ECOHYDRAULICS
ÉCOHYDRAULIQUE
2000

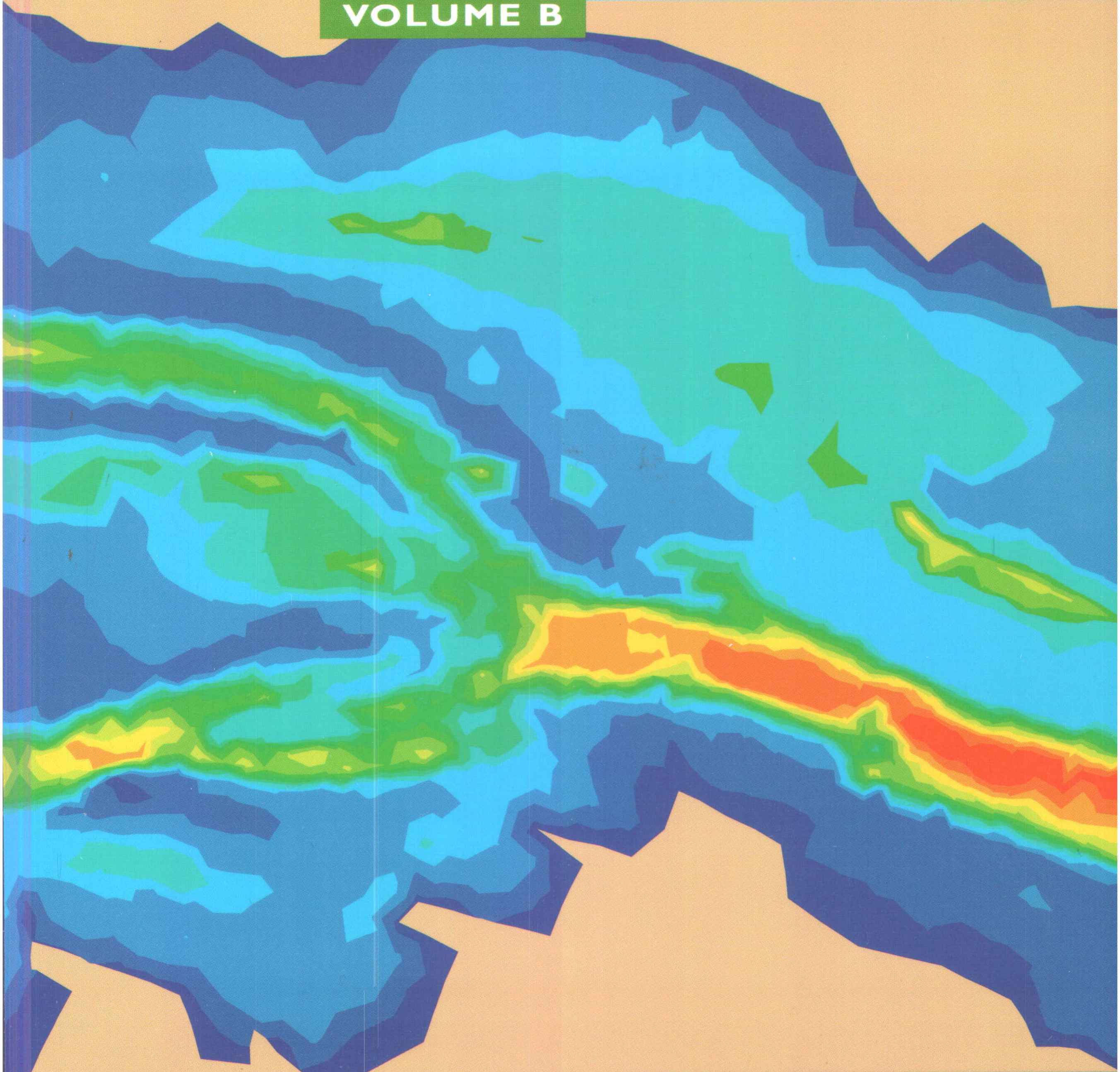
2nd INTERNATIONAL SYMPOSIUM ON HABITAT HYDRAULICS

2e SYMPOSIUM INTERNATIONAL SUR L'HYDRAULIQUE ET LES HABITATS

QUÉBEC, JUNE / JUIN 1996

PROCEEDINGS / COMPTES-RENDUS

VOLUME B



Editors / Éditeurs
M. Leclerc A. Boudreault
H. Capra Y. Côté
S. Valentin

Published by
Publié par
INRS-EAU

Co-published with
Co-publié avec
FQSA
IAHR
AIRH

Thanks to Ecohydraulics 2000 partners
Merci aux partenaires d'Écohydraulique 2000

INRS Institut National de
la Recherche Scientifique

IAHR AIRH
**International Association
for Hydraulic Research**

***Association Internationale
de Recherches Hydrauliques***

FQSA
Fédération Québécoise pour
le Saumon Atlantique

Environnement Canada **Environment Canada**

ENVIRONNEMENT
ET FAUNE
QUÉBEC

Fisheries and Oceans **Pêches et Océans**

Gouvernement du Québec
Ministère des Ressources naturelles

Hydro Québec

Cemagref
Institut de recherche
pour l'ingénierie
de l'agriculture
et de l'environnement

INTERNATIONAL WATER RESOURCES ASSOCIATION
ASSOCIATION INTERNATIONALE DES RESSOURCES EN EAU
IWRA AIRE AIREH

HMS ENERGY

Transports Québec

ALCAN

GENIVAR
GROUPE CONSEIL

EDF
Electricité de France

ROCHE

Sépaq

Ducks Unlimited Canada

CSCE SCGC
The Canadian Society for Civil Engineering
La Société canadienne de génie civil

AFS *American Fisheries Society*

COVER PAGE PICTURE / IMAGE DE LA PAGE COUVERTURE
Two-dimensional Numerical Field Model (NFM) for depth of Lake Saint-François, St. Lawrence River, Québec, Canada (Average flow of 7500 m^3/s) - See also the present Proceedings (Morin *et al*, Vol. B, pp. B187-B202)
Carte bidimensionnelle du Modèle Numérique de Terrain (MNT) de la profondeur au Lac Saint-François, fleuve Saint-Laurent, Québec, Canada (Débit moyen de 7500 m^3/s)- Présents Comptes-rendus (Morin et al, Vol. B, pp. B365-B380)

ECOHYDRAULICS
ÉCOHYDRAULIQUE
2000

Proceedings / *Comptes-rendus*

2nd International Symposium on Habitat Hydraulics

2ième Symposium international sur l'hydraulique et les habitats

Québec, June/*juin* 1996

Volume B

Editors / *Éditeurs* :
MICHEL LECLERC
Institut national de la recherche scientifique - Eau, Québec, Canada
HERVÉ CAPRA and / *et* **SYLVIE VALENTIN**
Cemagref, France
ANDRÉ BOUDREAULT
GÉNIVAR, Groupe-conseil Inc., Québec, Canada
YVON CÔTÉ
Ministère de l'Environnement et de la Faune du Québec, Québec, Canada

INRS

Published by / *Publiés par* : **INRS-Eau** / Québec / 1996
Co-published with / *Co-publiés avec* : **FQSA, IAHR/*AIRH***

Addresses of editors / *Adresses des éditeurs*

Michel Leclerc
Institut national de la Recherche scientifique - Eau
2800 Einstein, Bureau 105
C.P. 7500, Sainte-Foy (Qc),
Canada, G1V 4C7 E-mail:
michel_leclerc@inrs-eau.uquebec.ca

André Boudreault
GÉNIVAR, Groupe-Conseil Inc.
5355 des Gradins, Québec (Qc),
Canada, G2J 1C8
E-mail: aboudreault@genivar.com

Hervé Capra and / *et* **Sylvie Valentin**
Cemagref
3 Quai Chauveau
69336, Lyon, Cedex,
France
E-mail: sylvie.valentin@cemagref.fr
herve_capra@cemagref.fr

Yvon Côté
Direction générale de la ressource faunique
Ministère de l'Environnement et de la Faune du Québec
150 Boul. René-Lévesque est, 6^e étage, Boîte 90
Québec (Qc), Canada, G1R 4Y1

Copyright / *Droits d'auteurs*

Unless already published in the following journals, papers should refered to as follow /
À moins d'avoir été publiés dans les revues qui suivent les articles publiés dans ces comptes-rendus devront être cités de la manière suivante :
In: Leclerc M. *et al* (1996). Proceedings of the second IAHR Symposium on Habitats Hydraulics, Ecohydraulics 2000 or / *ou*
Dans: Leclerc M. et al (1996). Comptes-rendus du deuxième Symposium international de l'AIRH sur l'hydraulique et les habitats - Écohydraulique 2000

Complementary publications / *Publications complémentaires*

Selected papers will also be published in / *Une sélection de papiers feront l'objet de publication dans* :

Regulated Rivers : Research and Management
Editor / *Éditeur:* G.E. Petts
School of Geography, University of Birmingham
Edgbaston, Birmingham B15 2TT
United Kingdom

Revue des Sciences de l'eau
Editor / *Éditeur :* P. G. C. Campbell
Institut National de la Recherche Scientifique -Eau
2800 Rue Einstein,
P.O.B. / B.P 7500, Ste-Foy (Qc), Canada, G1V 4C7

Fisheries Management and Ecology
Editor / *Éditeur :* I. G. Cowx
Hull International Fisheries Institute
University of Hull
Hull HU6 7RX, United Kingdom

ISBN 2-89146-379-X: set / *ensemble* volumes A and / *et* B
ISBN 2-89146-380-3 : volume A
ISBN 2-89146-381-1 : volume B
Dépot légal, Bibliothèque nationale du Québec

Additional copies / *Copies additionnelles*

INRS-Eau, Service de la documentation, 2800 Einstein, C.P. 7500, Sainte-Foy (Qc) Canada G1V 4C7, FAX: +1 (418) 654-2600

Graphism for cover page and logo / *Graphisme pour le logo et la page couverture* : Graphidée
Printed in Québec City by / *Imprimé à Québec par* : Les impressions 03 inc.

Local organizing committee / *Comité local d'organisation*

Michel Leclerc	Chairman / Président Institut National de la Recherche Scientifique - Eau
Bernard Beaudin	Vice-president / *Vice-président* Fédération Québécoise pour le Saumon Atlantique
Ronald Greendale	Coordinator /*Coordonnateur* Consultant
Hervé Capra	Co-editor / *Co-éditeur* Cemagref-Lyon
Sylvie Valentin	Co-editor / *Co-éditeur* Cemagref-Lyon
André Boudreault	Editorial committee / *Comité éditorial* GÉNIVAR, Groupe-Conseil Inc.
Yvon Côté	Editorial committee / *Comité éditorial* Ministère de l'Environnement et de la Faune du Québec
Isabelle Couture	Communications Consultant / *Consultante*
Johanne Desrosiers	*Secretary / Secrétariat* *Institut national de la recherche scientifique - Eau*
Florent Gagné	Tresurer / *Trésorier* Fédération Québécoise pour le Saumon Atlantique
Yvan Vigneault	Excursions Ministère des Pêches et des Océans du Canada

Scientific committee / *Comité scientifique*

Baglinière, J.L., FR
Bovee, K. USA
Cunjak, R.A., CA (NB)
Fedorov, P., RUS
Hardy, T.B., USA
Jowett, I., NZ
Lachat, B., SW
Manoha, B., FR
Nestler, J., USA
Sabaton, C., FR
Stalnaker, C.B., USA
Tanida, K. , JA
Vigneault, Y. , CA (Qc)
Boudreault, A. CA (Qc)
Côté, Y. , CA (Qc)
Dodson, J., CA (Qc)
Gore, J.A., USA
Heggenes, J., NO
Katopodis, C., CA (Ma)
Larinier, M., FR.
McMahon T.A., AUS
Petts, G.E., UK
Saltveit, S.J., NO
Starosolszky, Ô., HUN
Tsujimoto, T., JA
Wesche, T.A., USA
Bechara, J., ARG
Cowx, I. UK
Di Silvio, G., IT
Gustard, A., UK
Hicks, B.J. , NZ
Killingtveit, Å. ,NO
Leclerc, M., CA (Qc)
Nakamura, S., JA
Prowse, T.D., CA (Sa)
Souchon, Y., FR
Swales, S., AUS
Vaskinn, K.A., NO

PREFACE

The harmonious coexistence of hydraulic structures with the natural environment is the focus of this second symposium on ecohydraulics organized under IAHR'S sponsorship. The first symposium was held in Trondheim, Norway in August 1994. Approximately equal numbers of hydraulicists and biologists sharing an interest in the rational and ecological use of the aquatic environment attended ECOHYDRAULICS 2000. Hydraulic works may bring about disastrous consequences for fish communities by destroying or modifying important habitats. The main objective of the Symposium ECOHYDRAULICS 2000 was to make available new scientific knowledge, state of the art analysis tools and advantageous technical solutions to prevent or limit these negative impacts. We hope that this goal was achieved. As chairman of the event, I take this opportunity to acknowledge all persons, sometime in shadow, who participated to the success of ECOHYDRAULICS 2000.

PRÉFACE

La coexistence harmonieuse des ouvrages hydrauliques et du milieu naturel est le thème principal d'un deuxième Symposium spécialisé organisé sous les auspices de l'AIRH sur ce sujet, le premier ayant été tenu à Trondheim, Norvège en août 1994. Cette seconde rencontre internationale a mis en présence un nombre sensiblement égal d'hydrauliciens et de biologistes désirant concilier des objectifs d'utilisation rationnelle et écologique du milieu aquatique. Les interventions hydrauliques peuvent avoir des conséquences désastreuses sur les communautés piscicoles par la destruction des habitats qui les supportent. Le but du Symposium était d'offrir des connaissances, des outils d'analyse et des solutions techniques avantageuses pour éviter ou limiter ces impacts négatifs. Nous sommes très heureux d'avoir eu le privilège de participer à la réalisation de cette noble cause. À titre de président du Symposium, je tiens à féliciter et à remercier toutes les personnes qui ont participé, parfois dans l'ombre, à l'organisation d'ÉCOHYDRAULIQUE 2000.

Michel Leclerc

Michel Leclerc
Chairman / *Président*

Table of Contents / Table des matières

VOLUME A

Direct and remote effects of civil works
Effets directs et à distance des ouvrages de génie

The Effect of Gas Supersaturation on Fish Health Below Yacyreta Dam (Parana River, Argentina) A3

Bechara J. A., H. A. Domitrovic, C. F. Quintana, J. P. Roux, W. R. Jacobo and G. Gavilan

Downstream Effects of Hydroelectric Impoundment on River Macrophyte Communities A13

Bernez I. and J. Haury

Anthropogenic Impacts, Hydrological Variations and Vegetation Changes in Stream Corridors of Northern Alps A25

Girel J. , G. Pautou and B. Garguet-Duport

Hydrobiological Aspects of Impounding at Petit Saut (French Guiana) A37

Horeau V., A. Champeau, A. Grégoire and P. Cerdan

Biological Motivation of Water Level Regulation in Large Shallow Lake Vortsjärv (Estonia) A47

Nõges T. and P. Nõges

Integrated System Analysis: Effects of Water Management on the Habitat Suitability for Key Species in the Sea of Azov A59

De Vries M. B.

Changes of Flood Dynamics Regulating Floodplain System Functioning on the Garonne River, South-West France A71

Steiger J.

A Preliminary Telemetry Investigation on the Obstacles to Anadromous Salmonids Migration in Spawning Streams of the Belgian Ardennes (River Meuse Basin) A83

Ovidio M., C. Birtles, E. Baras and J. C. Philippart

Colonisation Dynamics of a Catchment Area by Eel. Characterization of Migrating Populations in a Free Access River A89

Legault A.

The La Grande Hydroelectric Project and Eelgrass Monitoring in Coastal James Bay

Julien M., J. J. Fournier, R. Lalumière and C. Lemieux

Coastal - estuarian - fluvial interactions
Interactions fluviales - estuariennes - côtières

Morphosedimentological Evolution of the Shoreline in Pointe-Taillon Conservation Park, Lac Saint-Jean, Québec A113

Denis R.

Modeling Water Availability to Riparian Vegetation in an Impacted River System A121
Birkhead A. L., A. L. Van Coller , C. S. James and G. L. Heritage

Stratification of the Miramichi Estuary: Impact of Fresh Water Discharge A133
Saint-Hilaire, A., S.C. Courtenay, A. Locke and D. Booth

Control of Ecological Situation in the Basin of a Tidal Power Station (Kislaya Bay, Barents sea) A145
Fedorov M. P. and M. B. Shilin

Two Layer Finite Element Numerical Model for Simulating Salmon River Estuaries A155
Robert J. L., F. Anctil, Y. Ouellet

Land uses impacts
Impacts de l'utilisation du territoire

Hydro-Agricultural Management in the Rhône River Delta, France: Consequences on Dissolved and Solid Fluxes, Potential Impact on Fish Populations A167
Chauvelon P., A. Sandoz, G. Poizat and A. J. Crivelli

Response of a Natural River Valley Wetland to Supplementary Runoff and Pollutant Load from Urban Waste Water Discharge A177
Nõges P.

Role of abiotic variables
Rôle des variables abiotiques

Analysis of Flow Velocity Fluctuations in Different Macrophyte Banks in a Natural Open Channel A191
Stephan U. and U. Wychera

Laboratory Investigations of Near-Bed Hydraulics in a Reconstructed Cobble-Bed Stream Section A203
Young W.J.

Characterizing the Acoustical Environment of Streams A215
Stewardson M., C. Gippel and I. O'Neill

Measurement of Burst Swimming Performance in Wild Atlantic Salmon *(Salmo salar)* Using Coded Radio-transmitted Signals A227
Colavecchia M. V., C. Katopodis, R. Goosney, D. A. Scruton and R. S. McKinley

The Effect of Velocity and Turbulence on the Growth of Periphyton in a Cobble-Bed Stream: The Five-Stone Experiment A239
Goring D. and B. Biggs

Friction Factors for Vegetation A251
Kouwen N. and M. Fathi-Mogadam

Geomorphology and habitats
La géomorphologie et les habitats

Invited lecture / *Conférence sur invitation*

Fluvial Geomorphology and Fish Habitat: Implications for River Restoration A261
Kellerhalls R. and M. Miles

Translating Channel Morphology into Hydraulic Habitat: Application of the Hydraulic Biotope Concept to an Assessment of Discharge Related Habitat Changes A281
Rowntree K.M. and R. A. Wadeson

Fish Habitat and Micro Structure of Flow in Gravel Bed Stream A293
Tsujimoto T.

Between Reach Contrasts in Channel Form, Hydraulics, Lateral Stability and Riffle Sand Content in an Unstable Gravel Bed River: Implications for Atlantic Salmon Habitat A305
Lapointe M. F. and B. Payne

Modeling of Instream Flow Needs: the Link between Sediment and Aquatic Habitat A319
Milhous R. T.

Modelling the Response of River Geomorphology and Riverine Vegetation to Water Resources Development in the Sabie River, South Africa A331
James C.S., C. R. Nicolson, A. W. Van Niekerk and G.L. Heritage

Mechanics of Sediment Transport for Estimating Habitable Condition of Bivalves in Beach A343
Gotoh H. and T. Sakai

A Model of Fish Assemblage Response to Hydraulic and Fluvial Geomorphological Change in a Pool-Rapid Channel Type A355
Heritage G. L., A. W. van Niekerk and D. Weeks

Importance of Gravel Bars as Spawning Grounds and Nurseries for European Running Water Cyprinids A367
Baras E., J. C. Philippart and J. Nindaba

Winter Geomorphological Processes in the Sainte-Anne River (Sainte-Anne-de-La Pérade, Québec) and their Impact on the Migratory Behaviour of Atlantic Tomcod *(Microgadus tomcod)* A379
Bergeron N., A. G. Roy, D. Chaumont

River-Bed Degradation Influenced by Growth of Vegetation Along a Stream A389
Tsujimoto T. and T. Kitamura

Basic Morphological Process Due to Deposition of Suspended Sediment Affected by Vegetation A395
Tsujimoto T., T. Kitamura and S. Murakami

Ecohydrology
Écohydrologie

Importance of Hydrological Models in Habitat Issues: Comparison Between a Stochastic and a Deterministic at Catamaran Brook (New-Brunswick) A409

Saint-Hilaire A., D. Caissie, G. Morin and N. El-Jabi

Likelihood Expression of Hydrological Structuring Events For Brown Trout Population Dynamics: Methodological Approach A421

Breil P., H. Capra

Fish overwintering
Hivernage des poissons

Structure of Turbulent Flow in an Ice-Covered Tidal Environment A435

Martin N., A. G. Roy and N. Bergeron

On Under-Ice River Plumes in James and Hudson Bays A447

Messier D. and F. Anctil

Winter Flow Characteristics in Cold Regions A459

Hamilton S. A., R. D. Moore and K. Wiebe

Effect of Ice Formation on the Physical Stream Environment and the Overwinter Distribution of Presmolt Atlantic Salmon Parr A471

Whalen K. G. and D. L. Parrish

Impacts of logging operations
Impacts de l'exploitation forestière

Impacts of Logging Operation on Lake Populations of Brook Trout (*Salvelinus fontinalis*) and their Habitats in the Mastigouche Wildlife Reserve A485

Bérubé P. and F. Lévesque

Conservation of wetlands and shore habitats
Conservation des milieux humides et côtiers

Effective Shore Protection with Integrated Environmental Enhancement A501

Hall K. R.

Hydraulic and Ecological Considerations for Coastal Realignment A513

Nottage A. S., K. E. Carpenter and T. J. Chesher

The Effect of Seawalls on Coastal Morphology A525

Holmes P. and M. A. L. Neshaei

Minimum flow and adapted hydrological regime
Débits réservés et hydrologie adaptée

A Retrospective Assessment of a Regulated Flow Regimen for a Newfoundland (Canada) River A533

Scruton D.A. and L. J. LeDrew

The Basic Flow: An Alternative Approach to Calculate Minimum Environmental Instream Flows A547

Palau A. and J. Alcazar

Recommending Variable Flow Values for Fish A559

Locke A. G. H.

Use of Wetted Perimeter in Defining Minimum Environmental Flows A571

Gippel C. and M. Stewardson

Long-term Changes in Flood Protection Level and Fish Habitat Quality in a Mountainous Stream After a Major Flood Disaster A583

Nakamura S., T. Kutsumi, H. Asakusa, M. Sekine and M. Kubota

Ecological rehabilitation of the River Meuse: A Framework for Assessment of Ecological Effects of Discharges A595

Semmekrot S., M. J. J. Kerkhofs , J. C. G. van Liebergen

Habitats and comprehensive management
Habitats et gestion intégrée des ressources

Invited lecture / *Conférence sur invitation* :

Goals, Indicators, and Examples of River Basin Studies based on Ecological Considerations A609

Shen, H. W.

The Habitat Evaluation Procedure in the Policy Analysis of Inland Waters in the Netherlands: Towards Ecological Rehabilitation A619

Duel H., B. Pedroli and W. E. M. Laane

Towards a Regionalization of Aquatic Habitat: Distribution of Mesohabitats at the Scale of a Large Basin A631

Cohen P., H. Andriamahefa and J. G. Wasson

A Reference Framework for the Integrated Ecological Management of the Saint-Charles Watershed, Québec, Canada A643

Gerardin V. and J. P. Ducruc

Elaboration of an Ecological Information System with Spatial Reference. Example of the Anti-larval Mosquitos Control A653

Garguet-Duport B., G. Pautou and J. Girel

Aquatic Gap Analysis: Tool for Watershed Scale Assessment of Fluvial Habitat and Biodiversity A665

Meixler M. S ., M. B. Bain and G. H. Galbreath

Peak flow management
Effet des éclusées et débits de pointe

Study of the Juvenile Community in the Brown Trout *(Salmo trutta fario)* in Hydropeaking Situations A673

Liebig H., P. Lim, A. Belaud and S. Lek

The Effect of Managed Hydropower Peaking on the Physical Habitat, Benthos and Fish Fauna in the Bregenzerach, a Nival 6th Order River in Austria A685

Parasiewicz P., S. Schmutz and O. Moog

Influence of Hydropeaking on the Structure and Dynamics of Invertebrate Populations in a Mountain Stream A699

Cereghino R. and P. Lavandier

Towards a Method to Reconstruct a Natural Hydrograph for Defining Ecologically Acceptable Discharge Fluctuations A711

Salverda P., M. J. J. Kerkhofs, P. J. J. Verbraak and J. D. Klein

Impact mitigation measures
Mesures d'atténuation d'impacts

Controlling Algal Blooming in Withdrawal Zone Using Vertical Curtains: an Application to Terauchi Dam Reservoir, Japan A725

Asaeda T., D. G. N. Priyantha, S. Saitoh and K. Gotoh

Aeration and Minimum Flow Enhancements at Hydropower Dams A737

Harshbarger E. D., W. D. Proctor, B. Hadjerioua and G. E. Hauser

EKOS - a Software Program for Analysing Banks Used by Animal Species A743

Reinhold J. O. and J. T. Klein Breteler

Water quality and habitats
La qualité de l'eau et les habitats

Water Quality Assessment and Management Using Numerical Simulation Tools A753

Marche C. and M. Castonguay

Intermittent Release of Sewage Disposal for Ecosystem Protection in the Coastal Zone A759

Koutitonsky V.G., N. K. Gidas and M.B. Gidas

A Decision Support System for Managing an Urban Aquatic Ecosystem A769

Lynch D. and K. Wiebe

Chemical and Biotic Indices in River Environment Quality Assessment. The Case of Belice River (Western Sicily) A779

Nasello C. and S. Nicosia

Interdisciplinary approaches - Partnerships
Approches interdisciplinaires - Partenariat

Management and Use of Rivers in Norway: Measures to Alleviate the Impact of Hydropower Development A789

Grande R. and L. Storset

The Succession of the Marchfeld Canal - a Newly Created Watercourse A797

Ernegger T.

Wetlands in Nigeria: Characteristics and Available Options for their Utilization and Conservation A811

Balogun I.

Environmental Impact Assessment Methodology Based on a Reliability Knowledge Management A829

Rochette A. and B. Robert

Nature-Friendly Design and Maintenance of Watercourses. A Way to Promote Ecological Engineering in the Ministry of the Flemish Community A835

Claus K. and L. Janssens

Formation of the International Aquatic Modelling Group A845

Hardy, T. B

National Wildlife Areas: Essential Habitats on the St Lawrence A851

Mercier Y.

Wetlands Conservation Along the St-Lawrence River: an Examplary Partnership A865

Mercier Y.

Impact Analysis and Restoration Planning Using the Riverine Community Habitat Assessment and Restoration Concept (RCHARC) A871

Nestler J. M., L. T. Schneider, D. Latka and P. Johnson

Index of subjects / *Index des sujets* (Volumes A and / *et* B) A877

Index of authors / *Index des auteurs* (Volumes A and / *et* B) A889

VOLUME B

Instream flow needs methodology
Utilisation des modèles d'habitat

Geomorphological and Hydraulic Controls on Atlantic Salmon Spawning Habitat in a Tributary of the River Dee, Scotland B3

Moir H., C. Soulsby and A. Youngson

An Evaluation of Habitat Improvement Structures in an Experimental Channel in Newfoundland B15

Mitchell J., R. S. McKinley, G. Power and D. A. Scruton

A Study on the Method of Evaluating Fish Habitat Environment by PCA under Various River Environmental Factors B27

Matsuzaki H. and N. Tamai

A Comparison of Habitat Suitability Indices Developed From Daytime Versus Nightime Observations for Atlantic Salmon in a Regulated Newfoundland Stream B33

LeDrew L.J., D. A. Scruton, R. S. McKinley and G. Power

Spatial Relations Between Lotic Fishes, Benthic Macroinvertebrates and the Stream Habitat: Toward a GIS-Assisted Approach B45

Muotka T., A. Mäki-Petäys and P. Kreivi

Weighted Usable Volume for Young of the Year Non-game Fish in the Marchfeldkanal, Austria B55

Schmutz S., H. Mader and G. Unfer

Description of a Technique for Multiscale Analyses: Spatial Scaling and Habitat Selection in Riverine Habitats B67

Bult T. P., R. L. Haedrich and D. C. Schneider

Effect of Hydraulic Variables on Migratory Behaviour of Atlantic Salmon *(Salmo salar)* in the Moisie River, Québec B81

Lévesque F., M. Proulx and G. Corfa

A Perspective to Predict Habitat Suitability of Lotic Fish : Linking Statistical Hydraulic with Multivariate Habitat Use Models B95

Lamouroux N., H. Capra and M. Pouilly

Classifying Stream Habitat Using Fish Community Analyses B107

Bain M. B. and J. G. Knight

Equal Area Line-transect Sampling for Smallmouth Bass Habitat Suitability Criteria in the Susquehanna River, Pennylvania B119

Allen M. A.

Selection of an Ecological Reference Flow : Support by Modeling Habitat of a Key Species, the Northern Pike B133

Monfort O., P. Breil and P. Boët

Habitat modeling
Modélisation des microhabitats

Invited lecture / *Conférence sur invitation*

Perspectives on Two-Dimensional River Habitat Models: the PHABSIM Experience B149

Bovee K.

Ecological Evaluation of Instream Flow Regulations Based on Temporal and Spatial Variability of Bottom Shear Stress and Hydraulic Habitat Quality B163

Jorde K.

Natural/Artificial Floods Connected with River Habitat B175

Peviani M., I. Saccardo, A. Crosato and G. Gentili

Integrated Two-Dimensional Macrophytes-Hydrodynamics Modeling: Application to Lake Saint-François (St-Lawrence River, Québec, Canada) B187

Morin, J., M. Leclerc, Y. Secretan and P. Boudreau

The Application of Habitat Mapping to Identify Representative PHABSIM Sites on the River Tavy, Devon, UK B203

Maddock I. and D. Bird

The Habitat Modeling Framework. A tool for creating habitat analysis programs B215

Alfredsen K. and A. Killingtveit

Predicting Habitat Components for Semi-Natural Rivers in the United Kingdom B227

Fox P., M. Naura and P. Raven

Application of the Instream Flow Incremental Methodology to Conservation Flow for Freshwater Fishes in Japan B239

Tamai N., H. R. Kim and Y. Kawahara

Determining Habitat Criteria for the Endangered Fountain Darter Through Aquatic Mapping and Hydrologic Modeling B251

Bartsch N., C. P. Gubala and T. B. Hardy

Methods for Describing In-Channel Habitat Features: Engineering Processes to Fractal Geometry Concepts B263

Sutton V. and J.M. Nestler

Using Hydraulic and Water Quality Modeling Output for Instream Flow Studies B275

Schneider L.T. and J.M. Nestler

Towards the Notion of Trophic Value. Concepts and Approaches. The Brown Trout Example *(Salmo truta)* B283

Dumont B., G. Carrel, Y. Le Coarer, B. Rivier and G. Suard

The Relationships Between River Channel Morphometry, Complexity and Aquatic Habitat B299

Gubala C. P., J. M. Eilers and J. A. Bernert

Field Sampling Design and Spatial Scale in Habitat-Hydraulic Modeling: Comparison of Three Models B307

Scruton D.A., J. Heggenes, S. Valentin, A. Harby and T. H. Bakken

Methodology for Fish Assemblages Habitat Assessment in Large Rivers. Application in the Garonne River (France) B323

Pouilly M., Y. Souchon, Y. Le Coarer and D. Jouve

Spatiotemporal Use of the Riffle/Pool Unit by Fish in a Brittany Brook B341

Roussel, J. M. and A. Bardonnet

Microposition Choice in Stream-living Atlantic Salmon *(Salmo salar)* Parr and Brown Trout *(Salmo trutta)* : Habitat-Hydraulic Three-Dimensional Model and Test B353

Heggenes J., A. Harby and T. Bult

Validation strategies of habitat models
Stratégies de validation des modèles d'habitat

Two-Dimensional Habitat Model Validation Based on Spatial Fish Distribution : Application to Juvenile Atlantic Salmon of Moisie River (Québec, Canada) B365

Boudreau, P., G. Bourgeois, M. Leclerc, A. Boudreau and L. Belzile

Sensitivity Analysis of PHABSIM in a Small Atlantic Salmon Stream B381

Bourgeois G., D. Caissie and N. El-Jabi

Evaluation of One-Dimentional and Two-Dimensional Hydraulic Modeling in a Natural River and Implications in Instream Flow Assessment Methods B395

Tarbet K. L. and T. B. Hardy

Problems with the Use of IFIM for Salmonids and Guidelines for Future UK Studies B407

Bird D.J.

A Full-Scale Test to Validate the Contribution of the IFIM Procedure in the Choice of a Guaranteed Flow Downstream Hydrostations B419

Merle G. and J. Eon

Modelling Instream Flow Needs: the Effects of a Water Transfer Scheme on Macroinvertebrate Communities in the Receiving River Wear B431

Gibbins C. N., M. J. Jeffries, C. Soulsby and C. R. N. Elliott

Alternate modeling approaches
Nouvelles approches de modélisation

Invited lecture / *Conférence sur invitation*

The Future of Habitat Modeling B447

Hardy T. B.

Bioenergetics Model for Drift-Feeding Brown Trout B465

Hayes J.

Modeling Atlantic Salmon Population Dynamics in a Small Stream B477
Cunjak R. and J. Therrien

The Effect of Water Current on Foraging Behaviour of a Rheophilic Cyprinid, *Chondrostoma nasus*, During Ontogeny: Trade-off Energetic Benefit Swimming Costs B487
Flore L. and H. Keckeis

A Multivariate Model to Relate Hydrological, Chemical and Biological Variables to Salmonid Standing Crop in Italian Alpine Rivers B507
Annoni P., I. Saccardo, G. Gentili and L. Guzzi

Comparative Evaluation of Suitability Curve Based Habitat Modeling and a Mechanistic Based Bioenergetic Model Using Two-Dimensional Hydraulic Simulations in a Natural River System B519
Ludlow J. and T. B. Hardy

Presentation and First Applications of a Dynamic Population Model for Brown Trout (*Salmo trutta*). Aid to River Management B531
Sabaton C., L. Siegler, V. Gouraud, J. L. Baglinière and S. Manné

Assessing Impacts and Predicting Restoration Benefits of Flow Alterations in Rivers Developed for Hydroelectric Power Production B543
Bain M. B. and V. T. Travnichek

Habitat creation and restoration
Création et restauration d'habitat

Invited lecture / *Conférence sur invitation :*

Ecohydraulics: Challenges and Opportunities with Fish and Fish Habitat B555
Katopodis C.

Graelva, a River in Norway That Has Been "Raised" and Reconstructed on a New Rock Bed B579
Johnsen M. and T. O. Sandnæs

Improvements of the Retention Qualities of Pools in Salmon Rivers B585
Castonguay M. and C. Marche

Monitoring of Fisheries on the River Blackwater B595
Fisher K., H. R. Wallingford, R. Myers, J. Lyness and J. O'Sulliven

Application of the Physical Habitat Simulation (PHABSIM) Model as an Assessment Tool for Riverine Habitat Restoration Techniques B607
Elliott C.R.N., D. J. Willis and M. C. Acreman

Effects of Instream Enhancement Structures on Brown Trout Habitat Availability in a Channelized Boreal River: a PHABSIM Approach B619
Huusko A. and T. Yrjänä

Ecohydraulics of Spur Dikes in Ushizu River, Kyushu Region, Japan B631
Tamai N., Y. Kawahara, M. Aoki, A. Matumoto, Y. Ishikawa and M. Yasuda

An Analysis of Artificial Riffles and Enhancement of Benthic Community Diversity by Physical Habitat Simulation (PHABSIM) and Direct Observation B643

Gore J. A., D. J. Crawford and D. S. Addison

Construction of Generating Station at the Mercier Dam: Feasibility Study of a Man-Made Walleye Stream Spawning Habitat Used as a Mitigative Measure B655

Gagnon B., M. Gendron and I. Thériault

Problematics of Shoreline Habitat Erosion Along the St.-Lawrence River: Magnitude and Potential Solutions B669

Lehoux D.

Bioengineering for shore protection
Techniques végétales de protection des berges

Invited lecture / *Conférence sur invitation*:

Bioengineering Techniques to Prevent Shore Erosion B683

Lachat, B.

Artificial Silting of Shore Marshes: the Example of the Bay of Rivière-du-Loup, Québec, Canada B705

Bertrand P., B. Massicotte, J. P. Beaumont and D. Lehoux

The Stabilization of the Ottawa River Banks: a Comparative Study Between Bioengineering Methods and a Method Based on Natural Gravel Embankments Installed in Winter B717

Bérubé M., G. Gagnon, L. Hardy and L. Bariteau

Bank-Protecting Functions of Common Reed and Ditch Reed B729

Fukuoka S. and A. Watanabe

Design and Management of Banks: an Integrated Approach B741

Reinhold J. O. and J. T. Klein Breteler

Dynamics and Restoration of Lakeside Reedbelts in a Prealpine Lake (Lake Constance, Germany) B753

Ostendorp W., H. Stark, P. Krummscheid-Plankert and A. Pier

Bioengineering Technics Applied on an Humid Area Biological Recovery and Clearing B767

Ferreira-Machado, M. T. and A. Telhado

Implementation of New Technologies to Control Erosion Along the Banks of St-Lawrence River B775

Massicotte M. C.

Alcan Québec:Lake Saint-Jean (Québec, Canada) Shoreline Stabilization Program and Preservation of Natural Habitats B783

Dupuis S.

Fishways, By-pass structures
Passes migratoires, systèmes d'évitement

Invited lecture / *Conférence sur invitation* :

Integrated Management of a Restored Salmonids Habitat and Hydroelectrical Resources on the Jacques-Cartier River (Québec, Canada) B797

Beaulieu, C.

Upstream Passage of Migratory Fish Through Navigation Locks B829

Zylberblat M. and J. Y. Menella

Flow Structure of Stream-Type Fishway B843

Tsujimoto T. and Y. Shimizu

Naturally Shaped Fishway of the Nagara River Estuary Barrage B855

Komura S., S. Kazusa and T. Hara

Hydraulics of Riffles and their Applications to Fishways B861

Komura S.

Fish and Lamprey Passage in a Combined Denil and Vertical Slot Fishway of the Regulated River Kemijoki B873

Laine A., R. Kamula and J. Hooli

The Timing of the Cycle in Borland Fosh Locks: Optimization of the Duration of the Fish Passage Phase B885

Bochechas J.

Smolt Behaviour and Dowstream Fish By-pass Efficiency at Small Hydroelectric Plants in France B891

Larinier M. and F. Travade

Fish Community Structure, Distribution Patterns and Migration in the Vicinity of Different Types of Fishways on the Enz River (Germany) B903

Jansen W., B. Kappus, J. Böhmer, T. Beiter and H. Rahmann

Guiding Salmonids in a Big River to Fixed Fish Traps or to Fish Passages B915

Herva M.

Integrating Structural and Behavioral Fish Protection at a Major Pumped-Storage Hydropower Dam B929

Nestler J.M., G.R. Ploskey, L.T. Schneider and G. Weeks

Hydro-Electric Development in Ireland: Problems for Atlantic Salmon (Salmo salar) and Some Solutions B941

O'Farrell M., M. Murphy , W. O'Connor, L. O'Sullivan, D. Quigley, J. Browne and N. O'Maoileidigh

Design of Simple Denil and Deepened Alaska Steeppass Fishways B953

Odeh M. and A. Haro

Fish Migration from Two Regulated Lakes to Outcoming Rivers Monitored by Hydroacoustics **B967**
Vehanen T., P. Hyvärinen and A. Mäki-Petäys

Index of subjects / *Index des sujets* (Volumes A and / *et* B) B979

Index of authors / *Index des auteurs* (Volumes A and / *et* B) B991

Instream flow needs methodology

Utilisation des modèles d'habitat

Geomorphological and hydraulic controls on Atlantic salmon spawning habitat in a tributary of the River Dee, Scotland.

Hamish Moir & Chris Soulsby
Dept. of Geography,
University of Aberdeen,
Elphinstone Road,
Aberdeen, AB9 2UF,
Scotland

Alan Youngson
S.O.A.E.F.D.,
Marine Laboratory,
Torry,
Aberdeen,
Scotland

ABSTRACT

Atlantic salmon (*Salmo salar*) are an important resource within the River Dee catchment, north-east Scotland. However, numbers of adult fish returning to spawn have been declining in recent years. The Girnock Burn, a tributary of the Dee, has been used as a monitoring site for salmon production since 1966. During this period, data on annual spawning distributions along the river has been recorded, revealing areas of the river which are preferred, others which are used intermittently and others which are never used for spawning. This paper outlines the main physical controls on spawning habitat at two scales in the river; the tributary and the reach. At the tributary scale, reach bed slope has a significant influence on spawning distribution. Also at this scale, river flow during the spawning period has a significant influence on the subsequent distribution of redds. At the reach scale, local hydraulic (depth, velocity and Froude number) and sedimentary (particle size distribution) controls become increasingly important. Mean utilised depth, velocity and Froude number were 0.248m, 0.536ms^{-1} and 0.344 respectively. Mean utilised surface and sub-surface sediment sizes were 33.5mm and 15.2mm respectively. It is suggested that the Froude number may be a useful single descriptor of hydraulic habitat, having the advantage that it is dimentionless and is thus comparable between different sizes of river and fish species. It is suggested that in order to make conclusions on habitat preference, habitat utilisation has to be related to habitat availability at the same river flow. Potential implications of the results in relation to fisheries management are also discussed.

KEY-WORDS: habitat / hydraulics / sediments / hydrology / Atlantic salmon / fluvial geomorphology

INTRODUCTION

Atlantic salmon (*Salmo salar*) are an important natural resource in a number of Scottish rivers, most notably the Dee, Spey, Tweed and Tay (Maitland *et al*, 1994). In the case of the River Dee in north-east Scotland, industries associated with the salmon fishery contribute £6.3 million per annum to the local economy (Dee Salmon Fishing Improvement Association, 1994).

Although the Dee is still considered to be the best spring salmon river in Scotland, its spring run has, in recent years, been in decline. The most important areas for the production of spring fish are the tributaries in the upper region of the catchment (Hay, 1995). One such tributary is the Girnock Burn which has been used as a key monitoring site for salmon production since 1966 by SOAEFD Freshwater Fisheries Laboratory. During this period data on the spawning distribution of salmon has been gathered on an annual basis. This has revealed that there are discrete parts of the river that are used every year for spawning, some that are never used and others that are used for a number of years and then abandoned. Thus, spawning appears to be suitable only in certain areas, which may or may not be stable.

This paper describes the preliminary findings of a research programme which seeks to explain the physical controls that govern the distribution of spawning habitat and to subsequently suggest management criteria for developing artificial spawning areas and to optimise the existing spawning potential of streams. The identification and quantification of the physical controls on the spawning of Atlantic salmon is carried out at two scales; the tributary and the river reach. At the tributary scale, the influence of slope and stream flow on the distribution of spawning sites are examined. At the smaller reach scale, the influence of local hydraulic and sedimentary variables are investigated.

STUDY AREA

The Girnock Burn drains 30km^2 of the Dee catchment, joining the mainstem river near the town of Ballater (Figure 1). It rises at an altitude of about 570m on the north-east flanks of the Lochnagar massif and joins the Dee at about 230m. The slope calculated between the heights at 10% and 85% of the river length is 17.4m/km.

The geology of the catchment is quite complex for the area; granites dominate, particularly in the upper catchment. In lower areas, schists and other metamorphic rocks are more extensive, including an outcrop of serpentenite in the north-east of the catchment. There are also a number of smaller igneous intrusions, notably diorite.

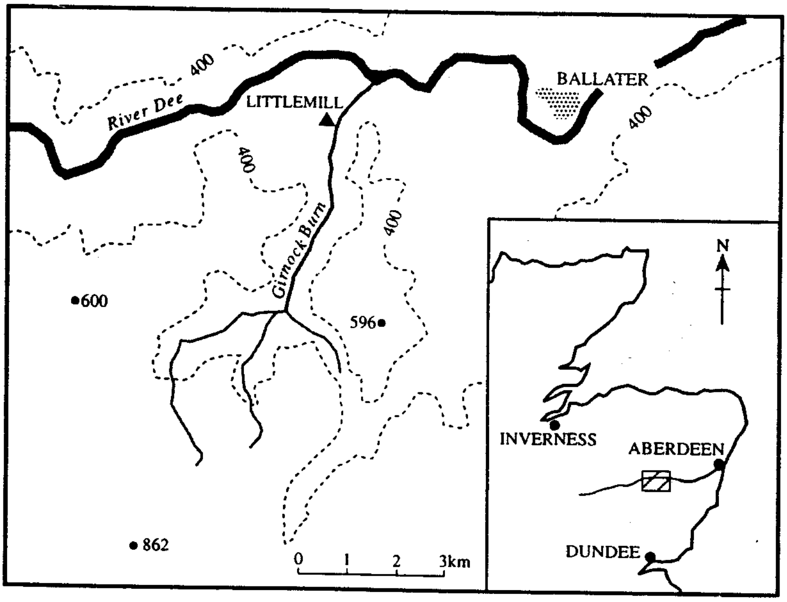


Figure 1: Study area

The climate and flow characteristics of the Girnock catchment are highly variable and exhibit strong seasonality. Maximum and minimum air temperatures recorded in the exceptional year of 1995 were 31°C and -27°C. The catchment receives about 1100mm of precipitation annually, with the summer months (May - August) being driest. Much of the precipitation, up to 25%, falls as snow in the winter months (Warren, 1985). The river has been gauged since 1969 at Littlemill (Figure 1). The river has a mean annual discharge of $0.5m^3s^{-1}$ although flows between June and August rarely exceed $0.1m^3s^{-1}$. About 85% of high flows with a 5 year return interval ($29m^3s^{-1}$) or greater have occurred between October and March. Many of these, especially between January and March, involve snowmelt.

The landuse in the catchment is predominantly heather (*Calluna vulgaris*) moorland used for rough grazing. There has been some commercial afforestation in the early 1990s but this is confined to small areas of previously farmed land. The catchment lies on an extensive sporting estate, so that in the autumn and winter grouse shooting and deer stalking are important activities.

METHODS

Data on the distribution of salmon spawning sites or redds have been gathered intermittently since 1966. During the spawning season, one or more redd counts are carried out along the length on the stream. During a count, each time a redd is observed, its longitudinal position on the river is noted. This position is given as a digital distance which is obtained from a base map. From this data, detailed redd maps and frequency-position histograms can be constructed. Spawning positions are measured upstream from the fish trap at Littlemill (Figure 1). The fish trap also permits the adult salmon entering the river to spawn each season to be counted, sexed and have their length measured.

The entire length of the Girnock where salmon are known to have spawned was levelled using standard surveying equipment in order to obtain data on bed slope. The distance between staff positions along the bed was on average 27m.

Sediment sampling was carried out during summer months when flows were low. Sampling sites were selected from data on previous spawning locations. This data was in the form of the longitudinal positions on the river (as described above) and photographs of individual redds. Two different sediment sampling techniques were employed. Firstly, to characterise the surface of the sediment, the Wolman-walk technique was used (Wolman, 1954). Eighteen sites were sampled in this manner. Approximately 150 particles were measured at each sampling site. Secondly, bulk sampling was used to characterise the subsurface sediment. Six sites were sampled using this technique. In this case a stilling basin was placed around the area of the bed to be sampled so that the loss of fine material was minimised. Sediment was dug out to a depth of approximately 0.3m, which has been shown to be the maximum depth to which salmon will excavate redds (Kondolf, 1988). The minimum mass of sample required to provide a good representation of the sediment characteristics was based on the criteria described by Church *et al* (1987). The required sample mass for a maximum sized particle with an intermediate axis diameter of 128mm (the largest size present in the spawning sediment examined) to be no more than 5% of the total mass was 60kg. At each site, this minimum mass of sediment was obtained from 3-5 replicate samples to account for any spatial heterogeneity within the sampling area. The average mass of each composite sample was 82.1kg.

Hydraulic measurements were carried out as close to spawning time as possible. Since the majority of redds are created during the night, it was usually not possible to measure hydraulic variables immediately after spawning had terminated. Rather, measurements were taken at the first occasion that a redd was identified, which was usually within a day of construction. Velocity and depth were measured immediately upstream of the pot over undisturbed sediment (Crisp and Carling, 1989). Three to five measurements were taken around this region and averages were calculated. As in many other studies, velocity was measured at 0.6 of the depth, the approximate position of mean velocity within a water column (Richards, 1982). Using the depth and velocity data, the mean Froude number ($= v/(gd)^{0.5}$, where v = flow velocity, d = flow depth and g = gravitational acceleration = $9.81ms^{-2}$) at each spawning site was calculated.

RESULTS AND DISCUSSION

Tributary Scale

Spatial Distribution of Spawning Sites

Annual redd count data permitted the examination of the cumulative spatial distribution of spawning sites on the Girnock (Figure 2). It is apparent that there are certain areas preferred for spawning, particularly in the middle reaches of the river. The highest peaks on the plot relate to very localised areas where there are frequently high spawning densities. Although the resolution on the x-axis is 100m, many of these popular spawning area have a longitudinal extent of as little as 5m. Infrequent use of other areas for spawning may be due to the occurrence of temporary, unstable pockets of spawning calibre sediment or opportunistic spawning in sub-optimal locations caused by very high numbers of spawning fish or reduced habitat availability due to low flows which are both known to limit the availability of spawning habitat (Shirvell and Dungey, 1983).

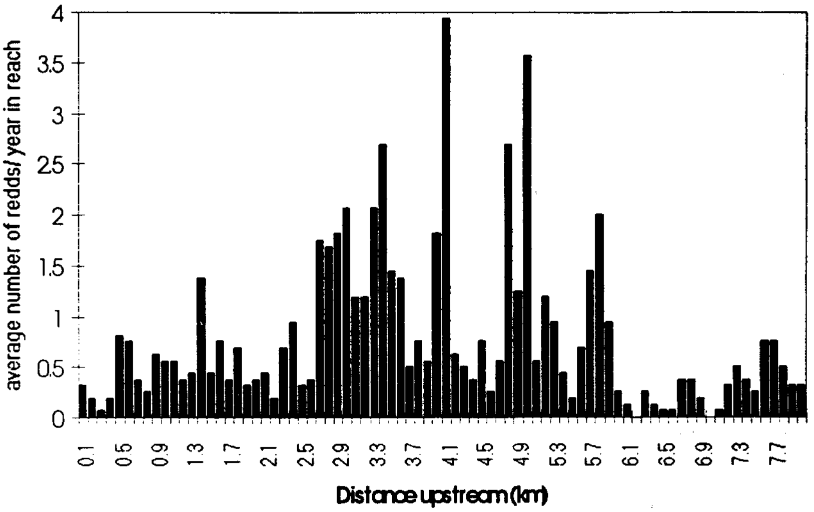


Figure 2: Diagram of cumulative annual redd distributions at the Girnock Burn. Distance upstream measured from fish trap at Littlemill

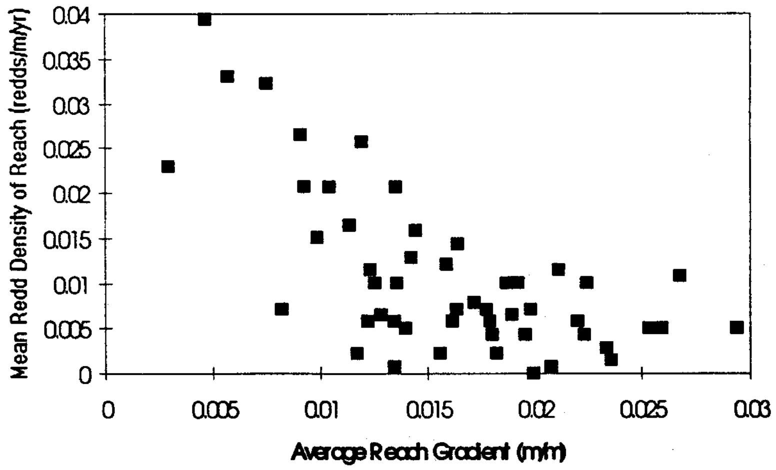


Figure 3: Influence of mean reach slope on mean reach redd densities

Figure 3 shows a plot of mean reach slope against mean reach redd densities. A characteristic reach length had to be assumed for the river as a whole. A resolution of about 140m was chosen and the number of spawning sites present in each of these sections was counted for the entire redd distribution record. There is a significant ($p<0.001$) negative relationship between mean reach slope and average annual redd densities, with reaches of lower mean slope tending to have a greater density of spawning sites. This probably reflects the reaches of lower slope having hydraulic conditions which facilitate the accumulation of redd-calibre sediment.

Inter-annual Variation in Spawning Location

Annual redd counts have shown that there is considerable variation in the number and location of spawning sites between years. There is a strong correlation ($p<0.001$) between the number of females entering the river to spawn and the amount of redds counted each year. However, this does not account for all the variation in redd counts between years or the difference in annual spawning distributions. When flow statistics for the spawning period (within the last week in October and the first three weeks in November) are correlated with redd location data, several significant relationships are apparent (Figure 4). A significant positive relationship ($p<0.025$) exists between the median flow during the spawning period (Q_{50}) and the mean number of redds created per female salmon that entered the river (Figure 4a). A significant positive relationship ($p<0.001$) also exists between median flow and the median position of the annual spawning distribution (Figure 4b). Finally, a positive relationship ($p<0.005$) between Q_{95} (a low flow statistic) and the upstream limit of spawning is apparent (Figure 4c). These findings suggest that river flow during the spawning season appears to significantly affect spawning behaviour.

The reasons for these relationships are not known, although several hypotheses can be suggested. In the case of the relationship between median flow and the number of redds created per female it may be that during periods of higher flow there is more habitat available to fish so that construction of multiple redds is permitted. For the other two relationships, it is hypothesised that low flow limits the spawning of salmon in the upper reaches of the stream as salmon may be prevented from reaching upstream locations. This could be due to their upstream migration being hindered by physical obstructions (such as a steep, bouldery reach which is impassable in low flow), or that there is a dearth of suitable spawning habitat in the upper parts of the catchment when flows are low. Work is currently being carried out to test these hypotheses.

These findings show that higher flows (excluding extreme floods) during the spawning season are preferable. Increased redds produced by each spawning female is desirable from a fisheries management perspective since the deposited eggs will not all be clustered at one site, thus increasing the chances of eggs surviving to emergence. The increased production of salmonids is also favoured when redds are situated as far up the catchment as possible, since salmon parr tend to populate the areas of a stream downstream from their spawning site (Hay, pers. comm.). If low flows prevent spawning in the upper parts of the catchment, then those areas of the stream will be unoccupied by parr in the following year. Also, a greater spread of a given number of redds along the length of the stream will help reduce density dependant mortalities upon emergence.

Reach Scale

Sediment Characteristics of Spawning Habitat

Salmon are known to usually spawn in cobble and gravel-sized sediment (Gibson, 1993). Figure 5 gives quantitative data on sedimentary spawning habitat from surface and bulk sampling analysis, in terms of composite frequency histograms. The range of median sizes (D_{50}) obtained from surface sampling is 24 to 40mm, with a mean of 33.5mm. The range of D_{50} of the subsurface samples is 12 to 19mm, with a mean of 15.2mm. All of these figures are within the gravel size class. These figures are in general agreement with the findings of Kondolf and Wolman (1993), who plotted the size of sub-surface spawning sediment against fish length for data obtained from the literature and their own research. Figure 6 incorporates this along with the sedimentary data gathered at the Girnock. Although there is considerable scatter in the relationship, upper and lower bounds to the data are quite clear. The upper boundary suggests that larger fish are able to spawn in coarser sediment, which is in agreement with the findings of Crisp (1993). This is thought to be due to the fact that larger fish can produce a greater suction force on the river bed with there tails, thus lifting larger particles and that they can spawn in faster velocities, which help to entrain larger sediment (Kondolf and Wolman, 1993). The Girnock data plots near the lower end of the range of median sediment sizes, especially when fish length is considered. The value of female body length for the Girnock is the median statistic from measurements obtained at the fish trap, as described above. It is notable that both the other plots relating to Atlantic salmon in Figure 6 have very similar values of D_{50}. Figure 5 also shows that the range of percentage fines (material <1mm diameter) content of the bulk samples is 3.4 to 9.6%, with a mean of 7.0%. These figures are all lower than the limits quoted for poor quality spawning gravel, which usually range between 15-20% (Crisp, 1993; Chapman, 1988). Levels of fine sediment above these

Figure 4a (p<0.025)

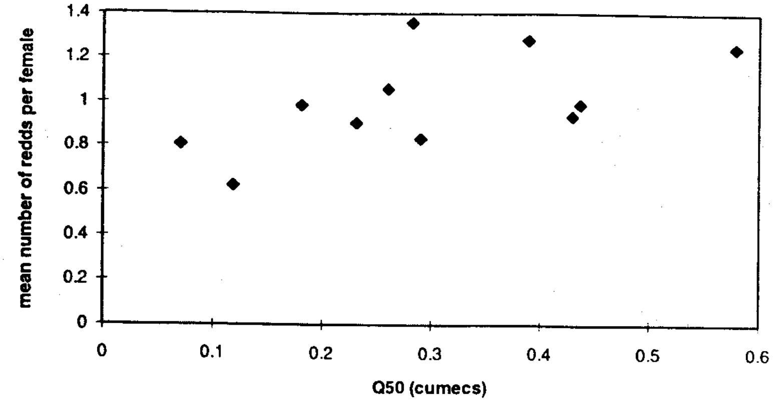


Figure 4b (p<0.001)

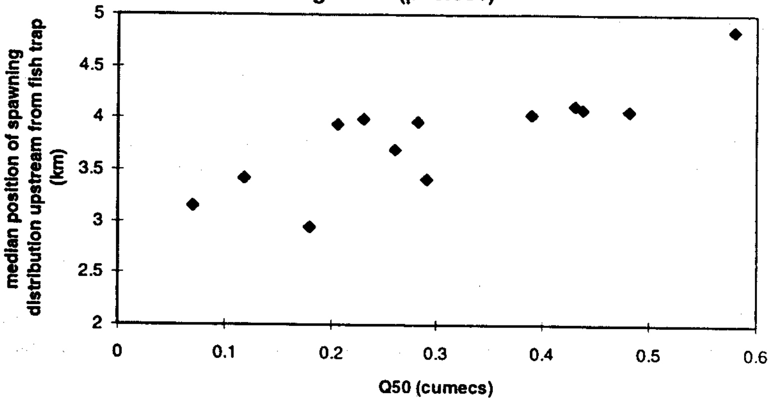


Figure 4c (p<0.005)

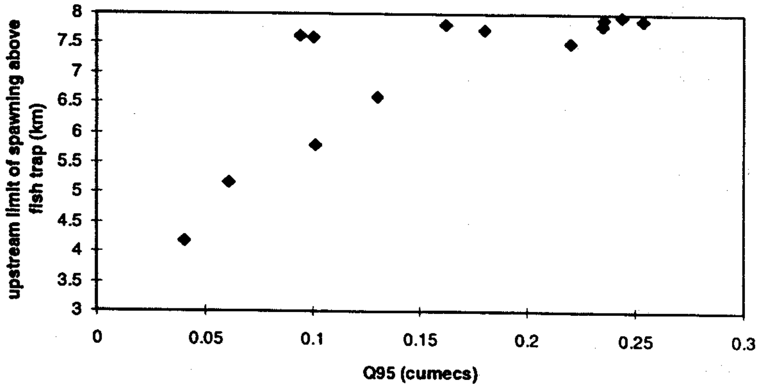


Figure 4a-c: Significant relationships between flow and aspects of spawning behaviour

levels reduce the permeability of spawning gravels, limiting the amount of dissolved oxygen reaching embryos and the rate at which waste products are removed, thereby increasing embryo mortality (Adams and Beschta, 1980).

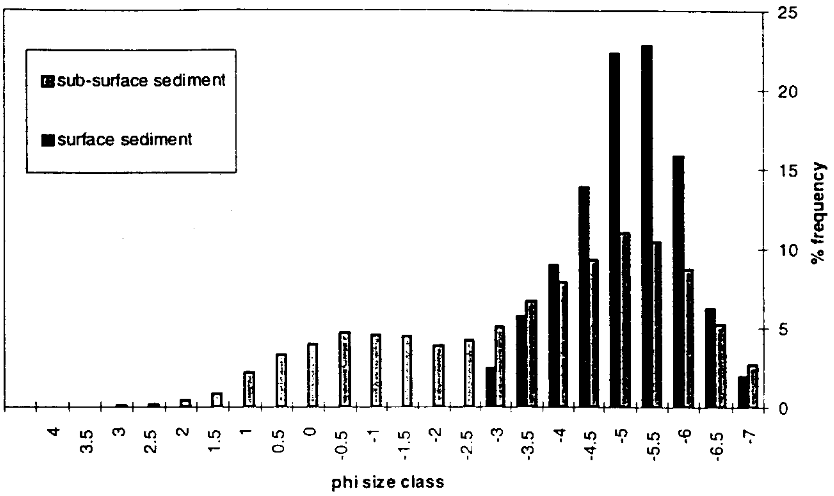


Figure 5: Frequency distribution of sediment sizes in composite surface and sub-surface samples at the Girnock Burn

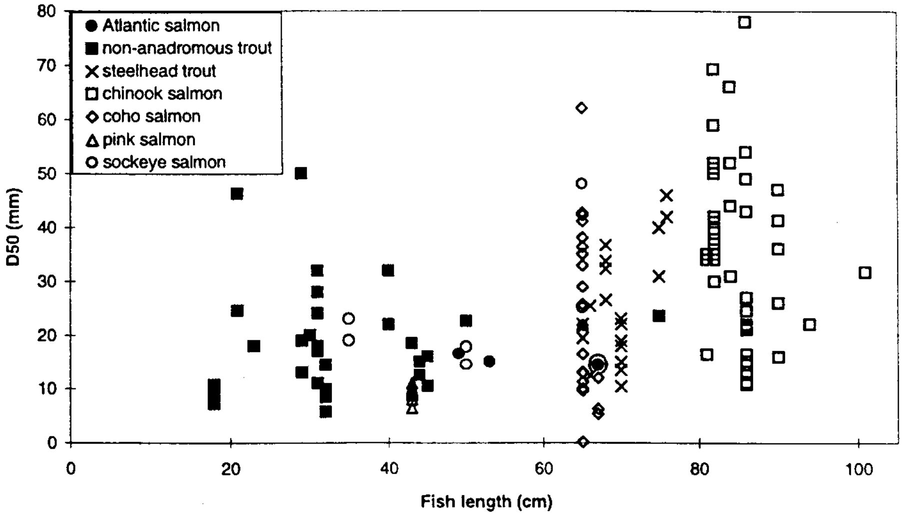


Figure 6: Plot of median sediment diameter against female fish length. Girnock data is circled. (After Kondolf and Wolman, 1993)

Hydraulic Controls on Spawning Sites

Figure 7 shows frequency diagrams for the depth, velocity and Froude number utilisation by spawning salmon. Mean utilised depth, velocity and Froude number at the Girnock are 0.248m, 0.536ms^{-1} and 0.344 respectively. These figures are well within the ranges of hydraulic data presented for salmon in the literature.

Figure 8 shows mean velocity, depth and Froude number utilisation at the Girnock along with selected data from the literature. The general trend is that as the utilisation velocity increases, so does the depth, which follows the lines of constant Froude number. Indeed, the trend line through all of the data points (y = 0.848x +

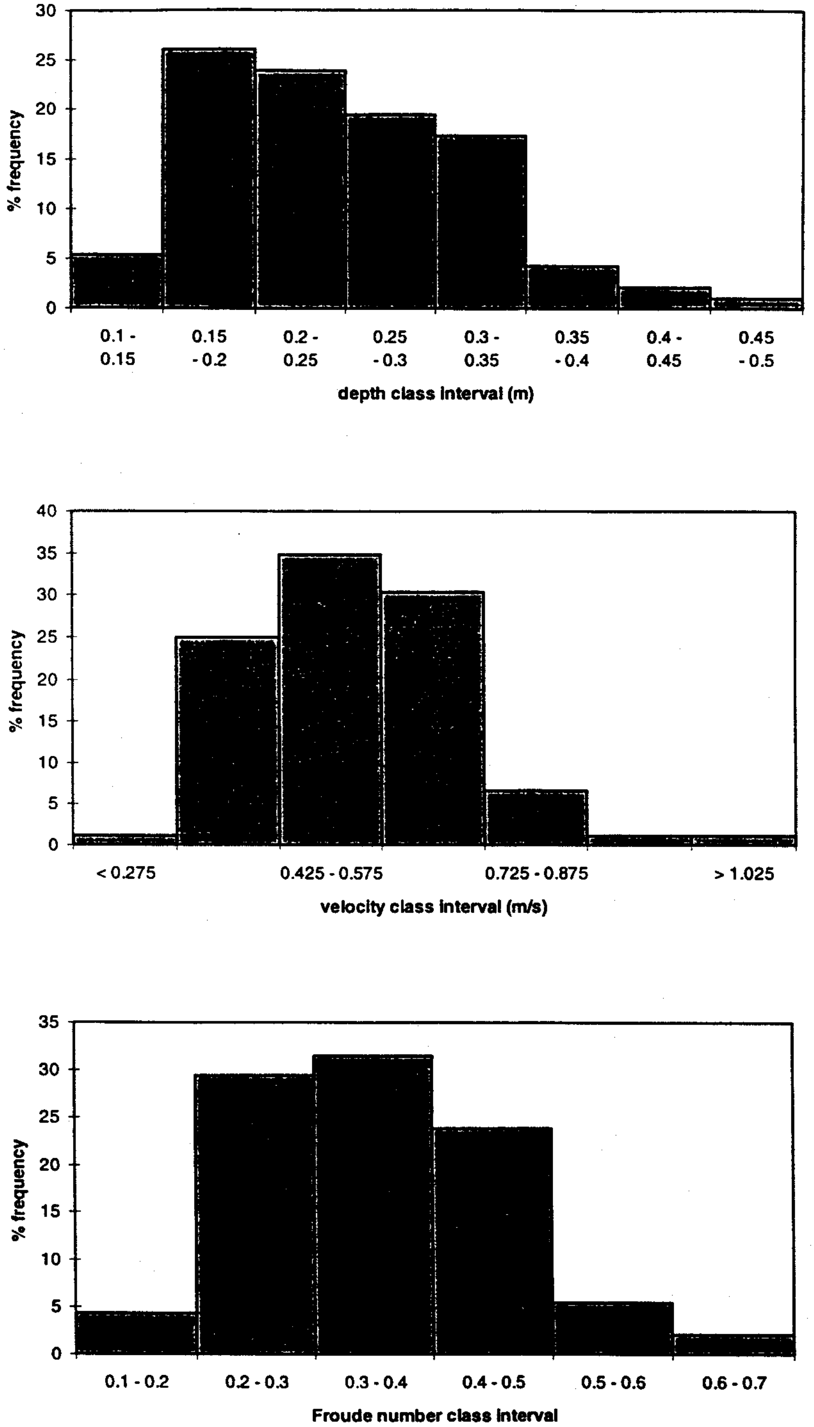


Figure 7: Frequency diagrams of depth, velocity and Froude number utilisation by Atlantic salmon at the Girnock Burn.

0.225, $r^2 = 0.431$, $p < 0.025$) is very similar to the line of constant Froude number 0.3 ($y = 0.850x + 0.251$, $r^2 = 0.991$ for a linear fit). This suggests that fish may select a particular combination of depth and velocity values, which is described by the Froude number. The Froude number also has the advantage that it is dimentionless and can therefore be applied to a range of sizes of river and fish species. As Crisp and Carling (1989) have shown, larger salmonids will tend to spawn in deeper and faster flowing water than smaller fish. However, it may be the case that Froude number utilisation for small and large salmonids are quite similar. The Froude number may thus be a more useful general descriptor of salmonid habitat than the individual combination of depth and velocity.

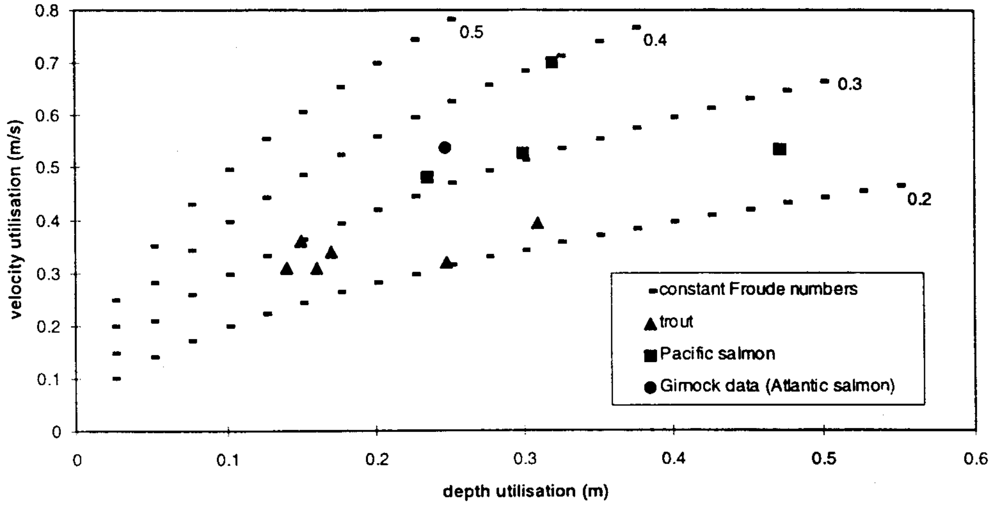


Figure 8: Plot of mean velocity and depth utilisation by salmonids from data obtained at the Girnock Burn and from the existing literature (After Burner, 1953; Grost *et al*, 1990; Kondolf, 1988; Parsons and Hubert, 1988; Reiser, 1976; Reiser and Wesche, 1977; Shirvell and Dungey, 1983; Witzel and MacCrimmon, 1983)

A problem with the data presented above is that it only represents habitat utilisation. Habitat utilisation is a function of habitat availability and habitat preference, variables which are not taken into account (deGraff and Bain, 1986; Parsons and Hubert, 1988). Many studies do not take habitat availability into account when characterising hydraulic habitat (e.g. Witzel and MacCrimmon, 1983; Crisp and Carling, 1989) and the ones that do (e.g. Parsons and Hubert, 1988; Grost *et al*, 1990) tend to treat it as a static variable. However, large temporal fluctuations in available habitat can occur with moderate changes in river flow (Stalnaker *et al*, 1994). Furthermore, as has been shown above, river flow at the Girnock during the spawning period significantly affects the subsequent redd distribution. Thus, if utilisation curves are to be calibrated for habitat availability in order to develop preference relationships, then measurements of habitat utilisation and availability should be carried out at the same river flow. Alternatively, a hydraulic model could be used to calculate the available habitat for the flow under which habitat utilisation was measured.

IMPLICATIONS

This study has shown that Atlantic salmon spawning habitat in the Girnock has distinct physical characteristics, which are in broad agreement with data in the literature. At the tributary scale average reach bed slope and river flow during the spawning period appear to affect the distribution of spawning sites. At the local scale, the nature of hydraulic (flow depth, velocity and Froude number) and sedimentary controls on spawning habitat are important. The Froude number also seems to be a good descriptor, since it is dimentionless and can be applied to a range of sizes of river and fish species. It must be stressed, however, that these are only the physical controls on spawning habitat. Other biological variables such as predation and intra-species competition may have a significant, sometimes dominant effect on spawning site location. Also, it is important to remember that habitat availability probably has an influence on habitat utilisation. It may be that in an upland stream such as the Girnock, habitat availability has a greater control on spawning site selection that habitat preferences of salmon. It is clear that variations in flow and the number of spawners within a river mean that fish have to be flexible in their selection of spawning habitat. Thus, salmonids may spawn within the constraints imposed upon them by the physical character and flow regime of the river and within broader species and size related habitat tolerances. However, habitat availability is something that was not taken into account in this study, though the influence of this factor is currently being examined. One of the ways in which the influence of habitat availability on habitat use is to be examined is by the application of the model PHABSIM, which is also being employed to test the validity of the results on physical characteristics of spawning habitat described, above.

The preservation of natural geomorphological features is essential to the maintenance of the spawning habitat in river channels. A catchment based approach to sediment management should be adopted in order to limit the detrimental effects of fine sediment on spawning habitat produced by landuse practices such as afforestation, moorland gripping, road construction and over-grazing (Werrity *et al*, 1994). The information on the physical characteristics of spawning habitat described above can be used as a guideline in the creation of artificial spawning channels. However, it must be stressed that the data presented is only from one stream. Information from other tributaries of the Dee and other nearby rivers is currently being gathered to examine if and how salmon spawning habitat utilisation varies with location. There may indeed be considerable differences in utilisation between streams, although tributary differences in habitat availability may account for this. The effects of river flow on the spawning distribution outlined above may be useful in a regulated rivers whereby flows during the spawning season could be controlled to optimise conditions for a favourable distribution of redds.

REFERENCES

Adams, J.N. and R.L. Beschta (1980). Gravel bed composition in Oregon coastal streams. *Canadian Journal of Fisheries and Aquatic Science.* 37, pp. 1514-1521

Burner, C.J. (1951). Characteristics of spawning nests of Columbia River salmon. *U.S. Fish and Wildlife Service Bulletin.* 61, pp. 97-110

Chapman, D.W. (1988). Critical review of variables used to define effects of fines in redds of large salmonids. *Transactions of the American Fisheries Society.* 117(1), pp. 1-21

Church, M.A., D.G. McLean and J.F. Wolcott (1987). River bed gravels: sampling and analysis. in Thorne, C.R., J.C. Bathurst and R.D. Hey (eds) *Sediment transport in gravel-bed rivers.* pp. 43-87

Crisp, D.T. (1993). The environmental requirements of salmon and trout in fresh water. *Freshwater Forum.* 3(3), pp. 176-202

Crisp, D.T. and Carling, P.A. (1989). Observations on siting, dimensions and structure of salmonid redds. *Journal of Fish Biology.* 34, pp. 119-134

Dee Salmon Fishing Improvement Association (1994) *Dee Salmon Action Plan.*

deGraff, D.A. and L.H. Bain (1986). Habitat use by and preferences of juvenile Atlantic salmon in two Newfoundland streams. *Transactions of the American Fisheries Society.* 115, pp. 671-681

Gibson, R.J. (1993). The Atlantic salmon in fresh water: spawning , rearing and production. *Reviews in Fish Biology and Fisheries.* 3, pp. 39-73

Grost, R.T., W.A. Hubert, T.A. Wesche (1990). Redd site selection by brown trout in Douglas Creek, Wyoming. *Journal of Freshwater Ecology.* 5(3), pp. 365-371

Hay, D.W. (1995) The current status of salmon stocks in the River Dee. in *Salmon in the Dee catchment: the scientific basis for management.* Atlantic Salmon Trust, Pitlochry, pp. 3-8

Hoopes, D.T. (1972). Selection of spawning sites by sockeye salmon in small streams. *Fishery Bulletin.* 70(2), pp. 447-458

Kondolf, G.M. (1988). *Salmonid spawning gravels: a geomorphic perspective on their size distribution, modification by spawning fish and criteria for gravel quality.* Ph.D. thesis dissertation. Department of Geography and Environmental Engineering, John Hopkins University. pp. 396

Kondolf, G.M. and M.G. Wolman (1993). The sizes of salmonid spawning gravels. *Water Resources Research.* 29(7), pp. 2275-2285

Maitland, P.S., P.J. Boon and D.S. McLusky (eds.) (1994). *The fresh waters of Scotland: a national resource of international significance.* pp.639

Parsons, B.G.M. and W.A. Hubert (1988). Influence of habitat availability on spawning site selection by kokanees in streams. *North American Journal of Fisheries Management,* 8, pp. 426-431

Reiser, D.W. (1976). *Determination of physical and hydraulic preferences of brown and brook trout in the selection of spawning locations.* M.Sc. thesis, University of Wyoming. pp. 112

Reiser, D.W. and T.A. Wesche (1977). Determination of physical and hydraulic preferences of brown and brook trout in the selection of spawning locations. Water Resources Research Insititute, *Water Resources Series 64,* Laramie, Wyoming. pp. 100

Richards, K.S. (1982). *Rivers: form and process in alluvial channels.* pp. 358

Shirvell, C.S. and R.G. Dungey (1983). Micro-habitats chosen by brown trout for feeding and spawning in rivers. *Transactions of the American Fisheries Society.* 112(3), pp. 355-367

Stalnaker, C.B., K.D. Bovee and T.J. Waddle (1994) The importance of the temporal aspects of habitat hydraulics in fish populations. *Proceeding of the 1st international symposium on habitat hydraulics.* pp. 1-11

Warren, J.S. (1985). Hydrology of the river Dee and its tributaries. in D. Jenkins (ed.) *The biology and management of the River Dee.* pp. 23-28

Werrity, A., V. Brazier, J.E. Gordon and J. McManus (1994). Geomorphology. in P.S. Maitland, P.J. Boon and D.S. McLusky (eds.) *The Freshwaters of Scotland: a national resource of international significance.* pp. 65-88

Witzel, L.D. and H.R. MacCrimmon (1983). Redd-site selection by brook trout and brown trout in southwestern Ontario streams. *Transactions of the American Fisheries Society.* 112, pp. 760-761

Wolman, M.G. (1954). A method of sampling coarse river-bed material. *Transactions of the American Geophysical Union.* 35(6), pp. 951-956

An Evaluation of Habitat Improvement Structures in an Experimental Channel in Newfoundland

J. Mitchell[1], R. S. McKinley[1], G. Power[1], and D. A. Scruton[2]

[1] Department of Biology, University of Waterloo, Waterloo, Ontario, N2L 3G1, Canada
[2] Department of Fisheries and Oceans, Science Branch, P.O. Box 5667, St. John's, Newfoundland, A1C 5X1, Canada

ABSTRACT

Distributional patterns and microhabitat selection of juvenile Atlantic salmon (*Salmo salar*) were investigated in relation to habitat improvement structures in a controlled flow experimental channel at Noel Paul's Brook, Newfoundland. The channel consisted of six replicates each containing three randomly arranged treatments. Each replicate included a control with no habitat modification, a mid-channel treatment with a boulder cluster and low-head barrier dam, and a stream bank treatment with undercut banks and wing deflectors. Daytime bank observations and night counting were used to characterize selection for microhabitat features created by the mid-channel and stream bank improvement structures. The influence of size class, density, discharge and diurnal/nocturnal differences on microhabitat selection were also evaluated. Results showed that at natural densities and low constant discharge salmon parr preferred the stream bank and control over the mid-channel treatments. There were no significant differences when densities were increased (1.5 times) as fish were displaced into less preferred treatments. In a discharge experiment, results showed that at low and medium discharge (0.032 and 0.063 m^3s^{-1}, respectively) no preferences were observed. At high discharge (0.13 m^3s^{-1}) mid-channel was the treatment of choice followed by control and stream bank treatments. In all experiments, greater depths were selected in the stream bank treatment, and the salmon in the mid-channel treatment consistently selected positions closer to cover. At increased densities, higher bottom and nose velocities were selected in the control treatment. During the day salmon selected larger substrates than at night. Among treatments largest substrates were selected in the mid-channel treatment. Three size classes were compared ($FL \leq 5$, $5 < FL \leq 10$ and $FL > 10$ cm). Results showed that larger salmon parr preferred greater depths and were found closer to the improvement structures while young of year selected greater focal depths (height above substrate) and were found at greater distances from the structures. Young of year preferred smaller substrates at all discharges. Results from this study can be used to evaluate habitat features and make recommendations suitable for each life stage of Atlantic salmon.

KEY WORDS: Habitat improvement / *Salmo salar* / Newfoundland / microhabitat / distribution / discharge

INTRODUCTION

Habitat improvement structures are devices used to repair or improve damaged fish habitat and, in some instances, to increase the productive capacity of natural habitats. Many rivers in Newfoundland have been affected by human activities. As a consequence, problems such as channelization, unstable stream banks, heavily silted gravels, and areas completely lacking in instream and overhanging cover, arise. It is obviously important to restore such degraded habitats using the best available techniques to maintain and increase fish productivity. Stream restoration techniques including pool creation, placement of instream and overhanging cover, bank stabilization, and revegetation must be assessed and the problems associated with degraded habitats must be documented to justify the costs involved.

Fish habitat improvement has been common practice in North America for decades but has not been widely used or tested under Newfoundland and Labrador conditions (Bourgeois *et al.*, 1993). Desirable microhabitat conditions can be created by improving the habitat and many techniques have been used with varying degrees of success (Saunders and Smith, 1962; Hunt, 1976; House and Boehne, 1985, 1986; Knudsen and Dilley, 1987; Moore and Gregory, 1988; Armantrout, 1991; Binns, 1994). However, the question arises whether habitat improvement techniques used elsewhere in North America, and developed primarily for trout species, are applicable to local conditions and species, in particular Atlantic salmon. One aim of habitat improvement in natural streams in Newfoundland is to increase productivity to benefit the sport fishery which is a multi-million dollar industry. It is therefore important to determine whether these techniques are appropriate and if they can be improved or modified in any way before they are used in Newfoundland waters.

This study involved the investigation of both macrohabitat use and selection for microhabitat features, created by mid-channel and stream bank improvement structures, by juvenile Atlantic salmon (*Salmo salar*). The objectives were, firstly, to determine the applicability of habitat improvement techniques to local conditions and species by evaluating habitat preferences in a broad distributional context under varying discharges and densities; secondly, to evaluate microhabitat conditions selected by juvenile Atlantic salmon (*Salmo salar*); and thirdly, to determine the influence of size class and day/night on microhabitat selection and choice of position in relation to the improvement structures. Such information is required to develop habitat suitability curves of salmon parr under the various experimental conditions. Results from this study can be used to evaluate habitat features and make recommendations suitable for each life stage of Atlantic salmon.

STUDY SITE

The Exploits River is the longest river and the largest watershed on the island of Newfoundland. The river is 267 km long and drains an area of 11,272 km^2. An abandoned spawning channel at an incubation facility at Noel Paul's Brook, a large tributary of the middle Exploits, was modified in 1990 (Bourgeois *et al.*, 1993). The channel was brought to uniform gradient (0.41% slope) with an average width of 3m (2.0-3.4 m) and gravel substrate (2.5 - 7.5 cm). The banks of the channel were stabilized with large rip-rap (boulders 15.0 - 60.0 cm diameter). The channel was then divided into six replicates of approximately 20 x 3 m. Each replicate consisted of a control, mid-channel and stream bank treatment (Figure 1).

The control received no further modification and consisted exclusively of rip-rap along the banks and gravel substrate. The mid-channel treatment contained a low-head barrier dam and a boulder cluster of five large boulders (30.0 - 60.0 cm diameter). The stream bank treatment contained two artificial undercut banks (mini-lunkers) and two wing deflectors. The channel was further modified, in 1994, to include overhanging cover in the first three stream bank treatments. Three blinds, constructed of a basic wooden frame covered with camouflage netting, were placed on the streambank, one at each treatment. Treatments were separated by wooden frames into which screens (0.5 cm mesh) could be inserted. This allowed the treatments to be either open or partitioned from each other. At medium discharge, the average depths for the control, mid channel and streambank treatments were 19.92, 23.55 and 23.58 cm, respectively. Average water column velocities were 0.157, 0.164 and 0.174 $m.s^{-1}$ for the control, mid-channel and stream bank treatments, respectively.

A 50m branch of Noel Paul's Brook, behind an island approximately 500 m below the dam, was selected as a reference. The average depth and water column velocity were 18.59 cm and 0.125 $m.s^{-1}$, respectively.

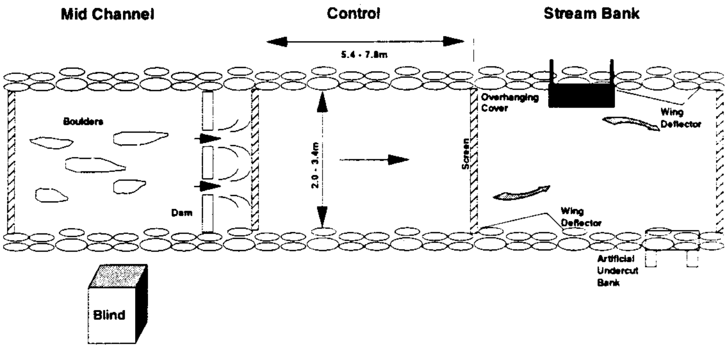


Figure 1. Schematic representation of one replicate showing the control, mid-channel and stream bank treatments.

MATERIALS AND METHODS

Three experiments were performed in 1995. The first experiment involved the natural density of fish in the channel (68 fish/$100m^2$). No fish were introduced or removed from the channel. Fish took up residence voluntarily, and there were no restrictions to the movement of fish in, out or within the channel (i.e. no screens were placed). In the second experiment, the channel was stocked with fish to represent a 1.5 times increase in the natural density (99 fish/$100m^2$), and the uppermost and lowermost screens were kept in place to restrict fish movement to the experimental channel. The fish were allowed at least three days to distribute themselves within the channel before any observations were made. These two experiments were performed in June and July and consisted of five day and five night observations. The temperature throughout these experiments ranged from 12.9-23.5° C and the channel was held at a constant low to medium discharge.

The channel was stocked at the natural fish density in the third experiment, and the fish were exposed to three different discharges: low (0.032 m^3s^{-1}), medium (0.063 m^3s^{-1}) and high (0.13 m^3s^{-1}). The discharge was controlled by an intake valve which was manually opened to increase the flow. This experiment consisted of five night observations at each of the flows and took place from August to October. The temperature range during this experiment was 4.0-18.5° C. Day observations were not possible due to the fish becoming more nocturnal as temperatures decreased.

Day observations, in the channel, consisted of two 30 minute observations at each blind between 9am and 3pm. Polarized sunglasses were used to minimize glare from the water surface and to increase visibility. During observations the holding position, species, length (cm) and focal depth (height off substrate in cm) of each fish were noted and the positions were then marked. Markers consisted of button tags attached to 7/16 inch nuts. Features measured at each holding position included depth (cm), bottom velocity ($m.s^{-1}$ at 1.5cm above substrate), nose velocity ($m.s^{-1}$), water column velocity ($m.s^{-1}$ at 0.6 depth), substrate diameter (cm), distance from nearest structure (cm), distance from cover (cm), and distance from nearest fish (cm). Velocity was measured with an American Sigma velocity meter.

The distribution and number of fish in the experimental channel and reference site were determined by night counting. Night counts were performed 2 hours after sunset at approximately 12 am. Counts began at the downstream end and consisted of two people walking slowly up either side of the channel or the reference. Light was provided by a Coleman Powerhouse Lantern 290A700C which was swept slowly over the water surface. Fish were counted and divided into three sizeclasses: $FL \leq 5cm$, $5 < FL \leq 10cm$, and $FL > 10cm$ (where FL=fork length). Double counting was avoided by nudging counted fish downstream. Holding positions of fish in those treatments with blinds were marked (as above) and the features were measured the following day.

At the completion of each experiment the screens were placed during the night to block off each treatment type. The next day the fish were removed from each treatment by electrofishing with a Smith Root VIII A, 12 volt DC unit. Each fish was identified, measured to the nearest mm and weighed to the nearest 0.1 g using an Acculab Model 5001 electronic digital balance. Scale samples were collected from fish measuring 5cm or more. The following night, a count was conducted to determine the number of fish left after electrofishing.

Habitat availability was measured in the three treatments by recording depth, bottom velocity, water column velocity and substrate diameter along equally spaced transects. Habitat availability and use were derived from frequency analysis (Bovee, 1986). Habitat preference was computed as the ratio between the observed relative frequencies from the habitat use and availability histograms at each interval. The resulting ratio was then normalized and a curve fitted to the mid-point at each interval.

RESULTS

Distributional patterns

At natural densities, the number of fish counted in the channel and the reference site were 50.44±7.77 and 10.68±1.13 fish/100m^2, respectively (average ±S.D.). There was a significant difference in the number of fish per treatment (as estimated by night counting) in the channel (ANOVA P=0.021). The number of fish differed significantly between the mid-channel and the stream bank treatments. On average (±S.D.), 10.57±2.94, 8.53±4.45 and 11.06±3.38 fish were counted in the control, mid-channel and stream bank treatments, respectively (Figure 2).

On average, the number of fish counted in each treatment at 1.5 times the natural density were 8.90±3.61, 9.50±5.15 and 8.73±4.94 in the control, mid-channel and stream bank treatments, respectively (Figure 2). The results showed that there were no significant differences between the treatments (P=0.796).

Results showed that at natural densities and varying discharge the number of fish counted in each varied from one replicate to the next (replicate x treatment interaction P<0.001) and therefore there was no overall treatment effect. However, there was a significant discharge effect (P<0.001). The average number of fish counted decreased as the discharge increased, 7.46±2.87, 6.53±2.49 and 5.16±2.14 for low, medium and high discharge, respectively. This occurred due to visibility problems with increasing water turbulence. Using electrofishing data at high discharge, a significant difference was found between the treatments (P=0.011). On average the number of fish counted were 9.33±2.66, 11.00±2.61 and 6.17±1.94 in the control, mid-channel and stream bank treatments, respectively (Figure 3).

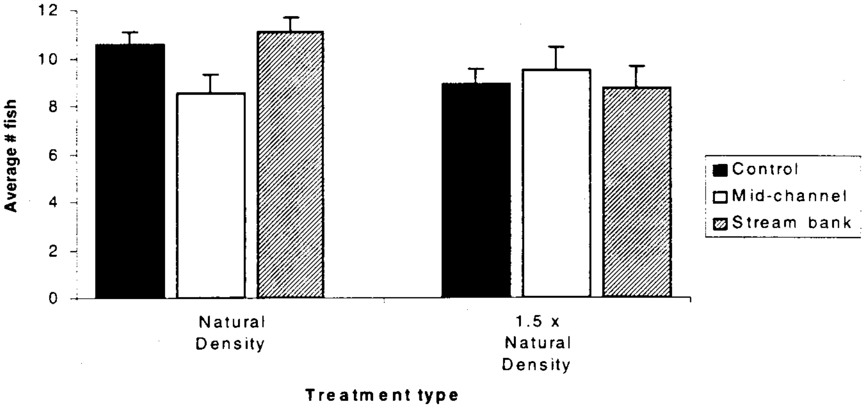


Figure 2. Average number of fish (±S.E.) in the control, mid-channel and stream bank treatments at natural (Expl) and 1.5 x natural density (Exp 2).

Microhabitat use

Using a randomized-block design, a 2-way ANOVA was performed using treatment and time (day and night) as the main effects. For microhabitat variables of Atlantic salmon, at natural density, the results showed that there were differences between treatments in the selection of depth (P=0.001) and distance to the nearest cover (P=0.008). Greater depths were selected in the stream bank treatment (21.6±3.12 cm) and the salmon, in the mid-channel treatment, consistently selected positions closer to cover (34.54±19.19 cm) than in the other treatments (Figure 4). There was also a diurnal difference in the selection of substrate size (P=0.0004). During the day, salmon selected larger substrates (9.8±6.35 cm) than at night (5.64±2.36 cm) (Figure 5).

The influence of size class on microhabitat selection was determined using size class and time as the main effects. The results showed that there were differences between the 3 size classes in the selection of depth (P=0.005), focal depth (P=0.04), and distance to the nearest structure (P=0.001). The larger salmon parr (FL>10cm) preferred greater depths (21.42±4.14 cm) and were found closer to the improvement structures (57.31±15.56 cm). The young of year selected greater focal depths (4.92±2.39 cm) and were found at greater distances from the structures (110.17±21.04 cm).

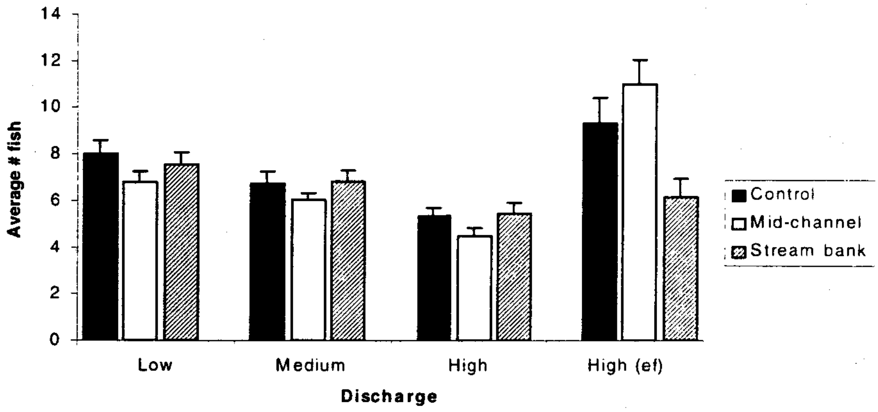


Figure 3. Average number of fish (±S.E.) counted in the control, mid-channel and stream bank treatments at low (0.032 m^3s^{-1}), medium (0.063 m^3s^{-1}) and high (0.13 m^3s^{-1}) discharge. Electrofishing data (ef) is also shown for high discharge.

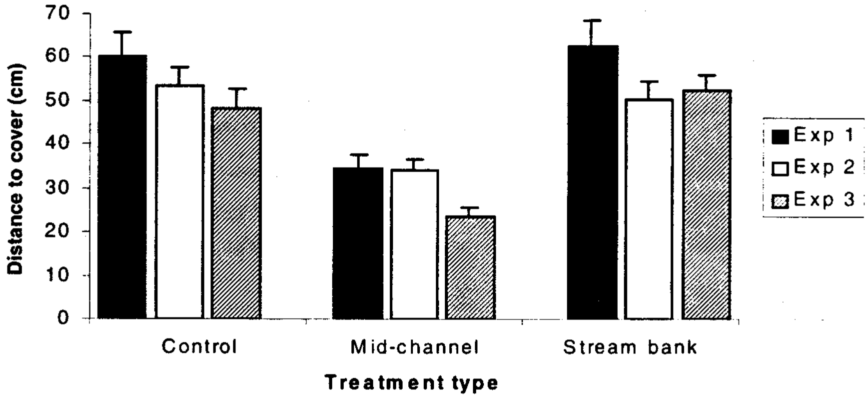


Figure 4. The average distance to the nearest cover (±S.E.) selected by juvenile Atlantic salmon (*Salmo salar*) in the control, mid-channel and stream bank treatments at natural (Exp 1), 1.5 x natural (Exp 2) densities and at natural densities with variable discharge (Exp 3).

At increased densities, there were significant differences between treatments in the selection of depth (P=0.005), bottom velocity (P=0.046), nose velocity (P=0.028) and distance to the nearest cover (P=0.001) (Figure 4). As in experiment #1, greater depths were selected in the stream bank treatment (21.02±1.82 cm) and positions were selected closer to cover (34.28±4.07 cm) in the mid-channel treatment. Higher bottom and nose velocities were selected in the control treatment, 0.087±0.042 and 0.095±0.04 $m.s^{-1}$, respectively.

In the discharge experiment, a 2-way ANOVA was performed using treatment and flow as the two main effects. For microhabitat variables, there was a significant difference between the treatments and flows in the selection of depth (P=0.001), substrate size (P=0.014), distance to the nearest cover (P<0.001) (Figure 4) and distance to the nearest fish (P=0.046). Again greater depths were selected in the stream bank treatment (25.11±3.41 cm) and as the flow increased so did the depth selection. Among the treatments the largest substrates and positions closest to cover were selected in the mid-channel treatment at all flows. As the flows increased the distances to the nearest fish also increased.

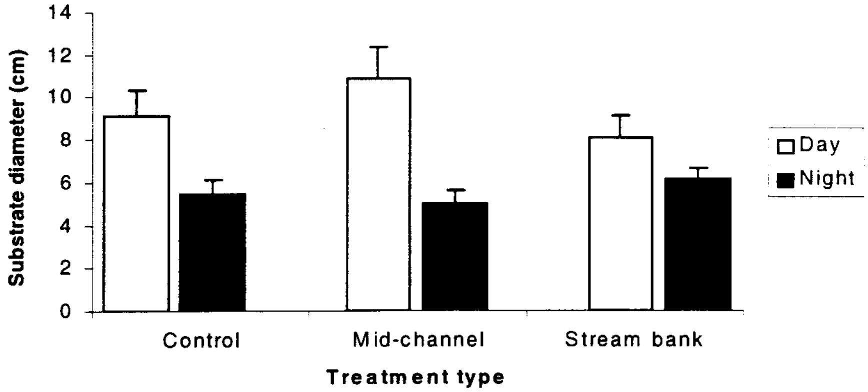


Figure 5. Day and night differences in the selection of substrate diameter and distance to nearest structure (average±S.E.) by 3 size classes of juvenile Atlantic salmon (*Salmo* salar) in the control, mid-channel and stream bank treatments.

Habitat use and availability

Compared to the available habitat in the control treatment, salmon preferred areas with shallow depths (10-15 cm), low bottom velocities (0.03-0.05 $m.s^{-1}$) and high water column velocities (0.14-0.16 $m.s^{-1}$). In the mid-channel treatment, salmon preferred depths of 15-18 cm, bottom velocities of 0.05-0.15 $m.s^{-1}$, and slow water column velocities (0-0.05 $m.s^{-1}$). Salmon preferred deeper areas (15-21 cm), low bottom velocities (0.03-0.06 $m.s^{-1}$) and high water column velocities (0.09-0.16 $m.s^{-1}$) in the stream bank treatment.

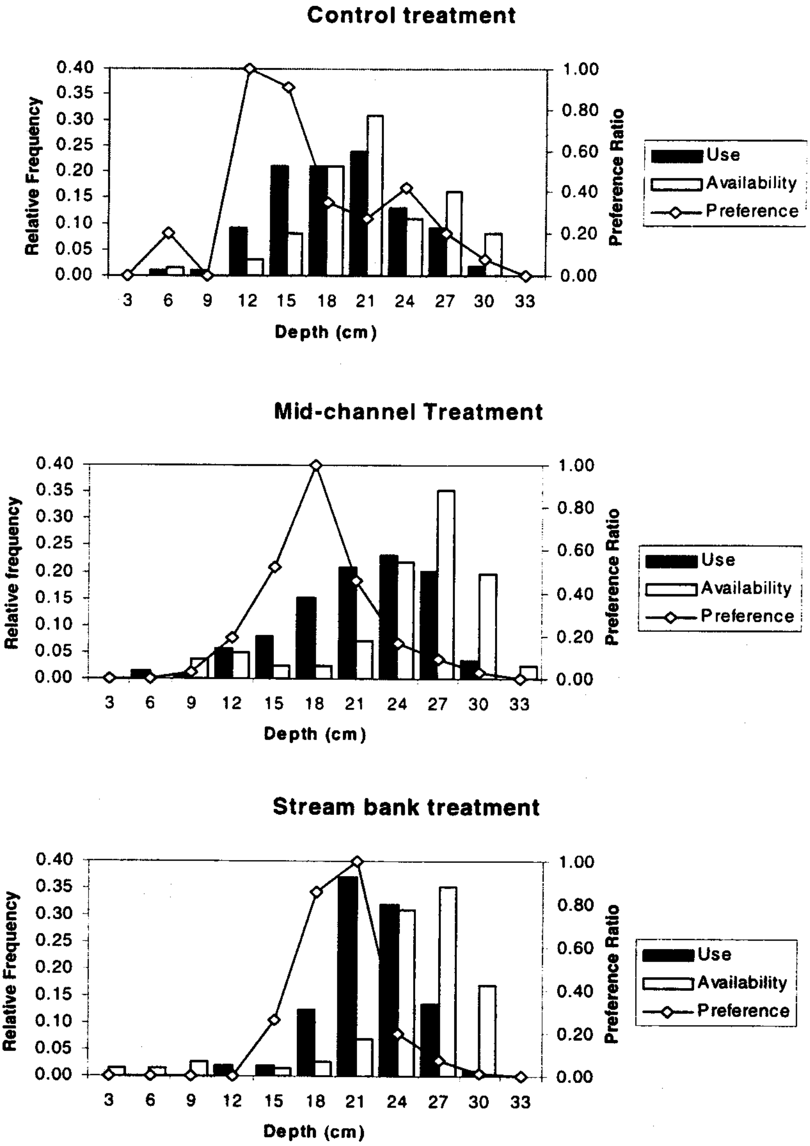


Figure 6. Habitat availability, use and preference of depth (cm) for all size classes in the control (N=110), mid-channel (N=137) and stream bank treatments (N=103) for Exp 1 and Exp 2 combined.

DISCUSSION

The improvement structures used in this study were chosen to create habitat features suitable for the indigenous species of Newfoundland (Buchanan *et al.*, 1989). One such species, Atlantic salmon (*Salmo salar*), has been studied for many years and their habitat requirements are relatively well known (Keenleyside, 1962; Gibson and Power, 1975; Symons and Helan, 1978; Rimmer *et* al., 1984; DeGraaf and Bain, 1986). In general, salmon tend to occupy riffle areas in the centre of streams with a gravel or cobble substrate. However, habitat preference varies depending on the age or size class of the fish. The principle variables influencing habitat use are nose velocity in the summer (DeGraaf and Bain, 1986; Morantz *et al.*, 1987) and substrate size and water depth in autumn (Rimmer *et al.*, 1984). The habitat improvement structures were evaluated to determine whether suitable microhabitat features were created.

The mid-channel treatment consisted of a low-head barrier dam and a boulder cluster. Low-head barrier dams are used to create pools with turbulent surface water that could serve as cover for larger salmonids and to increase oxygen content in the water. Boulders are simple, inexpensive and natural looking. The placement of boulders instream can increase availability and diversity of habitat, provide protective cover for juveniles, and increase the amount of substrate for settlement of benthic invertebrates. Results demonstrated that the mid-channel treatment did not serve its purpose at lower discharges, and as a result was not the treatment of choice. However, as the discharge increased, more salmon took up residence in this treatment. The mid-channel treatment was also preferred by juvenile Atlantic salmon in an initial study (Bourgeois *et al.*, 1993). At increased flows, the large boulders and plunge pools created by the low-head barrier dams, provided a greater diversity in habitat with respect to velocity and cover. The increased surface water turbulence provided more cover and distinct holding areas, with a higher degree of visual isolation than the other treatments. Fish were found further from each other in the mid-channel treatment at higher discharges, because there was more available cover and shelter due to the increased surface water turbulence.

The stream bank treatment contained undercut banks and wing deflectors. In theory, this treatment was more suited to juvenile trout which tend to occupy stream margins with riparian, overhanging and instream cover (Gibson and Power, 1975). Mini-lunkers are used to create artificial undercut banks to increase cover for juvenile and adult salmonids in prime feeding and holding areas. Wing deflectors are intended to increase the quality of salmonid habitat by accelerating channelized streams to their natural meander pattern by improving the sinuosity, increasing the velocity and inducing scouring of deep channels. In our experiment, however, the structures did not create these features to the extent intended. Some scouring occurred but to a minimal extent. This was likely due to the fact that the channel was not subjected to the natural variation in discharge that would have resulted in greater diversity (Bourgeois *et al.*, 1993). Even though the stream bank treatment was the treatment of choice at low to medium discharge, the distinction between the treatments was minimal. The low-head barrier dams in the mid-channel treatments were not effective in creating cover via plunge pools. Therefore, cover was provided by the boulders and rip-rap along the channel banks. However, at low flow, the undercut banks and wing deflectors in the stream bank treatments provided cover and protection as well as the rip-rap along the banks, and so the salmon found more cover and protection there. At increased discharge, fish tended to aggregate in the low flow areas behind the wing deflectors, since there were few other areas providing shelter from the flows in this treatment. As a result, this treatment was avoided at high flows.

In all three experiments, salmon were found closer to cover in the mid-channel treatment, and selected greater depths in the stream bank treatments. The positions selected within the mid-channel treatment were always near cover due to the placement of the boulders in the centre of the treatment. Greater depths were preferred in the stream bank treatment because increased velocities at wing deflectors caused some scouring, and many salmon selected positions at the tips of the deflectors.

At natural densities, salmon preferred larger substrates during the day than at night. Substrate as cover is an important variable for juvenile salmonids (Gibson, 1978; Rimmer *et al.*, 1984; Heggenes, 1988). The larger the substrate the more cover available in the interstitial spaces. During the day, juvenile salmonids are at risk from visual predators, such as piscivorous birds, and therefore need the protection and cover provided by the larger substrates. However, at night substrate size becomes less important as the risk from visual predators decreases. Heggenes (1988) found similar results with brown trout (*Salmo trutta*). Larger substrates can also provide shelter from high water velocities (Gibson, 1978). This was evident when larger substrates were selected in the mid-channel treatment at high discharge. The low temperatures seen during this experiment also caused the salmon to undergo a summer-autumn transition in microhabitat selection. When temperatures fall below 10°C juvenile salmon tend to hide in crevices under stones and they become more nocturnal (Gibson, 1978; Rimmer *et al.*, 1983, 1984; Heggenes and Saltveit, 1990; Fraser *et al.*, 1995). Not only were the salmon sheltering from the increased velocities, but they also began taking refuge in the substrate when the temperature dropped below 10°C.

The three size classes selected different depths with the larger parr selecting deeper areas. Although the values differ, due to the nature of the channel, these results are consistent with earlier works where habitat suitability curves were developed for juvenile Atlantic salmon (Morantz *et al.*, 1987; Scruton and Gibson, 1993). The larger parr were also found closer to improvement structures. As mentioned earlier, the structures could be used as cover and so the larger parr chose positions closer to the structures to take advantage of the varying velocities, depths and cover available to them. Mikheev *et al.* (1994) found that larger juvenile Atlantic salmon remained in shelter for longer periods of time to avoid risky situations. Grant and Noakes (1987) suggested that larger juvenile brook trout (*Salvelinus fontinalis*) were more conspicuous and vulnerable and so showed a greater wariness towards predators. Therefore, the larger parr in the channel selected positions close to the structures to avoid predators.

Habitat selection and distribution are not only dependent on the physical environment, but also related to a number of limiting factors including food availability (Chapman, 1966). The current patterns that developed around the improvement structures may have produced a funnelling effect of drift in the water column. Therefore, it would be advantageous to select a position close to the structures where there would be less energy expenditure in capturing food. Due to interspecific competition, the larger salmon parr selected the best positions with respect to food availability and cover; whereas the smaller subordinate fish avoided competition by feeding in less preferred areas (Fausch, 1984; Metcalfe *et al.*, 1986, 1989). Since salmon are territorial (Kalleberg, 1958; Keenleyside, 1962), the larger parr defended these prime positions and so the smaller salmon were prevented from using the structures.

Higher bottom and nose velocities were selected in the control treatment in the second experiment with increased densities. Higher velocities were more available in this treatment as there were no structures present to provide a diversity of velocities. Salmon were displaced into all treatments, and so less preferred positions were selected in the control treatment due to increased competition for the best positions. During this experiment, the numbers of fish counted in each treatment did not represent the 1.5 increase in density (Figure 2). Two possible explanations for this include: first, due to the increased numbers, the dominant fish took up residence quickly while the subordinates were forced into hiding to avoid the intense competition, thus minimizing energetic expenditure (Metcalfe *et al.*, 1986); and second, although the uppermost and lowermost screens were in place, the fish could have escaped the channel and lower numbers counted as a result.

As seen in this experiment, discharge is an important factor in the performance of habitat improvement structures. Sufficient flows are required to produce the desired effects, such as the scouring at wing deflectors and the creation of plunge pools and surface water turbulence by the low-head barrier dams. Habitat improvement techniques are not always successful and can cause harm if not properly planned and implemented (Frissell and Nawa, 1992). Therefore, both biological and physical parameters need to be taken into consideration when planning a stream improvement project. Biological considerations include: target species and life-stage involved, knowledge of limiting factors, and microhabitat features of the species' natural habitat. Physical considerations include channel morphology and hydraulic parameters.

In this study, the salmon did use the habitat improvement structures. The mid-channel treatment was successful in providing cover and low velocity microhabitats at higher discharge. The wing deflectors and undercut banks also provided cover in the stream bank treatment but to a lesser extent at higher flows than the mid-channel treatment. A diversity of depths were created by the structures and as a result the different size classes were able to select a variety of positions. However, for the structures to function optimally, sufficient discharge is required to produce the microhabitat features intended. Minimal differences were observed between all treatments at the lower discharges. When subjected to the natural variations in discharge of Newfoundland waters, the improvement structures will create a greater diversity of habitat, such as deeper plunge pools and more cover, which was observed to a limited extent under the controlled conditions in the experimental channel.

ACKNOWLEDGEMENTS

Funding for this study was provided by the Department of Fisheries and Oceans, St. John's, Newfoundland. We gratefully thank Paul Rose, Maria Colavecchia and Kim Connors for field assistance.

REFERENCES

Armatrout, N.B. (1991). Restructuring streams for anadromous salmonids. *Am. Fish. Soc. Symp.* 10: 136-149.

Binns, N.A. (1994). Long-term responses of trout and macrohabitats to habitat management in a Wyoming headwater stream. *North American Journal of Fisheries Management* 14: 87-98.

Bourgeois, C.E., D.A. Scruton, D.E. Stansbury and J.M. Green (1993). Preference of juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) for two types of habitat improvement structures, p. 103-108. *In* R.J. Gibson and R.E. Cutting [ed.] Production of juvenile Atlantic salmon, *Salmo salar*, in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. Instream Flow Information Paper 21. U.S. Fish Wildl. Serv. Biol. Rep. 86(7). 235 pp.

Buchanan, R.A. and D.A. Scruton (1989). A technical manual for small stream improvement and enhancement in Newfoundland and Labrador. Inshore Fisheries Development Agreement. pp. 108.

Chapman, D.W. (1966). Food and space as regulators of salmonid populations in streams. *Amer. Natur.* 100: 345-357.

DeGraaf, D.A. and L.H. Bain (1986). Habitat use and preference of juvenile Atlantic salmon in two Newfoundland rivers. *Trans. Am. Fish. Soc.* 115: 671-681.

Fausch, K.D. (1984). Profitable stream positions for salmonids: relating specific growth rate to net energy gain. *Can. J. Zool.* 62: 441-451.

Fraser, N.H.C., J. Heggenes, N.B. Metcalfe, and J.E. Thorpe (1995). Low summer temperatures cause juvenile Atlantic salmon to become nocturnal. *Can. J. Zool.* 73: 446-451.

Frissell, C.A. and R.K. Nawa (1992). Incidence and causes of physical failure of artificial habitat structures in streams of Western Oregon and Washington. *North American Journal of Fisheries Management* 12: 182-197.

Gibson, R.J. (1978). The behaviour of juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) with regard to temperature and water velocity. *Trans. Am. Fish. Soc.* 107(5): 703-712

Gibson, R.J. and G. Power (1975). Selection by brook trout (*Salvelinus fontinalis*) and juvenile Atlantic salmon (*Salmo salar*) of shade related to water depth. *J. Fish. Res. Bd. Can.* 32: 1652-1656.

Grant, J.W.A. and D.L.G. Noakes (1987). Escape behaviour and use of cover by young-of-the-year brook trout, *Salvelinus fontinalis*. *Can. J. Fish. Aquat. Sci.* 44: 1390-1396.

Heggenes, J. (1988). Substrate preferences of brown trout fry (*Salmo trutta*) in artificial stream channels. *Can. J. Fish. Aquat. Sci.* 45: 1801-1806.

Heggenes, J. and S.J. Saltveit (1990). Seasonal and spatial microhabitat selection and segregation in young Atlantic salmon, *Salmo salar* L., and brown trout, *Salmo trutta* L., in a Norwegian river. *J. Fish. Biol.* 36:707-720.

House, R.A. and P.L. Boehne (1985). Evaluation of instream enhancement structures for salmonid spawning and rearing in a coastal Oregon stream. *North American Journal of Fisheries Management* 5: 283-295.

House, R.A. and P.L. Boehne (1986). Effects of instream structures on salmonid habitat and population in Tobe Creek, Oregon. *North American Journal of Fisheries Management* 6: 38-46.

Hunt, R.L. (1976). A long-term evaluation of trout habitat development and its relation to improving management-related research. *Trans. Am. Fish. Soc.* 105(3): 361-364.

Kalleberg, H. (1958). Observations in a stream tank of territoriality and competition in juvenile salmon and trout (*Salmo salar* L. and *S. trutta* L.). *Fish. Bd. Swed. Inst. Freswat. Res. Drottingholm* 39: 55-98.

Keenleyside, M.H.A. (1962). Skin-diving observations of Atlantic salmon and brook trout in the Miramichi River, New Brunswick. *J. Fish. Res. Bd. Canada* 19(4): 625-634.

Knudsen, E.E. and S.J. Dilley (1987). Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. *North American Journal of Fisheries Management* 7: 351-356.

Metcalfe, N.B. (1986). Intraspecific variation in competitive ability and food intake in salmonids: consequences for energy budgets and growth rates. *J. Fish. Biol.* 28: 525-531.

Metcalfe, N.B., F.A. Huntingford, W.D. Graham and J.E. Thorpe (1989). Early social status and the development of life-history strategies in Atlantic salmon. *Proc. R. Soc. Lond.* 236: 7-19.

Mikheev, V.N., N.B. Metcalfe, F.A. Huntingford and J.E. Thorpe (1994). Size-related differences in behaviour and spatial distribution of juvenile Atlantic salmon in a novel environment. *J. Fish. Biol.* 45: 379-386.

Moore, K.M.S. and S.V. Gregory (1988). Response of young-of-year cutthroat trout to manipulation of habitat structure in a small stream. *Trans. Am. Fish. Soc.* 117: 162-170.

Morantz, D.L., R.K. Sweeney, C.S. Shirvell, and D.A. Longard (1987). Selection of microhabitat in summer by juvenile Atlantic salmon (*Salmo salar*). *Can. J. Fish. Aquat. Sci.* 44: 120-129.

Rimmer, D.M., U. Paim, and R.L. Saunders (1983). Autumnal habitat shift of juvenile Atlantic salmon (*Salmo salar*) in a small river. *Can. J. Fish. Aquat. Sci.* 41: 469-475.

Rimmer, D.M., U. Paim, and R.L. Saunders (1984). Changes in the selection of microhabitat by juvenile Atlantic salmon (*Salmo salar*) at the summer-autumn transition in a small river. *Can. J. Fish. Aquat. Sci.* 41: 469-475.

Saunders, J.W. and M.W. Smith (1962). Physical alteration of stream habitat to improve brook trout production. *Trans. Am. Fish. Soc.* 91: 185-188.

Scruton, D.A. and R.J. Gibson (1993). The development of habitat suitability curves for juvenile Atlantic salmon (*Salmo salar*) in riverine habitat in insular Newfoundland, Canada. *In* R.J. Gibson and R.E. Cutting [ed.] Production of juvenile Atlantic salmon, *Salmo salar*, in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118.

Symons, P.E. and M. Heland (1978). Stream habitats and behavioural interactions of under yearling and yearling Atlantic salmon (*Salmo salar*). *J. Fish. Res. Bd. Can.* 35: 175-183.

A Study on the method of evaluating fish habitat environment by PCA under various river environmental factors

Hironori MATSUZAKI
Department of Civil Engineering, University of Tokyo
7-3-1, Hongo, Bunkyo-ku, 113 Japan
E-mail: matuzaki@hydra.t.u-tokyo.ac.jp

Nobuyuki TAMAI
Department of Civil Engineering, University of Tokyo
7-3-1, Hongo, Bunkyo-ku, 113 Japan
E-mail: tamai@hydra.t.u-tokyo.ac.jp

ABSTRACT

This paper presents a new method for evaluating fish habitat environments. This method has improved three following faults of IFIM which is a current method. 1)The change in the river environment is considered to be a flow discharge change. 2)Only three factors, flow velocity, depth, and substrate are assumed to represent the river environment. 3)The influence by mutual relation of environmental factors is disregarded.

The proposal method is especially effective in the region like Japan where environments are various in each section along the river, and there are various fish species. Authors observed amount of fish habitat and river environments through year in one river. When the obtained data was analyzed by the principal component analysis(PCA) method, mutual relation between environmental factors and the amount of fish habitat was clarified.

And, this analysis result explains fishes' actual modes of life of each fish species every season. That is, how which environmental factors exert the influence on the amount of fish habitat is shown. Authors adapt the characteristic of such a principal component analysis for our proposal method by weighting factors according to the size of the vector of the factor loading. The method of fish habitat potential was calculated by using the weighted principal factor method and fishes' preference curves which are used also with IFIM.

KEY-WORDS: various environmental factors, various fish species, principal component analysis, factor loading, biological diversity, ecological niche

INTRODUCTION

A biological diversity is thought to be related stabilizing of the ecosystem at present. The living thing need to adjust to the a new environment when some environmental changes occur such as temperature rise, poison mixing, and entering into of living thing from another place. Abundant biological diversity has a high application ability to environmental changes. Therefore, various environments with various living things expand the habitat place by generating the struggle for existence among individuals and species. In other words, the ecological niche expands. And the variety of such an environment raise biological diversity. Consequentially, potential of the ecosystem is maintained.

Japan belongs to the temperate monsoon zone, and various fish species which adjusts to various environments inhabit in the region where the river environment changes remarkably. The observed shape of the preference curve actually becomes approximately similar to a physiological preference curve in the river where the fish species are few. On the other hand when various fish inhabit, observed curves are considerably biased from the physiological preference curve. Because, even if physiological suitability is somewhat sacrificed, the species request the scene of existence to the place where the interspecies competition is few if the load by the competition with another species is larger than physiological suitability. There is IFIM(Instream Flow Incremental Methodology) in the one to represent the method of the habitat environment evaluations in the river. This is the one developed to evaluate the habitat environment of salmon originally in North America. It is a method of calculating the habitat potential only by the flowing quantity change (Nestler et *al.*,1989).

We know that hydraulic parameters affect the fish habitat potential most. However, as for other environmental factors we need to judge empirically in each time. The method of the habitat environment evaluation in the river where many kinds of fish inhabit under various environments is desired such a current status.

FILED OBSERVATION

The field observation was done in Oto river(basin area:271km^2, length:34km, mean slope of upstream:1/45, mean slope of downstream:1/350) is located in the central districts in Japan as shown in Figure 1.

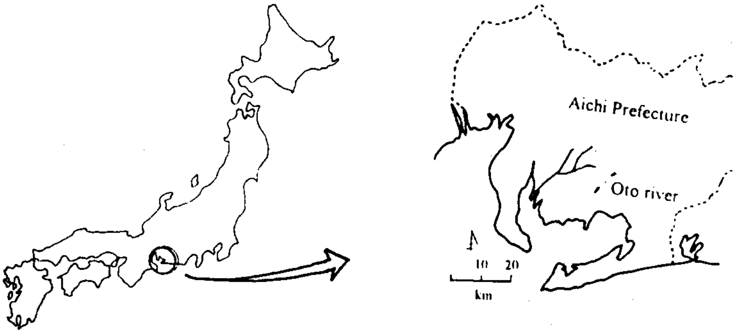


FIGURE 1: Map of Japan showing the location of Oto river field observed

The observed fish species was 29 kinds. The pale chub(Zacco platypus), the dark club(Zacco temminckii) and the river lizard goby(Rhinogobius flumineus) were observed in all the observation stage through a year, and there were a lot of numbers. Therefor, these tree kinds of fish were judged to be fish species by which Oto river was represented. Moreover, six factors such as the velocity, the depth, the substrate value, the cover rate, the vegetation value, and the distance between riffles and pools were selected as a representative factor which showed the river environment.

Depth and flow velocity are mean values of each observation stage. The substrate value is calculated from the amount of substrate and index number(detritus:1, mud:2, silt:3, sand:4, fine gravel:5, gravel:6, boulder:7, bedrock:8). The vegetation value is calculated from the length of riverside vegetation and the index number(none:1, grass:2, tree:3). The cover rate is value by which the area of riverside vegetation shadow to river surface divided by the total area.

METHOD OF EVALUATING HABITAT ENVIRONMENT

Principal Component Analysis(PCA)

The above mentioned habitat density of three fish species is used to be a criterion variate, six environmental factors are used to be an explanatory variate, and principal component analysis is done. Principal component analysis is one of analyses used to analyze related to between a lot of factors and to consolidate the factor. In principal component analysis, the original data is consolidated in a new factor like the 1st principal ingredient and the 2nd principal ingredient, etc. reducing the loss of information on the original data as much as possible by examining the relations between factors. The contribution rate which is the numerical value by how a new factor explains original information is shown can be calculated. The contribution rate of a new factor to the n-th principal ingredient is added and this value is called an accumulative contribution rate. A new factor to the n-th principal ingredient is selected based on this accumulative contribution rate. In general, it is a standard that the accumulative contribution rate exceeds about 80%.

Moreover, the correlation coefficient of selected new factor M pieces and explanatory variate is called the factor loading. N pieces of environmental factors which influences the habitat density in six original environmental factors are selected from the vector relation of the factor loading. Principal component analysis can be used to know not only consolidating the factor but also mutual relation of a multi variate data not clear of the relation.

Weighted Principal Factor Method

M pieces of new factors are assumed to be coordinate axis, and the size of the vector of the factor loading is defined to be weight of the selected original environmental factor. The highest size of vector is matched to one and then other weights should be adjusted. The preference curve to the selected factor which can be used also with IFIM can be made, or the one which has already been made is good for use. As mentioned above is done and fish habitat potential S is defined by the following equation based on the obtained numerical value.

(1)
$$S = \frac{1}{N}\sum_{i=1}^{N}\{a_i \times f_i\}$$

where S : inhabiting potential , N : number of selected factors, a_i : weight,
f_i : inhabiting probability obtained from preference curve

RESULTS AND DISCUSSION

The accumulative contribution rates of the 1st principal ingredient and the 2nd principal ingredient obtained as a result of principal component analysis to three fish species are shown in Table 1.

TABLE 1: The accumulative contribution rate till the 2nd principal ingredient (%)

fish species	spring	summer	autumn	winter
pale chub	68.7	69.4	65.5	62.9
dark club	68.3	70.9	64.3	67.9
river lizard goby	69.8	69.3	70.7	67.8

It is understood that almost values are from 65% to 70%. That is, these two principal ingredients explain 65-70% of the original data. This rate is not so enough statistically. It is easy to increase principal ingredient and to raise the rate. Actually, the third principal ingredient let the accumulative contribution rate raise up to 80%. However, it is no use to raise the rate terribly. The reason is that the original environmental factors selected at the end increase, and then the meaning by which the factors are consolidated is lost. Moreover, the contribution rate contains fuzziness as long as the object is living things even if mathematical significant exists. A little environmental fuzziness is rather convenient for various fish species because they expand own ecological niche by the competition among individuals and species.

Next, an environmental factor which exerts the influence on the amount of habitat is chosen. First of all, thinking about the direction of the vector of the factor loading to each environmental factors on the plane where the 1st and the 2nd principal ingredient are coordinate axes. These six vectors show the relation among environmental factors and the amount of fish habitat in information consolidated in the 1st and the 2nd principal ingredient. It can be judged that vectors have a mutually independent relation when the vectors meet with right angles. The other way, the parallel vectors can show subordination and influence mutually. For example in Figure 2, it can be understood that the substrate rate and the distance between riffles and pools influence on the 1st principal ingredient largely.

In the same way, Axes of the pale chub are assumed. The axis along the vector of the habitat density of the pale chub is defined as 1st axis, and the other which is at right angles with 1st axis is defined as 2nd axis. Environmental factors of the vector along the 1st axis influence on the habitat density, the other hand those which along the 2nd axis have few influence. It was confirmed that the mode of life every season for instanc, laying eggs and preying, agree with results of analysis for each fish species. Selected environmental factors and weights are shown in Table 2.

Taking an example of Figure 2, how to decide of habitat potential is shown. The preference curve to the velocity is shown as Figure 3, and one to the depth is shown as Figure 4. The curve of Figure 3 is assumed *f(v)* and one of Figure 4 is assumed *g(d)*The weight of the velocity is 1.00 and the weight of depth is 0.82 from Table 2. Then the habitat potential becomes equation (2). It can be imagined as Figure 5.

TABLE 2: Selected factors and weight

fish species	velocity	depth	substrate value	cover rate	vegetation value	distance between riffles and pools
pale chub (spring,summer)	1.00	0.82				
pale chub (autumn,winter)	0.94	1.00		0.80	0.89	
dark club (though year)			1.00	0.71		0.83
river lizard goby (though year)	1.00	0.67	0.77			

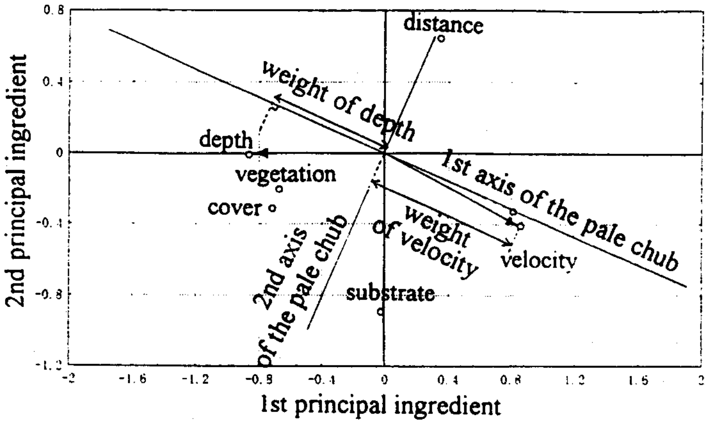


FIGURE 2: The factor loading vectors of environmental factors obtained by PCA

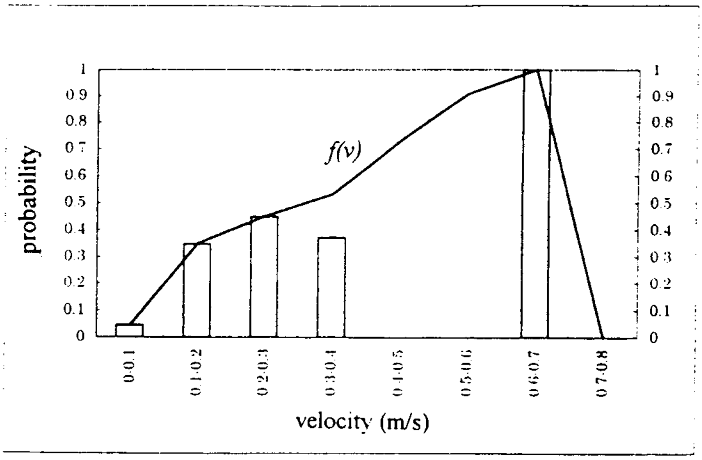


FIGURE 3: The preference curve of the pale chub (spring, summer) to the velocity

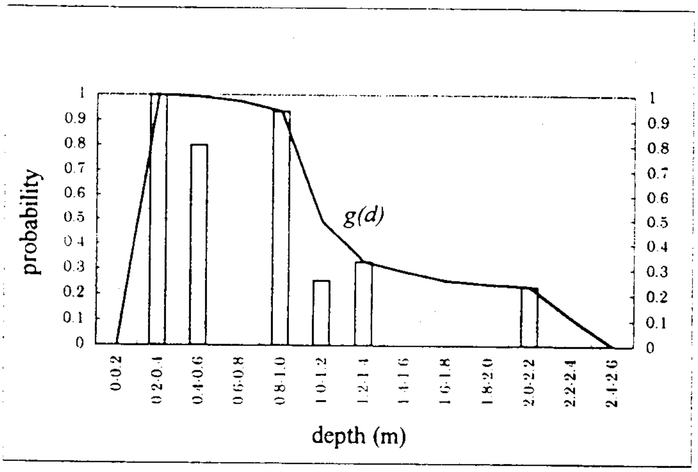


FIGURE 4: The preference curve of the pale chub (spring, summer) to the depth

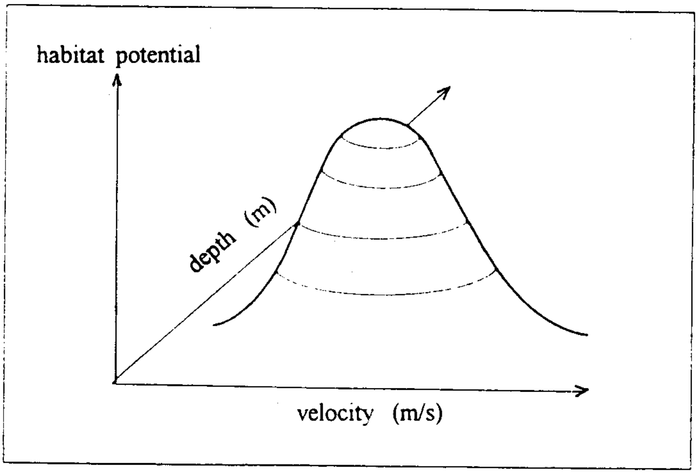


FIGURE 5: The habitat potential (imaginary)

(2) $$S = \{1.00 \times f(v) + 0.82 \times g(d)\} / 2$$

CONCLUSION

The purpose of this study is to have proposed the method of the habitat environment evaluation in the river where various fish species inhabit. The data obtained by field observation has be analyzed by the principal component analysis method. The result of analysis explains relation between the amount of the fish habitat and the river environments clearly. And the result has shown fishes' actual modes of life. Moreover, the relation of an environmental factor which influenced the amount of fish habitat is able to be known intuitively by plotting the analysis result in the two-dimensional graph.

The evaluation method proposed, weighted principal factor method simulates fish's habitat potential as follows.
1) The principal factors with a large influence are selected from some river environmental factors to the amount of habitat of each fish species by principal component analysis.
2) The weight setting based on the size of the vector of the factor loading is done to each selected environmental factor.
3) Adding the values which are obtained by weighting probabilities from the preference curve to selected environmental factors is defined habitat potential.

There are three following advantages when the proposed evaluation method are compared with IFIM. The index which shows various river environments can be taken. The proposed method can evaluate not only a river cross profile but also a whole river or a river section.

ACKNOWLEDGMENTS

Authors like to express our appreciation to technical official Mr. Morita and students in our laboratory for help with field observation. Moreover, the help of the river bureau of the Aichi Prefectural government and the Okazaki public work office in an offer of some data sets.

REFERENCE

Jhon M.Nestler, Robert T. Milhous, and James B. Layzer(1989).Instream Habitat Modeling Techniques, James A. Gore and Geoffrey E. Petts editors, Alternative in Regulated River Management, CRC Press, Baca Raton, Florida

A COMPARISON OF HABITAT SUITABILITY INDICES DEVELOPED FROM DAYTIME VERSUS NIGHTTIME OBSERVATIONS FOR ATLANTIC SALMON IN A REGULATED NEWFOUNDLAND STREAM

L. J. LeDrew[1]

D. A. Scruton
Fisheries and Oceans
P.O. Box 5667, St. John's, Newfoundland, Canada, A1C 5X1

R. S. McKinley[1]

G. Power[1]

[1]Department of Biology, University of Waterloo, Waterloo, Ontario, Canada, N21 3G1

ABSTRACT

Investigations were conducted in July - August, 1995 at West Salmon River, Newfoundland, Canada, to determine: (i) if there is a significant difference in diurnal habitat use/preference during summer by juvenile Atlantic salmon, *Salmo salar*, in a regulated stream; and (ii) whether or not habitat suitability indices (HSI) should be based on criteria which reflect temporal as well as spatial variation in habitat selection. The West Salmon River, is part of the Upper Salmon Hydroelectric Development, and is under a regulated flow regime. Microhabitat use by Atlantic salmon was examined through snorkelling observations conducted during daylight and nighttime hours. Microhabitat variables measured at the focal point of individually observed fish included: water depth; mean water column velocity; focal point velocity; and substrate particle size. Two 30 m x 4 m transects were established in each of three representative habitat types. A total of 55 underwater surveys were conducted over four time intervals and resulted in the observation of 970 Atlantic salmon. HSI curves were developed for Atlantic salmon fry for each of four distinct (6 hour) daily time intervals. The distributions of age classes observed during the nighttime were substantially different from daytime observations. Ninety one percent of all the age 1+ and 88% of all age 2+ salmon recorded were observed at night. The nocturnal behaviour of salmon parr (> 1+) was attributed to avian predator avoidance and was not temperature dependent. Significant differences were found in mean column velocity, depth and substrate use by 0+ salmon at night as compared to day (Kolmogorov-Smirnov two sample test). Results indicate that nighttime as well as daytime surveys of habitat selection behaviour should be considered in construction of habitat suitability indices in order to accommodate diurnal changes in habitat use.

KEY-WORDS: Habitat / Suitability / Nocturnal / Atlantic Salmon / Preference/ Day / Night.

INTRODUCTION

Temporal shifts in behaviour and habitat use by Atlantic salmon during the summer period has received limited attention to date. However, diurnal shifts in habitat use by age-0 brown trout (*Salmo trutta)* are known to occur. Chaston (1968) and Heggenes (1988) suggested that age-0 brown trout hide in the substrate more during the day than at night. Age-0 brown trout may move closer to the stream edge at night in response to larger fish that are active and feeding as do age-0 rainbow trout (*Oncorhryncus mykiss*) (Campbell and Neuner, 1985). Hubert et al. (1994) observed diurnal shifts in summer habitat use by age-0 brown trout. Locations of age-0 brown trout varied between day and night, moving closer to the stream edge at night where current velocities and water depths were less than at sites they occupied during the day. This shift in microhabitat may serve to avoid predators (Powers, 1987). Campbell and Neuner (1985) suggest that age-0 rainbow trout move closer to the stream edge at night in response to larger trout moving inshore to feed.

In his discussion on day/night habitat selection, Heggenes (1994) notes that an interesting phenomenon are profound night/day changes in behaviour and habitat selection in winter, both for brown trout and Atlantic salmon. While passive sheltering in the substrate or aggregating in pools previously was considered the principal winter activity in trout and salmon, this turns out to be daytime behaviour only. Both trout and salmon emerge from their daytime sheltering microhabitats as soon as it gets dark, are active during the night and disappear again as soon as it gets light in the morning (Heggenes 1994, Fraser et al. 1993,). This diurnal behaviour is consistent throughout the winter at low temperatures, and starts when the water temperature falls below 10°C, and is thereafter regulated by light (Heggenes 1994). Considerable spatial and temporal variation in habitat selection is therefore possible and probable, perhaps reflecting a wide tolerated spatial niche but a narrower preferred niche in trout and salmon. Selection is likely to be stronger for important physical factors in short supply, which may vary both in space and time (Heggenes 1994).

These changes in habitat selection and behaviour between summer and winter and day and night, governed by temperature and light, is a result of temperature-dependent changes in the relative costs and benefits of different behavioral strategies (Heggenes 1994). Trout and salmon are vulnerable to a great deal of predation, especially visual feeders (mergansers, osprey etc.) however, their susceptibility is reduced when sheltered in the substrate. When not sheltered, burst swimming speed is the main defence strategy utilized, however, trout and salmon being poikilotherms inhabiting cold water environments, are probably unable to attain maximum burst speed and therefore, are most at risk during periods of low temperature during daylight, hence a switch to nocturnal foraging. In an in-vitro experiment to investigate temperature-dependent switching between diurnal and nocturnal foraging in Atlantic salmon, Fraser et al (1993) demonstrated that juvenile salmon show rapid changes in their daily activity patterns in response to alterations in environmental temperature. At higher water temperatures (typical of late spring, summer and autumn) the fish are active and obtain most of their food during daylight hours. During this time they are aggressive to neighbouring fish and may defend territories. At night they often remain out of refuges and exposed in the water column, but feed relatively little and are unaggressive. Decreasing temperatures caused a drop in daytime foraging rates, however nocturnal foraging rates were independent of temperature, and at low temperatures the fish were making over three times as many feeding sorties by night as by day.

Campbell and Neuner (1995) and Heggenes (1994) suggest that there are indications of possible differences in habitat selection between day and night also in summer in salmonids, although less pronounced. Heggenes (1994) also speculates that brown trout may have a wider spatial niche during the night (in summer) and may also use shallow and slow water at night, often closer to the streambanks. In a Nova Scotia stream, LaCroix (1995) reported the use of slightly deeper water during the day by age 0+ salmon and brook trout, which he attributed to the requirement for greater cover during the day.

Heggenes (1994) concludes that if there is a diurnal shift in habitat selection, then habitat selection models must be developed incorporating diurnal light dependent habitat selection regimes.

Investigations were conducted in July - August, 1995 at West Salmon River, Newfoundland, to determine: (i) if there is a difference in diurnal habitat use/preference during summer by juvenile Atlantic salmon, in a regulated stream; and (ii) whether or not habitat suitability indices (HSI) should be based on criteria which reflect temporal as well as spatial variation in habitat selection.

METHODS

Study Area

The West Salmon River, located in south-central Newfoundland, is part of the Upper Salmon Hydroelectric Development (Figure 1). Prior to the construction of the hydroelectric development, West Salmon River flowed out of Cold Spring and inflowed to Godaleich Pond 9 km downstream and was the main spawning and nursery area for both upstream and downstream salmonid populations (Beak 1980). The construction of the West Salmon Dam in 1982 eliminated movement of salmonids downstream from Cold Spring Pond, and resulted in the loss of spawning and rearing habitat due to flooding (upstream of the dam) and de-watering (downstream of the dam). Landlocked Atlantic salmon, brook trout and Arctic charr *Salvelinus alpinus* are the only salmonids inhabiting the Upper Salmon watershed, with Atlantic salmon and brook trout dominating these fluvial habitats (Beak 1980).

In order to protect the valuable fish habitat downstream from the dam a compensatory flow release agreement was negotiated between Newfoundland and Labrador Hydro and the Department of Fisheries and Oceans. A 2-level controlled flow regime based on the Tennant's Montana Method (Tennant 1975) was adopted which specifies a flow of 1.3 $m^3 s^{-1}$ (20% MAF) between December 1 and May 31, and 2.6 $m^3 s^{-1}$ (40% MAF) between June 1 and November 31. A flow release structure was incorporated into West Salmon Dam which is able to vary flows from 0 - 7.0 $m^3 s^{-1}$. The flow throughout the study period was 2.6 $m^3 s^{-1}$.

The study area was between kilometers 0.5 and 1.5, downstream from the Dam (Figure 2).

Data Collection

Six transects were established in three different habitat types: two in each of pocket water; low gradient riffle; and run habitat (McCain et al. 1990). These habitat types were the predominant habitat in the river (Jacques Whitford

Environment, 1996). Transects, 4 m x 30 m (where possible), were established in an upstream direction at a 30° angle from the perpendicular to the shoreline. A Hugrun continuous recording thermograph recorded water temperatures throughout the study period.

Habitat Use

Habitat use data were collected by snorkelling within the transects. Two snorkellers surveyed the transect at a time, one in the downstream half and one in the upstream half. The downstream snorkeller would begin surveying first, starting at the shore and moving out into the stream, followed by the upstream snorkeller. The downstream snorkeller would stay slightly ahead of the upstream snorkeller throughout the survey to avoid the latter interfering with the visibility of the downstream snorkeller by disturbing silt, etc. The snorkellers communicated to each other the presence of any fish moving in or out of the transect, thus minimizing the potential for double counting. Numbered fluorescent markers were placed on the stream bottom below the head (snout) of each fish observed. The following data were then reported to a recorder on the shoreline: marker number; species; year class; (subjectively determined from fish length); fish length (cm); activity, (whether holding station or active); and position (cm) (height above substrate).

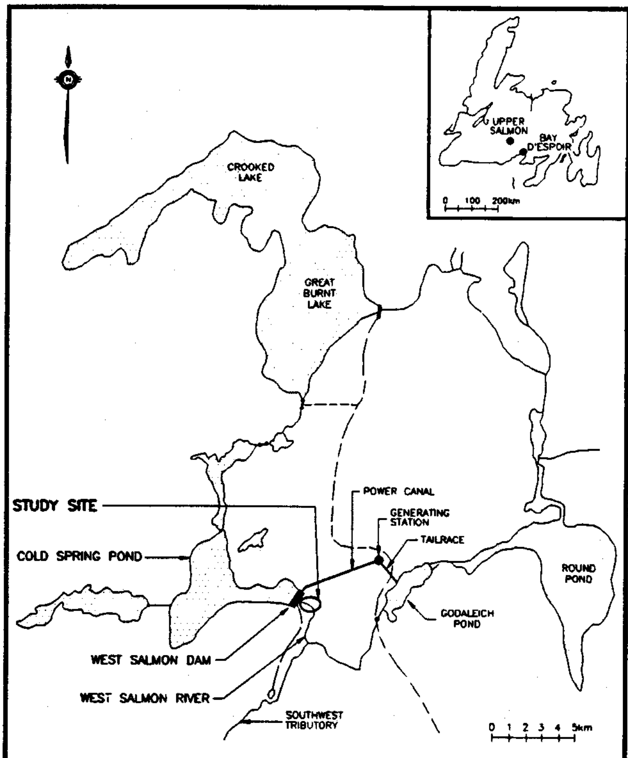


FIGURE 1: WEST SALMON RIVER STUDY AREA, UPPER SALMON HYDROELECTRIC DEVELOPMENT

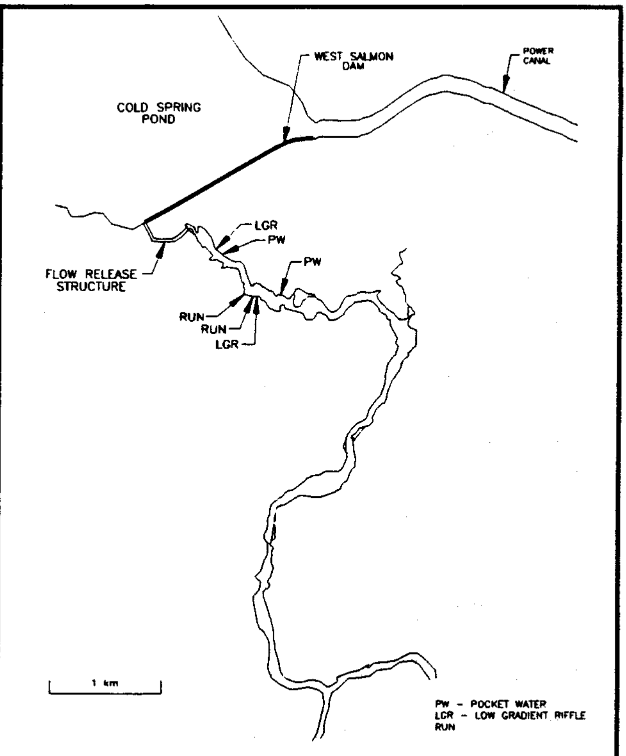


FIGURE 2: WEST SALMON RIVER STUDY SITE, JULY – AUGUST 1995

Illumination for nighttime observations was provided by a light was suspended from a continuous overhead rope which was fed through pulleys attached to posts at each end of the transect. A dual beam 9 volt fluorescent light was suspended from the rope and moved along the transect ahead of the snorkellers by the recorder on shore. The snorkellers also used handheld underwater flashlights to examine hard to view spots and shadows. Neon glow sticks were attached to the snorkels so that the recorder could see the location of the snorkellers and move the light appropriately.

Upon completion of snorkelling, or on the following day in the case of night surveys, the following data were recorded at each marker: marker number; total water depth (cm); bottom water velocity (2 cm above bottom) cm s^{-1}; nose (focal point) velocity cm s^{-1}; mean column velocity (at 60% depth) cm s^{-1}; and substrate classification directly under the marker (Local) and the predominant class within a 30 cm grid centered on the marker (General). The substrate classification was based on particle size measurements divided into 9 classes: Class 1, sand (0 - 6 mm); Class 2, pebble (6 mm - 16 mm); Class 3, small gravel (17 mm - 32 mm); Class 4, gravel (33 mm - 65 mm); Class 5, small cobble (66 mm - 128 mm); Class 6, cobble (129 mm - 255 mm); Class 7, large cobble (256 mm - 384 mm); Class 8, boulder (385 mm - 512 mm); and Class 9, large boulder (> 512 mm). Measuring sticks displaying the substrate scale were used to facilitate substrate classification. Water velocities were determined with a Marsh-McBirney Flo-Mate (Model 2000) electromagnetic flow meter.

Time Intervals

Snorkelling observations were conducted during four time intervals: interval #1, 1000 - 1600 hrs.; interval #2, 1600 - 2200 hrs.; interval #3, 2200 - 0400 hrs.; and interval #4, 0400 - 1000 hrs. An emphasis was placed on intervals #1 and #3 to minimize any potential influence of daytime/nighttime transition.

Validation

It was assumed that (i) the snorkelling did not change the habitat selection behaviour of the fish, (ii) the illumination used for nighttime surveys did not change the habitat selection behaviour of the fish, and (iii) habitat use observations were independent. To test these assumptions a validation experiment was conducted. A trap (Coral) capable of being triggered remotely and capturing all fish present within a subset of the river was designed and installed in the river for this purpose. These results are not reported here but did validate these assumptions.

Statistical Analysis

The Kolmogorov-Smirnov two sample test was used to compare the frequency distributions of age 0+ salmon. The frequency distributions were grouped into intervals prior to statistical analysis as per Siegle (1956). The null hypothesis was rejected at the 0.10 critical value, as per Warren (1986). Unless otherwise noted, daytime data refers to interval 1 in the following discussion.

Construction of Habitat Suitability Index Curves

The method of constructing habitat suitability index (HSI) curves based on use of different microhabitat variables using frequency analysis was conducted as described by Bovee and Cochnauer (1977). The number of individuals associated with each habitat parameter was tallied and assigned to each parameter (i.e. histogram). Optimal values of each variable were those associated with the most fish. Observed frequencies were divided by the mean of the optimum range to produce a normalized habitat use-curve. Optimal values of the habitat were assigned a value of 1.0 (maximum suitability) and values of the habitat not associated with any fish were assigned a value of 0 (least or unsuitable habitat).

Suitability curves were smoothed with stepwise polynomial regression. The aptness of each step or polynomial "order" was statistically evaluated by goodness of fit to the data (measured by the multiple correlation coefficient, R^2), the relative reduction in standard error given each additional order, and the random characteristics of the residual errors (with a serial correlation test). The stepwise procedure iteratively added an additional variable (or order) to the regression model until the addition of the next variable no longer caused a statistically significant reduction in residual error. The last significant model was inspected to see if residuals were randomly distributed and if the derived curve realistically described the observed frequency distribution. The curves generated by the polynomial models were normalized to 1.0 to produce habitat suitability values. Polynomial regression models typically do not fit outlying data points very well, therefore the derived curves which included such observations were subjectively smoothed to better represent suitability in those areas.

RESULTS

Fifty-five underwater transect snorkelling surveys were conducted during the period July 6 to August 24, 1995. With one exception (Pocket Water 1 which was surveyed 10 times) all the transects were surveyed an equal number of times (9). As per the study design, an emphasis was placed on intervals 1 and 3 with 23 and 19 surveys respectively. The duration of each survey was dependent on the number of fish observed, and was usually completed in 20 - 40 minutes. Observations were made and habitat use data collected on 1,077 fish, of which 970 were Atlantic salmon and 107 were brook trout *Salvelinus fontinalis* (Table 1).

Diurnal Variation in Observations

Table 2 provides a summary of the age classes and time intervals of the Atlantic salmon observations. Of the 970 salmon observed, 551 (56%) were observed at night (Interval 3). Of the remainder 202 (20.8%) were observed in Interval 1, 161 (16.6%) in Interval 2 and 56 (5.7%) in Interval 4. The distribution of age classes within time intervals 1, 2 and 4 were not significantly different from each other ($P > 0.05$), however the distribution in Interval 3 was significantly different from each of the other three time intervals ($P < 0.05$). This difference appears to be due to the variation in the proportion of age 0+ (35.6%) and 1+ (40.8%) salmon observed during the night compared to observations during the day (84.2% and 5.5%, respectively) (Figure 2). Sixty four percent of all the 0+ salmon observed (n = 549) were observed during the day, whereas, 91% of all the 1+ salmon (n = 248), and 88% of all 2+ (n = 79) were observed at nighttime.

Table 1. Snorkelling observations, July 6 - August 24, 1995, West Salmon River

Interval	Trout	Salmon	Total
1	9	202	211
2	5	161	166
3	91	551	642
4	2	56	58
Total	107	970	1077

Table 2. Summary of observations of Atlantic salmon by age class and time interval at West Salmon River July 6 - August 24, 1995.

Age Class	Day Time			Night Time	Total
	Interval 1	Interval 2	Interval 4	Interval 3	
0+	171	138	44	196	549
1+	10	6	7	225	248
2+	4	4	3	79	90
3+	17	13	2	51	83
Total	202	161	56	551	970

Habitat Suitability Indices

Mean column velocity - Most fry were observed at mean column velocities between 0 cm s^{-1} and 40 cm s^{-1} during both time periods. However, the frequency distribution of mean column velocity use during daytime was significantly different than nighttime use ($P < 0.05$). The HSI curve for mean column velocity exhibited an optimum range of 20 to 25 cm s^{-1} during the day and a lower optimum suitability of 15 to 20 cm s^{-1} during nighttime (Figure 3).

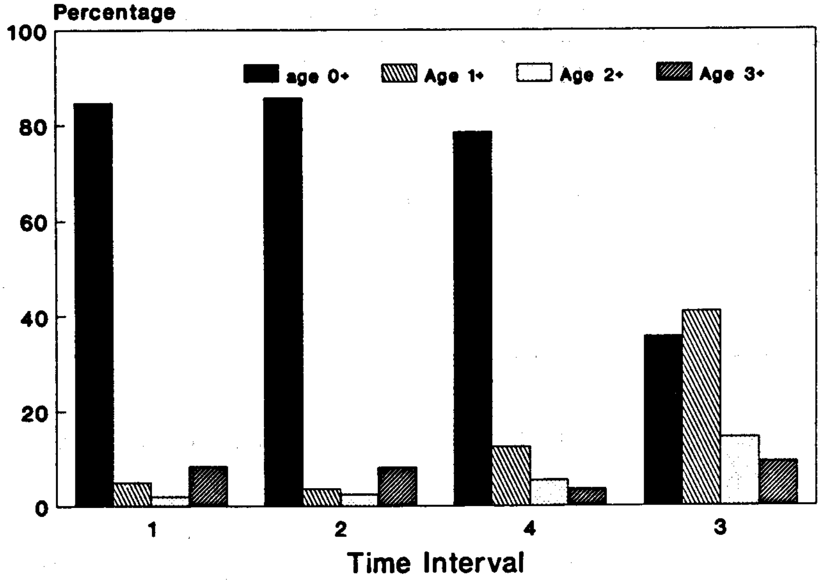


Figure 2. Age Distribution of observations by time interval

Nose velocity - The nose or focal point velocity of most fry was between 0 cm s^{-1} and 20 cm s^{-1} for both time periods and there was no significant difference between the nighttime and daytime frequency distributions for use ($P < 0.10$). The HSI curves for both periods showed an optimum at 0 to 5 cm s^{-1} (Figure 3).

Water Depth - Salmon fry were observed at focal positions in water depths of 9 cm to 85 cm during daytime and 4 cm to 73 cm at night, however most of the fish were seen at depths between 25 cm and 45 cm during both time periods. A significant difference was demonstrated between night and daytime depth use ($P < 0.10$). The HSI curve shows a strong single optimum peak at 40 cm for daytime while a wide optimum range, from 25 - 40 cm, was demonstrated at night (Figure 3).

Substrate (Local) - There was no significant difference in local substrate use between night and day ($P < 0.10$). The optimum suitability for both time periods was sand.

Substrate (General) - A significant difference was demonstrated between night and day general substrate use ($P < 0.05$), (Figure 3). Cobble was used most often in both time intervals, however, gravel and small cobble also received high usage in the day whereas small cobble and large cobble received high usage at night. The habitat suitability curves showed a similar trend.

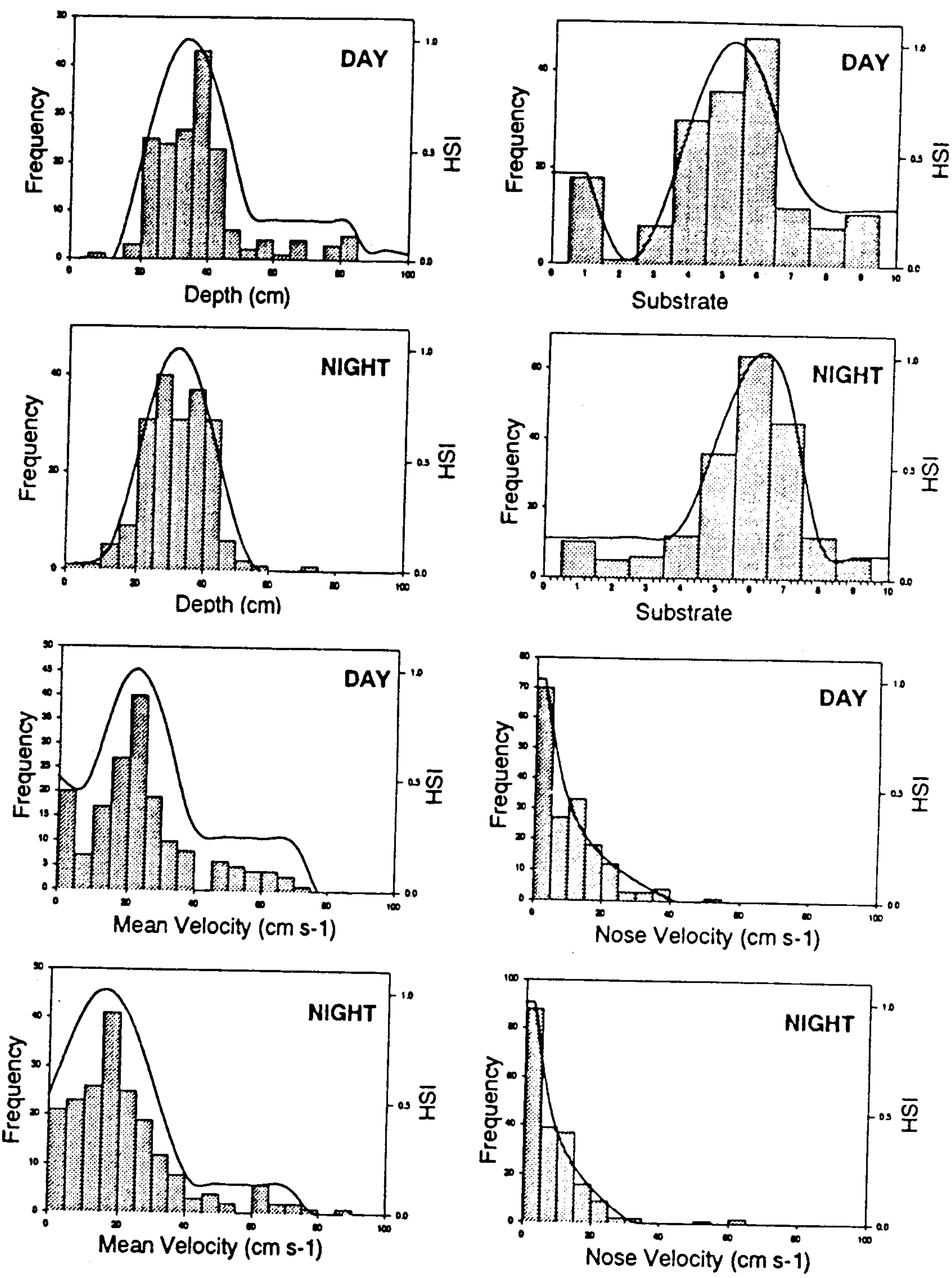


Figure 3. Habitat suitability curves and habitat use histograms for age 0+ Atlantic salmon.

DISCUSSION

A nocturnal behaviour pattern was exhibited by Atlantic salmon parr in West Salmon River. This was demonstrated by the large majority, 91% and 88% of all age 1+ and age 2+ parr, respectively, observed during the nighttime snorkelling surveys. A similar trend was not observed for 0+ fry, indicating a more diurnal strategy for that year class. Prior studies have demonstrated profound night/day changes in habitat selection behaviour by brown trout and Atlantic salmon (Heggenes 1994, Fraser et al 1993). These switches in behaviour have been associated with seasonal differences (summer/winter), seasonal transitions, (summer/autumn, Rimmer et al. 1983) and have been shown to be temperature dependent with 10°C being the critical threshold (Fraser et al. 1993). Although previously thought to be related to either an inherent annual rhythm or to photoperiod, Fraser et al (1995) demonstrated through laboratory experiments and field observations that neither is solely responsible, and determined that the temperature dependent shift to nocturnalism was explained by a suppression of daytime activity rather than an increase in activity at night. A reduced capacity to escape visual feeding predators (e.g. osprey *Pandion haliaetus*) due to low water temperatures had been speculated as influencing temporal changes in behaviour (Fraser et al. 1993). Salmon, being poikilotherms probably experience a reduction in burst speed which is necessary for escape from predators under cold water conditions, and therefore, during daylight hours are most at risk at low water temperatures.

Water temperatures recorded at West Salmon River during this study do not support the temperature dependent theory. The mean daily minimum water temperature throughout the study period was 16.9°C (range 14.5 -18.75 °C) with minimal diurnal variation (maximum of 2.7°C) as a result of regulation. The temporal shift in behaviour was therefore not the result of cold temperatures at night at West Salmon River, and other plausible explanations must be explored. The daily presence of ospreys feeding at the study site lends some credence to a predator avoidance theory. The West Salmon River, for the most part, is wide (30 m - 50 m) and shallow (<1.5 m) and provides excellent visibility for airborne predators, such as osprey and mergansers. At the end of the field season, (August 26th), the age 1+ and 2+ parr ranged in length from 66 mm to 143 mm, and would be more visible and thus make easier targets than the smaller 0+ fry (< 56 mm) during the daytime. Therefore, a behaviourial switch to nocturnalism by the larger, more vulnerable age classes is a logical response to airborne predators. Daytime behaviour would therefore entail seeking refuge in the substrate, which would account for the lack of daytime observations of parr by the snorkellers. Given that a similar behaviourial response would not be required by the smaller fry, the age class differential between day and night is understandable.

From a study of 6 rivers in Nova Scotia and 2 rivers in New Brunswick, Morantz et al.(1987) reported that Atlantic salmon fry selected water depths between 20 and 40 cm. Under similar discharge conditions as at West Salmon River (2.6 m^3/s), the work by deGraaf and Bain (1986) at North Arm River (2.3 m^3/s) found that Atlantic salmon fry preferred a depth of 15 to 25 cm. Fry suitability curves for station depth data for 18 Newfoundland rivers, demonstrated optimum suitability in the range of 15 - 20 cm with steady declines in suitability for depths greater than 20 cm (Scruton and Gibson 1993). The depth suitability curves for salmon fry at West Salmon River for both time periods showed an optimum range of 25 - 45 cm and is within the range reported in these studies.

Morantz et al. (1987) demonstrated optimal nose velocity for salmon fry from New Brunswick and Nova Scotia rivers to be in the range of 5 to 10 cm s^{-1}, and that they frequently selected zones of slow flow (0 - 5 cm s^{-1}), particularly at stream edges or behind large cobbles. deGraaf and Bain (1986) reported that nose velocity selection differed between habitat types for both fry and parr. Nose velocity selection was higher in riffle habitat, as opposed to slower moving habitat, and nose velocities selected were lower than the mean of available habitat in the faster flowing North Arm River. In the slower flowing North Harbour River (mean discharge 0.4 m^3/s) both fry and parr selected nose velocities were greater than the mean of available flows. At West Salmon River, the optimum range of selected nose velocity was between 0 and 5 cm s^{-1}, less than those previously reported for salmon fry in these other Atlantic Canada streams.

The classification of substrate use is probably one of the most important variables affecting fish habitat selection. However, the conventional technique of measuring substrate class directly beneath the focal point, "local substrate" in this study, is of questionable value in describing habitat selection. Variations in substrates is important since individuals in summer often hold positions close to a "home stone" (Rimmer et al. 1984). The home stone is used by the young salmon as a velocity shield from which it positions itself in an optimum velocity location for feeding and/or resting, often slightly downstream of the stone itself. The substrate directly below the focal point is often the sediment and fine particles aggregated immediately downstream of the home stone and is meaningless as an indicator of habitat selection. Because it is the dominant habitat variable which determines or influences all the other variables measured, the home stone must be included as a habitat selection parameter. The measurement of a 30 cm grid centered at the focal point incorporates the home stone into the description of substrate habitat and provides a measurement of arguably a more relevant determinant of habitat suitability. This measurement referred to as "general substrate" in this study showed a shift by salmon fry to larger home stones at night, or least the tendency to move further up into the water column utilizing large boulders as home stones. This is reflected in the habitat suitability curves.

Generally, age 0+ salmon exhibited a wider tolerance for all the variables measured during the day as compared to the night. The habitat suitability curves were similar for both time periods for nose velocity, depth, and local substrate. Subtle shifts in habitat suitability curves between day and night were apparent for mean column velocity and general substrate selection.

This study, conducted in a regulated Newfoundland stream, has demonstrated that diurnal shifts in habitat selection and behaviour by juvenile Atlantic salmon are not restricted to the winter period, as previously thought, but occur throughout the summer season and are not temperature dependent. The results of this study indicate that nighttime as well as daytime surveys should be considered for the construction of habitat suitability indices in order to accommodate diurnal changes in habitat use.

REFERENCES

Beak, 1980, Fisheries Investigations for the Upper Salmon Hydroelectric Development. Unpublished report prepared for Newfoundland and Labrador Hydro, St. John's, Newfoundland.

Bovee, K.D. and T. Cochnauer. 1977. Development and evaluation of weighted criteria, probability-of-use curves for instream flow assessments: fisheries. Instream Flow Information Paper 3. U.S.D.I. Fish. Wildl. Serv., Office of Biol. Serv. FWS/OBS-77/63: 39 p.

Campbell, R.F., and J. H. Neuner, 1985. Seasonal and Diurnal Shifts in Habitat Utilization by Resident Rainbow

Trout in Western Washington Cascade Mountains Streams. In F.W. Olson, R.G. White, R.H. Hamre, Proceedings of the Symposium on Small Hydropower and Fisheries. American Fisheries Society, Bethesda, Maryland USA; 39-48.

Chaston, I., 1968. Influence of Light on Activity of Brown Trout (*Salmo trutta*). J. Fish. Res. Bd. Can. 25; 1285-1289.

deGraaf, D.A. and L.H. Bain. 1986. Habitat Use by and Preferences of Juvenile Atlantic Salmon in Two Newfoundland Rivers. *Trans. Am. Fish. Soc.* 115: 671-681.

Fraser, Neil H.C., N.B. Metcalfe and J. E. Thorpe. 1993. Temperature-dependent Switch Between Diurnal and Nocturnal Foraging in Salmon. Proc. R. Soc. Lond. B. p.135-139.

Fraser, Neil H.C., J. Heggenes, N.B. Metcalfe and J. E. Thorpe. 1995. Low Summer Temperature Cause Juvenile Atlantic Salmon to become Nocturnal. *Can. J. Zool.* 73; 446- 451.

Heggenes, J., 1988. Substrate Preferences of Brown Trout fry (*Salmo trutta*) in Artificial Stream Channels. *Can. J. Fish. Aquat. Sci.* 45; 1801-1806.

Heggenes, J. 1994. Physical Habitat Selection by Brown Trout (*Salmo trutta*) and Young Atlantic Salmon (*Salmo salar*) in Spatially and Temporally Heterogeneous Streams: Implications for Hydraulic Modelling. p. 12 - 30. In Proceedings of the 1st International Symposium on Habitat Hydraulics. The Norwegian Institute of Technology Trondheim, Norway. August 18 - 20, 1994. 637 p.

Jacques Whitford Environment Ltd., 1996. Evaluation of Instream Flow Needs Assessment Methodologies in Newfoundland; 2. West Salmon River. Report Prepared Under the Canada-Newfoundland Agreement Respecting Water Resource Management and Green Plan, Habitat Action Plan. The Department of Environment, Government of Newfoundland and Labrador, Environment Canada, Fisheries and Oceans, St. John's, Newfoundland. 38 p. + app.

LaCroix, G. L. , D. Hood, J. Smith. 1995. Stability of Microhabitat Use by Brook Trout and Juvenile Atlantic Salmon after Stream Acidification. *Trans. Am. Fish. Soc.* 124: 588-598.

McCain, M. D. Fuller, L. Decker, and K. Overton. 1990. Stream Habitat Classification and Inventory Procedures for Northern California. FHR Currents Technical Bulletin No. 1. U.S. Department of Agriculture Forest Service Pacific Northwest Region. 15 p.

Morantz D.L., R.K. Sweeney, C.S. Shirvell, and D.A. Longard, 1987. Selection of Microhabitat in Summer by Juvenile Atlantic Salmon (*Salmo salar*). *Can. J. Fish. Aquat. Sci.* 44: 120-129.

Powers, M.E., 1987. Predator Avoidance by Grazing Fishes in Temperate and Tropical Streams; Importance of Stream Depth and Prey Size. *In* W.C. Kerfoot and A. Sih (eds.) Predation Direct and Indirect Impacts on Aquatic Communities. University Press of New England, Hanover, New hampshire, USA; 333-351.

Rimmer, D.M., U. Paim, and R.L. Saunders. 1984. Changes in the Selection of Microhabitat by Juvenile Atlantic Salmon (*Salmo salar*) at the Summer-Autumn Transition in a Small River. *Can J. Fish. Aquat. Sci.* 41: 469-475.

Scruton D.A. and R.J. Gibson. 1993. The Development of Habitat Suitability Curves for Juvenile Atlantic Salmon (*Salmo salar*) in Riverine Habitat in Insular Newfoundland, Canada, p 149-161. *In* R.J. Gibson and R.E. Cutting (ed.) Production of Juvenile Atlantic Salmon, *Salmo salar*, in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118.

Siegel, S. 1956. Nonparametric Statistics for the Behavioral Sciences. McGraw-Hill Book Company, 1956.

Tennant, D.L., 1975. Instream flow regimens for fish, Wildlife, Recreation and Related Environmental Resources. U.S. Fish Wildlife Service, Billings Montana. 30 p.

Warren, W. G. 1986. On the Presentation of Statistical Analysis: Reason or Ritual. Can J. For. Res. 16; 1185-1191.

SPATIAL RELATIONS BETWEEN LOTIC FISHES, BENTHIC MACROINVERTEBRATES AND THE STREAM HABITAT: TOWARD A GIS-ASSISTED APPROACH

Timo Muotka
Dept. of Biological and Environmental Sciences, University of Jyväskylä, PO Box 35, 40351 Jyväskylä, Finland;
e-mail: tmuotka@jylk.jyu.fi

Aki Mäki-Petäys
Finnish Game and Fisheries Research Institute, Kainuu Research Unit, 88300 Paltamo, Finland

Petri Kreivi
Stream Ecology Group, Dept. of Biology, University of Oulu, PO Box 333, 90571 Oulu, Finland

ABSTRACT

There are usually no natural or 'characteristic' scales for ecological patterns and processes. Accordingly, results are highly scale-dependent, and extrapolation across scales is inadvisable. These problems can be circumvented by adopting a multi-scale approach. In this study, we quantified the spatial distributions of stream fishes, their food resources and habitat characteristics in a riffle of a boreal stream. The study site (23 x 4.5 m) was divided into 279 grid cells, and the abundances of juvenile brown trout and arctic bullhead were quantified in each cell by electrofishing. In addition, four habitat variables (depth, water velocity, substrate size, in-stream vegetation) were measured from the centre of each quadrat, and benthic samples were collected from every other quadrat. Shaded contour plots, showing the spatial patterns of the habitat variables within the 'sampling window' were produced. Fish distributions were superimposed on these contours to produce overlay maps depicting the spatial configuration of the sample space. Bases on the visual inspection of these maps, both fish species were clearly non-randomly distributed within the sample area. More importantly, the two species showed a tendency for spatial avoidance: high-density patches of trout and sculpin were in separate sections of the riffle. The spatial interaction between juvenile trout and bullhead thus appears highly scale-dependent: inhibition at small to intermediate scales (individual fish to intraspecific aggregations), and coherence at larger scales (stream sections). Bullhead was more closely associated with the spatial variation of food resources: larger bullhead (>4 cm) preferred stream areas with highest densities of semi-sessile macroinvertebrates. For trout, the clearest patterns were found in relation to water depth and substrate size. We are currently using this data set to generate statistical models for situations with more than two layers of spatially-referenced data. This undertaking will consist of the interactive use of the Geographical Information System (GIS) for data input, storage and graphical presentation, and statistical modelling based on simplifications via conditional independencies. We believe such a technique should prove valuable in enhancing our understanding of the spatial heterogeneity and scaling in lotic consumer-resource interactions.

KEY-WORDS: Scale / Spatial heterogeneity / Streams / Brown trout / Arctic bullhead / Benthic invertebrates / Predator-prey interactions / Geographical Information Systems

INTRODUCTION

Habitat models used for predicting the amount of stream area suitable for game fishes are based on the measurement of a few key habitat features. It is obvious, however, that habitat selection by fish is also affected by biotic factors like the presence of competitors and predators and, perhaps most importantly, the availability of food resources. Even though many authors have pointed out the potential bias caused from not including food availability into the models (e.g. Orth, 1987; Gore & Nestler, 1988), the development of more realistic versions has been hindered by lack of information on the role of biotic factors in stream fish habitat selection. Even correlative field studies reporting the effects of prey availability on fish habitat use are conspiciously lacking.

All natural environments are spatially and temporally heterogeneous systems. It is thus not surprising that habitat selection, like indeed most ecological phenomena, is scale-variant: what is seen at one scale of observation may not recur at other scales. It is usually difficult, however, to determine a priori the appropriate scale for a study, and the best option may then be to conduct the study, regardless of its objectives, on a variety of spatial scales (Levin, 1992; Ives et al., 1993). Streams are notoriously heterogeneous environments, where organisms exhibit patchy distributions on a spatially and temporally variable physical arena. It thus seems obvious that fish - environment relationships, as well as associations between fish species or age-classes, should be examined across multiple scales. Yet, to our knowledge, there are no previous attempts to adopt such a multi-scale approach in a study of habitat selection by stream fishes. Here, we report the sampling strategy and preliminary results from a field study using a 'landscape approach' to examine habitat use by stream fishes in relation to physical habitat variables and prey resources.

MATERIAL AND METHODS

The field study was conducted in the River Kuusinkijoki, northeastern Finland. The mean discharge of the river is 10 m^3/s, and it is characterized by wide fluctuations in flow. The peak flow (up to 65 m^3/s) occurs in late May. The average population density of juvenile brown trout (*Salmo trutta* L.) at our study site is 0.75 ind./m^2, which is considerably higher than in other parts of the river system (Huusko and Korhonen, 1993). The stream bed at the study site is dominated by cobbles and small boulders, interspersed with coarse gravel. The stream bottom is densely covered with aquatic vegetation, mainly *Ranunculus* sp. and aquatic mosses. Brown trout and Arctic bullhead (*Cottus poecilopus* Heckel) are the dominant fish species, although other species, especially grayling (*Thymallus thymallus* L.) and European minnow (*Phoxinus phoxinus* L.) may also occur in relatively high numbers in the deepest stream pools.

Our sampling site is located in the Raatekoski rapids (66° 17' N, 29° 54'E) on the lower course of the river. We delineated a rectangular sampling area of 23 m x 4.5 m, consisting of a grid of 279 cells, each 0.75 m x 0.5 m (Fig. 1). This cell size was chosen to reduce the potential disturbance from sampling in a cell to invertebrates/fishes in adjacent cells. We electrofished the entire study area on 13 August 1994, recording the presence and number of trout and bullhead in each cell (point electrofishing, see Moyle and Baltz, 1985; Heggenes, 1989). Each fish was assigned to either of two size-classes (trout: 5-10 cm, 10-15 cm; bullhead: <5 cm, 5-8 cm), roughly corresponding to age classes 0+ and 1+ (Huusko and Korhonen, 1993). To reduce the effects of positive galvanotaxis, we used a DC backpack electroshocker with low voltage (300 V) and a 9-cm-diameter anode. It has been recently shown by Copp (1989) that the approximate area of the effective field of

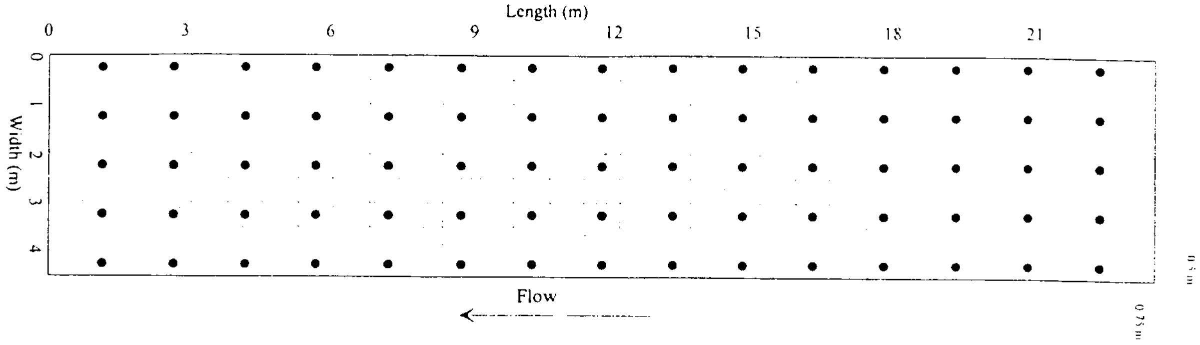


Figure 1: A schematic presentation of the study section. Black dots denote positions of the colonization substrates used for benthic sampling.

a 10-cm anode is 0.071 m^2. The "fright bias", which may cause displacement of individuals from their original positions, was minimized by the point electrofishing procedure used for locating fish (e.g. Heggenes, 1989).

After fish sampling was completed, four microhabitat variables (depth of water, water velocity, substrate size, percent cover of instream vegetation) were measured in each cell. Water depth was measured to the nearest centimeter with a wading rod. Water velocity was measured at 0.6-depth with a Schiltknecht Mini Water type 624 w-m/l flowmeter fitted with a 20 mm propeller. Water depth and velocity were determined at the centre of each grid cell. Dominant substrate size and percent cover of submerged vegetation were estimated visually for the entire cell. Substrate was classified according to a modified Wentworth scale, using categories 4-10 (sand to boulder) of the 13 particle size categories in Heggenes' (1989) classification.

We used dark-coloured paving bricks (14 x 13 cm) as colonization substrates for benthic invertebrates, since it has been shown that, given sufficient colonization time for periphytic algae, such bricks mimic natural stream stones reasonably well (Douglas & Lake 1994). One brick was placed in the centre of every other cell (see Fig. 1) three weeks before fish sampling took place. To reduce the immediate effects of disturbance ineluctably caused by electrofishing, benthos was sampled 10 days after fish sampling. Benthic sampling was performed by placing a frame with a net (mesh size 0.25 mm) behind a brick, which was then quickly moved into the net. Invertebrates dislodged from the stone and trapped by the net were included in the sample. Invertebrates were preserved in 70% ethanol in the field and later identified to a feasible (mainly generic) taxonomical level in the laboratory.

This data set is currently under a detailed analysis whereby methods are being developed for an effective presentation and statistical analysis of spatially dependent, multi-layered data. Here we present preliminary results in the form of overlay maps showing the spatial patterns in the distribution of different variables within the "sampling window". For this purpose, we used a SYSTAT smoothing option (Systat Inc. 1992), the DWLS (Distance Weighted Least Squares) smoothing. This method produces shaded contour plots with fill patterns from white (empty) to black (fill) in even gradations, determined by the height of the function at a given grid cell. The DWLS smoothing fits a contour through a set of points by least squares. As a result, this method produces a locally weighted contour plot running through all the points in the sample space (McLain, 1974).

RESULTS AND DISCUSSION

As representative examples of spatial variation in physical habitat variables, we present here the contours for water depth and substrate size in relation to brown trout positions within the sample space (Figs. 2A and B, respectively). It appears that larger trout preferentially use the deepest available stream areas with cobble to boulder substrates. This result is consistent with the previous findings of Mäki-Petäys et al. (1996) from the same stream system. The "bigger fish-deeper habitat" pattern has been repeatedly reported in numerous studies of stream fish habitat selection (e.g. Bohlin, 1977; Greenberg et al., 1994; Power, 1987; Harvey and Stewart, 1991). Nonetheless, even though there was not much variation in water depth within our study site, the overlay map (Fig. 2A) shows remarkably well the tendency for the largest trout to occupy the deepest available stream areas.

The spatial relations between fish and their prey resources can not be unambiguosly detected on the basis of a visual inspection of Figs. 3A and B. Microhabitat patches totally devoid of macroinvertebrates were clearly avoided by fish, but only the largest bullhead (>5 cm) seemed to show any aggregation to areas with highest densities of benthos, especially semi-sessile invertebrates (larval stages of blackflies and filter-feeding caddisflies; see Fig. 3B). Sculpins are pursuing, non-visual predators that rely on tactical and/or hydrodynamic cues for prey detection (e.g. Hoekstra and Janssen, 1985). Furthermore, there is some evidence showing that sculpin is a size-selective predator, favouring the largest prey items available (Newman and Waters, 1984; Allan and Malmqvist ,1989). Net-spinning caddis larvae of the families Polycentropodidae and Hydropsychidae are certainly among the largest of the food items available for sculpin in our study reach. These semi-sessile invertebrates are apparently relatively easy to capture, but especially the trichopteran larvae may be too big for the smaller bullheads to handle and ingest without prohibitive costs. Although these mechanistic explanations to the spatial patterns among bullhead and their prey seem plausible, they are at present mere speculations, because to our knowledge there are no ample laboratory studies documenting the mechanisms of prey selection in sculpins.

The apparent lack of aggregation by brown trout with its prey may partly result from the field methods used to estimate prey availability. Trout is a sit-and-wait predator, which hunts visually for drifting prey. In contrast to bullhead, which is strictly a benthic-feeding fish (e.g. Straskraba et al. 1966), trout takes its prey mainly from the water column (Elliott 1994). Although there are some indications that epibenthic feeding may be more prevalent in trout than commonly thought (Tippets & Moyle 1978), a valid description of food availability for trout may necessitate the quantification of macroinvertebrate drift. On later sampling occasions we have collected drift using somewhat sparser sampling grid, and even gut contents of fish caught from known stream positions, and we anticipate these new data sets to contribute considerably to our understanding of the spatial relations between stream fishes and their prey in systems similar to our sampling site.

Both fish species were clearly non-randomly distributed within the sample area. At the scale approximating the size of the mapped area, both sculpin and trout seemed to form three or four relatively distinct clusters, with regular spacing among individuals within these high density aggregations. It is, however, even more interesting that the two species seemed to show spatial avoidance: high-density aggregations of trout and sculpin were clearly in separate sections of the sample space (Fig. 3A). The two species were located in a same grid cell on only seven occasions which, considering the high general density of fishes in the study section, is a remarkably low frequency of co-occurrences. If there really is spatial competition between the two species, as previously

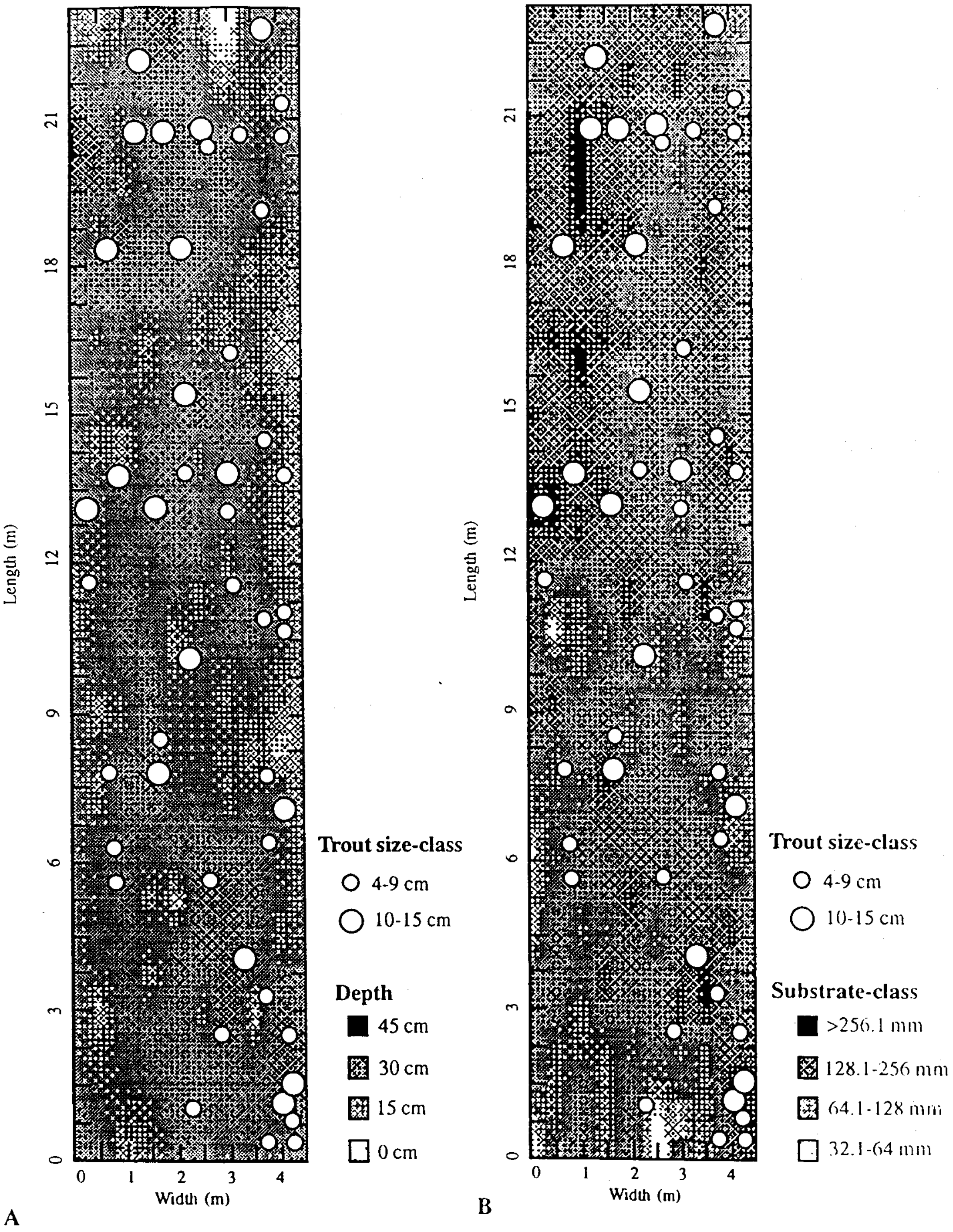


Figure 2. Overlay maps of the study section showing the distributions of brown trout size-classes in relation to contour plots of depth (A) and substrate size (B).

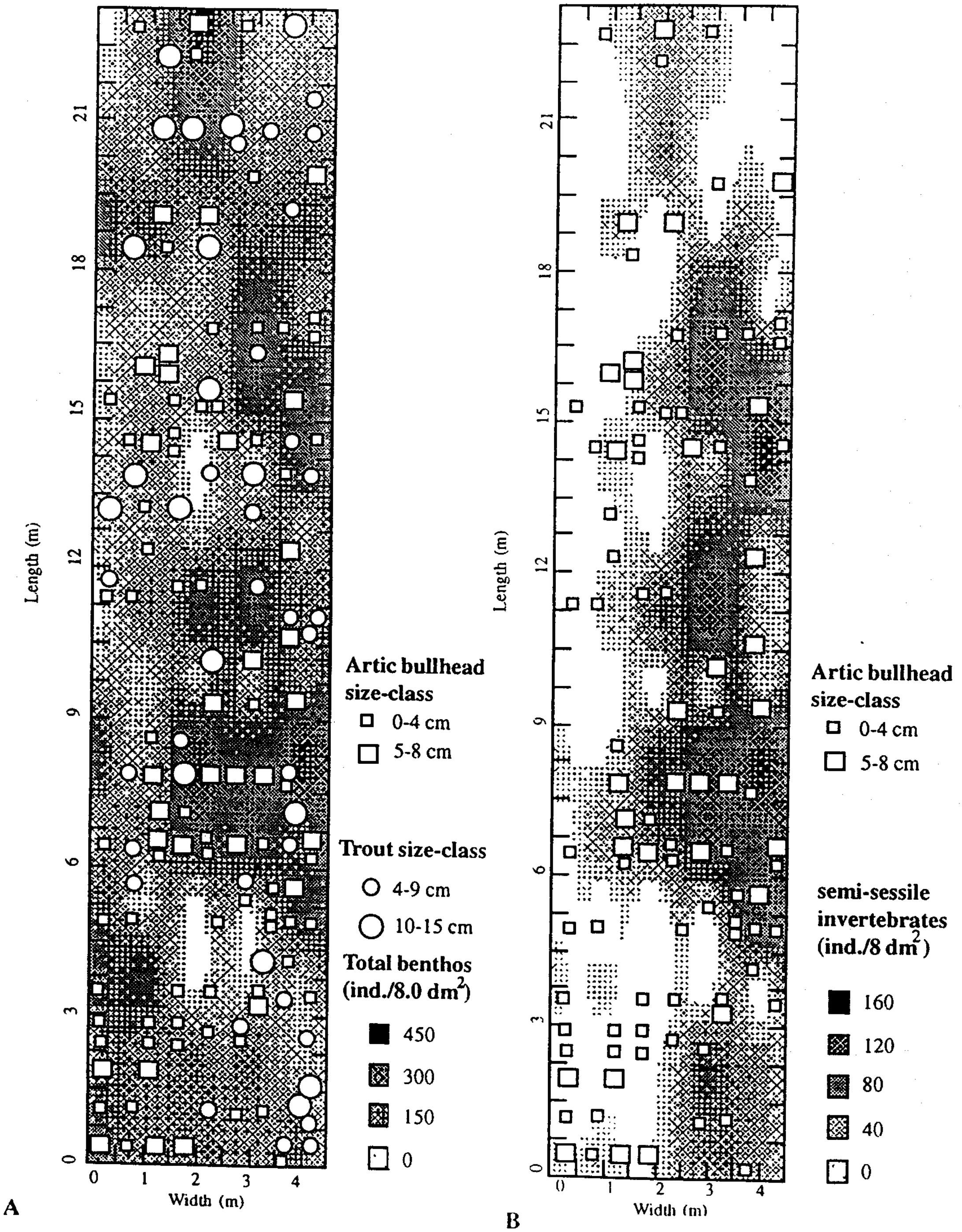


Figure 3. Overlay maps of the study section showing the distributions of both fish species in relation to contour plots of macroinvertebrate densities (A); bullhead in relation to densities of semi-sessile invertebrates (B).

density patches of semi-sessile invertebrates (Figs 3A and B). The spatial interaction between these two core species of fish assemblages in boreal streams appears to be highly scale dependent: inhibition at small to intermediate scales (individual fish to intraspecific aggregations), and coherence at larger scales (stream sections).

The antipredatory behaviours and relative mobilities of prey clearly must be considered in any study of predator-prey spatial correlations. If spatial dependences are to be found, they should be between fish and semi-sessile prey (see Sih, 1984). More mobile prey, like lotic mayflies, may easily evade attacks by fish (Scrimgeour and Culp, 1994; Tikkanen et al., 1994). It has even been suggested that these mayflies are able to track the relative predation risks at separate stream patches, and shift their distributions accordingly (Tikkanen et al., 1994; Forrester 1995). Thus, from a fish point of view, mayflies and other mobile invertebrates may form highly ephemeral prey patches which disperse as soon as the predator enters a patch (for a similar explanation for the lack of aggregation by stonefly predators with their prey, see Peckarsky and Penton, 1985). Mobile prey may thus use stream habitat based on their current estimate of the relative benefits (in terms of food acquisition) and costs (predation risk) in local patches; this in turn would add importantly to the spatial and temporal heterogeneity of stream habitat resources as experienced by lotic fishes.

Previous studies on the spatial aspects of predator-prey (or more widely, consumer-resource) interactions in streams have commonly employed areal survey designs using sampling plots of a fixed size. If the plots are sparsely distributed, the observations (counts of individuals, measurements of environmental variables) are approximately independent, and standard statistical methods can be used. An often ignored drawback of this approach is that there is usually no a priori information concerning the ecologically important scales of the interaction. Instead, the size of the quadrat defines the scale of investigation, and true associations between variables (e.g. interspecific relations) may be completely lost, or even worse, artificial associations that do not exist in reality, may emerge. In geostatistics, this is known as the problem of 'change of support' (e.g. Openshaw and Taylor, 1979; Arbia, 1989): a natural (or 'characteristic') scale for studying a process is lacking, or is at least different for different processes . A multiscale design based on mapped data avoids this bias by considering many different scales simultaneously. An adequate tool for the input, storage, graphical output and preliminary analysis of such data is the Geographical Information System (GIS; for a recent review of the applications of GIS in ecology, see Johnson, 1990). GIS-techniques have been intensively used by resource managers e.g. to assess the effects of broad-scale land use patterns on aquatic biota, but to our knowledge it has not been previously applied to analysis of within-stream heterogeneity at a scale of separate stream sections (but see Cooper et al., 1996). In GIS, spatially referenced data for each response variable is stored in separate layers, which can then be overlaid to examine associations, and the scales of association, between the variables. If sampling is conducted in the field, uncontrolled variability will always be present. This can to some extent be allowed for by collecting spatially-referenced data on potentially important covariates (e.g. physical habitat variables in a study of consumer-resource interactions), which are then also stored as separate layers in the GIS. In lotic ecology, it is sometimes possible to map a variable, e.g. the distribution of individuals within a sampling window, accurately as a point pattern (e.g. Muotka & Penttinen 1994), but in most cases some level of data aggregation is needed. For example, when electrofishing is used for fish sampling, the best one can obtain is a grid map of fish distributions. Furthermore, randomly or systematically positioned sampling units must be used to obtain data on habitat variables. Kriging or other- geostatistical interpolation methods can then be applied to provide estimates for unredorded locations (e.g. Rossi et al., 1992). The layers describing different response variables may thus be of different forms, but are still subject to visualization in GIS.

The statistical analysis of associations between different layers of a GIS presents a formidable challenge. The existing geostatistical methods, e.g. cross-variograms and cross-correlograms, can deal with the joint spatial distributions of two variables. We are currently utilizing the data presented here for statistical modelling in situations with more than two multilayered variables in a GIS context. This is performed using a series of simplifications via conditional independencies. As an example, let F (fish), P (prey) and H (habitat) represent the variables of interest, whose variation within a sample space is quantified. Their joint distribution $[F,P|H)]$, conditioned by the exogeneous habitat variables, is partitioned into a set of conditional distributions, e.g. $[F|P,H]$ and $[P|F,H]$, in simultaneous modelling. If one assumes that the prey does not display antipredatory behaviours (either because it does not possess adequate behavioural strategies toward a particular predator, or risk for the prey is negligible), a simplification $[F,P|H] = [P|H][F|P,H]$ is obtained. The model can be elaborated even further: for example, F (fish) can be partitioned into two components, F_T and F_B, corresponding to different fish species (trout and bullhead, respectively). In any case, the conditional models can be integrated using an iterative algorithm based on the Markov chain Monte Carlo-method.

In summary, the GIS-assisted approach described above will consist of the interactive use of the Geographical Information System and statistical modelling to produce effective analytical tools for spatially dependent, multilayered and multiscaled data. We believe such a technique may prove valuable in advancing our understanding of the spatial heterogeneity of lotic consumer-resource systems. Since heterogeneity and scale are tightly intertwined in all natural environments (e.g. Levin, 1992; Dutilleul and Legendre, 1993), our approach should also help to detect the relevant scales at which different processes operate. There is no doubt that manipulative experiments will be needed to separate the effects of individual factors on the patterns observed (e.g. the effects of prey availability vs. physical habitat characteristics on fish habitat selection) and, to this end, description of a pattern represents a mere starting point for more process-oriented studies. Furthermore, as Cooper et al. (1996) have pointed out, more studies are needed on the patch-scale dynamics (input-output budgets) of stream consumers, as well as on the movement patterns of fish and their prey in the heterogeneous 'microlandscapes' of stream habitats. Investigation of animal movement patterns across a range of spatial scales may provide a mechanistic link between ecological processes and the patterns of spatial heterogeneity of stream environments. This approach has proved useful in terrestial scaling studies (Wiens and Milne, 1989; Crist and Wiens, 1994; With, 1994), and although the tracking of individual stream consumers is bound to be a difficult and labour-intensive venture, we believe that such observations should be conducted whenever possible.

ACKNOWLEDGEMENTS

We are indebted to A. Penttinen for his irreplaceable contributions to our endeavours in the scaling of patterns and processes in lotic ecosystems. A. Huusko is acknowledged for all sorts of logistical support, and for sharing our belief that ecological scaling is of crucial importance not only for theoreticians but also for the purposes of stream management. Department of Animal Ecology, University of Umeå, provided a pleasant atmosphere to TM during the writing of this paper. Kuusamo Aquaculture and Oulanka Biological Station provided facilities and lodging during the field work. Finally, we express our gratitude to M. Hirvesmieli for her continual mental support. This is a contribution from the Finnish National Graduate School on Fish Biology and Fisheries.

REFERENCES

Allan, J.D. and B. Malmqvist (1989). Diel activity of Gammarus pulex (Crustacea) in a South Swedish stream: comparison of drift catches vs baited traps. *Hydrobiologia* 179, pp. 73-80.

Arbia, G. (1989). Spatial data configuration in statistical analysis of regional economic and related problems. Klower Academic Pbl. London.

Bohlin, T. (1977). Habitat selection and intercohort competition of juvenile sea-trout Salmo trutta. *Oikos* 29, pp. 112-117.

Cooper, S.D., L. Barmuta, O. Sarnelle, K. Kratz and S. Diehl (1996). The description of patchiness and its application to the study of biological interactions in streams. *Journal of the North American Benthological Society* (in press).

Copp, G.H. (1989). Electrofishing for fish larvae and 0+juveniles: equipment modifications for increased efficiency with short fishes. *Aquaculture and Fisheries Management* 20,pp.177-186.

Crist, T.O. and J.A. Wiens (1994). Scale effects of vegetation on forager movement and seed harvesting by ants. *Oikos* 69,pp. 37-46.

Douglas, M. and P.S. Lake (1994). Species richness of stream stones: an investigation of the mechanisms generating the species-area relationship. *Oikos* 69,pp. 387-396.

Dutilleul,P. and P. Legendre (1993). Spatial heterogeneity against heteroscedasticity: an ecological paradigm versus a statistical concept. *Oikos* 66,pp. 152-171.

Elliott, J.M. (1994). Quantitative ecology and the brown trout. Oxford University Press, Oxford.

Forrester, G.E. (1994). Diel patterns of drift by five species of mayfly at different level of fish predation. *Canadian Journal of Fisheries and Aquatic Sciences* 51,pp. 2549-2557.

Gaudin, P. and L. Caillere (1990). Microdistribution of Cottus gobio L. and fry of Salmo trutta L. in a first order stream. *Polskie Archiwum fur Hydrobiologie* 37,pp.81-94.

Gore, J.A. and J.M. Nestler (1988). Instream flow studies in perspective. *Regulated Rivers* 2,pp. 93-101.

Greenberg,L. P. Svendsen and A. Harby (1994). Availability of microhabitats and their use by brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in the River Vojmån, Sweden. *Proceedings of the 1st International Symposium on Habitat Hydraulics,* pp. 606-624.

Harvey, B.C. and A.J. Stewart (1991). Fish size and habitat depth relationships in headwater streams. *Oecologia* 87,pp. 336-342.

Heggenes, J. (1989). Physical habitat selection by brown trout (Salmo trutta) in riverine systems. *Nordic Journal of Freshwater Research* 64,pp. 74-90.

Hoekstra, D. and J. Janssen (1985). Non-visual feeding behavior of the mottled sculpin, Cottus bairdi, in Lake Michigan. *Environmental Biology of Fishes* 12,pp. 111-117..

Huusko, A. and P. Korhonen (1993). Population densities of young stages of brown trout (*Salmo trutta* L.) in the Oulankajoki river system. *Oulanka Reports* 12,pp. 125-127.

Ives, A.R., P. Kareiva and R. Perry (1993). Response of a predator to variation in prey density at three hierarchical scales: lady beetles feeding on aphids. *Ecology* 74,pp. 1929-1938.

Johnson, L.B. (1990). Analyzing spatial and temporal phenomena using geographical information systems. *Landscape Ecology* 4,pp. 31-43.

Levin, S.A. (1992) The problem of pattern and scale in ecology. *Ecology* 73,pp. 1943-1967.

McLain, D.H. (1974). Drawing contours from arbitrary data points. *The Computer Journal* 17,pp. 318-324.

Moyle, P.B. and D.M. Baltz (1985). Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow determinations. *Transactions of the North American Fisheries Society* 114,pp. 695-704.

Muotka, T. and A. Pemttinen (1994). Detecting small-scale spatial patterns in lotic predator-prey relationships: statistical methods and a case study. *Canadian Journal of Fisheries and Aquatic Sciences* 51,pp. 2210- 2218.

Mäki-Petäys, A., T. Muotka, A. Huusko, P. Tikkanen and P. Kreivi (1996). Seasonal changes in microhabitat use and preference by juvenile brown trout in a northern boreal river. *Manuscript* (in review).

Newman, R.M. and T.F. Waters (1984). Size selective predation on Gammarus pseudolimnaeus by trout and sculpins. *Ecology* 65,pp. 1535-1545.

Openshaw, S. and P.J. Taylor (1979). A million or so of correlated coefficients: three experiments on the modifiable unit problem. In Wringley, N. and R.J. Bennett (eds). *Statistical applications in the spatial sciences.* Pion, London.

Orth, D.J. (1987). Ecological considerations in the development and application of instream flow-habitat models. *Regulated Rivers* 1,pp. 171-181.

Peckarsky, B.L. and M.A. Penton (1985). Is predaceous stonefly behavior affected by competition? *Ecology* 66,pp. 1717-1728.

Power, M. (1987). Predator avoidance by grazing fishes in temperate and tropical streams: importance of stream depth and prey size. In Kerfoot, W.C. and A. Sih (eds). *Predation: direct and indirect impacts on aquatic communities.* University press of New England, Hanover, pp. 333-351.

Rossi, R.E., D.J. Mulla, A.G. Journel and A.H. Franz (1992). Geostatistical tools for modeling and interpreting ecological spatial dependence. *Ecological Monographs* 62, pp. 277-314.

Scrimgeour, G.J. and J.M. Culp (1994) Foraging and evading predators: the effect of predator species on a behavioural trade-off by a lotic mayfly. *Oikos* 69, 71-79.

Sih, A. (1984). The behavioral response race between predaor and prey. *American Naturalist* 123, pp. 143-150.

Straskraba, M., J. Chiar, S. Frank and V. Hruska (1966). Contribution to the problem of food competition among the sculpin, minnow and brown trout. *Journal of Animal Ecology* 35, pp. 303-311.

SYSTAT Inc. (1992). *SYSTAT for windows: Graphics*, Version 5. Evanston, Il.

Tippets, W.E. and P.B. Moyle (1978). Epibenthic feeding by rainbow trout (*Salmo gairdneri*) in the McCloud River, California. *Journal of Animal Ecology* 47, 549-559.

Tikkanen, P., T. Muotka and A. Huhta (1994). Predator detection and avoidance by lotic mayfly nymphs of different size. *Oecologia* 99, pp. 252-259.

Wiens, J.A. and B.T. Milne (1989). Scaling of 'landscapes' in landscape ecology, or, landscape ecology from a beetle's perspective. *Landscape Ecology* 3, pp. 87-96.

With, K.A. (1994). Using fractal analysis to assess how species perceive landscape structure. *Landscape Ecology* 9, pp. 25-36.

Weighted usable volume for young of the year non-game fish in the Marchfeldkanal, Austria

Stefan SCHMUTZ*, Helmut MADER# & Günther UNFER*
* Dept. of Hydrobiology, Fisheries and Aquaculture
Univ. of Agriculture and Natural Resources, Vienna
Dept. of Water Management, Hydrology and Hydraulic Engineering
Univ. of Agriculture and Natural Resources, Vienna
Austria
h524p2@edv1.boku.ac.at

ABSTRACT

From an ecological point of view the Marchfeldkanal (MFK), a nature-like irrigation channel, represents a side branch of the Danube River. In the summer of 1992 the first flooding of the constructed channel took place. The process of fish colonization and community succession is being investigated for a 5 year period within a large interdisciplinary research program. Because fish colonization was shown to be almost exclusively the result of young-of-the-year fish drifting in from the Danube River, the microhabitat requirements of juvenile fish have become one of the main objectives of our research.

We estimated micro habitat use in terms of water depth and flow velocity by electrofishing corresponding to the point sampling method. In 1995, during June and July, we sampled 1,202 micro habitats on 5 different dates. On the average, we found young-of-the-year fish in every third sample point. Fish used primarily areas of low velocities (< 2 cm/s) along the river banks and in backwater coves. We found juvenile fish of 16 different species (taxa). The sampled habitats were dominated by cyprinid species (*Rutilus rutilus, Gobio gobio* and *Chondrostoma nasus*) and a species of gobids (*Proterorhinus marmoratus*) also were frequent.

Habitat availability was investigated at three different flow conditions (2, 4 and 6 m^3/s); this range comprises more than 90% of the possible flow. Two representative stretches (about 100 m long), a nature-like and a channelized, were selected and within each, 5 profiles were measured.

Predicted weighted usable areas based on both water depth and velocity were compared with predicted weighted usable volumes across varying discharges and differing channel morphology.

KEY-WORDS: micro habitat utilization; depth, velocity; *Chondrostoma nasus, Gobio gobio, Rutilus rutilus, Proterorhinus marmoratus*; suitability indices; weighted usable area, weighted usable volume; Danube

INTRODUCTION

Only in the past few years the ecological requirements of Danubian fish species have been under large-scale scientific investigation. Studies on the habitat requirements of these species began in the early 1980's. The most important results of this research were that general habitat requirements of these species could be defined (Schiemer, 1988; Schiemer & Waidbacher, 1992), but detailed information about life-stage-specific habitat requirements remains scarce for most of these species (i.e. Spindler, 1988; Schiemer, et al., 1991; Copp et al., 1994).

The juvenile life stage plays an important role within the dynamics of fish populations. Juvenile micro habitat availability often represents a limiting factor for recruitment. Habitat models, based on species suitability curves and usable area, have been developed to estimate juvenile habitat needs but these models are primarily built on data from salmonid fishes (Bovee, 1982). Although some habitat modeling has been accomplished for warm water fish species (Gore et al., 1992) there is still a large deficit in knowledge concerning the habitat requirements of these species. Due to physiologically limitations (i.e. Dabrowski, 1986; Kaufmann & Wieser, 1992) these juvenile fish often occupy comparatively narrow niches in riverine systems. Typical juvenile micro habitats are lentic areas along the banks, in coves, oxbows, floodplain potholes, etc.. As these juvenile micro habitat areas are comparatively rare and show a patched distribution an accurate estimation of their availability is required to attain reasonable model results.

In general, fish utilize habitat in a three dimensional sense. Therefore, micro habitat use of fish could be considered more in terms of volume than in terms of area. Although some of the current habitat models predict use in terms of volume estimations most published studies focus only on area estimations.

Our goals were to develop habitat utilization curves to calculate and compare weighted usable habitat areas and volumes at varying discharges in order to assess habitat quality at two different morphologically structured study sites.

METHODS

Study area

The inlet of the MFK is located in the vicinity of Vienna. After 18 km, at Deutsch Wagram, the channel joins the Rußbach, which after 37 km itself flows into the Danube close to the Austrian-Slovakian border.

Within the frame of the interdisciplinary research project „Marchfeldkanalsystem" (Grubinger & Ernegger, 1994) the colonization and development of a recently constructed side-channel of the Danube is under a five-year investigation. This channel, which was flooded in 1992, provides water for irrigation and ground water infiltration purposes in the Marchfeld area. Due to the near-natural characteristics of the channel, a rapid colonization of organisms into the channel was expected.

The ecologically oriented construction provides various habitat structures in most parts of the channel; although some un-structured, monotone stretches exist as well. Due to the controlled flow in the channel the use of habitat by fish can be investigated under pre-defined flow conditions.

The immigration of juvenile fish into the channel through the inlet plays an important role for colonization, as the results indicated in the first years after initial flooding (Schmutz, et al. 1994). Although adult fish were scarce in the first two years, juveniles occurred in large numbers. The suitability of the created artificial habitats for early life stages is one of the main topics of our investigation program.

From an ecological point of view the MFK-system corresponds to a branch or side channel of the Danube with strongly regulated flow conditions. The system is fed with water from the Danube in the vicinity of Vienna through a screened inlet. The system is divided into two distinct parts. The upper part is the actual MFK and is about 19 km long with a newly created river bed. The MFK is designed to handle a maximum flow of 15 m^3/s. Its discharge is dependent on the water level of the Danube and is currently 2-6 m^3/s. The channel width is between 10 and 20 meters, maximum water depths range from 0.7 m to 1.8 m, and mean flow velocity is 0.3 - 0.9 m/s. The flow and water surface elevation are regulated by three weirs, each of which has a fish bypass. Due to a low gradient, the channel resembles a meta-potamal (lowland) river. The channel was originally filled with homogenous gravel substrate and subsequently fine substrates aggregated along the banks and in the coves.

Habitat suitability

For the micro habitat sampling we selected three representative stretches along the channel, each 200 m in length. Study site number one (code RS3) in the upper part of the channel displays various morphological features such as a meander, a gravel bar, and a cove. The main morphological features of study site number two (RS 4) are a meander and a 50 m long island. The monotone main channel of study site number three (RS 5) is accompanied by a 100 m-long cove (fig. 1).

In 1993, preliminary results concerning habitat use indicated that early life stages only occurred along the shore line and in coves, where the flow velocities are low and the water depth does not exceed 40 cm (Schmutz et al., 1994).

Within the 200 m long study sites we identified unique meso habitats by the following characteristics: steep or flat river bank; main channel, side arm or cove; rip-rap or vegetated bank; mean water depth and velocity; submerged vegetation (algae and macrophytes) and submerged roots, reeds, immersed vegetation and woody debris.

The size of the meso habitats ranged from 3-6 m^2. The number of sampled meso habitats of each type was chosen according to their relative occurrence within the study stretches. In each meso habitat we sampled 6 equally spaced micro habitats.

Micro habitat sampling was done according point sampling methods described by Persat & Copp (1988). We used a back-pack DC- electro shocker (1.5 kW) with a small anode (10 cm diameter, 1 mm mesh-sized net). Shocked fish were caught with a dip net (1 mm mesh size), counted and fixed in a 4 % formalin solution. Fish species were identified in the lab with the help of a dissecting scope. For data analyses we excluded fishes larger than 5 cm.

After each point sample we recorded water depth and flow velocity using an inductive two-dimensional current meter (DELFT Hydraulics®, p.-e.m.s.). The current meter was adjusted to automatically calculate a 10-second-interval-mean at a measurement frequency of 10 Hz for the two dimensions. Within an EXCEL 5.0® spreadsheet the resultant flow velocities of the two dimensional measurements were calculated. Due to the homogenous substrate composition along the banks, micro habitat surveys were done without recording substrate.

We sampled micro habitats 5 times between 23.06.1995 and 25.07.1995 and pooled the data for further computations. Habitat usability curves were directly derived from frequency graphs on which the frequency of individuals caught were plotted against the water depths and flow velocities used.

Available Habitat

Habitat availability was estimated using point measurements along transects at site 4 and 5 during 2, 4 and 6 m^3/s discharges. This range of discharge comprises more than 90% of the possible flow. Water depth and flow velocity were measured according the micro habitat point sampling procedure along the bank up to depths of 40 cm. With the help of a provisional footbridge we took flow velocity measurements at 4 cross-section points, at 10, 20, 30 and 40 cm total water depth. At each cross-section point we measured up to 7 points in the water column, at 0, 3, 8, 13, 23, 32 and 40 cm distance from the bottom, depending on water depth. At one transect, across the entire channel, the validity of the pre-designated discharge was controlled at each site.

For the computation of the conventional weighted useable area (WUA) we calculated the arithmetic mean of the measured column velocities. For the computation of the weighted useable volumes (WUV) we used the program SURFER®, which interpolated velocities between the measured points and so provided velocity isotach plots.

RESULTS

Figure 2 shows a representative cross section of the morphological/hydrological results at 2, 4 and 6 m^3/s discharge in site number 4. At a discharge of 2 m^3/s nearly the entire cross-section contains flow velocities <6 cm/s. At 4 m^3/s, the ratio between low and high velocities is equal, higher velocities occur outside of 1 m distance to the bank. At 6 m^3/s, higher velocities clearly dominate, low velocities were not found more than a half a meter to the shore line, or, are restricted to a 5 cm-thick layer above the bottom.

In site 3, 4 and 5, the juvenile stages (smaller than 5 cm) of 16 different fish species (taxa) were caught (Tab. 1). We sampled 1,224 points in four days and caught 1,198 individuals during June and July (23-25 June, 3-4 July, 11-13 July, 17-19 July, 24-25 July). The CPUE averaged about 1 fish per sample. Two thirds of the caught individuals had total lengths between 10 and 30 mm (Fig. 3).

For the four dominant (>10% of total catch) species we plotted utilization curves for flow velocity and depth as cumulative frequencies of sampled fish (Fig. 4). All four species primarily used areas of very low velocities (0-2 cm/s) in a comparatively narrow range and low water depth (5-20 cm) in a broader range.

Habitat use varied among species. Although all species showed the highest index at the lowest velocity class (0-2 cm/s) rheophilous species such as *Chondrostoma nasus* (n=173) and *Gobio gobio* (n=111) used higher velocities than the stagnophilous *Rutilus rutilus* (n=355). The bottom-dwelling *Proterorhinus marmoratus* (n=89) used the near-bottom layer and therefore can withstand higher velocities. Also, in terms of water depth, there was a potential segregation between the species. *Ch. nasus* and *G. gobio* used the shallow (5-10 cm) areas, while *P. marmoratus* and *R. rutilus* also occurred in deeper areas (5-30 cm).

Figure 5 shows the flow velocity/water depth plot based on micro habitat samples. Within the measured band along the bank there was no relationship between these two parameters.

Figure 6 shows the velocity based weighted usable areas (v-WUA) at a discharge of 2, 4 and 6 m^3/s. The usable habitat in site 5 (channelized) was only a small proportion of what we found for site 4 (meandering).

Figure 7 shows the velocity based, weighted usable volumes (v-WUV) at discharges of 2, 4 and 6 m^3/s. The v-WUV of site 4 was highest at 2 m^3/s and was reduced to 80 % and 77 % at 4 m^3/s and 6 m^3/s. The v-WUV of site 5 was also highest at 2 m^3/s, diminishes to 70 % at 4 m^3/s and increased again to 98 % at 6 m^3/s.

Also the v-WUA of site 4 and 5 showed a similar decrease from 2 to 4 m^3/s discharge but increased again at site 4 and nearly tripled at site 5 at 6 m^3/s. The ratio of usable habitat between site 4 and 5 was higher for v-WU$\underline{V}$ than for v-WU$\underline{A}$, at all discharges. At 6 m^3/s, the ratio in v-WU$\underline{V}$ between site 4 and 5 was 6.4, while using v-WU$\underline{A}$ it was only 3.6.

Figure 8 shows the water depth-based, weighted usable volume (d-WUV) for different discharges. The d-WU$\underline{A}$ values for site 4 are a little bit lower and for site 5 much higher than the v-WU$\underline{V}$.

Table 1: Fish species list (taxa) of micro habitat sampling

Abramis ballerus
Abramis brama
Alburnus alburnus
Aspius aspius
Barbus barbus
Blicca bjoerkna
Carassius auratus
Chondrostoma nasus
Cyprinidae (unkeyed)
Gasterosteus aculeatus
Gobio sp. (unkeyed)
Leuciscus sp. (unkeyed
Percidae (unkeyed)
Proterorhinus marmoratus
Rutilus rutilus
Scardinius erythrophthalmus

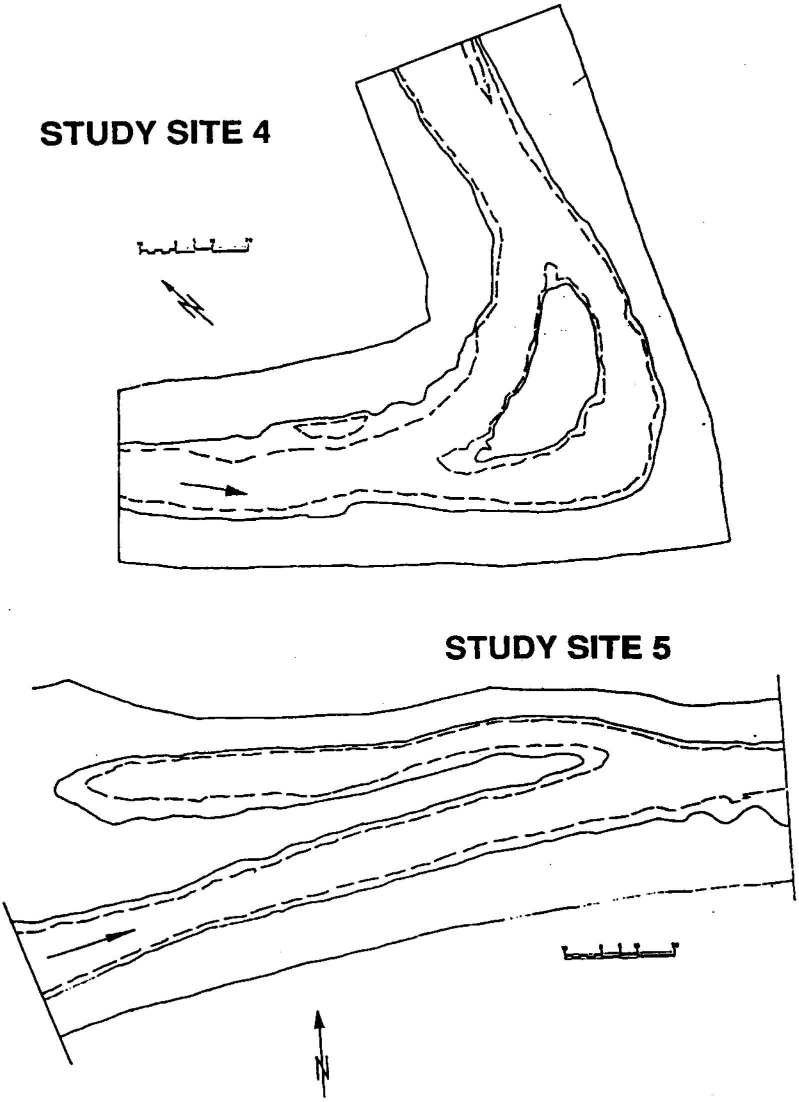


Figure 1: Schematic map of study site 4 (meandering) and site 5 (channelized).

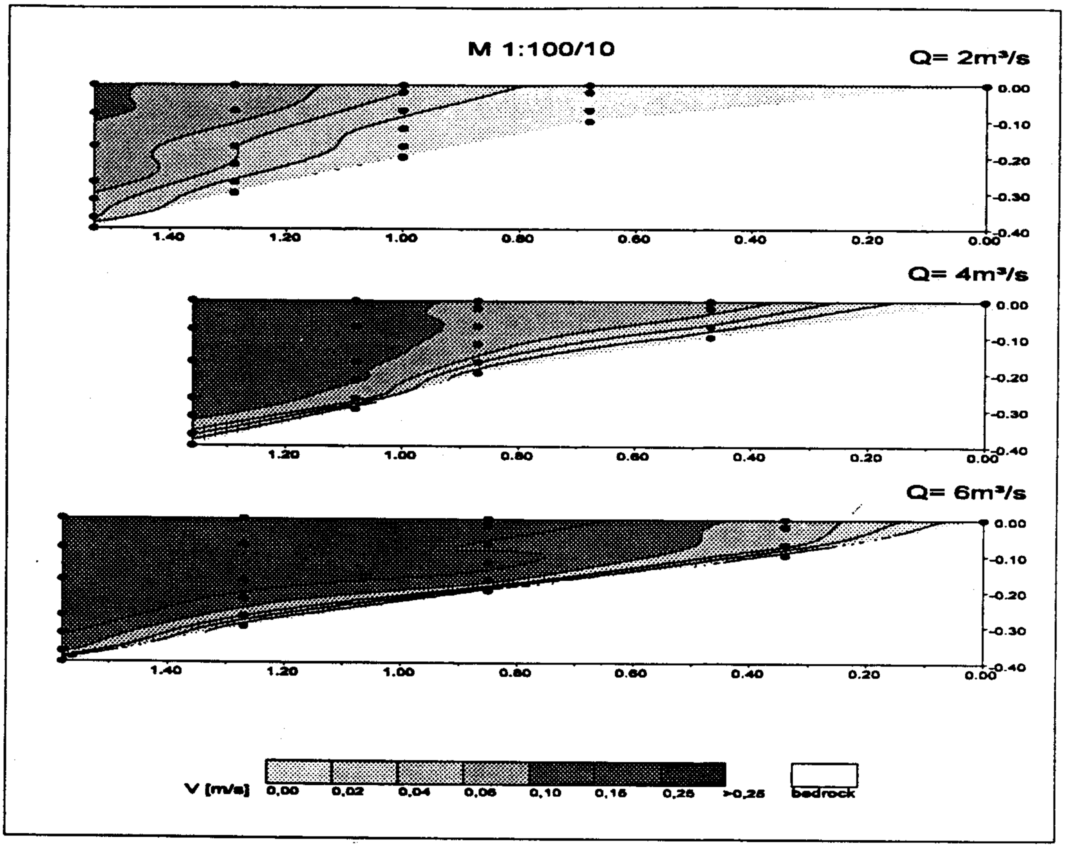


Figure 2: Example of the morphological/hydraulic conditions in a representative cross-section of site 4. Dots represent measured velocity points (scale 1:100 and 10 times excessive).

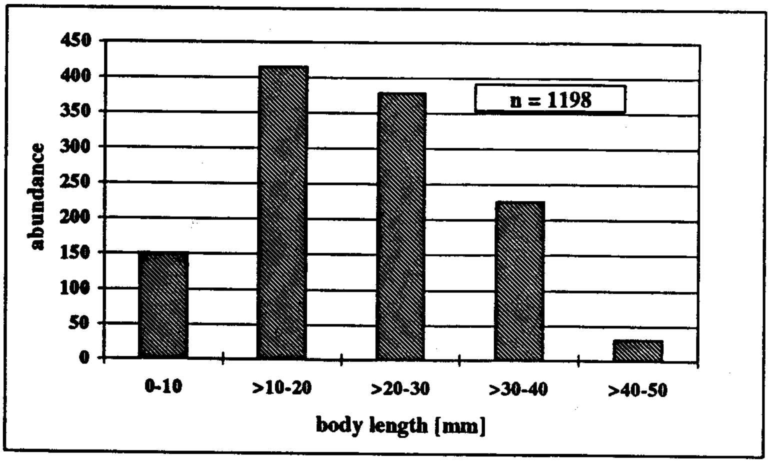


Figure 3: Total-length/frequency-plot of juvenile fish caught by micro habitat point sampling

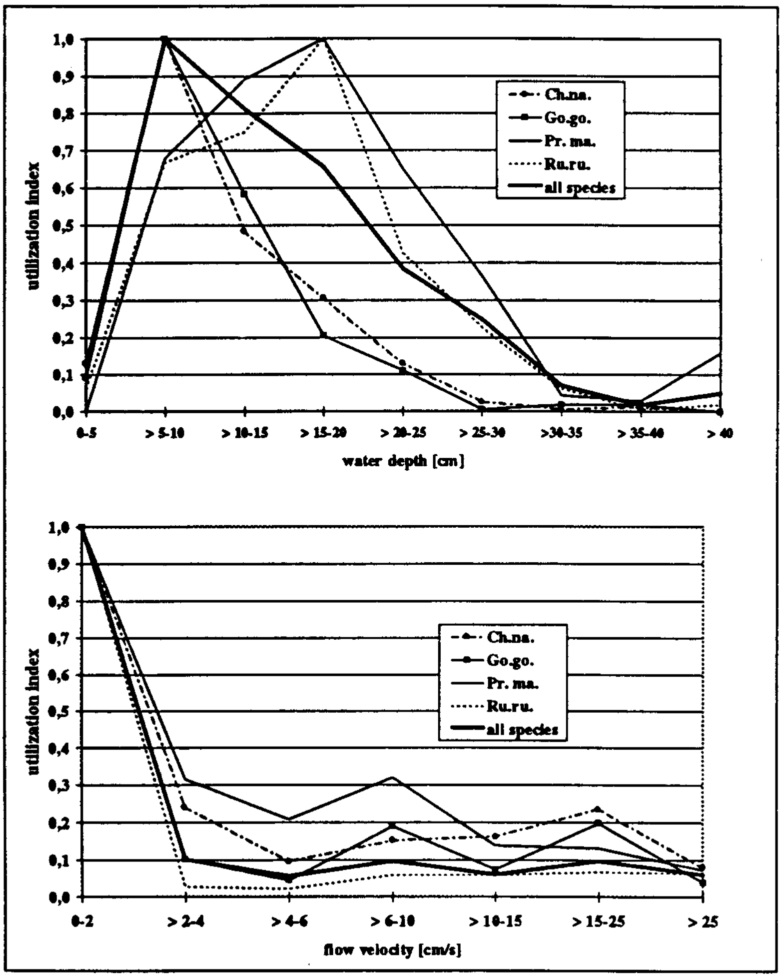


Figure 4: Microhabitat use (depth and flow velocity) by juveniles for four different species and altogether (16 taxa)

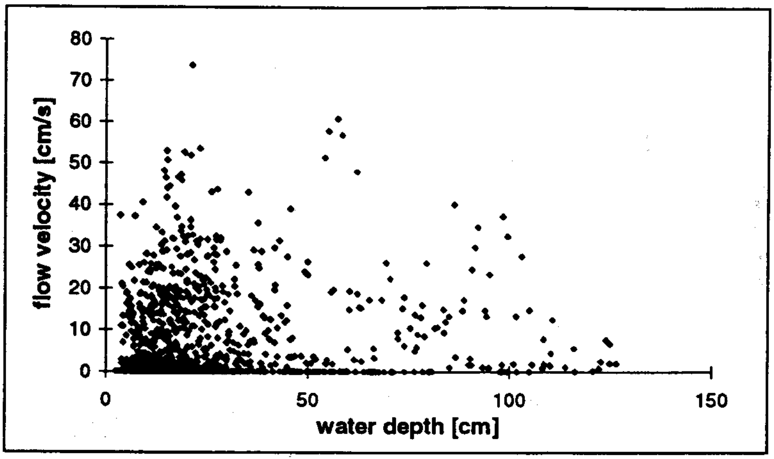


Figure 5: Velocity/depth-plot of micro habitat sampling points

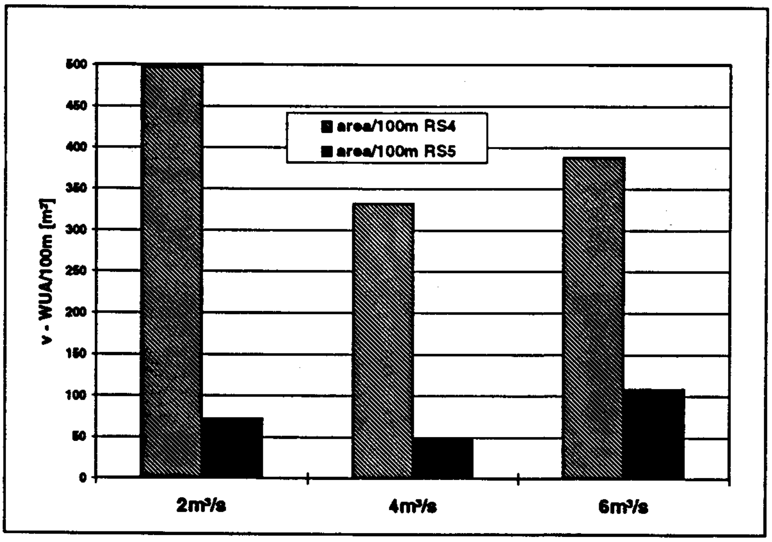


Figure 6: Comparison of flow velocity-based, weighted usable areas (v-WUA) for juvenile fish at site 4 (meandering) and site 5 (channelized) at low (2 m³/s) and mean (4 and 6 m³/s) flow conditions

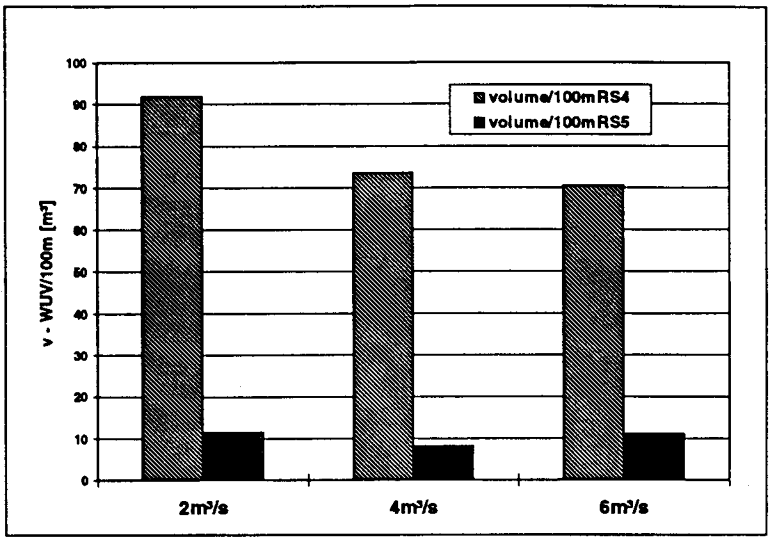


Figure 7: Comparison of flow velocity-based, weighted-usable volumes (v-WUV) for juvenile fish at site 4 (meandering) and site 5 (channelized) at low (2 m^3/s) and mean (4 and 6 m^3/s) flow conditions

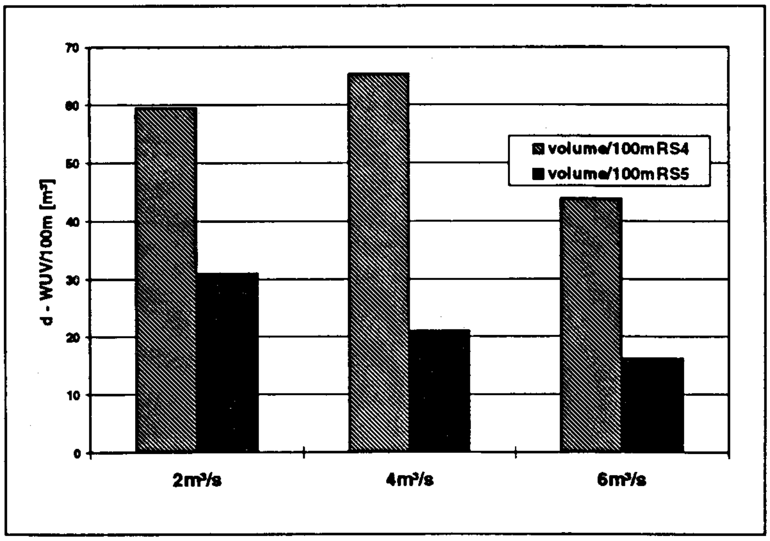


Figure 8: Comparison of water depth-based, weighted-usable areas (d-WUV) for juvenile fish at site 4 (meandering) and site 5 (channelized) at low (2 m^3/s) and mean (4 and 6 m^3/s) flow conditions

DISCUSSION

Habitat suitability

For the calculation of the WUA, we chose the most abundant, juvenile fish species as target organisms. If we compare the species distribution of the juveniles with that of the entire fish fauna we get a better view of the validity of our micro habitat sampling method. Altogether, including other investigations at the channel, we found 43 fish species (adults and juveniles) within the MFK-system in 1995 (Schmutz & Unfer, in prep). The fish fauna of the MFK channel itself was dominated by 7 species, representing more than 80 % of the total catch. These species also occurred as juveniles. The species frequencies of the juveniles caught by the micro habitat sampling differ from that of the adults. Three of the dominant adult species also were abundant as juveniles (*G. gobio, R. rutilus* and *P. marmoratus.*). Due to the lack of adults the high densities of *Ch. nasus* only can only be explained by their drift from the Danube into the MFK (Schmutz et al., 1994). Based on this accompanying data we assume that sampling methods were sufficient to estimate microhabitat use of abundant juveniles. Despite the overall high densities of juveniles the sampling of about 1,200 points provided satisfactory data for only 4 species. In order to calculate reasonably accurate utilization curves for rarer species, a larger sampling effort would be needed.

In general, an increase in distance from the bank and consequently deeper water depth results in higher flow velocities. Since we were able to produce unambiguous habitat use curves for both velocity and depth, and these two parameters were independent, both parameters influence microhabitat choice.

The limitation of usable habitat by flow velocity is physiologically based. Juvenile fish cannot withstand flow velocities 2 to 7 times higher than their body length for longer periods (i.e. Kaufmann & Wieser, 1992). The cruising speed is only a fraction of that and therefore juvenile fish select the lowest available velocities. On the other hand we caught a fairly high proportion of fish in areas of higher velocities; this could not be simply explained by methodological biases. Sometimes, the measured velocities used exceeded the maximum swimming capacity of observed fish. We assume that, in at least a proportion of these cases fish were not resident but drifting. This assumption is supported by the fact that most of our larval and juvenile fish species exhibit a distinct drift phase during their early development (SCHMUTZ et al., in prep).

Ch. nasus and *G. gobio* show higher utilization indices at higher velocities than *R. rutilus*. This agrees to the rheophilous classification of these two species.

Care must be taken in interpreting the utilization curve of *P. marmoratus*. Due to our sampling method, fish were caught either in the upper or lower part of the water column but not directly at the bottom. The bottom-dwelling *P. marmoratus* were obviously attracted to the anode and caught in places where they normally do not occur. Therefore, we believe that the comparatively high utilization values for velocity in the range between 2 and 25 cm/s are overestimated. So we believe that *P. marmoratus* is not rheophilous as our data suggest and the „semi-rheophilous“ classification of Copp et al (1994) and the „indifferent“ of Schiemer & Waidbacher (1992) seems to be more appropriate.

R. rutilus obviously show the highest preference to low velocities. Even low velocities of 2-4 cm/s show a use index of only 0.03. Garner (1995) found that within 4 pre-defined velocity categories, *R. rutilus* preferred „null-velocity“ areas but also occurred in areas of „weak, medium and strong“ velocities. These data were obtained by visual velocity estimations and therefore the comparability to our data is limited.

The utilization index in terms of water depth indicates a habitat partitioning. *Ch. nasus* and *G. gobio* mainly use shallow areas 5-10 cm-deep, *P. marmoratus* and *R. rutilus* mainly use 15-20 cm-deep water. Little is known about habitat partitioning during the early life history. Further studies on this topic are needed.

WUA/WUV

At site 4 the widened profile and the island provide large shallow areas at low flow conditions (2 m^3/s discharge). When discharge increases to 4 m^3/s some additional flat areas are flooded but also flow velocity increases. At 6 m^3/s both, water depth and flow velocity are increased resulting in a reduction of WUV.

At site 5, an increase in discharge results in a decrease of WUA except in terms of v-WUV, where inundated vegetation offers a velocity refuge and hence produces as much habitat as during low flow conditions. At site 4, due to the small proportion of inundated vegetation in comparison to the total WUV, this effect is negligible.

Due to the diverse morphology at site 4, the v-WUV is up to 10 times, and the d-WUV up to 3 times, higher than at site 5 (monotone). Therefore, the differences in usable habitat between site 4 and 5 can be more obviously demonstrated by the parameter flow velocity. At site 4, in each of the calculated WUV, the v-WUV is higher than the d-WUV. At site 5, it is vice versa. In site 4 (widened channel), water depth is apparently the more limiting factor than in site 5 (straightened, narrow site), where the flow velocity is the more limiting factor.

Future analyses will focus in combining the flow velocity and water depth index and comparing these results with our single-factor-based interpretation. Additional microhabitat parameters, such as vegetation and woody debris, should also be considered in ongoing studies. Finally, both WUA and WUV must be validated as predictors for fish abundance.

ACKNOWLEDGEMENTS

We wish to thank Christian Wiesner for supporting the data analysing and the Austrian Scientific Foundation for funding the project.

REFERENCES

Bovee, K.D. (1982): A Guide to Stream Habitat Analysis Using the Instream Flow Incremental Metholodgy, FWS/OBS-82/26, Biological Services Program, U.S. Fish and Wildlife Service, Fort Collins, CO.

Copp, G., H. G. Guti, B. Rovný & J. Çerný, (1994): Hierarchical analysis of habitat use by 0+ juvenile fish in Hungarian/Slovak flood plain of the Danube River. *Environmental Biology of Fishes* 40, 329-348.

Dabrowski, K., (1986): Energy utilization during swimmng and cost of locomotion in larval and juvenile fish. *J. Appl. Ichthyol*3:117- 124.

Garner, P., (1995): Suitability indices for juvenile 0+ roach (Rutilus rutilus, L.) using point abundance sampling data. *Regul. Riv.* Vol. 10, 99-104.

Gore, J.A., J.B. Layzer & I.A. Russell, (1992): Non-traditional applications of instream flow techniques for conserving habitat of biota in the Sabie River of Southern Africa. In: P.J. Boon, et al (Ed.), River conservation and management. John Wiley & Sons Ltd., pp161-177.

Grubinger, H., & T. Ernegger, (1994): Interdisziplinäre Forschung Marchfeldkanal. Ein Schwerpunktsprojekt des Fonds zur Förderung der Wissenschaftlichen Forschung in Wien. *Wetter und Leben*, 46/4: 185-198.

Kaufmann, R. & W. Wieser (1992): Influence of temperature and ambient oxygen on the swimming energetics of cyprinid larvae and juveniles. *Environmental Biology of Fishes* - 33, S. 87 - 95.

Persat, H. & G. H. Copp, (1988): Electrofishing and point abundance sampling for the ichthyology of large rivers. In Cowx, I. G. (Ed.) Developments in Electrofishing. Fishing News Book, Hull. pp 203-215.

Schiemer, F., T. Spindler, H. Wintersberger, A. Schneider & A. Chovanec, (1991): Fish fry associations: Important indicators for the ecological status of large rivers. *Verh.Int.Ver.Limnol.* 24, 2497-2500.

Schiemer, F. & H. Waidbacher, (1992): Strategies for Conservation of a Danube Fish Fauna. River Conservation and Management. Edited: P.J. BOON et al. John Wiley & Sons Ltd..

Schiemer, F., (1988): Gefährdete Cypriniden - Indikatoren für die ökologische Intaktheit von Flußsystemen. *Natur und Landschaft*, 63. Jg. Heft 9, 370-373.

Schmutz, S.& G. Unfer, (in prep): Interdisziplinärer Forschungsschwerpunkt Marchfeldkanalsystem - Fischökologie. 3. Zwischenbericht. Universität für Bodenkultur Wien.

Schmutz, S., A. Zitek & Ch. Dorninger, (in prep): A new automatic drift sampler for juvenile fish.

Schmutz, S., S. Matheisz, A. Pohn, J. Rathgeb & G. Unfer (1994): Erstbesiedelung des Marchfeldkanals aus fischökologischer Sicht. *Österreichs Fischerei*. 47/7, 158-178.

Spindler, T., (1988): Ökologie der Brutfische in der Donau bei Wien. Dissertation Univ. Wien. 119pp.

DESCRIPTION OF A TECHNIQUE FOR MULTISCALE ANALYSES: SPATIAL SCALING AND HABITAT SELECTION IN RIVERINE HABITATS

T.P. Bult[1], R.L. Haedrich[1], and D.C. Schneider[2]
[1] Department of Biology, Memorial University of Newfoundland, St. John's, Newfoundland, A1C 5S7
[2] Ocean Sciences Centre, Memorial University of Newfoundland, St. John's, Newfoundland, A1C 5S7
Canada
e-mail T.P. Bult: tbutt@kean.ucs.mun.ca

ABSTRACT

We argue that a single scale approach in habitat modelling fails to appreciate the complexity of habitat selection behaviour by fish, that conclusions from habitat modelling studies are scale dependent, and that this situation limits managerial effectiveness. Therefore, multiscale analyses are crucial to an understanding of habitat selection behaviour, for habitat modelling, and for management. We propose a quantitative multiscale technique based on frequency analysis and randomization to study habitat selection by fish in riverine habitats. The technique can be used over any range of spatial scales in an environment with irregular boundaries. We illustrate the approach using simulated distributions and field data on juvenile Atlantic salmon (*Salmo salar*) distributions, obtained by snorkelling in North Harbour River, Newfoundland, Canada. We suggest that current habitat models could be improved by a more explicit use of scale, which would improve assessment and prescription of habitat requirements of fish. Future habitat selection studies should focus on the identification of scales that are most appropriate to management questions.

KEY-WORDS: scale / spatial pattern / patchiness / heterogeneity / point pattern / habitat selection / habitat modelling / habitat assessment / juvenile Atlantic salmon (Salmo salar) / randomization

INTRODUCTION

It has long been recognized that the scale of measurement is one of the most important aspects influencing the results of ecological studies. Recent publications re-iterate the importance of scale (Wiens 1989, Menge and Olson 1990, Holling 1992, Levin 1992, Horne and Schneider 1995). Nevertheless, most empirical ecological studies are characterized by the use of single or few measurement scales and a rather implicit use of scaling. Often, the measurement scale chosen is not the result of a quantitative multiscale approach, but is based on the biological intuition of the researcher and logistical constraints. This informal treatment of scaling carries over into habitat modelling and habitat selection studies, despite a general perception of the importance of scaling (e.g. Frissel et al. 1986, Minshall 1988).

This informal approach arises from a combination of reasons. The first reason is that "scale" has numerous meanings in the ecological literature. In this paper, we recognise "scale" as "the resolution within the range of a measured quantity" (Schneider 1994). A second reason could be an unfamiliarity among ecologists with the mathematical tools available to deal with scaling, in spite of several publications on the subject (Platt and Denman 1975, Ripley 1981, Greig-Smith 1983, Upton and Fingleton 1985, Legendre and Fortin 1989, Schneider 1994, Horne and Schneider 1995). A third reason is that multiscale analyses generally require large data sets, collected over a range of scales. This has confined empirical multiscale analyses to studies based on techniques such as echo sounding or remote sensing, which generate large amounts of data (e.g. Weber et al. 1986, Horne 1994).

A number of mathematical tools are available to assess patchiness of fish distributions and associations of fish with habitats at a range of scales. Some of these tools can be used to cover a fixed number of scales; others can be used to examine a wide range of scales simultaneously.

To assess patchiness at a fixed scale, a variety of indices were developed based on variance to mean ratios, such as Morisita's index (Morisita 1959), Lloyd's index of mean crowding (Lloyd 1967), or the exponent of Taylor's Power law (Taylor 1961). To assess associations of fish with habitats at a fixed scale, correlation, regression, and frequency analyses can be used. These techniques are generally not applied over a range of scales, although all can be. Methods that examine patchiness over a range of scales, rather than a few selected scales, are pattern analysis (Greig-Smith 1983), second-order neighbourhood analysis (Getis and Franklin 1987) and spectral analysis (Platt and Denman 1975). Coherence analysis explicitly examines associations over a range of scales.

None of the techniques mentioned above is suitable for habitat selection studies in riverine habitats. Multiscale analyses based on variance to mean indices, correlation, regression or frequency analysis only provide information at a few explicitly stated scales (e.g. Downes et al. 1993). No information is obtained if the scales of the observations were set at scales of maximum variance or maximum association. Pattern analysis, second-order neighbourhood analysis, spectral analysis and coherence analysis could theoretically be used to cover a wide range of scales. However, the irregular system boundaries of riverine habitats and the sensitivity of the methods to low densities where zeros are common (Fasham 1978, Upton and Fingleton 1885) limits their use in empirical studies.

The irregularity of system boundaries in rivers precludes the use of 2 dimensional spectral analysis or 2 dimensional coherence analysis, because these methods can only be used for rectangular distribution maps. The irregularity of system boundaries also limits the use of pattern analysis because of the difficulty of positioning random or nested quadrats and limits the use of second-order neighbourhood analysis because of the difficulty of obtaining boundary corrections.

The sensitivity of the methods for low densities precludes interpretation of the data at small spatial scales. This sensitivity varies for transect and grid system approaches.

Transect and grid systems are characterised by their length, width, resolution and range. When using a transect approach, it makes sense to use a transect width that is similar to the smallest transect resolution, unless the smallest transect resolution is larger than the river width. In this case the transect width will equal the river width. Habitat selection studies generally operate at spatial resolutions far smaller than the river width (micro-habitat modelling) to several times the river width (macro-habitat modelling). To cover this range of spatial scales in multiscale analyses, the transect width will be small. Because of the small transect width, a transect approach will show a high occurrence of zero counts at very small spatial resolutions. This prevents interpretation of the data at these scales unless a large number of transects is studied.

The analysis of grid based data is also sensitive to zero counts. However, when decreasing the spatial resolution of data from grid systems, the occurrence of zeros declines more rapidly than in a transect approach. For example, halving the transect resolution will double the average number of observations per bin, but doubling the width of cells in a grid system approach will quadruple the number of observations per cell. Multiscale analyses at small spatial resolutions in environments of low densities will therefore be more effective when a grid system is used rather than a transect system.

Because of the irregularity of the boundaries of riverine habitats and the high occurrence of zeros at small spatial resolutions, there is a need for a method for analysing patchiness of fish distributions and associations of fish with habitats over a wide range of spatial scales, from far smaller than the river width to several times the river width, in a grid system approach. In this paper we present a quantitative multiscale approach to study the distribution and associations of fish with habitats over this range of spatial scales. This method is based on frequency analysis and randomization. The method will be illustrated using simulated fish distributions as well as field data collected in North Harbour River, Newfoundland, in 1994. The North Harbour River data will only be used in this paper to illustrate the method.

METHODS

Study Site

A selected reach of North Harbour River was studied. North Harbour River is located on the Avalon Peninsula of Newfoundland, Canada (47°11'30" N, 53°37'0" W). The river drains 86 km^2 of mostly forested land underlain by Precambrian volcanic and sedimentary rock. The fish community in North Harbour River is composed of Atlantic salmon *Salmo salar*, brook trout *Salvelinus fontinalis*, brown trout *Salmo trutta*, threespine stickleback *Gasterosteus aculeatus*, and American eel *Anguilla rostrata*. The river is further described by DeGraaf and Bain (1986).

The study reach was approximately 5 km upstream from the river mouth and consisted of a combination of riffle, run and pool habitats. The length of the study reach was 120 m and the average width 9 m. The fish community at the study site is predominantly juvenile Atlantic salmon. Brook and brown trout were relatively rare ($<10\%$ by number).

Habitat Mapping

The habitat mapping was done using metal T-posts as reference points in an XY-grid covering the study section. The entire study reach was mapped in terms of substrate, water depth (cm), water velocity (cm sec^{-1}) and cover. In this paper only the depth data are used.

To facilitate mapping, a 1*1 m PVC frame was used. This frame was divided into 4 (50*50 cm) and 9 (33*33 cm) cells with coloured twine. The frame was positioned in the XY-grid using measuring tapes and the reference T-posts. Next, the habitat observations were done at the centre of the cells within the frame: Substrate and cover were mapped with a resolution of 9 observations per square meter. Depths were mapped with a resolution of 4 observations per square meter. Water velocities (at 60% of depth and bottom) were mapped with a resolution of 1 observation per square meter, without the use of a frame.

Fish Distribution And Habitat

Fish distribution surveys were made on 17 August and 25 August, 1994. One survey took approximately 6 hours. The weather conditions during the surveys were similar. The water temperatures during the course of both surveys varied from 16 to 21 °C.

Fish were observed by snorkelling in an upstream direction in a zigzag pattern, intended to minimize disturbance of the fish. Observed fish positions were marked using numbered weights. Data recorded when snorkelling included: species, age class (0+, 1+, 2+, >2+; estimated from size), height above bottom (cm), and activity (moving, holding position). All fish observed within the section were recorded.

After snorkelling, the numbered markers were mapped relative to the XY-grid, to the nearest 5 cm. Water velocities, snout velocities (cm sec^{-1}), depth, cover and substrate were mapped at the locations of the fish that held positions.

Both the habitat mapping and the fish distribution surveys were done at a specific water level. The discharge at this water level was estimated at 0.25 m^3 s^{-1}. This level was chosen because it was the most prevalent discharge in the summer of 1994.

Data Analyses

A FORTRAN program was written to address a series of questions.

1. How are fish positioned relative to each other? This was addressed by computing spatial autocorrelations of fish positions.
2. How are fish of group 1 positioned relative to fish of group 2? This was addressed by computing spatial associations between positions of two groups of fish (cohorts, species).
3. How are fish distributed relative to a previously recorded distribution? This was addressed by computing spatial associations between fish positions recorded on separate surveys.
4. How are fish distributed relative to the distribution of a habitat feature? This was addressed by

computing spatial associations of fish positions with identified habitat features.

5. How are habitat features positioned relative to each other? This was addressed by computing spatial autocorrelations of habitat features.

In the following text, these questions will be referred to as components 1-5. The computational flow was similar for each component, as demonstrated below by an example based on component 4.

Component 4 compares differences in the habitat surrounding observed fish positions with the habitat surrounding computer-generated random fish positions. This comparison is made over a range of ambit radii (Figure 1).

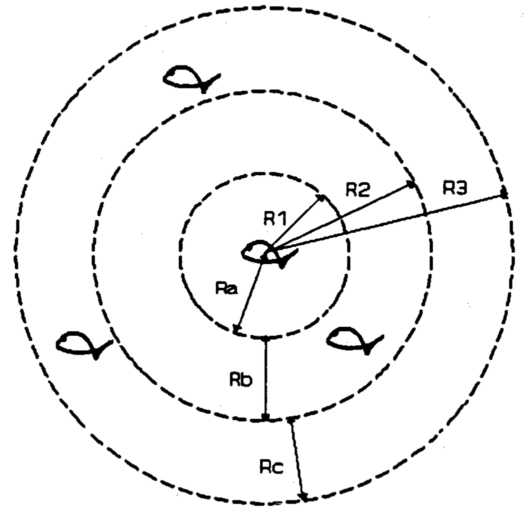


Figure 1. Illustration of the ambit concept. The multiscale analyses can be done at increasingly larger ambit radii (R1-R2-R3) or at increasingly larger distance slots (Ra-Rb-Rc).

In the FORTRAN program, the Contact of each observed individual fish with the habitat feature of interest is calculated first. Contact is quantified as the percentage occurrence of a specified class of a habitat variable (e.g. depth class, substrate class) from a map of evenly positioned habitat observations. From this, the average Contact is calculated by averaging the Contact across all individual fish observations:

$$C_{obs,i} = \frac{1}{n} * \sum_{f=1}^{n} \frac{Nh_{if}}{Nt_{if}} \tag{1}$$

$C_{obs,i}$: average Contact of fish with specified habitat at ambit radius i

n: total number of fish observed

Nh_{if}: number of observations of specified habitat within distance i from fish position f

Nt_{if}: total number of habitat observations within distance i from fish position f

To evaluate the observed Contact relative to a random outcome, a randomized fish distribution is created by randomly repositioning all fish within the study site for each survey separately. The random distributions were created using the FORTRAN system-supplied random number generator, upgraded using the shuffle procedure (Press et al. 1986) to break up possible sequential correlations. From the randomized distributions, the average Contact (C_{ran}) is calculated for each survey and as an average of all surveys. This procedure is repeated 500 times. For each of these 500 randomized distributions, C_{obs} is compared to C_{ran}. From this, p-values are derived that can be used as selection criteria to test if the Relative Contact is significantly different from 0, i.e. if the observed distribution is significantly different from the randomized distribution. Finally, an average C_{ran} is calculated based on all 500 observations of C_{ran}. From this, the Relative Contact at radius i is calculated (RC_i):

$$RC_i = \log_{10}(C_{obs,i}) - \log_{10}(C_{ran,i}) \tag{2}$$

By calculating the Relative Contact over a range of ambit radii, an impression can be obtained of how fish are associated with habitat features over a range of spatial scales. Figure 2 shows the steps involved in the multiscale program.

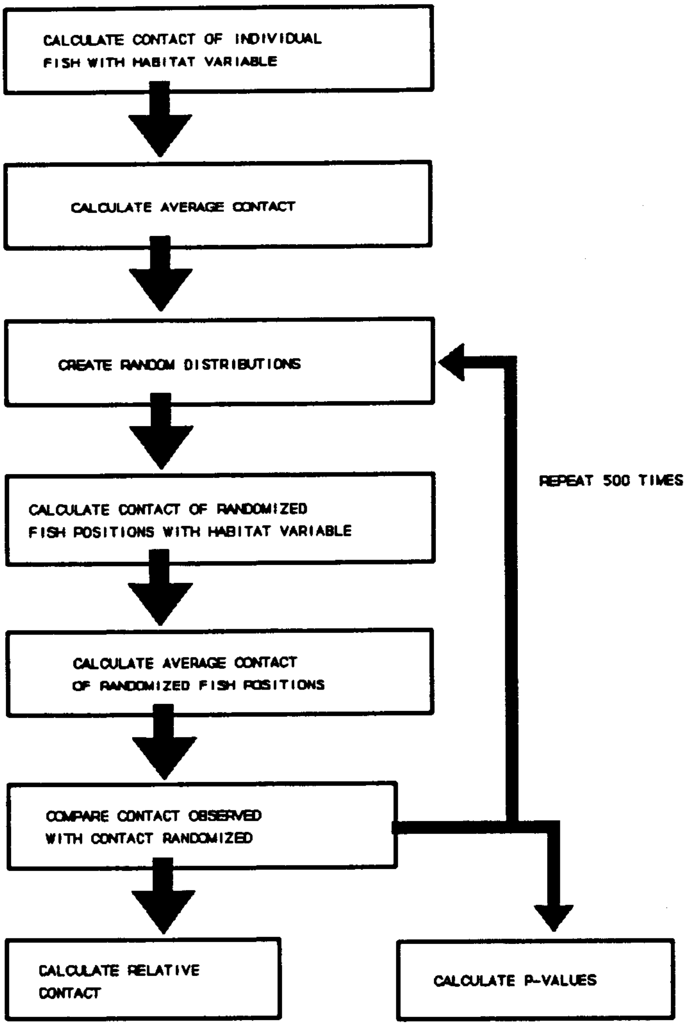


Figure 2. Flow of calculations used to investigate habitat associations of fish over a range of spatial scales.

A few aspects of the general procedure for components 1, 2, 3 and 5 differ from 4. For analyses aimed at investigating spatial autocorrelations of fish positions (component 1), or analyses aimed at investigating spatial associations of different fish populations (components 2-3), Contact is quantified by using fish densities. To obtain density estimates, the program creates evenly distributed dummy positions within the study site. The ratio of fish observations versus dummy positions is subsequently used as an estimate of fish densities.

To facilitate the comparison of separate surveys for component 1, which might differ in the number of fish observations, all density estimates are rescaled as a percentage of the number of fish observed per survey minus one. For component 2-3, all densities are rescaled as a percentage of the total number of fish observed per survey. C_{obs} will therefore provide an estimate of the Contact of an average fish in a particular group with the other fish of the same group (component 1) or with fish of another group (components 2-3).

The randomized fish distributions in components 2 and 3 are created by randomly repositioning only one of the two fish groups. For component 5, the randomized distribution is created by randomly allocating the habitat observations to the positions where these habitat observations were made, using sampling without replacement.

Note that in a situation of 2 surveys, 3 estimates for RC are obtained per ambit radius in components 1, 2 and 4 (one for each survey separate and one based on both surveys). One estimate per ambit radius is obtained for components 3 and 5.

All analyses are done over a range of ambit radii in two ways: at increasingly larger ambit radii (1) and at increasingly larger distance slots (2), as in Figure 1. The smallest ambit radius will differ among analyses, due to differences in resolution of the mapping of the habitat and fish distributions: For associations of fish positions, the smallest ambit radius should exceed 5 cm to ensure that at least one other possible fish position is within the ambit of each possible fish position. For associations of fish with depth, the smallest ambit radius should exceed 36 cm ($=\sqrt{(25^2+25^2)}$) to ensure that at least one depth observation is within the ambit of each possible fish position. For analyses aimed at spatial autocorrelation of depth observations, the smallest ambit radius should exceed 50 cm to ensure that at least one other habitat observation is within the ambit of each habitat observation.

The value of Relative Contact allows positive associations ($RC>0$), indicating preference behaviour, to be distinguished from negative associations ($RC<0$), indicating avoidance behaviour (applications 2, 3 and 4), and clumping ($RC>0$) from random ($RC=0$) and repulsion ($RC<0$) (applications 1 and 5). For example, a Relative Contact of 1 indicates that the average fish observed has 10 times more contact with a particular habitat feature compared to an average fish of the randomized distribution. A Relative Contact of -1 indicates that the average fish observed has 10 times less contact with a particular habitat feature compared to an average fish of the randomized distribution. The RC is therefore more readily interpretable than the Habitat Preference Indices used in many habitat selection studies.

Analyses Done

To test components 1 and 5 against a known situation, a total of 290 fish were evenly distributed by the program within 6 randomly positioned schools in a 100*100 m area (=group 1 fish). The minimum distance between fish in a school was 1.5 m. The radius of the (circular) schools was 6 m. The Relative Contact was quantified at increasingly larger ambit radii. The Relative Contact was expected to show a minimum at small spatial scales (ambit radius <1.5 m, $RC<0$) due to the minimum fish distance and a maximum at intermediate spatial scales (ambit radius = 10^1 m, $RC>0$) due to schooling. The Relative Contact was expected to decline to 0 at spatial scales larger than 10^1 m.

To test components 2 and 3 against a known situation, a total of 292 fish were evenly distributed within 6 randomly positioned schools in an 100*100 m area (=group 2 fish) in addition to the distribution of the previous example. There was no overlap between schools. The minimum distance between fish in a school was 1.5 m. The radius of the (circular) schools was 6 m. The Relative Contact between fish of group 1 with fish of group 2 was quantified at increasingly larger ambit radii. The Relative Contact was expected to show a minimum at small spatial scales (ambit radius $<10^1$ m, $RC<0$) due to the spatial separation of schools. The Relative Contact was expected to approach 0 at ambit radii larger than 10^1 m.

To test component 1 against an observed situation, the spatial autocorrelation of the 0+ salmon distribution of both surveys in the North Harbour river study reach was investigated. It was expected that the RC would be negative at small ambit radii (< 15 cm) due to spacing behaviour and competitive interactions. At larger ambit radii (15 cm - 2 m) a positive and gradually declining RC was expected due to selection of the 0+ salmon for primarily small scale (< 1 m^2) environmental features.

To test component 4 against an observed situation, the depth preference of the 0+ salmon was investigated over a range of spatial scales (maximum ambit radius=10 m) using the distribution data of both surveys. For this, the depth observations were rescaled into 6 depth classes: class 1: depth [0-4] cm; class 2: <4-8] cm; class 3: <8-16] cm; class 4: <16-32] cm; class 5: <32-64] cm; class 6: <64-∞] cm. It was expected that the 0+ salmon would be positively associated with the intermediate depth classes (16-32 cm) and negatively associated with the shallow depth classes (0-8 cm) at small spatial scales, as observed in other studies (DeGraaf and Bain 1986, Heggenes 1990). At larger spatial scales however (>4-7 m), the reverse was expected due to avoidance of pool areas and selection for riffle/run areas by the fish.

To test component 4 against a known situation, the 0+ salmon were randomly repositioned within the North Harbour river study section for both surveys. Associations of these distributions with depths were investigated over a range of spatial scales (maximum ambit radius = 10 m). It was expected that the RC would not be significantly different from 0 across all spatial scales investigated.

RESULTS

A total of 977 juvenile salmon, 47 brook trout and 8 brown trout were observed during both surveys. The distribution of these fish over the study section was very similar on both days. Figure 3 illustrates the distributions of the 0+ juvenile salmon on 17 August and 25 August, 1994. The percent occurrence of the depth classes 1-6 was 7%, 12%, 26%, 39%, 15% and 1% respectively.

The proposed multiscale approach detected scale dependent patterns in the simulated distributions of the schooling fish (Figure 4): The Relative Contact for the spatial autocorrelations of group 1 fish showed a minimum at small spatial scales (ambit radius $<10^0$ m, RC<0) and a maximum at intermediate spatial scales (ambit radius = 10^0 to 10^1 m, RC>0). The Relative Contact declined to 0 at larger spatial scales. The Relative Contact for the spatial associations between the first and the second group of schooling fish showed a minimum at small spatial scales (ambit radius $<10^1$ m, RC<0). At larger spatial scales (ambit radius $>10^1$ m) the RC approached 0.

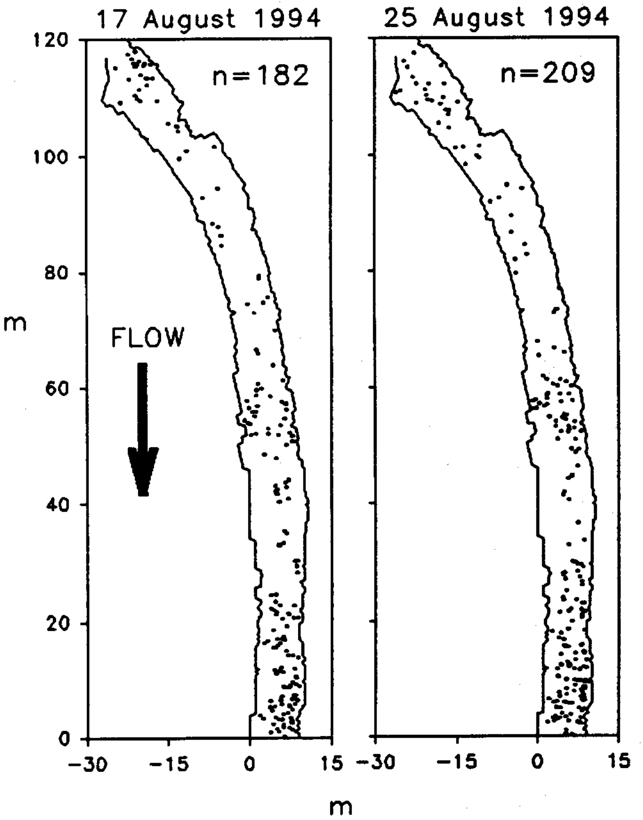


Figure 3. Distribution of 0+ Atlantic salmon, observed by snorkelling, on 17 August and 25 August, 1994, in North Harbour River, Newfoundland, Canada.

When applied to field data, the multiscale approach showed that the 0+ salmon distribution was not significantly different from random at small spatial scales (ambit radii < 0.2 m) (Figure 5). At larger spatial scales the distribution of 0+ salmon was clumped (RC>0). The RC reached a maximum at an ambit radius of 0.7 m (RC=0.43). The 0+ salmon were negatively associated with shallow depths (0-8 cm) and positively with intermediate depths (8-32 cm) at small spatial scales (RC=−0.66 and 0.19 for depth class 1 and 4 respectively, at an ambit radius of 40 cm, Figure 6). However, at large spatial scales (ambit radii > 4-6 m) the 0+ salmon were positively associated with shallow depths (RC=0.10 and 0.08 for depth class 1 and 2 respectively, at an ambit radius of 10 m). The associations were most different from random at small spatial scales. The randomized 0+ salmon distributions were not significantly associated with any of the depth classes (Figure 7).

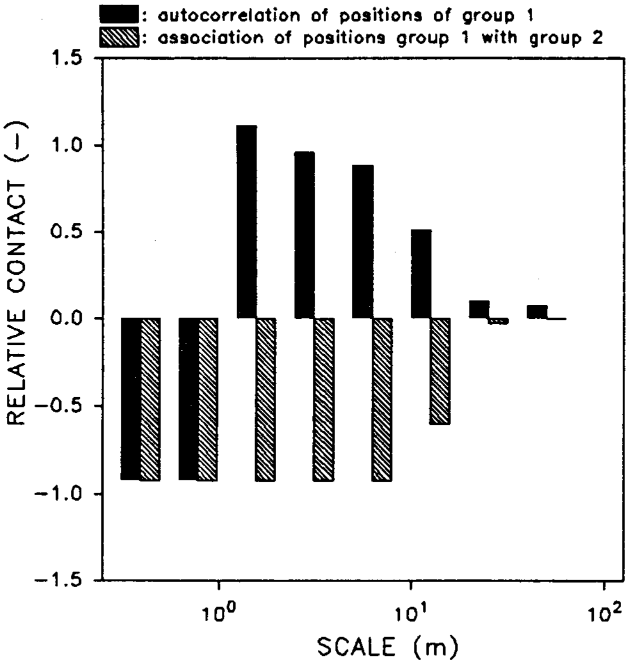


Figure 4. Spatial autocorrelation of group one fish positions (1) and spatial association of group one with group two fish positions (2) at increasingly larger ambit radii, quantified in terms of Relative Contact.

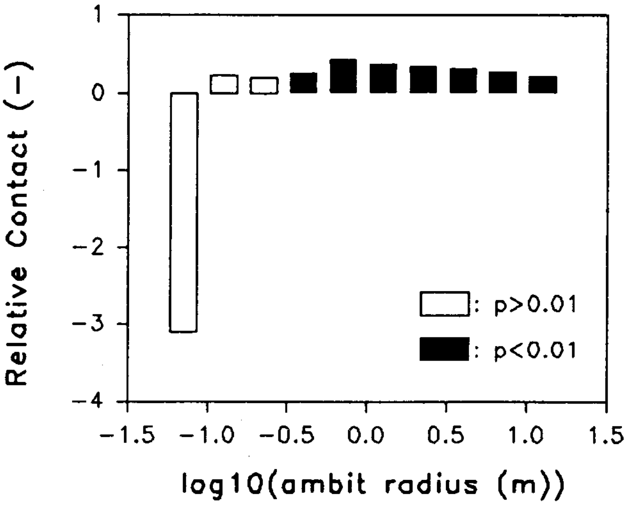


Figure 5. Spatial autocorrelation of the 0+ Atlantic salmon distributions as observed on 17 and 25 August, 1994, in North Harbour River, quantified in terms of Relative Contact at increasingly larger ambit radii.

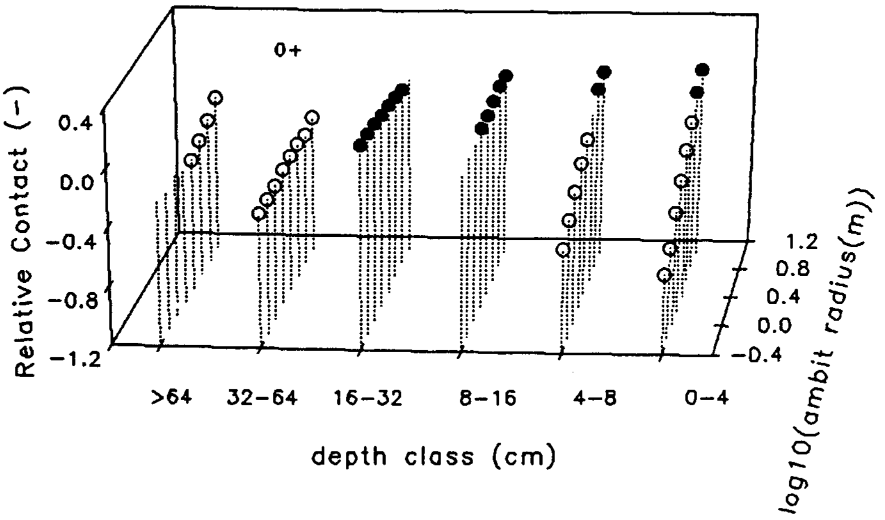


Figure 6. Spatial association of the 0+ juvenile salmon with depth as observed on 17 August and 25 August, 1994, in North Harbour River, Newfoundland, Canada. Spatial associations were quantified as the Relative Contact with a range of depth classes at increasingly larger ambit radii. Closed dots depict "significant" positive associations, open dots represent "significant" negative associations. A screening criterion of α=0.01 was used to separate "significant" from "non-significant".

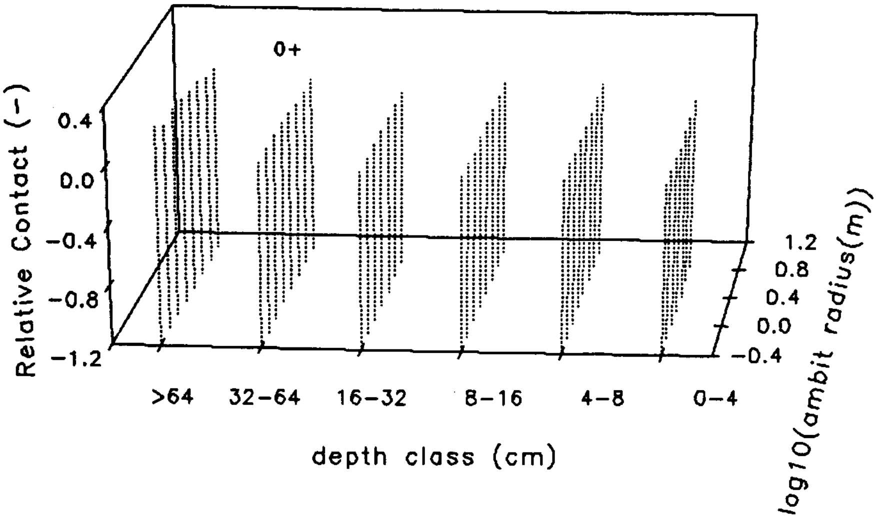


Figure 7. Spatial association of the randomized 0+ juvenile salmon with depth as observed on 17 August and 25 August, 1994, in North Harbour River, Newfoundland, Canada. Spatial associations were quantified as the Relative Contact with a range of depth classes at increasingly larger ambit radii. None of the Relative Contact estimates was "significantly" different from zero. A screening criterion of α=0.01 was used to separate "significant" from "non-significant".

DISCUSSION

Fish are associated with their environment over a range of spatial scales. This is because they react to their environment at a range of spatial and temporal scales, because of the propagation of effects from one scale to another and because of the interaction of biological and physical processes. The selection of salmonids for specific holding positions (small scale) with relatively low snout velocities in areas of high current conditions (larger scale) where drift is concentrated (Chapman and Bjornn 1969, Everest and Chapman 1972, Fausch and White 1981) is an example of selection behaviour operating at more than a single spatial scale. The effect of small-scale refuge locations (small spatial scale event) for rare flood events or ice scour (small temporal / large spatial scale events) for fish occupying a much larger area (large spatio-temporal scale effect) (Erman et al. 1988, Fausch and Bramblett 1991, Pearsons et al. 1992), and the process of expansion and contraction, where large scale distributions are influenced by small scale habitat selection processes (MacCall 1990), are examples of the propagation of effects across scales. The river continuum concept, where a continuous gradient of physical conditions influences gradients of organism distributions (Vannote et al. 1980, Statzner and Higler 1985), is an example of the interaction of biological and physical processes.

Because organisms are associated with their environment at a range of spatial and temporal scales, a comprehensive understanding of factors affecting the distribution and abundance of fish can only be achieved by studying factors affecting fish distributions at a range of scales, rather than one or even a few selected scales. In addition, multiscale approaches can potentially serve to integrate knowledge obtained from studies operating at a wide variety of spatial and temporal scales, such as micro-habitat studies, macro-habitat studies, stock-recruitment studies and movement studies.

The proposed multiscale technique offers several advantages over existing techniques. The technique can be used over any range of spatial scales in an environment with irregular boundaries. A theoretical distribution, which is needed for and greatly simplifies significance testing, is generated within the program. The technique can be used to analyze associations of fish with habitats as well as to test for patchiness. The program concept is simple. Results based on various distribution surveys can be assimilated into the analysis. The results in terms of Relative Contact are easy to visualize. The model is easy to adjust for transect data.

Disadvantages of the method are that the analyses require much computing time. The habitat mapping has to be done at evenly-spaced positions and is labour intensive. A grid system approach generally assumes that the distributional heterogeneity is isodiametric. In riverine habitats, however, the distributional heterogeneity is likely to be elongated in the flow direction. A transect approach would be amenable to test if patterns differ among transect orientations. However, the fish distribution in riverine habitats is inherently 2 dimensional rather than linear. In addition, it is important to note that the study reach has two types of boundaries: a real boundary by way of the shorelines and an imposed boundary by way of the start and end of the study reach. In the analyses it is assumed that the region outside the imposed boundaries has a spatial pattern similar to that of nearby areas within the boundaries. If this assumption cannot be met, the results should be limited to the areas that are located at a distance of the ambit radius of interest from the imposed boundaries. The problem of imposed system boundaries applies to any multiscale approach, such as spectral analysis, pattern analysis and second-order neighbourhood analysis.

We have shown that the proposed multiscale approach detects differences in patchiness of fish distributions and associations of fish with habitats at various spatial scales in simulated as well as in field data. Conclusions with respect to patchiness of distributions ranged from clumped (RC > 0) to random (RC = 0) or repulsed (RC < 0) (Figure 5). Conclusions with respect to habitat associations varied from positive to negative (Figure 6), indicating either preference or avoidance behaviour respectively.

The changes in patchiness of the 0+ salmon distributions across spatial scales were probably due to a combination of habitat selection for small scale environmental features, to small scale spacing behaviour and possibly to competitive interactions at small spatial scales (Figure 5). The 0+ salmon preferred intermediate depths and avoided shallow depths at small spatial scales, but preferred shallow depths at larger spatial scales (Figure 7). This pattern is probably due to a combination of avoidance of shallow depths at small spatial scales, preference for riffle areas that have a high number of shallow depth observations, and avoidance of pool habitats.

These results show that conclusions with respect to the distribution of juvenile salmon depend on scale. A micro-habitat approach would lead to the conclusion that 0+ salmon avoid shallow depths and are repulsed. A macro-habitat approach would lead to the conclusion that juvenile salmon prefer shallow depths and are clumped.

These results imply that the scale of measurement will determine the perceived relative importance of a habitat variable in habitat selection behaviour. Therefore, Habitat Suitability Indices and Habitat Use indices, commonly used in habitat modelling approaches, must also depend on scale. From this, it is obvious that managerial actions will differ based on the scale of measurement of the study used to support managerial decisions. The results also emphasize the fact that interpretation of results should be limited to the spatial scales over which the study was conducted.

A single scale approach in habitat modelling, being it either a "macro" or "micro" approach, fails to incorporate the complexity of habitat selection behaviour. Current habitat models could be improved by a more explicit use of scale. This would improve possibilities for assessing and prescribing habitat requirements of fish. Future habitat selection studies should focus on the identification of spatial scales that are most effective in explaining observed fish distributions, before extensive management studies are completed.

ACKNOWLEDGEMENTS

This study was supported via a National Science and Engineering Research Council (NSERC) grant to R.L. Haedrich and the Canadian Centre for Fisheries Innovations. A discussion with L. Zedel and the first author on autocorrelation techniques was the source of inspiration. The manuscript was greatly improved by the comments from M. Simpson, J. Horne, D. Methven and J. Ellis. We would like to thank D.A. Scruton of the Department of Fisheries and Oceans in St. John's, Newfoundland, for providing the field assistance of T. Bowdring and J. Maddock. In addition, we gratefully acknowledge the field assistance of the following people: T. Pittman, E. Trawbridge, L. Janes, J.K. Bult, J. Robinson, G. Crutcher, and M. Windsor.

REFERENCES

Chapman, D.W., and T.C. Bjornn (1969). Distribution of salmonids in streams, with special reference to food and feeding. In: *Symposium on Salmon and Trout in Streams*. T.G. Northcote (Ed.). H.R. McMillan lectures in fisheries. University of British Columbia. Vancouver. pp. 153-176.

DeGraaf, D.A., and L.H. Bain (1986). Habitat use by and preferences of juvenile Atlantic salmon in two Newfoundland rivers. *Transactions of the American Fisheries Society*. 115(5), pp. 671-681.

Downes, B.J., P.S. Lake, and E.S.G. Schreiber (1993). Spatial variation in the distribution of stream invertebrates: implications of patchiness for models of community organization. *Freshwater Biology*. 30, pp. 119-132.

Erman, D.C., E.D. Andrews, and M. Yoder-Williams (1988). Effects of winter floods on fishes in the Sierra Nevada. *Canadian Journal of Fisheries and Aquatic Sciences*. 45, pp. 2195-2200.

Everest, F.H., D.W. Chapman (1972). Habitat selection and spatial interaction by juvenile chinook salmon and steelhead trout in two Idaho streams. *Journal of the Fisheries Research Board of Canada*. 29, pp. 91-100.

Fasham, M.J. (1978). The application of some stochastic process to the study of plankton patchiness. In: *Spatial Pattern in Plankton Communities*. J.H. Steele (ed.). Plenum Press. New York. pp. 131-156.

Fausch, K.D., and R.J. White (1981). Competition between brook trout (Salvelinus fontinalis) and brown trout (Salmo trutta) for positions in a Michigan stream. *Canadian Journal of Fisheries and Aquatic Sciences*. 38, pp. 1220-1227.

Fausch, K.D., and R.G. Bramblett (1991). Disturbance and fish communities in intermittent tributaries of a western great plains river. *Copeia*. 1991, pp. 659-674.

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley (1986). A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management*. 10(2), pp. 199-214.

Getis, A., and J. Franklin (1987). Second-order neighborhood analysis of mapped point patterns. *Ecology*. 68(3), pp. 473-477.

Greig-Smith, P. (1983). *Quantitative Plant Ecology*. Third edition. Blackwell Scientific Publications. Oxford.

Heggenes, J. (1990). Habitat utilization and preferences in juvenile Atlantic salmon (Salmo salar) instreams. *Regulated Rivers: Research and Management*. 5, pp. 341-354.

Holling, C.S. (1992). Cross-scale morphology, geometry, and dynamics of ecosystems. *Ecological Monographs*. 62, pp. 447-502.

Horne, J.K. (1994). Spatial variance of capelin (Mallotus vilosus) in coastal Newfoundland waters. *Journal of the Northwest Atlantic Fisheries Science*. 17, pp. 33-43.

Horne, J.K., and D.C. Schneider (1995). Spatial variance in ecology. *Oikos*. 74(1), pp. 18-26.

Legendre, P., and M.J. Fortin (1989). Spatial pattern and ecological analysis. *Vegetatio*. 80, pp. 107-138.

Levin, S.A. (1992). The problem of pattern and scale in ecology. *Ecology*. 73(6), pp. 1943-1967.

Lloyd, M. (1967). Mean crowding. *Journal of Animal Ecology*. 36, pp. 1-30.

MacCall, A.D. (1990). *Dynamic Geography of Marine Populations*. University of Washington Press. New Yersey.

Menge, B.A. and Olson, A.M. (1990). Role of scale and environmental factors in regulation of community structure. *Trends in Ecology and Evolution*. 5, pp. 52-57.

Minshall, G.W. (1988). Stream ecosystem theory: a global perspective. *Journal of the North American Benthological Society*. 7(4), pp. 263-288.

Morisita, M. (1959). Measuring of the dispersion of individuals and analysis of the distributional patterns. *Memoirs of the Faculty of Science, Kyushu Univ. (E)*. 2, pp. 215-235.

Pearsons, T.N., H.W. Li, and G.A. Lamberti (1992). Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. *Transactions of the American Fisheries Society*. 121, pp. 427-436.

Platt, T. and Denman, K.L. (1975). Spectral analysis in ecology. *Annual Review of Ecology and Systematics*. 6, pp. 189-210.

Press, W.H., B.P. Flannery, S.A. Teukolsky, and W.T. Vetterling (1986). *Numerical Recipes in FORTRAN: The Art of Scientific Computing*. Cambridge University Press. Cambridge.

Ripley, B.D. (1981). *Spatial Statistics*. John Wiley. New York.

Schneider, D.C. (1994). *Quantitative Ecology. Spatial and Temporal Scaling*. Academic Press. San Diego. California.

Statzner, B., and B. Higler (1985). Questions and comments on the river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*. 42, pp. 1038-1044.

Taylor, L.R. (1961). Aggregation, variance and the mean. *Nature*. 189, pp. 732-735.

Upton, G. and B. Fingleton (1985). *Spatial Data Analysis by Example. Volume I. Point Pattern and Quantitative Data*. John Wiley and Sons. New York.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing (1980). The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences*. 37, pp. 130-137.

Weber, L.H., S.Z. El-Sayed, and I. Hampton (1986). The variance spectra of phytoplankton, krill and water temperature in the Antarctic ocean south of Africa. *Deep Sea Research*. 33(10), pp. 1327-1343.

Wiens, J.A. (1989). Spatial scaling in ecology. *Functional Ecology*. 3, pp. 385-397.

EFFECT OF HYDRAULIC VARIABLES ON MIGRATORY BEHAVIOUR OF ATLANTIC SALMON (*Salmo salar*) IN THE MOISIE RIVER, QUÉBEC

Frédéric Lévesque
Groupe-conseil Génivar inc.
5355, boul. des Gradins, Québec (Québec) G2J 1C8
E.mail : flevesque@genivar.com

Marcel Proulx
Groupe-conseil Génivar inc.
5355, boul. des Gradins, Québec (Québec) G2J 1C8
E.mail : mproulx@genivar.com

and

Geneviève Corfa
Vice-présidence Ingénierie et Services
Hydro-Québec
855, rue Sainte-Catherine Est, 12e étage, Montréal (Québec) H2L 4P5

ABSTRACT

Atlantic salmon (*Salmo salar* L.) migration was studied from 1992 to 1995 by echosounding and underwater video-camera in the Moisie River. Hydraulicity in the area is characterized by fast water, laminar flow and a mean water discharge estimate of 431 $m^3 \cdot s^{-1}$ (min-max = 166 - 1095 $m^3 \cdot s^{-1}$) for the sampling period. When flow was low, Atlantic salmon rapidly progressed upstream and the peak of the run was stronger and occurred earlier in the season. When flow was high, the migration slowed down, the peak was weaker and the run was spread out through the season. In these conditions, salmon stayed for a longer period of time below major obstacles. Fish travelled closer to the banks when flow was high, and stayed farther away when low. Summer floods stopped fish migration but it proceeded again as soon as the highest level was receding. Flow velocity altered migratory routes used by salmon as well as it's behaviour when moving upstream. Salmon travelled close to the shore in areas where water velocity was usually lower than 1 $m \cdot s^{-1}$, and swam close to the river bed at a mean height ranging from 15,5 cm to 35,9 cm depending on the discharge. Mean speed recorded during the study ranged from 0,25 $m \cdot s^{-1}$ to 0,64 $m \cdot s^{-1}$. Results demonstrated that salmon kept its cruse speed and remained near the bottom where water velocities were slower when the river water flow are high. Swimming speed increased as water temperature went up. This study brings new information on Atlantic salmon behaviour according to flow management.

INTRODUCTION

Water flow and water temperature are environmental variables that were often reported to influence the migration of adult Atlantic salmon *(Salmo salar)* (Hayes, 1953; Banks, 1969; Elson, 1969; Alabaster, 1970; Hellawell, 1976; Jensen *et al.*, 1989). Change in water flow may regulate fish movement, migratory speed and direction of migration while water temperature is an important factor initiating up or downstream migration of several fish species (Jonsson, 1991).

Upstream migration of Atlantic salmon has been studied in many ways. Most of these studies were based upon direct observation of major obstacle like waterfall or fish ladder and upon indirect observation made between two points in the river course. Data was obtained from the presence or absence of fish from one pool to another, by using a counting fence (Alabaster, 1970; Webb and Hawkins, 1989), a resistivity fish-counter (Hellawell *et al.*, 1974; Elson and Tuomi, 1975; Cragg-Hine, 1985) or telemetric survey (Webb and Hawkins, 1989; Tremblay, 1991). It was stated that relationship based on counting passages over falls or counting-fences introduces uncertainty as to whether the observed pattern is influenced by the obstruction itself and if a different pattern might not prevail in its absence (Hellawell, 1976). Few study were based upon direct observation in stream section were fish move freely and swim naturally.

In this paper, we studied the relationship between the Atlantic salmon run and the water flow and water temperature in a channel section of the Moisie River between 1992 and 1995. The main objective is to analyse direct observation on the behaviour of upstream fish during the migration season through the use of hydroacoustic and underwater camera data.

SAMPLING SITE

The sampling site is located (50°28'N - 68°08'W) 48,6 km from the mouth of the Moisie River (Figure 1). The river course in this area is 145 m wide and is characterised by a laminar flow. Mean depth at high level of water is around 3 m and flow velocity is about 1,34 to 1,61 $m \cdot s^{-1}$. The site has a single channel sloping uniformly from each bank with a hard bottom mainly composed of cobble stones and pebbles with a small fraction of gravel. Because there is no sport fishing in this area, boat traffic is low and could not likely affect fish behaviour. The location is also downstream of observed spawning sites for Atlantic salmon (Belzile, 1994), which reduces the probability that individual fish are turning around up and down or demonstrating low activity when they move through the sampling zone.

DATA COLLECTION

Water flow was obtained from a level-flow curve transformation of level-gauge measurements collected by the Ministère de l'Environnement et de la Faune 15,5 km downstream of the sampling site. There is no major tributaries between the level-gauge station and the sampling site. Since the water moved at a mean speed of 5,3 $km \cdot h^{-1}$, there is a slight delay of about 3 hrs between the sampling site and the limnimetric station. Real time water flow can be estimated with water level data from the sampling site by the following preliminary equation :

$$Q = 1.697\ (H + 3.96)^{3.3757} \quad (r^2 = 0{,}993) \tag{1}$$

where Q is flow ($m^3 \cdot s^{-1}$) and H is water level (m). It should be used with caution since preliminary observations showed that it slightly overestimated water flow when lower than 250 $m^3 \cdot s^{-1}$. Water level was measured every morning around 07:30 hrs with a water level-gauge set in front of the camp. Water temperature was also taken at the same time every day with a minimum-maximum thermometer fixed on the end of the left shore fence.

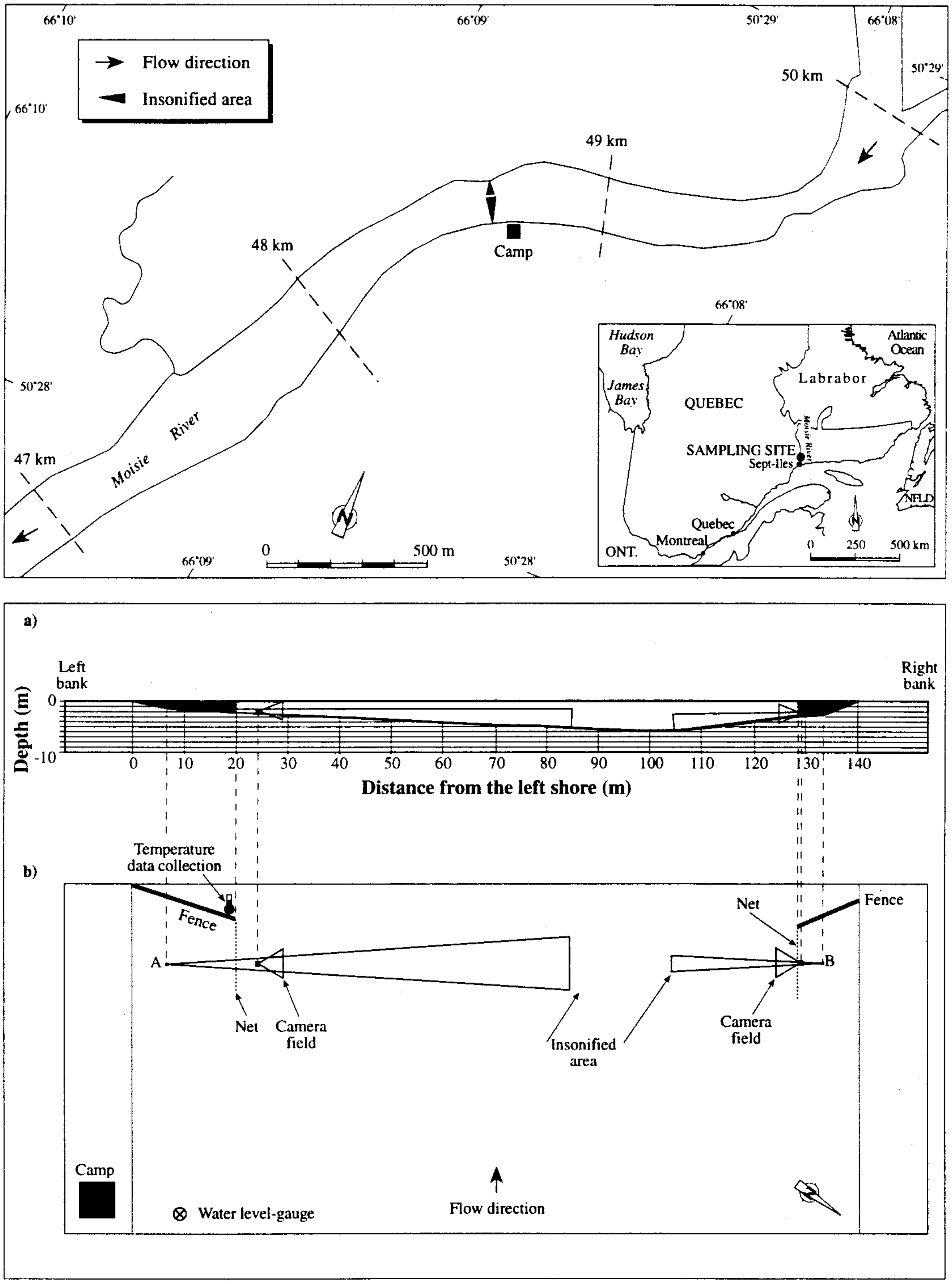


Figure 1. Site location (upper panel) and sampling design (lower panel) of the Atlantic Salmon run study on the Moisie River.

Flow velocity was investigated every 10 m through the total width of the river course on August 19, 1994. A Teledyne No 622-E currentmeter was used with an hand-winch and a streamlined weight to measure river current from a boat.

The number of fish, their distance from shore and their diel distribution where obtained daily from echosounding. A fixed, sampling location designed with side oriented transducer on each side of the river, were used to collect data (Figure 1). Echosounding was done with a dual beam echosounder model ES 2000 from BioSonics. Two side-by-side transducers set close to both banks made it possible to get information on fish direction. The echosounder worked 24 hrs each day from mid-June until the end of August (Table 1). Based on results from 1992 to 1994 population size estimates show that nearly all tracked fish where Atlantic salmon (Lévesque and Harte, 1994; Lévesque and Proulx, 1993; Proulx and Lévesque, 1994; Proulx *et al.*, 1995).

Table 1: Hydroacoustic and underwater video camera sampling schedule on the Moisie River (1992-1995).

	Sampling period	
Year	**Hydroacoustic**	**Video camera**
1992	06/15 - 09/06	07/08 - 07/18
1993	06-12 - 09/07	07/31 - 09/06
1994	06/22 - 08/18	07/01 - 08/17
1995	06/10 - 08/18	06/18 - 08/18

Height of fish over the river bed and swimming speed could be estimated from underwater camera data. Because the water is of a yellow-brown color and often turbid, sampling was conducted only in daytime even if the cameras were very sensitive. Cameras, one Hitachi (0,003 lux) and one Panasonic (0,02 lux), operated between 05:00 hr through 21:00 hr. Sampling periods are given in table 1. Images were recorded on videotape and analysed later through the use of a Panasonic time laps tape recorder model AG-6040.

FLOW VELOCITY

Atlantic salmon travelling upstream swims naturally close to the bottom of the Moisie River (Figure 2). Mean distance over the river bed range between 15,5 cm and 35,9 cm during the study (Table 2). Results from previous studies also show that most fish stays close to the bottom, in a range well below mean velocity (Hellawell *et al.*, 1974).

Average of swimming depth has been drawn on the flow velocity curve to find which water speed Atlantic salmon prefers when travelling upstream (Figure 3). Swimming depth was estimated with the geometric mean because fish distribution over the river bed was positively skewed (Figure 2). Flow velocity data was collected at a high discharge rate (593 $m^3 \cdot s^{-1}$) to insure that observed values were sufficiently critical to give an appropriate estimate for this parameter. Only fish recorded on videotape when waterflow was over 605 $m^3 \cdot s^{-1}$, the observed value at the main station (km 33,1), are included in the analysis. Range of flow velocity within which salmon travels fall between 0 and 0,8 $m \cdot s^{-1}$, well below the mean velocity at 0,6 H (Figure 3).

Water speed measured near the surface (0,2 H) and near the bottom (0,8 H) at the end limit on both side of the river where fish was detected, in 1994, averaged 1,56 $m \cdot s^{-1}$ and 1,07 $m \cdot s^{-1}$ respectively. Water flow was over 605 $m^3 \cdot s^{-1}$ most of the time during the salmon run of 1994. These observations suggest that fish is moving over the river bed in area where current is generally lower than 1 $m \cdot s^{-1}$. Webb and Hawkins (1989) reported that Atlantic salmon tended to avoid the areas where velocity is greater than 1 $m \cdot s^{-1}$ or traversed them by a burst swim.

Table 2: Depth, swimming speed and shore distance of fish as seen on video and detected on sonar in the Moisie River, and results from Kruskal-Wallis test (1992 to 1995)

Year	Distance over the river bed (cm)	Swimming speed ($m \cdot s^{-1}$)	Distance from the shore (m)		Mean flow ($m^3 \cdot s^{-1}$)[1]	Mean water temperature (°C)[2]
			Left	Right		
1992	28,3 ± 17,5 ^{a} (n = 55)	0,640 ± 0,200 ^{a} (n = 50)	n/a	n/a	596,0 ± 212,6 ^{a} (n = 71)	13,3 ± 1,8 ^{a} (n = 70)
1993	23,9 ± 13,6 ^{a} (n = 22)	0,509 ± 0,266^{ab} (n = 24)	38,6 ± 10,1 ^{a} (n = 391)	17,8 ± 2,3 ^{a} (n = 1335)	388,2 ± 79,7 ^{b} (n = 90)	15,9 ± 2,0 ^{b} (n = 70)
1994	15,5 ± 9,2 ^{b} (n = 259)	0,452 ± 0,144^{bc} (n = 249)	29,5 ± 7,8 ^{b} (n = 80)	11,0 ± 2,8 ^{b} (n = 760	684,0 ± 191,4 ^{c} (n = 90)	14,9 ± 2,4 ^{c} (n = 67)
1995	35,9 ± 19,4 ^{c} (n = 144)	0,248 ± 0,123 ^{d} (n = 100)	51,3 ± 14,8 ^{c} (n = 737)	20,0 ± 4,9 ^{c} (n = 1022)	284,9 ± 134,0 ^{d} (n = 90)	17,2 ± 2,7 ^{d} (n = 70)

Note: Statistics are the mean plus or minus the standard deviation; values with the same letter are not significantly different (p = 0,05).

[1] 06/10 - 09/07

[2] 09/10 - 19/08

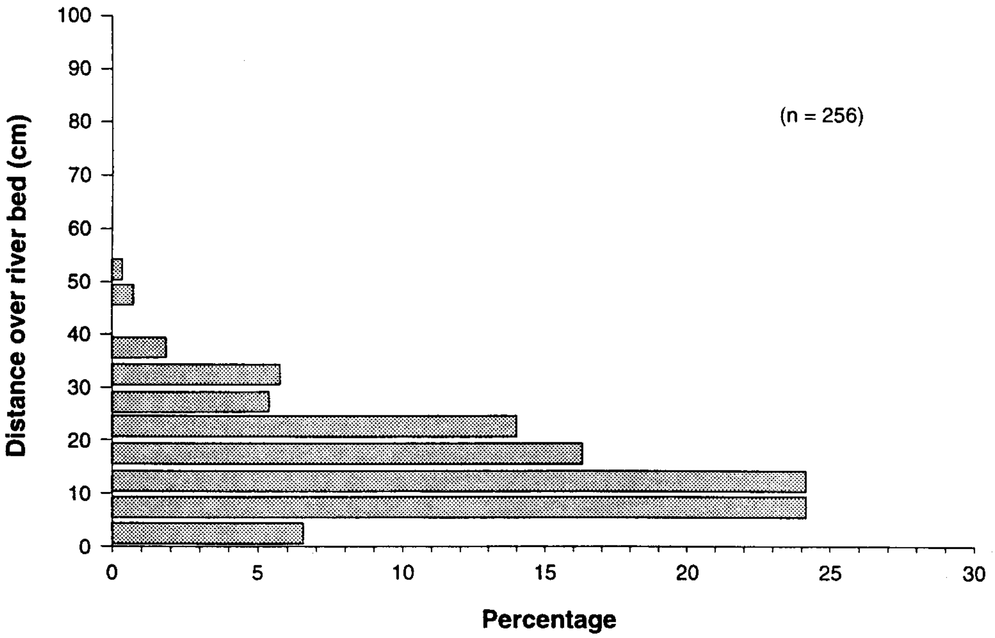


Figure 2. Swimming distance over the river bed of Atlantic Salmon travelling upstream in the Moisie River (1994).

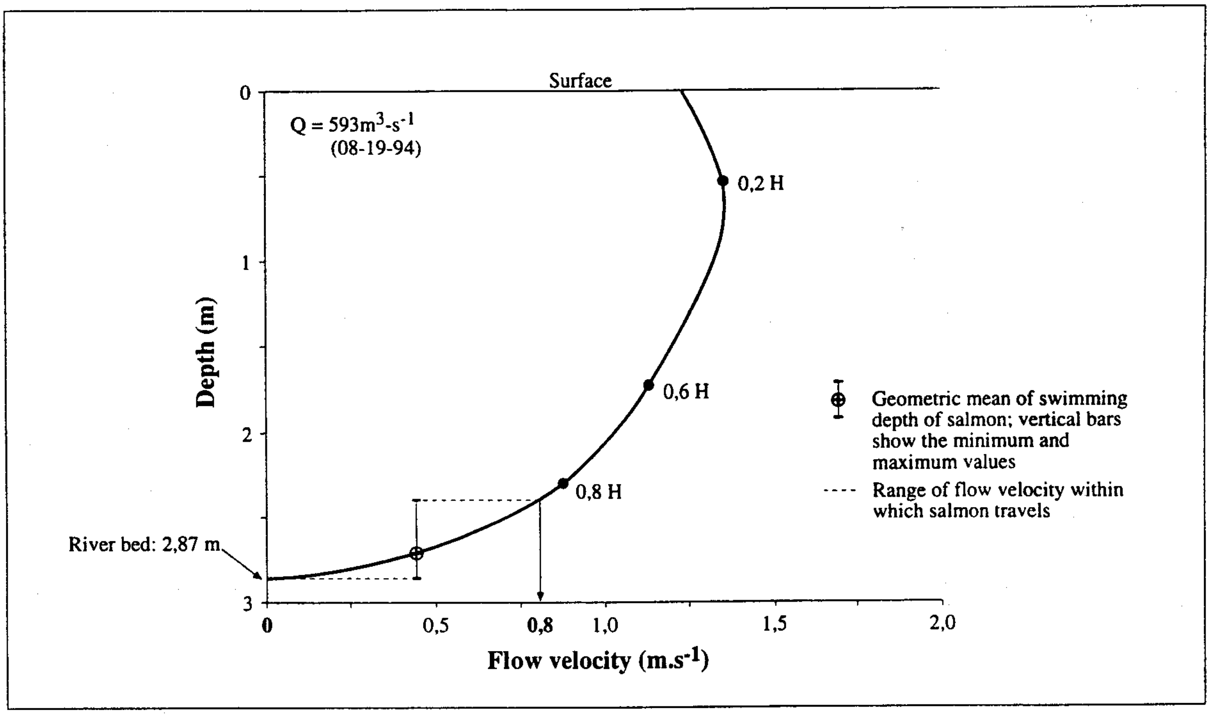


Figure 3. Theoric outline of flow velocity measured at 0,2 H, 0,6 H and 0,8 H from the surface in the major fish passage of the Moisie River.

WATER FLOW

Fish run

Water flow was significantly different ($p < 0,05$) from one season to another (Table 2). In 1992, water level was highly variable, from very high to very low in a short period of time (Figure 4). The water level variation has much less range in the following season. The water level was very high in 1994 and very low in 1995.

Overall, the migratory run is taking place generally in the same manner every year. It starts slowly in June, peaks at an optimal level in July and gradually falls down in August to end around the first week of September. Groups of fish enter into the sample area in successive waves of variable amplitude. Observation shows that fish begins to be detected at the sampling site when water flow is under 600 $m^3 \cdot s^{-1}$. A major downstream obstacle, a large, long and very rough rapid located between the kilometres 25 and 33, appears to regulate the period of fish entry into the study area, although the migratory run can be also controlled by photoperiod and water temperature. Data from sport fishing support this observation as catches increase just down below large rapids when water discharge is high.

Interseason differences of water flow influence the migratory run schedule. The high water level observed at the beginning of 1992 and 1994 campaign somewhat delayed the run. In 1995, migration was concentrated at the beginning of July since water level was very low as it was in the remaining sampling season. Two thirds (67 %)

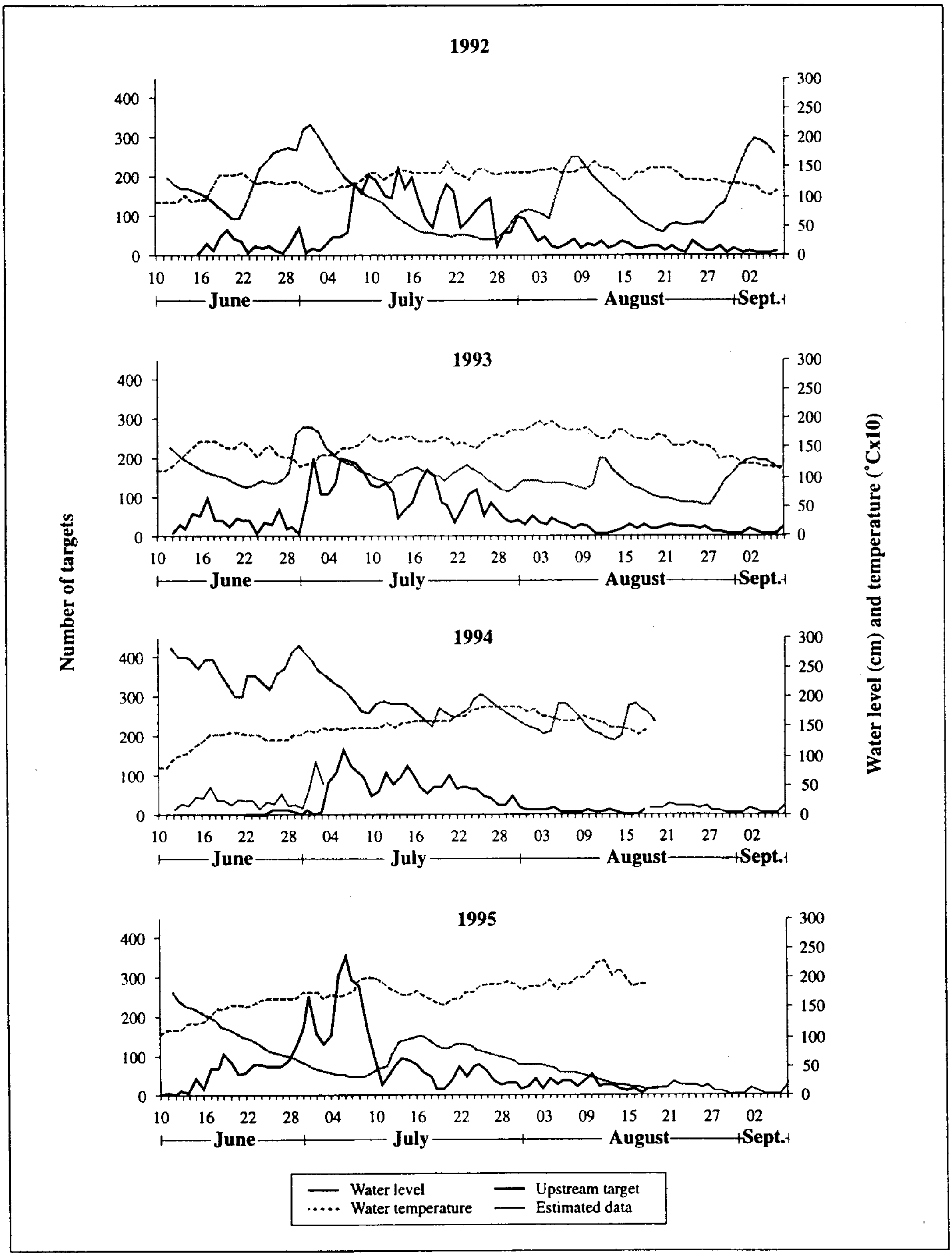


Figure 4. Migratory run timing of Moisie River's Atlantic Salmon as detected with hydroacoustic technique (1992 to 1995).

of the fish migrated upstream before the 10th of July in that season, while only 25 % (1992), 28 % (1994) and 46 % (1993) of the migration was complete at the same time in the previous years. The 1993 fish migration is considered to show normal pattern of the run timing.

Intra-season fluctuation of water flow altered events which characterised the salmon run. Atlantic salmon stops its migration when water level raises abruptly and starts again when it stabilizes or begins to lower. This phenomena has already been observed in many water courses (Jonsson, 1991). When the water raises flow velocity increases and drops where the water level reaches it highest point. Steep increases of flow slows down the upstream migration since water velocity is too high. It migration starts again as soon as the flow velocity comes into the range in which Atlantic salmon prefers to travel.

Horizontal distribution

Mean distance from both shore to which salmon migrates was significantly different between each year (Table 2). Salmon traveled near the shore where the flow velocity was generally slower (Figure 5). When water flow was high, as in 1994, salmon concentrated closer to the shore whereas they tented to be much more spread out toward the center of the river when it was low, as, for example, in 1995. Similar horizontal distribution of Pacific salmon species in relationship to water flow and velocity has already been reported before (Gaudet, 1990).

Horizontal distribution in 1992 was directed closer to the main channel than in 1993. This was observed even if mean water flow was higher in 1992 when compared to 1993, since fish were mainly detected during periods where water level was low (Figure 4).

Diel distribution

Water flow also influences the diel distribution (Figure 6). Salmon moves upstream mainly by night and in the morning, with major peaks observed at dawn and dusk. This behaviour is particularly shown in 1993, which was previously assumed to be a normal run. It is similar to the 1992 diel distribution, even if the mean water flow was higher. In 1994, fish travelled mostly during daytime since discharge was high and water unusually turbid. Very few fish were tracked in the afternoon of 1995 as water flow was abnormally low. These observations lead us to conclude that most upstream movement occurred during the hours of darkness when water is low and clear. However, during floods and turbid water periods, night time patterns are replaced in favour of a greater movement during the hours of daylight, as suggest by Hellawell *et al.* (1974).

WATER TEMPERATURE

Mean water temperature was significantly different between the years of the study (Table 2). Temperature profiles and run timing patterns of salmon migration did not show any clear link (Figure 4). It seems that there is no relationship between the migratory waves, peak or ditch, with the increase or decrease of water temperature. However, the first small peak of the run in June always occurred when water temperature increases rapidly from about 10°C to 14°C. Hellawell *et al.* (1974) stressed that there was no optimal temperature associated with movement in his study.

HYDRAULIC VARIABLES AND FISH BEHAVIOUR

Depth, swimming speed and distance from the shore during migration of the Atlantic salmon in the Moisie River vary between years, according to water flow and water temperature (Table 2). Mean depth and mean distance from the shore to which fish were swimming show an inverse relationship with water flow ($p < 0,05$). Moreover, swimming speed also appears to be inversely correlated with water temperature.

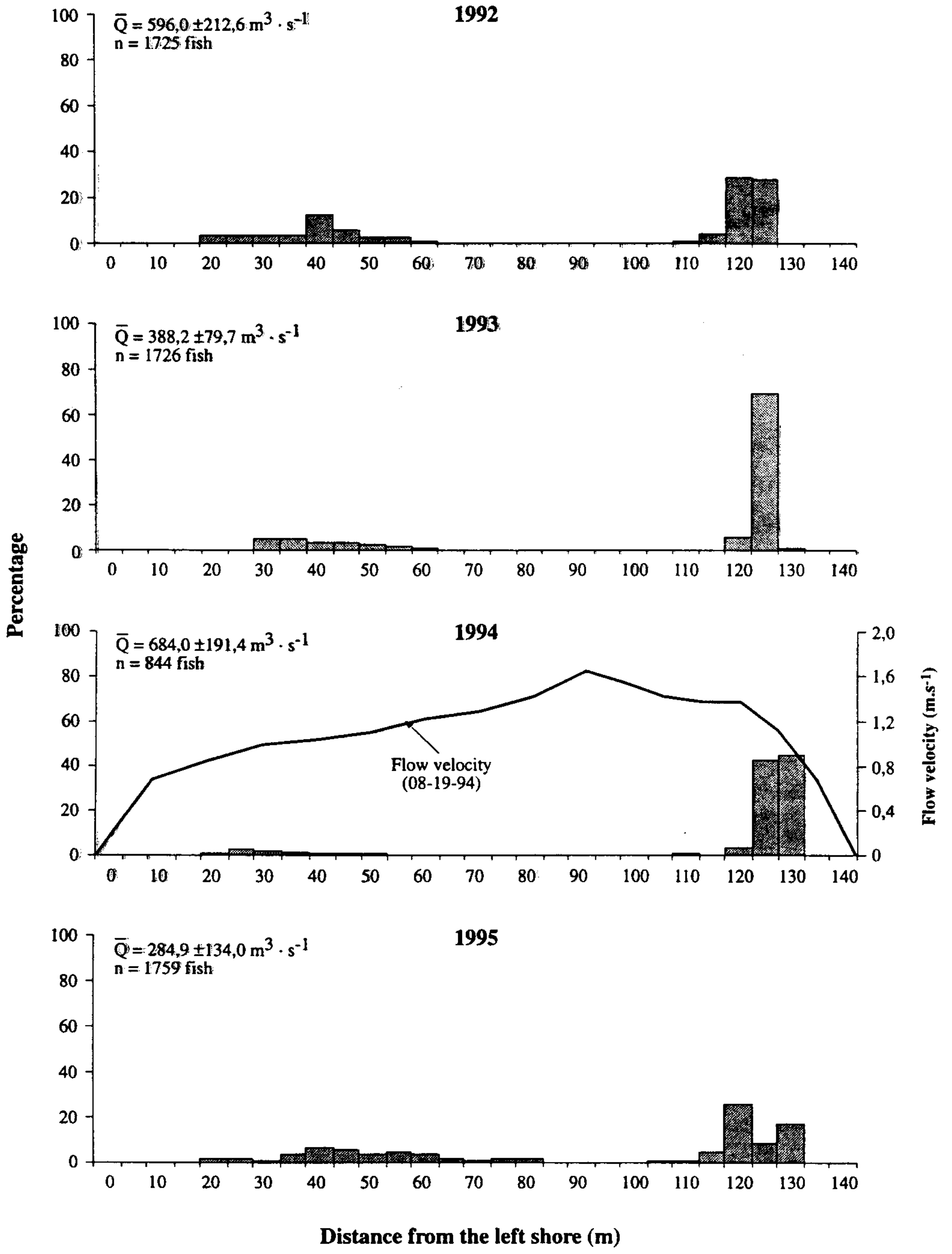


Figure 5. Horizontal distributions of fish tracked upstream, Moisie River (1992 to 1995).

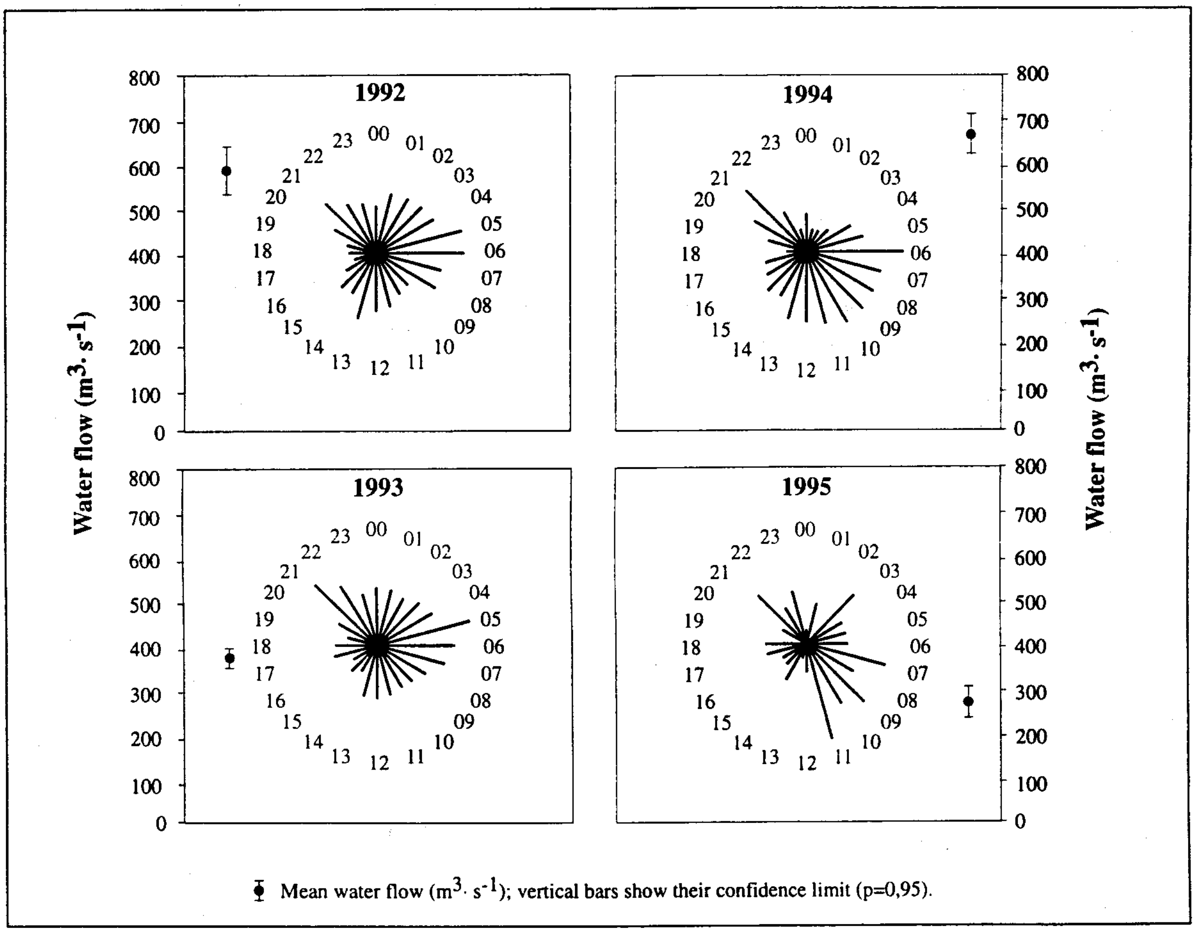


Figure 6. Diel distributions of fish detected upstream in the Moisie River (1992 to 1995).

Physical and biological data collected in the Moisie River (temperature, mean daily flows, mean daily distance of fish from both shores, daily number, depth and swimming speed of fish ascending the river) have been examined by principal component analysis. This statistical technique allows to determine statistically significant factors of variation in the data set and to provides correlations (Pearson's) between basic variables and factors, each explaining a part of the total variability of the data. Thus, basic variables can be ranked in order of importance depending on which factor it is correlated to. Data matrix is centered and reduced before analysis, allowing variation of different types of data (count, ratio and continuous scale) to be rigorously compared and correlated. Varimax procedures have been applied in order to maximize the fitting of the principal components to the major variability axis of the data.

The first factor of variation explained 44,2% of the total variation of the data set and is directly correlated with the distance of migrating salmon from the shore (left side: 0.89; rigth side: 0.78) and inversely correlated with mean daily flow (-0.81) (Table 3). Therefore, as the flow rate raises, the salmon swims closer to the shore (Figure 7).

Table 3. Varimax rotated factor matrix of correlation coefficient from the principal component analysis.

	Factor			
Variable	1	2	3	4
Temperature	0,40110	0,89954	0,11895	0,03372
Flow rate	-0,80509	-0,31813	-0,21568	-0,27160
Leftshore distance	0,88966	0,00964	0,16800	0,07227
Rightshore distance	0,77796	-0,11096	-0,20819	0,37142
Daily number	0,06001	-0,07219	0,93461	0,20219
Height	0,30517	0,09539	0,25573	0,89007
Speed	-0,40204	0,79278	-0,37672	0,09125
Percentage of explained variance	44,2*	23,0*	14,2	8,6

* Broken stick-test; significant (Legendre and Legendre, 1984).

The second major axis of variation is composed of the correlation with temperature (0.90) and swimming speed (0.79), which indicates that salmon increases it's speed of migration as the temperature raises. This factor explained 23.0% of the total variation of the data set. It can be explain as a fact that metabolism of fish rise with an increase of water temperature.

The third and fourth factors illustrates a better trend of daily number and height from the bottom respectively for each particular year of sampling (table 3). However, they cannot be interpreted as well since they do not reach the minimum level setted by the broken stick test (Legendre and Legendre, 1984). At this level, variations are only randomly generated. The fact that daily number of salmon is not correlated with daily flow is not surprising since the number of ascending salmon in a river in a year depends upon the production conditions in the river and the conditions encountered in the ocean a few years before.

Daily variations in the fish passage were not correlated with changes in discharge or temperature, as previously report by Hellawell (1976). We have seen before that steep increase of flow reduced migratory activity, but these events do not happen very often in the season. However, increase of temperature stimulate fish migration. Mean daily progress of fish carrying a radio-transmitter was significantly correlated with the mean water temperature between the fish location survied in previous study on the Moisie River, but it only explained a small part of the variance (Tremblay, 1991).

Our results show a major single mode occurring on July in the pattern of movement that is quite consistent in their timing from year to year (figure 4). Overall, our study supports the previous findings (Hellawell et al., 1974; Hellawell, 1976; Cragg-Hine, 1985; Jensen et al., 1989) which let us concludes that the primary factor in Atlantic salmon migration is established by a temporal function. The second factor in importance is the discharge and the third one is the temperature. Incident light might have direct link with fish movement but this environmental variable is probably more influenced by flood as an autocorrelated variable.

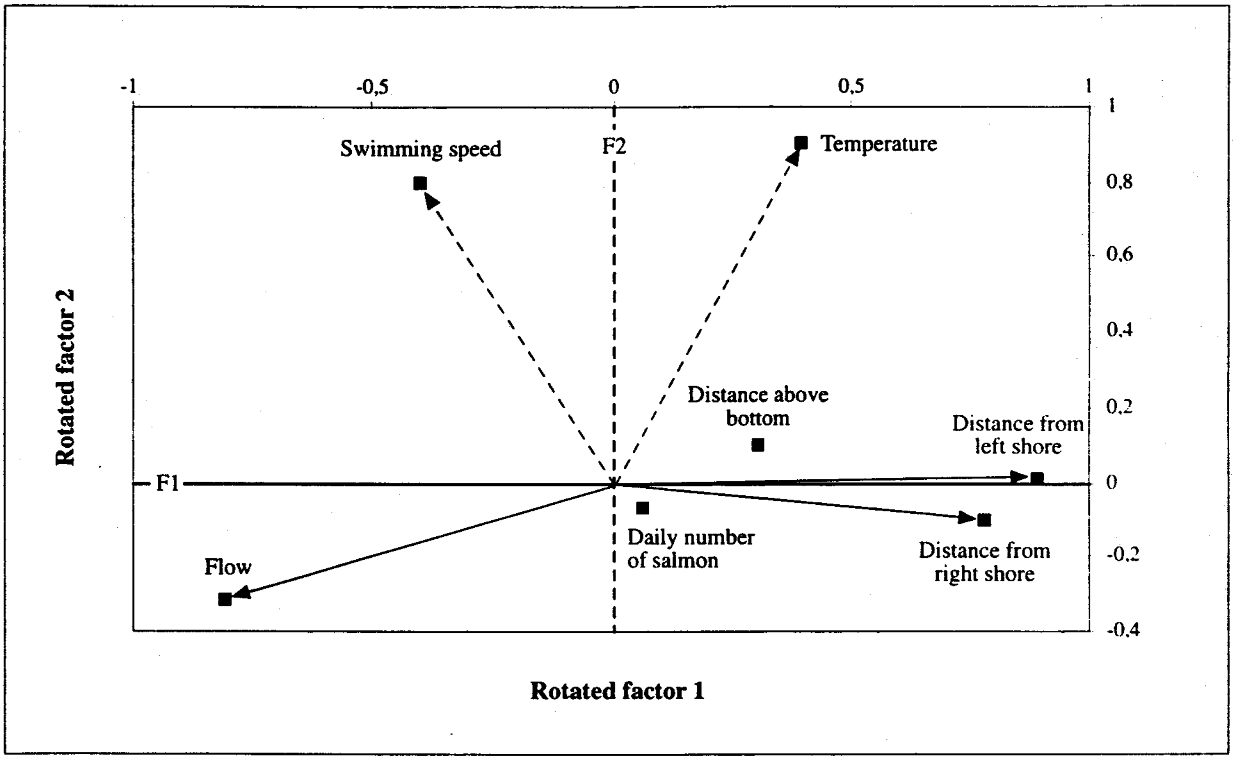


Figure 7. Result of the principal component analysis illustrated by vector projection of hydraulic and biogical variables.

ACKNOWLEDGEMENTS

This study was made possible thanks to the financial support of Hydro-Québec. The authors gratefully acknowledge their colleagues at Groupe-conseil Génivar inc. for their assistance, advice and comments on the first manuscript. We especially wish to address our sincere thanks to Gilles Tremblay who is responsible for the biometrical analysis.

REFERENCES

Alabaster, J.S. 1970. River flow and upstream movement and catch of migratory salmonids. J. Fish. Biol. 2:1-13.

Banks, W. 1969. A review of the literature on the upstream migration of adult salmonids. J. Fish. Biol. 1:85-136.

Belzile, L. 1994. Étude des frayères à saumon dans la rivière Moisie. Rapport des campagnes de 1988, 1989 et 1993. Présenté à la vice-présidence Environnement, Hydro-Québec, par le Groupe Environnement Shooner inc., 46 p. et annexes.

Cragg-Hine. D. 1985. The assessment of the flow requirements for upstream migration of salmonids in some rivers of north-west England. p. 209-215. *In* J. Alabaster (ed.). Habitat Modification and Freshwater Fisheries. Butterworth, London.

Elson, P.F. 1969. High temperature and river ascent by Atlantic salmon Int. Council Explor. Sea. C.M. 1969/M 12:1-9.

Elson, P.F. and A.L.W. Tuomi. 1975. The Foyle fisheries : New basis for rational management. LM Press Ltd, Lurgan N., Ireland 224 p.

Gaudet, D.M. 1990. Enumeration of migrating salmon populations using fixed-location sonar counters. Rapp. P.-v. Réun. Cons. int. Explor. Mer, 189:197-209.

Hayes, F.R. 1953. Artificial freshets and other factors controlling the ascent and population of Atlantic salmon in the La Have River, Nova Scotia. Bull. Fish. Res. Board Can. 99: 47 p.

Hellawell, J.M. 1976. River management and the migratory behaviour of salmonids. Fisheries Management 7:57-60.

Hellawell, J.M., H. Leatham and G.I. Williams. 1974. The upstream migratory behaviour of salmonids in the River Frome, Dorset. J. Fish. Biol. 6:729-744.

Jensen, A.J., B.O. Johnsen and L.P. Hansen. 1989. Effect of river flow and water temperature on the upstream migration of adults Atlantic salmon *Salmo salar* L. in the River Vefsna, northern Norway, p. 140-146. *In* E. Brannon and B. Jonsson (Eds). Salmonid migration and distribution symposium. Univ. Washington, Seattle.

Jonsson, N. 1991. Influence of water flow, water temperature and light on fish migration rivers. Nord. J. Freshwat. Res. 66:20-35.

Legendre, L. and P. Legendre. 1984. Écologie numérique. Tome 2. La structure des données écologiques. Masson et P.U.L. 254 p.

Lévesque, F. and M. Proulx. 1993. Étude de la montaison du saumon atlantique (*Salmo salar*) par méthode hydroacoustique dans la rivière Moisie: rapport des opérations 1990, 1991 et 1992. Présenté à la vice-présidence Environnement, Hydro-Québec, par le Groupe Environnement Shooner inc. 73 p. et annexes.

Lévesque, F. and M.K. Harte. 1994. Practical application of hydroacoustic sensing in rivers : The case-study of the Atlantic salmon (*Salmo salar* L.) spawning run as investigated by dual-beam echosounding in the Moisie River, Quebec. Bull. Fr. Pêche Pisc. 332:33-41.

Proulx, M. and F. Lévesque. 1994. Étude de la montaison du saumon atlantique (*Salmo salar*) par méthode hydroacoustique dans la rivière Moisie en 1993. Rapport présenté à la vice-présidence Environnement, Hydro-Québec, par le Groupe Environnement Shooner inc. 56 p. et annexes.

Proulx, M., F., Lévesque and L. Belzile. 1995. Évaluation de la montaison du saumon atlantique (*Salmo salar*) par méthode hydroacoustique et capture-marquage-recapture dans la rivière Moisie en 1994. Rapport présenté à la Vice-présidence Environnement, Hydro-Québec par le Groupe Environnement Shooner, 106 p. et annexes.

Tremblay, G. 1991. Projet Sainte-Marguerite. Suivi par télémétrie de quarante-trois saumons sur la rivière Moisie en 1990. Rapport complémentaire à l'étude synthèse présenté à la Vice-présidence Environnement d'Hydro-Québec. 42 p. et annexes.

Webb, J. and A.D. Hawkins. 1989. The movement and spawning behaviour of adult salmon in the Girnock Burn, a tributary of the Aberdeenshire Dee, 1986. Scot. Fish. Res. Rep. 40:1-41.

A PERSPECTIVE TO PREDICT HABITAT SUITABILITY OF LOTIC FISH: LINKING STATISTICAL HYDRAULIC WITH MULTIVARIATE HABITAT USE MODELS.

Nicolas LAMOUROUX
ESA CNRS 5023 Ecologie des Eaux Douces et des Grands Fleuves / Compagnie Nationale du Rhône
Université Lyon 1, F-69622 Villeurbanne Cedex, France
lamourou@biomserv.univ-lyon1.fr

Hervé CAPRA
CEMAGREF, Laboratoire d'Hydroécologie Quantitative, Division Biologie des Ecosystèmes Aquatiques
3 bis quai Chauveau, CP220, F-69336 Lyon Cedex 09, France

Marc POUILLY
ORSTOM, L'Institut Français de Recherche Scientifique pour le Développement en Coopération
Bat. 403, Université Lyon 1, F-69622 Villeurbanne Cedex, France

ABSTRACT

Quantitative estimates of habitat suitability in a stream reach generally result from the coupling of a hydraulic habitat model and a biological model of habitat use. The choice of each of these models has led to much controversy and discussion. Nevertheless, most habitat studies on lotic fish use a deterministic hydraulic model and univariate suitability curves. The objective of this contribution is to present a new, alternative method, which relates statistical hydraulic models to multivariate habitat use models for fish.

Statistical hydraulic models aim at predicting the frequency distributions of hydraulic variables such as velocity or water depth within stream reaches. Their main advantage is the simplicity of their input variables: discharge and average characteristics of the stream. The multivariate habitat use models involved in our approach take into account the local variability of fish habitat. Based on regression analyses, they predict habitat suitability as a function of the frequency distribution of point velocity, water depth and roughness within the local fish habitat.

We demonstrate how these two model types can be related. Indeed, under certain conditions, multivariate habitat use models developped at the scale of the local fish habitat remain valid at the scale of the whole stream reach, i.e. the scale of the hydraulic description. The resulting model is an attractive tool to estimate habitat suitability in a stream reach as a function of discharge, which is demonstrated using an example on two species (barbel, chub) at different life stages in a regulated reach of the French Rhône river.

The main limitations of our approach are due to mathematical constraints associated with the linkage of the two modeling approaches, and current difficulties in transfering biological models from one stream to others because of insufficient data. Nevertheless, our first simulations strongly encourage 1) the development of statistical approaches to describe hydraulic variables, and 2) the study of multivariate habitat use models that apply to a large variety of streams.

KEY-WORDS: Minimum flow / Habitat Modeling / Statistical Hydraulics / Habitat use models / Fish

INTRODUCTION

Several interactive factors influence stream living organisms at various spatial and temporal scales (Orth, 1995; Bleed, 1987; Armitage, 1989). Among these factors the hydraulic variables play a central role, because they are directly affected by flow regulation. Considering that species show significant preferences for certain values of point velocity, local depth, bottom stress and substrate elements size (Bovee, 1982; Gore and Judy, 1981; Stazner *et al.*, 1988), the hydraulic component of the habitat is expected to influence species distribution. These observations led to a wide use of quantitative methods to modeling stream habitat suitability, such as the Instream Flow Incremental Methodology (IFIM), which are based on the coupling of a hydraulic habitat model and biological preference models (Bovee, 1982; Nestler *et al.*, 1989).

The choice of each of these models has led to much discussion (Mathur *et al.*, 1985; Gore and Nestler, 1988; Osborne *et al.*, 1988). In most cases, the hydraulic model is derived from the hydraulic engineering practice (Nestler *et al.*, 1989; Bechara *et al.*, 1994), which is poorly adapted to complex flow simulations especially under low flow conditions (Osborne *et al.*, 1988; Beebe, 1996). These numerical simulations require precise topographic samples (Olsen and Stokseth, 1994), experience (Nestler *et al.*, 1989), and describe the hydraulic variables at a spatial scale that generally differs from the scale of the biological model (Heggenes, 1994). The wide use of univariate suitability curves as biological models (Bovee, 1982) was criticized for not considering the hydraulic variables covariances and biological interactions (Mathur *et al.*, 1985; Gore and Nestler, 1988), and for defining the hydraulic habitat by a single value for each variable, whereas the real habitat of species consists in a distribution of microhabitat conditions (Bain, 1995). Discussion finally concerned the predictive ability of quantitative instream flow assessment methods (Mathur *et al.*, 1985; Gore and Nestler, 1988). A major cause of such discussion is that other factors than local hydraulic conditions interact to influence fish density in stream reaches, making it difficult to validate the instream flow assessment methods by comparison with fish samples (Gore and Nestler, 1988). Thus, only few validation studies were proposed, generally concerning salmonids (Capra *et al.*, 1995).

For all these reasons, existing instream flow assessment methods should be considered as evolutionary tools. In particular, simpler approaches would allow wider applications and accelerate the validation process which consists, among others, in a better quantification of the part plaid by hydraulics in stream ecosystems functioning (Nestler *et al.*, 1989; Orth and Leonard, 1990; Orth, 1995). Osborne *et al.* (1988) proposed the use of regression techniques to simulate the evolution with discharge of the distributions of hydraulic variables (velocity, depth) using unexpensive measurements in representative sections of the reach. One other possibility to simplify the method is to replace the complex hydraulic numerical simulation by the description of the hydraulic habitat in the probability domain. Lamouroux *et al.* (1992) and Lamouroux *et al.* (1995) demonstrated that the frequency distributions of hydraulic variables show common patterns among different stream reaches, and can be modeled using simple measurements (Lamouroux, 1995). Such properties open interesting perspectives in describing the hydraulics conditions at a given site. However, the question of coupling such statistical hydraulic models with biological models in order to derive instream flow requirements methods remains open. Therefore, we present a theoretical model of habitat suitability that links statistical hydraulic models with multivariate habitat use models for fish. Because both models propose solutions to some of the above criticisms but are new and in progress, the method we present should be considered as a perspective to derive alternatives in instream flow needs modeling. An illustrative example is given for a French river to demonstrate the potential of our approach, while a detailed discussion on the material used in the example and the results obtained is beyond the scope of this contribution. In addition, the advantages and limits of our alternative method are discussed.

DEFINITIONS

Definition Of A Stream Reach And Reach Elements

In this contribution we refer to a stream reach as several pool-riffle successions (Newbury and Gaboury, 1993). We define reach elements as spatial subdivisions of the reach. The reach element is the spatial scale at which the fish habitat use model is developped (Figure 1). We emphasize that these reach elements can be defined at any spatial scale. They can be elements with a fixed surface, or cells of a hydraulic numerical model, or groups of cells. In any case, the reach is supposed to be a sum of reach elements.

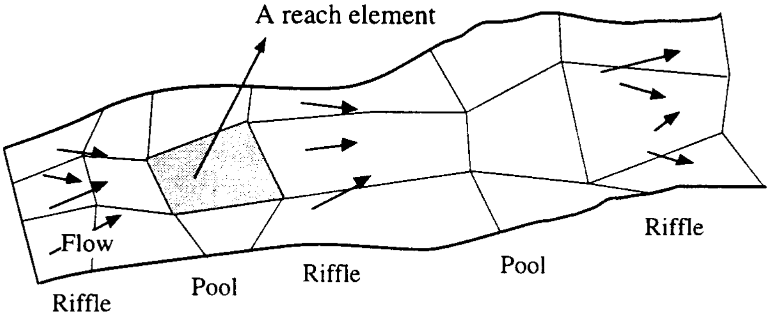


Figure 1: Plan view of a stream reach divided into reach elements. The fish habitat use model is developped at the scale of the reach elements.

Definition Of The Habitat Variability Within A Reach Element

In quantitative methods derived from IFIM (Bovee, 1982; Nestler *et al.*, 1989), fish habitat in a reach element is described by the mean values of different physical variables such as flow velocity, water depth, and substrate particles size. In the approach we present here, the habitat is not defined by these mean values, but by the frequency distributions of the physical variables in the reach element. This approach allows to take into account the spatial variability of the local habitat which is an essential element for fish according to the concept of energetic costs optimisation (Fausch, 1984; Facey and Grossman, 1992). Therefore, each physical variable describing the habitat is divided into several classes which have a given weight within the reach element:

Let P_i ($i = 1...n$) be the physical variables, where n is the number of variables describing the habitat.

Let P_{ij} ($j = 1...c_i$) be classes of the variable P_i, where c_i is the number of classes of the variable P_i.

Let w_{ijk} ($k = 1...r$) be the weight associated with the class P_{ij} within the reach element k, where r is the number of reach elements within the reach. This weight can either be the area or the volume of water whose values of the variable P_i fall in the class P_{ij}. The choice of weighting a class P_{ij} by the area or the volume it represents depends on the nature of the variable P_i: for example, water depth, mean velocities on a vertical, or substrate particles sizes are generally variables whose classes are weighted by the area they represent. However, point velocity classes can be associated with the volume of water they represent. In any case, the weights w_{ijk} are simply derived from the frequency distribution of the habitat variable within the reach element, and the dimensions of this element.

We define the habitat within the reach element k as the values of w_{ijk}, which are expressions of the frequency distributions of the habitat variables within the reach element (Figure 2).

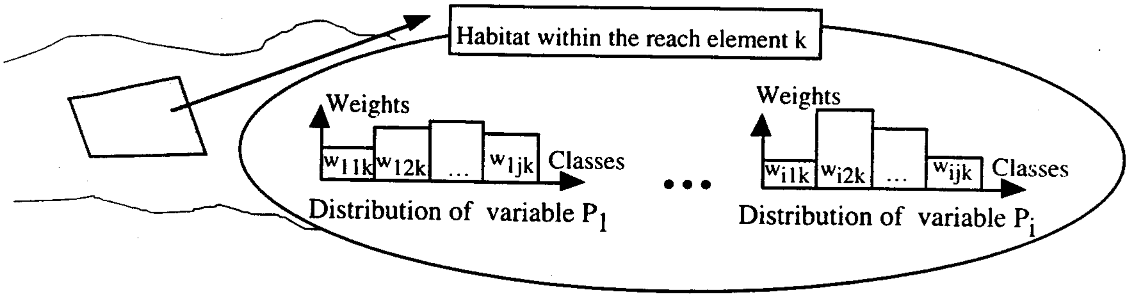


Figure 2: The habitat variability in a reach element is described by distributions of habitat variables.

MULTIVARIATE HABITAT USE MODELS FOR FISH INVOLVED IN OUR APPROACH

We define I_k ($k = 1...r$) as the suitability index estimated by the biological model as a function of the habitat variability within the reach element k. In our approach, we consider multivariate habitat use models that predict I_k as a linear combination of the different values of w_{ijk} ($i=1...n$, $j=1...c_i$), as follows:

$$(1) \qquad I_k = \sum_{i=1}^{i=n} \sum_{j=1}^{j=c_i} a_{ij} \cdot w_{ijk} + \text{error}$$

where a_{ij} can be obtained from multiple regression analyses. In Equation 1, the index is expressed as a linear function of the weights of the habitat variables classes. Thus, contrary to a model defining the index as a linear combination of the habitat variables themselves, the model of Equation 1 can reflect a preference for intermediate values of the habitat variables. It is therefore flexible in reproducing various species utilization niches (Hutchinson, 1957). In addition, it takes into account the covariances of the different weights of habitat variables classes, because these covariances influence the estimation of the coefficients a_{ij}.

Capra (1995) and Pouilly (1994) collected typical data to built such biological models on French streams of various sizes varying from streams whose dominant fish was brown trout (*Salmo trutta Fario*, L. 1758) to larger streams characterised by barbel (*Barbus barbus*, L. 1758). Bain (1995) also described means to randomly investigate microhabitats with a fixed surface. Analyses of this type of data (Pouilly and Souchon, 1994) already indicated that multivariate habitat use models based on the local frequency distribution of habitat variables could have better predictive properties than univariate suitability curves.

STATISTICAL HYDRAULIC MODELS

Statistical hydraulic models provide estimates of the frequency distributions of shear stress (Lamouroux et al., 1992) or point velocity (Lamouroux et al., 1995) at the scale of the whole stream reach. They have been developped on streams with discharge below bankfull discharge, up to 20 m^3/s. However, they are expected to remain valid for higher discharges because they are based on dimensionless descriptors of the reach that reflect common patterns in streams of different sizes. Their input variables are discharge, mean depth, mean width, and mean substrate particles size of the reach. These models are based on the observations that the frequency distributions of hydraulic variables show constant patterns among very different streams, and strongly depend on the average characteristics of the reach. Ongoing works describe comparable behaviour concerning the frequency distributions of water depth, which is with stress and velocity the third important hydraulic variable for living organisms in streams (Heggenes, 1994; Lamouroux et al, 1992).

If the evolutions of the average depth and the average width as a function of discharge are known, statistical hydraulic models provide estimates of the frequency distributions of hydraulic variables within the reach at any discharge. For example, Lamouroux (1995) proposed to fit depth-discharge and width-discharge relationships to power laws (see Leopold *et al.*, 1964), using measurements of about 100 depths and 15 widths within the reach at 2 different discharges. Thus, using statistical hydraulic models, affordable measurements make it possible to estimate the distributions of hydraulic variables at any discharge, without complex topographic sampling, without difficult velocity or stress measurements, and without experience of hydraulic models calibration (Figure 3).

Let W_{ij} ($i = 1...n$; $j = 1...c_i$) be the weight (surface or volume) associated with the class P_{ij} within the whole reach. Statistical hydraulic models estimate the values of the weights W_{ij} using simple measurements (Figure 3).

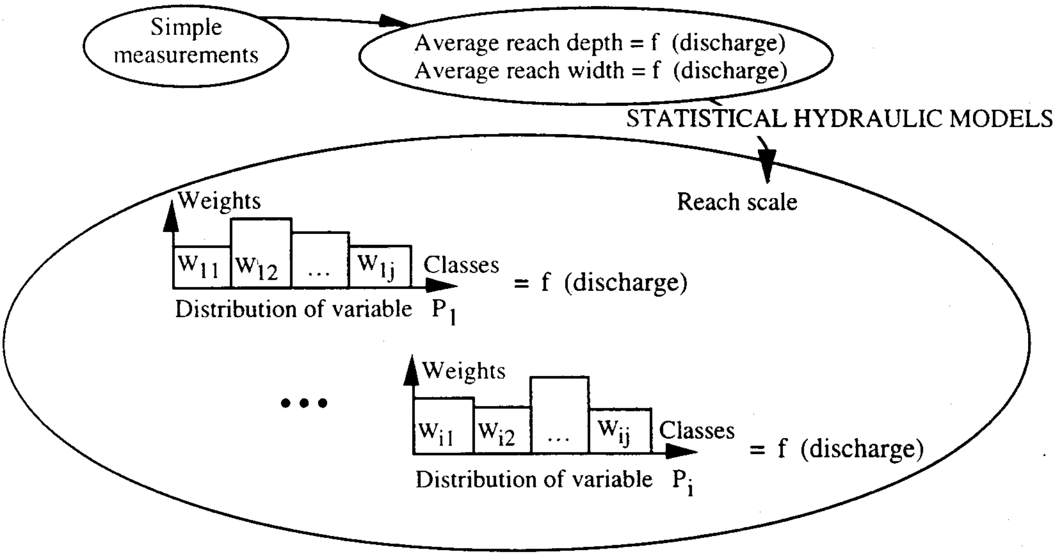


Figure 3: Predicting distributions of hydraulic variables in a reach using statistical hydraulic models.

COUPLING STATISTICAL HYDRAULIC MODELS AND MULTIVARIATE HABITAT USE MODELS

In this section, we demonstrate that, though developped at a local scale of the reach elements, the multivariate habitat use models defined in Equation 1 can be applied at the scale of the whole reach. With this aim in view, we define a total suitability index on the stream reach, I, as the sum of the suitability indexes predicted on the different reach elements using Equation 1. Thus, we have:

(2)
$$I = \sum_{k=1}^{k=r} I_k = \sum_{k=1}^{k=r} \left(\sum_{i=1}^{i=n} \sum_{j=1}^{j=c_i} a_{ij} \cdot w_{ijk} \right) + \text{error}$$

Which can be rewritten as:

(3)
$$I = \sum_{i=1}^{i=n} \sum_{j=1}^{j=c_i} a_{ij} \cdot \left(\sum_{k=1}^{k=r} w_{ijk} \right) + \text{error}$$

or finally:

(4
$$I = \sum_{i=1}^{i=n} \sum_{j=1}^{j=c_i} a_{ij} \,.\, W_{ij} + \text{error}$$

because for ($i = 1 \ldots n; j=1 \ldots c_i$), we have:

(5)
$$W_{ij} = \sum_{k=1}^{k=r} w_{ijk}$$

Equation 4 is similar to Equation 1, but applies to the whole reach and not to reach elements. Therefore, the multivariate habitat use model defined on reach elements can be applied to the whole stream reach, exactly as if the reach was a single reach element. This mathematical property entirely proceeds from the linear form of Equation 1, which allows to sum the weights of a given habitat variable class among the different reach elements when summing the indexes. Its major consequence is that a suitability index can be calculated for the reach as a function of the frequency distribution of the physical variables P_i within the whole reach. Thus, in order to predict the evolution of this index with discharge in a given reach, it is necessary to estimate the volumes or the surfaces associated with the different classes of habitat variables at different discharges. This is exactly the kind of information provided by statistical hydraulic models. Therefore, the coupling can be performed between multivariate habitat use models and statistical hydraulic models following the scheme of Figure 4.

In Figure 4, statistical models provide the distributions in the reach of hydraulic variables considered independently, i.e. whose covariances are not described. However, our model takes into account the covariances between the weights of the classes of hydraulic variables in reach elements, because these covariances influence the values obtained for the coefficients a_{ij}. Therefore, the computation of Equation 1 integrates the dependant behaviour of the hydraulic variables, and this information is implicitely taken into account when using the biological model.

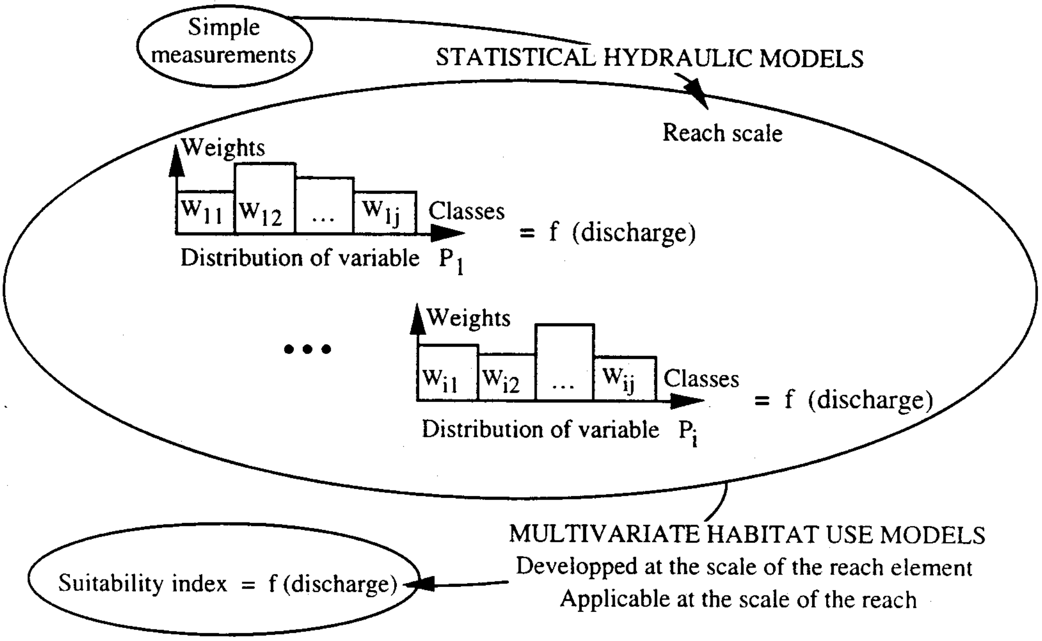


Figure 4: Coupling statistical hydraulic models with multivariate habitat use models.

EXAMPLE ON A FRENCH STREAM

Choice Of Biological Models

The multivariate habitat use models of our example were computed using the data of Pouilly (1994) which contain biological and physical descriptions of 668 reach elements sampled in three French streams of the Rhône river basin, the Ain, the Rhône and the Ardèche rivers. The reach elements were defined at a spatial scale of 5 to 50 m2 (Capra, 1995; Pouilly, 1994). Within each reach element, whose area and volume were estimated, the fishes were collected by electro-fishing, identified and measured. Then, the frequency distributions of point velocity, water depth, and substrate particles size were measured using approximatively 10 representative points. The three subsets of data corresponding to the different streams were analysed together in order to derive habitat use models matching Equation 1, because the information on only one stream for a given species was not always sufficient enough to reflect significant trends. For this analyses, two habitat variables were considered: point velocity and water depth.

Five classes of point velocity were defined ([0,0.05[; [0.05,0.2[; [0.2,0.4[; [0.4,0.8[; [0.8,∞[; in ms^{-1}). In each reach element k, these classes where weighted by the volumes of water w_{11k} to w_{15k}.

Four classes of water depth were defined ([0,0.2[; [0.2,0.4[; [0.4,0.8[; [0.8,∞[; in m). In each reach element k, these classes where weighted by the areas w_{21k} to w_{25k}.

The total volume of a reach element was noted:

(6)
$$v_k = \sum_{j=1}^{j=5} w_{1jk}$$

Following the requirements of Equation 1, we derived habitat use models as :

(7)
$$I_k = v_k \,.\, \ln(1+1000\, d_k) = \sum_{j=1}^{j=5} a_{1j}.w_{1jk} + \sum_{j=1}^{j=4} a_{2j}.w_{2jk} + \text{error}$$

where d_k is the fish density within the reach element. In Equation 7, the index I_k is defined as a linear combination of areas and volumes associated to classes of hydraulic variables. Thus, if we double the area and volume of a given reach element without changing the frequency distributions of depth and velocity, the estimated index in this new reach element would also be doubled. Therefore, it is natural to express the index as a function of fish density, which is independant of the reach element size, multiplied by the area or the volume of the reach element. The latter was chosen in our example.

A robust computation of the coefficients a_{ij} using multiple regression requires that the explicative variables w_{ijk} and the dependant variable I_k do not have distributions among the different reach elements with isolated extreme values. Such extreme values in regression analyses strongly influence the estimation of regression coefficients, and interfere heavily with the robustness of the regressive model (Saporta, 1978). In our example, such problems existed but were minimized by i) a log-tranformation of the fish densities in the definition of the index I_k (Equation 7), and ii) the tranformation of Equation 7 in a form independant of volumes of reach elements. Thus, the coefficients aij were estimated using a multiple regression performed on the new following expression of the model:

(8)
$$I_k/v_k = \ln(1+1000\, d_k) = \sum_{j=1}^{j=5} a_{1j}.\, w_{1jk}/v_k + \sum_{j=1}^{j=4} a_{2j}.\, w_{2jk}/v_k + \text{error}$$

Equation 8 reflects better than Equation 7 that the chosen model fits a function of fish density to a linear combination of the frequencies of point velocity classes, and the areas associated with water depth classes in a volume unit. Models were derived for two characteristic species of the sampled streams, both restricted to a given size class: large barbels (*Barbus barbus*, L. 1758; length > 220 mm), and small chubs (*Leuciscus cephalus*, L. 1758; length < 80 mm) (Table 1).

Table 1: Habitat use models (Equation 8) coefficients for barbels (length > 220mm) and chubs (length < 80 mm). Regression coefficients significantly different from zero (p<0.05) are underlined. The variance explained (r^2) and the probability associated with the absence of a linear relation (p) are provided.

species	a11	a12	a13	a14	a15	a21	a22	a23	a24	r^2	p
barbel	0.51	1.01	1.12	$\underline{1.39}$	$\underline{1.43}$	$\underline{-0.16}$	-0.07	0.01	-0.33	0.05	<0.001
chub	1.20	0.16	-0.75	$\underline{-1.78}$	$\underline{-1.55}$	$\underline{0.48}$	$\underline{0.77}$	$\underline{0.91}$	1.10	0.21	<0.001

The low proportions of variance explained by these regressive models (r^2 in Table 1) indicate that it is difficult to predict fish densities (log transformed) in a reach element in a precise manner. Nevertheless, the low values of p in Table 1 demonstrate that the models reveal significant trends in fish response to hydraulic conditions in reach elements, i.e. they reflect that the probability distributions of density differ for reach elements whose hydraulic characteristics are different.

Application to the Rhône River.

The Rhône in Montelimar (south of Lyon, France) is a stream reach with a highly reduced discharge. The minimum discharge is 15 m^3/s whereas the natural average discharge was about 1500 m^3/s. However, the river morphology is close to a natural one, and the flow is organized into several pool-riffle successions. The substrate particles sizes were visually observed in several hundred points, and had an average of 0.15 m. In addition, regular surveys of depth and width distributions performed by the Compagnie Nationale du Rhône (CNR) which is responsible for the stream management, helped to derive depth-discharge and width-discharge relationships for the reach. For discharges below bankfull, these relationships were (with 95% confidence limits of the slope coefficients provided in brackets):

$$\ln(\text{Depth}) = -0.87 + 0.30\ [\pm 0.05]\ .\ \ln(\text{Discharge}) \qquad r^2 = 0.95\ ;\ n = 11 \tag{9}$$

$$\ln(\text{Width}) = 4.4 + 0.14\ [\pm 0.01]\ .\ \ln(\text{Discharge}) \qquad r^2 = 0.99\ ;\ n = 11 \tag{10}$$

Equation 9 and 10 are linear forms of power laws linking depth and width to discharge (see Leopold *et al.*, 1964). The high values of r^2 in Equations 9 and 10 indicates the appropriateness of power models to describe depth-discharge and width-discharge relationships for discharges below bankfull discharge. It illustrates that Equations 9 and 10 could have been computed using simple depth and width measurements at two different discharges, as suggested by Lamouroux (1995).

Following the scheme of Figure 4, we used Equations 9 and 10 to derive the average characteristics of the reach. Then, a statistical hydraulic model (Lamouroux *et al,* 1995) was used to estimate the velocity frequency distributions within the reach as a function of discharge. A statistical model for depth distribution modeling is not available yet, but depth distributions were measured at different discharges in the reach by the CNR. Thus, the evolution with discharge of the weights associated with the different depth classes were directly interpolated from measurements. Finally, the biological models of Table 1 provided estimates of the suitability indexes I associated with large barbels and small chubs at different discharges.

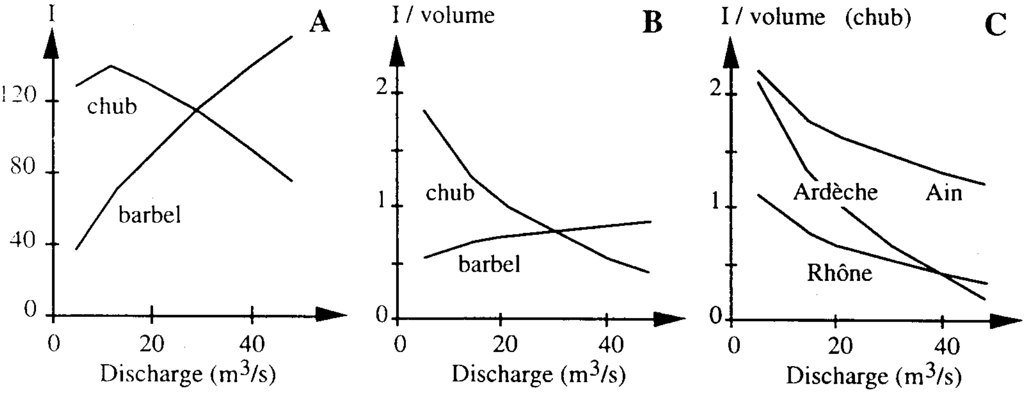


Figure 5: Suitability indexes (A) and volumic indexes (B) as a function of discharge in the Rhône River, calculated using habitat use models computed for three streams. Volumic indexes for chub corresponding to biological models estimated on each stream considered separately are provided (C).

Figure 5 shows examples of simulations for discharges between 5 and 50 m^3/s. The suitability indexes I (calculated for a unit of stream length) increases with discharge for large barbels and follows a bell curve for small chubs (Figure 5A). I is a linear combination of volumes and areas within the reach, as already discussed for I_k in Equation 7. Thus, a quality index in a given volume can be computed by dividing I by the volume of water of the reach (Figure 5B). In our example, this volumic index increases for large barbels, which is the more rheophilic species, and decreases for small chubs with discharge.

Error analyses

An essential point when using curves similar to those of Figure 5 is to quantify uncertainty around the curves. This uncertainty has different sources. First, physical and biological measurements have their own uncertainty. In particular, fish samples are biased by the electrofishing methodology (Bovee, 1982). A second source of uncertainty proceeds from an application of the different models out of their validation field, i.e. without a strict respect of the basic hypothesis of the models. Finally, a last source of uncertainty is that of the different models. In our example, this last source of uncertainty has three origins: i) errors in calculating the average dimensions of the streams (Equations 9 and 10), ii) errors in deriving the distributions of hydraulic variables using statistical models, and iii) errors associated with the habitat use models (Equation 7).

A complete estimation of all errors is a very complex problem which is beyond the scope of this contribution, where the example is given for illustrative purpose. However, to give an idea of the order of magnitude of errors, we give rapid estimates of the error due to the uncertainty of the chub habitat use model of Table 1. For this purpose, new suitability curves were derived for small chubs using exclusively the data from each of the three sampled streams (Figure 5C). These simulations indicate that the three curves show significant deviations in value from the general curve developped for the basin (Figure 5B). However, the general curve roughly reproduces the common patterns among the three streams. Although part of these deviations can be attributed to insufficient available data in each stream, they suggest that our models, as widely used instream flow assessment methodologies, are not expected to provide precise estimates of species densities in a given stream: our models aim at reflecting general trends in the hydraulic behaviour of streams and in fish response to the hydraulic conditions which are common to the different streams. Therefore, the generality of the biological model among different streams has to be properly tested.

DISCUSSION

Statistical hydraulic models simplify the description of the habitat variables distributions (Lamouroux, 1995). We have demonstrated that they can be coupled with multivariate habitat use models in order to derive predictions of the impact on fish communities of discharge management. Using an example, we emphasized that applications of our modeling approach require that much attention must be paid to the study of errors, and that the models have been properly validated by confrontation with data to quantify their ability in reflecting significant trends for the whole fish population.

The main limits of our approach are the following: 1) Coupling statistical hydraulic models with multivariate habitat use models dictates that the biological models match the form of Equation 1, for mathematical reasons. These constraints limit the flexibility in the choice of the model predicting fish suitability indexes. 2) Though they properly apply to reaches where the discharge is regulated, statistical models require that the morphology of the stream reach be quasi-natural. They cannot be used in flows strongly affected by civil works. In addition, part of these models are difficult to validate for high discharges where the hydraulic measurements are difficult, and models concerning the water depth are ongoing works. Finally, 3) because the idea of introducing spatial variability in the description of the hydraulic habitat is recent, few data exist and do not describe all possible combinations of hydraulic habitats in a large variety of streams. As a consequence, multivariate habitat use models involved in our approach are at the moment hardly transferable from one stream to another without losses in accuracy. Further field samples and analyses are required to find models as general as possible, whose accuracy is known and predictive power is tested.

However, our approach proposes alternative solutions that answer several criticisms of widely used instream flow assessment methodologies. First, statistical hydraulic models do not require long topographic sampling and strong experience in hydraulics. Thus, they strongly simplify the method and can accelerate applications and validations (Nestler *et al.*, 1989; Orth and Leonard, 1990). In contrast to deterministic hydraulic models, they provide estimates of point variables and not of mean cell values. Second, the multivariate biological models we use take into account the interdependance of hydraulic variables (Mathur *et al.*, 1985). Because their input is the variability of the local fish habitat, they respect ecological concepts derived from the ecological niche concept (Hutchinson, 1957) or energy optimisation theories (Fausch, 1984; Facey and Grossman, 1992).

For these reasons, our first simulations strongly encourage 1) the development of statistical descriptions of the hydraulic conditions; 2) the study of multivariate habitat use models that take into account the habitat variability and predict fish communities in a similar manner for a large variety of streams. More generally, statistical descriptions in hydraulics as well as models of fish sensitivity to habitat variability can be developped at various spatial and temporal scales different from those used in our example. Thus, such approaches offer numerous perpectives of research.

REFERENCES

Armitage, P.D. (1989). The application of a classification and prediction technique based on macroinvertebrates to assess the effects of river regulation. In Gore, J.A., and G.E. Petts (eds), *Alternatives in Regulated River Management*, CRC Press, Boca Raton, Florida, USA, pp. 267-293.

Bain, M.B. (1995). Habitat at the local scale: Multivariate patterns for stream fishes. *Bulletin Français de la Pêche et de la Pisciculture.* 337/338/339, pp. 165-177.

Bechara, J.A., M. Leclerc, P. Boudreau, and L. Belzile (1994). A numerical method for modeling the dynamics of the spawning habitat of landlocked salmon. *Proceedings of the 1st International Symposium on Habitat Hydraulics*, Norvegian Institute of Technology, Trondheim, Norway. pp. 170-184.

Beebe, J.T. (1996). Fluid speed variability and the importance to managing fish habitat in rivers. *Regulated Rivers: Research and Management.* 12, pp. 63-79.

Bleed, A.S. (1987). Limitations of concepts used to determine instream flow requirements for habitat maintenance. *Water Resources Bulletin.* 23(6), pp. 1173-1178.

Bovee, K.D. (1982). A guide to stream habitat analysis using the instream flow incremental methodology. *Instream Flow Information Paper 12*, U.S. Fish and Wildlife Service, Fort Collins, Colorado, USA. 248 p.

Capra, H. (1995). Amélioration des modèles predictifs d'habitat de la truite fario : échelles d'échantillonage et intégration des chroniques hydrologiques. Thèse de Doctorat, Université Lyon 1, Cemagref BEA-LHQ, 281 p.

Capra, H., S. Valentin, and P. Breil (1995). Chroniques d'habitat et dynamique de populations de truite. *Bulletin Français de la Pêche et de la Pisciculture.* 337/338/339, pp. 337-344.

Facey, D.E., and G.D. Grossman (1992). The relationship between water velocity, energetic costs, and microhabitat use in four North American stream fishes. *Hydrobiologia.* 239, pp.1-6.

Fausch, K.D. (1984). Profitable stream positions for salmonids: relating specific growth rate to net energy gain. *Canadian Journal of Zoology.* 62, pp.441-451.

Gore, J.A., and R.D. Judy (1981). Predictive models of benthic macroinvertebrate density for use in instream flow and regulated flow management. *Canadian Journal of Fisheries and Aquatic Sciences*. 38, pp. 1363-1370.

Gore, J.A., and J.M. Nestler (1988). Instream flow studies in perspective. *Regulated Rivers: Research and Management*. 5, pp. 129-138.

Heggenes, J. (1994). Physical habitat selection by brown trout (*Salmo trutta*) and young atlantic salmon (*S. salar*) in spatially and temporally heterogeneous streams: implications for hydraulic modelling. *Proceedings of the 1st International Symposium on Habitat Hydraulics*, Norvegian Institute of Technology, Trondheim, Norway. pp. 12-30.

Hutchinson, G.E. (1957). Concluding remarks. Cold Spring Harbor Symposium on Quantitative Biology. 22, pp.415-427.

Lamouroux, N. (1995). Les modèles statistiques de description de l'habitat hydraulique : des outils pour l'écologie. *Bulletin Français de la Pêche et de la Pisciculture*. 337/338/339, pp. 157-163.

Lamouroux, N., Y. Souchon and E.Herouin (1995). Predicting velocity distributions in stream reaches. *Water Resources Research*. 31(9), pp. 2367-2375.

Lamouroux, N., B. Statzner, U. Fuchs, F. Kohmann, and U. Schmedtje (1992). An unconventional approach to modeling spatial and temporal variability of local shear stress in stream segments. *Water Resources Research*. 28, pp. 3251-3258.

Leopold, L.B., M.G. Wolman, and J.P. Miller (1964). *Fluvial processes in geomorphology*, W.H. Freeman & Co., San Fancisco, California. 522p.

Mathur, D., W.H. Bason, E.J. Purdy, and C.A. Silver (1985). A critique of the instream flow incremental methodology. *Canadian Journal of Fisheries and Aquatic Sciences*. 42, pp. 825-831.

Nestler, J.M., R.T. Milhous, and J.B. Layzer (1989). Instream habitat modeling techniques. In Gore, J.A., and G.E. Petts (eds), *Alternatives in Regulated River Management*, CRC Press, Boca Raton, Florida, USA, pp. 295-315.

Newbury, R., and M. Gaboury (1993). Exploration and rehabilitation of hydraulic habitats in streams using principles of fluvial behaviour. *Freshwater Biology*. 29, pp. 195-210.

Olsen, N.R.B., and S.M. Stokseth (1994). Three-dimensional modeling of hydraulic habitat in rivers with large bed roughness. *Proceedings of the 1st International Symposium on Habitat Hydraulics*, Norvegian Institute of Technology, Trondheim, Norway. pp. 99-112.

Orth, D.J. (1995). Food web influences on fish population responses to instream flow. *Bulletin Français de la Pêche et de la Pisciculture*. 337/338/339, pp. 317-328.

Orth, D.J., and P.M. Leonard (1990). Comparison of discharge methods and habitat optimization for recommending instream flows to protect fish habitat. *Regulated Rivers: Research and Management*. 5, pp. 129-138.

Osborne, L.L., M.J. Wiley, and R.W. Larimore (1988). Assessment of the water surface profile model: Accuracy of predicted instream fish habitat conditions in low-gradient, warmwater streams. *Regulated Rivers: Research and Management*. 2, pp. 619-631.

Pouilly, M. (1994). Relations entre l'habitat physique et les poissons des zones à cyprinidés rhéophiles dans trois cours d'eau du bassin Rhôdanien : vers une simulation de la capacité d'accueil pour les peuplements. Thèse de Doctorat, Université Lyon 1, Cemagref BEA-LHQ, 256 p.

Pouilly, M., and Y. Souchon (1994). Simulation de l'habitat physique du Barbeau fluviatile (*barbus barbus*, L.1758): Choix des modèles biologiques et sensibilité de la réponse. *Bulletin Français de la Pêche et de la Pisciculture*. 334, pp. 213-225.

Saporta, G. (1978). *Théories et Méthodes de la Statistique*, Publications de l'Institut Français du Pétrole, Editions Technip, Paris . 386 p.

Statzner, B., J.A. Gore, and V.H. Resh (1988). Hydraulic stream ecology: observed patterns and potential applications. *Journal of the North American Benthological Society*. 7(4), pp. 307-360.

Classifying Stream Habitat Using Fish Community Analyses

Mark B. Bain and Jeremy G. Knight
New York Cooperative Fish and Wildlife Research Unit
Department of Natural Resources
Fernow Hall - Cornell University
Ithaca, New York 14853
USA
MBB1@Cornell.edu

ABSTRACT

Fish and water management agencies would benefit from a classification system that simplifies relations between fish communities and physical habitat in rivers and streams. A practical method should be simple, appropriately handle high species diversity, and be transferrable across a wide range of streams. We developed a habitat classification model based on the microdistribution of fish species and life stages in seven small rivers and large streams in the Southeastern United States. Fish were sampled using 2-m^2 prepositioned area electrofishing frames and microhabitat variables (water depth, current velocity, substrate type, and instream cover) were recorded within the corners of these discrete samples. Fish and microhabitat data from the 10 sites were analyzed using multivariate analysis of variance (MANOVA) to determine if each species was using a distinct microhabitat. A principle components analyses (PCA) was performed on pooled data from all study sites to generalize species-habitat relations and identify a robust pattern.

A total of 1,578 samples and 13,524 fish were recorded, including 79 species across the 10 study sites. MANOVA's performed on 59 species indicated that most species showed significant microhabitat associations; however, some species were less consistent than others in habitat use across study sites. Species included in the pooled PCA plot were grouped based on generalized patterns seen in the multivariate habitat plot and on the species-specific results of the MANOVA models. We identified 5 habitat-use assemblages and their corresponding microhabitats by compromising species-specific precision to identify a simple multispecies-habitat pattern. The types of habitat used by each of the 5 fish assemblages were: shallow-fast habitat (e.g., ≤35 cm, ≥0.55 cm/s); slow-cover habitat (e.g., ≤0.20 cm/s, occurrence of structure); deep-fast habitat (e.g., >0.35 cm, >0.45 cm/s); shallow-slow habitat (e.g., < 0.35 cm, < 0.35 cm/s); and shallow-coarse habitat (e.g., <0.35, gravel or smaller). The five types of habitat provide a simplified species-habitat model for assessing and managing physical change in streams. Discrete area fish samples were categorized using the microhabitat classification. Some samples whose attributes fell outside the range of the five habitat types were considered unclassified habitat and these generally contained few fish. Many samples fit multiple habitat classes indicating that is probably not possible to develop exclusive habitat categories for a diversity of fish species. Shallow-slow and shallow-coarse habitat types tended to possess the most species and highest densities of fish. These will most likely be the habitats of interest for impact assessment because studies have shown that impacts to fish are selective and most pronounced on fish using shallow, shoreline areas of streams and rivers.

KEY-WORDS: Microhabitat / communities / multivariate statistics / habitat classification / diversity / habitat modeling / Southeastern USA / fish classification / habitat use /

INTRODUCTION

Natural resource agencies need widely usable tools for assessing the effects of streamflow and habitat changes on rivers and streams for the purpose of maintaining or recovering indigenous riverine fishes. A practical method for assessing flow regulation impacts will require some means of handling the high fish diversity and complex physical structure of many large streams and rivers. One solution is based on identifying a generalized pattern in the relationship between stream habitat and stream fish assemblages. The development of microhabitat types for fish with a corresponding classification of instream habitat (physical variables such as depth, velocity, substrate and cover) types would provide a means to analyze and predict the effects of habitat change and flow alteration

Aggregating species into functional or structural groups is common in ecology (e.g., carnivores) and fisheries management (e.g., forage fishes) and this practice has been of great practical and analytic value. Stream fish have been categorized into groups based on common trophic roles (Schlosser 1985), foraging modes (Horwitz 1978), life history characteristics (Winemiller and Rose 1992), and morphology (Gatz 1979). However, categorizing stream fish into groups based on microhabitat use is new, with only a few studies (Lobb and Orth 1991; Aadland 1993) reported to date. Stream fish respond to changes in habitat caused by flow regulation, and the effects of flow changes have been identified using patterns of habitat use by fish (Schlosser 1987; Bain et al. 1988; Leonard and Orth 1988; Kinsolving and Bain 1993, Travnicheck et al. 1995). This paper presents a generalized assemblage-habitat framework developed using field data on species-habitat relations from 10 sites on 7 streams with diverse fish communities and widely variable instream habitats and natural flow regimes. This framework extends the practice of classifying fish into groups by combining it with microhabitat classification to provide an tool for assessing complex habitats with high fish diversity.

METHODS

STUDY STREAMS

Ten study site on seven streams were identified in the Alabama River basin (Alabama, USA) that have U. S. Geological Survey streamflow gages, good water quality, and a diversity of habitats and flow regimes. The Alabama River basin has a diverse fish fauna (more than 157 species) and includes six ecoregions (Omernik 1987). A wide range of habitat types and habitat dynamics were incorporated in the study by selecting streams with wide variation in physical structure and flow regime. Study streams ranged from rocky, high gradient streams with little annual change in channel shape, to streams with shifting sand channels that continually changed shape through time. While we tried to select streams with roughly similar mean annual flows (3 - 8 m^3/s) , we intentionally selected streams with a variability of flow regimes (26 - 143 % CV). All of the selected streams have records of good water quality and no major environmental degradation.

Field Sampling

Fish and microhabitat data were collected in discrete areas using pre-positioned area electrofishing frames (Bain et al. 1985). Electorfishing frames measuring 1 x 2 m were placed along six fixed transects selected to include common instream habitat. Four frames were positioned across the transects so that two midstream areas and two stream margin areas were sampled. Midstream samples were taken at one-third and two-thirds of the stream width, and stream margin samples were taken adjacent to the bank and approximately 1 m off the bank respectively. Usually 24 samples were taken at a site on a given collection date.

Sampling was conducted at base flow discharges from late May through early September in 1989 and 1990. Fish were collected by electrifying a frame for approximately 20 to 30 seconds using a 750 Watt AC generator coupled to a variable voltage (approximately 900 volts maximum) transformer. Immediately after activating the power supply, a 3.0 x 1.2 m seine with 3.2 mm mesh size was placed downstream of the frame so that stunned fish were swept into it. With the power active, one person dislodged substrate within the frame to better capture benthic fishes. When current velocities were negligible, or in areas where a seine was difficult to set, dip nets were sufficient for fish collection. Specimens larger than 20 mm were identified to species.

Habitat parameters recorded at each sample included depth (cm), substrate type (coded), and water velocity (cm/sec). Each of these measurements were recorded at the four corners of sampled frames. Water velocity was measured with a Marsh-McBirney current meter set at 0.6 water depth (mean velocities for a vertical position) with a calibrated wading rod. Water depth was also recorded with the wading rod. Cover was recorded as the number of objects larger than a pencil's diameter, laying across the frame's perimeter. Substrate was coded as follows: 1= clay (<0.06 mm); 2=silt (0.06-0.1 mm); 3=fine to coarse sand (0.1-1 mm); 4= fine to coarse gravel (1-60 mm); 5=cobble (60-150 mm); 6=boulder (>150 mm); and 7=bedrock consisting of hard, flat or uplifted surfaces including hard clay surfaces.

Data Analysis

Each sample provided habitat attributes and the numbers of each fish species collected. Water depth, current velocity, and substrate type were analyzed using means of the four observations per sample. Samples containing no fish were also included in the data analysis. These data were analyzed first as 10 stream data sets and then as one data set with samples from all streams combined. Microhabitat attributes of occupied samples were correlated with total length of each captured fish by species to determine if habitat use was related to fish size. Species which showed high correlations (i.e. $r^2 > 0.90$) were split in to two size classes: large and small; the exact break depended on the species.

Relations between abiotic habitat attributes and the presence or absence of fish in samples were determined using a multivariate analysis of variance (MANOVA). This statistical procedure tested the hypothesis that multivariate habitat composition was similar among samples with and without a species. Initially the full null model (all variables) was tested and variables were eliminated on the basis of canonical loadings and changes in the F-ratio statistic for the model. The most concise but effective model for defining habitat was reported for each species. If no model was significant, the p-value for the full model was reported. Different models were tested until we concluded that there was no simpler model (fewer variables) that could distinguish used and unused habitat without sacrificing effective discrimination. If the best model had only one variable, then it was reported as an analysis of variance (ANOVA) model. These results were summarized, along with length and weight statistics and associated microhabitat statistics, for all common species collected across the 10 stream sites.

Principal components analysis (PCA) was used to illustrate the position of each fish species in habitat space. Only species that were found to use a specific microhabitat (significant MANOVA results) were used in the PCA analysis. PCA generates uncorrelated, multivariate habitat combinations (principle components) in order of decreasing importance (% of variance explained). Eigenvectors reported in each analysis were used to interpret the general orientation of important components. The important habitat components (eigenvalues >1.0, total variance >65%) were used to plot sample and species locations in multivariate habitat space. Species' mean locations in habitat space were determined by weighting each sample's factor score by the abundance of that species in the sample. Species which occurred in less than 5% of the total samples at a stream site were omitted from this analysis.

To identify generalized species-habitat patterns, PCA and cluster analyses were applied to the entire data set (all streams combined) using only specialist species. Specialist were species consistently found to use specific microhabitats in the MANOVA analyses. Species found to use habitat broadly (no significant MANOVA models) were considered habitat generalists and as such were eliminated from this analysis. Microhabitat-use assemblages were identified using the results of species MANOVA's, the pooled PCA, and a pooled cluster analysis. Species were separated into habitat-use assemblages based on similarities in microhabitat specialization. These species were then used as descriptors for our habitat types; i.e., we used the species present in a microhabitat-use group (descriptor species) to identify (1) the primary attributes of a habitat type (e.g. shallow-fast and shallow-cover), and (2) the quantitative criteria for defining a habitat type (e.g., water < 30 cm deep). Criteria were established by visually scanning multivariate habitat space, with samples containing one microhabitat-use group at a time, until the selected rsnge of habitat included a majority of samples containing the descriptor species. For each quantitative habitat criteria, we were satisfied with endpoints which captured at least 80% of samples containing any individuals in a descriptor group. This procedure was done using a high-resolution color monitor on a Macintosh IIci microcomputer running an exploratory statistics package (Data Desk® 1989) with manipulative multivariate graphical capability.

Once habitat types were defined, we compared the relative occurrence and density of species by habitat type as well as by stream sites. Using species densities by site and by habitat type, we determined those species which were dependent on a particular habitat type. Dependent species were those species which occurred in samples classified as a particular habitat type and whose densities in that habitat were at least 50% of its total density across stream sites. The relationship between fish community composition and instream habitat composition was then finalized and summarized.

RESULTS

In 1989 and 1990 we collected a total of 1,578 samples containing 13,524 fish from 10 stream sites. The total number of samples taken at each site ranged from 63 to 242. Overall, 79 species were collected, but stream sites varied considerably in species and abundances (15 - 40). Comparisons of habitat attributes (water depth, current velocity, substrate and cover) among the study sites indicated that sites with the highest mean cover values also tended to have the greatest mean depths. Stream sites with high mean substrate sizes (i.e., ranging from gravel-cobble) tended to have low mean current velocities (<30 cm/sec) and greater habitat variability than the sandy stream sites. In general, all habitat characteristics varied widely within, and among, the stream sites. MANOVA's performed on 59 species (those in 5% or more of the samples from a stream site) identified the important microhabitat attributes which could successfully predict the presence of a species (a subset of these results are in Table 1). Most (51) species had significant habitat associations.

Table 1. Sample MANOVA results for species that were later classified in group A (swift, coarse, shallow and no cover microhabitat) in Figure 1. Habitat attributes are shown that were part of a model to distinguish used from unused microhabitat. The abbreviations for the extremes of the habitat attributes are: SH = shallow, FA = fast, and CO = coarse. No significant model is shown by "ns" and a "-" indicates inadequate data for analysis.

	Study (acronym labels)									
Species	LRN	TER	HIL1	HIL2	HAT1	HAT2	OAK	UPH1	UPH2	LRS
Largescale stoneroller	SH/FA	SH/FA	SH/CO	ns	-	FA	CO	SH/FA	ns	-
Speckled chub (large)	-	-	-	-	-	-	-	FA/CO	-	-
Freckled madtom	-	-	FA/CO	SH	-	-	-	-	-	-
Speckled madtom	FA/CO	-	FA	-	CO	FA/SH	SH/CO	-	-	-
Greenbreast darter	SH/CO/FA	-	FA/CO/SH	FA/SH	CO	FA/SH	-	SH/FA/CO	-	-
Rock darter	-	-	-	-	-	ns	SH/FA	-	-	-
Banded sculpin	CO	FA/SH	-	-	-	-	-	-	-	-

Scientific names in order are: *Campostoma oligolepis, Macrhybopsis aestivalis, Noturus nocturnus, Noturus leptocanthus, Etheostoma jordani, Etheostoma rupestre, Cottus carolinae.*

Generalized Habitat Types

A data set pooled across all study sites included 51 species, 11,442 individuals and 1,578 samples with associated microhabitat attributes. A PCA of the pooled data yielded two components that incorporated about 60% of the total variance in microhabitat conditions. The first component was primarily composed of cover and current velocity and the second component was dominated by water depth and substrate coarseness (Figure 1). These two components described a two dimensional habitat space in which microhabitats ranged from fast current with no cover to slow current with abundant cover (component 1) and deep water with fine substrate to shallow water with coarse substrate (component 2). The plot of mean (specialist) species locations in habitat space revealed a general pattern of stream fish habitat use and the pooled range of habitat conditions. An abundance of species were concentrated in shallow microhabitats with coarse substrate and these varied along the water velocity and cover gradient. Those species which were located toward the slow, deep, high cover end of the available microhabitats were the sunfishes (*Centrarchidae*), small and large blacktail redhorse (*Moxostoma poecilurum*), mosquitofish (*Gambusia affinis*), pretty shiner (*Lythrurus bellus*) and blackspotted topminnow (*Fundulus olivaceus*). Conversely the species concentrated in the swift, coarse, shallow and no cover end of the available microhabitats are shown in Table 1. These two groups might be designated respectively as pool and riffle groups; however, this view would be oversimplified and would not account for the majority of intermediate species in habitat space.

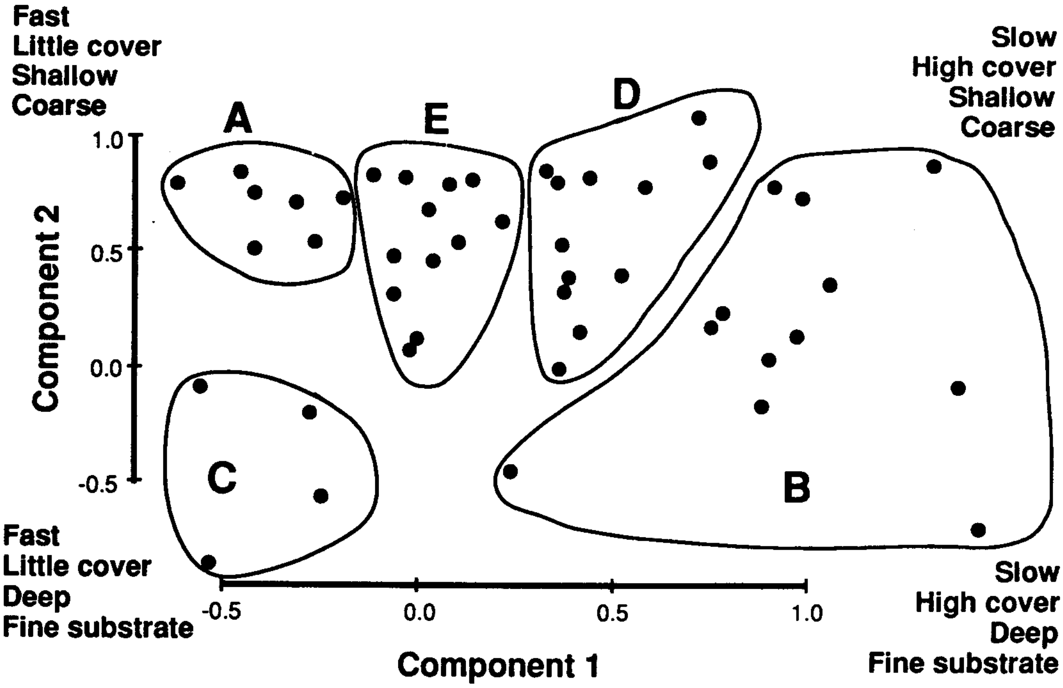


Figure 1. Mean species locations (solid circles) in habitat space. The irregular circles labeled with letters A through E encompass species with similar habitat use. These species groups were used to define habitat types A through E.

Species included in the pooled PCA plot were grouped based on similarity of mean microhabitat use and the species-specific results of the MANOVA models (examples in Table 1). Species which tended to be similar in microhabitat use in the grand PCA plot were characterized according to their significant habitat associations determined from the MANOVA models. For example, many species co-occurred in shallow microhabitats and MANOVA models indicated that some of these shallow-water species were primarily oriented to fast currents (e. g., speckled madtom) and some to cover. Consequently, subgroups could be formed within the species group concentrated in shallow water. We minimized the number of species groups by generalizing the patterns seen in the multivariate habitat plot and MANOVA models.

Species groups were interpreted as descriptors of a particular type of habitat (habitat-use assemblages), and their presence in samples was used to develop quantitative definitions of habitat types. We identified five habitat-use assemblages by balancing species-specific precision with the simplicity of a multispecies-habitat pattern. These were: a shallow-fast habitat-use group (habitat type A; letters referenced in Figure 1); a slow-high cover group (type B); a deep-fast group (C); a shallow-slow group (D); and a shallow-coarse group (E). The habitat types corresponded with important areas for species in a multivariate habitat space, consequently there was overlap between some habitat types for specific habitat attributes. A criterion for distinguishing deep from shallow habitat in all habitat types was a depth of 35 cm. However, no single criteria defined slow and fast current speed. Fast current was ≥55 cm/s and >45 cm/s for habitat types A and C, and slow current speed was ≤20 cm/s and <35 cm/s for types B and D. For habitat type B, a cover criterion was defined as present or absent and in type E coarse substrate was defined as ≥ 4 (gravel and larger).

Distribution of Fish by Habitat Type

Using the criteria for habitat types, we assigned each sample to one or more of the 5 habitat types, or as unclassified when the habitat attributes of that sample could not be classified. The number of samples in each type of habitat varied among the study sites and in some cases one or more habitat types were absent at a site. The percentage of samples in unclassified habitat was generally high (26 - 58%). Habitat type A comprised the smallest percentage of samples at a site and was absent at one site. The percentage of samples at a site classified in habitat types B and C was generally less than 25% of the total samples. Habitat type C was absent at two study sites. Habitat types D and E comprised the greatest percentage of habitat samples at a site, usually including greater than 25% of the samples.

A comparison of the number of species and density of fish by habitat type and study site indicated how fish were distributed among habitats. It is important to note two things when comparing fish-habitat relations this way. First, because the habitat types are non-exclusive, fish densities in each habitat type are not different subsets of a whole. Some fish were counted in multiple habitat types. Second densities were based on numbers, not weight, hence they tend to be biased toward smaller more abundant species. However, given that the vast majority of stream fish were small, less than 100 mm, very few species were affected by this bias. Habitat types D and E had the highest densities and diversity of fish. The number of species using a habitat type tended to be lowest in habitat A. Habitat types which were well represented at a stream site (i.e., >25% of the samples) usually possessed high fish densities.

DISCUSSION

Using study streams and sites which encompassed a wide range of habitats and flow regimes, we were successful in analyzing species-habitat patterns in a diversity of stream environments. The study sites varied considerably in habitat composition with some sites lacking entire categories of habitat. Where multiple sites were studied on a single stream, there remained great variation in habitat composition. Difference in total fish densities varied nine fold among the study sites and species richness more than a three fold.

Most of the fish species captured in enough samples for microhabitat analysis were found to specialize on some set of microhabitat conditions. Some species were very consistent in habitat use across different streams. The largescale stoneroller, speckled madtom, and greenbreast darter were consistently concentrated in microhabitats characterized as shallow with fast current and coarse substrate. Longear sunfish (*Lepomis megalotis*) were consistently concentrated in microhabitats that had slow current speed and abundant cover. However, some species had highly inconsistent habitat use patterns among the study sites. The bullhead minnow (*Pimephales vigilax*) was found to be concentrated in slow microhabitats at some sites and fast microhabitats at other sites. Spotted bass (*Micropterus punctulatus*) were concentrated in shallow water at two sites and this species used no particular set of microhabitat conditions at 4 sites.

Most of the 59 species analyzed were intermediate among these extremes and were found to be somewhat consistent, but not uniform, in habitat use. These results suggest that the range of available habitats at a stream site has an important effect on microhabitat use for most species. Therefore, the use of a particular set of criteria to define microhabitat needs for individual species does not appear feasible for species-rich, warmwater streams.

Despite variation in species-specific habitat use among study sites and size-classes, there were some general patterns evident. Fish concentrated in microhabitats with coarse substrate tended to specialize on fast current and shallow water. Likewise, species that were oriented to cover tended to use microhabitats that had slow current speed and fine substrate. Overall, most species were concentrated in shallow water with other important habitat attributes being substrate and current speed. Generalized habitat-use trends allowed us to identify species groups or habitat-use assemblages. However, species-habitat trends did not consistently apply at the family level. The diverse families (Cyprinidae, percidae, and Catostomidae) had members that were concentrated in specific microhabitats, but as a family they spanned nearly all habitat types. Consequently, coping with the high species diversity of warmwater streams and rivers using taxonomic levels more general than species does not appear to be a solution.

While habitat-use trends found by inspecting results from ten site-specific analyses suggest some general patterns, this kind of analysis obscures the absolute habitat conditions associated with any single species. For example, some species were found to be concentrated in swift samples taken at a study site, but these results do not provide a basis for defining a generalized velocity criteria for all study sites. We incorporated absolute similarities in habitat use by pooling all study sites and species in one analysis. Such an analysis sacrifices precision in identifying species-specific habitat use and eliminates any variation in habitat use due to the unique nature of instream habitat at a given stream site. However, such a compromise was necessary to produce a generalized pattern of fish-habitat relations that could be applied to a variety of streams and rivers.

The analysis of habitat-use patterns across study sites yielded five habitat-use assemblages which appeared distinct in their microhabitats. Some species were strongly oriented to cover, and others were restricted to microhabitats that were shallow. Both of these two species groups were found in habitats that had cover in shallow water even though our results indicated they were captured there in response to different habitat attributes. We could not find a way to categorize fish into groups using exclusive types of habitat because most species used very specific habitat defined by one or two attributes. Therefore, the five habitat-use assemblages often overlapped on microhabitat attribute which was not exclusive to any one habitat type. This finding was probably the most unexpected but clear result of this study.

Having identified five habitat-use assemblages, corresponding habitat criteria were identified. These criteria define the type of habitat that we assume are needed to support each group of fish. The resulting five habitat types provide a basis for assessing and managing changes in stream habitat in a manner that emphasizes the most sensitive components of a stream fish community. Overall, the species group-habitat type classification represents a simple model of the pattern between fish and habitat in the warmwater streams we studied. By incorporating data from study streams with very different macrohabitat features and flow regimes, we assume this generalized pattern is robust and potentially applicable to a wide range of streams and rivers.

Categories of habitat are more useful than categories of fish (i.e., habitat-use assemblages) as a basis for an impact assessment method. In streams outside those we studied, the species composition will vary, but the basic habitat types should not. Also, past habitat-based studies of streams with altered physical habitat (reviewed in Bain and Boltz 1989) suggest that the effects on fish are selective and dependent on habitats used by a species. The habitat types identified here are few in number for practicality and generality. Nevertheless, the habitat types included all mean habitat conditions of species that specialized on some type of microhabitat.

Habitat A was used by species generally regarded as riffle fishes (as in habitat descriptions of regional "fishes of" publications) and habitat A was defined as shallow (<35 cm) and fast (≥55 cm/s). Habitat B included slow (≤ 20 cm/s) current speeds with cover and it was important to many sunfishes as well as other species. Habitat C was important for a few sucker species and was defined as deep (≥35 cm) and fast (>45 cm/s). Habitat D was heavily used by many species in a variety of fish families and was defined as shallow (<35 cm) and slow (<35 cm/s). Like habitat D, habitat E was heavily used by a variety of species which occurred in shallow water (<35 cm), but habitat E was also defined as having coarse substrate (gravel and larger). Microhabitats that were outside the range of the five habitat types were collectively termed unclassified habitat and samples in these habitats generally contained low densities of fish.

Our habitat framework qualitatively corresponds with findings of stream fish-habitat research in the Midwest (Schlosser 1987; Aadland 1993), Great Plains (Peters et al. 1989; Peters and Holland 1990), Southeast (Ross et al. 1987), and Northeast (Bain et al. 1988; Lobb and Orth 1991). It is difficult to compare such quantitative studies with ours because of differences in methods, objectives, and analyses. However, fish-habitat research on the Platte River in Nebraska did employ similar methods and analyses (Peters et al. 1989, Peters and Holland 1990). In this large, turbid river, slow and shallow habitats (<30 cm deep, <30 cm/s) contained the majority of fish and species. Another group of fish used shallow and fast habitats (<30 cm, >30 cm/s). Work by Bain et al. (1988) in a New England river also indicated that most species and fish were in shallow (mean depth = 30 cm) and slow (mean surface current = 8 cm/s) microhabitats. As in this study, Aadland's (1993) identified six habitat types with variable habitat criteria but none of his types overlap, and his classification criteria are numerically quite different than ours. Finally, Lobb and Orth (1991) identified four exclusive habitat types by dividing habitat space (defined by water depth, current velocity, substrate, vegetation) into quadrants. Despite some similarities with past studies, no previous study has classified fish and habitat using all common habitat attributes (depth, current, substrate, cover), data from a variety of stream systems with natural flow regimes, and with an analytic approach that allows non-exclusive habitat types.

While the habitat framework identified some trends in the relations between fish and habitat, the generality of the framework contributes to some weaknesses. Habitat types that overlapped in space (actual habitat conditions) frequently had similar species composition and fish densities because some fish were being counted in multiple habitat types. The very abundant species ranked among the most common fish in several habitats because even at relatively low densities they were numerous relative to many specialist species. For example, the largescale stoneroller was was generally concentrated in very fast and shallow habitat (habitat A), it was often one of the most common fish in other habitat types. Finally, a few species that appeared to be descriptors of one type of habitat were later found to be concentrated in another habitat type at a particular stream site. Such weaknesses can be expected in any generalized classification of fish and habitat. Whether such weaknesses are acceptable trade-offs for greater utility depends on additional research on the applicability of the framework to other streams, and the usefulness of the framework for identifying the effects of habitat alterations.

REFERENCES

Aadland, L. P. 1993. Stream habitat types: their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* 13:790-806.

Bain, M. B., and J. M. Boltz. 1989. Regulated streamflow and warmwater stream fish: a general hypothesis and research agenda. U. S. Fish and Wildlife Service Biological Report 89(18).

Bain, M. B., J. T. Finn and H. E. Booke. 1985. A quantitative method for sampling riverine microhabitats by electrofishing. *North American Journal of Fisheries Management* 5:489-493.

Bain, M. B., J. T. Finn and H. E. Booke. 1988. Streamflow regulation and fish community structure. *Ecology* 69:382-392.

Gatz, A. J., Jr. 1979. Community organization in fishes as indicated by morphological features. *Ecology* 60:711-718.

Horwitz, R. J. 1978. Temporal variability patterns and the distributional patterns of stream fishes. *Ecological Monographs* 48:307-321.

Kinsolving, A. D., and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. *Ecological Applications* 3:531-544.

Leonard, P., and D. J. Orth. 1988. Use of habitat guilds of fishes to determine instream flow requirements. *North American Journal of Fisheries Management* 8:399-409.

Lobb, M. D., and D. J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. *Transactions of the American Fisheries Society* 120:65-78.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77:118-125.

Peters, E. J., R. S. Holland, M. A. Callam, and D. L. Bunnell. 1989. Platte River suitability criteria. Nebraska Game and Parks Commission Technical Series 17, Lincoln, Nebraska, USA.

Peters, E. J., and R. S. Holland. 1990. Temporal variation in fish community abundance in the lower Platte River, Nebraska (abstract). 52nd Midwest Fish and Wildlife Conference Program.

Ross, S. T. , J. A. Baker, and K. E. Clark. 1987. Microhabitat partitioning of southeastern stream fishes: temporal and spatial predictability. Pages 42-60 in W. J. Matthews and D. J. Heins, editors. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, OK, USA.

Schlosser, I. J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* 66:1484-1490.

Schlosser, I. J. 1987. A conceptual framework for fish communities in small warmwater streams. p. 17-24 in W. J. Matthews and D. J. Heins (editors). Community and evolutionary ecology of North

American stream fishes. University of Oklahoma Press, Norman, OK.

Travnicheck, V. H., M. B. Bain, and M. J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124:836-844.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life-history diversification in North American fishes: implications for population regulation. *Canadian Journal of Fisheries and Aquatic Sciences* 49:2196-2218.

EQUAL AREA LINE-TRANSECT SAMPLING FOR SMALLMOUTH BASS HABITAT SUITABILITY CRITERIA IN THE SUSQUEHANNA RIVER, PENNSYLVANIA

Mark A. Allen
Thomas R. Payne & Associates
850 G Street, Suite J
Arcata, California 95521 USA
trpa@northcoast.com

ABSTRACT

Two new sampling methods were applied to the collection of data on spawning and rearing smallmouth bass (*Micropterus dolomieui*) in the mainstem Susquehanna River, Pennsylvania. Both line-transect sampling and equal habitat area sampling are believed to reduce problems inherent with previous methods of collecting data and developing habitat suitability index (HSI) criteria for use with aquatic habitat modeling. The study was designed to minimize the biasing effects of habitat availability, observer subjectivity, and fish disturbance through the use of equal area sampling within habitat type strata, transect randomization, and direct observation of life stages. A 7.5km reach of the Susquehanna River was stratified into five habitat types based on depth, velocity, substrate, and instream cover characteristics. Stratification allowed a controlled distribution of effort with a reasonable assurance that all available habitat characteristics would be sampled. Line-transect sampling is an effective subsampling technique for large rivers that allows random placement and equal effort allocation within habitat type strata. Direct observation of transects using snorkeling or SCUBA gear ensured accurate location and description of chosen microhabitats. Direct observation of 81 transects yielded microhabitat data at focal positions of 50 spawning nests, 196 young-of-year bass, and 129 juvenile and adult bass. Comparison of the Susquehanna River data with habitat suitability models currently in use revealed several important differences in habitat use by smallmouth for spawning and rearing. A close association was observed between instream cover and spawning and fry rearing which could have important consequences for habitat modeling.

KEY-WORDS: Smallmouth Bass/Microhabitat/Habitat Suitability/Instream Flows/Equal-Area/Line-Transect

INTRODUCTION

A key component to modeling the relationship between streamflows and fish habitat with the Instream Flow Incremental Methodology (IFIM) is information describing the microhabitat requirements of the affected fish species. In the IFIM context, these requirements are typically described by measurement of water depths, water velocities, bottom substrate materials, and cover characteristics at the focal positions where fish are observed. Analysis and interpretation of these microhabitat observations produce Habitat Suitability Index (HSI) criteria. The biological responses of fish to stream habitat (described by HSI) are interpreted with predicted physical responses of the river to changes in streamflow or surface elevations (by log-log rating, step-backwater, or other hydraulic models) to evaluate potential changes in fish populations.

IFIM results can be very sensitive to differences in HSI (Modde and Hardy, 1992), and HSI in turn are known to be very sensitive to differences in data collection (sampling designs and observation methods, Orth *et al.*, 1982; Bovee, 1986) and analysis (curve smoothing, Cheslak and Garcia, 1988), and in factors that affect fish habitat selection, including fish species and size (Probst *et al.*, 1984), fish behavior (Gosse and Helm, 1982), fish species composition (Schlosser, 1987), season and water temperature (Munther, 1970; Todd and Rabeni, 1989), streamflows (Bain *et al.*, 1982), population densities (Fraser and Sise, 1980), food availability (Wilzbach, 1985), and habitat availability (Orth *et al.*, 1982).

The effects of resource availability on use by organisms has been a long-standing problem in the fields of ecology (Ivlev, 1961; Johnson, 1980), and has been a major topic in the development of HSI where several methods have been employed to account for these effects (Baldridge and Amos, 1982; Rubin *et al.*, 1991). Although disagreement occurs over methods to account for habitat availability in HSI (Morhardt and Hanson, 1988; Bartholow and Slauson, 1990), most researchers acknowledge the potential biases inherent in HSI and many IFIM studies are required to develop site-specific HSI data, or to test the transferability of existing data to the locality in question (Thomas and Bovee, 1993).

This habitat suitability study was conducted on the mainstem Susquehanna River in eastern Pennsylvania at the site of a proposed dam expansion (Figure 1). The proposed development would increase water surface elevations of the Susquehanna by 5m at the dam site and would affect elevations and low-flow depth and velocity characteristics for approximately 7.5km kilometers upstream. The large size and unique habitat features of the study area posed concern over the applicability of published HSI curves for smallmouth bass (*Micropterus dolomieui*), which were developed from smaller streams with different habitat characteristics (Edwards *et al.*, 1983). The objective of this study was to evaluate the applicability of generic HSI criteria for smallmouth bass spawning and rearing for the Susquehanna study area using site specific data collected by unbiased sampling methodologies. HSI data were collected in the Susquehanna according to an equal-area, habitat stratified sampling scheme designed to account for the biasing effects of habitat availability. Direct observation of focal positions along randomly selected strip transects minimized sampling biases associated with observer preconceptions and measurement biases associated with imprecise identification of focal positions.

STUDY AREA

The study area included the mainstem Susquehanna River from the existing Dock Street Dam at Harrisburg, Pennsylvania, upstream approximately 7.5km to the Rockville Bridge (Figure 1). Limited sampling was also conducted in York Haven Pond located on the Susquehanna 19m downstream of the Dock Street Dam. In the Harrisburg area, the mainstem Susquehanna averaged 1.2km in width and exhibits mean monthly streamflows of 322 to 2,153 cubic meters per second. Despite it's width, the Susquehanna rarely exceeds 2.5m in depth during low-flow conditions and much of the river is less than 1m deep. The expansive shallows with emergent islands and low velocities produce dense beds of aquatic macrophytes (*Justica americana*).

The study area is a highly productive fishery for smallmouth bass, and fish exceeding 2kg are not uncommon in angler catches. Rock bass (*Ambloplites rupestris*), channel catfish (*Ictalurus punctatus*), walleye (*Stizostedion vitreum*), and muskellunge (*Esox masquinongy*) are other sought-after gamefish in the study area. Gizzard shad (*Dorosoma cepedianum*) is the principle forage species. The heavily wooded islands common to the study area provide important nesting habitats for many species of herons and egrets and provide complexity to the aquatic habitat (Figure 1).

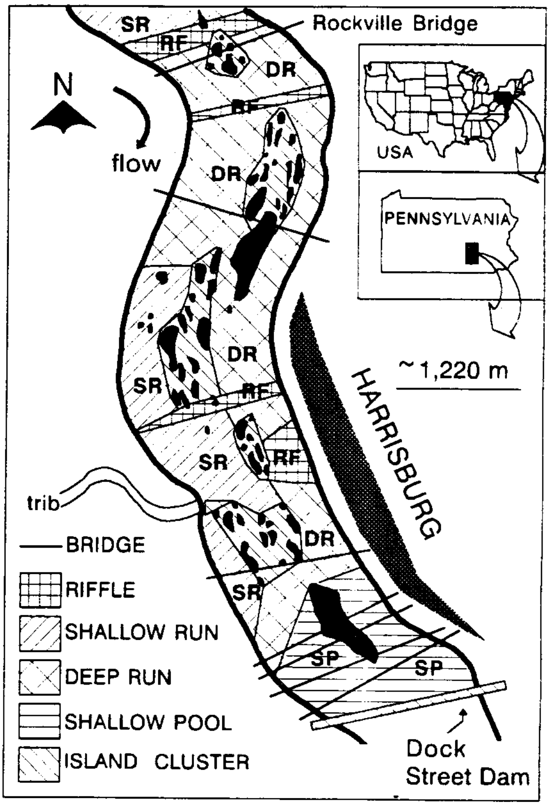


Figure 1. Susquehanna River study area used for the collection of smallmouth bass HSI criteria, 1989 and 1991.

METHODS

A minimum of fifty micro-habitat observations per life stage was desired to verify published HSI criteria for smallmouth bass (Edwards *et al.*, 1983). Randomly selected strip transects were sampled in May 1989 and in May and August 1991 within habitat-type strata by snorkeling or SCUBA diving to identify focal positions of spawning and rearing smallmouth bass.

Habitat Stratification

The project area was stratified into five macro-habitat types on the basis of depth, velocity, substrate, shoreline, and aquatic vegetation components (Figure 1). Aquatic habitats were generally classified as deep/slow (shallow pool), deep/fast (deep run), shallow/slow (shallow run), and shallow/fast (riffle). Island cluster habitat represented a heterogeneous mixture of the above habitats with a relatively large proportion of shoreline area. Because the project area lacked water exceeding 2m in depth, a sixth habitat stratum (deep pool) was selected from the lower end of York Haven Pond, a Susquehanna River impoundment located downstream of the project area. The inclusion of deep pool habitat was intended to cover a wider range of possible habitat types and to better represent probable post-project habitat conditions. Stratification allowed a controlled distribution of sampling effort with a reasonable assurance that all available habitat characteristics would be represented.

Equal-Area Sampling

The concept of equal-area sampling is based on the supposition that fish densities within a given habitat type will be governed by the suitability of that habitat type. By sampling with equal effort in all available habitat types, pooled HSI data will be weighted by the fish densities present within each type. Consequently, the habitats that are most suitable will be most represented in the pooled data. This methodology is based on several assumptions, including: 1) effective habitat stratification, 2) equal recruitment potential and survival between habitat types, and 3) equal observability between habitat types. The validity of the second assumption in a reach as large as the Susquehanna study area is unknown, however average distances traveled by radio-tagged smallmouth bass in the study area was between 2km and 4km during spring and summer months (Environmental Research and Consulting, 1993), and high flows during the spawning season were expected to distribute fry away from spawning area (Larimore, 1975). Equal recruitment between the mostly free-flowing project area and the impounded York Haven Pond was also assumed, although potential differences prevent a confident interpretation of deep water use by spawning or rearing bass.

Transect Selection

Spawning and rearing observations were made within each habitat strata along randomly located strip transects. The starting point of each transect was selected by locating random coordinate points on a detailed project map. Transect positions were located on-site by triangulating and rangefinding to geographic features. Each transect extended 90m to 150m perpendicular to flow. In 1989, transects were delineated with a weighted cable; in 1991 transects were sampled by following a compass heading.

Field Techniques

After a transect was delineated, a diver wearing snorkeling or SCUBA gear moved slowly along the transect line while searching for rearing fish or spawning nests. Swift water transects required the use of heavy dive weights, a hand-held grapnel anchor, and crampons (ice climbing cleats) to maintain position. Fish exhibiting an alarmed or unnatural behavior were not included in the analysis; focal positions of all other fish or spawning nests were marked with a weighted and numbered buoy. Size of each observed fish was estimated by reference to an underwater ruler. All spawning nests were inspected for the presence of eggs or fry. Fry were classified as sac fry (transparent and benthic), black benthic fry, or black suspended fry.

Focal positions of rearing fish were eye-estimated (height above the substrate), focal positions for all spawning nests were measured 15cm above the substrate. On transects requiring SCUBA, focal velocities were measured by the diver at the estimated focal height with a hand-held pygmy current meter attached to a one meter probe. Adjacent current velocities were also measured at focal height two feet to either side of observed nests. Substrate and instream cover characteristics were evaluated at each fish or nest observation point. Substrate was classified with a continuous code of dominant and adjacent size class composition (Table 1) at the focal position under the observed rearing fish or within the spawning nest. For spawning observations, substrate was also evaluated approximately one foot adjacent to the nest. Instream cover, if present, was identified by type with it's distance and direction from the focal position.

After completion of each transect, the boat or wading crew returned to each marker and measured total depth, mean column water velocity, and focal velocity (for shallow observations). Overhead cover and surface turbulence was also evaluated at each focal position. Additional data recorded at each transect were diver search width, water temperature, water clarity (measured with a Secchi disc), dive time, transect length and direction, weather conditions, and river stage (morning and evening).

Table 1. Type, size, and code of substrates used to describe focal and nest positions. Code includes the two most dominant size classes, with the percent compostion of the larger size class (i.e. a code 4.3 is 70% sand with 30% gravel; 6.8 is 20% cobble with 80% boulder. From Bovee and Cochnauer (1977).

Substrate Type	Size (mm)	Code
Vegetation		1
Mud / Clay	<.05	2
Silt	.05-.5	3
Sand	.5-6.4	4
Gravel	6.4-76.2	5
Cobble	76.2-304.8	6
Boulder	>304.8	7
Bedrock		8

RESULTS AND DISCUSSION

Effort Allocation

Table 2. Allocation of sample effort (in lineal meters of transect) by year, season, and habitat type.

	Spring Spawning		Summer Rearing		
Habitat Type	1989	1991	1989	1991	Totals
Riffle	305	945	0	610	1,860
Shallow Run	610	640	0	610	1,860
Deep Run	914	245	0	610	1,769
Shallow Pool	802	488	152	457	1,899
Deep Pool	0	1,067	0	762	1,829
Island Cluster	1,059	741	0	0	1,800
Totals	3,690	4,126	152	3,049	11,017

High flows and poor water clarity during the springs of 1989 and 1991 restricted most of the sampling effort to the middle third of the river channel, making equalization of effort within the macro-habitat strata difficult. In 1989, effort was largely allocated to island clusters and deep pool habitat was not sampled. Total spring sampling effort for the two years was approximately 1,250 lineal meters of transect per macro-habitat type, although deep pools were somewhat under-sampled and island clusters were over-sampled (Table 2). Summer sampling was largely restricted to 1991 due to a heavy algal bloom which occurred in 1989. Summer sampling was conducted along approximately 600 meters per habitat type, except for deep pools and island clusters which were sampled accordingly to produce a combined spring and summer effort of 1,800 meters per habitat type (Table 2). Spawning and rearing transects were distributed throughout most of the study area (Figure 2) and in York Hven Pond.

Environmental Conditions

River discharge differed substantially during the two spawning survey periods. In 1989, streamflows fluctuated between 1,421 and 809 cubic meters per second, whereas stages during the 1991 survey dropped steadily from 849 to 464 cubic meters per second. Water temperatures also differed between years. In 1989, water temperatures generally remained below 20°C. In 1991, a heat wave rapidly increased temperatures from 21°C to 24°C, which appeared to accelerate both spawning and incubation. The wide range of conditions sampled was considered to improve the quality of the data by increasing the variability of habitats available to spawning fish. Spring water visibilities were typically only one to two meters which restricted the location on non-spawning fish.

Summer rearing data was collected during stable streamflows of 105 cubic meters per second and a water temperature of 27°C. Visibility was generally less than three meters and sometimes less than two meters due to suspended organic matter and turbid inflow along the Harrisburg shoreline.

Smallmouth Bass Spawning

Fifty nest observations were collected for spawning smallmouth bass in the Susquehanna River. Twenty-two nests were observed in 1989 and another 28 in 1991, two of which were found in the deep water habitat at York Haven Pond (Table 3). Although transect survey methodologies were not designed to yield precise estimates of nest or fish densities (transect search widths were eye estimated), some comparisons between habitat types may be informative.

Overall, a mean of 0.63 nests were observed per 100m² in the Harrisburg and York Haven areas (Table 3). Estimated nest densities ranged from a low of 0.06/100m² in deep runs to a high of 1.29/100m² in island clusters. Despite the large difference in mean densities, transects with no nest observations were common in all habitat types.

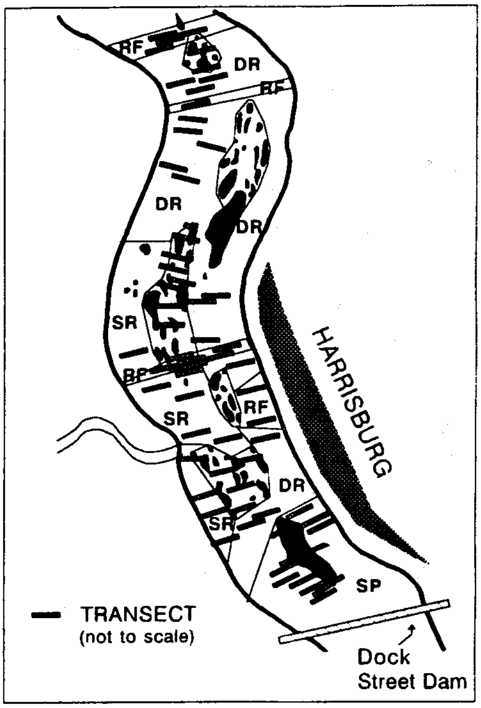


Figure 2. Approximate location of spawning and rearing transects within the study area.

Table 3. Total number and mean density (#/100m²) of smallmouth bass nests observed along spring transects, by habitat type.

Habitat Type	# Nests	Mean Nest Density	Density Range
Riffles	9	0.80	0 - 2.05
Shallow Runs	4	0.32	0 - 2.87
Deep Runs	1	0.06	0 - 1.02
Shallow Pool	10	0.69	0 - 2.58
Deep Pool	2	0.30	0 - 1.80
Island Clusters	24	1.29	0 - 5.13
Totals	50	0.63	0 - 5.13

Nest Activity

The majority of nests observed in 1989 contained eggs (20 of 22 nests), and no fry were observed. In 1991 four nests contained eggs and 21 nests contained fry in various stages of development (transparent sac-fry - 2 nests; black benthic fry - 11 nests; and black suspended fry - 8 nests). Three of the 1991 nests contained no eggs, fry, or guarding adult. However, the clean substrate within these nests suggested recent residence and therefore data were collected. Adult bass were observed in the proximity of 12 of 22 nests in 1989 and 13 of 28 nests in 1991. The actual proportion of nests with resident adults was probably underestimated due to disturbance by the diver. The estimated fork length of adult bass found over nests ranged from 15 to 50 cm in 1989 (mean=31.4cm) and 22 to 35 cm in 1991 (mean=27.6cm).

Nest Depth

Smallmouth bass nests observed in the Susquehanna ranged in depth from 37cm to 152cm and averaged 102cm in depth (Figure 3). The two York Haven nests occurred at approximately 260cm deep. Nests were typically deeper in 1989 (mean=106cm) than in 1991 (mean= 86cm), possibly due to the higher flows encountered that year. The Susquehanna data showed somewhat higher suitability of shallow water than the HSI curve suggested by the United States Fish & Wildlife Service (FWS, Edwards *et al.*, 1983). Nest observations in York Haven Pool indicated that deep water can be suitable for spawning, but data from the Susquehanna and other waters (Surber, 1943; Stone *et al.*, 1954; Cleary, 1956; Coble, 1975; Monahan, 1991) do not verify keeping suitability maximum to depths of 175 cm, or to infinity as suggested by the FWS clear water option (Edwards *et al.*, 1983).

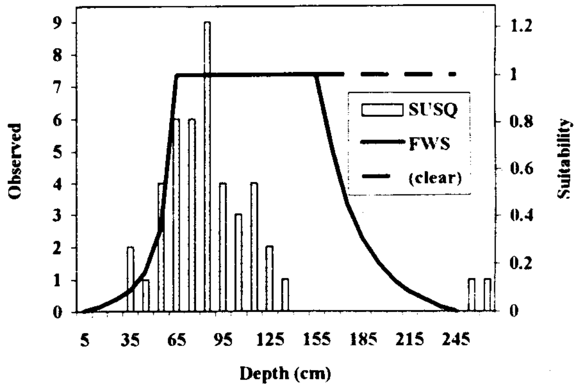


Figure 3. Depths measured at spawning nests, with FWS HSI curves.

Nest Velocity

Mean column velocities at nest sites ranged from zero to 58cm/s and averaged 16 cm/s (Figure 4). High flows during spawning in 1989 produced much higher nest velocities (mean = 26cm/s) than observed in 1991 (mean = 8cm/s). Despite the observed difference in mean column velocities, bottom velocities at nest sites were similar in 1989 (mean= 6cm/s) and in 1991 (mean = 4cm/s). The low and relatively constant bottom velocities measured at nest sites was largely due to the upstream presence of submerged cover. Smallmouth bass are known to require low bottom velocities for successful reproduction (Surber, 1943; Cleary, 1956; Coble, 1975; Pflieger, 1975; Simonson and Swenson, 1990). Differences in adjacent bottom velocities measured 60cm to the side of nest cavities in 1991 ranged from zero to 13cm/s (mean = 3cm/s), but were generally less than 5cm/s. The distribution of mean column velocities measured at Susquehanna River nests did not conform to the HSI curve suggested by the FWS (Edwards *et al.*, 1985), which indicates much higher suitability for velocities exceeding 25cm/s (Figure 4). In lotic systems, however, bottom velocities at many nest sites may be determined by the upstream presence of instream cover rather than by mean column velocities over the nest. This imprecise association between mean column velocities and bottom velocities, of which the latter is probably most critical for nest selection and success, makes prediction of spawning suitability difficult with current instream flow methodologies.

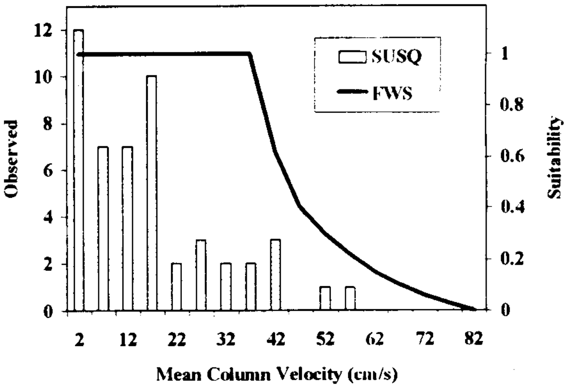


Figure 4. Mean clumn velocities measured at spawning nests, with FWS HSI curve.

Nest Cover

All but four of the 50 observed nests (92%) were associated with some form of instream cover (Figure 5). Most nests (78%) were located immediately downstream from the cover element. Cover types used by nesting bass were aquatic vegetation (46% - mostly water willow), woody debris and tree branches or roots (14%), boulders (24%), and cobbles (8%). Overhead and turbulence cover were present in 6% and 18% of all nest observations, respectively. Nests were typically located in close proximity to the cover element (mean distance =23cm). Sixty percent of observed nests were less than 20cm from cover (Figure 6).

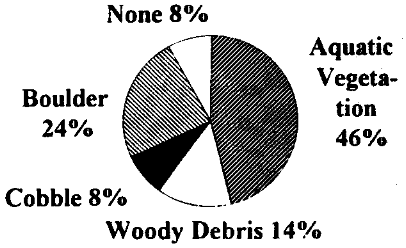


Figure 5. Percent composition of cover types found near smallmouth bass nests.

Close proximity to cover was found to be an important aspect of nest sites in both lotic (Simonson and Swenson, 1990) and lentic (Vogele, 1981) environments. Instream cover may provide protection from excessive velocities (see above) or provide security for fry (Livingstone and Rabeni, 1989) or spawning adults (Cleary, 1956).

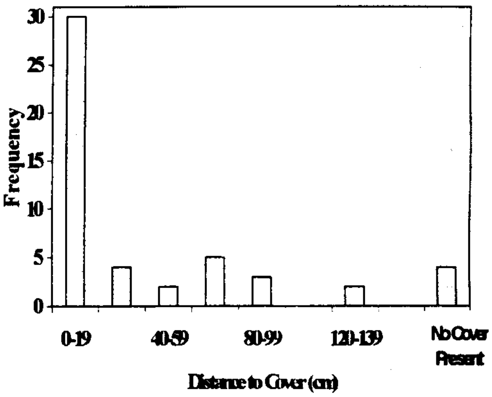


Figure 6. Distance measured from smallmouth bass nests to the nearest form of instream cover.

Nest Substrate

Gravel was the predominant substrate type within smallmouth bass nests, accounting for 56% of the observations (Figure 7). Cobble was dominant in 12% of nests, vegetation (water willow stems and roots) in 10%, and fines (sand or silt) in the remaining 22%. Differences in substrate composition within and adjacent to nests are readily apparent. The difference in nest and adjacent substrates illustrates the cleaning ability of spawning bass, and the need to account for underlying materials when evaluating substrate suitability (some nests were excavated in 5cm of silt). In general, the substrates found within Susquehanna nests were consistent with those described by the FWS HSI curve (Edwards *et al.*, 1985) and with other studies (Coble, 1975; Pflieger, 1975; Vogele, 1981; Monahan, 1991). Susquehanna nests showed more fines (silt) which accumulated in older and recently abandoned nests, and several nests were found with eggs attached to water willow roots.

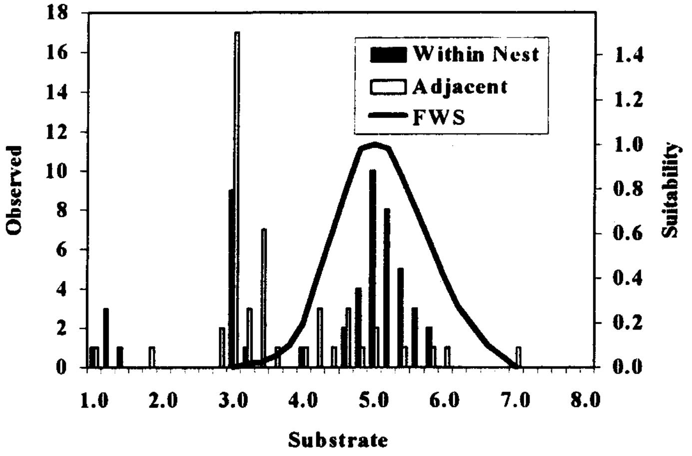


Figure 7. Substrate characteristics found within and one foot adjacent to smallmouth bass spawning nests.

Smallmouth Bass Rearing

Length Frequency

Four hundred and fifty-five rearing bass were observed during spring and summer surveys.

Observed bass ranged in length from 5cm to 46cm (Figure 8). The dominant size class was composed of young-of-year (YOY) bass (5 - 10.9cm); the remaining fish represented juvenile (*i.e.* yearling, 11-19.9cm) and adult bass (20+ cm). Because juvenile bass were frequently seen in close proximity to adult fish, and because juvenile and adult bass used similar depths and velocities, these fish were grouped together for further analysis (YLG+). Similar findings were made by Orth *et al.* (1982). HSI data were collected at focal positions of 196 YOY and 129 YLG+ bass.

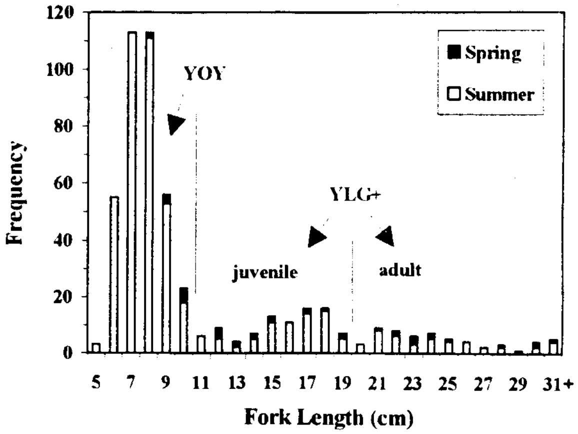


Figure 8. Length frequency of smallmouth bass observed in the Susquehanna River during spring and summer surveys, 1989 and 1991. Young-of-year (YOY, <11cm fork length) and juvenile+adult (YLG+, >10cm) life stage classifications are shown..

YOY or YLG+ bass were observed on all but two summer transects. Densities of YOY were far higher in riffles than in other habitat types (Table 4). Variability in YLG+ densities between habitat types was much less than observed with YOY. Overall densities were estimated at 10 YOY/100m^2 and 3 YLG+/100m^2. Qualitative estimates of YOY densities in Courtois Creek averaged approximately one-third of the Susquehanna value (Pflieger, 1975), however estimated YLG+ densities in the Susquehanna were lower than densities reported from many smaller rivers (Paragamian and Coble, 1975).

Table 4. Total number and mean density (#/100m^2) of YOY and YLG+ smallmouth bass observed along summer transects, by habitat type.

Habitat Type	# YOY	Mean YOY Density	Density Range	# YLG+	Mean YLG+ Density	Density Range
Riffles	242	30	10 - 56	34	4	0 - 8
Shallow Runs	54	7	0 - 22	4	1	0 - 1
Deep Runs	21	3	0 - 8	17	2	0 - 5
Shallow Pool	24	4	0 - 9	27	5	4 - 8
Deep Pools	10	1	0 - 3	22	3	2 - 3
Totals	351	10	0 - 56	104	3	0 - 8

Rearing Depth

The observed depth distributions of YOY and YLG+ bass were noticeably different. Mean depth of YOY focal positions was 53cm (Figure 9), whereas YLG+ fish were observed at a mean depth of 131cm (Figure 10). Although range in depth was similar, YOY frequently inhabited water less than 30cm deep and were uncommon at depths exceeding 150cm. In contrast, YLG+ were not observed in water less than 30cm but were relatively abundant in the deeper depths. The distinct shallow water peak for YLG+ at 35cm is composed of observations from riffle habitats and represents both juvenile and adult bass.

Comparison of Susquehanna data with FWS HSI curves for fry, juvenile, and adult bass show few similarities. Both FWS curves show low suitabilities for depths less than 40cm, where numerous YOY were observed in the Susquehanna. The practice of maintaining maximum suitabilities in deeper water for bass fry does not appear valid for the Susquehanna River. Despite considerable effort in greater depths, few fry were found in the presence of larger

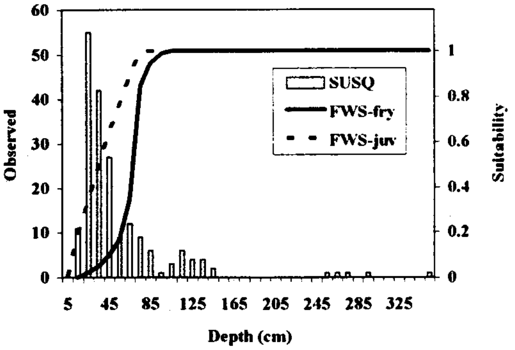


Figure 9. Depths measured at YOY focal positions, with FWS HSI curves.

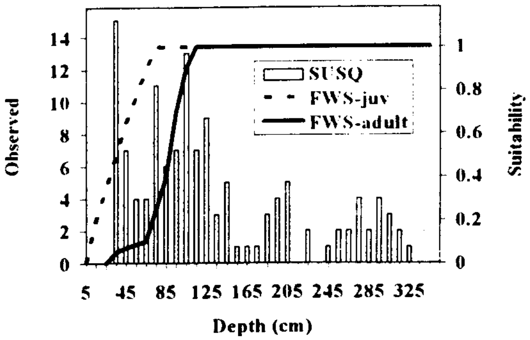


Figure 10. Depths measured at YLG+ focal positions, with FWS HSI curves.

bass. Other HSI studies have also suggested low suitability of deeper water for fry (Bain *et al.*, 1982; Monahan, 1991).

The FWS HSI curves for juvenile and adult bass also appear to under-rate the importance of shallow water, where numerous bass of both size classes were observed holding focal positions in riffle habitats (Figure 10). These FWS curves were derived from electrofishing data, which might be expected to disturb large fish away from such exposed positions. Despite the use of electrofishing, Orth *et al.* (1982) also captured numerous adult bass in water less than 50cm. The suitability of shallow water for adult bass may be enhanced in larger river systems, as many of the Susquehanna observations were made hundreds of meters away from the nearest streambank and its associated community of predators. Turbulence cover was also present at many of these shallow water observations. The FWS practice of keeping depth suitability at maximum for juvenile and adult bass may be valid for the Susquehanna. Studies in other large rivers have shown substantial use of deep water by adult bass (Munther, 1970).

Rearing Velocity

Mean column velocities measured at focal points averaged 15cm/s for YOY bass and 20cm/s for YLG+ bass (Figures 11 and 12). The frequency distributions were fairly similar, with larger fish somewhat more abundant in faster water.

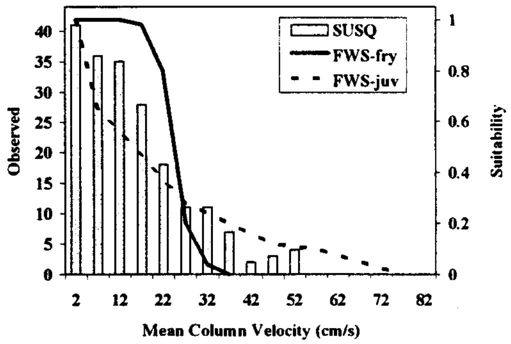


Figure 11. Mean column velocities measured at YOY focal positions, with FWS HSI curves.

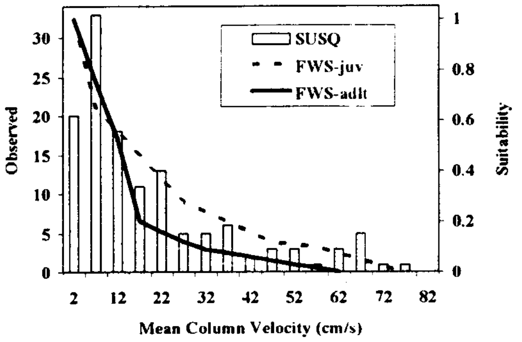


Figure 12. Mean column velocities measured at YLG+ focal positions, with FWS HSI curves.

The observed similarities were probably due to the relatively large size of the YOY bass, most of which exceeded seven cm in length. Relatively few YOY or YLG+ were found in velocities exceeding 50cm/s, which is consistent with findings in other rivers (Orth *et al.*, 1982; Rankin, 1986; Todd and Rabeni, 1989; Monahan, 1991). Several studies have shown that velocities less than 25cm/s are optimum for bass young-of-year (Bain *et al.*, 1982; Simonson and Swenson, 1990). The FWS curve describing velocity suitabilities for fry (Figure 11) matches the Susquehanna data fairly well until approximately 30cm/s, at which point the FWS juvenile curve shows a more similar trend. Because bass often select locations behind submerged objects, these fish can find suitable habitat in mean column velocities thought to exceed critical levels. The observed use of velocities by Susquehanna YLG+ was closely matched by the FWS HSI curve for juvenile bass (Figure 12).

Rearing Cover

Young-of-year and juvenile/adult bass were frequently observed in close association with instream cover (Figure 13). The cover association was particularly strong with the YOY population, where only 7% of observations were made in the absence of observable cover (Figure 14). In contrast, 48% of observed YLG+ were observed away from cover. Rock substrates, such as cobbles, boulders, or bedrock ledges, were the most common form of instream object cover observed near focal positions. Instream vegetation (mostly water willow) was the closest source of object cover for 12% of YOY and 6% of YLG+ bass. Woody debris or tree roots were seen near 5-7% of observations. Overhead cover in the form of surface turbulence or overhanging objects was observed at only 2% and 6% of all YOY and YLG+ observations, respectively.

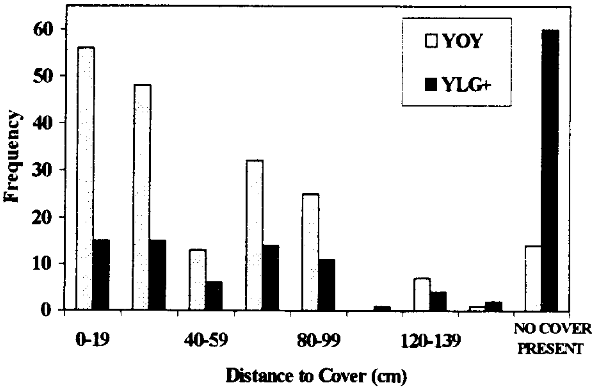


Figure 13. Distance measured from focal positions of YOY and YLG+ bass to the nearest form of instream cover.

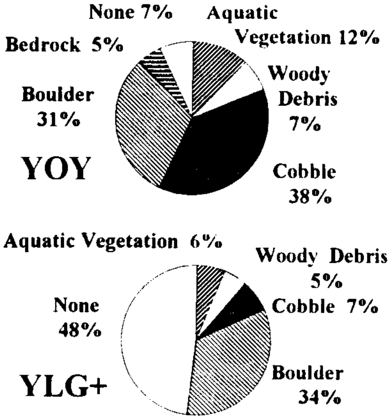


Figure 14. Percent composition of cover types observed near YOY and YLG+ focal positions.

Cover is frequently cited as an important habitat component for smallmouth bass (Probst *et al.*, 1984; Todd and Rabeni, 1989). In large rivers such as the Susquehanna, woody debris and other riparian-related cover types are relatively rare and many fish rely on rock substrates and aquatic vegetation for protection from predators or excessive velocities (Munther, 1970; Simonson and Swenson, 1990).

Rearing Substrate

Young-of-year smallmouth bass were most frequently seen over sand and gravel substrates, but cobble, boulder, and bedrock were also used (Figure 15). YLG+ bass used a similar range of substrates with a somewhat greater use of the fine substrates characteristic of deeper water (Figure 16). Sand and gravel substrates were more dominant in the Susquehanna data than found in other studies (Munther, 1970; Orth *et al.*, 1982; Rankin, 1986; Todd and Rabeni, 1989, Monahan, 1991). The FWS HSI curves for fry and juvenile/adult bass also emphasized higher suitabilities of larger substrate materials. Some of these discrepancies may be due to the procedures used to classify substrate. In the Susquehanna data, substrate was characterized within a 30cm radius of focal positions, which may not have accounted for the presence of larger materials further away. YOY, for instance, were seldom found away from cover components, which often included nearby cobbles and boulders (Figure 14)..

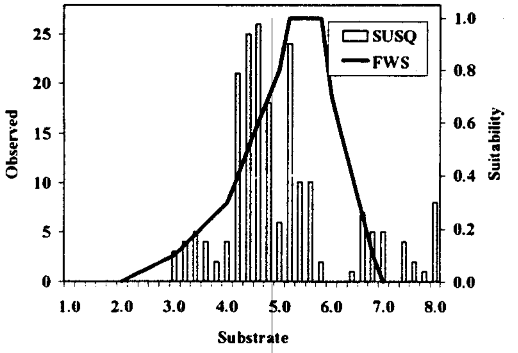


Figure 15. Substrate characteristics beneath YOY focal positions, with FWS fry HSI curve.

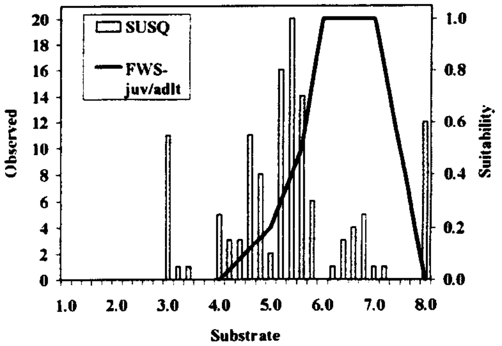


Figure 16. Substrate characteristics beneath YLG+ focal positions, with FWS HSI curve.

LITERATURE CITED

Bain, M.B, J.T. Finn, L.J Gerardi, M.R. Ross and W.P. Saunders. (1982). An evaluation of methodologies for assessing the effects of flow fluctuations on stream fish. United States Department of Energy, Oak Ridge National Laboratory, Oak Ridge, Tennessee. DOE/DOI-F&WS-20733-1. 199 pp.

Baldridge, J.E. and D. Amos. (1982). A technique for determining fish habitat suitability criteria: a comparison between habitat utilization and availability. Pp. 251-258 In N.B. Armantrout (editor). *Acquisition and utilization of aquatic habitat inventory information.* Portland, Oregon. 376 pp.

Bartholow, J. and W. Slauson. (1990). Comments: Questions on habitat preference. *North American Journal of Fisheries Management.* 10 p. 362.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. *Instream Flow Information Paper No. 21.* United States Fish and Wildlife Service. Biological Report 86(7). 235 pp.

Bovee, K.D. and T. Cochnauer. (1977). Development and evaluation of weighted criteria, probability-of-use curves for instream flow assessments: fisheries. *Instream Flow Information Paper No. 3.* United States Fish and Wildlife Service, FWS/OBS-77/63. 38 pp.

Cheslak, E.F. and J.C. Garcia. (1988). An evaluation of the effects of various smoothing and curve-fitting techniques on the accuracy of suitability functions. Pp 259-283 In K. Bovee and J.R. Zuboy (editors). *Proceedings of a workshop on the development and evaluation of habitat suitability criteria.* United States Fish and Wildlife Service, Biological Report 88(11). 407 pp.

Cleary, R.E. (1956). Observations on factors affecting smallmouth bass production in Iowa. *Journal of Wildlife Management.* 20(4) pp. 353-359.

Coble, D.W. (1975). Smallmouth bass. Pp 21-33 In H. Clepper (editor). *Black bass biology and management. National symposium on the biology and management of centrarchid basses.* Sport Fishing Institute, Washington, D.C.

Edwards, E.A., G. Gebhart and O.E. Maughan.(1983). Habitat suitability information: smallmouth bass. United States Fish and Wildlife Service, FWS/OBS-82/10.36. 47 pp.

Environmental Research and Consulting. (1993). Movement and habitat use of radio-tagged smallmouth bass in the Susquehanna River near Harrisburg and in York Haven Pool, Pennsylvania. Environmental Research and Consulting, Chadds Ford, Pennsylvania.

Fraser, D.F. and T.E. Sise. (1980). Observations on stream minnows in a patchy environment: a test of a theory of habitat distribution. *Ecology.* 61(4) pp. 790-797.

Gosse, J.C. and W.T. Helm. (1982). A method for measuring microhabitat components for lotic fishes and its application with regard to brown trout. Pp. 138-149 In N.B. Armantrout (editor). *Acquisition and utilization of aquatic habitat inventory information.* Portland, Oregon. 376 pp.

Ivlev, V.S. (1961). Experimental ecology of the feeding of fishes. Yale University Press, New Haven, Connecticut.

Johnson, D.H. (1980). The comparison of usage and availability measurements for evaluating resource preference. *Ecology.* 61(1) pp. 65-71.

Larimore, R.W. (1975). Visual and tactile orientation of smallmouth bass fry under floodwater conditions. Pp. 323-332 In H. Clepper (editor). *Black bass biology and management. National symposium on the biology and management of centrarchid basses.* Sport Fishing Institute, Washington, D.C.

Livingstone, A.C. and C.F. Rabeni. (1989). Food habit relations which determine the success of young-of-year smallmouth bass in Ozark streams. Abstract in *First International Smallmouth Bass Symposium.* Nashville, Tennessee.

Modde, T. and T.B. Hardy. (1992). Influence of different microhabitat criteria on salmonid habitat simulation. *Rivers.* 3(1) pp. 37-44.

Monahan, J.T. (1991). Development of habitat suitability data for smallmouth bass (*Micropterus dolomieui*) and rock bass (*Ambloplites rupestris*) in the Huron River, Michigan. M.S. Thesis, Michigan State University, East Lansing, Michigan. 130 pp.

Morhardt, J.E. and D.F. Hanson. (1988). Habitat availability considerations in the development of suitability criteria. Pp 392-403 In K. Bovee and J.R. Zuboy (editors). *Proceedings of a workshop on the development and evaluation of habitat suitability criteria.* United States Fish and Wildlife Service, Biological Report 88(11). 407 pp.

Munther, G.L. (1970). Movement and distribution of smallmouth bass in the Middle Snake River. *Transactions of the American Fisheries Society.* 99 pp. 44-53.

Orth, D.J., R.N. Jones and O. Eugene Maughan. (1982). Considerations in the development of curves for habitat suitability criteria. Pp. 124-133 In N.B. Armantrout (editor). *Acquisition and utilization of aquatic habitat inventory information.* Portland, Oregon. 376 pp.

Paragamian, V.L. and D.W. Coble. (1975). Vital statistics of smallmouth bass in two Wisconsin rivers, and other waters. *Journal of Wildlife Management.* 39(1) pp. 201-210.

Pflieger, W.L. (1975). Reproduction and survival of the smallmouth bass in Courtois Creek. Pp. 231-239 In H. Clepper (editor). *Black bass biology and management. National symposium on the biology and management of centrarchid basses.* Sport Fishing Institute, Washington, D.C.

Probst, W.E., C.F. Rabeni, W.G. Covington and R.E. Marteney. (1984). Resource use by stream-dwelling rock bass and smallmouth bass. *Transactions of the American Fisheries Society.* 113 pp. 283-294.

Rankin, E.T. (1986). Habitat selection by smallmouth bass in response to physical characteristics in a natural stream. *Transactions of the American Fisheries Society.* 115 pp. 322-334.

Rubin, S.P., T.C. Bjornn and B. Dennis. (1991). Habitat suitability curves for juvenile chinook salmon and steelhead using a habitat-oriented approach. *Rivers.* 2 pp. 12-29.

Schlosser, I.J. (1987). The role of predation in age- and size-related habitat use by stream fishes. *Ecology.* 68(3) pp. 651-659.

Simonson, T.D. and W.A. Swenson. (1990). Critical stream velocities for young-of-year smallmouth bass in relation to habitat use. *Transactions of the American Fisheries Society.* 119 pp. 902-909.

Stone, U.B., D.G. Pasko and R.M. Roecker. (1954). A study of Lake Ontario-St. Lawrence River smallmouth bass. *New York Fish and Game Journal.* 1(1) pp. 1-26.

Surber, E.W. (1943). Observations on the natural and artificial propagation of the smallmouth black bass. *Micropterus dolomieui. Transactions of the American Fisheries Society.* 72 pp. 233-245.

Thomas, J.A. and K.D. Bovee. (1993). Application and testing of a procedure to evaluate transferability of habitat suitability criteria. *Regulated Rivers: Research and Management.* 8 pp. 285-294.

Todd, B.L. and C.F. Rabeni. (1989). Movement and habitat use by stream-dwelling smallmouth bass. *Transactions of the American Fisheries Society.* 118 pp. 229-242.

Vogele, L.E. (1981). Reproduction of smallmouth bass, *Micropterus dolomieui*, in Bull Shoals Lake, Arkansas. U.S. Fish and Wildlife Service, Technical Paper 106. Washington, D.C. 15 pp.

Wilzbach, M.A. (1985). Relative roles of food abundance and cover in determining the habitat distribution of stream-dwelling cutthroat trout (*Salmo clarki*). Canadian Journal of Fisheries and Aquatic Sciences. 42 pp. 1668-1672.

SELECTION OF AN ECOLOGICAL REFERENCE FLOW: SUPPORT BY MODELING THE HABITAT OF A KEY SPECIES, THE NORTHERN PIKE

Olivier Monfort
Cemagref, division Qualité des Eaux
14, avenue de Saint-Mandé
75012 Paris, France

Pascal Breil
Cemagref, Hydrologie/Hydraulique
3 bis quai Chauveau
C.P. 220, 69336 Lyon 09
France

Philippe Boet
Cemagref, division Qualité des Eaux
14, avenue de Saint-Mandé
75012 Paris, France

ABSTRACT

The regulation by dams and reservoirs of the upper reaches of the Seine and its major tributaries, the Aube and Marne rivers, has greatly modified the natural flow characteristics of the river by reducing the periods and durations of recurring floods. The impacts of these regulation works on the ecosystem are numerous; they mostly affect the structure of fish communities.

This paper attempts to provide elements for more ecological management practices of these hydraulic works, by proposing modelisation of the reproduction habitat of pike (*Esox lucius*), a species highly representative of this type of river and its functionnal relations with the flood plain. Pike favours flooded grasslands, closely linked to the importance of annual floods, as spawning grounds. The idea is to use pike as a reference species to adapt the French version of PHABSIM.

The method requires a global assessment of the quality and number of aquatic habitats on the Seine near Pont-sur-Seine, which are available for the first development stages of pike, and then produce a model to estimate the potential carrying capacity or weighted usable areas (WUA) under different flow conditions, by coupling measurements of water level, flow speed and vegetation cover with pike habitat preference curves. Results have revealed, in terms of WUA, the impacts of regulation structures between 1966 and 1989.

KEY-WORDS: Fish / Habitat / Dams / Management / Seine / Pike

INTRODUCTION

This paper results from a six month study by a graduate student. The objective of the study was to define a preliminary ecological reference flow in a river by modeling the habitat of a key species, the northern pike. The biological cycle of the pike, especially the reproduction phase and early stages of development, requires a variety of specific habitats such as cut-off meanders, oxbows, flooded grasslands, etc. The occurrence and good functionning of these habitats regulate the production and survival rate of pike larvae. Pike is an interesting biological model reflecting the ecological conditions of the river, particularly because it uses the relations between primary and secondary river beds.

Our study contributes to a better comprehension of the hydrosystem and assesses the impacts of flow regulation in the upper reaches of the Seine hydrographic basin. The Seine and Aube dams and reservoirs, in operation since 1966 and 1990 respectively, were designed and commissionned to supply drinking water to Paris, to maintain minimum low-water levels and to control recurring floods. Their current operation mode induces changes in natural flow characteristics, a decrease of flood durations and, ultimately, a generalized disturbance of the hydrosystem.

This study was conducted in the course of research projects of the "Corridor fluvial" group of PIREN-SEINE; the global objective of the research projects is to integrate the interactions between the river and the various habitats along the alluvial plain within the ecological models of the Seine hydrographics.

DAMS AND RESERVOIRS ON THE SEINE

Constructed and managed by the "Institution Interdépartementale des Barrages-réservoirs du Bassin de la Seine (IIBRS)", or "Grands Lacs de la Seine (GlS)", these works have a double objective: flood control and maintenance of minimum low-water flows, while supplying sufficient drinking water reserves for the metropolitain Paris, as a predominant concern. These objectives may appear contradictory. In fact, the dams stock water from December to June (by topping eventual flood peaks), and then release water from July to November following a rule of reservoir ranking (translated by a "filling-flushing" curve). Depending on the operation mode, the flow characteristics of the river are modified and the natural discharge, to which biological components have adapted, is disturbed. Established on the upper reaches of the Seine and of its tributaries, Aube and Marne, the dams and reservoirs greatly affect the natural flow characteristics.

KEY SPECIES: PIKE

Pike prefers flooded natural grasslands for spawning. It uses wetlands, ditches, small grass-filled brooks in the alluvial plains, and cut-off meanders in contact with the river during winter and spring floods (February to April). Thus, pike require easy access to and from these associated zones. The occurrence and proper functionning of these zones regulate pike larvae production and survival rate. Pike is thus a good indicator of proper lateral water exchanges with the main river bed.

Therefore, the study of pike reproduction should inform on the biological potential of flooded plains along the Seine and highlight eventual misfunctions of the river system. This justifies our selection of pike as the reference species.

STUDY AREA

An adequate study site was required to validate our method. Pont-sur-Seine is located 150 km from Paris. This site comprises 7 km of river, bordered upstream by Périgny-la-Rose and downstream by the town of Nogent-sur-Seine. On a ten-kilometer segment, the river has remained largely natural despite hydraulic works and farming on

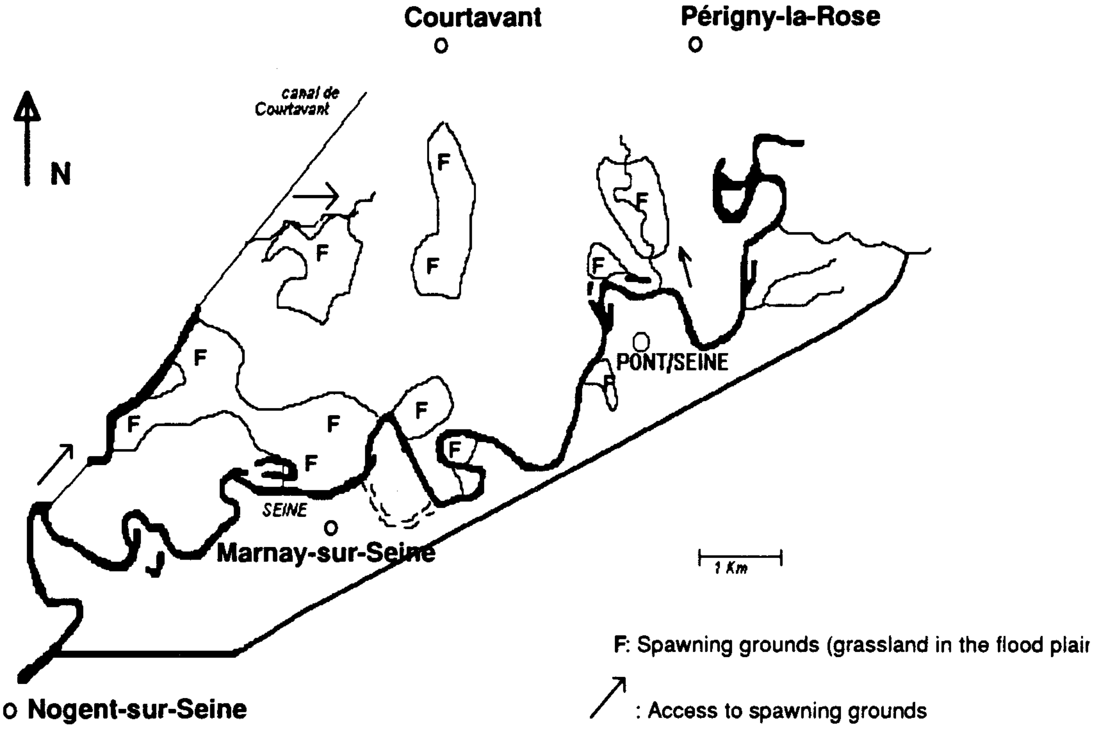


Figure 1 : Diagram of the main potentiel pike spawning grounds at Pont-sur-Seine on the Seine

a portion of the territory. The main river bed remains a flood expansion zone and there is a functionnal lateral component of the hydrosystem. Cut-off meanders, oxbrows and flooded grasslands are abundant. Vegetation along the main river bed comprises oakwoods, poplar forests, farmlands and grasslands in sufficient proportions to provide adequate pike reproduction habitats (Figure 1). In addition, water quality is considered good in the Pont-sur-Seine area.

The total surface area of the plain is approximately 2912 ha, 7.3 km long by 4 km wide.

MODELIZATION OF PIKE REPRODUCTION HABITAT

Modelization of the reproduction habitat of pike, a key species in this type of river environment, was attempted. It consisted in adapting the PHABSIM method to a river in its flood plain using pike as a reference species (Figure 2).

The method consists in coupling a hydraulic model and a biological model in order to quantify, in relation with discharge values, the potential carrying capacity of a stream section for a given species at different stages of development (life cycle): spawning, fry, juveniles, adults. In our case, we studied the potential areas adequate for

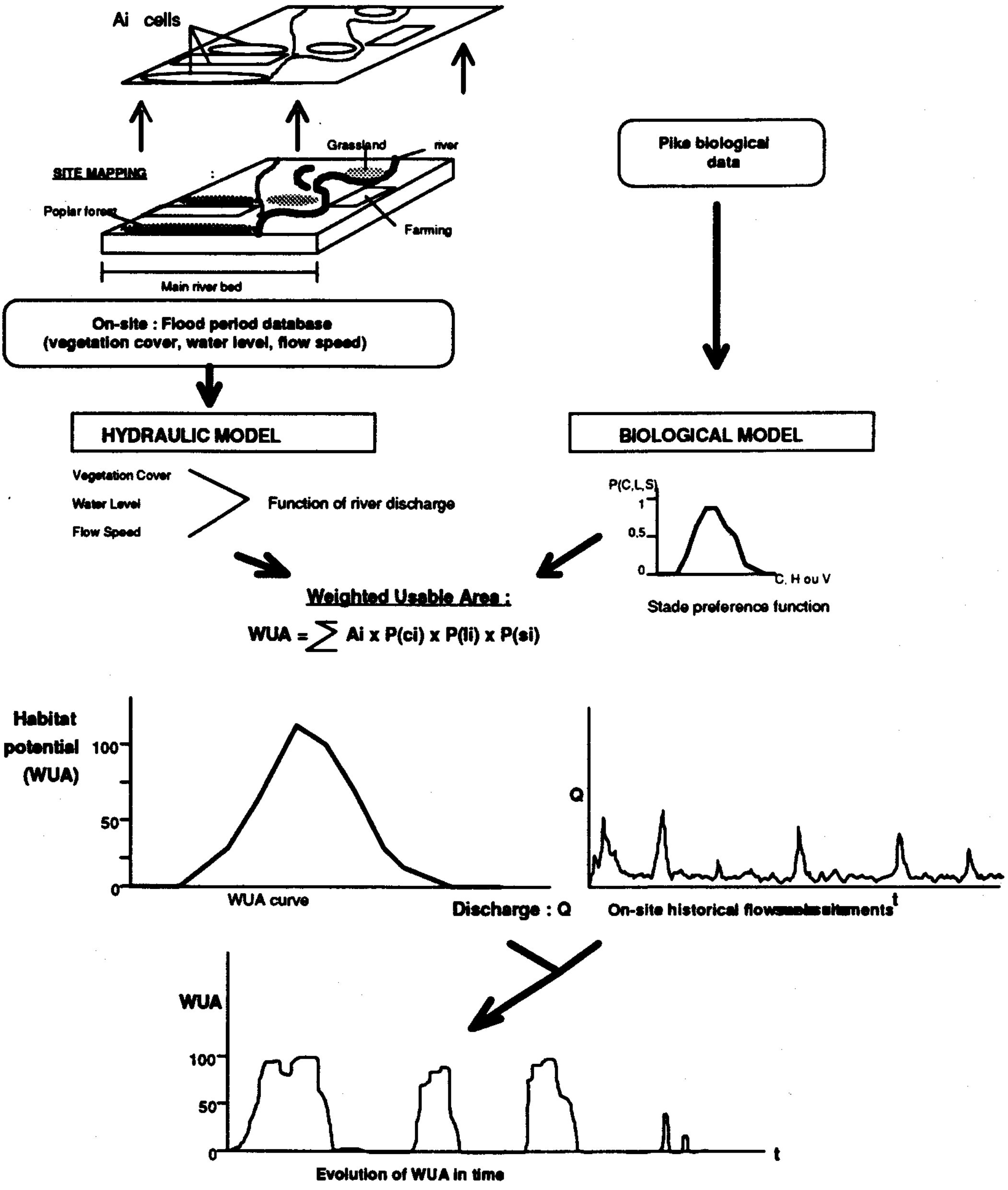


Figure 2: Diagram of the pike reproduction habitat model

pike reproduction.

Globally, the goal is to provide quantitative assistance for the selection of an optimal reference flow which satisfies pike requirements while protecting the interests of riverfront residents. At this scale, very accurate values are not essential; it is preferable to provide values relating to various scenarios in order to find a sound ecological reference flow.

Pike biological model

A biological model essentially reflects the relationship between relative density of the species and some habitat variables provided by the hydraulic model. The biological criteria used to establish the pike preference curve were selected on the basis of a literature review. The criteria retained are: nature of the vegetation cover (C), water level (L) and flow speed (S); although water temperature is critical during the embryonic development, it was not retained because it is considered an uncontrollable parameter influenced by weather conditions. Water quality is globaly satisfactory in the area; therefore it was not taken into account in the model.

For each value of the variables, preference coefficients, between 0 and 1, are attributed.

Vegetation cover

Pike is somehow an opportunistic spawner and vegetation cover is not of utmost importance at spawning sites. However a substrate lined with submerged terrestrial vegetation is prefered over a substrate of tree stumps and dead wood. These preferences are based on Fortin *et al.* (1982) who studied the mean egg density laid on different substrates: grasslands, shrubby and herbaceous vegetation, forest with herbaceous vegetation, forest with a leaf litter, emerging aquatic vegetation, etc.

The maximum preference coefficient is granted to grasslands; for other substrate, values are calculated by a simple rule of three (Table 1 and Figure 3).

Table 1.Influence of the vegetation cover on the density of eggs laid (from Fortin; 1982) and preference coefficient.

Vegetation cover	Egg density(n/m^2)	Preference coefficient
Grassland	2.12 to 6.98	1
Shrubs with herbaceous stratum	3.40 to 4.67	0.8
Forest with herbaceous vegetation	0 to 2	0.2
Forest with dead leaf litter	0+	0
Emergent aquatic vegetation	1 to 2.34	0.3
Bare or ploughed soil	0	0

Water level

During the first two stages of life cycle, pike is vulnerable to water level variations, especially to level drops threatening essential habitats. Immobile eggs and larvae are particularly exposed. At these stages, pike prefers water levels between 30 and 50 cm. Conversely, alevins are mobile and prefer water levels between 40 and 60 cm.

Two curves are drawn to account for these elements (Figure 3), assuming a decrease by 30% of the preference coefficient for each 10 centimeter layer on either side of the optimal water levels for each stage (Table 2).

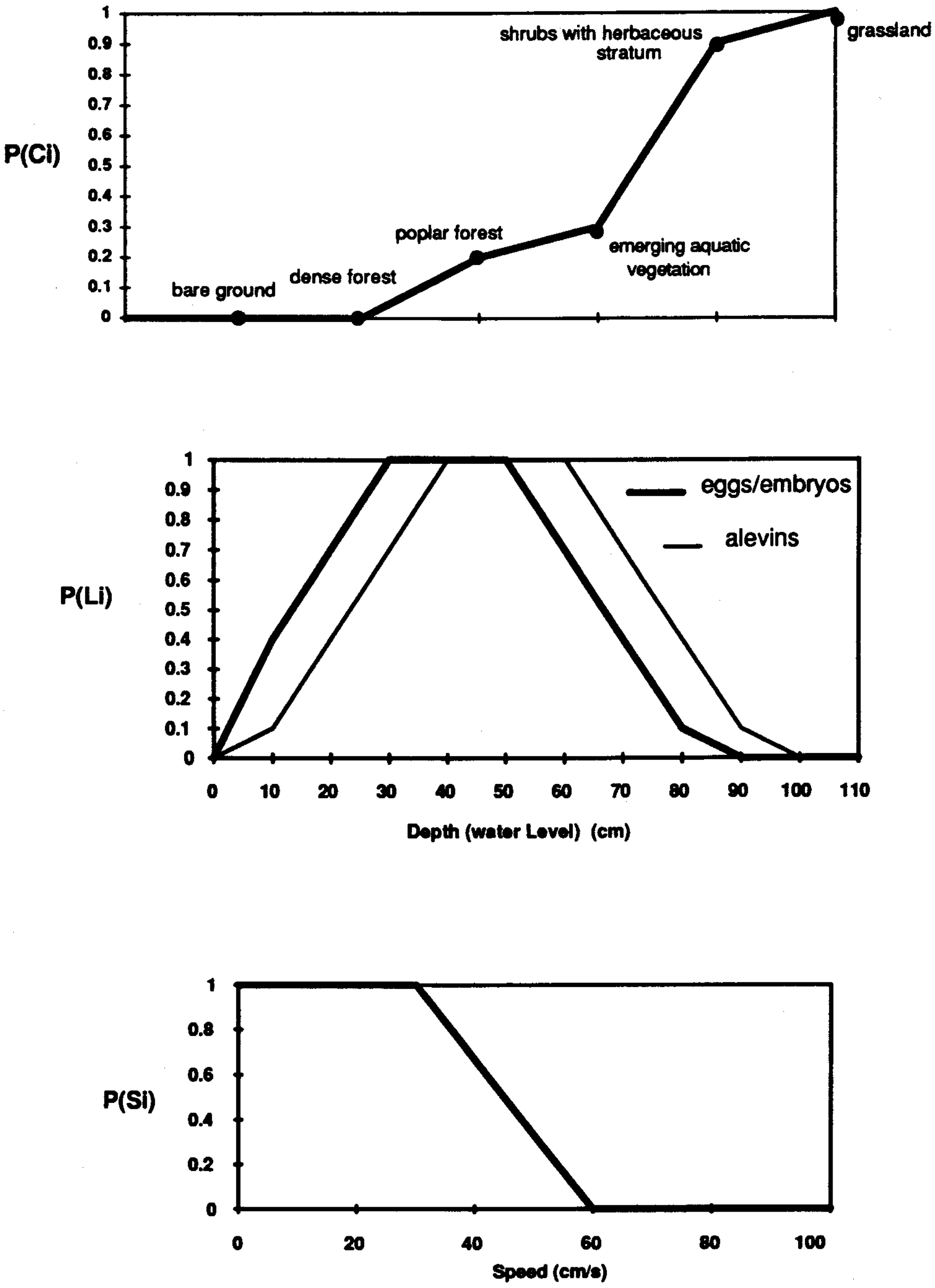


Figure 3 : Pike preferences curves for vegetation cover, water level and flow speed.

Table 2.Preference coefficients of the pike first two stages of development for different water levels.

Water level (cm)	**0**	**10**	**20**	**30**	**40**	**50**	**60**	**70**	**80**	**90**	**100**
Eggs/embryos	0	0.4	0.7	1	1	1	0.7	0.4	0.1	0	0
Alevins	0	0.1	0.4	0.7	1	1	1	0.7	0.4	0.1	0

Flow speeds

In its first stages of development, pike prefers slow moving water. According to Inskip (1982), the ideal flow speed lies between 0 and 5 cm/s. However, speeds between 5 and 30 cm/s are still adequate for efficient spawning and normal embryos and alevins development. Pike reproduction is hampered by flow speeds above 30 cm/s.

The speed preference curve (Figure 3) considers 60 cm/s as the upper limit above which pike spawning does not occur at the site.

Hydraulic model

This step consists in defining the values of morphodynamic variables (L, C, S) and their variations within the flood plain along the main river bed at Pont-sur-Seine, in relation with river discharges.

Relationship between discharge and flooded surfaces

The potential flooded surfaces between immediately downstream from Conflans-sur-Seine and Nogent-sur-Seine were estimated for 5 discharge ranges (175, 303, 422, 250 and 608 m^3/s) from several sets of aerial photographs taken by IGN in the course of the impact assessment study of the EDF generating station at Norgent-sur-Seine, combined by a study by SOGREAH (1983). For each discharge, observed surfaces were plotted on a contour map, scale 1:25 000, and measured with a digital planimeter. The total surface area of the studied plain is approximately 2912 ha, 7.3 km long by 4 km wide (Table 3).

Table 3.Evolution of flooded surface in relation with river discharge

Discharge (m^3/s)	**100**	**175**	**250**	**303**	**383**	**422**	**608**
Flooded surfaces (ha)	0	734	1712	2311	2562	2612	2769
% of flood plain	0	25	58	79	87	89	95

Habitat variable estimates

The flood plain was then divided according to vegetation cover. This produces a mosaic picture made of elementary zones, called cells, for which the surface and the three variables (C, L, S) are measured at different discharge values.

Vegetation cover was compiled from a 1:25 000 cartography done in 1983 by the Muséum National d'Histoire Naturelle and updated by recent observations made by geographers working on Corridor Fluvial (Levassor *et al.* 1995), and by a ground-truthing experiment conducted at the end of the 1995 summer. Along the main river bed, four major vegetation covers were identified: grassland, oakwood, poplar forest and farmland. Homogenous zones were determined for each type of cover and their surfaces were measured with a digital planimeter. The evolution of flooded surfaces of each vegetation cover was then measured in relation with river discharge (Table 4). This was accomplished by superimposing, for each discharge value, the map of flooded surfaces on the map of vegetation cover.

Table 4.Evolution of grassland and poplar forest flooded surfaces in relation with river discharge.

Discharge (m^3/s)	**100**	**175**	**250**	**303**	**383**	**422**	**608**
Grassland (ha)	0	216	235	238	238	238	238
Poplar forest (ha)	0	213	590	760	831	834	856

The relationship between water level and river discharge could not be established only on the basis of the depths measured at various points located near the main channel and in the flood plain (SOGREAH, 1983). These measurements were extrapolated to the whole plain from the 1:25 000 IGN map. Taking into account the topography, surfaces considered homogenous in altitude were delimited around the measurement points. Thus, water level was estimated by "zones". However, these estimates were in part supported by water level measurements taken on January 7 and 8, 1994, during a peak flood period, for a discharge of 288 m^3/s along a transverse transect between Pont-sur-Seine and Courtavant (Levassor *et al.* 1994). For each discharge value, a map of water levels (1:25 000) was drawn.

Flow speed - Data on flow speed are scarce. Only a few flow speed measurements along the transect between Pont-sur-Seine and Courtavant are provided by Levassor *et al.* (1994). A mean flow speed of 1.88 m/s was measured in the Seine secondary bed during a discharge of 288 m^3/s. Flow speed can reach 1.5 m/s in the major bed drainage ditches, which are privileged flow axis. Everywhere else along the transect, flow speeds are below 0.3 m/s, not limiting pike spawning and egg hatching. Conversely, when the discharges exceed 400 m^3/s (SOGREAH 1983), flow speeds are above the tolerance threshold for the survival of pike eggs.

Because of the lack of data available, only three classes of values are considered. From 100 to 300 m^3/s discharges, flow speed is assumed to be below 0.3 m/s over most of the flood plain. Between 300 and 400 m^3/s, flow speeds increase and reach between 0.3 and 0.6 m/s, limiting pike reproduction. Above 400 m^3/s, egg survival and development are impossible, with the exception of a few remote sites protected by dense rugged vegetation.

Model coupling

Data on the characteristics of the flood plain in relation with the river discharge are processed in a hydraulyc model coupled with the biological model. The resulting hydraulic parameters are used to measure the surfaces of habitat usable by fish (Weighted Usable Area = WUA). For each i cell, defined from similar vegetation cover, WUAi equals the surface Ai multiplied by the preference coefficients read on the preference curvers P(Ci), P(Li) and P(Si):

$$WUAi = Ai \times P(Ci) \times P(Li) \times P(Si)$$

For a given discharge, the total WUA for the study area equals the summation of all WUAi:

$$WUAtot = \Sigma WUAi$$

RESULTS

This calculation for different discharges provides a WUA (Q) curve for the study area (Figure 4). Two WUA (Q) curves are drawn, respectively related to the first two stages of development of pike (eggs/embryos and alevins); but since the two curves are similar, only the first one is used in the model.

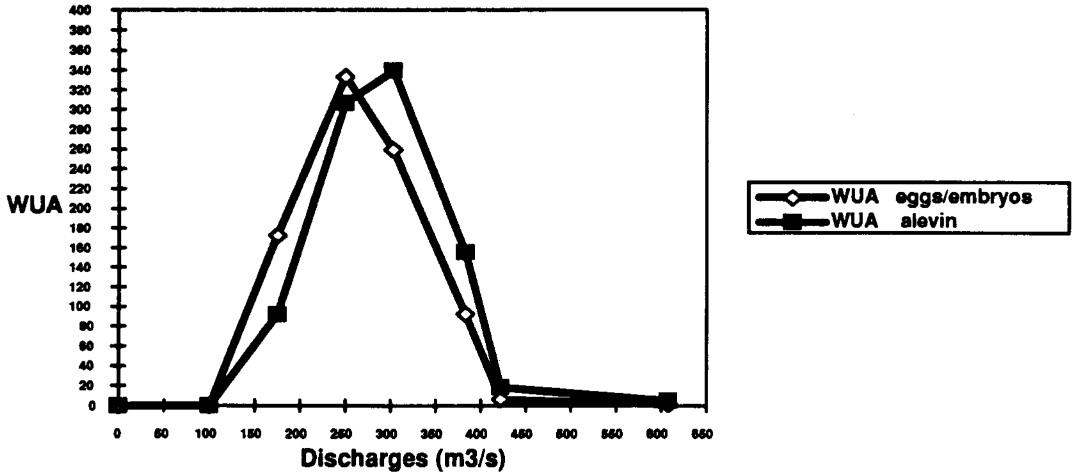


Figure 4 : WUA(Q) curve for the study site.

The maximum WUA is reached for a discharge of 250 to 300 m^3/s. The lower limit of 100 m^3/s corresponds to the bankful discharge. Above 400 m^3/s, the site is not suitable for pike reproduction; water levels and flow speeds at these discharges do not meet the habitat requirements for pike reproduction.

Optimum discharge

The evolution of grassland flooded surfaces in relation with river discharges shows that a maximum of 235 ha is reached at a discharge of 250 m^3/s (Table 4). But such a large area may not be biologically essential. Although mature fish present in the river all year around prefer flooded grasslands for spawning, fish abundance is necessarily limited in the river and it is likely that there is a threshold beyond which additional flooded grasslands are superfluous. Pike density assessments in lakes are commonly found in the literature but rather rarely for river populations; also data available often include all individuals, adults and juveniles (Table 5).

Table 5. Pike density estimated in various environments (from J.P. Roat, 1988).

Location	**Density (nb/ha)**	**Reference**
Wisconsin Lakes (Poland)	6.2 to 7.4	Priegel and Krohn (1975)
River Pilica (Poland)	26	Penczack, Zalewski and Molinski (1976)
River Pilica (Poland)	11 to 12	Mann and Penczack (1984)
Oulujoki River (Finland)	4.5	Lind and Kaukoranta (1975)
River Nene (England)	200	Hart and Pitcher (1973)
River Stour (England)	21 to 127	Mann (1980)
River Frome (England)	127 to 227	Mann (1980)
Mean	56 - 86	

We shall consider a density of 4.5 adults/ha (adults are 3^+ years), although this is a low density when compared to other environments. This value leads to an estimated 189 mature fish at the study site, where the submerged river bed is approximately 42 ha (60 m mean width and 7 km long). Since the sex-ratio is usually 1 in nature, 94 females are potentially present. With 8 females/ha of flooded grasslands (Inskip 1982), only 11.8 ha of grasslands are actually required.

In fact, about 238 ha of grassland are found over the whole study area (Table 4) and the vegetation cover does not appear to be a limitation factor. For a discharge of 175 m^3/s, 216 ha of grasslands are flooded, almost the total surface. Reproduction habitats are then available for about 80 adult pikes per ha of river surface. This value is close to the mean density reported in table 5; it even seems rather high. A discharge of 175 m^3/s would thus be sufficient, since the increase in grassland flooded surfaces at greater discharges does not imply a proportionnal increase in site carrying capacity for spawning pikes.

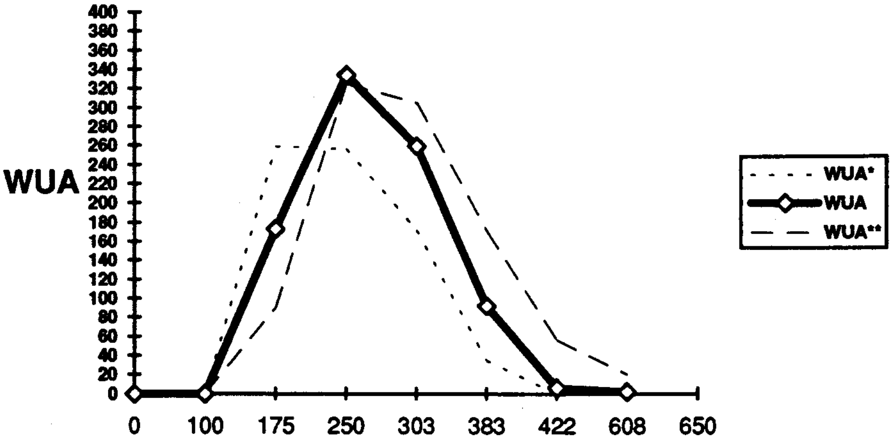


Figure 4' : Comparison of different WUA curves (WUA* : water level at -10 % , WUA : water level at + 10 %)**

Sensitivity analysis

First, the "weight" of each morphodynamic vairable (vegetation cover (C), water level (L) and flow speed (S) in the drawing of the WUA(Q) curve was assessed. For this purpose, we calculated two new WUA curves: a first, WUA (Q, C, L) using only C and L; a second, WUA (Q, C, S) using only C and S. The resulting curves were compared to the WUA (Q) curve which integrates all three variables. The WUA (Q, C, L) curve is shaped like the WUA (Q) curve, while the WUA (Q, C, S) curve is very different, particularly for higher discharges. Therefore the "weight" of the variable flow speed is low in the development of the WUA (Q) curve. Conversely, the water level variable is dominant in the calculation of the WUA (Q) at the Pont-sur-Seine site.

We then proceeded with a sensitivity analysis on water levels at + and -10% of the value on the curve. The resulting WUA (Q) curve shape are similar to the mean WUA (Q) curve (Figure 4').

Utilization of discharge time series

A 33-year discharge time series (1960-1993) was compiled from available information in the Hydro database for three stations: Arcis-sur-Aube, Méry-sur-Seine and Pont-sur-Seine (Figure 5). The geomorphological and rain conditions are similar all over the Seine and Aube watershed, above their confluence. This situation is expressed by similar hydrographs at the two stations; the frequency of floods is the same, only the intensity differs. Consequently, floods recorded at Pont-sur-Seine are "simple floods" (Figure 6).

It has been possible to recreate the discharges at Pont-sur-Seine for the period 1960-1965, from data recorded at Méry-sur-Seine or at Arcis-sur-Aube. The following linear regressions were calculated:

$$Q(\text{Pont}) = 2.4\ Q\ (\text{Méry})$$
$$Q(\text{Pont}) = 2.2\ Q\ (\text{Arcis})$$

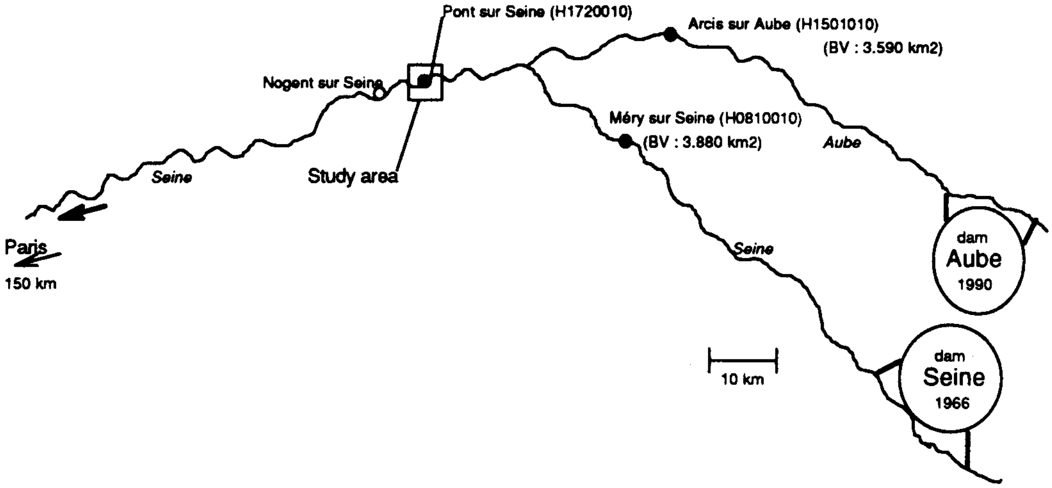


Figure 5 : Location of gauging stations

The impact of river regulation by the Seine and Aube dams-reservoirs, respectively commissionned in 1966 and 1990, can be assessed with these two relations, by recreating different types of discharge series at Pont-sur-Seine between the two dates:

-natural flow characteristics, simulated by the relation Q(Pont) = 2.2 Q(Arcis) which corresponds to the Aube flow characteristics prior to 1990;
-flow charactéristics regulated by a single dam, which corresponds to the actual flow measured at Pont-sur-Seine since the commissionning of the Seine dam in 1966;
-flow characteristics regulated by two dams, simulated by the relation Q(Pont) = 2.4 Q(Méry).

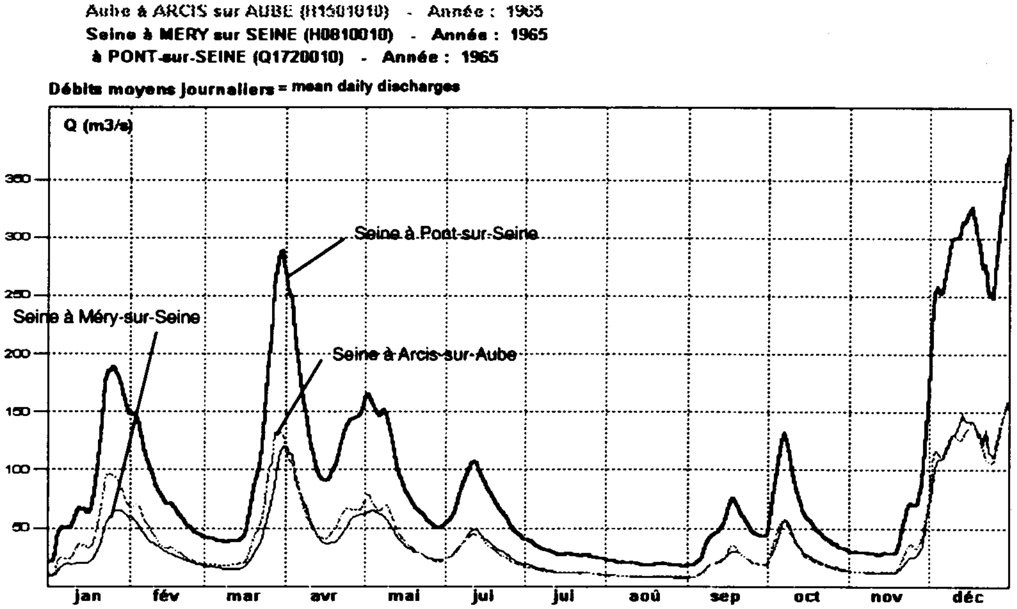


Figure 6 : Simple floods recorded at Pont-sur-Seine

The discharge time series Q(t) are then coupled with the pike WUA(Q) curve to get WUA(Q) time series for a temporal analysis. Since the period suitable for pike spawning and the first two life stages spans from mid-February to mid-April, only this period (60 days) is considered (Figure 7).

The resulting WUA(t) curves indicate strong variations which make difficult direct interpretation in terms of habitat available to pike (Figure 7). The WUA(+) time series must be classified and adjusted: for each year, daily WUA during the period considered are ranked by decreasing order. This representation supplies a measure of the time span (%) during which a given value of WUA is exceeded, during the period considered (Valentin 1995).

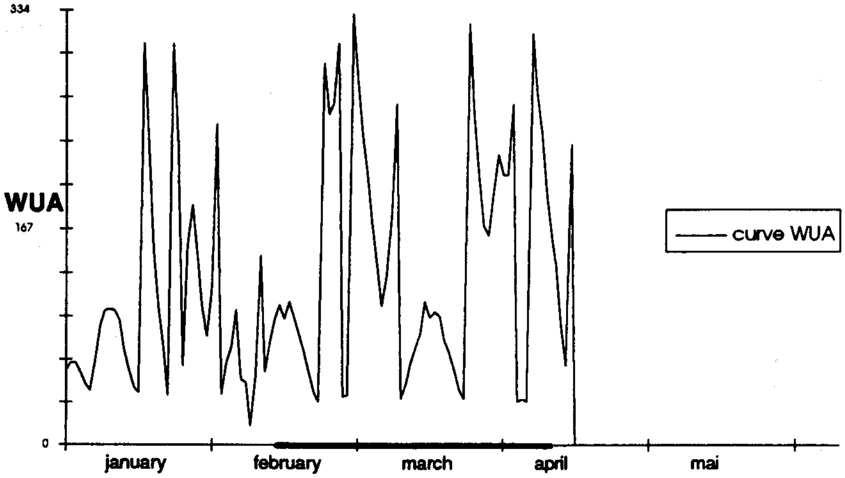


Figure 7 : Example of a WUA(t) time serves in natural flow conditions - year 1980 -

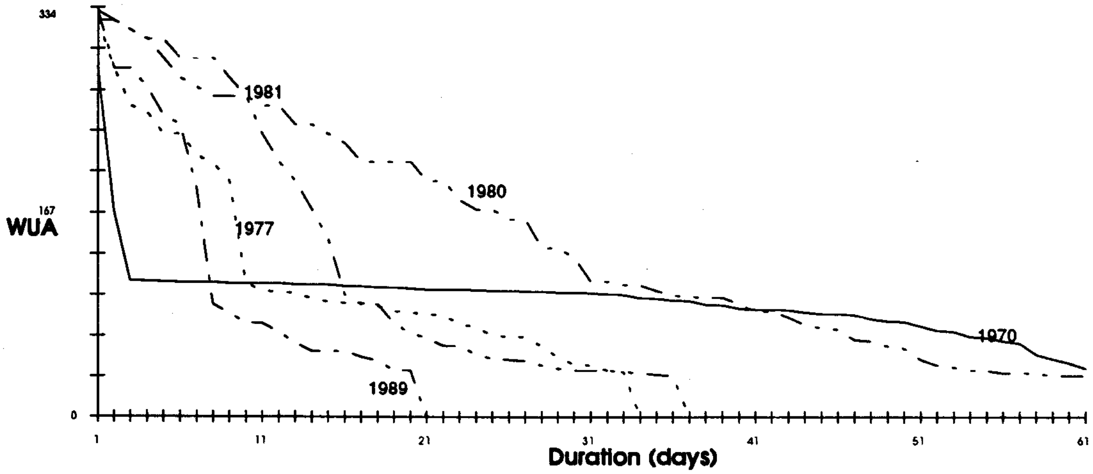


Figure 8 : Diagram of the influence of interannual variations on ranked WUA(t) curves.

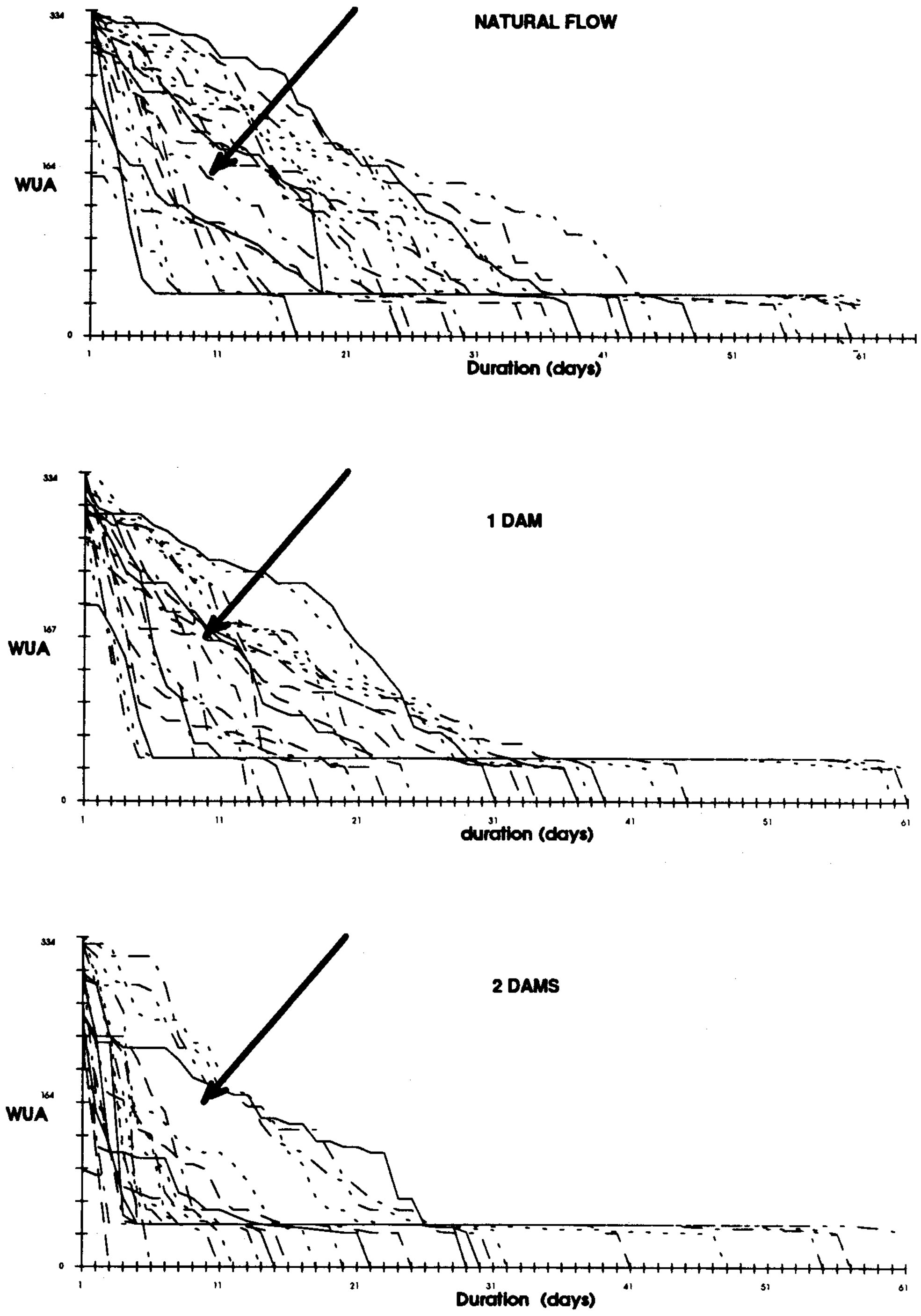


Figure 9 : Comparison of ranked WUA(t) curves between 1966 and 1989 for various discharge simulations.

Because of natural climate variations, these curves are naturally very different from year to year (Figure 8). For example under simulated natural flow characteristics, the 50 WUA value -half of the maximum WUA- is reached or exceeded during 27 days (45% of duration) in the year 1980, while it is reached for only 2 days (3% of duration) in 1970. Thus, interannual variations mat be very important.

Nevertheless, the effect of regulation works can be assessed by comparing the resulting curves for the various discharge simulations (natural, one dam, two dams) between 1966 and 1989. When compared to simulated natural conditions, a significant and general decrease of the ranked- WUA values is recorded for flow characteristics with 1 and 2 dams (Figure 9). In addition, from one suitable year out of two for pike reproduction under natural conditions, the regulation by two dams results in one suitable year out of five (right end of the curves) and, more important, in four unsuitable years (left end of the curves) inexistant before.

PERSPECTIVES AND CONCLUSIONS

The objective of this study was to develop a method to assess the evolution of the "physical habitat" of a flooded plain for a key species (pike) under different flood discharge conditions. Applied to the Seine flood plain at Pont-sur-Seine, this research may appear academic but, in fact, an ecological reference flow was quantified, which also answers, in part, strong social concerns.

Of course, there are several uncertainties associated with field data. Compared to the microhabitat method (PHABSIM), the level of precision required at this scale is not as high. However, the change in scale implies necessarily to consider the morphodynamic complexity of the river.

Applying this modified PHABSIM has produced satisfactory results; pike is a suitable species for this adapted method.

However, this experiment concerns a specific study area and results are valid only in this context. Trying to apply this method to other sectors suitable for pike (Marne, Oise...) over the whole watershed remains delicate because of problematic field data collection.

Finally, the next step is the biological validation. At this scale, it is impossible to obtain a fine validation such as that done on small streams (Pouilly, 1995). In our case, uncontrolled variability factors are numerous ("dry" year, temperature, etc.). Only on the long-term can we hope to harmonize pike densities with optimized management practices at dams and reservoirs.

REFERENCES

Fortin, R., Dumont, P. et Fournier, H. (1982). Reproduction et force des classes d'âge du grand Brochet (Esox lucius L.) dans le Haut Richelieu. *Can.J.Zool* pp.227-240.

Inskip, P.D. (1982). Habitat suitability index models : northern pike. *U.S. Fish and Wildlife Service.* FWS/OBS-82/10,17. pp.1-40

Levassor, A. et al. (1994). Les corridors fluviaux dans le bassin de la Seine. PIREN-Seine, thème "corridor fluvial" .Rapport 1993/II.

Pouilly, M., Valentin, S. (1995). Méthode des microhabitats : principes et protocoles d'application. *Bull.Fr. Pêche Piscic.* (n° 337/338/339). pp.329-336.

Raat, J.P. (1988). Synopsis of biological data on the northern pike Esox lucius. *FAO Fisheries Synopsis*.(30).Rev.2

Valentin, S.(1995). Variabilité artificielle des conditions d'habitat et conséquences sur les peuplements aquatiques : effets écologiques des éclusées hydoélectriques en rivière.(52-95). Université Claude Bernard-Lyon pp284.

Habitat modeling

Modélisation des microhabitats

Perspectives on Two-Dimensional River Habitat Models: the PHABSIM Experience

Ken D. Bovee
National Biological Service
4512 McMurry Avenue
Fort Collins, Colorado 80525
USA

ABSTRACT

Despite the apparent superiority of two-dimensional hydraulics, the real advantage of this technology lies in the realm of spatially explicit habitat models. Two-dimensional habitat models will enable biologists to develop "top-down" habitat suitability criteria by examining patterns of habitat availability and use at a mesohabitat scale. Spatially explicit models also allow the calculation of different measures of environmental heterogeneity, which may explain variations in the structure and function of riverine communities. An exciting potential use of two-dimensional habitat models is the ability to describe spatial and temporal heterogeneity in a single metric.

Obstacles to the widespread use of two-dimensional technologies include user inertia, lack of biological validation, lack of a consumer-support infrastructure, and unreasonable expectations. The intended users of this technology are managers and planners, who view new technologies as means to an end, not researchers who see the technology as an end in itself. This user community has invested a large amount of time and energy into the use of one-dimensional methodologies (e.g., PHABSIM) and a certain amount of disinterest or resistance to two-dimensional technologies can be anticipated. Acceptance of two-dimensional technologies will be facilitated by biological validation and the development of a user-support infrastructure. Biological validation is necessary to help decision-makers interpret the consequences of a change in a habitat metric. If two-dimensional models do not provide better biological information than existing one-dimensional models, there may be little incentive to adopt their use. The existing procedures for developing two-dimensional models are unstructured and confusing. Managers and planners are intolerant of user-hostile software, and may not attempt to use two-dimensional models unless they are provided with intelligible documentation, training, technical assistance, and software support. The research and development community should be enthusiastic about two-dimensional hydraulic and habitat simulation. However, we must be careful to avoid overselling the technology, making unsubstantiated claims about its capabilities, or fostering unrealistic expectations among ourselves and our clients.

INTRODUCTION

For the past two decades, the Physical Habitat Simulation System (PHABSIM) has been the mainstay of microhabitat simulations in the Instream Flow Incremental Methodology (IFIM). Recently, a number of authors (e.g., Leclerc et al. 1995; companion articles in these proceedings) have suggested replacing or augmenting PHABSIM with two-dimensional hydraulic and habitat models. Two-dimensional models have great potential to become major advancements in the state-of-the-art of stream habitat assessments. However, there may exist a number of unrecognized obstacles to be overcome in order for the technology to advance from a tool with great potential to an operational reality. Despite a litany of limitations, a host of vocal detractors, and a tendency to be obtuse and user-hostile, PHABSIM has been a useful and popular river habitat assessment tool. Our experiences in developing, supporting, and validating PHABSIM provide a unique perspective with respect to the opportunities and potential pitfalls involved with the maturation of a habitat assessment technology. Experience is said to be a wonderful thing because it allows us to recognize mistakes when we make them again. The purpose of my discussion is to share our experiences related to the development of PHABSIM, with the hope that we can take advantage of the opportunities of two-dimensional habitat modeling, but to recognize our mistakes before we make them.

THE OPPORTUNITIES

Many of the papers presented in these proceedings extoll the advantages of two-dimensional hydraulic models compared to their one-dimensional counterparts. I do not intend to devote much time or space on hydraulics, except to agree in principle with most of the assertions made by the promoters of two-dimensional hydraulic models. It seems intuitive that we should be able to simulate streamflow characteristics in complex channels much better with two-dimensional models. Islands and braided streams should not present the challenge with two-dimensional models that they do with one-dimensional hydraulics. We can even dispense with disputes over the ability to describe rivers with transects because two-dimensional models do not need to use them (although we may have merely advanced this argument into another dimension).

Despite the great potential of two-dimensional hydraulic models, I submit that the real advancement of the state-of-the-art will occur through the adaptation of two-dimensional habitat models. It is reasonable to expect that two-dimensional habitat models will provide better estimates of the amount of microhabitat for a target species (e.g., WUA in PHABSIM), simply because they provide a more accurate depiction of the stream. If this were the only advantage, however, it might not be much of an improvement over one-dimensional approaches. The true value of two-dimensional habitat models lies in the fact that they are spatially explicit, a characteristic that will allow the use of a whole suite of new habitat metrics.

Spatially Explicit Habitat Models

The definition of additional or new habitat types with a spatially explicit habitat model is made possible by habitat mapping (Figure 1). A habitat map can be used to quantify the area of habitats that are directly important to a species, but it also shows all of the other types present in the habitat mosaic. We postulate that the mosaic of different habitat types may be of greater importance in some communities than the abundance of any single habitat type. We know, for example, that there exist many combinations of habitats that are important to stream fauna: (a) eddy currents adjacent to high velocity chutes (Rinne 1991; Freeman and Grossman 1993), (b) velocity shelters associated with high velocity currents, used by fishes ranging from salmonids to paddlefish (Fausch and White 1981; Southall and Hubert 1984), (c) mid-channel pools or runs that border pools containing structural cover, important to smallmouth bass (Rabeni and Jacobson 1993), (d) pools abutting gravelly riffles, a combination used by spawning redhorse suckers (Kwak and Skelly 1992), (e) connections between floodplain wetlands and main channel habitats, critical for successful recruitment of endemic species of the Colorado River system (Stanford 1994), (f) shallow, low-velocity areas with emergent vegetation adjacent to higher velocity riffle or pool habitats, extensively used by young-of-year fishes (Lobb and Orth 1991, Rabeni and Jacobson 1993, Bovee et al. 1994).

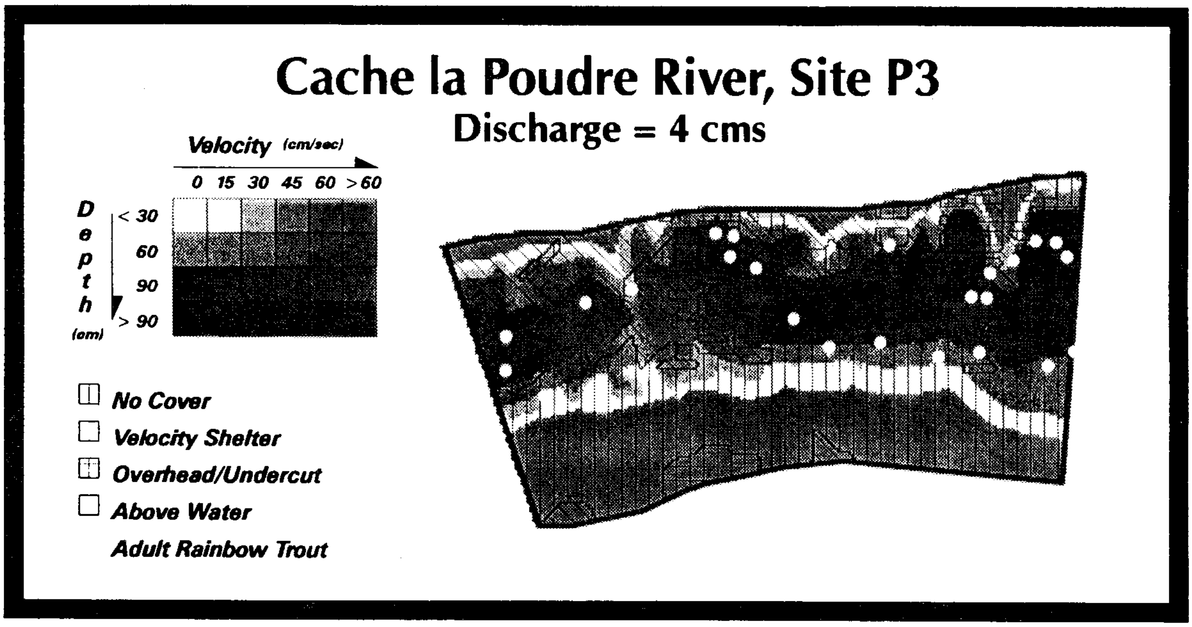


Figure 1: Spatially explicit habitat map of a shallow pool in the Cache la Poudre River, Colorado, at a discharge of 4 m^3/sec. Patterns of occupation by adult rainbow trout (white dots) show an association with the edges of pools, structural cover, and high-efficiency feeding stations.

Some habitat combinations might best be quantified by the amount of contact, or edge, between habitat types. The length of interface, (e.g., between eddy and high velocity currents) may give an index to the potential for exchange of animals and resources between habitats. In fact, the contact zone between two different habitat types may constitute a unique habitat type in itself (Ranney et al. 1981, Noss 1983, Lovejoy et al. 1986, Harris 1988, Yahner 1988, Malcolm 1994), wherein edge effects near the contact create habitat and community characteristics of neither of the adjoining habitats. Quantifying habitat juxtaposition, extent of edges and habitat connectivity may provide habitat-based measures of ecosystem function in streams, but will require spatially explicit habitat models.

The spatial and temporal variability of habitat can be collectively described within the rubric of patch dynamics (Pringle et al. 1988; Townsend 1989). Li and Reynolds (1994) recognize five components of spatial heterogeneity that may be linked to functional responses of species: (1) the number of patch types, (2) the proportion of each type, (3) spatial arrangement, (4) patch shape, and (5) contrast between neighboring patches. The number of patch types is related to habitat diversity, with more patch types indicative of higher resource diversity. The proportion of each type may determine the dominance or lack of critical resources or habitat types. The spatial distribution of resources may affect species dispersal, colonization potential, or foraging efficiency. Patch size and shape reflect the relative importance of edge effects; small, irregular patches will be more affected by edge phenomena than large, simple patches. Contrast between neighboring patches may modify the effects of edges (either by accentuation or attenuation), affecting the capability of the species to disperse. Contrast between neighboring patches may be potentially useful in determining indexes of fragmentation and connectivity. In addition, the health of an aquatic community can be affected by the relationships between aquatic and terrestrial ecotones, by large-scale spatial relations among habitats (especially as they affect movement), and by the availability of refugia from harsh environmental conditions (Schlosser 1995). A few representative examples of patch indexes that can be obtained from two-dimensional habitat models are summarized in Table 1.

Table 1: Spatially explicit habitat metrics commonly used in landscape ecology.

Habitat Metric	Description
Fractal dimension (D) (Burrough 1986; O'Neill et al. 1988)	Describes the complexity of shapes on the landscape. A mosaic of simple geometric shapes will result in a small fractal dimension, but D will be large if the landscape contains many complex, irregular patches.
Dominance (D1) (O'Neill et al. 1988)	Measures the extent to which one or a few patch types dominate a landscape, expressed as the deviation of patch proportions from the most evenly distributed mosaic possible). In landscapes dominated by few patch types, D1 will be large, but in areas containing many equal-sized patches, D1 will approach zero.
Contagion (D2) (O'Neill et al. 1988; Turner et al. 1989; Li and Reynolds 1994)	Represents the degree to which patches are clumped or aggregated, and can be used to evaluate *anisotropism*, or directionality in aggregating patterns. Contagion is expressed as the probability that a unit of patch type, *I*, will be adjacent to a unit of patch type, *j*, relative to a maximum in which all adjacency probabilities are equal. High values of D_2 indicate a mosaic of large, contiguous patches, whereas low values depict a highly dissected patchwork.
Edge effect Malcolm (1994)	Can be used to represent juxtapositions (e.g., the length of edge of habitat type x in contact with habitat type y) or patch complexity (e.g., as the ratio between perimeter and area of a patch). The area affected by an edge may also be calculated, so that the edge effect may be considered a habitat type in its own right.

"Top-down" Habitat Suitability Criteria

Historically, habitat suitability criteria for PHABSIM have been developed following a "bottom-up" strategy: data from observations of individual fish were extrapolated upward to represent populations, and even entire species where databases were especially thin. The following illustrates the general procedure for developing habitat suitability criteria for depths utilized by adult rainbow trout, according to a bottom-up sampling strategy (Bovee 1986):

1. By snorkeling or other observational technique, 100-200 locations in the stream, occupied by adult rainbow trout, are marked with a numbered tag or buoy.
2. The depth at each marked location is measured to the nearest 3-5 cm.
3. A histogram is developed by tallying the frequencies of adult rainbow trout associated with each depth increment.
4. By a variety of techniques (e.g., running means, non-linear maximum likelihood functions, freehand drawing), a curve is drawn to fit the general shape of the histogram
5. The curve is normalized, such that the mode of the frequency distribution is assigned a suitability of 1.0 and all lesser suitabilities assigned proportionately.

The foregoing procedure seems to have been fairly successful for species with well-defined habitat requirements, such as for drift feeding salmonids, darters, and other obligate or facultative riverine species. However, accuracy and transferability of the criteria suffered when:

1. The habitat requirements and tolerances of the target organism were extremely plastic.
2. The habitat requirements and tolerances of the target organism were actually determined by the juxtaposition or interspersion of several microhabitat types (i.e., microhabitat tolerances were plastic, but mesohabitat requirements were tightly constrained).
3. The target organism was too rare or widely dispersed to observe a sufficient number of individuals to develop habitat suitability criteria.
4. The biological relevance of the habitat information was unknown or not demonstrated.

By overlaying information about the habitat mosaic with information on species distributions and abundances, it will be possible to examine habitat utilization following a "top-down" strategy. Rather than looking for patterns of habitat use one fish and one variable at a time, attributes of community structure and function (e.g., species diversity, Index of Biotic Integrity) can be used directly in habitat assessments. The use of more powerful statistical analyses, such as Canonical Correspondence Analysis (Palmer 1993) can facilitate investigatations of habitat use along a multivariate gradient of environmental variables. Traditionally, these gradients have consisted of first-order environmental variables, such as depth, velocity, and substrate particle size. The use of two-dimensional habitat models, however, may encourage the use of "synthetic variables" to generate a gradient of spatially-explicit indexes (e.g., contagion or fractal dimension). These spatial indexes may expose relations between the habitat mosaic and biological characteristics that are undetectable using "typical" microhabitat variables.

A top-down strategy for habitat suitability criteria focuses at the mesohabitat scale, which reduces the need for high precision sampling techniques (e.g., observing habitat use by snorkeling). Using a bottom-up sampling strategy, it is very important to avoid disturbing the fish while in the process of observing them. Another difficult aspect of criteria development using the bottom-up strategy is that the exact location of the fish at the time of sampling must be determined as precisely as possible. These restrictions make it very difficult to sample or observe fish in deep, fast, or turbid water. Some of the most effective equipment for sampling fish could not be used because it caused excessive disturbance or because it was impossible to tell exactly where the organism was when it was sampled. A top-down strategy can minimize these problems because mesohabitat conditions are related to characteristics of populations or communities, not just to individual fish. The exact location of the fish at the time of sampling is much less significant with this sampling approach.

Investigating Ecological Questions

Since about the mid-1970s, a lively debate has taken place among community ecologists regarding the roles of biotic versus abiotic factors in determining community structure and function. Grossman et al. (1982) described the two antagonists in this debate as the "determinists" and the "stochasticists." Following, is a simplified synopsis of the debate:

1. The determinists hypothesize that biotic mechanisms, such as predation and competition, are responsible for different patterns of community composition and function (MacArthur 1965; Cody and Diamond 1975; Gatz 1979; Moyle and Vondracek 1985). In essence, the deterministic argument is based on the observation that different species of plants and animals tend to specialize in resource utilization in rather well-defined niche spaces. By providing a greater variety of niche spaces, it is possible to support a greater variety of species in a particular environment. In stream ecology, the determinists offer as evidence the findings that the diversity of fish species is correlated with habitat diversity (Gorman and Karr 1978), that patterns of fish community structure tend to parallel patterns in stream habitat structure (Schlosser 1982), and that differences in species assemblages in different parts of a stream reflect differences in physical characteristics of the stream (Moyle and Vondracek 1985).
2. The stochasticists postulate that the physicochemical environment is rarely stable long enough to allow a competitive equilibrium to develop among species. Thus, species abundances are determined largely through differential responses to unpredictable environmental changes rather than through biological interactions. These random catastrophes (e.g., floods, droughts, fires) maintain populations at densities well below those needed for competition to occur with sufficient intensity to affect community structure and function. During each of these "ecological bottleneck" periods, many individuals will be exterminated and only certain individuals or species will survive. When the ecological crunch is relieved, resources will be abundantly available to the survivors (Wiens 1977; Connell 1980). In support of abiotic regulation of communities, the stochasticists provide evidence that the highest levels of species diversity occur in environments that experience moderate rates of intense disturbance (Connell 1978; Huston 1979; Grossman et al. 1982). Diversity tends to be lower in extremely stable or extremely variable environments.

Both sides of the determinist-stochasticist debate appear to have compelling arguments. The problem , however, may not be so much one of competing ecological theories, but rather, the result of an inability to depict spatial and temporal variability at the same time. The use of spatially explicit modeling may allow us to develop this type of metric.

The habitat mosaic changes instantly whenever there is a change in discharge, so the various indexes of spatial heterogeneity will also vary as a function of discharge. For example, the relationship between discharge and habitat diversity (Shannon and Weaver 1962) is similar to the familiar flow-habitat relationship derived from PHABSIM (Figure 2b). The similarity of the two functions means that variations in habitat diversity can be analyzed over time, using exactly the same habitat time series techniques that are utilized routinely in the IFIM. A time series of habitat diversity can be constructed using the stream hydrograph and the relationship between flow and habitat diversity (Figure 2).

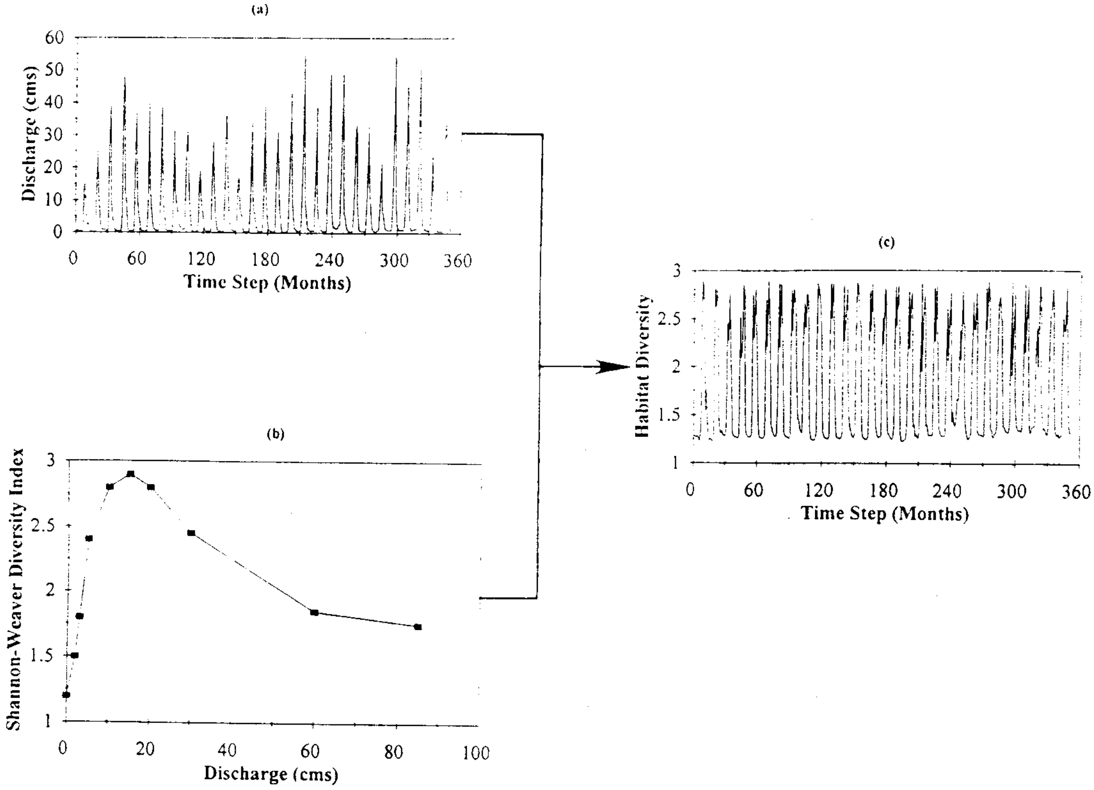


Figure 2: Development of a habitat-diversity time series for the Cache la Poudre River, Colorado: (a) a discharge is read for the first time step from the hydrologic time series, (b) the habitat diversity index corresponding to the discharge for the first time step is determined, and (c) the habitat index is transcribed to the time step on the habitat diversity time series.

In a manner similar to the development of a flow duration curve (Riggs 1968), time series data for habitat diversity can be arranged and plotted as a habitat-diversity duration curve (Figure 3). The duration curve allows a relatively quick and easy interpretation of temporal variability. With the axes oriented as they are in Figure 3, a horizontal line means that the habitat is extremely stable, whereas a near-vertical line exemplifies a highly variable environment. Several indexes can be used to depict variability of a time series: the coefficient of variation (the ratio between the standard deviation and mean), the ratio between the habitat values at the 10% and 90% exceedance probabilities, or simply the average slope of the duration curve over the range of probabilities. The important point is that this type of analysis enables the development of a metric that describes the temporal variability of spatial variability.

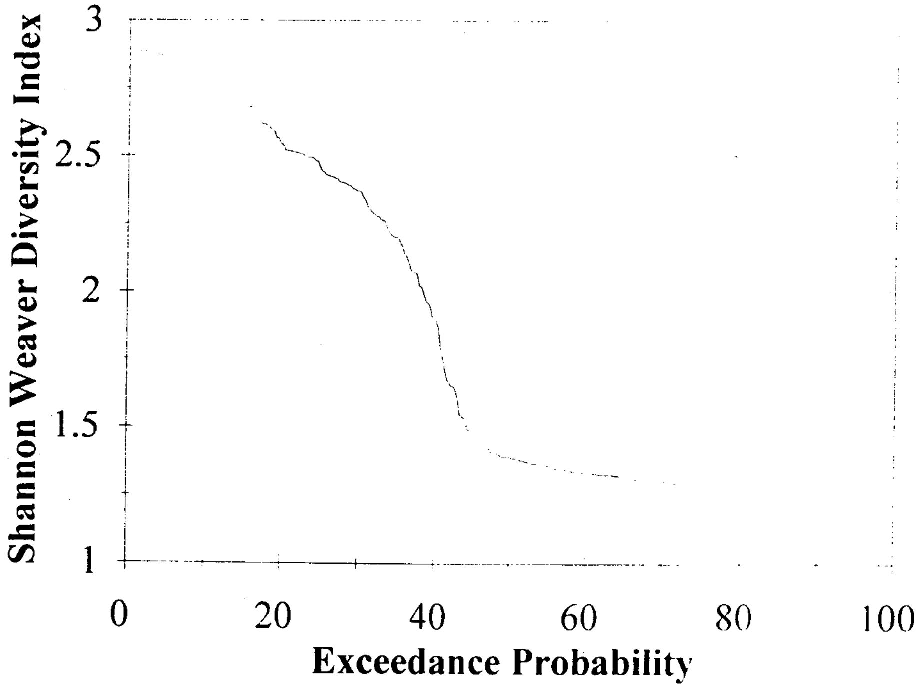


Figure 3: Habitat-diversity duration curve for the Cache la Poudre River, Colorado.

The ability to assess habitat variability in time and space may provide valuable new insights into the mechanisms by which populations and communities are regulated. Hypotheses that could be examined using two-dimensional habitat models include, for example:

1. Fish and invertebrate communities in structurally diverse streams with stable hydrographs will exhibit a high degree of resource partitioning and will contain a large proportion of specialist species (e.g., Poff and Allan 1995).
2. Streams that are dominated by one or two large patch types will tend to contain a few specialists species adapted to conditions at the interior of the patch, or generalists across patches.
3. Communities in streams exhibiting large temporal variability in habitat availability or heterogeneity will be dominated by generalists and species with high recolonization potential. Species assemblages will exhibit great flux, but will be highly resilient following disturbances.
4. Critical resources, as represented by small, unique patches, may be in short supply where the dominance index (DI) is large.
5. Edge effects are maximized in landscapes having many same-sized patches.
6. Where the stream is fragmented into numerous patch types and connectivity is low, accessibility to patch types may be more important than the amount of critical habitat available to a species.

These are just a few examples of the types of questions that can be addressed using spatially explicit habitat models. The most important point of this discussion, however, is that by describing the relationships between habitat indexes and a population- or community-level response, we are essentially validating our models as we develop them. Incremental changes in habitat metrics should be translatable into some type of known biological response, which may help biologists to postulate and test for underlying ecological mechanisms. Thus, mangement is not the only benefactor of this technology; basic ecological research should benefit from it as well.

THE PITFALLS

The concepts for two-dimensional habitat modeling have been around for a long time. Two-dimensional hydraulics models have been available since the early 1970s, and spatially-explicit habitat models in the form of Geographic Information Systems (GIS) were perfected in the early 1980s. The new aspect is the marriage of the two types of technology into a coherent habitat assessment tool. Before two-dimensional habitat assessments become an operational reality, however, I believe that several obstacles will need to be overcome.

The Sunk-Cost Inertial Braking System

It is a safe wager that most of the participants in this symposium are in the research and development business. With respect to developing an operational technology, it is vitally important to recognize that our clients are planners and managers, not other researchers. I believe that this distinction is routinely overlooked by people involved in the development of new technologies. A researcher might be willing to investigate a new technology simply because it is interesting or it hasthe potential to become a useful product. To researchers and developers, technological advancement is often an end in itself. To people in planning and management, however, technology is only a means to an end. Therefore, if a new technology is not reliable or supported, user friendly, and more utilitarian for decision-making than what the manager is currently using, it will probably collect dust on a shelf somewhere.

Over the last decade and a half, hundreds of people have received training in the care and feeding of PHABSIM. The technology has become institutionalized to the extent that an industry has developed around it. The industry has learned to deal with the foibles and weaknesses of PHABSIM and has been doing just fine without a two-dimensional model. These people are not likely to embrace two-dimensional modeling just because it can produce nice graphics. In fact, two-dimensional modeling may be considered by some as a threat, especially if it is promoted as a replacement to PHABSIM rather than as an attractive alternative.

The Validation Conundrum

The second barrier to the widespread adoption of two-dimensional modeling is "validation," or determining whether the output has biological significance. One of the problems we encountered with PHABSIM was that the technology had been on the market for nearly a decade before we got around to validating it. This created two problems. First, the absence of biological relationships made it difficult to interpret the output from PHABSIM. Consequently, managers were forced to rely on policy and rules rather than on scientific understanding. Many of these policies and rules undoubtedly appeared arbitrary and capricious to developers and applicants. The second problem that arose from a lack of rigorous biological testing was that the methodology became vulnerable to criticism (Mathur et al. 1985; Shirvell 1986; Scott and Shirvell 1987). Although a methodology should be defendable under criticism and scientific scrutiny, the lack of validation placed the burden of proof on the manager who used PHABSIM, rather than on the instigators of the problem. To this day, the number and intensity of critical articles about PHABSIM tends to increase dramatically just before a major instream flow case goes to arbitration.

Recently conducted studies have provided some insights into the biological meaning of PHABSIM output (Nehring and Anderson 1993; Bovee et al. 1994). Presumably, these findings are transferable for similar habitat metrics derived from a two-dimensional model. However, if the output from two-dimensional models provides no more or no better biological information than what we can currently derive from PHABSIM, there may be little incentive to adopt the new technology.

With the current popularity of terms like biodiversity, ecosystem management, and biological integrity, one would think that two-dimensional modeling is predestined to fill the niche of community-level analysis. Unfortunately, we are as ignorant today about the meaning of a contagion index to a community of river fishes as we were about the meaning of weighted usable area to adult trout populations ten years ago. Currently, a large amount of energy is being expended in developing and improving the hydraulic component of this technology. A lesser amount of energy is being directed at the habitat analysis, and less yet toward testing the biological relationships. I propose that this emphasis is exactly backward. We already know much about two-dimensional hydraulics, comparatively speaking. We know considerably less about two-dimensional habitat mapping and metrics. We know virtually nothing about the biological meaning of these habitat metrics, other than what we have learned from one-dimensional modeling. I am encouraged that a few small efforts have been initiated in the last year or so, but I believe that a much broader effort is warranted.

Furthermore, we should simultaneously examine the policy-relevance of this information to decision-makers. As a concept, biodiversity seems to be similar to pornography. No one defines it the same, but everyone seems to be able to recognize it when they see it. Simply because a decision-maker supports the concept of biodiversity does not necessarily mean that he or she will know what to do with a model that predicts how it will change in response to operational alternatives. The extirpation of endemic specialist species by rapid-colonizing generalists, for example, would be interpreted by many decision-makers as "eco-babble." Expressing the same information differently, (e.g., pallid sturgeons and blue suckers will be replaced by green sunfish and emerald shiners) may be more meaningful to some. To others, the most meaningful information might be that a particular operational scheme will result in an unpleasant encounter with the Endangered Species Act. As methodology designers, we must be acutely sensitive to the type of information that will be of greatest utility to the end user.

The Learning Curve Quagmire

Currently, the most charitable adjective I can think of to describe the process for developing a habitat map such as the one shown in Figure 1 is "Byzantine." The procedure I followed was probably more intricate than necessary, but it involved six different software packages, none of which was designed to interface with one another. In fact, the data could not even be processed from beginning to end on the same computer. On a user-friendliness scale, PHABSIM is absolutely chummy compared to two-dimensional modeling (at least, the way I did it).

If a client uses two-dimensional modeling routinely, an awkward process for conducting analyses might be tolerated because the user will have invested enough time to learn a system, regardless of its complexity. For some users, however, it may be several years between applications, which means that they will essentially have to re-learn the system every time they use it. Acceptance and use of two-dimensional river habitat modeling will be facilitated greatly if an infrastructure for software development, support, and transfer is established. Superior models may go unused if there is no reliable source of intelligible documentation, training, technical assistance, and software support.

Enthusiasm

We did several things right during the development of PHABSIM. We provided a tremendous amount of training and technical assistance to our clients. We incorporated clients' ideas and suggestions into our models. Where there were lapses in system capabilities, we developed work-arounds. In a word, we were enthusiastic about PHABSIM. Normally, enthusiasm is a good thing, but with respect to PHABSIM, excessive enthusiams may have encouraged unrealistic expectations among ourselves and our client community.

One lesson from the PHABSIM experience was that first impressions are extremely important. Users who were successful with PHABSIM (defined as anything from being able to make the models run to justifying flow recommendations) the first time they used it, became loyal defenders of the methodology. Often, these clients insisted on the use of PHABSIM, even in situations where another approach might have been easier or more applicable. In contrast, people who had a bad experience the first time they used PHABSIM (even if the difficulty was unrelated to methodology or software) tended to avoid it thereafter. Some became outspoken critics of the model, but many others simply could not bring themselves to trust model outputs again.

During the past two years, I have heard a number of claims regarding the superiority of two-dimensional modeling, mostly from the hydraulics component. Although I believe that most of these claims can ultimately be proven, to date relatively few have actually been substantiated. For example, consider the claim that less data, or more easily acquired data, are needed for two-dimensional hydraulic models than for one-dimensional models. In part, the reduced data requirement is based on the assertion that two-dimensional models can produce highly accurate velocity predictions without the calibration velocities that are so time-consuming in PHABSIM. I have two observations from my limited experience with two-dimensional hydraulics. First, the premise that calibration velocities are unnecessary has not been tested sufficiently in enough different kinds of streams to warrant such a claim. (Incidentally, calibration velocities are not always necessary in PHABSIM, either). Second, even if calibration velocities are not necessary, these data are more than compensated by the amount of bathymetric data required for a two-dimensional model. If bathymetric data can be collected using GPS, sonar, and data loggers, the large volume is not especially troublesome. However, large volumes of bathymetric data seem problematic if they must be collected with a total station, or worse, with a level. In practical terms, two-dimensional hydraulic modeling appears to be most applicable in rivers that are large enough to support data collection from a boat. In order for it to become operationally practical in small streams, we must first devise economical methods for collecting bathymetric and cover-polygon data (e.g., the use of low-level stereoscopy or multi-spectral videography) that require a little ground-truthing rather than intensive surveying.

SUMMARY

In summary, two-dimensional hydraulic and habitat modeling offer great opportunities to further the state-of-the-art in river habitat assessments. Advancements in hydraulic simulation notwithstanding, the greatest opportunities for advancement lie in the areas of two-dimensional habitat analysis and associated ecological research. Two-dimensional hydraulics allow us to do things better than we could with one-dimensional hydraulics: two-dimensional habitat analysis allows us to do things that were impossible in one dimension. Opportunities provided by this newly developing technology include:

- Application to riverine environments of spatially explicit habitat metrics from landscape ecology.
- Development of "top-down" habitat suitability criteria by examining patterns of habitat utilization and availability at the mesohabitat scale.
- Derivation of new metrics that describe variabilility of habitat in time and space, simultaneously.
- Testing of numerous hypotheses regarding linkages between spatial and temporal habitat variability and characteristics of aquatic communities, such as biodiversity, persistence, and resiliency to disturbance.

It is important to remember, that no matter how interesting two-dimensional technology is to the research and development community, the intended end-users are managers and planners. Several barriers exist that will impede the wholesale adoption of two-dimensional technology by the management and planning community. If not dealt with skillfully, these barriers may actually prevent successful technology transfer:

- User inertia in the form of previous investments in learning one-dimensional systems.
- Lack of biological testing and validation of new habitat metrics.
- Failure to provide a client-support infrastructure.
- Unrealistic expectations resulting from overzealous promotion of the technology.

This technology deserves our enthusiasm, but we must discipline ourselves against overselling. It is difficult to get a second chance to make a first impression.

ACKNOWLEDGEMENTS

I thank Mary Freeman, Don Orth, Sylvie Valentin, and Michel Leclerc for their helpful comments and contributions to earlier drafts of this paper. Bob Waltermire was responsible for teaching me how to create the 2-dimensional habitat map in Figure 1. His patience and perserverance are laudable (see "Learning Curve Quagmire).

REFERENCES

Bovee, K.D. 1986. Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. *U.S. Fish and Wildlife Service Biological Report* 86(7). 235 pp.

Bovee, K.D., T.J. Newcomb, and T.G. Coon. 1994. Relations between habitat variability and population dynamics of bass in the Huron River, Michigan. *National Biological Service Biological Report* 21. 63 pp.

Burrough, P.A. 1986. Principles of geographical information systems for land resources assessment. Clarendon, Oxford, U.K.

Cody, M.L., and J.M. Diamond, eds. 1975. Ecology and evolution of communities. Belknap Press of Harvard University Press.

Connell, J.H. 1978. Diversity in tropical rainforests and coral reefs. *Science* 199:1302-1310.

Connell, J.H. 1980. Diversity and the coevolution of competitors, or the ghost of competition past. *Oikos* 35:131-138.

Fausch, K.D., and R.J. White. 1981. Competition between brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) for positions in a Michigan stream. *Canadian Journal of Fisheries and Aquatic Sciences* 38:1220-1227.

Freeman, M. C. and G. D. Grossman. 1993. Effects of habitat availability on dispersion of a stream cyprinid. *Environmental Biology of Fishes* 37: 121-130.

Gatz, A.J., Jr. 1979. Community organization in fishes as indicated by morphological features. *Ecology* 60(4):711-718.

Gorman, O.T., and J.R. Karr. 1978. Habitat structure and stream fish communities. *Ecology* 59(3):507-515.

Grossman, G.D., P.B. Moyle, and J.O.Whitaker, Jr. 1982. Stochasticity in structural and functional characteristics of an Indiana stream fish assemblage: a test of community theory. *American Naturalist* 119:611-637.

Harris, L.D. 1988. Edge effects and conservation of biotic diversity. *Conservation Biology* 2:330-332.

Huston, M. 1979. A general hypothesis of species diversity. *American Naturalist* 113:81-101.

Kwak, T. J. and T. M. Skelly. 1992. Spawning habitat, behavior, and morphology as isolating mechanisms of the golden redhorse, *Moxostoma erythrurum*, and the black redhorse, *M. duquesnei*, two syntopic fishes. *Environmental Biology of Fishes* 34: 127-137.

Leclerc, M., A. Boudreault, J.A. Bechara, and G. Corfa. 1995. Two-dimensional hydrodynamic modeling: a neglected tool in the instream flow incremental methodology. *Transactions of the American Fisheries Society* 124(5):645-662.

Lobb, M. D. III and D. J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. *Transactions of the American Fisheries Society* 120: 65-78.

Li, H. and J.F. Reynolds. 1994. A simulation experiment to quantify spatial heterogeneity in categorical maps. *Ecology* 75(8):2446-2455.

Lovejoy, T.E., R.O. Bierregaard, Jr., A.B. Rylands, J.R. Malcolm, C.E. Quintela, L.H. Harper, K.S. Brown, Jr., A.H. Powell, G.V.N. Powell, H.O.R. Shubart, and M.B. Hays. 1986. Edge and other effects of isolation on Amazon forest fragments. Pages 257-285 *in* M.E. Soulé, editor. *Conservation Biology: the science of scarcity and diversity.* Sinauer Associates, Sunderland, MA.

MacArthur, R.H. 1965. Patterns of species diversity. *Biological Review* 40:510-533.

Malcolm, J.R. 1994. Edge effects in central Amazonian forest fragments. *Ecology* 75(8):2438-2445.

Mathur, D. W.H. Bason, E.J. Purdy, and C.A. Silver. 1985. A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences* 42:825-831.

Moyle, P.B., and B. Vondracek. 1985. Persistence and structure of the fish assemblage in a small California stream. *Ecology* 66(1):1-13.

Nehring, R.B., and R. M. Anderson. 1993. Determination of population-limiting critical salmonid habitats in Colorado streams using the Physical Habitat Simulation System. *Rivers* 4(1):1-19.

Noss, R.F. 1983. A regional landscape approach to maintain diversity. *BioScience* 33:700-706.

O'Neill, R.V., J.R. Krummel, R.H. Gardner, G. Sugihara, B. Jackson, D.L. DeAngelis, B.T. Milne, M.G. Turner, B. Zygmunt, S.W. Christensen, V.H. Dale, and R.L. Graham. 1988. Indices of landscape pattern. *Landscape Ecology* 1:153-162.

Palmer, M.W. 1993. Putting things in even better order: the advantages of Canonical Correspondence Analysis. *Ecology* 74(8):2215-2230.

Poff, N.L. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology* 76:606-627.

Pringle, C.M., R.J. Naiman, G. Bretschko, J.R. Karr, M.W. Webster, R.L. Welcomme, and M.J. Winterbourn. 1988. Patch dynamics in lotic systems: the stream as a mosaic. *Journal of the North American Benthological Society* 7:503-524.

Rabeni, C. F. and R. B. Jacobson. 1993. The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Freshwater Biology* 29: 211-220.

Ranney, J.W., M.C. Bruner, and J.B. Levenson. 1981. The importance of edge in the structure and dynamics of forest islands. Pages 67-95 *in* R.L. Burgess and D.M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York.

Riggs, H.C. 1968. Frequency curves. Techniques of water-resources investigations of the U.S. Geological Survey, Book 4, Chapter A-2. 15 pp.

Rinne, J. N. 1991. Habitat use by spikedace, *Meda fulgida* (Pisces: Cyprinidae) in southwestern streams with reference to probable habitat competition by red shiner *Notropis lutrensis* (Pisces: Cyprinidae). *Southwestern Naturalist* 36: 7-13.

Schlosser, I.J. 1982. Fish community structure and function along two habitat gradients in a headwater stream. *Ecological Monographs* 52(4):395-414.

Scott, D., and C.S. Shirvell. 1987. A critique of the Instream Flow Incremental Methodology and observation on flow determination in New Zealand. Pages 27-43 *in* J.F. Craig and J.B. Kemper, editors. Regulated streams: advances in ecology. Plenum Press, New York.

Shannon, C.E., and W. Weaver. 1962. The mathematical theory of communication. University of Illinois Press, Urbana. 125 pp.

Shirvell, C.S. 1986. Pitfalls of physical habitat simulation in the Instream Flow Incremental Methodology. *Canadian Fisheries and Aquatic Sciences Technical Report* 1460. 68 pp.

Southall, P. D. and W. A. Hubert. 1984. Habitat use by adult paddlefish in the upper Mississippi River. *Transactions of the American Fisheries Society* 113: 125-131.

Stanford, J.A. 1994. Instream flows to assist the recovery of endangered fishes of the upper Colorado River basin. *National Biological Survey Biological Report* 24. 47 pp.

Townsend, C.R. 1989. The patch dynamics concept of stream community ecology. *Journal of the North American Benthological Society* 8:36-50.

Turner, M.G., R. Costanza, and F.H. Sklar. 1989. Methods to evaluate the performance of spatial simulation models. *Ecological Modelling* 48:1-18.

Wiens, J.A. 1977. On competition and variable environments. *American Scientist* 65:590-597.

Yahner, R.H. 1988. Changes in wildlife communities near edges. *Conservation Biology* 2:333-339.

ECOLOGICAL EVALUATION OF INSTREAM FLOW REGULATIONS BASED ON TEMPORAL AND SPATIAL VARIABILITY OF BOTTOM SHEAR STRESS AND HYDRAULIC HABITAT QUALITY

Klaus Jorde
Institute for Hydraulic Engineering, University of Stuttgart
Pfaffenwaldring 61, D-70550 Stuttgart
Germany
e-mail: jorde@iws.uni-stuttgart.de

ABSTRACT

Reduced discharge in rivers influenced by diversion-type hydropower plants may cause severe ecologic damage. The residual flow in the diverted segment is influenced by the natural discharge at the diversion weir, the technical and operational characteristics of the hydropower system, and the minimum flow regulation. An ecological assessment of instream flow regulations requires knowledge of the relationships between the discharge in a river stretch and the parameters that influence habitat size and quality. This study focuses on the hydraulic pattern at the river bottom, especially the temporal and spatial variability of bottom shear stress and therefore the hydraulic habitat availability for invertebrates and bottom dwelling fish. The hydraulic pattern at the river bottom depends on river morphology and flow and can be measured with FST-hemispheres. Calibration functions between FST-hemisphere movement and the local shear stress depend on the substratum roughness, the flow process at the bottom and the local depth. Habitat suitability functions showing the correlation between the local abundance and FST-hemisphere-measurement or bottom shear stress have been found for many taxae. In this study "CASIMIR" (Computer Aided **SI**mulation **M**odel for **I**nstream flow **R**equirements in diverted streams), a set of computer models developped by the author, was used for the first time. CASIMIR includes seperate units to simulate flow regimes in river sections affected by hydropower use, analyse and simulate the variability of bottom shear stress and other ecologically significant parameters, and simulate hydraulic habitat availability at the river bottom. FST-hemisphere measurements were performed in two morphologically different river stretches affected by the same hydropower plant at six different discharges in each stretch. The transformed random shear stress data were analysed statistically and density functions were found that describe the spatial distribution of the bottom shear stress. As the pattern of bottom shear stress was measured at different flows, relationships could be established that describe the spatial distribution of bottom shear stress as a function of the flow. Results show that shear stress distribution as a function of flow in a river section can be described by a set of curves. The characteristics of these curves depend on river section morphology, the slope of the energy line and the thalweg seem to be important. Applying natural or simulated hydrographs in the CASIMIR-unit for bottom shear stress leads to the temporal and spatial variability of bottom shear stress. If habitat suitability functions for selected species are applied with the bottom shear stress model, the temporal and spatial variability of size and quality of hydraulic habitat can be determined. CASIMIR was then used to investigate the hydraulic consequences of different minimum flow regulations: constant and dynamic, in both sections, during average, wet and dry years. The influence of dynamic minimum flow requirements and the channel morphology on the dynamics of wetted areas, bottom shear stresses and habitat quality for different rheobiont and limnophilous invertebrates were investigated.

KEY-WORDS: / diversion / hydro power plant / instream flow / minimum flow / simulation model / river bottom / bottom shear stress / spatial variability / temporal variability / benthic habitat / habitat suitability / CASIMIR .

INTRODUCTION

Diversion type hydropower plants may cause severe ecologic damage in the diverted rivers. Instream flow regulations should aim to reduce or minimize negative ecologic impacts. In Europe simple hydrological calculations based on percentages of the mean annual flow or the mean minimum flow have been widely used (Giesecke 1994, DVWK 1995). Such approaches have little ecological background. In North America habitat related methods, such as the "Instream Flow Incremental Methodology" with its major component PHABSIM (Bovee 1986), have been widely used to quantify habitat availability for certain fish species as a function of the discharge in a river stretch. Hydropower use is mostly restricted to rivers of the Rhitral and Epipotamal. In these rivers most of the biologic processes take place at the river bottom or within the substratum (Schwoerbel 1971). A new habitat approach was developped which includes the temporal and spacial variability of bottom shear stress and hydraulic habitat availability for benthic organisms and bottom dwelling fish. The pattern of hydraulic forces and substratum is an extremely significant habitat quality parameter for organisms living at the river bottom (Ambühl 1959, Mutz 1989). The hydraulic forces at the river bottom are also responsible for sediment transport or sedimentation of fine materials and thus shape the surface of the river bottom (Rabeni & Minshall 1977). The temporal dynamic must be considered since natural rivers are dynamic systems and their communities are adapted to and depend on dynamic flow regimes (Ward & Stanford 1983, Townsend 1989, Schmedtje 1995). The hydraulic pattern at the river bottom depends on the discharge and on the river bed morphology. At a given discharge, FST-hemispheres can be used to measure the hydraulic pattern at the river bottom (Statzner & Müller 1989, Statzner *et. al.* 1991)). Under certain conditions FST-hemisphere-measurements can be calibrated against bottom shear stress (Dittrich & Schmedtje 1996, Jorde 1996). Calibration functions depend mostly on water depth, substratum roughness and applied placement techniques of the FST-hemispheres and ground plate. Although heavily discussed (Frutiger & Schib 1993) FST-hemispheres have been shown to be suitable for the characterisation of the hydraulic habitat an the river bottom. The abundance of certain benthic species showed greater correlation with the FST-measurement at the sampling points than with any other factor (Fuchs 1994, Schmedtje 1995).

THE KOCHER CASE STUDIE

Field measurements

The Kocher is a midland river in the south of germany. The study was conducted in the diverted river bed at the "Ohrnberg"-hydropower station. The diverted section of the river is 8 km long, has an average slope of 1,4 % and lies approximately 180 m above sea. The catchment area at the diversion weir is 1500 km^2, the mean annual flow is approximately 20 m^3/s. Figure 1 shows a map of the site. Within the original stream bed two morphologically different characteristic river stretches, each with a length of 1000 m, were chosen. Test stretch A (TS-A) is a straightened section with very long and deep pools between short riffles. Test stretch B (TS-B) is more natural with small meanders and short riffle-pool-sequences. The two TS were choosen to show the influence of river bed morphology on the interaction between discharge and shear stress pattern. First the thalweg of each TS was measured and monitored and water-level gauges were installed at

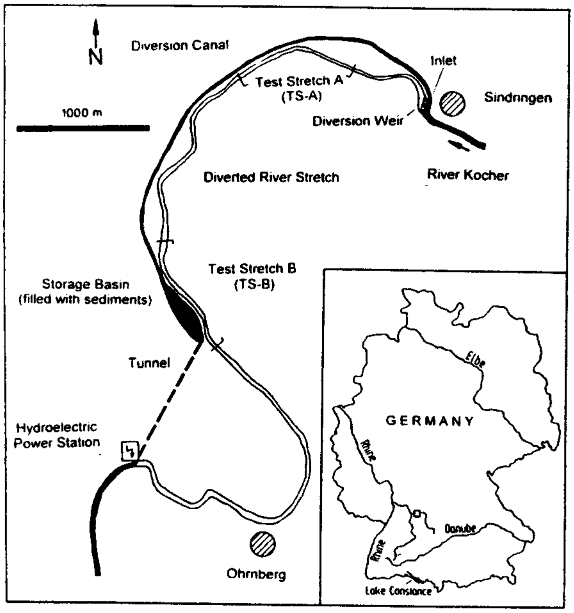


Fig. 1: location of investigation sites

50m intervalls to monitor water levels easily at different flows. Then series of 100 FST-measurements at randomly selected points were performed in both TS at 6 different discharge rates. The rate of discharges could be adjusted with a gate at the weir. The flow rates during the measurements are shown in Figure 2a and 2b. At each point the water depth was measured. The gauges and measurements of the inundated breadth at each flow every 10 m gave additional information about the aquatic volume in the TS. Biological investigations were focused on bethic communities.

Results from Field Measurements

Figures 2a and 2b show the results of the FST-measurements at each flow for TS-A and TS-B. Differences are visible but further statistical analysation is required for quantification. FST-hemisphere-measurements expressed in FST-

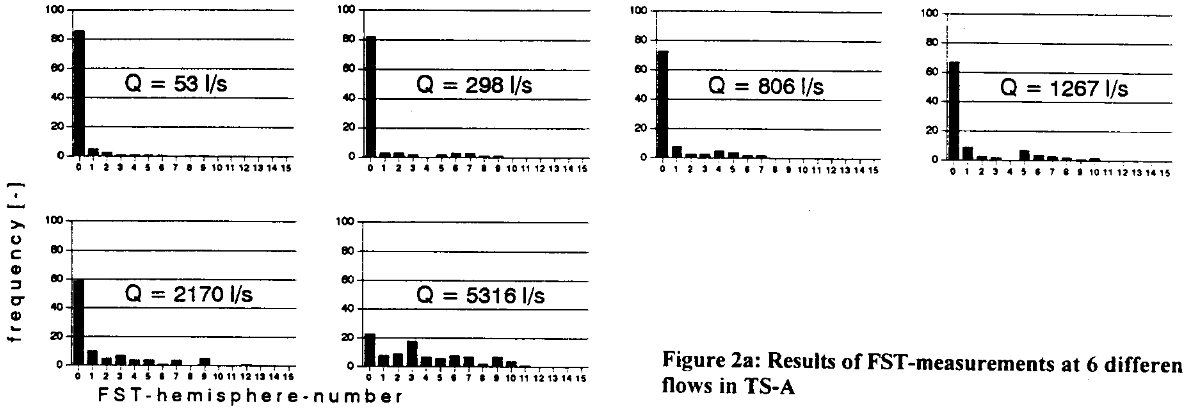


Figure 2a: Results of FST-measurements at 6 different flows in TS-A

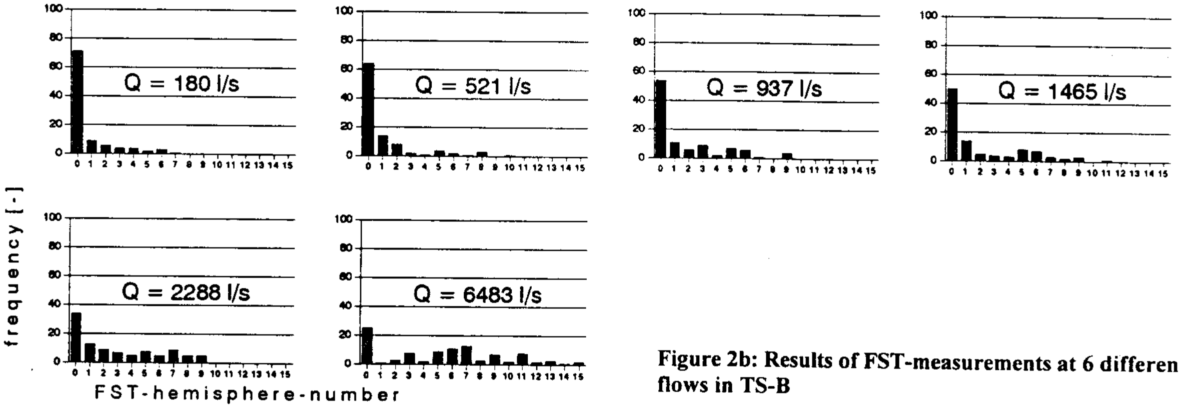


Figure 2b: Results of FST-measurements at 6 different flows in TS-B

numbers represent data on an ordinal scale. Statistical evaluation of ordinal data is only meaningful for a few operations such as calculating the median of a data series. For further evaluation it is necessary to transfer the FST-data into physical data, in this case bottom shear stress. The method of Smith (1975) which has been used by some authors to measure bottom shear stress should only be applied with restrictions in most natural streams. In this study the calibration functions for the ρ_i/τ_{lo}-relation (ρ_i = density of FST-hemisphere i in g/cm^3 and τ_{lo} = local shear stress in N/m^2) were derived from series of velocity-profiles over the FST-hemisphere ground plate measured in the Kocher TS (Jorde 1996). The calibration function that could be derived specifically for the Kocher FST-data was $\tau_{lo} = \rho_i^{3,69} - 0.465$. For hemisphere1-9 the function $\tau_{lo} = 8.45\rho_i - 8.18$ was used.

Figure 3 shows the results from two discharges (see Fig. 2b) in TS-B expressed in frequencies of FST-numbers (a,b) and corresponding local shear-stress using the given calibration function (c,d). Two observations can be made: 1) both series represent positively skewed unimodal distributions and not (as Fig. 3b would indicate) a bimodal distribution in the case of the higher discharge, and 2) reasonable statistical operations with FST-Numbers are severely restricted. The following simple example will illustrate this. The mean value of two single measurements, one FST-number 1 and the other FST-number 15 (see Fig. 2b) would be FST-number 8. The actual mean value of those two measurements (see Fig. 2d) is actually approximately 44 N/m^2 and thus between FST-hemisphere number 13 and 14. Therefore all further statistical calculations that are based on the arithmetic mean of the FST-hemisphere-numbers are meaningless.

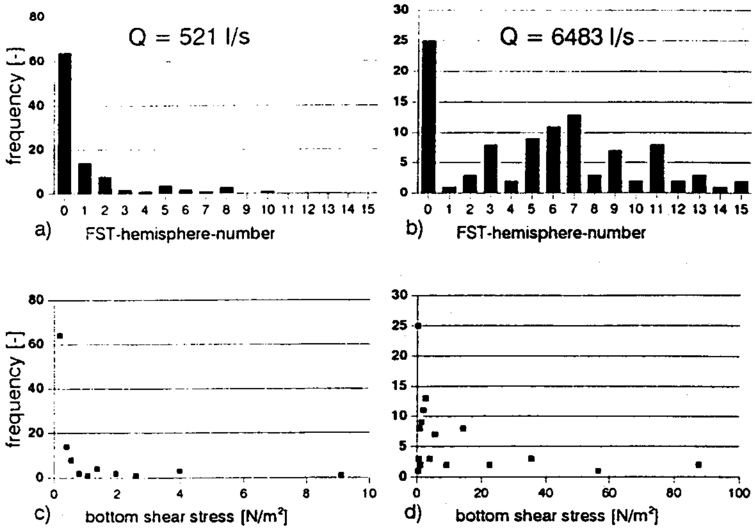


Figure 3: Results of FST-measurements at 2 different discharges expressed in FST-hemisphere-numbers and corresponding bottom shear stress (note different scales on both axis)

Statistical evaluation of the field results

A first attempt at describing the measured shear-stress series with 2- or 3-parametric gamma-, lognormal-, Weibull- or exponential-distributions did not lead to satisfactory results. In a second step the shear-stress data were divided in two ranges. Range 1 included all "0-measurements" (points where the lightest FST-hemisphere number 1 ist not moved by the flow are assigned the virtual FST-number 0). Range 2 included all measurements above 0. The threshold shear-stress-value between range 1 and range 2 is the critical shear-stress for the movement of the lightest hemisphere Nr. 1 which was calculated from the calibration function to 0.36 N/m^2. The nω^2- goodness of fit tests (Dyck 1980) showed that the data of range 2 could be very well discribed by 3-parametric lognormal (LN) distributions (Figure 4) with a threshold value of θ = 0.36. Figure 5 shows the 2-range-cumulative density function. R_0 is the share of R_1 where the flow is actually "zero", and was set to 0.75 R_1. R_2 ist the part of the measurements that can be described by the lognormal-distributions. These have the general form

$$(1) \quad f_\tau(\tau) = \frac{1}{\sigma_t(\tau-\theta)\sqrt{2\pi}}\, e^{-\frac{1}{2}\frac{(\ln(\tau-\theta)-\mu_t)^2}{2\sigma_t^2}}$$

with θ = threshold-, μ_t = scale- and σ_t = shape-parameter. The index t indicates that the parameters are calculated from the transformed data $t_j = \ln(\tau_j - \theta)$. μ_t is the arithmetic mean and σ_t is the variance of the transferred data t_j.

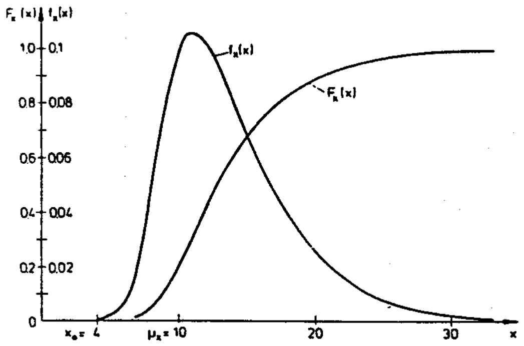


Figure 4: Example for a lognormal (LN) distribution with θ=4.0, σ_x=7 and μ_x=14 (after Plate 1993)

The density function of the LN-distribution is:

$$(2) \quad F_\tau(\tau) = \Phi\left(\frac{\ln(\tau - \theta) - \mu_t}{\sigma_t}\right)$$

with

$$(3) \quad \Phi(x) = \frac{1}{2\pi}\int_{-\infty}^{x} e^{-\frac{x^2}{2}}\,dx$$

where $\Phi(x)$ is the density function of the standard normal distribution (Plate 1993). The maximum-likelihood-method was used to estimate the density distributions for each set of empirical shear stress data. The results are shown as density functions for both ranges in Figure 6. The parameters of the density distributions of range 2 only for both TS-A and TS-B at 6 different flows and $n\omega^2$ test results are given in table 1 (Function values $F_\tau(\tau)$ have to be multiplied by R_2 and then added R_1 to obtain functions in Figure 6). For further calculations both ranges R_1 and R_2 are combined. R_1 and R_2 represent the portions of the river bottom area belonging to range 1 and range 2 with $R_1 + R_2 = 1$ and $R_0 \le R_1$ (Figure 5). Equ. (1) is valid for $\tau > \theta$ and

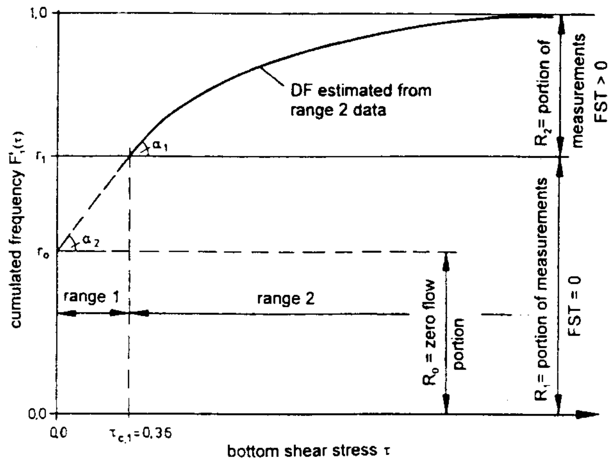


Figure 5: Division of the data into two ranges. Range 2 can be discribed with LN-distributions $F_\tau(\tau)$, $F'_\tau(\tau)$ is related to the whole range.

$$(4) \quad f_\tau(\tau) = \frac{R_1 - R_0}{\theta} \qquad if\ \tau \le \theta$$

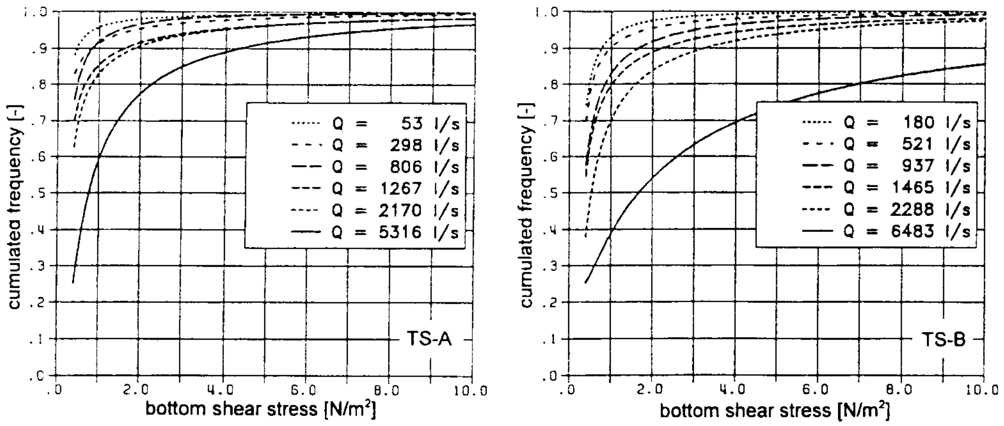


Figure 6: LN-distributions for range 2 data at 6 different flows in TS-A and TS-B (parameters see table 1)

Table 1: Parameters of LN-distributions and results of $n\omega^2$-tests

	Q [l/s]	shape σ	scale μ	nω²	R₂		Q [l/s]	shape σ	scale μ	nω²	R₂
TS-A	53	1.791	-1.339	0.084	0.14	TS-B	180	1.435	-1.479	0.187	0.29
	298	1.619	-0.564	0.113	0.18		521	1.745	-1.515	0.287	0.36
	806	1.510	-1.240	0.241	0.27		937	1.589	-0.979	0.201	0.46
	1267	1.850	-0.722	0.249	0.33		1465	1.776	-0.873	0.318	0.50
	2170	1.783	-0.834	0.121	0.41		2288	1.648	-0.641	0.284	0.66
	5316	1.516	-0.330	0.154	0.77		6483	1.531	0.933	0.191	0.75

From each of the densitiy functions $f_\tau(\tau)$ or their distribution functions $F_\tau(\tau)$ the relative frequency of a certain range of bottom shear stress can be derived. For these "hydraulic niches" which have a lower value τ_u and an upper value τ_o, the portion of the whole unit area 1 with a bottom shear stress between τ_u and τ_o can be derived from function $R(\tau)$:

(5) $$R_{\tau_u,\tau_o} = F_{\tau_{(\tau_o)}} - F_{\tau_{(\tau_u)}} = \frac{(R_1 - R_0)}{\theta}(\tau_o - \tau_u) \qquad if\ 0 < \tau \le \theta$$

and

(6) $$R_{\tau_u,\tau_o} = F_{\tau_{(\tau_o)}} - F_{\tau_{(\tau_u)}} = R_2 \int_{\tau_u}^{\tau_o} f_{\tau_{(t)}}\, d\tau \qquad if\ \theta < \tau < \infty$$

and

(7) $$R_{\tau=0} = R_0$$

Integration of the function $R(\tau)$ between $\tau = 0$ and $\tau = \infty$ will be 1. Those factors R must be multiplied by the entire inundated area, $A(Q)$, to obtain the actual river bottom area that belongs to the specific hydraulic niche. That is the spatial variability of the bottom shear stress at a certain flow Q in an investigated river stretch. Several approaches were used to find the general pattern of flow and shear stress distribution influenced by the specific river bed morphology. The one that gave the most reasonable results is shown in Figure 7. A linear interpolation between the values of $f_\tau(\tau)$ at two investigated discharges Q_i and Q_{i+1} leads to a plane $H(Q,\tau)$ from wich any portion R between some τ_u and τ_o at a certain flow $Q_{(t)}$ can be calculated by:

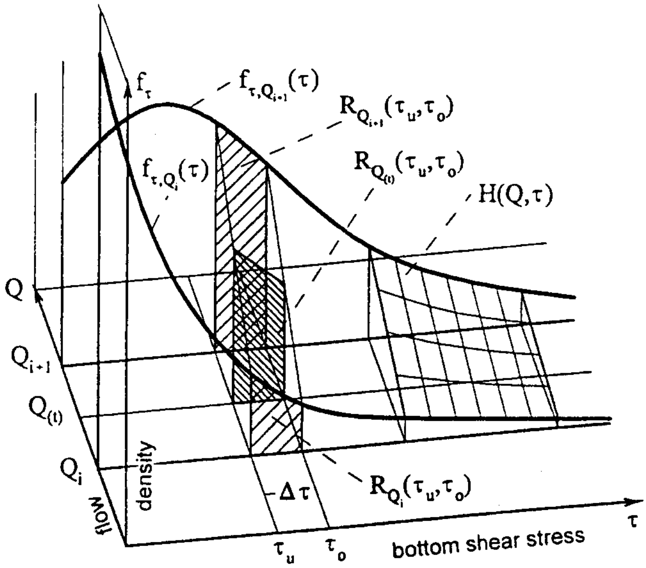


Figure 7: Calculation of the spatial variability of bottom shear stress at a (time dependant) discharge $Q_{(t)}$ from data of 2 in- vestigated discharges Q_i and Q_{i+1}.

(8) $$R_{Q_{(t)}(\tau_u,\tau_o)} = \frac{\int_{\tau_u}^{\tau_o} f_{\tau,Q_{i+1}}(\tau)\, d\tau - \int_{\tau_u}^{\tau_o} f_{\tau,Q_i}(\tau)\, d\tau}{Q_{i+1} - Q_i}(Q_{(t)} - Q_i) + \int_{\tau_u}^{\tau_o} f_{\tau,Q_i}(\tau)\, d\tau$$

or more easily with

(9) $$R_{Q_{(t)}(\tau_u,\tau_o)} = \frac{R_{Q_{i+1}(\tau_u,\tau_o)} - R_{Q_i(\tau_u,\tau_o)}}{Q_{i+1} - Q_i}(Q_{(t)} - Q_i) + R_{Q_i(\tau_u,\tau_o)}$$

Multiplication of $R_{Q}(\tau_u,\tau_o)$ by the inundated area $A_{Q(t)}$ leads to the general real area pattern of bottom shear stress as a function of flow. This information over the entire investigated flow range can be understood as a general "flow-shear-stress-model". All further investigations of temporal variability can be performed using these models by combining them with discharge time series. The time series used can be natural hydrographs, duration curves or, as in this case, simulated hydrographs for the diverted river stretches influenced by a hydropower plant.

SIMULATION MODEL CASIMIR

"CASIMIR" (Computer Aided SImulation Model for Instream flow Requirements in diverted streams) is a set of computer models developped by the author (Jorde 1996). It consists of serial modules: GAMA (registration of measured FST-values, calibration functions, statistical evaluation), WASKRA (simulation of hydropower plant, natural flow curves, head-flow-relation curves, turbine characteristics and minimum flow regulation), TAUSIM (combining the flow-shear-stress-model with time series of flow and thus calculating the temporal and spatial variability of bottom shear stress) and HABIM (combining results from TAUSIM with habitat suitability curves of selected species for habitat quality and availability simulation). As WASKRA simulates the hydropower plant and calculates the head, plant discharge, turbine efficiencies, energy output, and hydrographs for the diverted river stretch it is also used to assess economic impacts of minimum flow regulations. Any kind of minimum flow regulation, constant, time dependent or dynamic can be simulated by WASKRA. The hydrograph for the diverted river stretch is used to calculate the temporal variability of the shear stress pattern.

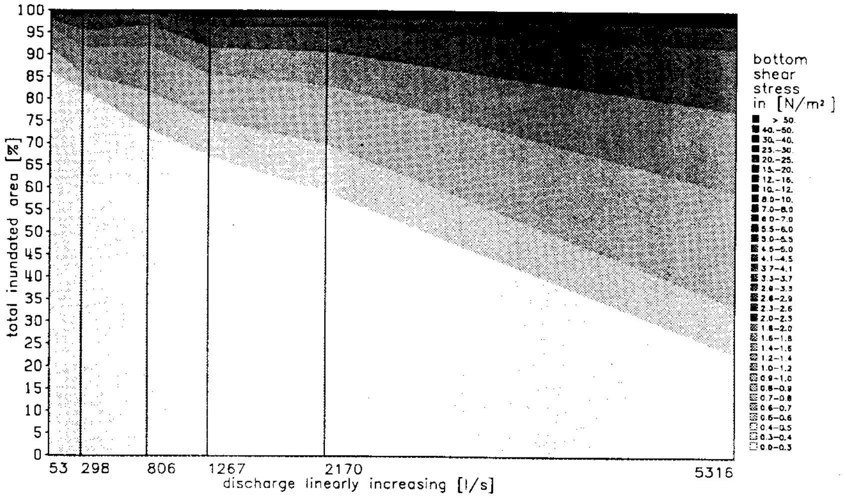


Figure 8: Shear stress distribution in TS-A over increasing discharge. The vertical lines indicate the flow rates where FST-measurements were done. The shear stress scale is not linear

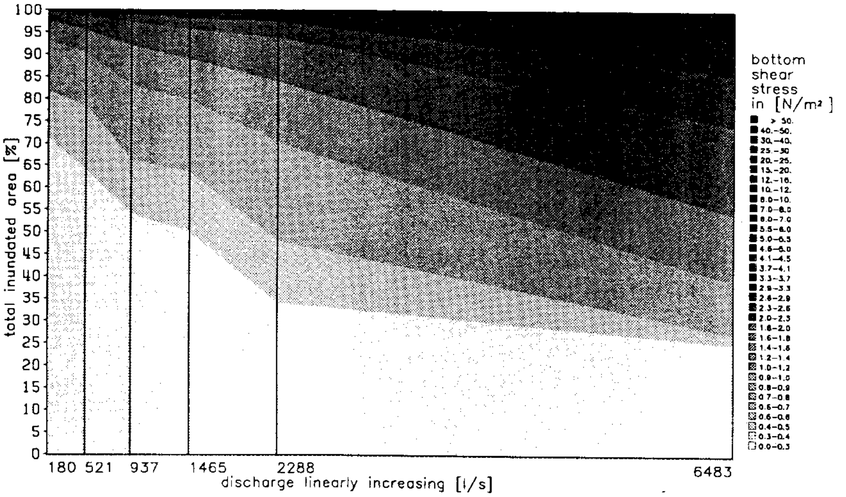


Figure 9: Shear stress distribution in TS-B over increasing discharge. The vertical lines indicate the flow rates where FST-measurements were done. The shear stress scale is not linear

GAMA and TAUSIM are used for practical computations of the shear stress pattern as described above. Figures 8 and 9 show the shear stress distribution as functions of flow in both TS-A and TS-B. A linearly increasing discharge between the minimum and maximum investigated flow in each TS was used for the simulation. The hydraulic range of natural-type TS-B is much larger than the one of regulated TS-A. Also, in TS-B areas with "zero flow" decrease much faster with rising discharge. The variability of bottom shear stress seems to follow certain rules which resemble sets of curves. The characteristics of those curves depend on the river bed morphology. At this time it is assumed that, besides the mean slope of the energy line, the thalweg has a most significant influence on those curves' characteristics. However, parameters could not be clearly established.

SIMULATION INVESTIGATION AND RESULTS

CASIMIR was used to simulate the impact of seven different minimum flow regulations including the status quo, constant, time dependent and natural-flow dependent damped regulations. Three hydrologically different years were used, the average year 1986, a wet (1987) and a dry year (1989). Results presented in this paper cover only the average year. The impact of the different morphology on the pattern of bottom shear stress should be shown and consequently the impact on hydraulic habitat size and quality at the river bottom for some selected species. Out of seven different minimum flow requirements three will be shown as examples. Alternative 1 (MF-1) is the status quo. Approximately 50 l/s run over a small fishway at the diversion weir or seep through under the weir. The flow at the beginning of TS-A is between 50 and 60 l/s. At the beginning of TS-B the flow is approximately 130 l/s higher due to seepage losses from the canal. Higher discharges occur only during natural discharges above 15 m³/s. These are the existing conditions for the past 70 years. Alternative 2 (MF-2) is a damped dynamic regulation where the minimum flow released at the diversion weir follows the natural flow at a certain damping rate to cause some natural dynamic in the diverted river bed. Alternative 3 (MF-3) is also a damped dynamic regulation but on a higher level. The regulations MF-2 and MF-3 are shown in Figure 10.

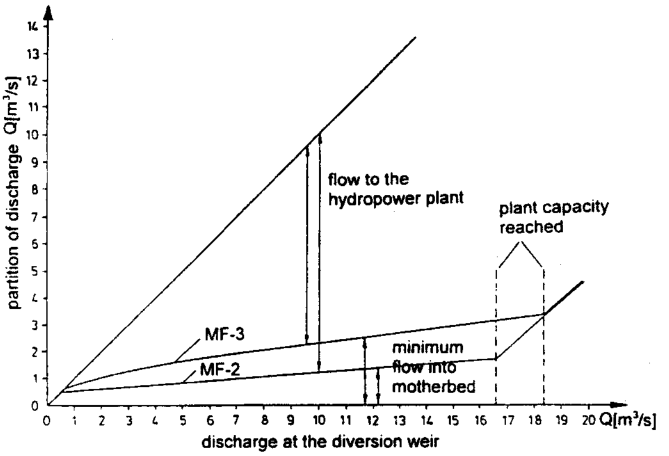


Figure 10: Damped dynamic minimum flow regulations MF-2 and MF-3

Bottom shear stress pattern

The temporal and spatial variability of bottom shear stress in TS-B for the average year 1986 under MF-1 (Figure 11) and MF-2 (Figure 12) conditions clearly differ. MF-1 causes a very monotone shear stress pattern during periods where the minimum flow regulation is ruling. The range of bottom shear stress reaches only 3 N/m², and only a very small portion of the river bottom shows higher shear stress values. The total inundated area is about 16000 m² with an average breadth of 16 m, 70% of that area is "zero flow area". Weir overflow periods cause a rapid change in shear stress pattern with shear stresses increasing to over 50 N/m². The inundated area increases to over 20000m². The end of weir overflow periods causes a rapid change back to MF-1 conditions within a day or two. There is an extreme lack of natural dynamics. MF-2 causes a much larger range of occurring bottom shear stresses up to more than 50 N/m² over small areas. Approximately 50 % (8500m²) of the inundated river bottom are subjected to shear stresses and there is an area of about 2000m² affected by a "natural dynamic" of being more or less frequently drowned or dry. Higher minimum flows (MF-3, no Fig.) cause higher dynamics in shear stress but reduced dynamic in inundated areas.

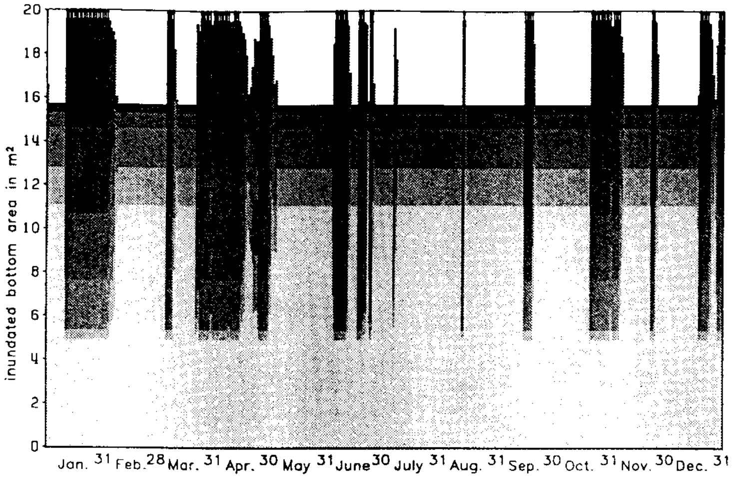
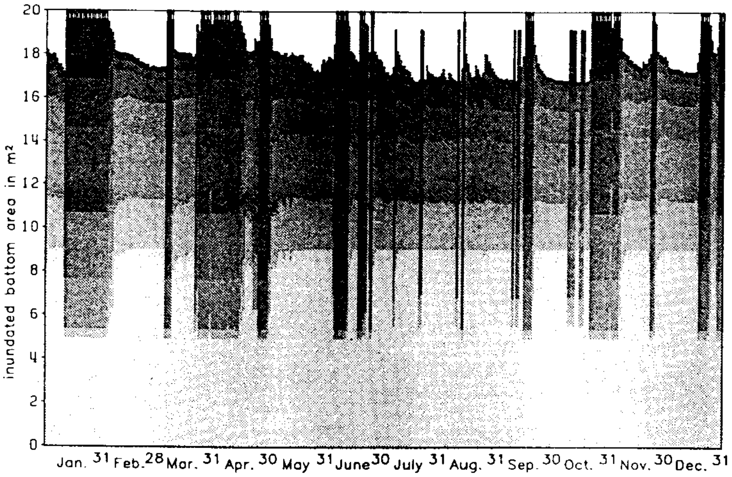


Figure 11: Temporal and spatial variability of bottom shear stress in TS-B due to MF-1, 1986



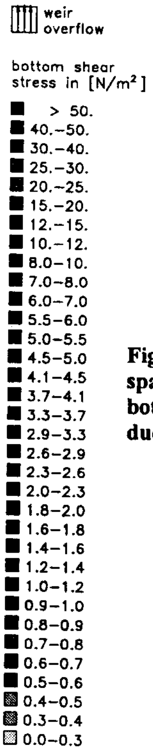


Figure 12: Temporal and spatial variability of bottom shear stress in TS-B due to MF-2, 1986

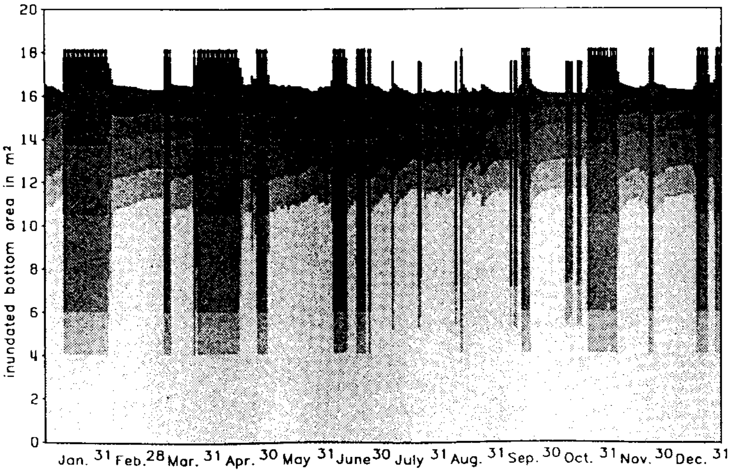


Figure 13: Temporal and spatial variability of bottom shear stress in TS-A due to MF-2, 1986

Figure 13 shows the results of MF-2 in the regulated TS-A. The range and dynamics of bottom shear stresses are far less developped and there is very little dynamics of inundated area because the river bed is very uniform. The simulation of wet and dry years showed that the hydrologic regime in combination with the MF-regulations causes different frequencies and periods of dynamic processes or disturbances. Dry periods in combination with constant MF-regulations can cause extremely unnatural conditions in the diverted river bed when the power plant is switched off because the available flow is less than minimum turbine capacity and the entire discharge runs over the weir.

Hydraulic Habitat Quality and Availability

Benthic organisms seem to react to overall forces at the substratum. These can be measured with FST-hemispheres (Schmedtje & Dittrich 1996). Other parameters, such as mean column velocities, have a roughness dependent relation to these forces and should therefore (from a hydraulic point of view) not be used as benthic habitat parameters unless geometric roughness parameters are included. Three benthic species have been selected as examples for hydraulic habitat simulation to assess the biological impact of the investigated MF-regulations. These are a limnophilous leech *Helobdella stagnalis*, and two caddis fly larvae, *Psychomia pusilla* and *Rhyacophila* cf. *dorsalis* (Figure 14). To calibrate FST-hemisphere based suitability curves in a habitat-quality-bottom-shear-stress relation the τ_{l_0}/ρ_l-function for the investigated river stretch must be used for the habitat suitability curves as well. These "preference curves" can then be combined with the shear stress pattern to obtain the temporal and spatial size and quality of available hydraulic habitats.

Figures 15 to 17 show the results for each investigated species at MF-1 and MF-2 conditions. *Helobdella stagnalis* (Figure 15) finds perfect hydraulic conditions over a very large area, particularly during MF-1 controlled periods. MF-2 or weir overflow conditions reduce its hydraulically suitable habitats considerably but there are still large areas remaining which offer best hydraulic conditions. During short weir overflow periods the animals move possibly deeper into the substratum. The hydraulic habitat availability for *Psychomia pusilla* (Figure 16) under MF-1 conditions show a few hundred m² within TS-B with acceptable or good hydraulic conditions. MF-2 conditions result in approximately double the area of suitable hydraulic habitats (compared to MF-1 conditions), but still about 75% of the inundated bottom area are hydraulically unsuitable for *psychomia pusilla*. The third considered species is the extremely rheophilous *Rhyacophila* cf. *dorsalis* (Figure 17). In the Kocher sampling sites it was found only at very few locations within TS-B riffle sections. The simulation shows that habitat availability is during MF-1 conditions extremely restricted in terms of quality and of size. Habitat quality remains below 30% over the entire domain and suitable hydraulic habitat is available only on approximately 300 m² (that is less than 2% of the inundated area). MF-2 causes a significant relative improvement in available habitat, however still on an extremely low level compared with the

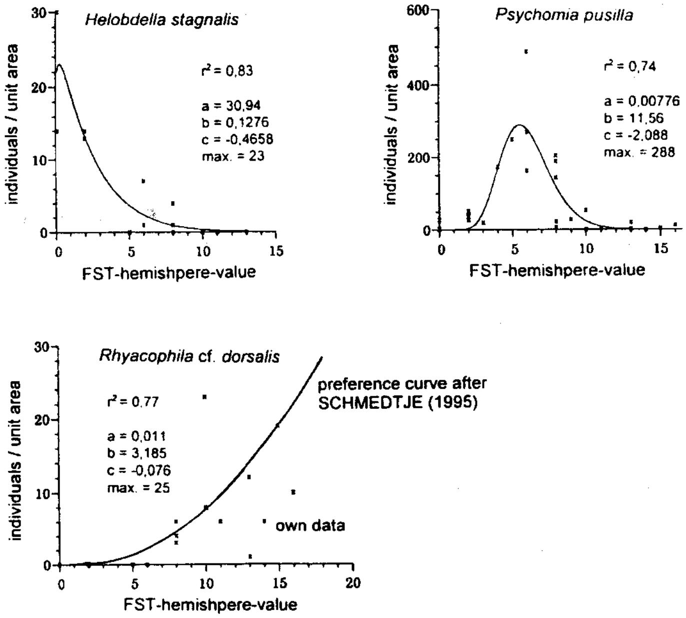


Figure 14: Habitat suitability curves for 3 benthic species derived from Kocher data and after Schmedtje(1995). a,b and c are the parameters of the function $f(x)=a\ x^b\ e^{cx}$.

other two species. Weir overflow periods reveal the hydraulic habitat availability during discharges of 5-6 m^3/s. This was the maximum investigated discharge and therefore the diagrams show that pattern during weir overflow.

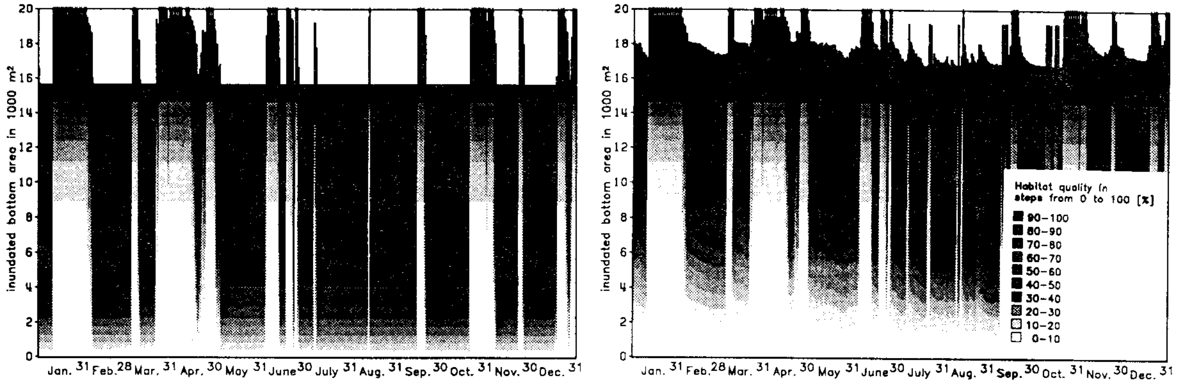


Figure 15: Temporal and spatial distribution of hydraulic habitat suitability classes for *Helobdella stagnalis* in TS-B, minimum flow regulation MF-1 (left) and MF-2 (right).

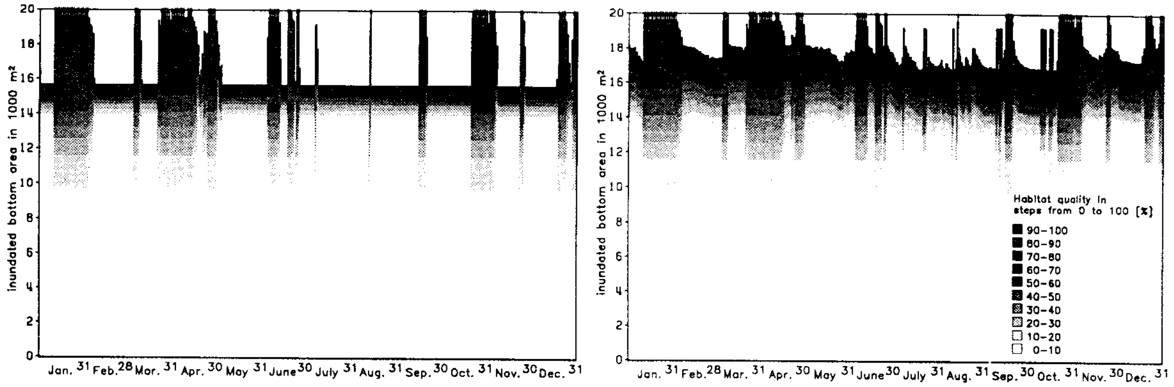


Figure 16: Temporal and spatial distribution of hydraulic habitat suitability classes for *Psychomia pusilla* in TS-B, minimum flow regulation MF-1 (left) and MF-2 (right).

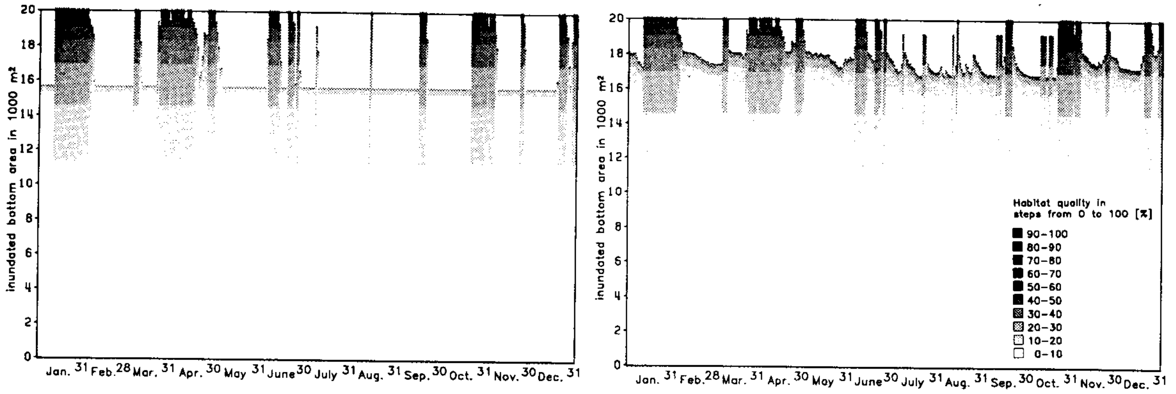


Figure 17: Temporal and spatial distribution of hydraulic habitat suitability classes for *Rhyacophila* cf. *dorsalis* in TS-B, minimum flow regulation MF-1 (left) and MF-2 (right).

For all species the MF-2 flow fluctuations in TS-B do not considerably affect habitat size and quality. Only very large fluctuations (comparison MF-1 conditions, MF-2 conditions, weir overflow) cause significant changes of the available habitat. Nevertheless, there is some dynamic and local disturbance due to the alternations of inundated area in TS-B which also also result in a local shift of certain hydraulic niches within the river bed. The morphologically highly variable TS-B flow fluctuations under MF-2 conditions cause a higher temporal and spatial variability of the flow pattern at the river bottom than in the regulated TS-A. TS-A requires much greater discharges to developp similar patterns. Hydraulic habitat simulations showed that the diversion of water causes an extreme distortion of available hydraulic habitat in favour of limnophilous species, and that this distortion can be compensed only to a small degree by minimum flow regulations.

From the simulations with CASIMIR it is concluded that minimum flow regulations have only a limited capacity to improve hydraulic habitat availability for rheophilous benthic species in both investigated river stretches. In the regulated section flow fluctuations cause only reduced heterogenity and dynamics in bottom shear stress pattern. It is therefore important to adapt minimum flow regulations to river bed morphology and eventually improve river bed morphology if high ecological standards are desired.

References

Ambühl, H. (1959): Die Bedeutung der Strömung als ökologischer Faktor. Schweiz. Zeitschrift für Hydrologie 21: 133-264.

Bovee, K.D. (1986): Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. - US Fish and Wildlife Service, Instream Flow Information Paper No. 21, US Fish and Wildlife Service Biologic Report 86(7), 235 pp.

Dittrich, A. & U. Schmedtje (1996): Indicating Shear Stress with FST-Hemispheres - Effects of Stream-Bottom Topography and Water Depth.-Freshwater Biology 34, pp.107-121.

DVWK (1995): Deutscher Verband für Wasserwirtschaft und Kulturbau, Merkblatt (Gelbdruck) "Gesichtspunkte zum Abfluß in Ausleitungsstrecken von Wasserkraftanlagen", Bonn.

Dyck S. (1977): Angewandte Hydrologie, Teil 1, 2. Auflage, Ernst & Sohn, Berlin.

Frutiger, A. & Schib J.-L. (1993): Limitations of FST hemispheres in lotic benthos research.-Freshwater Biology 30, pp.463-474.

Fuchs, U. (1994): Ökologische Grundlagen zur wasserwirtschaftlichen Planung von Abfluß und Morphologie kleinerer Fließgewässer. - Dissertation an der Fakultät für Bio- und Geowissenschaften, Universität Karlsruhe.

Giesecke, J. (1994): Kriterien für die Mindestwasserführung in Ausleitungsstrecken kleiner Wasserkraftanlagen.-Wasserwirtschaft 84 (6), S.338-341.

Jorde, K. (1996): Mindestwasserregelungen in Ausleitungsstrecken: Ein Simulationsmodell zur Beurteilung ökologischer und ökonomischer Auswirkungen.-Wasserwirtschaft 86(6), in prep.

Lamouroux N., B. Statzner, U. Fuchs, F. Kohmann & U. Schmedtje (1993): An Unconventional Approach to Modelling Spatial and Temporal Variability of Local Shear Stress in Stream Segments. - Water Ressources Research (28)12, pp. 3251-3258.

Mutz, M. (1989): Muster von Substrat, sohlnaher Strömung und Makrozoobenthos auf der Gewässersohle eines Mittelgebirgsbaches. - Dissertation, Universität Freiburg i.Br, 193 S.

Plate E.J. (1993): Statistik und angewandte Wahrscheinlichkeitslehre für Bauingenieure, Ernst & Sohn, Karlsruhe.

Rabeni, C.F. & G.W. Minshall (1977): Factors affecting microdistribution of stream benthic insects. Oikos 29, pp. 33-43.

Schmedtje, U. (1995): Beziehungen zwischender sohlnahen Strömung, dem Gewässerbett und dem Makrozoobenthos in Fließgewässern - Ökologische Grundlagen für die Beurteilung von Ausleitungsstrecken. Dissertation Institut für Zoologie und Limnologie, Universität Innsbruck, 231 S.

Schwoerbel, J. (1971): Einführung in die Limnologie, 6. Auflage, Gustav Fischer Verlag, Stuttgart.

Smith I.R. (1975): Turbulence in lakes and rivers, Freshwater Biological Association, Scientific publication No. 29, Edinburgh.

Statzner, B., F. Kohmann & A. G. Hildrew (1991): Calibration of FST-Hemispheres against bottom shear stress in a laboratory flume. - Freshwater Biology 26: 227-231.

Statzner, B. & R. Müller (1989): Standard hemispheres as indicators of flow characteristics in lothic benthos research. - Freshwater Biology 21: 445-459.

Townsend, C. R. (1989): The patch dynamics concept of stream community ecology. Journal of the North American Benthologial Society 8, pp. 36-50.

Ward J.V. & J.A. Stanford (1983): The intermediate disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems.- Dynamics of lotic ecosystems, T.D Fontaine & S.M. Bartell (Eds.), Ann Arbour Science, Ann Arboour, Michigan, pp. 347-356.

NATURAL/ARTIFICIAL FLOODS CONNECTED WITH RIVER HABITAT

Maximo A. Peviani
ISMES SpA
via Pastrengo 9, 24068 Seriate (BG)
Italy
E-mail: mpeviani@ismes.it

Italo Saccardo
ENEL SpA DRS CRIS
Corso del Popolo 245, 30170 Mestre (VE)
Italy
E-mail: saccardo@cris.enel.it

Alessandra Crosato
ISMES SpA
via Pastrengo 9, 24068 Seriate (BG)
Italy
E-mail: dta@ismes.it

Gaetano Gentili
GRAIA SRL
via Repubblica 2, 21020 Varano Borghi (VA)
Italy
E-mail: graia@interland.it

ABSTRACT

The present paper describes the basis of a large-scale analysis tool to assess the influence of hydraulic and morphological variations on river habitat. The methodology mainly concerns the assessment of changes due to natural/artificial floods in mountain areas. It joints the techniques usually applied by river morphologists with those used by biologists, with the common aim of preserving natural habitats when designing and managing hydraulic structures. It mainly consists of interfacing a hydro-morphological model and a habitat evaluation model.

Floods are short-term events having strong impacts on river networks, they produce both acute effects on the acquatic communities, by means of exposure to strong concentrations of suspended solids and high velocities, and long-term effects on the river habitat, related to the induced morphological changes and further long-term channel evolution. The effects on the riverine communities vary with the season in which the flood occurs, due to the presence of different development stages, like egg incubation and juveniles. Moreover the natural river regime itself has seasonal features. For this reason the same natural/artificial flood can be lethal, tolerable or even positive, (lack of water, high water temperatures...) depending on the moment in which it occurs.

KEY-WORDS: macrohabitat / river habitat / Alpine Region / mountain river / flood / suspended solids / river morphology / suitabily index / stress index / cover / transverse profiles / prediction model / Brown Trout / Grayling

1. INTRODUCTION

The habitat of a river stretch is governed by climate, surronding environment, quality of the water, as well as by the physical attributes of the water course. Any alterations of the hydro-morphological parameters, such as water depth, velocity field, sediment transport, river bottom composition, as well as any changes of the chemical and physical characteristics of the water, transform the river habitat. Besides, hydraulic and morphological variations may also influence parameters like water temperature and oxygen content, which contribute to the definition of river habitat.

River planimetry, as well as river cross-sectional shapes and substrate evolve continuously. This dynamic behaviour is related to both sediment transport and bank erosion processes. River morphology is therefore strongly dependent on the hydraulic and sedimentologic characteristics of the water course and is influenced by several factors, such as geology, vegetation cover and climate of the river valley. Morphological changes can be very slow and therefore detectable only by means of long-term observations, but also relatively fast, even sudden. In the last case they are related to flood events. Human interventions, such as deforestation, soil protection, river management, reservoir operations, sediment extraction and others, are always responsible for modifications of the hydraulic and morphological characteristics of the river. As an example, peaking reservoir operations are comparable with natural floods and can cause strong short-term changes, while the presence of the reservoir itself is responsible for long-term channel evolution through near-dam erosion and incision of the river bed.

A prediction tool for river hydraulic and morphological behaviour is therefore important for the study of river habitats. This is expecially true for mountain and piedmont rivers, where the hydraulic and morphological perturbations are the most limiting factors for the riverine habitat, usually being other factors, like organic water pollution, of a minor relevance. Such a tool could also be adopted for the analysis of the consequences of human interventions and for the selection of solutions and management policies which minimise the impact on river habitats.

Already existing tools for the analysis of river habitats are mostly related to the Instream Flow Incremental Methodology (Bovee, 1982, 1995) and have proved to be particularly adapted for detailed studies of short river reaches. These models are based on the assumption that the morphology of the water course remains constant and can be applied for a limited range of water discharges. The study based on a wider range of discharges would require an enormous amount of field observations. Besides, these models cannot be used to estimate the effects of peaking hydrological events, such as floods, which lead to strong variations of the water discharges and of the morphological and substrate characteristics. This is due to the impossibility of carrying on measurements during those events.

The need to perform spatial and temporal large-scale studies on rivers in the Alpine Region, taking into account both the hydraulic and morphological variations, led to the development of a new model, named HAFIMO, for the analysis of *macrohabitats,* to be used in combination with an already existing hydro-morphological model, MORIMOR (Di Silvio & Peviani, 1989, 1991). MORIMOR ia a *one-dimensional* model with mobile bed, which computes sediment transport as bed and suspended load. The output of the hydro-morphological model is a part of the input of the habitat model. The integrated use of both models has been designed to evaluate macrohabitat variations throughout a whole hydrological year, with particular enphasis on the study of the consequences of flood events.

Cross-stream variations of the physical parameters, like velocity and water depth, are responsible for the diversity of river habitat and therefore they cannot be neglected. For this reason the model HAFIMO uses transverse profiles for the description of cross-stream distributions. Since the hydro-morphological model is one-dimensional and therefore computes cross-sectional averaged values, post-processing procedures to estimates cross-stream variations are needed for the evaluation of river habitat.

The quality of the habitat is quantified by means of habitat suitability criteria based on the organisms adopted as biological indicators. For the study of peaking events also the acute effects on the riverine communities are estimated. The exposure to high suspended sediment concentrations is evaluated my means of a "stress index". Also velocity peaks and their effects are considered. Besides the model takes into account the different life stages of fishes (reproduction, incubation etc.), by means of calendars describing their typical biological year. The implication is that for certain species, such as Brown Trout and Grayling, the consequences of the same hydro-morphological event depend on the period in which it occurs. This allows for a better definition of river management policies, regarding for instance reservoir operations, water widthdrawals and sediment extractions, with the aim of preserving river habitat.

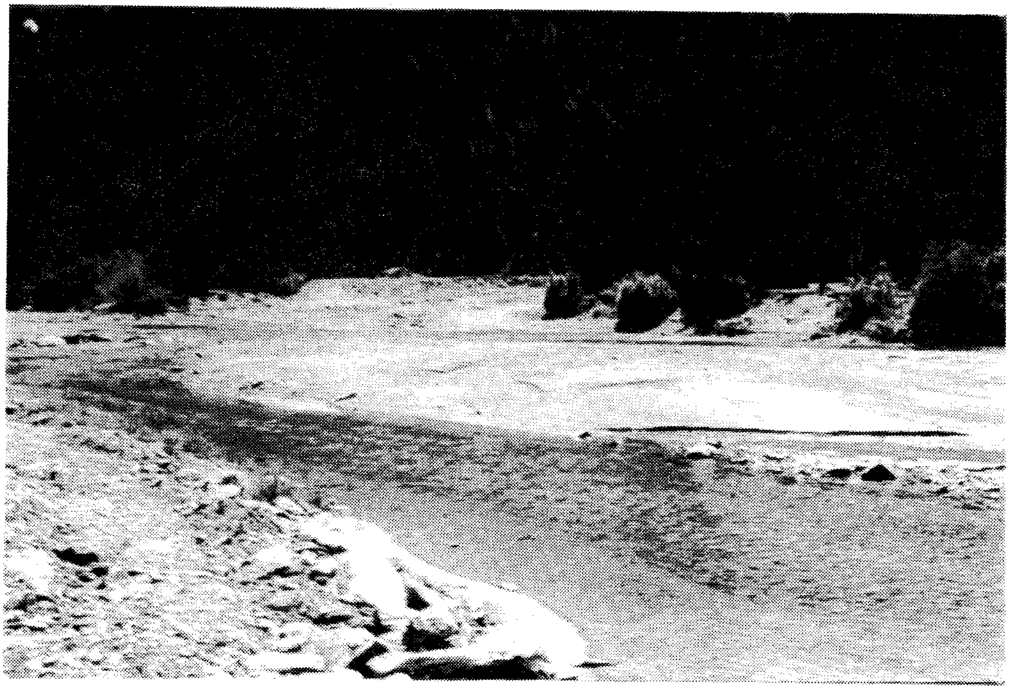


Figure 1: Typical pattern of Alpine rivers

2. MAIN CHARACTERISTICS OF ALPINE RIVERS

From the hydraulic and morphological point of view, mountain and piedmont rivers have a strongly dynamic behaviour. This is mainly due to their typically steep discharge hydrographs and to the strong time-variations of amount and gradation of sediment inputs. Mountain rivers mainly have coarse beds and steep longitudinal slopes with alternating pool/riffle patterns. Their planimetry is almost straight with only a few bends, which are mainly related to the geological characteristics of the valley. Piedmont rivers have milder slopes and typically braided patterns. They flow within an alluvial braid belt which participates in conveying water during high flow conditions. Floods in mountain and piedmont rivers are characterised by extreme peaks of velocity and suspended solid concentrations. In piedmont areas one of the consequences of floods can be river bed aggradation, mainly caused by deposition of fine sediments. Figure 1 shows a typical pattern of Alpine rivers.

Alpine rivers mainly present low temperature waters with a high oxygen content. The organic and industrial pollution is generally low. For what primary producers are concerned, the biological community is represented, almost exclusively, by peryphiton. This, together with the organic debris falling into the water, is the alimentary basis for the primary consumers, and in particular for the macrobenthic invertebrates. Secondary consumers are some species of macro-invertebrates and fishes. In the Alpine Region fishes are represented by a low number of species, mainly belonging to Salmonids, which require cold, well oxygenated and running waters. Macrobenthic invertebrates are represented by groups which are well adapted to stream currents, but exigent in terms of water oxygenation, such as Plecoptera, Ephemeroptera and Tricoptera.

3. PROPOSED METHODOLOGY

The proposed methodology consists of: i) the simulation of a certain hydrological event with the hydro-morphological model MORIMOR, ii) the evaluation of its consequences on river macrohabitat and riverine communities with the habitat model HAFIMO. The applicability is restricted to large-scale studies of *mountain and piedmont rivers* in the Alpine Region. The scheme of the interfacing of the two models is presented in Figure 2.

3.1 The hydro-morphological model

The hydro-morphological model MORIMOR has been especially designed for large-scale simulations (basin scale) of river networks in mountain areas. It is one-dimensional and time-dependent, for every computational cross-section and at any time-step, it simulates: hydraulics, sediment transport (bed and suspended load, granulometries) and channel morphology (water depth, longitudinal slope, bed level variations), taking into account discharge/width relations (based on observations). In case of rivers having compound sections, the model can treat a channel subdivision.

The code is based on a one-dimensional mathematical description of motion and continuity of water and sediment. The sediment is assumed non-uniform. The set of theoretical equations, reported in Di Silvio & Peviani ((1991) is numerically solved using a finite differences approximation. Besides the upstream and downstream boundary conditions, the model allows for lateral and internal boundaries, which can represent phenomena like rocky bottom, water and sediment inputs.

The implementation of the model requires the topographical and sedimentological descriptions of the drainage system (planimetry, cross-sections, grain size distributions of the bottom material) and the definition of initial and boundary conditions (granulometry and amount of sediment inputs, discharge hydrographs).

3.2 The habitat model

River habitat is described by means of two types of physical parameters, those resulting from the hydro-morphological simulation: velocity, water depth, granulometry of substrate, suspended sediment, bed level and longitudinal slope; and those deriving from direct measurements: temperature, cover and curvature of the river axis (temperature can also be computed by means of an independent model).

The habitat model HAFIMO evaluates habitat quality and its variations as a function of the hydro-morphological parameters. Water temperature and cover also contribute in quantifying habitat quality. The model also estimates the effects of peaking conditions on the acquatic communities.

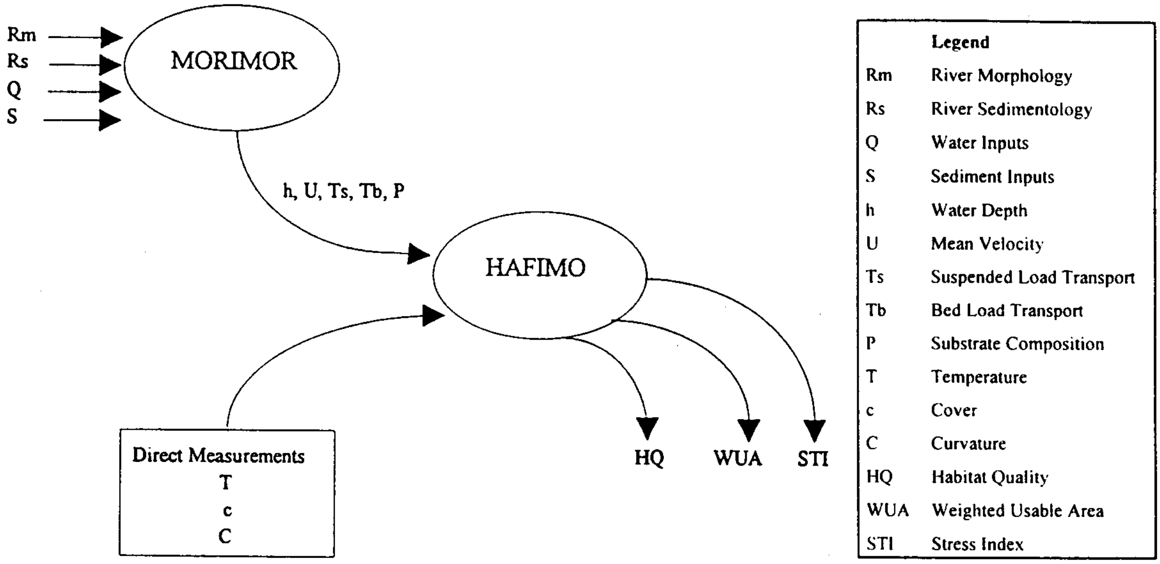


Figure 2: Sheme of the interfacing MORIMOR-HAFIMO.

3.2.1 Evaluation of Habitat Quality

Macrohabitat quality is quantified through the aggregation of the results obtained from the analysis of the parameters describing the microhabitat. The river reach is divided in cells, each of them characterised by a certain microhabitat. The physical parameters of the cell are compared against the habitat suitability criteria for a particular organism, chosen as biological indicator. Each species has its own requirements, with respect to habitat characteristics, and therefore the result is species-dependent. The values of each parameter (flow velocity, water depth, substrate, water temperature, etc.) are associated to a Suitability Index, SI, ranging between zero (unsuitable value) and one (optimal value).

A Compound Suitability Index for a certain biological indicator can be defined as:

$$CSI = (SI_{p1})(SI_{p2})(SI_{p3})...(SI_{pn})$$

In which SI_{p1},...SI_{pn} are the Suitability Indexes relative to parameter1... parameter n, respectively. CSI can be regarded as the weight to be attributed to each cell for a whole habitat quality evaluation. The sum of all cells in terms of wet area, multiplied by their weight, in terms of habitat quality, gives a "meter" with which to "measure" habitat quality of the river reach as a whole. The result is a fraction of the wet area of the whole river reach, A, which can be regarded as the available habitat for a certain species, and is called Weighted Usable Area:

$$WUA = \Sigma A_i CSI_i$$

In which A_i is the area of cell i and CSI_i is the Compound Suitability Index for cell i. This is the same approach as that adopted in the Instream Flow Incremental Methodology (Bovee 1982, 1995).

In order to allow an easy comparison among river reaches of different sizes, a dimensionless parameter HQ is defined:

$$HQ = WUA/A$$

The typical river segment to be studied with the proposed methodology has a length up to 25 km. The river segment is divided into reaches having length up to 250 m. Every river reach is assumed uniform. In transverse direction the cross-section is divided in 10 cells.

For every biological indicator, the model applies the suitability criteria of the life stages which can be found at the moment corresponding to the given physical conditions and computes:

- for each river reach in which the river segment is divided: duration curves of HQ and WUA and time-dependent values (the critical parameter is also indicated);
- for the whole river segment: averaged values of HQ and WUA.

3.2.2 Estimation of the effects of acute conditions

In case of *peaking events,* such as floods and flushing of reservoirs, it is important to quantify habitat quality *before* and *after* the event, but it is also necessary to estimate the effects of the acute conditions, which are present *during* the event. Acute conditions, such as high concentrations of suspended sediments and high velocities, can have catastrophic effects on the biological community.

In general the effects of suspended sediments on the living organisms are a function of concentration and duration of exposure. The effects on macrobenthic invertebrates can be: damages of their tissues due to the erosive action of suspended particles, suffocation, and catastrophic drift. The effects on fishes can be summarised as follows: *Behavioural:* modification of the behavioural characteristics; *Sub-Lethal:* physiological stress, damage of tissues; *Lethal:* mortality, size of population reduced.

Newcombe & MacDonald (1991) defined a Stress Index, which can be used to express the effects of suspended sediments on the organisms:

$$STI = \ln (C^*D)$$

in which C = Concentration (mg L^{-1}) and D = Duration (h).
For the different biological indicators the effects of the exposure to suspended sediments have been experimentally determined (though not exhaustively) as a function of the Stress Index.

Besideds that, high flow velocities can cause a catastrophic drift. The severity of the effects depend on value and duration of the extreme conditions and on the resistance of the organisms. For example, fishes (juveniles and adults) can better resist, because they can swim and find recover in zones were the velocity is lower. In the model the maximum velocity having a duration of at least two hours is compared to the resistance of the different biological indicators. The critical duration of two hours has been arbitrarily chosen assuming that for a minor duration fish could resist (i.e. finding recover).

The estimation of the effects of suspended sediments and high velocities concerns only the study of events occurring on short time-scales (days), like floods and peaking reservoir operations, for which the required computational time-step duration has a maximum of ten minutes. Acute conditions cannot be averaged on longer time intervals, like the computational time-steps required for studies on longer time-scales (order of magnitude of several days).

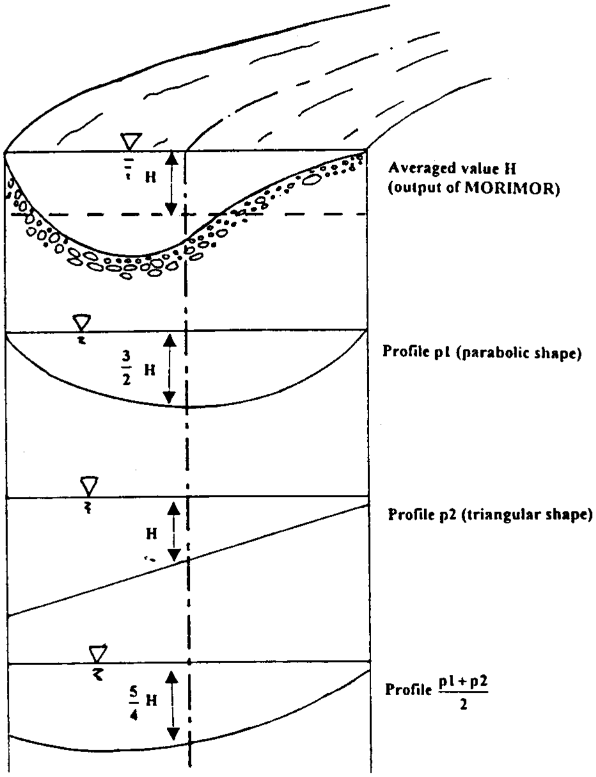


Figure 3: Schematisation of transverse bed levels variations by means of transverse profiles.

3.2.3 Post-processing of computed hydraulic parameters

The idro-morphological model is one-dimensional and therefore for each parameter it computes one single averaged value per cross-section. This is a strong approximation for the simulation of river hydro-morphologic behaviour. Moreover transverse variations proved to be of basic importance for the description of river habitats. This is true, for example, when establishing whether for a certain species the river reach is suitable for colonisation: if average conditions are unsuitable, near-bank conditions can be acceptable. Transverse variations of velocity are decisive in case of extreme events: even when the average value is above the resistance limit, certain fishes (adults or juveniles) can still resist close to the river banks, where the velocity is lower. For this reason the model HAFIMO is provided with transverse profiles which can approximate the cross-sectional distribution of stream velocity and water depth. Transverse profiles are based on equilibrium hypotheses and are strictly valid only in theoretical cases. Their range of validity is therefore restricted to those cases in which the real channel is not too far from the theoretical one.

For uniform channels, velocity and water depth variations are assumed to follow a profile which takes into account: i) the typical cross-sectional distribution of velocity and water depth in straight channels, which presents lower values close to the river banks and maximum values close to the channel centre-line, by means of a parabolic profile; ii) the influence of channel curvature, which is responsible for the typical triangular shape of cross-sections when the flow is subcritical, by means of a relation derived for fully developed bend flow Crosato (1990). The adopted transverse profiles are schematised in Figure 3. The concentration of suspended solids and the granulometry of the substrate are assumed uniform in the cross-section.

3.2.4 Biological indicators

The biological indicators which are adopted in the model HAFIMO are chosen among both fishes and macrobenthic invertebrates. Fishes are represented by the Brown Trout (*Salmo trutta*), which is the dominant species in alpine rivers, and by the Grayling (*Thymallus thymallus*), which is the characteristic species of piedmont reaches. Both the Brown Trout and the Grayling are intolerant to water quality alterations and to hydraulic and morphological perturbations. Adults of Brown Trout have a marked territorial behaviour and require, more than any other species of Trout, the presence of cover. On the contrary the Grayling is a gregarious species and do not need cover. Both species eat prevailingly macroinvertebrates. The biological calendars indicating the life stages of these fish species are given below.

Table 1: Brown Trout (*Salmo trutta*)

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
adult	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
juvenile	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
fry	◨	▨	▨	▨	▨	▨	▨	◧				
incubation	▨	▨	▨	◧							▨	▨
reproduction	▨										▨	▨

▨ = present

Table 2: Grayling (*Thymallus thymallus*)

	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
adult	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
juvenile	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨	▨
fry				◨	▨	▨	▨	▨				
incubation				▨	▨	◧						
reproduction				▨	▨							

▨ = present

Macrobenthic invertebrates are represented by the orders Plecoptera, Ephemeroptera, Trichoptera, Coleoptera and Diptera, which show modifications, in terms of abundance and relative distribution, to habitat alterations. Besides also two species of macroinvertebrates, Ephemerella and Hydropsyche, are adopted as biological indicators.

For the Brown Trout and the benthic macroinvertebrates, the adopted Suitability Indexes are based on the work of Gowan (1984) and of Raleigh et al. (1986), for the Grayling on the work of Hubert et al. (1985). Due to the lack of significant European contributions, all the tables adopted in the model HAFIMO are based on North American data. Therefore their applicability to European cases requires further studies.

The effects of suspended sediments on the chosen species have been summarised in tables, based on the data reported by Newcombe (1994). For the Brown Trout data proved to be scarce, and therefore they had to be integrated with those regarding the Rainbow Trout. The effects on Grayling had to be extrapolated from data collected for the Arctic Grayling. The study is far from being finished, observations on European species and environment, and further studies are needed.

3.2.5 Cover

Cover can be described as: hiding-place, refuge from predators, refuge from adverse conditions (high stream velocity). Cover can be provided by the presence of a boulder, a trunk of a tree, roots, a hole, etc. During peak conditions cover can be also the near-bank area where velocity is lower. Cover is defined in terms of usable area, that is in m^2.

The presence of cover increases with the granulometry of the substrate and with the water depth. In general the availability of cover varies with the discharge and therefore cover is a parameter of difficult estimation. Furthermore it requires field observations, since it cannot be computed by means of a hydro-morphological model and it should be evaluetad for every river reach.

Cover is a parameter of high importance for the quality of habitat. This is especially valid for the Brown Trout, for which it is one of the parameters to be taken into account for the habitat description. Cover has also a general role of decreasing the severity of the effects of extreme conditions and in defining microhabitats characterised by low velocities. Often, in preliminary model tests, stream velocity has proved to be the critical parameter for the habitat and sometimes this result was not realistic. Mountain and piedmont rivers have indeed high values of average velocity, but, on the other hand, they present also strong channel irregularities (small bends, holes, stagnation areas caused by instream obstacles, like boulders) where the velocity is lower and where the presence of some less-resistant species is evident. Those channel irregularities can be compared with the area usable as cover. They cannot be predicted by models and cannot be reproduced by means of transverse velocity profiles. Taking into account the transverse variations of velocity is anyway the first requirement, but this sometimes still gives unrealistically critical results. For this reason in the proposed methodology cover has been introduced in the estimation of SI_v (Suitability Index for velocity) through the following expression:

$$SI_v = SI_{v0}\,(A_i - A_{ic})/A_i + A_{ic}/A_i$$

In which A_i is the wet area of cell i; A_{ic} is the area usable as cover in cell i (for simplicity cover is assumed uniformly distributed in the cross section); SI_{v0} is the Suitability Index based on the value of computed velocity in the cell and valid for the area (A_i-A_{ic}); SI_v is the revised Suitability Index. The above relation implies that a

Suitability Index equal to one (optimal) is assigned to the area A_{ic}. The Weighted Usable Area with respect to velocity increases and has a minimum value equal to that of the area defined as cover.

The proposed relation gives the possibility of estimating the effects of drastical changes of cover availability, which can derive from river canalisation, bank protection works etc. It allows also for a more realistic determination of the effects of peak velocities during floods. In this case cover is assumed to take the role of refuge from adverse conditions, for *juvenile* and *adults* of fish only (it is assumed that the other life stages, eggs and fry, and macroinvertebrates cannot swim and therefore do not easily benefit from refuge areas).

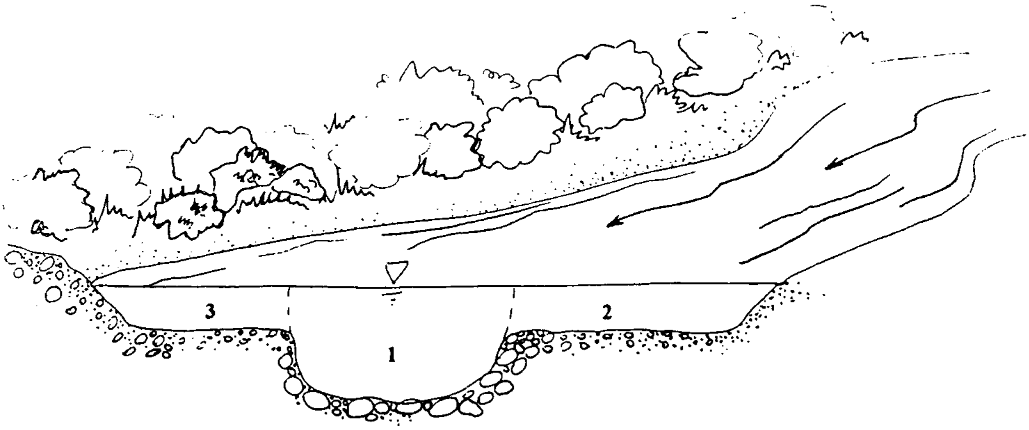


Figure 4: River with a compound section, subdivision in parallel channels.

4. APPLICATIONS

The study of river habitat changes regards short- to long-term simulations, in which the hydro-morphological parameters and habitat quality are computed at every time-step. Short-term simulations (duration of days) regard natural/artificial flood events. Medium-term studies (duration of months) regard a whole hydrological year or season, which can be based on averaged, measured or hypothetical hydrological data. Long-term simulations (duration of tens of years) generally regard environmental impact studies. In this case the evolution following human interventions should be compared with the initial conditions and with the natural trend. The short-medium-long-term morphological evolutions are reproduced by means of the hydro-morphological model. The comparisons are based on the analysis of habitat quality for a whole hydrological year, a single event, or the most critical situation.

When the river has a typical compound section, with a main stream and lateral channels (Figure 4) the whole channel can be divided into sub-channels. The model MORIMOR computes all the hydro-morphological parameters for every sub-channel. Every sub-channel will then be treated separately for the evaluation of habitat quality. In this case it is not advisable to apply the transverse profiles as described in Section 3.2.3, which are valid only for uniform channels.

The possible applications of the integrated model are:

- the study of the effects of natural floods on habitat quality and on the biological community;
- the study of habitat quality evolution throughout a whole (ordinary or exceptional) hydrological year or a season;
- the study of the effects of peaking reservoir operations on habitat quality and on the biological community;
- the study of the effects on habitat quality of river canalisation and bank protection works;
- the study of the effects on habitat quality of sediment extraction (deliver) from (into) the river channel;
- the study of the effects on habitat quality of river hydraulic regime modifications (water withdrawals, reservoir operations) and optimisation of water management policies with respect to the minimisation of the impact on river habitats;
- the study of long-term habitat modifications, due to slow hydraulic and morphological evolutions.

5. FURTHER DEVELOPMENTS AND DISCUSSION

The development of the presented methodology is still at an initial stage. Up to now the integrated model MORIMOR-HAFIMO has only been tested on hypothetical cases. Applications on existing rivers are needed for further improvements and calibrations.

The applicability of the adopted North American data to European cases must be assessed.

Habitat diversity cannot be neglected, and therefore cross-sectional variations must be estimated. Discussion is still open about how to take into account cross-sectional variations of velocity and water depth investigating the more convenient shematisation. This can result from theoretical transverse profiles and from field observations. Field measurements are therefore needed.

The need of vertical profiles for the estimation of the near-bottom velocity (important parameter for microhabitat of invertebrates) is also under discussion.

A preliminary conclusion at this stage of development is that the role of cover is of great importance for the definition of habitat and for the consequences of high velocities on the riverine communities. Unfortunately this parameter is difficult to estimate. A sensitive analysis of cover estimation is needed. Standard criteria for its estimation are also needed. Discussion is still open about how to take into account the influence of cover on defining macrohabitats, also in relation to velocity, and on decreasing the effects of velocity peaks.

REFERENCES

Bovee, K. D. 1982. A guide to stream habitat analysis using the Instream Flow Incremental Methodology. Instream *Flow Information Paper 12. U.S. Fish and Wildlife Service*, FWS./OBS-82/26.248 p.

Bovee, K. D. 1995 (editor). A comprehensive overview of the Instream Flow Incremental Methodology. *National Biological Service*, Fort Collins, Colorado, U.S.A.

Crosato, A. 1990. Simulation of meandering river processes. *Communications on Hydraulic and Geotechnical Engineering*, Report No 90-3, ISSN 0169-6548, Delft University of Technology, Faculty of Civil Engineering, The Netherlands.

Di Silvio G. & Peviani M. A. 1989. Modelling short- and long-term evolution of mountain rivers: an application to the Torrent Mallero (Italy). *Proceedings, International Workshop on Fluvial Hydraulics*. Trent, Italy.

Di Silvio G. & Peviani M. A. 1991. Transport of a mixture of sand and gravel in suspension and as bedload. Experiments and mathematical modelling. *Proceedings of the symposium on: The transport of suspended sediment and its mathematical modelling*. Florence, Italy, September 1991.

Gowan C. 1984. The impacts of irrigation water withdrawals on Brown Trout and two species of benthic macroinvertebrates in a typical southern Michigan stream. Master Science Thesis, Michigan State University, Department of Fisheries and Wildlife.

Hubert W. A., Helzner R. S., Lee L. A. & Nelson P. C. 1985. Habitat suitability index models and instream flow suitability curves: Arctic Grayling riverine populations. Biological Report 82(10.110). Wester Energy and Land Use team, Division of Biological Services, Research and Development, Fish and Wildlife Service, U.S. Department of the Interior, Washington DC 20240.

Newcombe C. P. 1994. Suspended sediments in acquatic ecosystem: III effects as a function of concentration and duration of exposure. British Columbia Ministry of Environment, Land and Parks, Habitat Protection Branch. Victoria, British Columbia, Canada, 298 pp.

Newcombe C. P. & MacDonald D. D. 1991. Effects of suspended sediments on acquatic ecosystems. In: *North American Journal of Fisheries Management*, No. 11, pp. 72-82.

Raleigh R. F., Zuckerman L. D. & Nelson P. C. 1986. Habitat suitability index models and instream suitability curves: Brown Trout. Biological report 82(10.124). National Ecology Center, Division of Wildlife and Contaminant Research, Research and Development, Fish and Wildlife Service, U.S. Department of the Interior, Washington DC 20240.

INTEGRATED TWO-DIMENSIONAL MACROPHYTES-HYDRODYNAMIC MODELING: APPLICATION TO LAKE SAINT-FRANÇOIS (ST.LAWRENCE RIVER, QUÉBEC, CANADA)

Jean Morin, Michel Leclerc, Yves Secretan and Paul Boudreau

Institut national de la Recherche scientifique - Eau
2800 Rue Einstein, C.P. 7500, Sainte-Foy (Qc),
Canada, G1V 4C7
Jean_Morin..., Michel_Leclerc..., Yves_Secretan..., Paul_Boudreau@inrs-eau.uquebec.ca

ABSTRACT

Lake Saint-François is the first fluvial lake downstream of the Great Lakes; this complex fluvio-lacustrine system reaches 8 km in its wider part, and extends for over 50 km. Aquatic macrophytes are abundant partially because of water level stabilization and increased nutrient loading from the Great Lakes. The influence of plants on flow modification during summer is very important and with a magnitude similar to ice in winter. In Lake Saint-François, emergent plants and especially submerged species have to be considered in order to simulate accurately hydrodynamic conditions.

Spatial distribution of plant species is linked to their instream flow preferences. This aspect is well shown by the occurrence of growth forms representing adaptation to abiotic conditions. Several abiotic variables are of importance: light penetration, wave energy, current velocity, nutrients in substrate and substrate physical characteristics are, at different levels, controlling the species and their biomass.

Field characterization of macrophytes was performed using echosounder transects in association with a submersible video camera. This technique allowed us to calibrate the echofacies for species identification and relative proportion, plant height and density. Eleven main assemblages were recognized on 83 different echofacies. These assemblages appear to be strongly correlated with abiotic conditions.

A basic interpretation key was set up in order to describe plant distribution over the entire lake. A precise bathymetric map has been built from a 292 000 points hydrographic database. Species, relative proportion, density and plant height were then interpolated between transects. These informations were used for adjusting the Manning's friction coefficient for each assemblages. Simulations of the flow fields with plants and in absence of plants show a contrasted pattern. During the summer (with plants), the flow is mainly concentrated in deep channels where velocities are clearly increased by approximately 20%, compared to spring-fall simulation (without plants).

KEY-WORDS: Two-Dimensional model / Macrophytes / Aquatic plants / Habitat modeling / Lake Saint-François / St. Lawrence River / Manning's coefficients / Hydrodynamic model / Physical habitat / Finite elements method / Québec / Canada

INTRODUCTION

Context

Lake Saint-François is a complex fluvio-lacustrine system that has been submitted to important modifications associated with industrialization. Politically, this part of the St. Lawrence River plays the role of a border; it is shared by two Canadian provinces, Ontario and Québec and by New York state (Figure 1). Lake Saint-François is the first fluvial lake of the St.Lawrence River, downstream of the Great Lakes. It is an important widening of the river. The lake has a maximum width of about reaches almost 8 km on its larger part, and nearly 50 km in length, covering a total area of 233 km^2 (Allan 1986; Lorrain *et al.* 1993). The mean flow discharge of the river at Cornwall is 7500 m^3/s, creating strong currents responsible for the lack of stratification in the lake.

Morphologically, Lake Saint-François can be subdivided in three main parts (Figure 2): a multi-channel riverine reach with large islands forming an archipelago at its upstream part; a middle part which presents the aspect of a sub-fluvial breaded reach, with deep channels and shallow water flat zones; and a lacustrine section on the downstream side where mean depth stands around 10 m with a 21 m deep navigation seaway. This complex morphology strongly determines the flow distribution over the lake. Water levels are heavily regulated for navigation purposes, by hydropower stations located upstream and downstream of the lake. The pristine regime used to allow a 60 cm annual variation of the level. Regulation reduced this interval to less than 12 cm on an annual basis which favoured luxuriant submerged plant growth.

The lake is covered with ice during winter, a factor which contributes to particular flow patterns during this season. Figure 3 illustrates the seasonal energy slope of the river in relation to the presence of ice during winter. The river increases its water level difference between upstream and downstream to compensate ice resistance and to allow an efficient flow transit through the system. Similarly, during summer, aquatic plants grow on area where physical conditions are acceptable or suitable for them. This results in a transient reinforcement of the flow patterns already present in the lake due to its morphology. Figure 3 also depicts this influence. For hydrodynamic modeling purposes, taking into account the aquatic plant resistance to flow is an absolute necessity in order to provide reliable results for application related to river enhancements, or for waste disposal and to water quality management.

This research focuses on the modeling strategy adopted to represent the distribution of aquatic plants and the effect of plants within a finite element hydrodynamic model, taking into account that the relation between plants and flow is non-linear from a modeling point of view. In fact, plant distribution is dependent on the flow field to grow and, at the same time, plants slow down the currents due to their resistance, thus contributing to the establishment of their own suitable microhabitat. This work is the first step toward a microhabitat modeling of aquatic plants.

This project was realized in the context of a multidisciplinary research program entitled "Ecosystem Recovery on the St. Lawrence", a multiuniversitary initiative lead by the Research Institute for Environment and Economy of the University of Ottawa and funded by the Canadian federal government "Three Councils Ecoresearch program".

This article presents successively the theoretical background behind plant suitable microhabitat modeling and its basic application in the cartography of aquatic plant distribution, hydrodynamic modeling and its formal aspects related to flow resistance, validation considerations, some of the numerical tools involved in the project, and finally, some hydrodynamic results, unfortunately not yet validated.

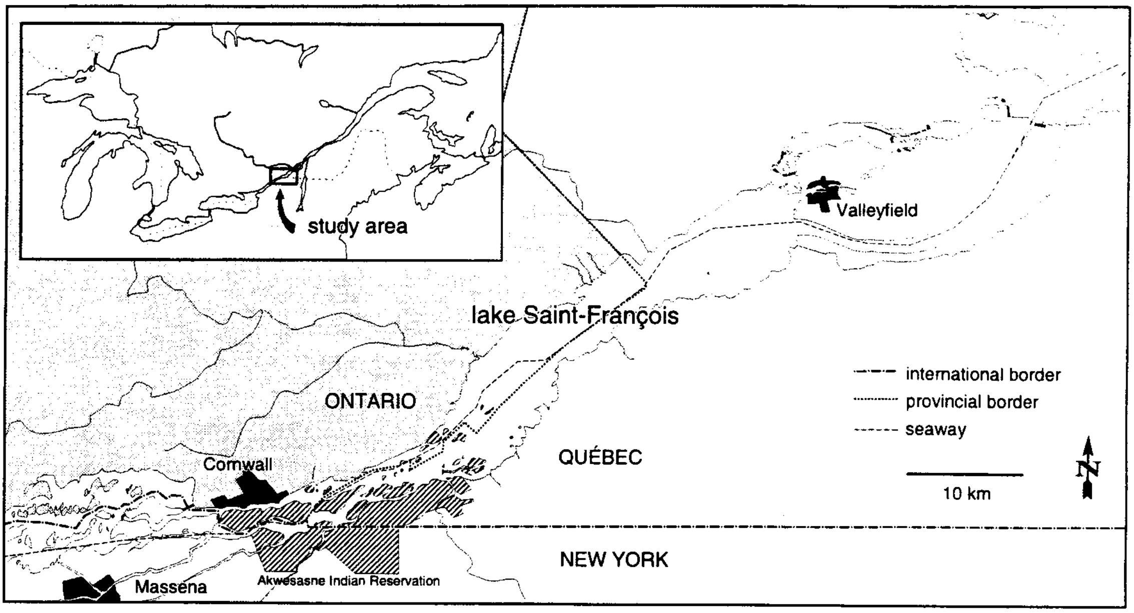


Figure 1: Location of Lake Saint-François (St. Lawrence River) (After Morin *et al.* 1994).

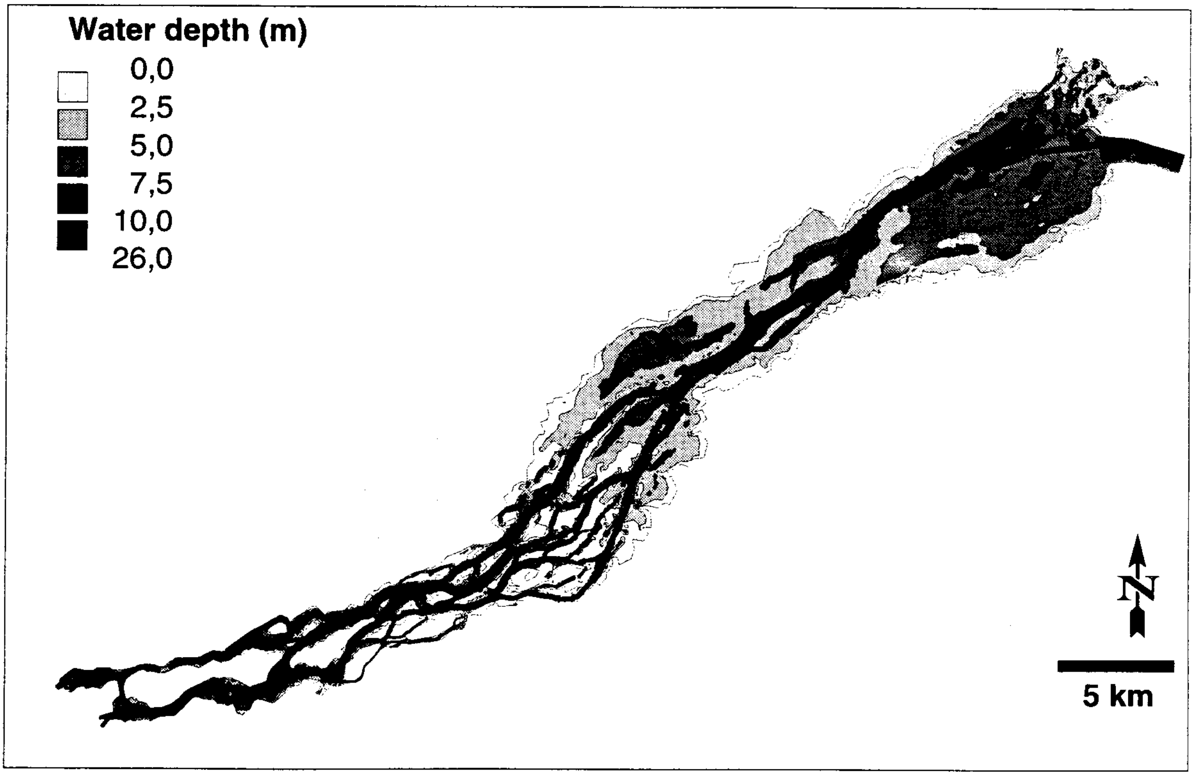


Figure 2: Bathymetry of Lake Saint-François

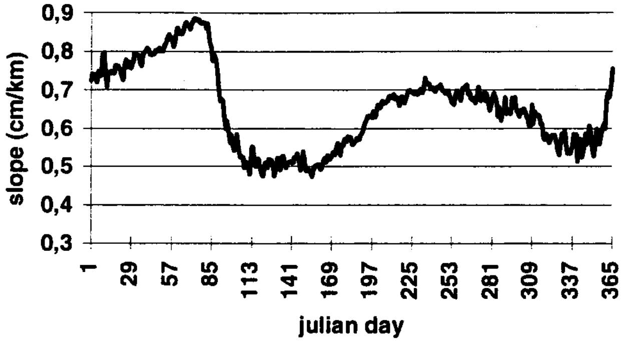


Figure 3 : Seasonal evolution of the energy slope in relation to ice and plant resistance in Lake Saint-François (After Morin *et al* 1994)

TOWARDS SIMULATION OF THE SUITABLE MICROHABITAT OF AQUATIC PLANTS

Plant ecology and abiotic factors

One can distinguish two main groups among aquatic plants, those which float freely with currents, and those which are rooted in the substrate. The second group have a significant influence on the currents by resisting to the flow. This second group splits again into three classes (Sculthorpe 1967; Wetzel 1975): plants with floating leaves (0.5 m <depth < 3 m), plants emerging above the water surface (depth < 1.5 m) and submerged plants growing to almost any depth where light penetrates. For hydrodynamic modeling purposes, only the last two really play a significant role on the main water body. From an ecological point of view, the emergent plant constitutes the ecotope between terrestrial ecosystem (eventually wetlands) and the hydrosystem.

The submerged plants were classified by using a distinction based on the growth form (Figure 4): *canopy-producer, erect-form, rosette-form, bent-down form* and *bottom-dwellers*. The canopy-producers concentrate their photosynthetic biomass near the water surface and their stems can reach 4.5 m. Erect-forms have long stems which can reach 4 m but are not forming canopy. Rosette-forms have their photosynthetic biomass located close to the bottom, their maximum length is 1.5 m but they do not reach the surface. The bent-down form is typical of an environment dominated by currents, their height is less than 1 m but could reach 3 m in length. Finally, bottom-dwellers are less than 0.5 m in height and form dense mat.

This classification is much more helpful to modelers than any other because it is more closely related to abiotic conditions (Chambers and Kalff 1987). For example, canopy-producers are growing in nutrient rich substrate while bottom-dwellers are favoured by low-nutrient substrate. The growth form could be considered as an adaptation to the abiotic conditions since rosette-forms are resistant to wave action and the bent-down forms are adapted to fast currents. Chambers and Kalff (1987) and Chambers (1987) mention that some species sharing the same growth form have also comparable abiotic preferences for light (depth) and nutrients (substrate). Inversely, similar abiotic conditions would generate similar growth forms, a principle that allows to start habitat modeling from physical conditions as independent variables.

Spatial distribution of macrophytes is related to their instream flow needs (or preferences) and consequently, to abiotic conditions in the water body. Figure 5 depicts the complex network of relationships that governs the interactions between submerged plants and their habitat in the aquatic environment (after Morin 1995). The main variables which contribute to habitat selection by plants are the substrate (in relation to nutrients availability), light penetration (turbidity & depth), wind stresses and waves (or, more simply the wind fetch), flow velocity, substrate (in relation to rooting) and seasonal water level variations, riverbed slope, etc...

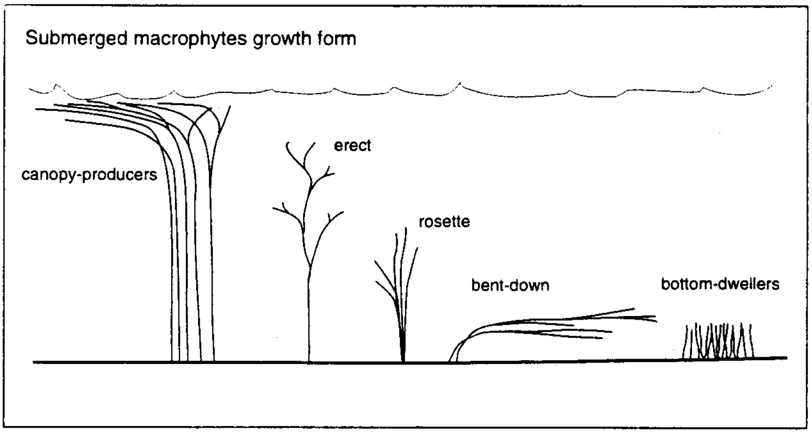


Figure 4: Schematic representation of aquatic plants growth forms

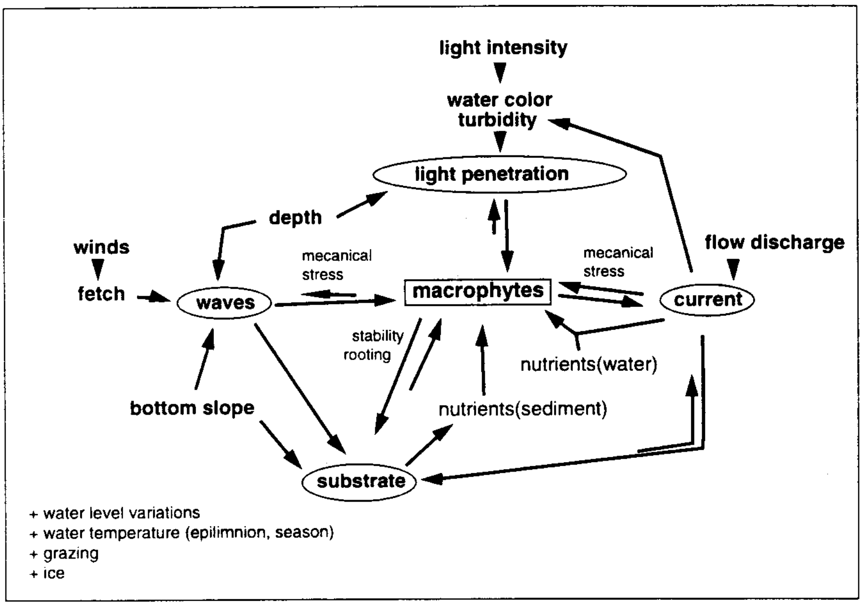


Figure 5: Conceptual representation of relationship between aquatic plants and their microhabitat

Role of abiotic factors

According to Duarte and Kalff (1986), *riverbed slopes* can affect plants microhabitat selection by indirectly influencing thickness of sediment, grain size and stability of deposits. Wave dissipation is also related to this factor (Duarte and Kalff 1988). In general, macrophytes seem to prefer slope conditions inferior to 15%. An adequate Numerical Field Model can easily provide the spatial distribution of this information in the flow domain.

Waves play a significant role in habitat suitability for plants. In fact, they apply mechanical stresses on them, but they also contribute to seed propagation (Keddy 1982). Waves affect directly the sedimentary dynamics within shallow water zones. When not controlled by current velocities, grain size and substrate composition are strongly correlated to wave energy dissipation on riverbed (Hakanson 1977; Keddy 1982, Pettigrew and Kalff 1991). Plants can also contribute to wave damping (Kobayashi *et al* 1993; Camfield 1977). In absence of a wave model, the wind fetch can be used in order to roughly estimate this factor. Again this data can be generated by spatial analysis within terrain morphology analytical tools.

Light penetration is related to turbidity and depth. It determines the maximum depth in a particular water body at which plants can grow (Dale 1986; Spence 1982; Wetzel 1975). The capacity to assimilate light energy varies from a plant category to another, a characterisic that determines their distribution (Chambers and Kalff 1985). Depth is provided directly by coupled NFM - hydrodynamic models. Turbidity may be interpreted as a global property of water bodies within a river (accounting for tributary plumes), or simulated with a sedimentation model able to calculate the concentration of suspended load.

Current velocity or turbulence can inhibit macrophyte growth by breaking leaves, uprooting, or making rooting difficult. Presence of currents reduces plant biomass and diversity without regards to substrate composition (Chambers *et al* 1991). Virtually, no plant can grow in water bodies with velocities higher than 1 m/s. However, macrophytes can modify the flow distribution by offering resistance to flow (Pettigrew and Kalff 1992). This resistance depends on the total surface of leaves. Hydrodynamic models can predict precisely current velocity as long as the resistance is well accounted for in the model parametrization.

Nutrients can also be a dominant factor for plant species distribution (Anderson and Kalff 1986). Sediment is the main source of nutrients to plants, over direct assimilation in the water column (Barko and Smart 1980; Carignan and Kalff 1980). As for currents, macrophytes contribute to sedimentation processes by reducing velocities and facilitating deposition of fine material which usually carry adsorbed nutrients, and their own organic matter after senescence. This effect results in increased habitat value for plants (Chambers and Prepas 1994; Pettigrew and Kalff 1991). The nutrient content in substrate is related to wave action and to current velocity (Chambers and Prepas 1990) implying that nutrients are associated to fine particles, and that this type of material settles in protected areas (Wilson and Keddy 1985).

Substrate composition, as previously stated, plays significant physical and chemical roles for plant habitat selection. Penetration and anchoring of roots necessitate a light and stable substrate. Thus, zones subject to erosion are not favorable conditions as plant microhabitat (Madsen and Adams 1989). Finally, *water level variation* has an important influence on macrophytes by exposing their structures to the air or to freezing.

Macrophytes in Lake Saint-François

In Lake Saint-François, macrophytes appear to have benefited, both in terms of diversity and biomass, from increased sources of nutrients related to human activity, and from water level regulation for navigation and hydropower generation (Owen and Wile 1975). The creation of very favourable abiotic conditions for plant growth were caused by 1) constant depths by stabilization of water levels, 2) alteration of current velocity by dredging, 3) reduction of ice scouring, 4) and by the anthropogenic increase of nutrients. The actual equilibrium in submerged plant composition includes mainly species listed in Table 1. *Scirpus* sp. is the most abundant emergent plant observed in the fluvio-lacustrine zone.

Table 1: List of most abundant species of submerged macrophytes in L. Saint-François and growth form

Growth form	Species
canopy-producer and erect	*Myriophyllum spicatum*
erect	*Elodea canadensis*
erect	*Ceratophyllum demersum*
erect	*Potamogeton richardsonii*
erect	*Heteranthera dubia*
bend-down and erect	*Potamogeton pectinatus*
rosette	*Alisma gramineus*
rosette	*Vallisneria americana*
bottom-dweller	*Nitella sp*

Field characterization of macrophytes

The influence of macrophytes on flow varies for each species, depending mainly on their total surface area. Therefore, a precise knowledge of their spatial distribution is a necessary condition for accurate flow simulation. The approach proposed herein for macrophytes mapping is an adaptation of one proposed in Fortin *et al* (1993) and Boudreau *et al.* (1994). On October 1995, the maximum growth phase for plants for that year, several echosounder profiles were obtained on transects (Figure 6) during a five day field campaign to provide the basic information for setting up the macrophytes map. Ponctual controls with a submersible camera were performed regularly during echosounder recording, especially at echofacies changes. Positioning was done by a DGPS providing a reliable spatial referencing output. Plant taxonomy has been verified by Norman Dignard (Min. Énergie et Ressources, Québec) on samples collected in the field.

An interpretation key of the specific response of echosoundings to particular patterns of plant association and growth forms was built by adapting Fortin's approach and it was calibrated with camera observations. Species composition, rough species proportion, plant height and relative evaluation of density were extracted from each assemblages. A total of eleven plants assemblages, when refined with relative proportions, density or plant height compose a grand total of 83 distinct echofacies. Some examples of echofacies and corresponding interpretation are depicted in Figure 7.

Macrophytes mapping: an "educated" interpolation task

We know that the most important factor for the occurrence of aquatic plant is light. Light intensity is function of turbidity and depth. Close to a sampled area, as turbidity remains constant, one can argue that light intensity is only a function of depth. Similar logics can be applied to other abiotic variables. Therefore, the use of a precise bathymetric map was essential, in order to allow for a precise interpolation between transect. We used a bathymetric map with 1 m depth isocontour intervals at a scale of 1:15 000. This map was drawn directly with a Delauney triangulation procedure from an extensive data base (see below Numerical Field Model or NFM).

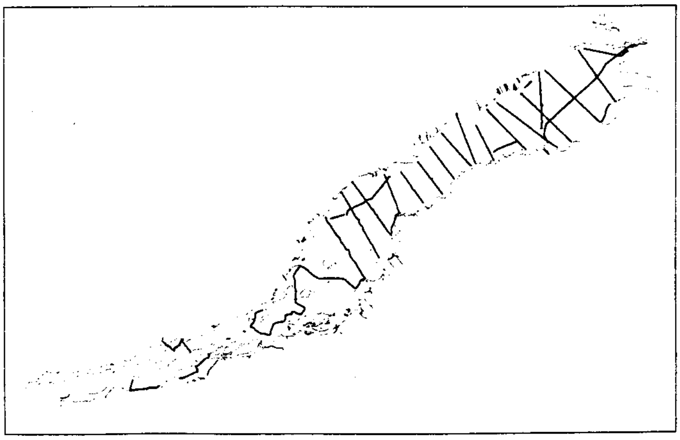


Figure 6 : Echosounding transects on Lake Saint-François (August 1994).

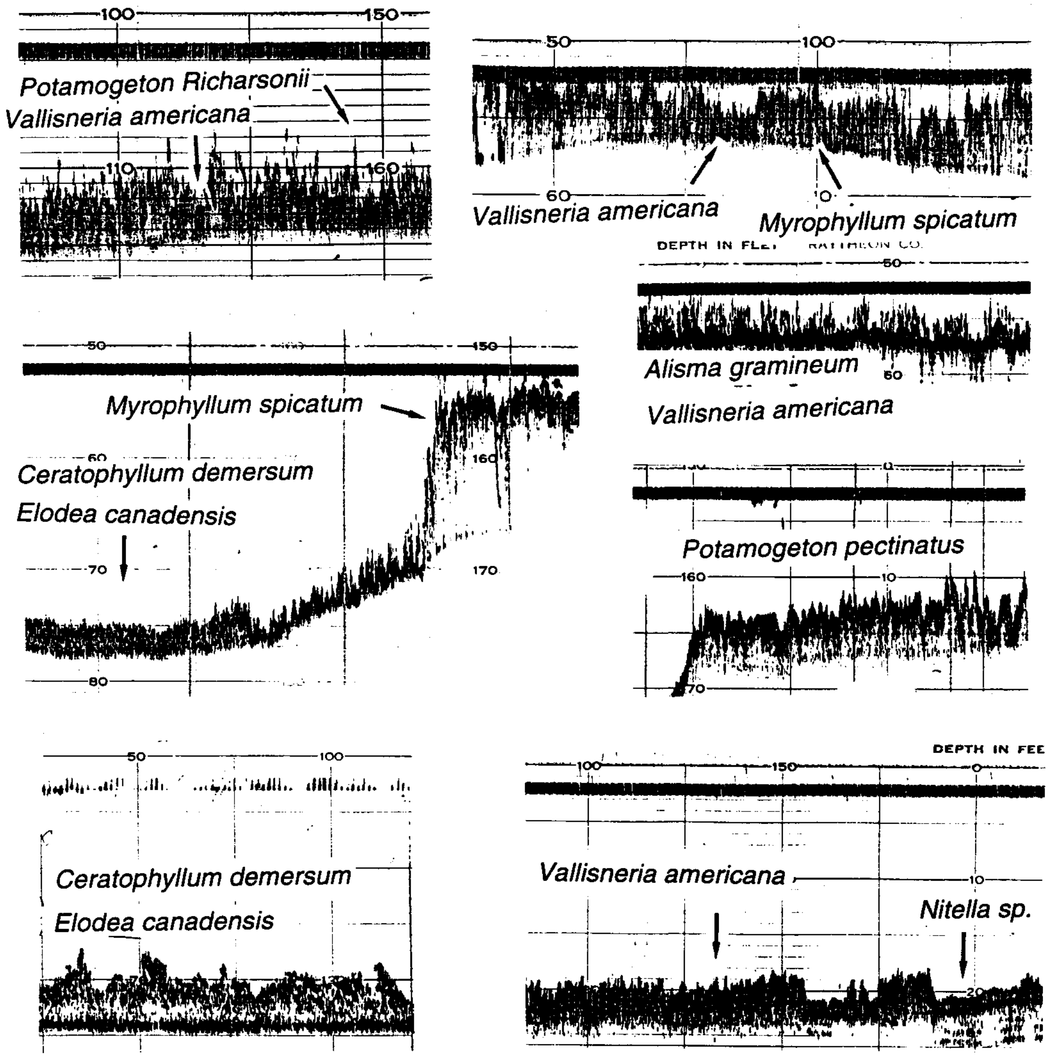


Figure 7 : Typical echofacies

Between the transects, plant distribution was estimated by using an interpolation procedure accounting for the visual interpretation of relevant abiotic variables available in the NFM data base. Rules governing the habitat selection by plants according to their growth form were also exploited. Provenance of these rules came first from the literature. But, it was mostly associated with a quick "learning curve" that took place while analyzing echofacies - vs - abiotic variable. A relatively small amount of the total echofacies data set was necessary to calibrate this conceptual interpretation and interpolation key. Depth, riverbed slopes, rough estimation of flow field and shelter effects, substrate composition, wind fetch and a rough estimation of nutrient distribution which participate to the application of the key are components of the NFM.

Just to give a brief idea on spatial distribution of plants, assemblages with *Vallisneria americana* as dominant species are found in areas with waves as the dominant abiotic factor, *Myriophyllum spicatum* assemblages are mainly related to nutrient rich substrates, *Potamogeton pectinatus* is found in areas dominated by currents, and *Nitella sp.* is present where other plants have difficulties to grow, mainly in deeper parts.

MODELING PLANT INFLUENCE ON HYDRODYNAMICS

Two-dimensional (2-D) hydrodynamic model

Briefly, as stated in Boudreau *et al.* (1996), the hydrodynamic model provides dynamic drying-wetting capabilities, an essential feature to deal with natural watercourse processes where the lateral flow boundaries move continuously with flow discharge or tides (Leclerc *et al,* 1990a, b). As with every 2-D horizontal model, velocities are vertically integrated (mean value). The model uses a "non-conservative" mathematical formulation (velocities and water levels as primitive state variables) based on St-Venant, or shallow water equations. It also uses a finite element (often called TIN for Triangular Irregular Network) spatial and numerical discretization scheme which is reputed to provide the best interpolation method to reproduce adequately the field data within the Numerical Field Model. The 2-D model in use in this research covers the entire range of forces and accelerations which occur in the natural flow equilibrium, e.g., local and convective acceleration, gravity (surface slope), riverbed, plant and ice resistance, wind stresses, turbulent shear stresses and compression, and Coriolis force.

Data representation: choosing a digital support

Explanatory variables can be provided through direct field sampling, and/or by predictive hydrodynamic models. For modeling purposes, all data sets must share a topologically homogeneous support (e.g., the same grid) to facilitate the computation tasks involving more than only one set. The Finite Element interpolation Method (FEM) proposed herein is comparable to generic Geographical Information Systems (GIS). In the case of field data, representation of information on a common digital support implies an interpolation procedure. The result constitutes the Numerical Field Model (NFM) obtained from the transfer of measured variable to the common data support.

Coupling hydrodynamic model with plant resistance

Resistance to flow by plants was the object of several studies (Haslam 1978; Manz and Westhoff 1988; Marshall and Westlake 1990, among others). Logarithmic velocity profile which usually characterize turbulent shear layer in open flow must be applied carefully in plant resistance dominated flows. Within the hydrodynamic model (Boudreau *et al* 1994), resistance of riverbed, ice and plants are being represented by a Manning's type formula.

(1) $$\tau_i = \frac{\rho g n^2 |V| u_i}{H^{1/3}}$$

Where τ_i represent the i^{th} (according to x, y coordinates, $i = 1{,}2$) component of resistance, ρ water density, g gravity, n the Manning roughness coefficient, V the velocity module (vertically integrated), u_i the i^{th} component of the velocity vector, and H the total depth.

This formula, very classical, thus relates quadratically resistance to velocity and a roughness coefficient, which is similar to "drag" type formulas. Bottom and plant resistance cumulate their influences according to a vectorial behavior.

(2) $$\tau_x = \tau_{xm} + \tau_{xb} \text{ and } \tau_y = \tau_{ym} + \tau_{yb}$$

Where $\tau_b$and τ_m are respectively bottom and macrophyte resistance modules.

As for the n for substrate (n_b), one currently relates this parameter to local mean rugosity or grain size. Concerning the parametrization of the plants (Manning's n_m) in the formula, it is necessary to take into account the main characteristics of plants which contribute to the resistance. Species, seasonal growth phase, density and relative area coverage in plant associations were the variables considered by Boudreau *et al.* (1994). Because there is a directly proportional relation between the total surface area of a plant and its resistance to flow, there is also a direct relation between plant resistance to flow and its height. In the actual work, plant heights have been measured and it is used in the Manning's coefficient calculation.

(3) $$n^2_{m_i} = C_i(t)\, P_i(r)\, D_{tot}(x,y)\, (\tfrac{h_i}{h_{i\max}})\, n^2_{m_i \max} \qquad \text{with} \quad \begin{matrix} 0 \le P_i(r) \le 1 \\ 0 \le D_i(x,y) \le 1 \end{matrix}$$

Where n_i represents the i^{th} species contribution to resistance, $n_{m_i \max}$ the maximum value provided by the species in the highest density, for its maximum height observed and during the maximum growth season, $C_i(t)$ is a growth phase factor varying between 0 and 1, h_i the local plant height at the maximum growth phase, $h_{i\max}$ the maximum *maximorum* height observed for a species, $P_i(r)$ the ratio of a species within an assemblage, and $D_i(x,y)$ a density factor related to relative area coverage of the assemblage.

$D_i(x,y)$ allows to modulate the spatial influence of plants according to its physical distribution. $P_i(r)$ permits to take in account the percentage of the i^{th} species in the assemblage and finally, the ratio h/h_{max} allows a modulation of the friction coefficient with plant height. Combination of C_i, P_i and D_i=1,0 and a height h_i corresponding to maximum growth phase generates the maximum influence of the species considered.

Consistently with (1), (2) and (3), n_{mi} for several plants (Number=M) species within an association cumulates quadratically to express to total influence of the considered association n_m.

(4) $$n_m^2 = \sum_{i=1}^{M} n_{mi}^2$$

Numerical Field Model (NFM)

Bathymetry in Lake Saint-François was characterized by the Canadian Hydrographic Service for navigation purposes. 292 000 measurement points were available in a digitized "field sheet" form, thus providing very convenient and precise estimation of riverbed morphology, and of geomorphologic features. This information was mobilized by a Delaunay's triangulation procedure as representation and interpolation tool, and used for setting up the hydrodynamic FEM (mesh). The result of this procedure is the Numerical Field Model. Accessibility to interactive visualization tools is necessary to set a NFM, especially if it uses the Finite Element Method. Geographical Information System can also be used for similar analysis.

For Lake Saint-François, a FEM was set up with the proposed methodology; the resulting mesh includes 27 618 triangular T6 elements (six interpolation nodes providing a quadratic approximation of variables) and thus, 57 468 nodes. Such a mesh generates more than 130 000 unknown variables which are solved by the algebraic resolution of flow equations. Nevertheless, this amount of data is much less than the original bathymetric data set.

Parametrization of plant resistance

One needs a reliable map of plants including their growth form and density and an estimation of n_{iMAX} in order to implement the proposed roughness algebraic closure. Our approach consists in four main steps:

- First estimation of plant distribution using a transect-based echosounding field characterization of plants;
- Simultaneous measuring of flow velocities for calibration and validation of hydrodynamic model;
- Bio-interpretation of transects data with respect to macrophytes preferences - vs - abiotic factors, educated interpolation between transects in order to set a map of plant growth forms;
- Calibration of roughness coefficient by using flow distribution measurements;
- Validation of hydrodynamic model.

We will see later how the predicted velocity values, together with other abiotic field data sets obtained in the field will be mobilized as independent variables in order to model mathematically the macrophytes distribution.

Field current measurements for calibration and validation

For calibrating and validating plants roughness parameters, velocity measurements were collected along the transects by using a ADCP (Acoustical Doppler Current Profiler). 15 000 measurement points, each offering the vertical velocity profile every 25 cm, resulted from this effort. For providing spatially homogeneous velocity data with regard to hydrodynamic model, a vertical integration of these profiles was required. Again, the huge size of this data set required the development of the proper tools to manipulate readily the information. The graphical package that resulted is called DOPPVISI. Distribution of velocities can be visualized either in plan or in vertical sections.

Dealing with roughness

Classification of the flow domain with respect to plant distribution and roughness allows the closure of the hydrodynamic model on the resistance parametrization. Table 2 shows the value of Manning's coefficient for each species and growth form at their maximum density, and for their maximum height. The values of Manning's coefficients used herein are modified from Boudreau *et al* (1994). They were used to represent resistance of plants in Lake Saint-Pierre, another fluvial lake of the St. Lawrence River system. This range of value was calibrated and validated on this lake by using a data set of 150 single velocities measurement points. Validation on Lake Saint-François will be much more precise taking into account the huge amount of data provided by the ADCP. As this activity is still in progress, the final results will be part of future communications. Figure 8 shows the distributed value of the plants roughness coefficient according to our parametrization scheme.

Table 2: Classification of plants roughness coefficients scheme used in Lake Saint-François.

Species (growth form)	Manning's coefficient (n_{max})	Maximum height (m)
EMERGING	0,200	surface
Myriophyllum spicatum (canopy)	0,200	surface (4,5)
Myriophyllum spicatum (erect)	0,150	3,0
Ceratophyllum demersum	0,100	1,5
Elodea canadensis	0,080	1,0
Heteranthera dubia	0,750	1,7
Potamogeton richardsonii	0,070	1,6
Vallisneria americana	0,060	1,2
Potamogeton pectinatus	0,030	0,8
Nitella sp.	0,020	0,5
Alisma gramineus	0,020	0,4

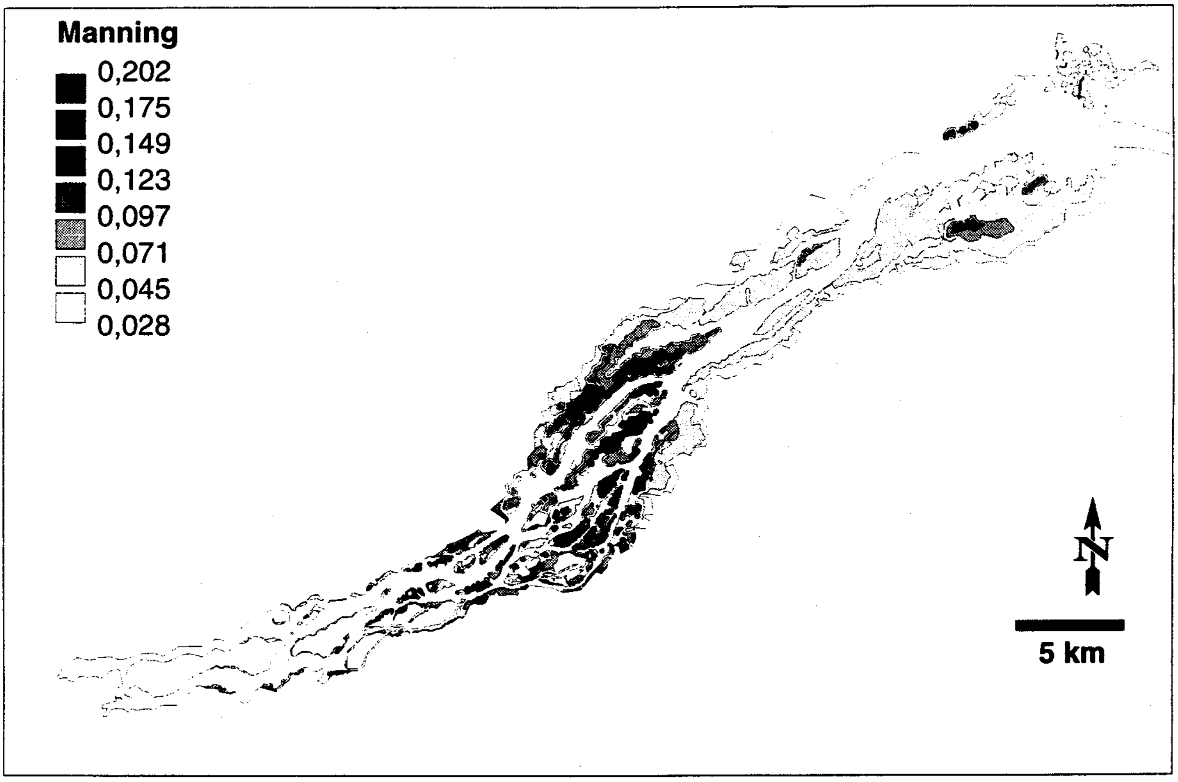


Figure 8 : Map of plants resistance as indicated by the global roughness coefficient : n_m

RESULTS

In order to demonstrate the contrasting flow patterns of the lake in presence of vegetation - vs - plant-free flowing conditions, e.g., late fall or early spring periods, Figures 9 and 10 depict velocity results of two hydrodynamic model simulations obtained with similar conditions, except for plant presence, in the model. Flow discharge through the model was imposed the average value of 7500 m^3/s. In these figures, only a part of the flow field located within the middle part of the lake is shown. This detailled view shows very distinctly local differences between the two simulations.

The maximum values of simulated velocities are within the plant-free deepest zones of the lake in the summer. Also noticeable are the numerous abrupt direction changes in currents. The structuring effect of plants on flow is dramatically demonstrated by these simulations. Already well-structured by a contrasting bathymetry, current patterns in Lake Saint-François are amplified or inhibited by the distribution of plants. Similar observations were made in Lake Saint-Pierre by Boudreau *et al* (1994), both in the field and in simulations.

DISCUSSION

Numerical modeling of complex flow fields like Lake Saint-François is a difficult task which involves a series of disciplinary knowledge form very distinct domains. Hydraulics, botany, hydrography, computer engineering, field work and numerical methods are the principal source of knowledge and know-how involved in such a project. The present work, as previously stated, has not yet completed its field validation phase for hydrodynamics. Proofed-results will be published in the final version of this article in a refereed journal.

The next challenging step will be to undertake the macrophytes microhabitat modeling for predicting plant distribution and biomass without using echosounder data. Rules derived by previous researchers and expertise obtained from echosounding interpretation will be exploited to produce a predictive mathematical model of plant distribution. Basic abiotic data will be provided through the NFM database. Validation of plant distribution model will be performed with some part of the actual data set and with new field characterization.

CONCLUSION

This work is a first step towards a predictive model of aquatic macrophytes habitat. Biomass, growth forms and possibly plant species appear predictable by this mean. Macrophytes of Lake Saint-François have been mapped using echosounder transects in association with a submersible video camera. Plants distribution between transects were interpreted using a basic habitat model in the form of an interpretation key using distributed abiotic variables as provided by a Numerical Field Model. We have extracted, from echosounding data, plant species composition, proportion, density and height. These heuristics were used in the estimation of the Manning's roughness coefficient.

Hydrodynamic simulations performed with friction coefficient for plants show that macrophytes tend to concentrate the flow in the main channels. In the future, we believe that plant microhabitat simulation will allow a rapid description of plant distribution and biomass, and consequently, improve the precision and the rapidity of field modeling.

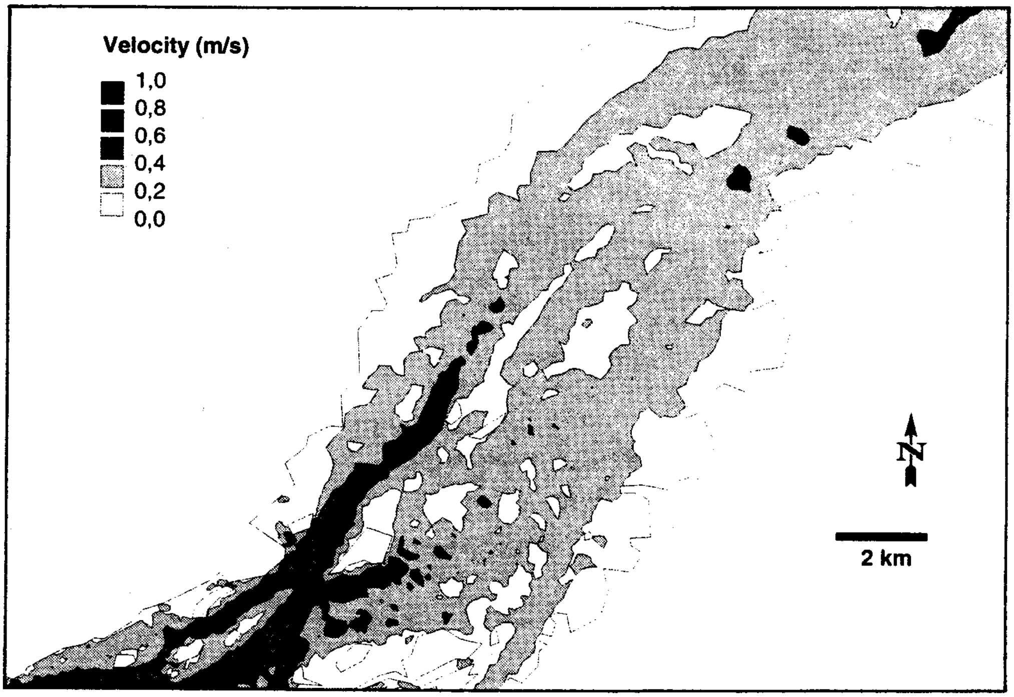


Figure 9: Simulation of flow velocity with spring-fall model. Notice: field-validation in progress

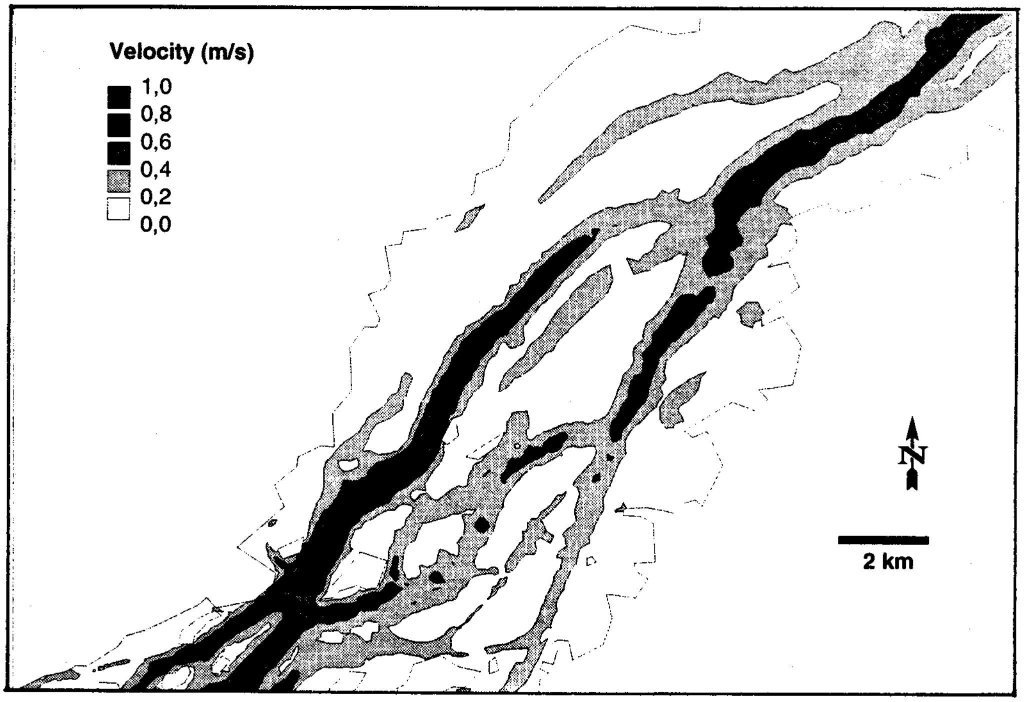


Figure 10: Simulation of flow velocity with summer model. Notice: field-validation in progress

ACKNOWLEDGMENTS

We thank Environment Canada (Jean-François Cantin, Paul-Emile Bergeron and Jacques Laroche) for field support, IREE (Philippe Crabbé), CNRC - Three Councils Ecoresearch Program for undertaking and funding the St. Lawrence Ecosystem Recovery project. Acknowledgement to Dr Richard Carignan (U of Montréal) for his judicious advices on research protocols, and to Norman Dignard (Min. Energie et Ressources, Québec) for his support in plant identification. This research has also benefited from the financial support of NSERC (Post-graduate scholarship to J.M. and operating grant to M.L.) and Fonds F.C.A.R. (scholarship to J.M).

REFERENCES

Allan, R.J., (1986). The limnological units of the Lower Great Lakes St.Lawrence corridor and their role in the source and aquatic fate of toxic contaminants. *Water Pollution Research* Journal *of Canada.* **21**: 168-186.

Anderson, R. M. & Kalff, J., (1986b). Regulation of submerged aquatic plant distribution in a uniform area of a weedbed. *Journal of Ecology.* **74**:953-961.

Barko, J. W. & Smart, R. M., (1980). Mobilization of sediment phosphorus by submersed freshwater macrophytes. *Freshwater Biol.* **10**: 229-238.

Boudreau, P., Leclerc, M. & Fortin, G., (1994). Modélisation hydrodynamiquue du lac Saint-Pierre, fleuve Saint-Laurent: l'influence de la végétation aquatique. *Revue canadienne de génie civil.* **21**: 471-489.

Boudreau, P., G. Bourgeois, M. Leclerc, A. Boudreault and L. Belzile (1996). Two-Dimensional Habitat Model validation based on spatial fish distribution : Application to juvenile atlantic salmon of Moisie river (Québec, Canada). The present Ecohydraulics 2000 Proceedings

Camfield, F. E., (1977). Wind-wave propagation over flooded, vegetated land. Tech. Paper No.77-12, Coast Engrg Res. Ctr., U.S. Army Engineer Waterways Experiment Station, Vicksburg, Miss.

Carignan, R. & Kalff, J. (1980). Phosphorus sources for aquatic weeds: water or sediments? *Science*, **207**: 987-989.

Chambers, P. A. & Kalff, J., (1985). Depth distribution and biomass of submerged aquatic macrophyte communities in relation to Secchi depth. *Can. J. Fish. Aquat. Sci.*, **42**: 701-709.

Chambers, P. A., (1987). Nearshore occurrence of submersed aquatic macrophytes in relation to wave action. *Can. J. Fish. Aquat. Sci.*, **44**: 1666-1669.

Chambers, P. A., & Prepas, E. E., (1990). Competition and coexistence in submerged aquatic plant communities: the effects of species interactions versus abiotic factors. *Freswater Biol.*, **23**: 541-550.

Chambers, P. A., & Prepas, E. E., (1994). Nutrient dynamics in riverbeds: the impact of sewage effluent and aquatic macrophytes. *Wat. Res.*, **28**:453-464.

Chambers, P. A., Prepas, E. E., Hamilton, H. R. & Bothwell, M. L., (1991). Current velocity and its effect on the growth of aquatic macrophytes in slow-flowing rivers. *Ecol. Appl.*, **1**:249-257.

Dale, H. M., (1986). Temperature and light: the determining factors in maximum depth distribution of aquatic macrophytes in Ontario, Canada. *Hydrobiologia*, **133**: 73-77.

Duarte, C. M. & Kalff, J., (1986). Littoral slope as predictor of the maximum biomass of submerged macrophyte communities. *Limnol. Oceanogr.*, **31**: 1072-1080.

Duarte, C. M. & Kalff, J., (1988). Influence of lake morphometry on the response of submerged macrophytes to sediment fertilization. *Can. J. Fish. Aquat. Sci.*, **45**: 216-221.

Fortin, G. R., Saint-Cyr, L., & Leclerc, M., (1993). Distribution of submersed macrophytes by echosounder tracings in Lake Saint-Pierre, Québec. *Journal of Aquatic Plant Management*, **31**: 232-240.

Hakanson, L., (1977). The influence of wind fetch, and water depth on the distribution of sediments in Lake Vanern, Sweden. *Canadian Journal of Earth Sciences*, **14**: 397-412.

Keddy, P. A. (1982). Quantifying within lake gradients of wave energy: interrelationships of wave energy, substrate particle size and shoreline plants in Axe lake, Ontario. *Aquatic Botany*, **14**: 41-58.

Kobayashi, N., Raichle, W. & Asano, T., (1993). Wave attenuation by vegetation. *J. of Waterway, Port, Coast. and Oc. Engrg., ASCE*, **109**: 115-117.

Leclerc, M., Bellemare, J.F., & Trusssard, S., (1990a). Simulation hydrodynamique de l'estuaire supérieur du fleuve Saint-Laurent (Canada) avec un modèle aux éléments finis couvrant-découvrant. *Revue canadienne de Génie Civil*, **17**: 739-751.

Leclerc, M., Dumas, G., Bellemare, J.F., & Dhatt, G., (1990b). A finite element model of estuarian and river flows with moving boundaries. *Advances in Water ressources*, **4**: 158-168.

Lorrain, S. Jarry, V. & Guertin, K., (1993). Répartition spatiale et évoluton temporelle des biphényles polychlorés et du mercure dans les sédiments du lac Saint-François; 1979-1989. Environnement Canada, Centre Saint-Laurent. 64 p.

Madsen, J. D. & Adams, M. S., (1989). The distribution of submerged aquatic macrophyte biomass in a eutrophic stream, Badfish Creek: the effect of environment. *Hydrobiologia*, **171**: 111-119.

Manz, D.H. & Westhoff, D.R (1988). Numerical analysis of the effects of aquatic weeds on the performance of irrigation conveyance systems. *Revue canadienne de génie civil.* **15**: 1- 13.

Marshall and Westlake (1990). Water velocities around water plants in chalk stream. *Folia Geobot. & Phytotax.* 25: 279-289.

Morin, J. (1995). Représentation de la distribution des macrophytes du lac Saint-François dans le cadre de la modélisation de l'hydrodynamique et de la propagation des contaminants. Ph.D. Thesis Proposal. 30 pp + 1 bibliographic appendice.

Morin J., Leclerc, M. & Boudreau, P. (1994). Lac Saint-François: les bases de la modélisation hydrodynamique. INRS-Eau, rapport de recherche No R-412, 66 p.

Owen and Wile (1975), Causes, consequenses and control of excessive aquatic plant growths in Lake St-Francis. Ontario Ministry of the environment, Southeastern region, Kingston.

Petticrew E. L. & Kalff, J., (1991). Prediction of surficial sediment composition in the littoral zones of lakes. *Limnol. Oceanogr.*, **36**: 384-392.

Petticrew E. L. & Kalff, J., (1992). Water flow and clay retention in submerged macrophyte beds. *Can. J. Fish. Aquat. Sci.*, **49**: 2483-2489.

Spence, D. H.N., (1982). The zonation of plants in freshwater lakes. *Adv. in Ecol. Res.*, **12**: 37-125.

Wetzel, R. G., (1975). Limnology. Saunders, Philadelphia. 743 pp.

Wilson, S. D. & Keddy, P.A., (1985). Plant zonation on a shoreline gradient: physiological response curves of component species. *Journal of Ecology*. **73**:851-860.

THE APPLICATION OF HABITAT MAPPING TO IDENTIFY REPRESENTATIVE PHABSIM SITES ON THE RIVER TAVY, DEVON, UK

Ian Maddock
Department of Geography
Worcester College of Higher Education
Henwick Grove
Worcester, WR2 6AJ
England
i.maddock@worc.ac.uk

Dave Bird
National Rivers Authority
South-West Region
Manley House
Kestrel Way
Exeter, EX2 7LQ
England

ABSTRACT

Many habitat modelling and instream flow studies are undertaken at the reach level (over a few hundred metres), yet for management purposes, these results must be extrapolated to longer sectors of river (over a few kilometres). In order to do this with any degree of certainty, a knowledge of the habitat types present and their extent throughout the river is essential. This paper describes the application of a methodology for defining representative reaches and transects for use with subsequent PHABSIM analysis on the River Tavy, Devon. The habitat mapping procedure was undertaken to identify the habitat types that must be sampled as part of the PHABSIM fieldwork and enable subsequent extrapolation of these results. Three sectors of river were identified based on existing hydrological information. Six typical mesohabitat types' present in the river were proposed (*i.e.* cascade, chute, riffle, shallow glide, deep glide and deep slack). Each sector was then habitat mapped to record the location and extent of each type. Transects were positioned where a change in mesohabitat type was evident or at 100m downstream from the last transect, whichever was the closer. Instream measurements consisted of channel and water width, maximum depth, maximum velocity and identification of the substrate sizes present. This resulted in the identification, measurement and mapping of 179 transects along 9km of river. These data were analysed to confirm the relevance of the habitat typology and assess the maximum velocity and depth characteristics of each type. The data also allowed the identification of the proportion of the river length and surface area occupied by each of the six mesohabitat types. Mesohabitat composition of the PHABSIM reaches currently being used to evaluate the impacts of flow regulation on habitat availability was based on these results. Reach specific results can then be extrapolated along the relevant sectors to ensure management is applied in the appropriate places.

KEY-WORDS: Habitat Mapping / IFIM / PHABSIM / Mesohabitat / Sector Scale / Reach Scale / River Tavy

INTRODUCTION

With a growing demand for water resources, an array of techniques have been developed to assess the impact of flow variations on instream habitat availability. These methods range from (1) simple 'desktop' methods, such as the Montana method (Tennant, 1976) that rely on an historical flow record, (2) 'mid-range' methods that utilise a small degree of site specific information (*e.g.*, Wetted Perimeter Method; Nelson, 1980) to (3) the more complex 'biological response' or 'incremental' methods (*e.g.*, Instream Flow Incremental Methodology (IFIM); Bovee, 1982) that require detailed fieldwork across several transects under different flows, computer-based hydraulic simulation and habitat simulation. Most of these techniques were originally developed in the USA, but with the declining flows, increasing demand for water and the recent drought (Marsh *et al.*, 1994), these techniques are now being utilised on an increasing scale in the UK (Petts and Maddock, 1994; Petts *et al.*, 1995). In particular, the IFIM has been tested to UK conditions (Johnson *et al.*, 1993) and is beginning to be used to determine water resource management decisions (Maddock and Petts, 1995).

The Importance of Scale in River Channel Assessment

In addition to each of these methods having different data requirements, many are applied over alternative spatial scales. It is widely accepted that the scale of approach to river investigations varies from the microhabitat level up to the catchment scale as shown in figure 1 (Frissell *et al.*, 1986; Petts, 1994). A river may be considered across a spectrum of scales which can be ordered into a hierarchy, each with a different degree of sensitivity and recovery time. Therefore, such a system not only defines the structural components of a river, but also recognises that the relative importance of factors controlling the short-term and long-term behaviour of streams changes with spatial scale.

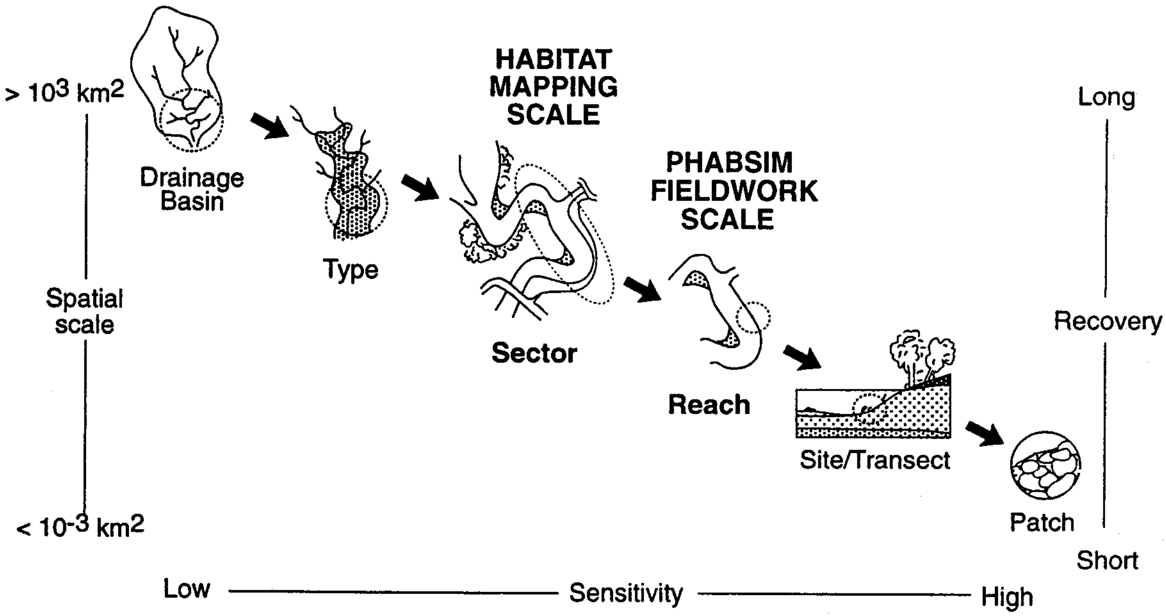


Figure 1: A functional classification of rivers based on scale (after Frissell *et al.*, 1986; and Petts, 1994)

Sectors and Reaches

As far as IFIM is concerned, two levels of assessment have been advocated that lie inbetween the extremes of scale (Bovee ,1982). Firstly the river may be divided into SECTORS (typically a few kms in length) which are defined by variables that instigate changes in hydrology and/or water quality, *e.g.*, at tributary confluences or the location of significant abstractions or augmentations. Within, each sector, water quality, sediment load, and hydrological regime are seen as largely invariant between sites. Secondly, each sector is divided into REACHES (typically a few hundred metres in length) on the basis of local variations in channel morphology or river-margin vegetation. Variations between individual reaches within a sector relates to local conditions (bank sediments, riparian vegetation *etc.*). On this basis, typical Physical Habitat Simulation Model fieldwork (PHABSIM - the computer based element of the IFIM) involves measurement at the REACH level, whereas management recommendations are more appropriate at the SECTOR level. Therefore extrapolation of results from the reach level to the sector level is necessary (Maddock *et al.*, 1995).

Habitat (or 'liveable space') can be defined as the local physical, chemical and biological features that provide an environment for the instream biota. It is affected by instream and surrounding topographical features, and is a major determinant of aquatic community potential. There is considerable evidence to suggest that both the quality and quantity of available habitat affect the structure and composition of resident biological communities (Hynes, 1968; Ward and Stanford, 1979; Meffe and Sheldon, 1988; Calow and Petts, 1994). Spatial scales in the context of habitat range from the 'macrohabitat' to the 'mesohabitat' and 'microhabitat' level (Bovee, 1982; Kershner and Snider, 1992).

Macrohabitat

At the broadest scale, the environment is relatively stable and biota are determined by the overall features of the region, its topography and altitude and its geomorphic/land-use pattern. This in turn produces a broad pattern of hydrology, temperature and chemistry. These features are called macrohabitat features; they determine the longitudinal distribution of various species (Bovee, 1982) and are akin to the drainage basin, type and sector scales discussed earlier. However, longitudinal zonation does not explain how stream reaches influence assemblages.

Mesohabitat

The distribution of habitat types within reaches has not received as much attention in the literature. Bisson *et al.* (1981) and Rosgen (1985) focused awareness on habitats as channel units by creating typologies based on hydrological and geomorphological features, followed by an examination of fish distributions among these habitats (Bisson *et al.*, 1988). At this scale of approach, habitat features of different river sections such as average flow velocity, morphological type (*e.g.*, riffle, pools) and bankside cover are deemed influential. Typical mesohabitats used in these studies include riffles, cascades, pools, glides and runs. The omission of this level of approach in many studies has been reiterated more recently by Kershner and Snider (1992). They stressed the need for instream flow studies to accurately characterise the mesohabitat characteristics of channel sectors based on objective, repeatable criteria.

Microhabitat

Habitat also includes the distribution of hydraulic and structural features comprising the actual living space of the organism at a much smaller scale called microhabitat (Bovee, 1982). For instance, the exact positions chosen by resident salmonids appears to be related to small scale physical characteristics, especially depth, velocity and substrate (Lewis, 1969; Shirvell and Dungey, 1983). Brown trout were found to prefer distinct velocities according to whether they were feeding or spawning and chose similar microhabitats regardless of the available habitat in different rivers (Shirvell and Dungey, 1983).

The Need for Habitat Mapping

It has long been recognised that to enable reach specific PHABSIM results to be applied and extrapolated to a longer sector of the river, it is necessary to undertake a 'habitat mapping' or 'reconnaissance level' survey at the sector level first (Bovee, 1982; Rabeni and Jacobson, 1993). However, many studies have neglected this initial sector level analysis and reaches have been selected on a professional judgement basis rather than on quantitative data. This in turn leads to difficulties in extrapolating reach results with any degree of certainty to larger stretches of river. Habitat mapping surveys incorporate a mixture of qualitative assessment and physical measurement to create a record of the form of a river based on field observation (Bisson *et al.*, 1981). They are designed to standardise the types of habitat observations that may be routinely made by geomorphologists or stream ecologists, but not usually recorded in such a disciplined way. The need, succinctly described by Thorne and Easton (1994), relies on information for:

1) the provision for scientific and repeatable observation and interpretation of channel morphology and instream habitat,

2) the supply of useful information for developing schemes to rehabilitate and restore geomorphic features in engineered streams in addition to highlighting those that need to be conserved in natural systems. This information can then be used to aid the classification and definition of mesohabitat types.

Habitat mapping involves walking the appropriate sector of river, identifying mesohabitats, noting their location and extent, and measuring their physical attributes, *e.g.*, water width, depth, velocity *etc.* Deciding on a classification of mesohabitat types should be completed before commencing the fieldwork. This can either be based on professional judgement; taken from the various sources available in the literature (*e.g.*, Bisson *et al.*, 1981; Rosgen, 1985; Modde *et al.*, 1991) or a mixture of both. Refinement of the typology may also be necessary after completion of the fieldwork in order to incorporate river specific circumstances. In other words, some types may be merged to simplify the typology if the original method was more complex than necessary (*e.g.*, combining intermediate glides with deep glides to give a single deep glide category). Others may have to be sub-divided to account for variability within the stream not recognised by the original types (*e.g.*, subdividing pools into plunge pools, lateral pools and scour pools). These decisions must take account of the tradeoff between creating a typology that has too many mesohabitats and is cumbersome to apply, and one that has too few and may omit an important category of mesohabitats. Re-classification can be achieved via professional judgement or more objective approaches (Jowett, 1993).

This information can then be used to identify the longitudinal distribution and total length/area and proportional length/area of each mesohabitat within the study sector. Subsequent analysis enables the mesohabitat composition of the PHABSIM reaches to be defined using the same criteria as the habitat mapping approach. This in turn facilitates a comparison between the types present within the PHABSIM reach and those evident in the sector as a whole so that the representative nature of the PHABSIM reaches can be critically assessed (Johnson and Elliott, 1993).

Habitat mapping methods place the emphasis on using minimal time and effort to cover long stretches of river and therefore must forego some of the level of detail and accuracy that is associated with more quantitative techniques applied at the reach level. Nevertheless, by including an element of physical measurement they provide a greater level of precision than subjective intuitive assessment and allow field analysis to be undertaken at the sector rather than reach scale.

The following section describes the application of habitat mapping on the River Tavy in order to:-

- characterise the mesohabitats present within discrete sectors,
- identify the longitudinal distribution and proportions of these mesohabitats within each sector, and,
- identify the appropriate mesohabitat composition of the PHABSIM reaches within each sector

HABITAT MAPPING ON THE RIVER TAVY

Catchment Characteristics

The River Tavy is an upland stream situated in south-west Devon, U.K. The headwaters rise in the Dartmoor National Park on the western flank of the Dartmoor Granite plateau and flow in a southerly direction to Plymouth where the river drains into the English Channel. Flows are monitored at Ludbrook gauging station where the catchment area upstream is 197.3km^2 and average daily flows are 6.04 m^{3}s^{-1} (figure 2). Land-use is dominated by moorland, rough grazing and low grade agriculture.

Definition of Sector Boundaries

River flows are regulated by a series of abstractions and augmentations. Water is abstracted at Tavy Cleave to Wheal Jewell Reservoir; and at Hill Bridge to Bennets Reservoir. Both reservoirs feed Mary Tavy Hydroelectric Power Station which discharges into Cholwell Brook, a tributary of the River Tavy. Water is also abstracted from the river near Tavistock where it supplies Morwellham Hydroelectric Power Station and is discharged into the neighbouring River Tamar.

Sector boundaries were positioned at the location of significant abstractions, augmentations or tributary confluences. Therefore, water quality, sediment load and the hydrological regime are largely invariant within each sector. Three sectors were identified and are defined in table 1.

Table 1: Sector boundaries

Sector No.	Upstream Boundary	Downstream Boundary
Sector 1	Nat Tor (NGR:SX546822)	Hill Bridge (NGR:SX532804)
Sector 2	Confluence with Wapsworthy Brook (NGR:SX531803)	Confluence with Cholwell Brook (NGR:SX509784)
Sector 3	Tavistock town centre (NGR:SX482743)	Confluence with the River Lumburn (NGR:SX466717)

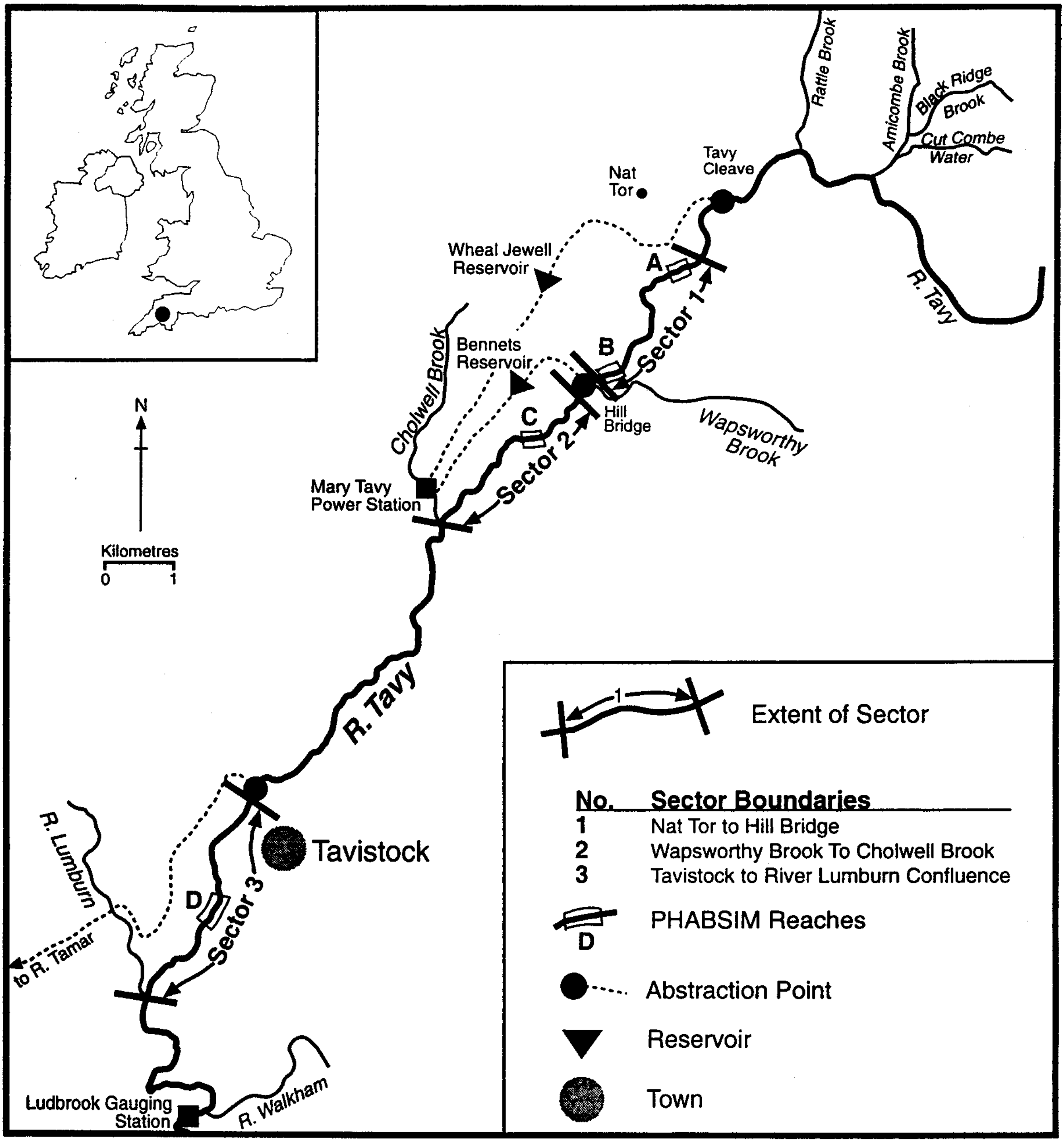


Figure 2: Location of sector boundaries and PHABSIM reaches within the River Tavy catchment

<u>Habitat Mapping Fieldwork</u>

Each sector was habitat mapped by a combination of visual assessment and physical measurement. Habitat mapping was undertaken during periods of stable 'medium' flow between 24th - 31st October 1995. Initial fieldwork involved identifying, measuring and mapping the location of six different mesohabitats present along the river (see table 2).

Table 2: Definition of mesohabitat types

Mesohabitat	Depth range (m)	Velocity range (m/s)	Description
CASCADE	highly variable	> 1	Steepest gradient. Step-like bed profile, with bedrock as substrate.
CHUTE	highly variable	> 0.45	Steep gradient with broken water surface and rapid flow velocities. Often bedrock or boulder substrate constraining wetted channel during low flow.
RIFFLE	≤ 0.5	> 0.8	Relatively steep water surface gradient, coarser bed material than local vicinity, some broken water. Usually of limited extent with deeper water evident both upstream and downstream.
SHALLOW GLIDE	≤ 0.5	< 1.3	Relatively smooth, low gradient water surface compared to riffle. Differentiated from deep glides by max. depth value. Visible flow clearly evident.
DEEP GLIDE	> 0.5	> 0.2 < 1.05	Relatively smooth, low gradient water surface differentiated from shallow glide by depth >0.5m.
DEEP SLACK	> 0.5	≤ 0.2	Smooth, low gradient water surface. Deep with little visible flow.

Typical/important mesohabitat types present were identified before fieldwork began. Habitat mapping is a technique that is designed to cover relatively long stretches of river in a short space of time. Inevitably there is a tradeoff when identifying the habitat types that will be mapped between the time taken to identify them and the level of precision. In other words, a system with too many types would make the mapping process too slow to cover long reaches, whereas too few types may fail to identify important habitats. The six types identified here were considered to provide a suitable balance between these two extremes.

The habitat mapping process was achieved in the field by starting at the upstream end and noting the first mesohabitat type and location. Instream measurements were then conducted to validate and clarify mesohabitat identification. The location of each transect was either at a position where a change in mesohabitat was evident or at 100m downstream from the last transect, whichever was the closer. For example, in sector 2, the upstream end was delimited and the transect defined as a shallow glide (as water depth was less than 0.5m and visible flow was clearly evident). The first change in mesohabitat type occurred at 28m downstream where visual assessment indicated depth had increased. Instream measurements confirmed this and the transect was defined as a deep glide. Measurements of channel width and water width were recorded to the nearest 10 cm and maximum depth across the transect was recorded to the nearest cm. Maximum velocity across the transect was identified to the nearest

0.01m/s using the E.F.S. Flowvane. This instrument was used to provide a rapid estimate of flow velocity and is designed to work within error bounds of ±5%. A subjective assessment was made of the substrate sizes present based on the percentage in each Wentworth (1922) size class. These measurements were then repeated at the next transect downstream and the distance inbetween each transect noted.

Final definition of transects into one of the six mesohabitat types was carried out by a review of the instream measurements and the subjective categorisation in the field. In certain circumstances the classification relies heavily on the visual assessment whereas for other categories the instream measurements prove more influential. For example, chutes and cascades were apparent from the bank and classification of these mesohabitats is primarily based on the visual assessment. Instream measurements in this case provide secondary/supportive evidence. Shallow glide transects may have similar depth and velocity characteristics to some chutes and riffles but the overall hydraulic conditions were clearly different and classification can be made based on visual assessment. Alternatively, the approximate boundary between 'shallow' and 'deep' in terms of glides is very subjective and more difficult to identify from the bank. The final boundary was arbitrarily defined at 0.5m and classification of these transects was achieved through the instream measurements for each. Therefore in this case, the classification relies more heavily on the physical measurements in addition to the visual assessment.

Habitat Mapping Results

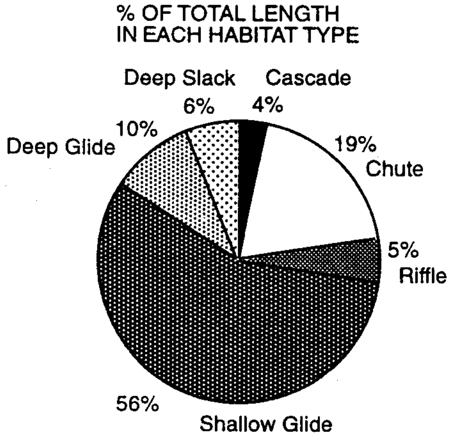
The entire survey involved the identification, measurement and mapping of 179 transects along a total of 9km of river. From these data, the total length and mean length of each mesohabitat type for each sector was calculated and is shown in table 3. However, these results are based on the length characteristics, and the PHABSIM output calculates usable habitat as a function of area. Therefore, to concur with this format, the habitat mapping length data were multiplied by the water width data to calculate the percentage of the total sector area in each mesohabitat type. Pie charts have been produced to illustrate the relative significance of each mesohabitat based on the length and area characteristics for each sector (figure 3). However, the area values could not be calculated for sector 1. Many cascades and chutes had no data for water width due to the inaccessible nature of the river at these transects and hence only length results are shown.

Shallow glides represent the most dominant mesohabitat type in all three sectors. In sector 1 chutes (19%) and deep glides (10%) are also prevalent. Deep slacks (6%), riffles (5%) and cascades (4%) are less frequent. Sector 2 is also dominated by shallow glide (31%) with deep glide (28%), chute (21%) and deep slack (18%) also significant. Riffles (20%) are the second most important type in sector 3, followed by chutes (17%) and deep glides (16%). Deep slack and cascade are present but rare (3% and 1% respectively). Based on this proportional data, the mesohabitat composition of the PHABSIM reaches were identified.

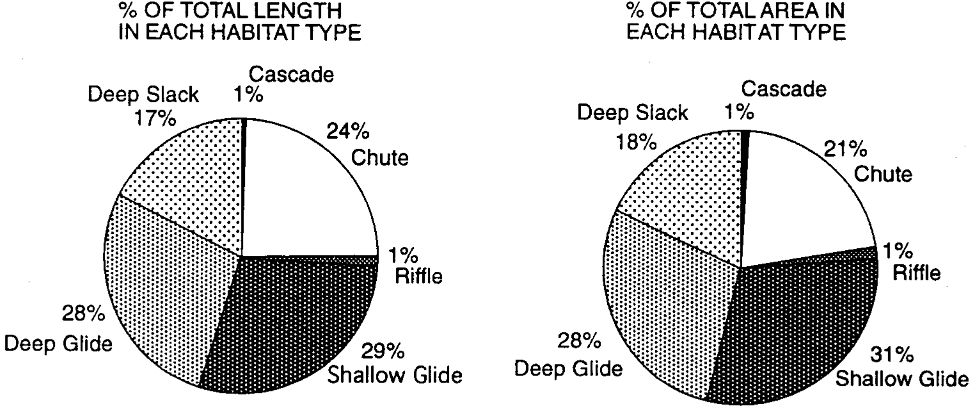
DEFINING THE MESOHABITAT COMPOSITION OF PHABSIM REACHES

Transects were selected to sample the mesohabitats present within each individual sector. The number of transects representing each mesohabitat type within the PHABSIM reaches were chosen to produce similar % weightings to those present within the sector as a whole. This study aimed to identify representative reaches. Mesohabitats represented by less than 5% of the total habitat length/area for that sector were considered to be 'rare' and hence are not represented by the PHABSIM transects in this case. It is recognised that these rare mesohabitat types may in

SECTOR 1



SECTOR 2



SECTOR 3

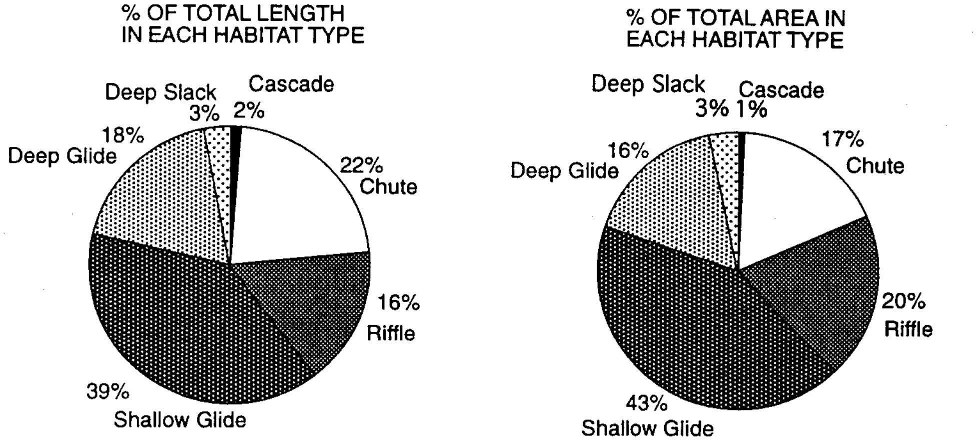


Figure 3: Proportions of mesohabitat types in each sector

fact be important for some species and a study wishing to identify 'critical' rather than 'representative ' reaches may also need to sample these types (*e.g.*, when wishing to protect the habitat of rare target species).

Table 3: Mesohabitat composition of sectors and number of transects used in each representative PHABSIM reach

Mesohabitat type	total length (m)	mean length (m)	total area (m^2)	mean area (m^2)	% of total in sector	no. of PHABSIM transects in reach	% weight in PHABSIM reach	Difference between sector & reach %
Sector 1								
cascade	88	29	-	-	4	-	-	-
chute	490	31	-	-	19	2	15	4
riffle	131	33	-	-	5	1	8	3
shallow glide	1440	53	-	-	56	7	54	2
deep glide	250	21	-	-	10	2	15	5
deep slack	154	39	-	-	6	1	8	2
TOTAL	2553	-	-	-	100	13	100	-
Sector 2								
cascade	20	20	400	400	1	-	-	-
chute	752	58	6637	511	21	2	17	4
riffle	41	41	447	447	1	-	-	-
shallow glide	894	75	9571	798	31	4	33	2
deep glide	861	62	8796	628	28	4	33	5
deep slack	535	54	5661	566	18	2	17	1
TOTAL	3103	-	31512	-	100	12	100	-
Sector 3								
cascade	53	26.5	431	216	1	-	-	-
chute	728	48.5	8238	549	17	2	18	1
riffle	551	50.1	9241	840	20	2	18	2
shallow glide	1301	59.1	20049	911	43	5	46	3
deep glide	616	61.6	7686	769	16	2	18	2
deep slack	96	48	1568	784	3	-	-	-
TOTAL	3345	-	47213	-	100	11	100	-

The mesohabitat composition of each sector is shown in table 3. For each sector, the proportion of the sector length/area is identified, followed by the number of transects used to represent each mesohabitat type within each PHABSIM reach. For example, a total of 13 PHABSIM transects were identified in sector 1 to sample the mesohabitats present. This number representing each type was chosen to provide a weighting that approximated to that in the sector as a whole. The selection also took into consideration the tradeoff between incorporating additional transects and the time required to collect the representative sample and was chosen to provide a workable balance between the two. The % weighting based on the PHABSIM transect is also shown followed by the difference in % weighting between the sector and reach scales of assessment. The % weighting based for each mesohabitat type in the PHABSIM reaches is no more than 5% different than that for the sector as a whole.

Different total numbers of PHABSIM transects were chosen for each reach depending on the complexity of the proportional weightings and the diversity of the mesohabitats present. For example, fewer transects were needed in sector 3 because the proportions of three of the four types present had similar weightings (chute, riffle and deep glide) and each one could be represented by two transects within the PHABSIM reach.

After PHABSIM data collection, calibration and simulation, transect-specific output (*e.g.*, discharge versus weighted usable area relationships, habitat-time series and habitat duration curves) will be combined for each reach to produce composite results. These results can then be applied from a management perspective at the sector scale with a knowledge that the PHABSIM output represent the mesohabitats present with the sector as a whole.

CONCLUSION

This review has attempted to reiterate the need for sector definition and habitat mapping to be completed before detailed reach-based habitat assessment work in order that results can be extrapolated with a greater degree of certainty. The method has been applied as the first stage of a PHABSIM study that is examining the impact of flow regulation on the River Tavy, UK. It is recognised that the identification of the mesohabitat types may be flow dependent, and work is ongoing to resurvey each sector under three flows that span the flow regime to address this question. The habitat mapping technique inevitably contains some subjectivity in order to identify the mesohabitat types, but instream measurements help to keep this to a minimum. It is an approach designed to cover relatively long stretches of river in a short time and so to fulfill this criteria, cannot involve highly detailed assessment and measurement. Further research will attempt to utlilise cluster analysis on the physical measurements (*e.g.*, depth and velocity) to further reduce the subjectivity used to classify each mesohabitat type. This may be particularly applicable when defining the depth boundary between 'shallow' and 'deep' glides and the velocity boundary between deep glide and deep slack. Nevertheless, the application of sector definition and habitat mapping as the first stage of assessment will enable PHABSIM reach results to be extrapolated along the relevant stretches of river to ensure that management recommendations are applied in the appropriate places.

REFERENCES

Bisson, P.A., J.L. Nielsen,, R.A. Palmason and L.E. Grove, (1981). A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. In, Armantrout, N.B. (ed) *Acquisition and Utilization of Aquatic Habitat Inventory Information.* American Fisheries Society. pp. 62-73.

Bisson, P.A., K. Sullivan and J.L. Nielsen (1988). Channel hydraulics, habitat use and body form of juvenile coho salmon, steelhead, and cutthroat trout in streams. *Transactions of the American Fisheries Society*, 117 pp. 262-273.

Bovee, K.D. (1982). A guide to stream habitat analysis using the Instream low Incremental Methodology. *US Fish and Wildlife Service Biological Services Program, Cooperative Instream Flow Service Group, Instream Flow Information Paper No. 12.* FWS/OBS-82-26. 248p.

Calow, P. and G.E. Petts (eds) (1994). *The Rivers Handbook vol.2.* Blackwell Scientific (Oxford).

Frissell, C.A., W.J. Liss, C.E. Warren and M.D. Hurley (1986). A hierarchical framework for stream classification: viewing streams in a watershed context. *Environmental Management.* 10 pp. 199-214.

Hynes, H.B.N. (1968). Further studies on the invertebrate fauna of a Welsh mountain stream. *Archives Hydrobiologia.* 65 pp. 360-379.

Johnson, I.W. and C.R.N. Elliott (1993). River Allen habitat mapping survey. *Report undertaken by the Institute of Hydrology for the National Rivers Authority Wessex Region.*

Johnson, I.W., C.R.N. Elliott and A. Gustard (1993). Ecologically acceptable flows. *Report undertaken by the Institute of Hydrology for the National Rivers Authority.*

Jowett, I. G. (1993). A method for objectively identifying pool, run, and riffle habitats from physical measurements. *New Zealand Journal of Marine and Freshwater Research.* 27 pp. 241-248.

Kershner, J.L. and W.M. Snider (1992). Importance of a habitat-level classification system to design instream flow studies. In, Boon, P.J., P. Calow and G.E. Petts (eds) *River Conservation and Management,* Wiley (Chichester) pp. 179-193.

Lewis, S.L. (1969). Physical factors influencing fish populations in pools of a trout stream. *Transactions of the American Fisheries Society.* 948 pp. 14-19.

Maddock, I.P. and G.E. Petts (1995). Instream flow requirements of the River Kennet. *Report Undertaken for the National Rivers Authority Thames Region.* Project Reference PAB/OSP/69.

Maddock, I.P, G.E. Petts and M. Bickerton (1995). River channel assessment - a method for defining channel sectors on the River Glen, Lincolnshire, UK. In, Petts, G.E. (ed) *Man's Influence on Freshwater Ecosystems and Water Use,* IAHS Publication No. 230, IAHS Press (Wallingford), pp. 219-226.

Marsh, T.J. R.A. Monkhouse, N.W. Arnell, M.L. Lees and Reynard, N.S. (1994). *The 1988-92 drought.* Institute of Hydrology. 79p.

Meffe, G.K. and A.L. Sheldon (1988). The influence of habitat structure on fish assemblage composition in southeastern blackwater streams. *American Midland Naturalist.* 120 pp. 225-240.

Modde, T., R.C. Ford and M.G. Parsons (1991). Use of a habitat-based stream classification system for categorizing trout biomass. *North American Journal of Fisheries Management.* 11 pp. 305-311.

Nelson, F.A. (1980). Evaluation of selected instream flow methods in Montana. *Proceedings of the Annual Conference of the Western Association of Fish and Wildlife Agencies.* pp. 412-432.

Petts, G.E. (1994). Rivers: dynamic components of catchment ecosystems. In, Calow, P. and G.E. Petts (eds) *The Rivers Handbook vol. 2.* Blackwell Scientific (Oxford) pp. 3-22.

Rabeni, C.F. and R.B. Jacobson (1993). The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Freshwater Biology.* 29 pp. 211-220.

Rosgen, D.L. (1985). A stream classification system. In, Johnson, R.R., C.D. Zeibell, D.R. Patton, P.F. Pfolliott and R.H. Hamre (eds) *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses.* United States Forest Service, General Technical Report M-120, Rocky Mountain Forest and Range Experimental Station, Fort Collins, Colorado.

Shirvell, C.S. and R.G. Dungey (1983). Microhabitats chosen by brown trout for feeding and spawning in rivers. *Transactions of the American Fisheries Society.* 112 pp. 355-367.

Tennant, D.L. (1976). Instream flow regimens for fish, wildlife, recreation and related environmental resources. *Fisheries.* 1 pp. 6-10.

Thorne, C.R. and K. Easton (1994). Geomorphological reconnaissance of the River Sence, Leicestershire for river restoration. *East Midland Geographer.* 17 pp. 40-50.

Ward, J.V. and J.A. Stanford (1979). *The Ecology of Regulated Streams.* Plenum Press (London) 348p.

Wentworth, C.K. (1922). A scale of grade and class terms for clastic sediments. *Journal of Geology.* 30 pp. 377-392.

The Habitat Modelling Framework
A tool for creating habitat analysis programs

Knut Alfredsen
Dept. of Hydraulic and Environmental Engineering.
Norwegian University of Science and Technology.
7034 Trondheim
Norway
Email: Knut.Alfredsen@ivb.unit.no

Ånund Killingtveit
Dept. of Hydraulic and Environmental Engineering.
Norwegian University of Science and Technology.
7034 Trondheim
Norway.
Aanund.Killingtveit@ivb.unit.no

ABSTRACT.

Habitat modelling is a still evolving field with development of new methods for analysis, characterisation of new parameters and addition of new species in the studies. This leads to frequent changes in the modelling tools, as new changes are implemented. In an traditional modelling environment frequent changes can be time consuming to implement, and they can also lead to maintenance problems in a model not designed for this type of incremental development. This paper describes a general object-oriented toolkit for development of habitat models for rivers based on a generalized grid structure. The toolkit has a strong focus on code reuse and extendibility, features which are important when we want tools for rapid application development and a high demand for future extensions. The paper looks at the background for the development and the elements in the framework and their collaboration in the total system structure. The use of the framework in model development is shown, with the implementation of the new version of the River System Simulator Habitat model used as an example.

KEY-WORDS: Object-orientation / Habitat modelling / Frameworks / Impact assessments / Program development / River System Simulator / Habitat /

1. INTRODUCTION

During the recent years habitat analysis has been done in several Norwegian rivers as a part of impact assessment studies. New research also provides new methods and new criteria's for the analysis. In the future the field of habitat analysis will broaden as more subjects and parameters are included in the study. For example the Norwegian habitat model in the River System Simulator (Killingtveit et.al. 1994) has undergone several changes over the last years as new methods and lotic parameters are characterised. The changes in the model structure can be an complex process, it is in most cases necessary to involve computer programmers to do this job. This is an unfortunate situation, since it slows down the development/testing process. It would be an advantage if the models were made to allow fast prototyping and development of new methods without deep knowledge of the underlying system structure. In a model that allows this form for "incremental development", hydraulic engineers and biologists can take more part in the development and testing than what is the case today.

To solve problems with continued changes and new developments there is a need for a tool that supports these requirements in a better way than the existing one. The requirements for the Habitat Modelling Framework development project is described below:

- Create a tool that facilitates changes in structure and methods easier than today. Basically the analysis and calculation method should be completely separate from the components describing the topology and structure of the river in question.
- Create an open system where scientists with little knowledge of the internal implementation can use the toolkit to test out new methods and teories.
- It must be possible for the user to define parameters used in the analysis without changing the storage structure and the parameter retrieval mechanism.
- Simplify maintenance and further development of the models.

To fulfil these requirements we decided to create a general toolkit for construction of habitat models, as part of a general frameworks for hydrological and hydraulic modelling under development at the Department of Hydraulic and Environmental Engineering, NTNU (Rinde, 1996 Alfredsen et.al., 1996). This tool was then used to implement the Habitat model with the existing set of analysis methods and a set of newly proposed analysis methods. An added benefit was that the 3D hydraulic model SSIIM (Olsen et.al., 1994) now was easily interfaced to the habitat analysis methods.

We used object oriented methods to construct the toolkit. This gives both technical advantages when creating a system that have a high demand for extendibility and reuse, and it gives the user of the system a natural way to create the system model since the components found in a habitat model fits well into an object model.

2. OBJECT-ORIENTATION AND O.O. FRAMEWORKS

The use of object-oriented analysis and design techniques is growing strongly among computer programmers around the world. The key concept of object orientation is to model the world with objects, each object represents a real world component. The component is defined by it's representation (state) and methods that responds to messages sent to the object or perform internal operations in the object. The representation is controlled by state variables, in most cases hidden from the user inside the object. The methods are a set of procedures and functions that the object understands. A user of the object will see only the message handling methods, thereby avoiding to have to deal with the difficulties of internal representation. This gives the programmer the opportunity to change the internal representation and implementation without having affecting the public interface against users of the object.

The object definition is found in a class. The class describes the representation and the methods of an object. In a program there can be several objects of the same class, everyone with an unique identifier. For example, if we model a river we will most likely divide it into several reaches represented as individual objects. Each river reach will be of the same class, but the objects used in the model will have a unique representation defined by it's internal state variables like cross sections, roughness coefficients and slope. Each river reach will have the same set of methods for calculation of hydraulic and habitat data, and they will pass message to each other when e.g. water are transported from one reach to the next.

In a model, we use collaborating objects to represent the real world system we want to simulate. These objects are grouped together in a structure where they communicate to each other through message passing. As long as we keep the message interface of the objects consistent, we can change objects with similar functionality without having to deal with the other objects in the structure. Another important aspect is the objects ability to inherit both representation and functions from other objects, a mechanism that can be used to specialise subtypes.

If we extend our river reach example from above, we can define lakes, reservoirs, channels and other objects found in a real world watershed and group them together in a structure. The objects will have methods to calculate internal states and to send water to the next module in the system. We will also use inheritance to specialise the different components, lake and reservoir objects will both inherit from a storage object that holds their common information.

A common tool in software development is libraries of functions that perform specified tasks. This concept is brought a step forward in object oriented methods where toolkits named frameworks are currently developed and used in several areas. A framework is a collection of objects that can be reused to create whole or part of applications. The main difference between a framework and a library is that in addition to being a library of components, the framework also defines co-operation among the objects. The user can extend the framework by inheritance and reimplementation of key functions (Booch, 1994). Frameworks are commonly used in programming of databases and graphical user interfaces.

3. DESIGN OF THE HABITAT MODELLING FRAMEWORK.

3.1 Analysis of the problem domain.

To be able to create a solid and usable framework, a thorough analysis of the problem domain is necessary. This analysis forms the base for the construction of the framework. The main result of the analysis is to identify the objects of the system, their roles in the model and the messages that they will need to respond to. To identify the system components, a traditional object identification approach is used. To describe the interaction between objects and the message flow, the Object Oriented Role Analysis and Modelling (OORAM) technique is used (Reenskaug, 1996). A role describes a specific task that an object performs in the system, and we will often see that a single object can take several different roles. If we look at our reservoir object from the example in chapter two, it will in one view of the system have the role as a data storage unit providing the user of the system with all necessary information about the reservoir. In another view of the system it will have the role of a water transport element, performing a routing operation.

Our problem domain is divided into three separate areas: system structure, analysis methods and simulation control. In addition objects concerned with datahandling and data structuring will be a part of the toolkit.

System structure

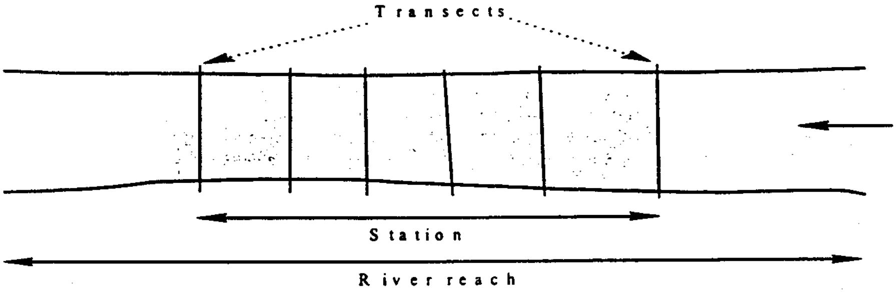


Figure 1: River reach

Figure 1 shows a typical river reach. A part of this reach is selected for habitat modelling. This **river reach** is further divided by **transects**, each transect has a set of **points** where data is collected. The stretch is called a **station** in the Habitat model. The station is divided into a **grid** of **cells** in which habitat analysis is done. Figure 2 shows examples of different grid structures. The boldfaced terms in the above sentences are potential objects describing the structure of the system. These six objects form the general structure part of the system.

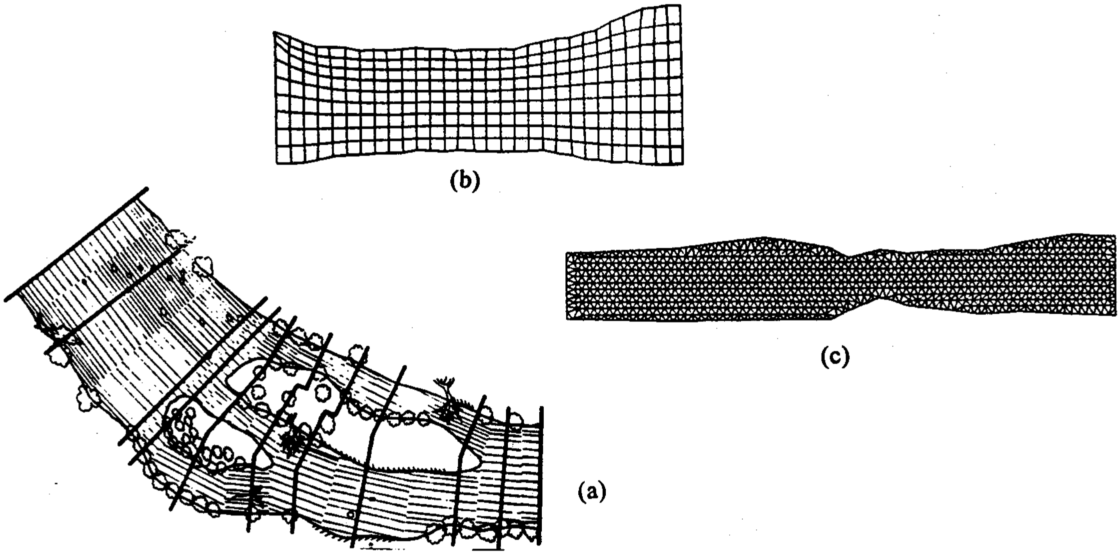


Figure 2: Possible grid configurations, a:Transect defined (After Bovee, 1982), b: SSIIM grid (After Olsen et.al. 1994), c: triangular grid (After Ghanem et.al. 1994)

Analysis methods

Each object can hold a set of methods for analysis. One of the requirements for the framework is that these methods should be separated from the structural information. We must thereby define these separately, and then associate them with the structural objects. We can for example have habitat analysis methods connected to the station object and volume calculation connected to cell objects.

Simulation control

The last part of the main system is the controller. The controller creates the system structure, assigns the methods and controls the simulation sequence.

System interaction

Analysis of the system interaction is done using OORAM (Reenskaug, 1996). We start by looking at the main system as a whole, then we break it down and analyse each of the system components. Figure 3 shows the collaboration of main roles in the system.

Each of the main roles will have one or more role-models on a lower level describing the different tasks that each component of the system must perform. Each object in the model can have different roles depending on the context it operates in. Definition of the different roles of an object is important also when we want to create a proper abstraction. If one object has too many roles we may consider to create subobjects by using inheritance. Figure 4 shows how the station object both has a role as a data storage unit and in another view of the model it calls the different analysis methods connected.

When all roles are defined the role-modelling concept goes on with work scenarios and message maps. These define the interaction between the objects and the necessary message handling that must be provided in each interface. As an example Figure 5 shows the transactions done when the simulation control object issues a simulate message. The roles found on figure 5 is a result of the combination (called synthesis in OORAM) of the two views shown in figure 4.

3.2 Framework Design

The habitat modelling concept is based on analysis of a defined river reach. The river reach class is defined in a general toolkit for hydraulic/hydrological modelling. It will therefore in the future be possible to connect the habitat model directly to other models that work in the scope of the general framework. To allow connection to other hydrological and hydraulic objects, the river reach class is derived from a general hydraulic class, which sole purpose is to work as a hook for connection of different objects.

The station is defined as a class related to river reach. This class must be overridden by the user of the framework, and a selection of functions must be reimplemented to create and operate the actual station.

The model works on a grid structure, where each cell is analysed, e.g. for weighted usable area. The grid may be defined in three dimensions. To store the cell information and to navigate the grid, a special datastructure, a cellcontainer is developed.

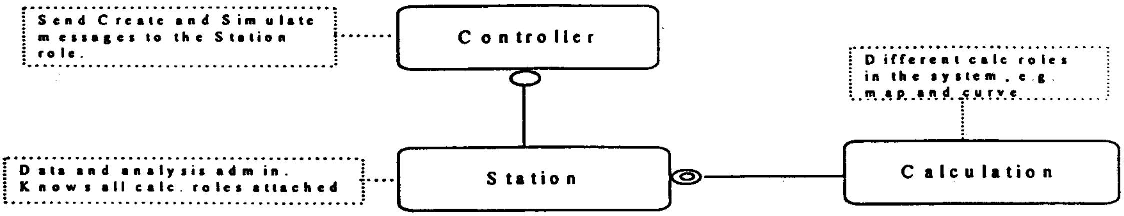


Figure 3: Main roles in framework

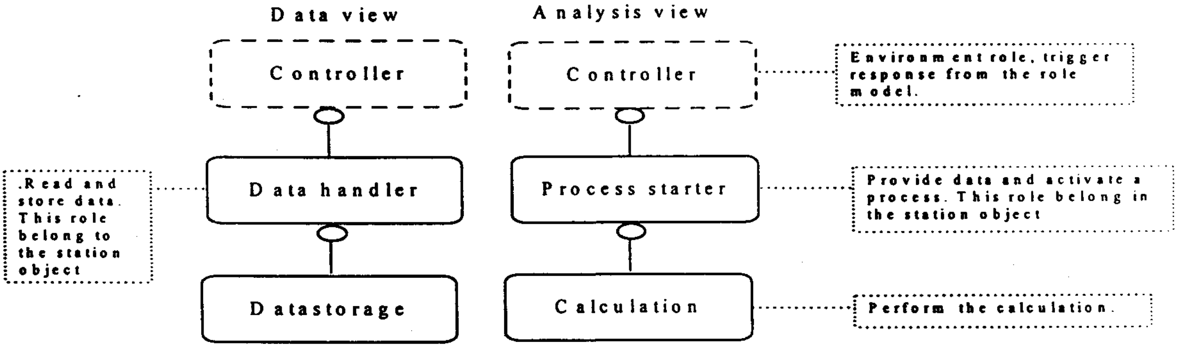


Figure 4: Station object roles

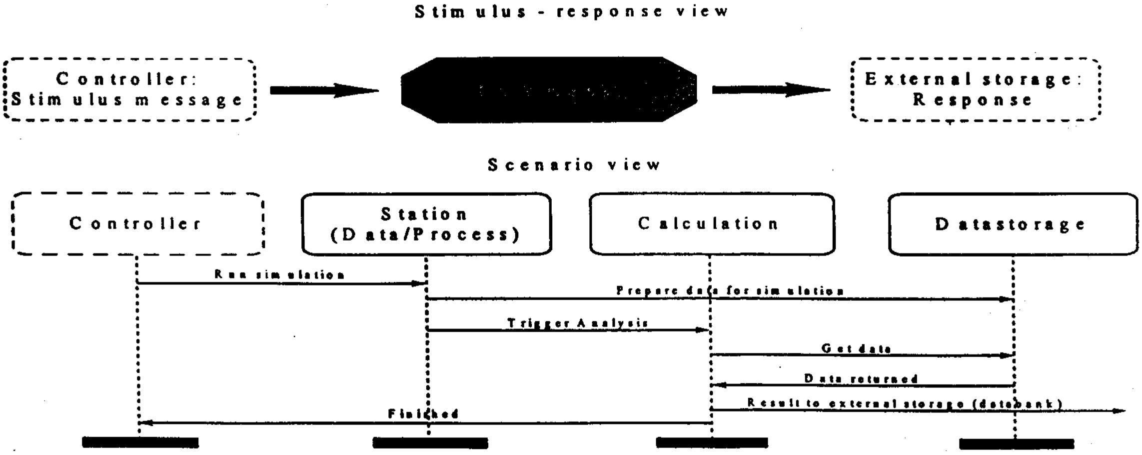


Figure 5: Scenario and stimulus response view of role models in figure 4. The views describe the detailed transactions of the combined role model.

The cell object is used to store references to data for the cell and to store the grid cells geometrical information. The cell only contains references to data, no actual datavalues are stored in the cell. This is important if we later want to use a different implementation of cell, or if we want to change the datastructure without changing the cell itself.

One of the key fetaures of the framework is to separate the structural information from the analysis methods. This makes it possible to change the analysis methods without having to change the structural objects. This is done by using a technique called the strategy design pattern, described by Gamma et al. (1995). This is a general method to separate processes from process users. Figure 6 shows an object diagram explaining how this technique is applied in the framework.

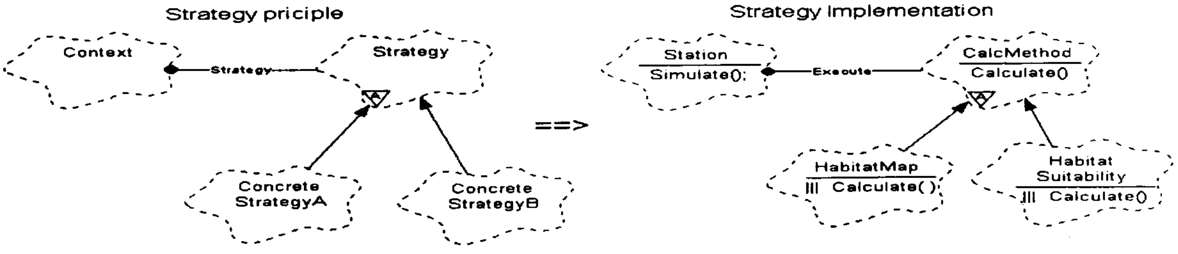


Figure 6: Definition of strategy design pattern, and it's use in the framework.

The framework provides the user with an interface to the controller. There are two main tasks in the controller, creation of the system structure and control of the calculation loop. Both these task must be implemented by the model developer.

A habitatmodel of this type normally generates a large amount of data from the run. To handle the flow, storage and retrieval of data a data handler is used. The data handler supports timeseries, two dimensional (x,y) data and three dimensional data. There is no code for data storage in the calculation methods. When a analysis method needs data, it sends a request to the data handler. The data handler currently supports SQL database and file storage.

Figure 7 shows the complete system structure of the framework.

5. EXAMPLE OF USING THE FRAMEWORK TO CREATE A HABITAT MODEL

The River System Simulator (Killingtveit, 1994) program system contains a habitat model which was based on the earlier RIMOS model (Vaskinn, 1985). In order to meet the needs of the scientists doing habitat analysis, a major upgrade of this model was necessary and has recently been implemented. To simplify this upgrade and to prepare the model for future extensions, the Habitat Modelling Framework (HMF) was used.

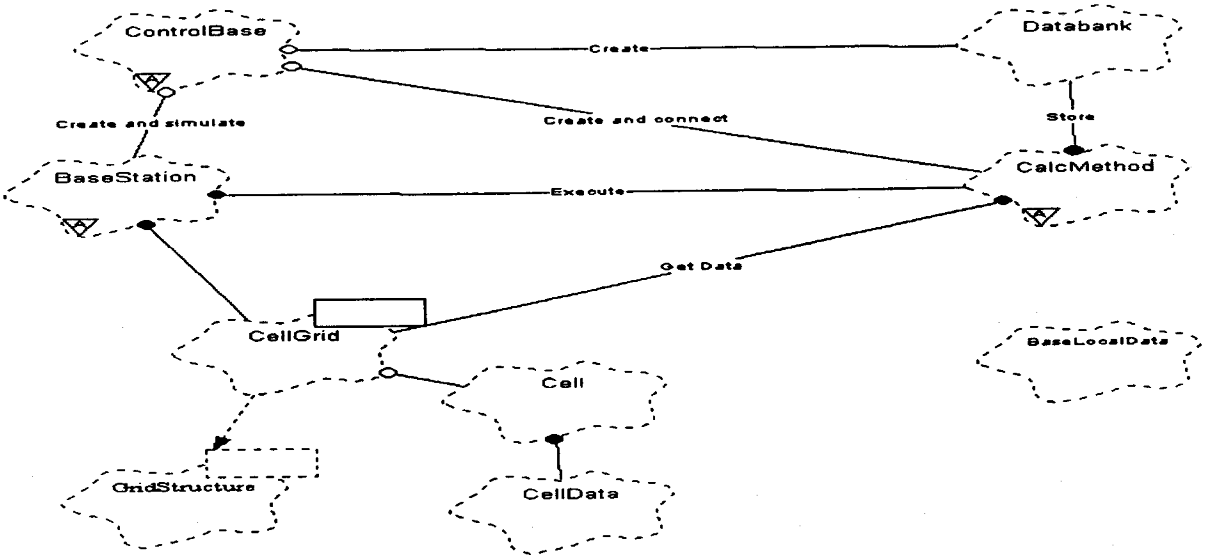


Figure 7: Class diagram for the habitat modelling framework.

The current Habitat model is based on calculations with the hydraulic model Hec-2 and some additional hydraulic modules. This version of Habitat is based on external use of those programs, but a future version will include all hydraulic modules as internal calculation methods. In this version the station object is generated using data from the external hydraulic models. This station object handles all grid initialisation and memory allocation.

All analysis methods are derived from the calculation method class. The base calculation method class contain the connections to the cell grid and the data handler. Table 1 list the analysis methods currently available in the Habitat model.

Table 1: Analysis methods defined in the Habitat model

Habitat Maps	Map of the river reach with colour codes describing the habitat values of the different cells.
Habitat Classification Curves	Habitat area as a function of discharge, classified as good, indifferent or unusable.
Habitat Timeseries	Timeseries of habitat based on discharge-habitatarea relations and discharge timeseries.
Weighted Usable Area	PHABSIM method (Bovee, 1982). This method computes a suitability index for each cell and multiplies it with the cell area. The total weighted usable area is then computed by adding all cell areas together.
Weighted habitat maps and classification curves.	Maps and classification curves where parameters are weighted together into one. HabitatValue = dw*f(d)+vw*f(v)+sw*f(s). dw,vw and sw are respectively weights for depth, velocity and substrate, the f() function gives the suitability index

To create a new habitat analysis or hydraulic method involves setting up some common parameters. A method generator program is created to do this for a standard method, so the procedure for generating a new method is as follows:

1. Use the method generator program to create the method shell. This shell contains the necessary links to the grid and the data handler. This function also generates the code that traverses the grid and retrieve data. The user should then implement the analysis part in the simulation method. Data storage is done by requesting a new data storage unit from the data handler, fill data into this and then return it to the data handler.
2. A function to create the method in the model is automatically created and stored in the method factory. The method factory is based on a technique for automatic object generation, defined by Gamma et.al. (1995)
3. Call the appropriate factory function in the controller to create the new method.

The main task of the controller is to create all parts of the model and to control the simulation loop. All creation is done by using the object factory (Gamma et.al. 1995) to create all objects, and then calling the initialisation function of each of them. The simulation loop control can be handled in several ways, it can be coded by the user, it can follow steering parameters defined by the user or it will execute all connected methods in the order they where instanciated. The last is used in the default controller, but the controller can be overridden if the user has specific needs. In the Habitat model a special controller is made that uses a control parameter block to create and execute the methods that the user wants to run. The parameters can be configured in such a way that the model can run interactively.

When the controller is made, all the main parts of the model is ready. Figure 8 shows the system structure for the HMF based habitat model.

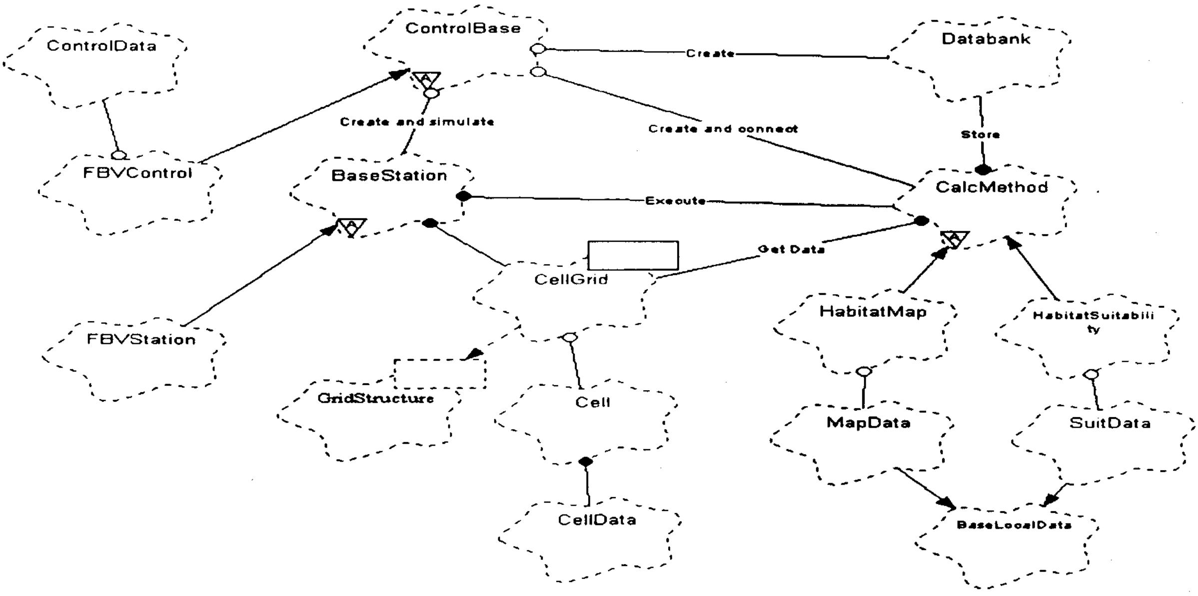


Figure 8:Class structure for the Habitat model

The SSIIM model (Olsen, 1996) has been used in several areas for hydraulic calculations. The model is based on a three dimensional grid structure, similar to the one used in the framework. The Habitat model was created to take advantage of the detailed simulation capabilities found in SSIIM. To integrate SSIIM, a SSIIM specific station class was derived, and the creation function was reimplemented so that it creates the station from SSIIM data. The habitat analysis methods are similar to the ones made for the Hec-2 based habitat model, no changes are necessary to make the methods available on the SSIIM generated data. This example illustrates how use of framework allows the users of the system to change a part of the system (here the station object) without interfering with the other parts. If new methods are necessary to take full advantage of the new data available in SSIIM these can be added as derivations of the calculation method class, still the original methods will be unchanged.

Figure 9 shows an example of results produced with the Habitat model.

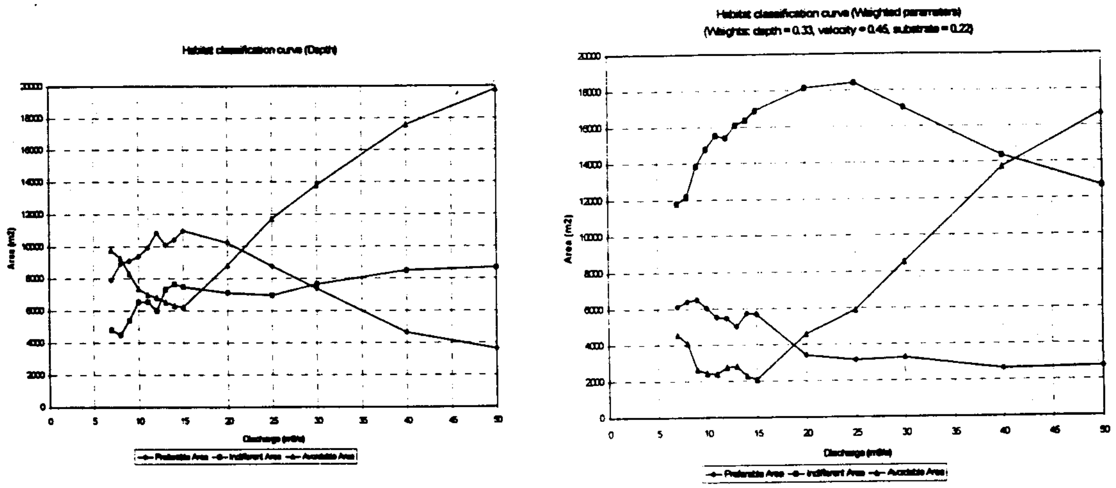


Figure 9: Examples of results produced by the Habitat model

7. CONCLUSION

A general object-oriented framework for creation of habitat analysis models has been created. The framework offers high degree of code reuse, adaptability to user defined problems and an open extensible system where new methodologies can be easily incorporated. The framework is based on a river reach divided into a rectangular grid structure.

New research in the field of habitat modelling required upgrades to the Habitat model in the River System Simulator. To simplify future extensions and to create a model that can be changed by users not familiar with the entire model structure, the framework was applied to the implementation. The use of the framework also made it possible to include hydraulic data from the 3D hydraulic model SSIIM into the Habitat model.

REFERENCES

1 Gamma, E., Helm, R., Johnson, R. and Vlissides, J. (1995) Design Patterns - Elements of Resusable Object-Oriented Software. Addison Wesley Professional Computing Series. ISBN 0-201-63361-2

2 Booch, G. (1994a) Designing an Application Framework. *Dr. Dobb's Journal.* 19(2) 24-30.

3 Reenskaug, T. (1996) Working with objects: The ooram software engineering method. Manning Publications Co. ISBN 1-884777-10-4.

4 Olsen, N.R.B. and Alfredsen, K.T. (1994) A Three-dimensional Numerical Model for Calculations of Hydraulic Conditions for Fish Habitat. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* Trondheim, Norway August 18-20 1994. P 113-124.

5 Vaskinn, Kjetil A. (1985) Fysisk Beskrivende Vassdragsmodell. SINTEF Report A1. Norsk Hydroteknisk Laboratorium, Trondheim 1985.

6 Olsen, N.R.B. (1996) SSIIM User's Manual, version 1.4. The Norwegian University of Science and Technolology, Division of Hydraulic and Environmental Engineering.

7 Killingtveit, Å. and Fossdal, M.L. (1994). The River System Simulator - An integrated model system for water resources planning and operation. *HYDROSOFT 94,* Porto Carras, Greece September 21-23 1994.

8 Rinde, T. (1996) PINE - a hydrological model with flexible model structure. To be presented at the Nordic Hydrological Conference, Akureyri, Iceland August 13-15 1996.

9 Alfredsen, K and Killingtveit, Å. (1996) Object-oriented framework for hydrological models. . To be presented at the Nordic Hydrological Conference, Akureyri, Iceland August 13-15 1996.

10 Booch, G. (1994b) Object-Oriented Analysis and Design with Applications, 2nd ed. Benjamin Cummings Publishing Company Inc. ISBN 0-8053-5340-2

11 Bovee, K.D. (1982) A guide to stream habitat analysis using the instream flow increamental methodology. *Instream Flow Informaton Paper No. 12.* Fort Collins, Colorado.

12 Ghanem, A. Steffler, P. Hicks, F. and Katopodis, C. Two-dimensional finite element flow modeling of physical fish habitat. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* Trondheim, Norway August 18-20 1994. P 84-98.

PREDICTING HABITAT COMPONENTS FOR SEMI-NATURAL RIVERS IN THE UNITED KINGDOM

Fox P.J.A.[1], M. Naura[1] and P. Raven[2]

Environment Agency, North West Region, Richard Fairclough House, Knutsford Road, Warrington WA4 1HG, UK.

[2]Environment Agency, Rivers House, Waterside Drive, Aztec West, Almondsbury, Bristol BS12 4UD, UK.

ABSTRACT

The Environment Agency (covering England and Wales) has developed, tested, and implemented a new survey methodology, based on recording the physical structure of channel, banks and adjacent land use of 500m stretches of river. It has been applied to a stratified random sample of watercourses throughout the UK and by December 1995 records from over 3400 had been compiled onto a National computer database.

One main purpose for data collection was the development of a classification of river habitat quality as a measure of wildlife value, its careful management being a statutory duty of the Agency. The aim was to compare the expected structure of natural sites with modified equivalents (commonly managed by resectioning for land drainage or flood defence purposes), the difference between expected and observed used to assess "naturalness".

The approach adopted was to interrogate channel substrate composition and flow related biotopes (for example, riffles), chosen as a key component of channel habitats, of semi-natural sites (n=944) to establish relationships between these and a set of stable descriptors (not affected by human management). Relationships were established with solid and drift geology, altitude, slope and distance from source and height of source. Thresholds for each were developed which together provided a simple set of rules for describing the substrate composition for these sites. These combinations were subsequently combined into 4 substrate groups. Differences of flow biotopes within these 4 were used to separate these further into a series of 9 river types, each describing a distinct semi-natural character.

The data for semi-natural sites were used to generate predictions for new sites, for each type. To validate these they were applied to a sub-set of semi-natural sites (n=236), each of which had been allocated to its respective type but excluded during construction of the prediction. Tested predictions for dominant substrate combination was correct on average 75% (n=236). Significant differences in the abundance of selected flow biotopes were observed in 33% of cases (n=27). Further work will identify the most accurate predictions for each type and these will be used to measure the distance from "naturalness" of modified sites, which will feature as part of the scoring system for wildlife value as part of a modification index.

KEY-WORDS: River structure / prediction / naturalness / channel substrate

INTRODUCTION AND AIMS.

The England and Wales Environment Agency (EA) has a statutory duty to further wildlife conservation and therefore an operational requirement for a classification of river habitat quality. The River Habitat Survey (RHS) methodology for field data collection was developed by the EA to provide site information for this development.

Papers are currently being prepared which describe the River Habitat Survey method (Fox *et al*, 1996, in prep) used to supply data for the analyses presented in this paper. This paper covers the development and preliminary validation of a model to predict the natural structure for British rivers, and describes the range within a series of river types. Further papers will use a wide range of predictions in the assessment of habitat quality.

Some contemporary approaches to classification use predictions for "naturalness" against which degraded sites are scored, a low score indicating a high degree of modification, for example RIVPACS (River InVertebrate Prediction and Classification System) system developed in Britain by the Institute for Freshwater Ecology (Wright *et al.*, 1993). Here, invertebrate presence is predicted from a series of environmental variables and sites are scored by comparing their performance against this. A similar approach was desired for RHS, the considerable dataset of sites from across Britain offering semi-natural sites from which predictions could be developed.

Much work has been done, for example Schumm (1977), describing river morphology and processes by dividing catchments into stretches or segments where physical character, described by channel substrate, flow types, channel and bank habitats and geomorphological features, are approximately similar. These divisions have often been used for descriptive purposes and have been applied to individual catchments. Some work has been done into inter-catchment comparison (Hack, 1957). This work indicated that bed material size is similar at sites on different catchments when these are classified according to their geology, slope and distance from source.

These ideas were used to develop and produce a predictive capability for natural structure of British watercourses. The approach adopted was to interrogate the least modified surveyed sites (termed semi-natural) to build predictions for selected habitat features, the occurrence and abundance of which could then be compared at modified sites. To link unmodified and modified sites data for a series of background variables (for example, slope) were provided. The selected habitats at semi-natural sites were linked to and described by these and a presumption made that at similar slopes at modified sites any observable differences in flow biotopes or substrate could be attributed to modification.

METHOD.

The RHS field data collection methodology comprises a four page form prompting surveyors at the river-side to answer standard questions, many appearing in the form of a tick-list. Field data collected at each 500m site produces approximately 400 separate records of habitat and structural features including predominant substrate of bed and banks, and flow types. The most important features of channel (substrate and flow types) and bank are collected from 10, 1m wide transects spaced at 50m intervals along the survey site. This method was adopted after tests from previous experimental work within the project indicated that this intensity of collection provided a adequate measure of site diversity without significant data redundancy (Fox *et al.*, in prep). Larger and rare habitat features, for example the presence of cascades and waterfalls, are recorded once in a "sweep-up" for the whole site. Riparian land use is also collected at this scale. Information in the sweep-up is recorded as absent, present or extensive (where abundance exceeds 33% of the available area).

Features were as closely defined as possible to reduce between-surveyor error and published and tested definitions used to increase confidence of applicability. For example, substrate are sized using an amalgamated Wentworth classification (Wentworth, 1922), ie., Boulders >256mm, Cobble 64-256mm, Gravel/Pebble 2-64mm, Sand 0.06-2mm and Silt <0.06mm. These were expanded to include clay (cohesive), bedrock and peat. The amalgamation was agreed through field trial which indicated a few difficulties, for example, in confidently recording differences between gravel and pebble.

By April 1996 the survey had been applied to over 3400 sites in the UK, comprising 2 sites per 10km^2 in England and Wales and 1 per 10 km^2 in Northern Ireland and Scotland. The stream order of a sample of surveyed sites (n=1521) sites was calculated and compared with a national survey of stream order generated by Smith and Lyle (1979), both using the Strahler method (Strahler, 1964). The proportions in each category were very similar, suggesting that this dataset is a representative sample of UK watercourses.

To complement field data collection a series of background variables were collected for all surveyed sites. These represented those variables which would be least affected by human management and included solid and drift geology, valley slope, altitude, distance from source and height of source. Standard methods were adopted and published maps used, a geographic information system has subsequently replaced this method. The resultant data was stored on MINITAB.

A simple algorithm was developed and applied to the data to select the semi-natural sites to remove those where channel realignment or bank reinforcement will have modified habitats. 1180 sites were selected representing approximately 35% of the data. Investigation indicated that these data over-represented the "upland" and steep slope channels within the data where land value or access reduced the desire for modification. Twenty percent of these sites were selected (n=236) and reserved for validation of the model subsequently developed, leaving 944 sites used within prediction development.

Channel substrate were selected from the field data to represent the core habitat component of watercourses. A number of reasons are given. Firstly, the practical consideration of attempting to interpret results rules-out the use of a large data set. Secondly, given a sub-set was necessary, channel substrate and flow biotopes produce key habitats themselves (Moon, 1939) and have been shown to be closely associated with others (Leopold *et al.*, 1964; Wolman, 1990; Padmore et al., 1995). Thirdly, the Environment Agency has little direct control over riparian land use, hence conservation effort focuses on channels.

Analyses first concentrated on the development of predictions for substrate. The first step was to identify and describe the variation in channel substrate using the background variables. Relationships between the background variables and channel substrate were established through a series of parametric and non-parametric tests.

Two very important results emerged from initial work. Firstly, a short-list of the best background variables were identified which described differences in channel substrate. These were solid and drift geology, altitude, slope, distance from source and height of source. Of these, solid geology was the most important, it was the only one which described the absence of certain substrate. Secondly, some mixture of substrate was recorded at almost all sites, a single substrate was recorded only 20% of the time, usually two were recorded at 3 or more transects. Hence, emphasis was placed on mixtures of two substrate.

These results dictated methods used for describing channel substrate variation. As solid geology was the best single descriptor for channel substrate each of the 73 solid geologies represented within the 944 sites was interrogated alone. To further describe substrate combinations within each geology, relationships with other background variables were investigated within each. A series of simple rules were constructed defining thresholds for other background variables, a cluster analysis was undertaken (Euclidean distance minimum variance) to assist this process. Nine approximately distinct substrate clusters were identified from the 944 sites and these were numbered (by decreasing substrate size combination) for this exercise.

The thresholds were used to help identify small groups, termed "sets", of sites with similar substrate composition. This work fully considered the within-geology variation, between-geology variation had not been studied. It became clear than the combinations of substrate within the sets were similar and that agglomeration of sets could be achieved. For this the mean occurrence for each substrate was generated for each of the resultant sets and these were subjected to a cluster analysis (Euclidean distance minimum variance) to investigate their similarity. The total variation explained by different numbers of clusters was investigated to guide the process of agglomeration. Resulting from this, and using the cluster analysis results, each set was characterised by the mixture of their two most common substrate. These were then amalgamated into 4 substrate groups. The occurrence of substrate was then tested on the sample data set. Two methods were used. Firstly, that either of the predicted dominant substrate were indeed dominant at "new" sites, and secondly, that either of those two substrate were present.

This concluded the description of differences between substrate, attention then focused on flow biotopes. The occurrence of glides, rapids and cascades at sites (as recorded in the sweep-up) within each of the four substrate groups was investigated using ANOVA. Three treatments for each flow biotope were tested (absent, present and extensive) with slope, altitude, distance from source and height of source. Investigation of variation within each indicated that differences occurred and could be correlated with some of the background variables. Accordingly, the four groups were split creating 9 semi-natural river types.

The predictive capability was then tested. Predictions for the flow biotopes were generated for each of the 9 types. The rules for definition of the river types were then applied to the validation data and the predictions were applied using Chi-squared tests. The success of prediction is expressed by the variation between the abundance of the selected features in the population with that of the sample, the null hypothesis being that the sample distribution matched the population.

RESULTS.

The decision to interrogate all solid geologies individually was justified on the basis that solid geology described the greatest amount of variation in substrate composition of sites. Table 1 illustrates this, showing the occurrence of different substrate categories for 5 solid geologies. This shows that some substrate are very rare or absent from some solid geologies, also note that at least two occur at a majority of sites in each geology. This indicates the prevalence of substrate mixes at sites.

Table 1: Showing the occurrence of substrate (expressed as a percentage), by solid geology.

Solid Geology Number	Bedrock	Boulder	Cobble	Gravel/ Pebble	Sand	Silt	Clay	Peat
34	41	75	71	58	13	8	0	0
72	40	23	67	85	6	6	0	0
81	32	38	70	58	8	15	1	0
94	33	56	67	56	28	56	20	0
108	0	0	3	86	31	59	24	0

Hence, each produced a slightly different substrate "fingerprint". The other background variables provided generally fewer significant relationships with substrate. The occurrence of significant results between these and the other background variables (for the same set of solid geologies as introduced in Table 1) is shown in Table 2.

Kruskal-Wallis tests were undertaken on the differences between substrate occurrence to illustrate that the occurrence of drift geology deposits describes some differences in substrate composition of watercourses running through them. This non-parametric test used as the distribution of substrate in highly skewed to the left. Using the examples shown above, within solid geology 72 (a sedimentary deposit of the silurian period) significantly more silt ($p \leq 0.05$) is found in watercourses running through alluvium drift deposits than those without drift. Also, for solid geology 81 (a sedimentary deposit of the carboniferous period) significantly more gravel/pebble is found ($p \leq 0.05$) in watercourse running through areas without drift than through peat deposits. Also, for this geology, significantly more cobbles ($p \leq 0.05$) are present in channels running through boulder clay deposits than through peat.

Table 2: Showing the occurrence of significant relationships ($p \leq 0.05$) between background variables and substrate, by solid geology.

Solid Geology Number	Drift	Altitude	Slope	Dist from Source	Height of Source
34				X	
72	X				X
81	X	X	X		X
94					X
108		X			X

Correlations between substrate abundance and the other variables were generated to investigate whether these describe more differences in substrate composition (these continual variables were transformed to normalise them; the natural log of altitude, slope and distance from source and the square root of height of source were used). Significant relationships were found, these too are illustrated in Table 2. This shows that increasing altitude of sites within solid geology 108 (a sedimentary deposit of the Palaeogene period) is significantly positively correlated with gravel/pebble and negatively correlated with silt. Increasing slope is significantly positively correlated with abundance of bedrock and boulders and negatively correlated with abundance of gravel/pebble in geology 81. Increasing distance from source is significantly positively correlated with abundance of boulder and negatively correlated with abundance of gravel/pebble within geology 34 (granite). Increasing height of source is significantly positively correlated with bedrock, boulder and cobble and negatively correlated with silt within geology 94 (a sedimentary deposit of the Jurassic period).

These illustrative results show that the background variables describe further the differences between substrate found at sites than solid geology alone. Their descriptive capability is not consistent across all geology groups and therefore was not treated as describing the same effect at one value (for example, all rivers with slope greater than 50m per km will have cobbles and boulders). Hence, thresholds did differ between solid geology in order to maximise variance explained.

The number of sites with each solid geology varied, those geologies which are rare in Britain or whose watercourse have suffered a high proportion of modification were poorly represented. Of the 73 geologies represented, 48 had 10 or less semi-natural sites. Hence, whilst it was possible to illustrate statistically the relationships between background variables and substrate within geologies represented by a large number of cases, the use of this technique in the definition of rules was confounded for most by small sample size. Therefore, many represent expert interpretation, based on an understanding of the underlying nature of the data and of the effects described by the background variables (eg, high slope describes larger substrate). A typical screen display is shown in Table 3. Substrate abundance at sites is represented by the number of transects where each was recorded as predominant, results of the cluster analysis are also shown (down the right margin).

Table 3: showing typical substrate composition of sites (expressed as number of recorded transects) with same solid geology, ordered by increasing slope.

Slope m/km	Bedrock	Boulder	Cobble	Gravel/ Pebble	Sand	Silt	Clay	Cluster results
2	0	0	1	3	0	6	0	6
2	1	1	3	5	0	0	0	4
6	0	0	3	4	0	2	0	6
14	0	0	8	2	0	0	0	4
15	0	2	8	0	0	0	0	3
27	2	6	2	0	0	0	0	2
27	0	2	7	0	0	1	0	3
35	0	5	5	0	0	0	0	3

This table shows the influence of slope on substrate combination and indicates a split at slope ≥15m/km, above this value all sites are dominated by some combination of boulders and cobbles. Similar thresholds were identified for each stable descriptor and applied to each solid geology to identify sets of sites within each with similar substrate composition. For example:

Solid Geology = 10 & Altitude≥160<490 & Slope <50m/km

This simple algorithm describes a set of natural watercourses which are characterised by some mixture of boulders and cobbles in their channels. Such rules identified 326 sets used slope as thresholds in their definition most frequently, 42 times, altitude 28, Distance from source 22, drift geology 16 and Height of source 14 times respectively. The rules agglomerated the 944 sites into a series of 326 sets.

Whilst the definition of the within geology variation was considered during this process the between geology variation was not. Many of the resultant sets were very similar and the 326 were condensed. The creation of four groups based on substrate was based on the results of a cluster analysis of the substrate fingerprints for the 326. Figure 1 shows the distance level between clusters resulting from a cluster analysis of the substrate of the 326 sets.

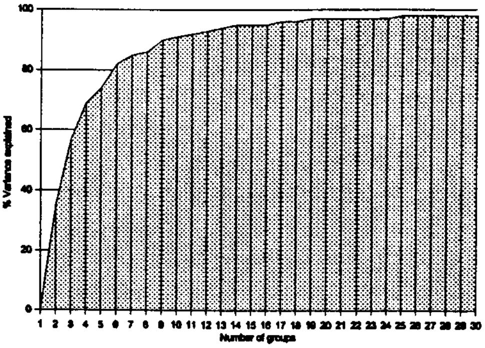


Figure 1 Showing the cumulative percentage variance explained within groups identified by cluster analysis (Euclidean distance minimum variance) on substrate of semi-natural sites.

Attention focused on the creation of around 4-6 groups, explaining approximately 70-80% of the variation. Investigation of the combinations of substrate within each of the 326 sets revealed that in 210 sets (71%) only one or a combination of two substrate was dominant (combined mean >5 transects). These represented mixtures of bedrock and boulder, boulder and cobble, cobble and gravel/pebble and gravel/pebble, sand and silt. Hence, four groups were created around these mixtures and each set was allocated to one using a series of simple algorithms. The remaining sets comprising small numbers of rarer combinations, for example, cobble with bedrock or silt with sand, were allocated to the nearest appropriate group. Group 1 comprised sites dominated by gravel/pebble and/or silt and/or sand. Group 2 comprised sites dominated by gravel/pebble and/or cobble; group 3 comprised sites dominated by boulder and/or cobble and group 4 comprised sites dominated by bedrock and/or boulder.

Tests of the predictive power of the 4 groups were made using the 20% of semi-natural sites put aside before prediction development. Two tests were conducted. Firstly, the presence of either of the two expected dominant substrate was tested and was found to be present in 88% of cases (n=236). Secondly, that one or other of the expected dominant substrate were dominant was found to be true in 77% of cases.

The results of the ANOVA of flow biotopes within each of the four groups showed that slope produced the greatest number of significant relationships. Increasing slope was observed to be highly significantly positively correlated ($p \leq 0.01$) with increasing abundance of cascades and negatively correlated with abundance of glides within all four groups. Whilst other background variables provided similar results, especially altitude and distance from source, significant results were not observed for all groups.

The 95% confidence intervals for slope for each level of abundance of cascades (absent, present, extensive) generated by the ANOVA tests provided clear indication of breaks within the four groups. Divisions were created within each between the relatively high and low slope sites at thresholds which would clearly put any site in one or other abundance category. Nine river types were created from this and a brief description for each is provided in Table 4. Co-incidentally, and resulting from the high correlations between slope and distance from source, the nine have distinct size differences. A size descriptor can be linked to distance from source as channel width as measured at the site is highly significantly positively correlated ($p \leq 0.01$) with distance from source.

Table 4. Showing the mean and standard error of mean for slope and distance from source, with substrate mix of the 9 river types.

River type	Slope descriptor	mean	(SEmean)	Substrate mix	Distance from source descriptor	mean	(SEmean)
1	v. low	0.97	(0.045)	gravel/pebble-sand-silt	rivers	24.99	(2.14)
2	low	6.56	(0.605)	gravel/pebble-sand-silt	streams	8.37	(0.58)
3	low	5.82	(0.237)	cobble-gravel	rivers	14.73	(1.07)
4	high	36.01	(2.47)	cobble-gravel	streams	3.42	(0.44)
5	low	8.34	(0.547)	boulder-cobble	rivers	16.56	(1.93)
6	medium	24.63	(0.831)	boulder-cobble	streams	4.91	(0.75)
7	high	88.30	(10.80)	boulder-cobble	streams	3.36	(0.41)
8	v. low	1.81	(0.345)	bedrock-boulder	rivers	24.99	(19.9)
9	high	40.20	(4.93)	bedrock-boulder	streams	6.98	(0.92)

A series of probabilities of occurrence of cascades, rapids and glides where then generated for the nine types and these where tested against the sample population from the validation data set to test whether the predictions were robust when tested on "new" semi-natural sites.

A total of 27 Chi-squared tests were conducted (9 types x 3 biotopes) and significant differences between the sample and the population were observed in 9 cases (33%), three for each biotope. Further predictions for other features will be generated and tested to expand the range available.

DISCUSSION.

The system appears to be able to predict substrate combinations with considerable success. The prediction of flow features is less successful, the fact that sites were grouped principally by their substrate composition is probably a factor here. The results presented in this paper are preliminary, further predictions will be generated and tested and a suite of the best for each type developed. These will be taken forward into the process of building a classification of modification, significant differences in habitat features between semi-natural and modified sites attributed to management. These changes should prove the level of habitat change consequent with management for flood defence and land drainage purposes.

A number of considerations need to be addressed in presenting these results. The predictions are built from the semi-natural sites. Totally unmodified sites comprised approximately 20% of sites surveyed, this was considered too small and minor bank modification was accepted (one transect per bank only) to increase the selection. It was recognised that this decision was not based on evidence that habitats had altered by this, however, extensive field experience suggests that major habitat modification would be unlikely.

The influence of some of the background variables, especially geology, slope and distance from source, on channel substrate is accepted (Hack, 1957). For others, particularly altitude, the relationship is apparent but not immediately explicable, is seems likely to be a surrogate for other effects, perhaps climate. Also, the high degree of correlation between the background variables may suggest that altitude is a surrogate for slope. Altitude was used because it helped describe variation in substrate composition. No attempt was made to describe and explain some relationships, especially those shown by drift geology, this was not a central demand of this work. It was sufficient to know they existed and how they effected substrate.

Cluster analyses were frequently used to indicate and assist development of associations between sites and background variables. The results were interpreted closely to establish the basis upon which clustering had occurred and from this strict algorithms developed upon which groupings were implemented. The level of transparency of working introduced by this lead to a clearer understanding of the synthesis of the system.

The overriding importance of solid geology in determining substrate composition was shown. The interrogation of sites by geology reflected this but lead to problems of dealing with small sample sizes. Whilst the general relationships between channel substrate and the other background variables was established this could not be proved statistically at this scale. Hence, the thresholds for most sets were determined using best judgement. It seems unlikely that further survey will completely eliminate this problem, a wider definition of semi-natural would increase site numbers but with a consequent concern over the introduction of modified site character.

ACKNOWLEDGEMENTS.

The authors wish to thank the following for important contributions to this work: Nigel Holmes, Mark Everard and Hugh Dawson.

REFERENCES.

Hack, J.T. (1957). Studies of longditudinal stream profiles in Virginia and Maryland. *US Geol. Survey Prof* Paper 294-B, pp, 97.

Leopold, L.B., M.G. Wolman and J.P. Miller (1964). *Fluvial processes in Geomorphology*. W.H. Freeman and Co, San Francisco and London, 522pp.

Moon,H.P. (1939). Aspects of the ecology of aquatic insects. *Transactions of the British Entomological Society.* 9 pp. 39-49.

Padmore, C.L., M.D.Newson and M.E. Charlton (1995). Instream habitat: geomorphological guidance for the identification and characterisation of biotopes. In K.M. Rowntree (ed) *The hydraulics of physical biotopes - terminology, inventory and calibration.* Report of workshop held at Citrusdal, SA, 4-7 February 1995.

Schumm, S. A. (1977). *The fluvial System.* Wiley Interscience, New York.

Smith, I. and A. Lyle (1979). Distribution of freshwater in Great Britain. Institute of Terrestrial Ecology, HMSO, London.

Strahler, A.N. (1964). "Geology. Part II. Quantitative geomorphology of drainage basins and channel networks". In V.T. Chow (ed) *Handbook of applied Hydrology,* McGraw Hill, New York. pp 4-39 - 4-76.

Wentworth, C.K. (1922). A scale of grade and class terms for clastic sediments. *Journal of Geology.* 30, pp.377-392.

Wolman, M.G. (1990). Pattern and origin of stepped-bed morphology in high gradient streams, Western Cascades, Oregon. *Geological Society of the American Bulletin.* 102 pp340-352.

Wright, J.F., M.T. Furse and P.D. Armitage (1993). RIVPACS - a technique for evaluating the biological quality of rivers in the U.K. *Eur. Water Pollut. Control.* 3(4) pp, 15-25.

Application of the Instream Flow Incremental Methodology to Conservation Flow for Freshwater Fishes in Japan

Nobuyuki TAMAI, Hyeong Ryeol KIM, and Yoshihisa KAWAHARA
Dept. of Civil Eng., Univ. of Tokyo, Bukyo-ku, Tokyo, 113, JAPAN
E-mail : khr@hydra.t.u-tokyo.ac.jp

ABSTRACT

Recently, the request for ecological aspect in river improvement works becomes gradually stronger in order to offer higher quality of natural space to people. We adopted the Instream Flow Incremental Methodology(IFIM) to determine a conservation flow for freshwater fishes in Japan. Hydraulic variables(velocity, depth, and substrate) for pale chub(*Zacco platypus*), dark club(*Zacco temminckii*), and river lizard goby (*Rhinogobius flumineus*) are utilized to derive a habitat suitability function for target fishes observed in 1994 in Oto River. And we surveyed water temperature and cover(i.e. shadowed area on water surface by trees along the riverside) during the summer in 1995 to enhance the IFIM model by introducing the relationship between these two habitat variables and density of target fishes. To apply the IFIM model to the study reach, we developed a one-dimensional mathematical model for a gradually varied flow. We derived accurate and objective habitat suitability functions between number of captured fishes and hydraulic variables(velocity, depth, and substrate) and found that cover was closely related to density of target fishes according to each habitat style. However, water temperature data during one day were not sufficient to explain a relation with density of target fishes. Variation in WUA(Weighted Usable Area), which shows a quantitative estimate of fish habitat, is found out to be approximately proportional to density of target fishes and a conservation flow for freshwater fishes in summer in Oto river is evaluated about 4m^3/sec by a one-dimensional mathematical model. This paper provides possibility to determine a conservation flow for freshwater fishes in Japan.

KEY-WORDS : IFIM, Conservation flow, habitat suitability function, macro-habitat variable, WUA, gradually varied flow

1. Introduction

Until recently river improvement works have been executed placing emphasis on two main objectives, that is flood control and water resources development. However, recently the request for preservation of riverine ecology to offer higher quality of open space becomes quite strong. Based on a social need, a new concept of nature-orinted river improvement works is introduced by the Ministry of Construction in Japan. But quantitative measures to evaluate ecological conditions in natural streams are far from completion. The Instream Flow Incremental Methodology (IFIM) is one method to evaluate a riverine habitat by ecology, specially fishes, which was developed by Aquatic System Branch of the National Ecology Research Center in U.S.(Brookes. 1988 ; Gore. 1989 ;Nestler et al. 1989 ; Orth and Maughan. 1982 ; Petts and Maddock. 1994). While the IFIM is useful to predict the effect of river improvement works and evaluate alternatives, it is still needed to determine utilization criteria of habitat variables objectively and precisely. And also improvement in a simulation technique is necessary to predict velocity and depth more accurately in natural streams. In general, one-dimensional hydraulic simulation models are used to predict velocity and depth in IFIM until recently. The velocity simulated is very sensitive to Weighted Usable Area(WUA) in IFIM(Milhous et al. 1989). However, its velocity by one-dimensional mathematical model is often inaccurate. To predict velocity more exactly in IFIM, two-dimensional mathematical model(Ghanem et al. 1994) and three-dimensional mathematical model(Olsen and Stokseth. 1994) were developed. In this study, first we analyzed utilization criteria of habitat variables based on several methods and found out a most suitable objective method for each target fish and each season. Secondly, influence of water temperature and cover was discussed to consider the relationship between fish habitat and macro-habitat variables(i.e. water temperature and cover). Finally, we determined a conservation flow for freshwater fishes in Japan by using a one-dimensional mathematical model. And we are developing a two-dimensional mathematical model for predicting changes in velocity and depth accurately under a specified discharge based on shallow water equations to improve the IFIM model.

2. Study area

Oto river(34°75'N, 137°10'E) which flows from east to west in Okazaki City, Aichi Prefecture in Japan, is a tributary of Yahagi river(Figure 1). This river has specific discharge of $0.032m^3/sec/km^2$ and catchment area of $271.1km^2$, and water course length is about 34km. The bottom slope of this river is 1/2,000~1/900 in downstream of Ohira weir and 1/400 in upstream of Ohira weir. The annual average rainfall for recent 5 years is 1,219mm. Water quality was not so good. BOD was 2.0*mg/l* in upstream but over 5.2*mg/l* in downstream(River Department, Aichi Prefecture, Japan.1995a).

3. Target fishes

The study reach in Oto river was from the junction with Otoko river to the junction with Yahagi river(See Fig. 1). 10 stations were selected in this study reach where each station was selected to include one pool and one riffle. The investigation for fishes were done for these stations during 1994 by capture, diving, and marking. Oto river supported

By the correlation coefficient between the relative frequency of captured fishes and each suitability index of utilization criteria obtained by four method, we determined a best utilization criteria of habitat variables for each target fish and each season. Figure 4 shows utilization criteria of velocity for pale chub in summer for each method. BEPM(D=20cm) was the method to determine utilization criteria of velocity at depth = 20cm. There were differences in the suitability indices of utilization criteria for each method, specially in 120~160cm/sec. In this case, univariate polynomial method was a most suitable method to determine utilization criteria of velocity in summer (the correlation coefficient was 0.999). A most suitable method to determine utilization criteria of habitat variables for each target fish and each season were presented in Table 1.

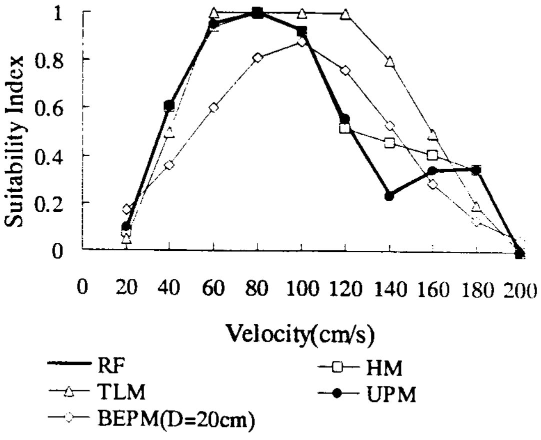


Figure 4 : Utilization criteria of velocity for pale club in summer for each method.

Table 1 : Most suitable method to determine utilization criteria of habitat variables for each target fish and each season in Oto river.

Fishes	habitat variables	spring	summer	autumn
pale chub	velocity	HM	UPM	UPM
	depth	HM	BEPM(v=80cm/s)	TLM
	substrate	UPM	HM	BEPM(v=60cm/s)
dark club	velocity	UPM	HM	UPM
	depth	UPM	UPM	HM
	substrate	UPM	HM	UPM
river lizard goby	velocity	UPM	UPM	UPM
	depth	HM	HM	UPM
	substrate	BEPM(v=80cm/s)	UPM	BEPM(v=40/s)

HM : Histogram Method, TLM : Tolerance Limits Method, UPM : Univariate Polynomial Method, and BEPM : Bivaiate Exponential Polynomial Method.

5. Relationship between macro-habitat variables and fish habitat

Water temperature and cover are macro-habitat variable that almost does not change transversely but changes in the

In case of univariate polynomial method(Bovee. 1986 ; Gore. 1989 ; Lambert and Hanson.1986) and bivariate exponential polynomial method(Bovee. 1986 ; Gore. 1981 ; Gore. 1989 ; Lambert and Hanson. 1986 ; Shirvell and Dungey. 1983), utilization criteria of habitat variables is obtained by solving the polynomial equation and the exponential polynomial equation for each method. Equation(1) is the basic equation for the univariate polynomial method and it is solved by least square method. The best equation was selected by the maximum coefficient of determination twice adjusted for degree of freedim($R^{\prime\prime 2}$). Equation(2) is the basic equation for the bivariate exponential polynomial method with or without interaction item and its best equation is also selected by $R^{\prime\prime 2}$. The partial differentials of SI in equation(2), that is $\partial SI/\partial X = \partial SI/\partial Y = 0$, are calculated to determine the value of X and Y that represented preferred habitat conditions.

$$SI=A0+A1*X+A2*X^2+A3*X^3... \qquad (1)$$

$$SI = \exp(A0+A1*X+A2*Y+A3*X^2+A4*Y^2+A5*X*Y) \qquad (2)$$

where, SI = suitability index, X and Y = habitat variables, X*Y = interaction between X and Y, and A0,A1,... = coefficient.

Figure 2 shows utilization criteria of velocity for pale chub in summer by univariate polynomial method(UPM). This utilization criteria was eighth order equation and $R^{\prime\prime 2}$ was 0.969. Figure 3 shows utilization criteria of velocity and depth for pale chub in summer by bivariate exponential polynomial method(BEPM). This utilization criteria had interaction item and $R^{\prime\prime 2}$ was 0.517.

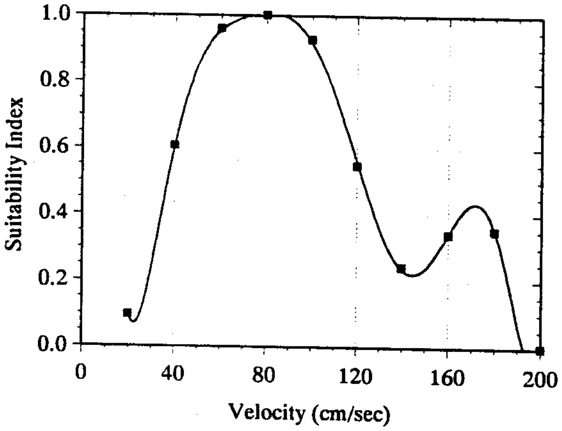


Figure 2:Utilization criteria of velocity for pale chub in summer by UPM.

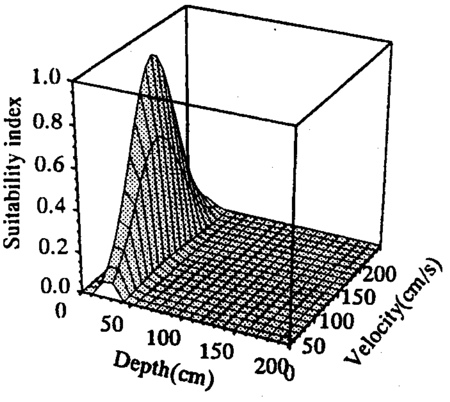


Figure 3:Utilization criteria of velocity and depth for pale chub in summer BEPM.

4.2 Comparison with utilization criteria for each method

a diverse fish community and 29 species were observed. By utilizing a report of fish habitat in Oto river(Okazaki work office, Aichi Prefecture, Japan.1994), we selected 3 species as target fishes to represent existing natural situation of Oto river. These target fishes were observed at all stations and in all seasons through a year except winter when the target fishes were rarely observed, that is pale chub (*Zacco platypus*), dark club (*Zacco temminckii*), and river lizard goby (*Rhinogobius flumineus*).

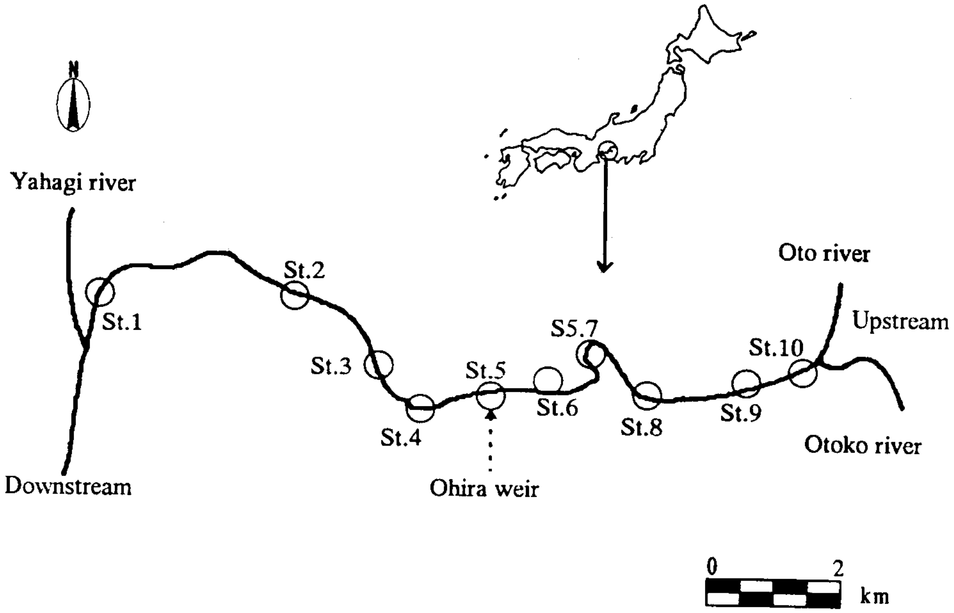


Figure 1 : Location of study sites on Oto river

4. Utilization criteria of habitat variables

To determine utilization criteria(or utilization function) of habitat variables(i.e. velocity, depth, and substrate) objectively and exactly, we utilized four methods based on Histogram Method, Tolerance Limits Method, Univariate Polynomial Method, and Bivariate Exponential Polynomial Method.

4.1 Consideration four methods to determine utilization criteria of habitat variables

In case of histogram method and tolerance limits method, the histogram of frequency of captured fishes is utilized. In histogram method, we determined utilization criteria of habitat variables by only connecting the middle point of each bar on histogram(Bovee. 1986). And in tolerance limits method, utilization criteria is obtained by assigning suitability index to the specified boundary of the frequency of captured fishes(Bovee. 1986 ; Thomas and Bovee. 1993). For example, suitability index of utilization criteria of 1.0 is assigned to 50% of frequency of captured fishes, 0.5 is assigned to 75%, 0.2 is assigned to 50%, and 0.1 is assigned to 95%.

longitudinal direction, although velocity, depth, and substrate are micro-habitat variables that may change significantly in both transverse and longitudinal directions in rivers(Nestler. 1989). We surveyed water temperature and cover in summer in 1995 to enhance the IFIM model by introducing the relationship between macro-habitat variables and fish habitat.

5.1 Water temperature and fish habitat

We selected seven stations(from station 3 to station 10) in the study reach and measured water temperature at two point, one was in riffle where the water depth was about 10cm and the other was in pool at the depth of 100cm from the water surface, at each station. We repeated the measurement three times a day(10 : 00~11 : 30,12 : 00 ~ 13 : 30, 14 : 00 ~ 15 : 30). Figure 5 shows variation of water temperature and density of target fishes. Station 10 was located at upstream end of the study reach. It was very difficult to capture the fishes in Station 7 because the water depth there, was about 6m. From figure 5, the difference in water temperature in pools was 1.1°C and variation of water temperature in riffle(the difference in water temperature was 2.8°C) was not related to density of target fishes. Water temperature data during one day were not sufficient to explain a relation with density of target fishes.

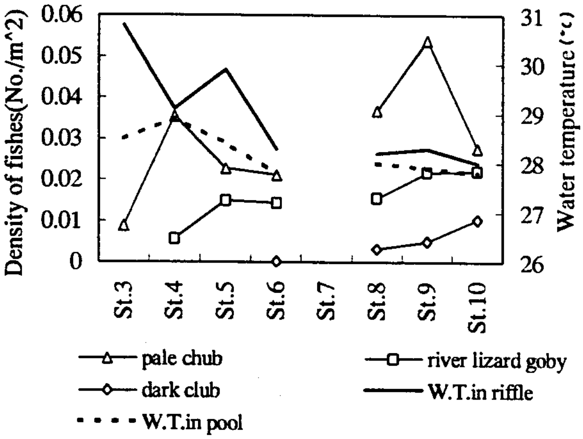


Figure 5 : Water temperature and density of target fishes in summer in Oto river.

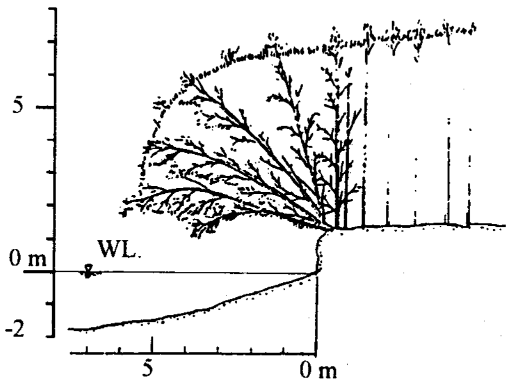


Figure 6 : Concept of cover(from River Department, Aichi Prefecture, Japan. 1995b)

5.2 Cover and fish habitat

We measured cover(i.e. shadowed area on water surface by trees along the riverside) by two cameras which provided streo image of cover at the same station with the measurement of water temperature during the summer in 1995. Station 4 and 7 were not measured. And by utilizing a reoprt of investigation for vegetation distribution in Oto river (River Department, Aichi Prefecture, Japan. 1995b), we also measured cover from the lengths of vegetation community along riverside and the length of vegetation community from waterfront toward river on both banks for each station, that was measured cover by direct survey. Figure 6 shows the concept of cover. Table 2 shows measured cover by a camera system and measred cover by direct survey in Oto river. The whole measured cover by a camera system was lower than measured cover by direct survey because error due to measurement methods in a camera system was very sensitive to the results. The variation of measured cover by a camera system in upstream was closely related to measured cover by direct survey. However, measured cover by a camera system in downstream was not related to measured cover by direct survey because measurement error in downstream due to wide river width had effect on measured cover in a camera system.

Table 2 : Measured cover by a camera system and measured cover by direct survey in Oto river.

Station	St.3	St.4	St.5	St.6	St.7	St.8	St.9	St.10
Area(m^2)	11,550	11,570	11,880	9,970	-	11,290	3,950	3,400
Measured cover by a camera system/Area (%)	1.69	-	0.78	3.84	-	0.74	3.04	6.59
Measured cover by direct survey/Area (%)	12.59	16.53	20.04	15.72	-	15.54	15.90	63.27

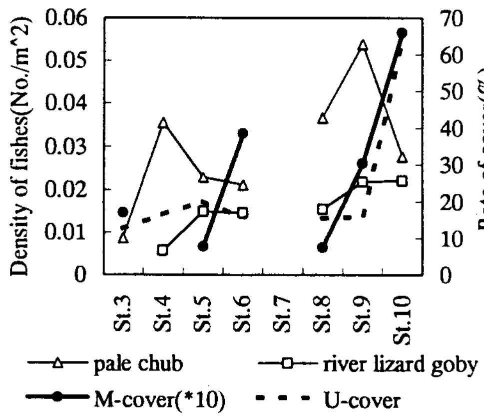


Figure 7 : Rate of cover and density of target fishes (pale chub and river lizard goby) in summer in Oto river.

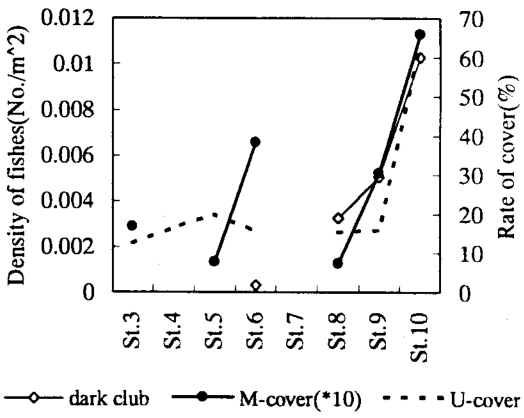


Figure 8 : Rate of cover and density of target fishes (dark club) in summer in Oto river.

Figure 7 and 8 show rate of cover(measured cover/area) and density of target fishes(number of captured fishes/area). Pale chub inhabitated in riffle, where the length of riffle in station 10 was about 56m although average length of riffle was about 210m in this study station and ate the algae lived in sunny place. Therefore, the density of pale chub was not related to cover. However, dark club inhabited in upstream and under the trees along the riverside and ate terrestrial insects from the trees and aquatic insects from the upstream. The density of dark club was closely related to cover. River lizard goby also was related to cover. The decrease of water temperature due to cover had effect on fish habitat(Barton and Taylor, 1985). In this study, however, water temperature data were not enough to discuss a relation with cover.

6. Evaluation of a conservation flow for target fishes by a one-dimensional mathematical model

To apply the IFIM to the study reach, a numerical simulation is necessary. RHABSIM(Riverine Habitat Simulation), that is an extensive conversion of PHABSIM(Physical Habitat Simulation), is used to apply the IFIM to the target stream. RHABSIM is a control program based on serveral modules : HYDSIM is one module for computing water surface levels and simulating velocity patterns. To predict water surface levels in HYDSIM, log-log regression model, channel conveyance model(or a uniform flow model), and step-backwater model(or a gradually varied flow model) are used(Payne. 1994). In this study, a one-dimensional mathematical model for a gradually varied flow is developed. Equation(3) was the governing equation of gradually varied flow(Chaudhry.1993). By standard step method, we simulated the water surface levels for a specified discharge.

$$(3) \qquad \frac{\partial h}{\partial x} = (I_o - \frac{n^2Q^2}{R^{4/3}A^2} + \frac{\alpha Q^2 h}{gA^3}\frac{\partial B}{\partial x}) / (1 - \frac{\alpha Q^2 B}{gA^3})$$

where, I_o = slope of channel bottom, n = Manning's roughness coefficient, Q = discharge, R = hydraulic radius, α = velocity-head coefficient, B = channel width, A = section area in channel, and g = gravity acceleration.

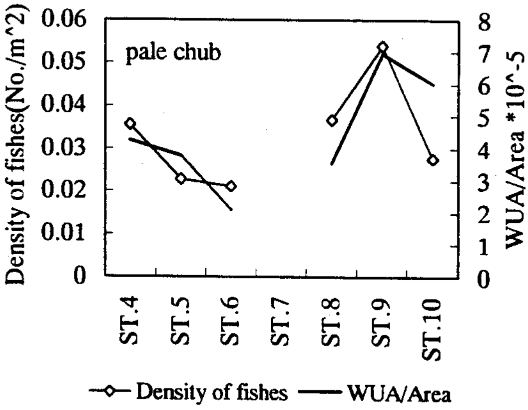


Figure 9 : WUA and density of target fishes (pale chub) at each station in summer.

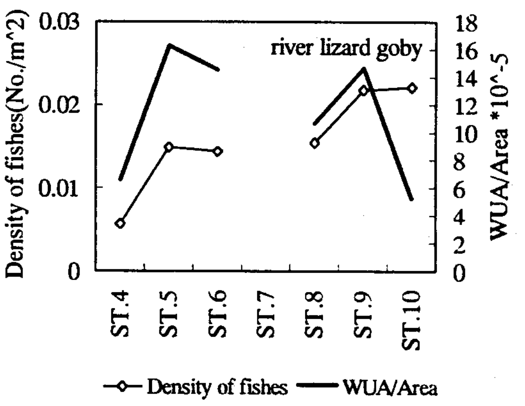


Figure 10 : WUA and density of target fishes (river lizard goby) at each station in summer.

For the water surface level simulated, we predicted velocity and depth. And by using these and substrate, we calculated Weighted usable area(WUA) at each station for each target fish and each season, which WUA is a quantitative estimate of fish habitat(Nestler et al.1989; Orth and Maughan.1982). Figure 9 and 10 show variation of WUA, which was divided by area and density of target fishes in summer in Oto river. WUA was closely related to density of target fishes except station 10 where the meandering part was about 120°. One-dimentional methematical model may not be appopriate to predict velocity and water depth in meandering part in natural streams.

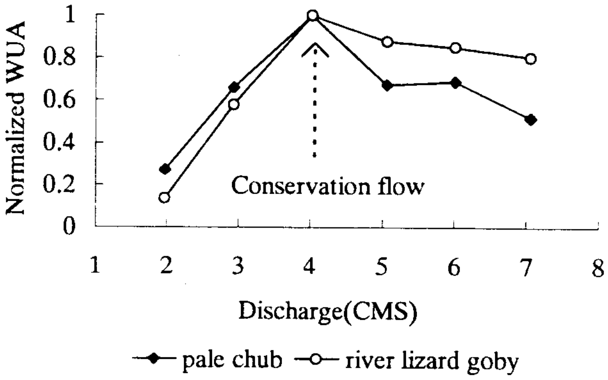


Figure 11 :Relationship between normalized WUA and discharges for target fishes in summer at station 5(Arrow represents conservation flow for target fishes.

These computations are repeated for each discharge of interest to make the relationship between WUA and discharges. From this relationship, a conservation flow is evaluated for each target fish and each season. Figure 11 shows variation of normalized WUA according to the change of discharges for pale chub and river lizard goby in summer at station 5 where was the typical station in Oto river. A conservation flow for pale chub was the same to that for river lizard goby. From this figure, a conservation flow for these fishes is evaluated about 4m^3/sec in summer in Oto river by a one-dimensional mathematical model.

7. Undergoing research work

Figure 12 illustrates a typical example of the contour lines of the bed level in Oto river reach. From this figure, it is clear that the river shows meandering with riffles and pools between main levees which also show heavy meandering. Therefore, the variation in streamwise velocity is large within a cross-section. Consequently it may not be appopriate to estimate the velocity and water depth for the IFIM using a one-dimensional mathematical model. It follows that a two-dimensional flow model is to be developed to reproduce the distributions of velocity and water depth in the river reach.

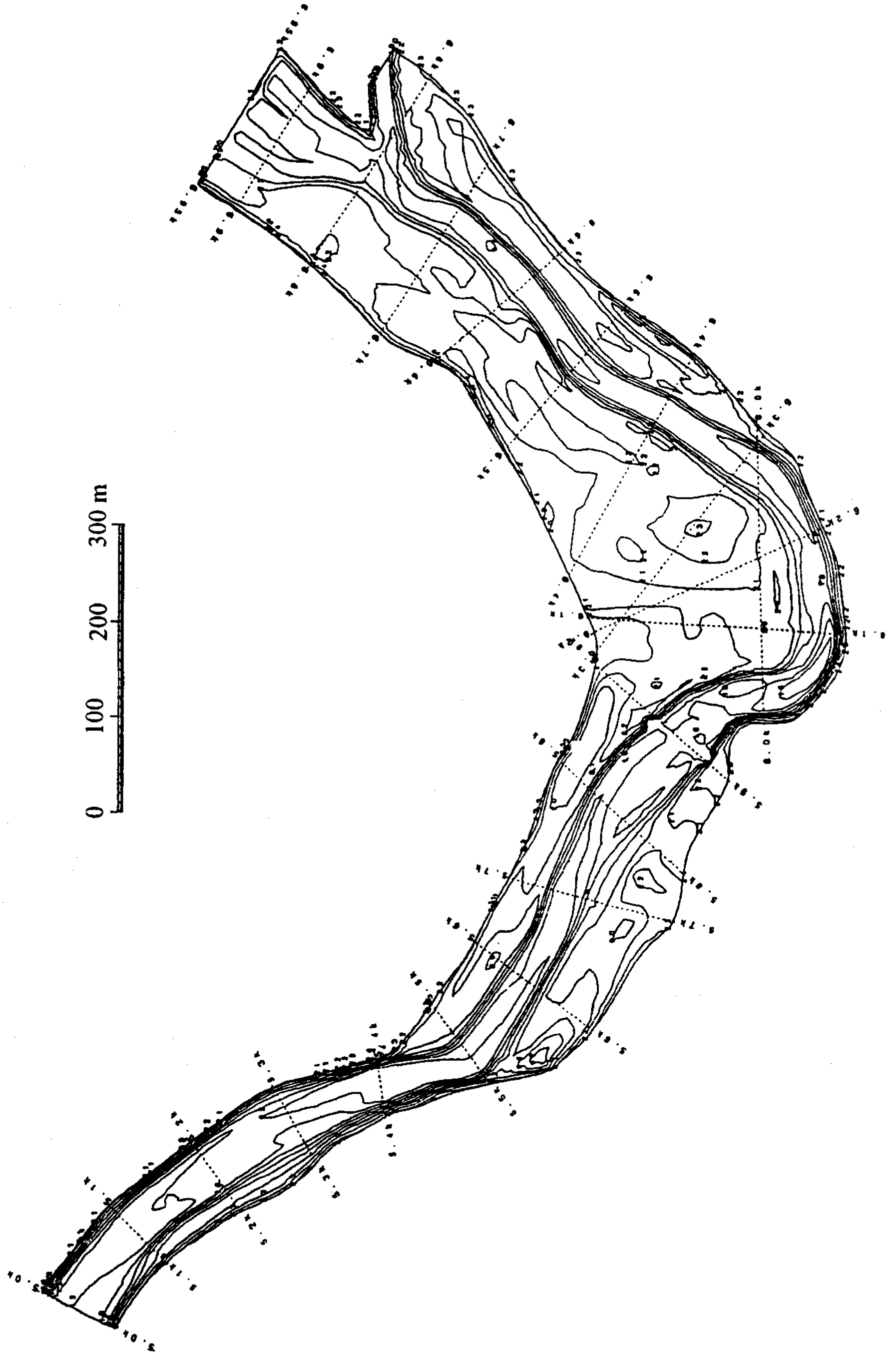


Figure 12 : Contour lines of the bed level in Oto river.

There are many ways to solve the depth-averaged velocity and water depth simultaneously. We are now develping a two-dimensional mathematical model for flow in rivers of complex geometry with special attention on the less grid-dependency of the calculated results. The model solves the shallow water equations together with the κ - ε model, where κ and ε stand for the depth-averaged turbulence kinetic energy and its dissipation rate. We use the finite volume method to descretize the basic equations and the SIMPLE algorithm to obtain the converged solutions of velocity and depth. The calculated results will be reported soon elsewhere.

8. Conclusion

We are able to conclude that first, a most suitable objective method to determine utilization criteria of habitat variables for each target fish and each season was obtained. In case of pale chub in summer, univariate polynomial method was a most suitable method for velocity, bivariate exponential polynomial method was a most suitable method for depth, and histogram method was a most suitable method for substrate(See Table 1). Secondly, cover was closely related to density of target fishes according to each habitat style. However, water temperature data during one day were not sufficient to explain a relation with density of target fishes. Finally, WUA was approximately proportional to density of target fishes except in a meandering part. And a conservation flow for freshwater fishes(i.e. pale chub and river lizard goby) was evaluated about 4m^3/sec in summer in Oto river by a one-dimensional mathematical model.
By this study, we showed possibility to determine a conservation flow for freshwater fishes in Japan.

Acknowledgments

The authors would like to express their thanks to Okazaki work office, Aichi Prefecture and FISCO Ltd. for providing the field data of the fishes. And also would like to thank to Research assistent Matuzaki, H., Mr. Morita, M., and Mr. Nakamura, U. in department of civil engineering, university of tokyo for their helpful supports in field measurement.

References

Barton,D.R. and Taylor,W.D.(1985). Dimensions of Riparian Buffer Strips required to maintain Trout habitat in Southern Ontario Streams. *North American Journal of Fisheries Management*. Vol.5.pp364-378.

Bovee,K.D.(1986). Development and Evaluation of Habitat Suitability Criteria for Use in the Instream Flow Incremental Methodology. FWS/OBS-86/7. Office of Biological Services. U.S. Fish and Wildlife Service.

Brookes,A.(1988). Channelized Rivers : perspectives for Environmental Management, John Wiley & Sons press. pp67-70

Chaudhry,M.H.(1993). Open-Channel Flow. Prentice-Hall. pp133-137.

Ghanem,A. et al(1994). Two-Dimensional Finite Element Flow Modeling of Physical Fish Habitat.*Proceeding of the 1st International Symposium on Habitat Hydraulics, IAHR.* Trondheim, Norway. pp84-98.

Gore,J.A.(1981). Predictive Models of Benthic Macroinvertebrate Density for Use in Insteram Flow Studies and Regulated Flow Management. *Canadian Journal of Fisheries and Aquatic Sciences*. Vlo.38. No.11. pp1363-

1370.

Gore,J.A.(1989). Models for predicting Benthic Macroinvertebrate Habitat Suitability Under Regulated River. Alternatives in Regulated River Management. CRC press. Boca Ration,Florida.pp253-265.

Lambert,T.R. and Hanson.D.F.(1989). Development of Habitat Suitability Criteria for Trout in Small Streams. *Regulated Rivers : Research and management*. Vol.3. pp291-303.

Milhous,R.T. et al(1989). Physical Habitat Simulation System Reference Manual-Version Ⅱ, Instream Flow Information Paper No.26. U.S. Fish and Wildlife Service. Biol.Rep.(9(16). p Ⅱ.127.

NAKAMURA,S. et al(1995), Field Research to obtain the IFIM's Preference Curves for Japanese Fresh Water Fishes. *Proceedings of Symposium on River Hydraulics and Environments*. JSCE. pp127-134. In Japan.

Nestler,J.M.et al(1989). Instream Habitat Modeling Technique,Gore,J.A. and Petts,G.E. editors. Alternatives in Regulated River Management. CRC press. Boca Ration,Florida.pp295-315.

Okazaki work office, Aichi Prefecture, Japan(1994). Report of Oto river in Heisei 6 year River Environmental Improvement Work. In Japan.

Olsen,N.R.B. and Stokseth,S.(1994). Three-Dimensional Medeling of Hydraulic Habitat in River with large bed roughness. *Proceeding of the 1st International Symposium on Habitat Hydraulics, IAHR.* Trondheim, Norway. pp99-112.

Orth,D.J. and Maughan,O.E.(1982).Evaluation of the Incremental Methodology for Recommending Instream Flows for Fishes. *Transactions of the American Fisheries Society*. Vol.3. No.4. pp413-445.

Payne,T.R.(1994). RHABSIM : User-Friendly Computer Model to calculate River Hydraulics and Aquatic Habitat. *Proceeding of the 1st International Symposium on Habitat Hydraulics, IAHR.* Trondheim, Norway. pp254-260.

Petts,G.E. and Maddock,I.(1994). Flow Allocation for In-river Needs,Calow,P. and Petts,G.E.editors. The River Handbook,Vol.2,pp289-308.

River Department, Aichi Prefecture, Japan(1995a). Summary of Oto river survey. In Japan.

River Department, Aichi Prefecture, Japan(1995b). Report of Investigation for Vegetation Distribution in Oto river. In Japan.

Shirvell,C.S. and Dungey,R.G.(1983). Macrohabitat Chosen by Brown Trout for Feeding and Spawning in Rivers. *Transactions of the American Fisheries Society*. Vol.112. No.3. pp355-367.

Thomas,J.A. and Bovee,K.D.(1993). Application and Testing of a Procedure to Evaluate Transferability of Habitat Suitability Criteria. *Regulated Rivers : Research and management*. Vol.8. pp285-294.

DETERMINING HABITAT CRITERIA FOR THE ENDANGERED FOUNTAIN DARTER THROUGH AQUATIC MAPPING AND HYDROLOGIC MODELING

Nathan Bartsch
Research Assistant, Institute for Natural Systems Engineering
Department of Civil and Environmental Engineering
Logan, UT 84322-4110
United States of America
danadana@aaron.cee.usu.edu

Chad P. Gubala
Associate Director, Institute for Natural Systems Engineering
Utah State University, Logan, UT 84322-4110
United States of America
chad@neptune.cee.usu.edu

Thomas B. Hardy
Director, Institute for Natural Systems Engineering
Utah State University, Logan, UT 84322-4110
United States of America
hardy@aaron.cee.usu.edu

ABSTRACT

The Comal River in Southern Texas, USA is a unique aquatic resource formed from natural springs of the Edwards Aquifer and represents one of two locations where the endangered Fountain darter (*Etheostoma fontanelles*) is found. In order to preserve and recover this species, the instream flow requirements must be determined and maintained. This paper highlights the application of advanced spatial data acquistion techniques, integration of 2-dimensional hydraulic modeling and multivariate flow dependant species criteria within the framework of the Instream Flow Incremental Methodology (IFIM). In particular, we highlight the application of an innovative combination of a shallow water hydroacoustic array combined with differential Global Position System (DGPS) survey equipment to delineate the spatial domain of the Comal River system. These data were utilized to construct a 3-dimensional finite difference grid which was developed through specialized krigging methods. This grid served as the map basis for delineating existing aquatic vegetation beds and the spatial distribution of substrates. The finite element grid was used to develop a one-dimensional water surface elevation model for the system and a 2-dimensional velocity simulation model. Stratified random sampling procedures were used to collect seasonal fisheries data which were analyzed by multivariate techniques to construct darter presence/absence equations based on hydraulic characteristics. The complete modeling framework was subsequently used for prediction of darter presence/absence at two different simulated flow rates.

KEY-WORDS: Instream Flow Incremental Methodology/ Habitat Modeling/ 2-Dimensional Hydraulics/ Hydroacoustics/ GPS/ Impact Assesment/ Endangered Species..

INTRODUCTION

The Instream Flow Incremental Methodology (IFIM) represents a conceptual framework for the evaluation of impacts of altered flow regimes or the assessment of instream flow requirements for target species (Bovee, 1995). The IFIM broadly considers the evaluation of the critical physical, chemical, and biological elements which may affect the flow dependant responses of target organisms in a particular system. Although the IFIM maintains an open architecture in terms of the specific analysis tools utilized, historically applications of have centered on the use of the Physical Habitat Simulation System (PHABSIM) (Milhous, 1989). PHABSIM focuses on physical habitat as the limiting factor and generally ignores other elements such as temperature or water quality. PHABSIM in particular has relied on the use of 1-D modeling techniques to simulate water surface elevations (WSEL) and velocities to predict the quantity and quality of fish habitat based on the target organisms prescribed suitabilities to depth, velocity and channel index (i.e. substrate and cover). With the resurgence of "more ecologically-based" assessment methods, new tools are being evaluated for data acquisition, hydraulic simulations, and habitat modeling options (e.g. Orth, 1995; Bain, 1995; companion articles in these proceedings). We explore the use of IFIM in light of several of these new analysis tools and modeling approaches to develop an assessment framework for instream flows for the endangered Fountain darter in the Comal River System. Given the limited space, we briefly outline the overall assessment framework applied in this study and then highlight the use of innovative linkages of hydroacoustics and GPS to acquire data amendable for one-dimensional and 2-dimensional hydraulic simulations. We then demonstrate the use of these tools to generate hydraulic model outputs which are used for assessing changes in the flow dependant quantity of darter habitat based on multi-variate prediction equations for darter locations under simulated flow conditions.

BACKGROUND

In general, the application of IFIM requires a delineation of the critical factors affecting the target organism(s), subsequent delineation of the these factors in the field, development, calibration, and validation of appropriate models, application of these models under alternative flow regimes, and negotiation or selection of an appropriate instream flow (Bovee, 1996). In terms of the Comal River and the endangered Fountain darter, available life history information suggests that the type and location of vegetation, temperature, dissolved oxygen, and velocity determine suitable conditions for habitat. Based on this information, a study was undertaken to broadly meet the following objectives: 1) characterize the aquatic environment within the Comal River; 2) develop suitable flow dependant "suitability criteria" for darters; and 3) evaluate flow ranges for their quantity and quality of darter habitat in order to recommend instream flow needs for the protection and recovery of the species. Figure 1 shows the overall IFIM assessment framework in light of the specific habitat requirements of darters and corresponding modeling components to meet the overall study objectives articulated above. In particular, we highlight the data acquisition, hydraulic modeling and integration of modeling components in this paper.

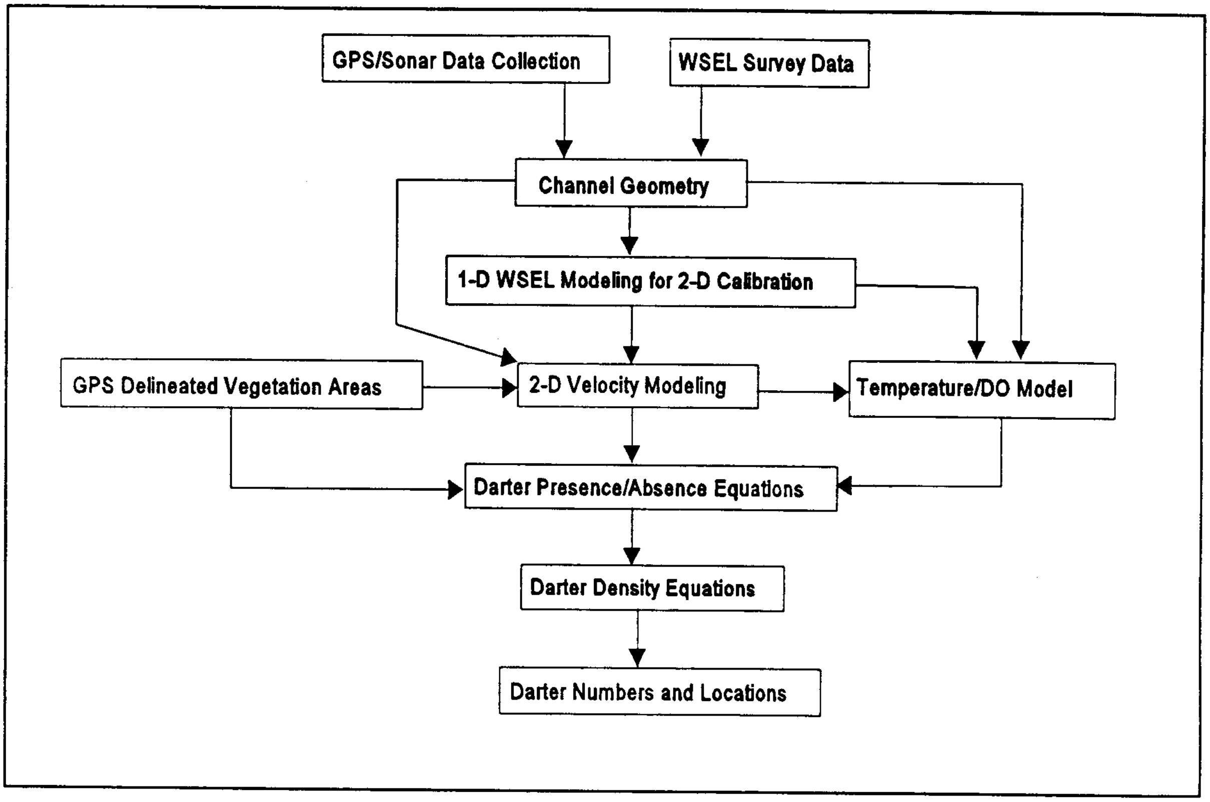


Figure 1. Model Flowchart

First, one area of current research focus is the application of 2-dimensional hydraulic models in place of the existing one-dimensional hydraulic simulation routines in PHABSIM (e.g. Leclerc et al., 1995). Historically, the application of one-dimensional hydraulic models required the simplifying assumption that water flows perpendicular to each cross section within the river channel. Unfortunately, complex channel geometries act to create velocity vectors that cannot often be modeled adequately in a 1-D simulation environment but can be represented by 2- or 3-dimensional hydraulic simulations. Use of these later models however, require accurate delineation of the spatial environment which in turn may allow more realistic multidirectional velocity components and thereby enable more sophisticated habitat evaluations. Traditional survey methods are standard practice in gathering 1-D cross section and river morphometry data. The ability and requirement of 2-D models for more spatially varied geometries creates the need for evaluation of new methods in data collection. Recent advancements in global positioning system (GPS) technology and the innovative use of dual frequency transducers in sonar applications have evolved to meet this need (Gubala et al., 1994). The use of GPS coupled with sonar allows for rapid, accurate and extremely intensive data collection and is one aspect highlighted in this paper. Second, historical applications of IFIM have relied on habitat modeling in PHABSIM which employ suitability index curves. These curves most often represent a univariate response gradient to depth, velocity and substrate based on fish observations. These curves are utilized to evaluate the simulated hydraulic properties at each discharge to derive and estimate of the quantity and quality of available habitat (i.e. Weighted Usable Area (WUA)). Recent research in habitat modeling approaches has emphasized the development and application of multi variate analyses of biological criteria for use in instream flow assessments (Orth, 1995; Bain, 1995). These approaches attempt to incorporate more variables which can account for critical factors in determining the location, quantity and quality of habitats or species locations for target organisms. In this study we utilize this approach to develop prediction equations for darter presence/absence as an alternative to more classical suitability index curve habitat modeling as used in PHABSIM.

Comal River System

The Comal River system is located in New Braunsfels, Texas, USA (Figure 2). Springs from the Edwards Aquifer originate along the northern and western boundary of the system and create Landa Lake where water exits to the east to form the Comal River. The mean discharge for the Comal springs from 1928 to 1972 has been estimated to range between 7.8 m^3/s to 8.3 m^3/s (Edwards Underground Water District; 1974, Buckner and Shelby, 1990). Declining spring flows have been tied to groundwater withdrawals within the recharge basin to meet municipal, industrial, and agricultural demands.

MATERIALS AND METHODS

Global Positioning System (GPS) and Hydroacoustic Data Collection System

A custom hydroacoustic set, coupled with differential global position system (DGPS) surveying equipment was designed and configured specifically for determining morphometries of small, shallow river systems, such as the Comal River and Landa Lake. A sonar signal processor controlled dual vertical beam transducers (200 kHz/5.5o and 28 kHz/35o), acquiring data at a ping rate of 6 times per second. The ASCII data stream transmitted from the hydroacoustic set was comprised of two identical and instantaneous streams from each transducer, including depth (recorded to the nearest 3 cm), bottom reflectivity, roughness and biomass indices (recorded to the nearest 1.0 decibel over a 0-120 scale) and a hexadecimal checksum for quality assurance (QA) purposes. The combination of these hydroacoustic parameters permitted the inference of bottom type along with detailed morphometry for hydraulic modeling purposes.

Differential Global Position System (DGPS) surveying technology was used to tag hydroacoustic data with horizontal positions. A pair of Trimble Pro-XLtm units, equipped with 4 MB RAM TDC-1 data loggers were used in this capacity. One unit, designated as a base, was located in close proximity (less than 1.5 kilometers) to the active rover. Maintaining a short distance between the base and rover enhanced precision by minimizing baseline drift. This paired DGPS system acquired and stored differentially corrected data at a rate of once per second. The data acquisition rate was matched precisely with that of the hydroacoustic set to eliminate interpolating bias. The raw data stream consisted of the full set of DGPS variables (UTM time, horizontal position, notes/attributes and precision estimates) along with the full suite of hydroacoustic data (depth, roughness, and hardness). River and lake levels did not vary enough to affect quality of depth measurements over the 6 days of data collection.

DGPS was also used in combination with other traditional forms of IFIM and surveying work to rectify all measurements to a single coordinate system. Known benchmarks and landmarks, as well as channel transect locations were tagged with DPGS positions and later imported into a GIS database for image processing and analyses. 95 percent of the data acquired in the field was determined to be within 0.75 m of actual locations on the ground. Approximately 4,000 data points collected simultaneously from the combined DGPS and hydroacoustic arrays and the supplemental rover data were subjected to a variety of analyses to assure quality and completeness. Approximately 2% of the raw data were censored during this QA check due to second echoes and transients. Verification and limited censorship of the raw data was conducted on site during the field data collection period.

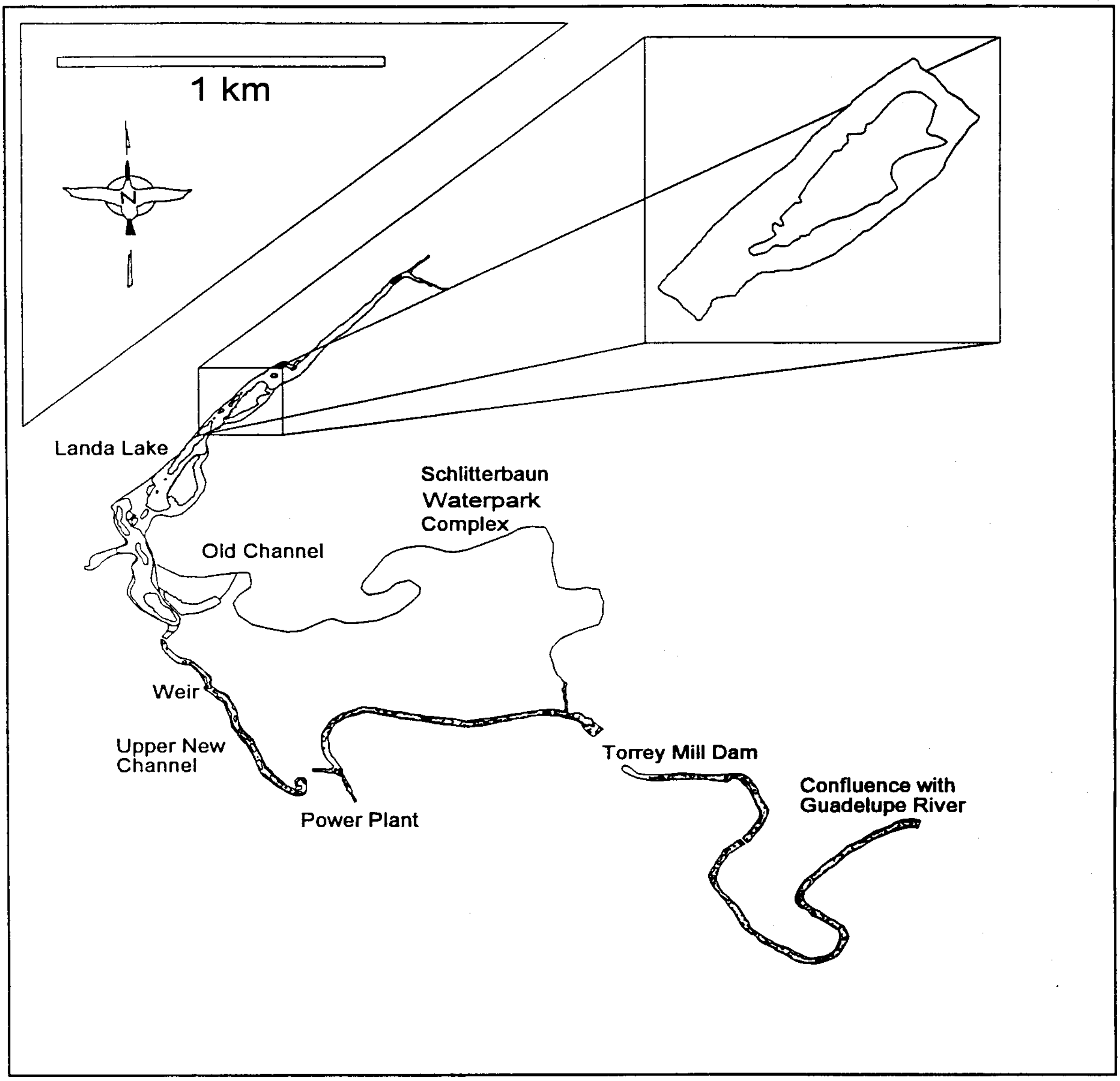
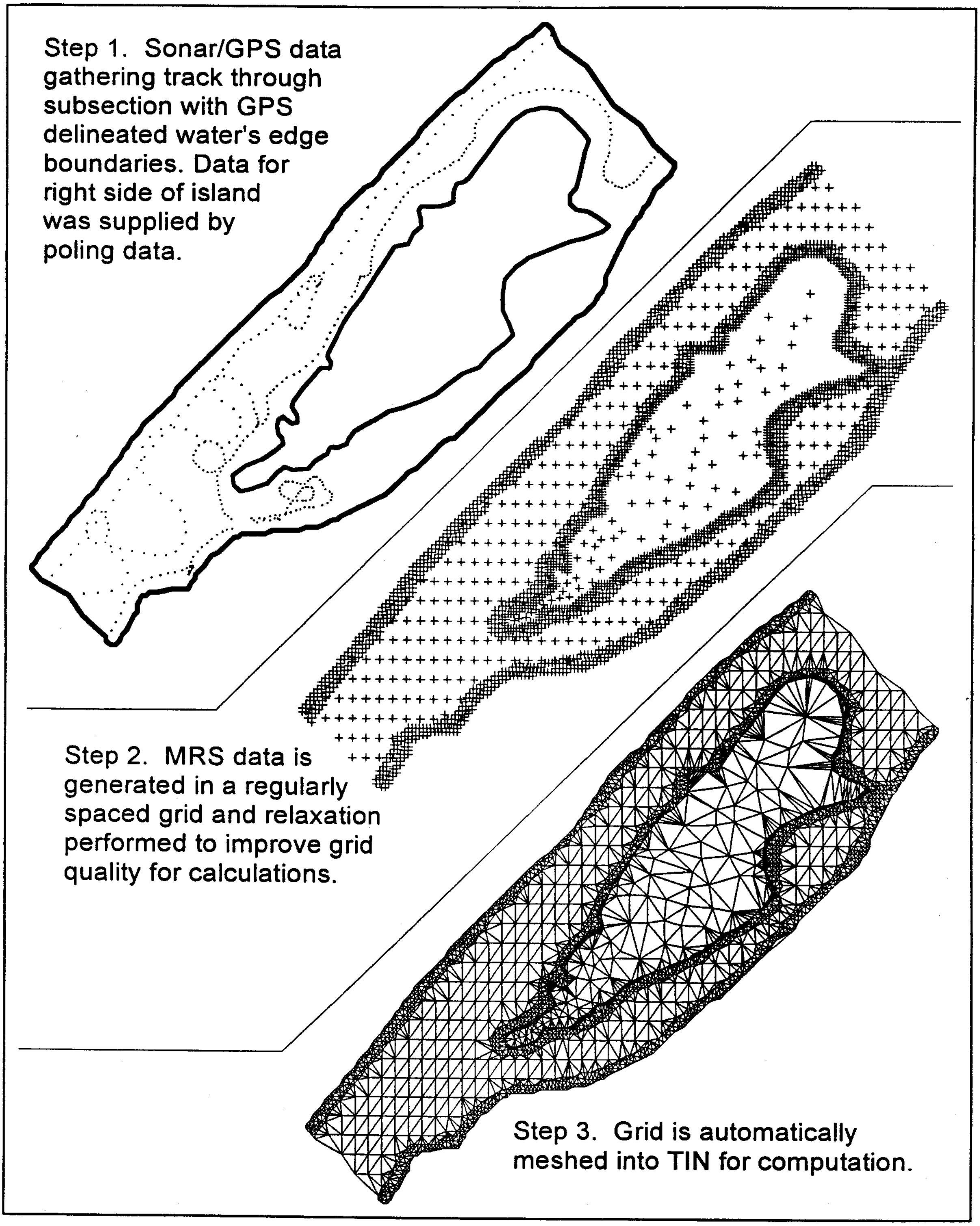


Figure 2: Site Layout and 2-D Modeling Subarea

Data Reduction and Analysis

Once an appropriate raw database was collected and verified, a bathymetric image was developed for display and digital import into CAD and hydrologic flow models. This process required the translation of the irregularly spaced raw data sets into regularly spaced finite difference or finite element grids, as shown in Figure 3. Interpolating algorithms, such as linear or quadratic krigging and cubic spline radial bias functions were used to accomplish this task. The general gridding procedure entailed an iterative application of appropriate methods and parameters until the most representative surface (MRS) had been created. The MRS is defined as the interpolated matrix which least deviates from the raw data. An estimate of the variance between the observed data and the mathematical representation of the surface was used as the



Step 1. Sonar/GPS data gathering track through subsection with GPS delineated water's edge boundaries. Data for right side of island was supplied by poling data.

Step 2. MRS data is generated in a regularly spaced grid and relaxation performed to improve grid quality for calculations.

Step 3. Grid is automatically meshed into TIN for computation.

Figure 3: 2-D TIN Generation Process

numeric metric for this analysis. The final MRS was rectified to a known elevation by combining the GPS/sonar horizontal locations and water depths with surveyed water surface elevations (WSEL's) throughout the system. The MRS was stored in a 0.9 x 0.9 meter format. The regularly spaced GPS/sonar depth data was then mated to the WSEL's obtained by traditional surveying methods. This created the topography of the lake bed for use in modeling other flowrates. Bank and island borders were recreated from GPS tracks walked by the researchers. Additional information such as distance to shore measurements were also recorded during GPS/sonar data collection. This data supplemented the GPS walked system boundaries. Bank and island elevations were assumed to be represented by the local WSEL plus a vertical offset.

1-D Modeling for 2-D Calibration

The 2-D tool used required an input beginning water surface elevation (WSEL) and flowrate (Q). Since WSEL-Q information only existed for one to two profiles, depending on the reach subsection in question, intermediate WSEL-Q information had to be modeled for use in the 2-D model. For this purpose, classical 1-D modeling, valid for WSEL's, was performed. The MRS showed its value in its use in the 1-D modeling when the tight 0.9 m spacing of the GPS/sonar data MRS was used for modeling cross sections (generated from the MRS) every 3 m along the channel. Thus, an accurate 1-D calibration tool for the 2-D model was able to use existing GPS/sonar generated morphometric data.

2-D Solution

The 0.9 x 0.9 m grid spacing of the MRS proved to be too data intensive for the 2-D tool being used due to computational memory requirements. The computer used was an SGI Indigo 2 workstation with 128 MEG RAM with scratch file space demands in excess of 270 Megabytes. To solve this problem, an alternative gridding procedure was employed. After the GPS/sonar data was converted into a regular grid, the regularly spaced data set was broken into three distinct data sets with grid node spacing ranging from the original 0.9 m up to a cubic spline smoothed topography of 3 m. A tight (1.2 x 1.2 m) river bank was used for purposes of boundary delineation along with a looser (2.9 x 2.9 m) river/lake bottom grid. The scale of the river and lake bathymetric changes was small enough to allow these grid spacings. This data set was used to create a triangulated irregular network (TIN) by automated computational methods. The 2-D model being used recommended a geometry in which no grid cell was greater than 50% larger than a bordering grid cell. Consequently, the TIN was 'relaxed', a process in which the grid cells are automatically resized and the grid node locations interpolated in order to create a more uniform cell size. The altered TIN was subsequently used for the solution of the following 2-D equations, resulting in 2-D water surface, depth and velocity profiles.

The 2-D model used solves the depth-integrated equations of fluid mass and momentum conservation in two horizontal directions (Hydraulics Laboratory, 1990):

$$\text{(1)}\quad \frac{\partial u}{\partial t}+u\frac{\partial u}{\partial x}+v\frac{\partial u}{\partial y}+g\frac{\partial h}{\partial x}+g\frac{\partial a_0}{\partial x}-\frac{\epsilon_{xx}}{\rho}\frac{\partial^2 u}{\partial x^2}-\frac{\epsilon_{xy}}{\rho}\frac{\partial^2 u}{\partial y^2}-2v\omega\sin\phi+\frac{gu}{C^2h}(u^2+v^2)^{\frac{1}{2}}-\frac{\epsilon V^2_a}{h}\cos\psi=0$$

$$\text{(2)}\quad \frac{\partial v}{\partial t}+u\frac{\partial v}{\partial x}+v\frac{\partial v}{\partial y}+g\frac{\partial h}{\partial y}+g\frac{\partial a_0}{\partial y}-\frac{\epsilon_{yx}}{\rho}\frac{\partial^2 v}{\partial x^2}-\frac{\epsilon_{yy}}{\rho}\frac{\partial^2 u}{\partial y^2}-2u\omega\sin\phi+\frac{gv}{C^2h}(u^2+v^2)^{\frac{1}{2}}-\frac{\epsilon V^2_a}{h}\sin\psi=0$$

$$\text{(3)}\quad \frac{\partial h}{\partial t}+\frac{\partial}{\partial x}(uh)+\frac{\partial}{\partial y}(vh)=0$$

where u = depth-integrated horizontal flow velocity in the x-direction,
t = time,
x = distance in the x-direction,
v = depth-integrated horizontal flow velocity in the y-direction,
y= distance in the y-direction (lateral),
g=acceleration due to gravity,
h = water depth,
a_0 = elevation of the bottom,
ϵ_{xx} = normal turbulent exchange coefficient in the x-direction,
ρ = fluid density,
ϵ_{xy} = tangential turbulent exchange coefficient in the x-direction,
ω = angular rate of earth's rotation,
ϕ = latitude,
C = Chezy roughness coefficient,
ξ = coefficient relating wind speed to stress exerted on the fluid,
Va = wind velocity,
ψ = angle between wind direction and x-axis,
ϵ_{yx} = tangential turbulent exchange coefficient in the y-direction, and
ϵ_{yy} = normal turbulent exchange coefficient in the y-direction.

Darter habitat was consequently evaluated based on bottom velocities according to the following equations (Chulick, 1995). The equation with the larger value determined the presence or absence of darters:

$$Absent = -1.7 - 2.2\ \log(velocity\ at\ 15\ cm\ above\ bottom) \tag{4}$$

$$Present = -3.4 - 4.3\ \log(velocity\ at\ 15\ cm\ above\ bottom) \tag{5}$$

Fisheries Collection and Analysis Methods

A stratified random sampling procedure was employed to sample 3 meter areas in uniform meso-scale habitat types using a drop net structure during the summer, fall, winter and spring of 1994. For each collection location, a complete depletion sample of all fish species was accomplished and fish type, numbers and lengths were recorded. At each location, vegetation type, height and aerial coverage, substrate, mean column velocity and velocity at 15 cm above the bottom, water temperature, conductivity, pH, and dissolved oxygen were recorded. Vegetation type, height, aerial coverage, and substrate were also noted for all adjacent 3 meter cell areas at each sample location. Macroinvertebrates and smaller fish were preserved in the field and identified in the lab. Subsequently, vegetation mapping of the entire Comal River was undertaken at 1 meter resolution using GPS survey equipment and rectified to the existing finite element grid described above. A total of 352 fish collection records were analyzed using a variety of parametric and non-parametric statistical procedures to develop statistically significant regression equations for predicting presence/absence of darters (Equations 4 and 5). In addition, data analyses generated prediction equations for density estimates based on an expanded set of cell attributes such as vegetation type, density, temperature, etc. However, given space limitations, only the presence/absence equations are evaluated in this paper and the density prediction equations are not considered. A complete description of the sampling procedures and analytical methods can be found in Chulick (1995).

RESULTS

The system wide depth contours based on the GPS linked hydroacoustic data are illustrated in Figure 2. A typical river section representing approximately 150 meters used in the hydraulic modeling from the middle portion of Landa Lake is shown in Figures 3 and 4. The use of 150 meter sections in the analyses was mandated by limitations of the array sizes of the finite element programs and corresponding computational burden as noted in the methods section. Corresponding 2-dimensional hydraulic simulation results for this same example area are shown in Figure 4 for flowrates of 3.5 and 6.4 m^3/s. Figure 4 also illustrates model results from the application of the multi variate prediction equations for the evaluation of darter presence/absence at the two different flow rates. The lower modeled flowrate caused some drying of the channel bottom along the right side (looking upstream) of the island which resulted in a loss of darter habitat in that area. Several deep pools did remain, however, which provided limited darter habitat according to the prediction equations. The higher 6.4 m^3/s flowrate allowed the water level to rise and fill the right channel, although velocities at the lower end of the channel proved too high for darters. Bottom velocities on the upper left side of the island proved too large for darters at both flowrates. The lower left area of the modeled subsection had low velocities and was deep enough to be darter habitat. The difference between the two flowrates' effects on darter habitat is particularly visible in this area. Vectors shown in Figure 4 indicate water velocity direction but not magnitude.

The results shown in Figures 3 and 4 clearly reveal that the high data density obtained from the hydroacoustic sampling procedure results in a well defined spatial geometry that is ideally suited for use of 2-dimensional hydraulic simulation of complex flow patterns over relatively small spatial domains. Furthermore, integration of these hydraulic model results with the multi variate species response equations indicates a high degree of spatial sensitivity of changes in the hydraulic regimes.

DISCUSSION

Study results clearly show the utility of linking GPS and hydroacoustics for obtaining high quality data over relatively large spatial domains. Data collection time was reduced from weeks to days while vastly improving lake and river system coverage at spatial resolutions on the order of 1 to 3 meters. This method of data collection was also shown to be ideal in terms of integrating vegetation mapping over the same spatial domain which incorporates a key element of the habitat requirements of darters. Limited field validation (currently in progress), has shown that the GPS/sonar data provided accurate spatial dependant channel geometries that permit linkages to both traditional 1-dimensional and the new 2-dimensional modeling tools in IFIM applications. Analysis software developed as part of this project permitted the extraction of 1-dimensional cross section profiles of the river channel at any arbitrary location for use in modeling the water surface profiles for input to the 2-dimensional hydraulic simulation routines and are compatible with existing hydraulic and habitat analyses of PHABSIM.

The high density spatial characterization of the river channel also results in improved spatial characterization of the hydraulic regime (i.e. velocities) based on the application of 2-dimensional hydraulic simulations. The lower modeled flowrate of 3.5 m^3/s resulted in velocity reduction and some drying out of the darter habitat areas, as reflected in the 2-D solutions. Consequently, darter habitat changes between the two flowrates. Water's edge boundaries were produced on a 1 - 3 meter scale, which aided in small scale delineation of darter habitat. Incorporation of submerged spring orifices (not illustrated) produced realistic velocity nets and is potentially critical in modeling these areas within Landa Lake for the habitat needs of salamanders which are spatially tied to these locations. The extension of the hydraulic modeling to 2-dimensional flow patterns is considered critical in terms of accounting for the complex flow patterns associated with spatial variations in the channel characteristics.

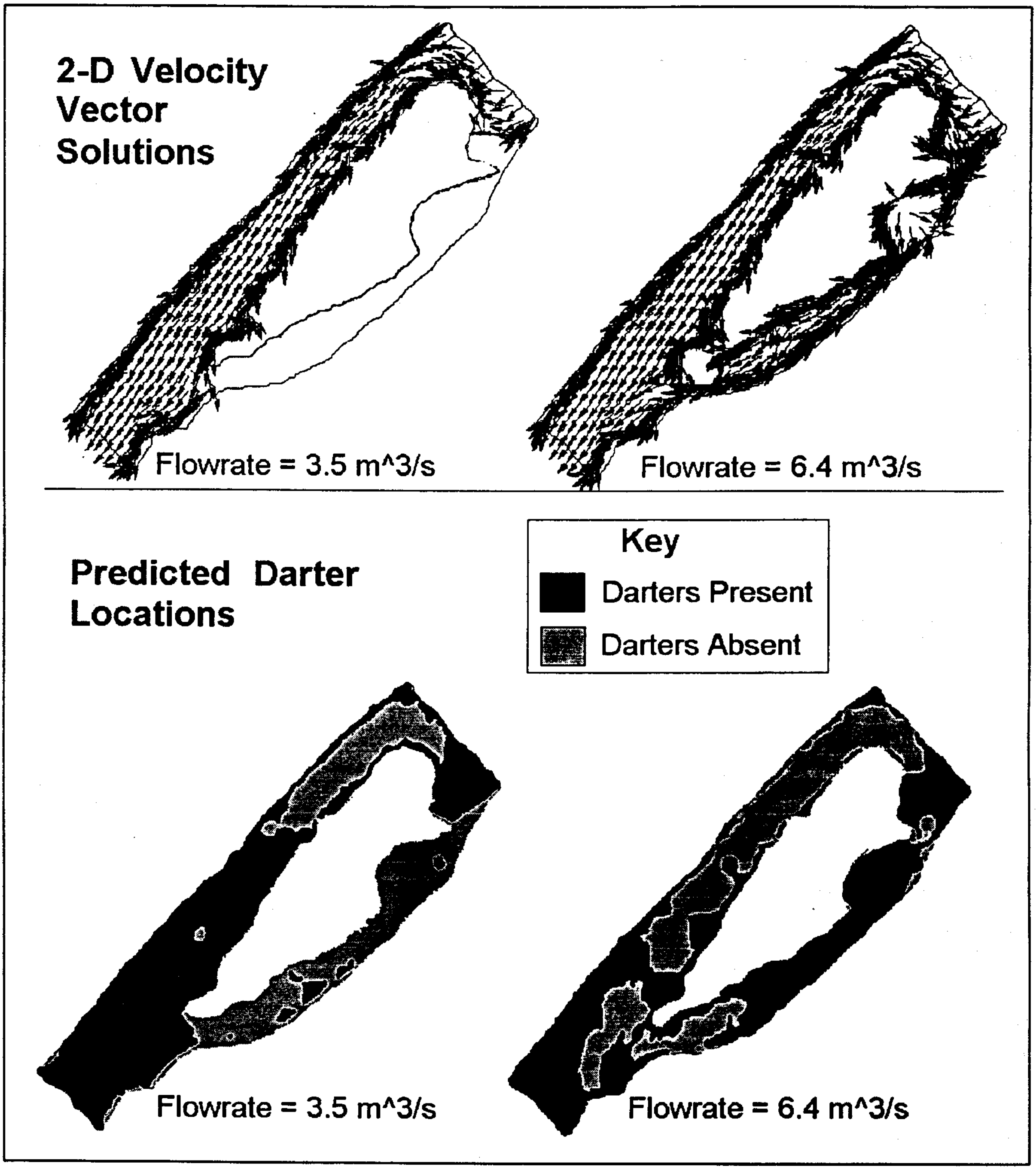


Figure 4. 2-D Velocity Vector Solution and Darter Habitat Prediction

The application of the multi variate derived darter presence/absence and density equations also showed a high degree of spatial sensitivity under simulated discharge conditions. The sensitivity of these predictions is directly related to the high density delineation of the spatial grid system and use of the resultant 2-dimensional hydraulic model outputs. Field validation for both the hydraulic simulation of velocities and predictions for both presence/absence and density of darters is currently targeted for the summer of 1996 during projected low spring flows due to drought and ground water abstractions. Another important aspect of this project was the demonstration of the linkages of these data acquisition and analysis tools within the existing IFIM framework based on readily available cost competitive equipment and commercially available software systems. This should provide encouragement for other investigators and consultants to consider the application of these types of tools and modeling procedures in other instream flow applications.

CONCLUSION

The IFIM provided and excellent framework for the development and application of innovative data acquisition and analysis tools to model the flow dependant characteristics of the Comal River for Fountain darters. The use of the GPS/sonar methods for gathering topographical data for river systems was demonstrated to result in a quick, efficient, and accurate data collection strategy. The use of 2-dimensional hydraulic modeling of velocities based on the resulting finite element grid is considered a substantial improvement over existing 1-dimensional velocity simulation methods and is directly related to the intensive spatial data derived in project. The GPS/sonar data collection method can be used for traditional 1-dimensional modeling, resulting in spacing of cross sections that is dramatically improved over traditional survey methods. Incorporation of a multi variate statistical model for prediction of darter presence/absence within the IFIM framework was illustrated and was demonstrated to have a high degree of sensitivity spatially to changes in the simulated hydraulic characteristics associated with changes in discharge. This is again attributed to the linkage between the spatially explicit delineation of the channel geometries and use of 2-dimensional hydraulic simulations. Reduction in the accuracy of the spatial delineations of the channel geometries is anticipated to reduce the utility of use of 2-dimensional hydraulic simulations since these models are highly dependant of the characterization of the spatial domain. The more complex the channel characteristics, the more intensive the data requirements are likely to be to accurately simulate the hydraulic characteristics using this class of models. Acquisition of the type and extent of spatially explicit channel characteristics and associated model output from 2-dimensional hydraulic simulations also provide expanded access to more innovative use of spatially explicit habitat metrics derived from landscape ecology (see Bovee; Hardy; these proceedings).

REFERENCES

Bain, M.B. (1995). Habitat at the local scale: Multivariate patterns for stream fishes. *Bulletin Français del la Peche et de la Pisciculture.* 337/338/339 pp. 165-177.

Bovee, K.D. (1995). A comprehensive overview of the instream flow incremental methodology. National Biological Service, Fort Collins, CO., 322pp.

Buckner, H.D. and W.J. Shelby. (1990). Water Resources Data for Texas, Water Year 1990, Volume 3: Colorado River Basin, Lavaca River Basin, Guadalupe River Basin, Nueces River Basin, Rio Grande Basin and intervening coastal basins, Water Data Report TX-9003. U.S. Geological Survey, Austin, TX.

Chulick, M.E. (1995) Statistical modelling methodology for the determination of habitat suitability and habitat preferences of the endangered fountain darter. Plan B Thesis. Utah State University, Logan, UT, USA 84341

Edwards Underground Water District. (1974). The Edwards Bulletin.

Gubala, C. P., C. Branch, N. Roundy and D. Landers (1994). Automated global positioning system charting of environmental attributes: a limnologic case study. *The Science of the Total Environment.* 148 pp. 83-92

Guyton, W.F. and Associates. (1979). Geohydrology of Comal, San Marcos and Hueco Springs, Report 234, Texas Department of Water Resources, Austin, Texas.

Hydraulics Laboratory. (1990). Gereralized Computer Program System for pen Channel Flow and Sedimentation, TABS System, Volume 1: General Overview, Department of the Army Waterways Experiment Stations, Corps of Engineers, Vicksburg, MS, USA

Leclerc, M., A. Boudreault, J.A. Bechara and G. Corfa (1995). Two-dimensional hydrodynamic modeling: A neglected tool in the instream flow incremental methodology. *Transactions of the American Fisheries Society.* 124(5) pp. 645-662.

Milhous, R.T., M.A. Updike and D.M Schneider (1989). Physical Habitat Simulation System Reference Manual - Version II, Instream Flow Information Paper No. 26. Biological Report 89(16), U.S. Dept. of the Interior, Fish and Wildlife Service, Research and Develoment, Washington, DC, USA 20240.

Orth, D.J. (1995). Food web influences on fish population responses to instream flow. *Bulletin Français del la Peche et de la Pisciculture.* 337/338/339 pp. 317-328.

METHODS FOR DESCRIBING IN-CHANNEL HABITAT FEATURES: ENGINEERING PROCESSES TO FRACTAL GEOMETRY CONCEPTS

Virginia K. Sutton
Baylor University
Department of Information Systems
Waco, Texas 76798
United States of America
Virginia_Sutton@baylor.edu

John Nestler
USAE Waterways Experiment Station
3909 Halls Ferry Road
Vicksburg, Mississippi 39180
United States of America
Nestlej@ex1.wes.army.mil

ABSTRACT

Quantitative description of spatial habitat patterns is often at the heart of ecological research in aquatic systems, especially for analyses of physical habitat. A common approach for describing physical habitat is tessellation, or the discretization of the channel into cells of approximately uniform size, and assigning each cell an average value for velocity and other characteristics. More innovative methods may use spectral analysis or fractal dimension to characterize underlying patterns in spatially complex geological features such as channel bed forms. Unfortunately, all of these methods lose information because they either force continuous data into a grid framework, assume that it has the form of a finite series of harmonic functions, or assume that complexity is constant over a range of scales. Understanding aquatic processes would improve if information about the scale of features in a channel could be preserved throughout the analysis instead of being discarded through simplifying assumptions as the first step in habitat analysis. We present methods, based on fractal geometry, that characterize complex spatial data sets with minimal use of assumptions or simplifying approximations. Our approach identifies dominant features in a set of coordinate data, locates the positions of such channel features in the cross section, and quantifies how features of different scales relate to one other. We apply this method to mathematical constructs having known characteristics, such as Cantor sets arrayed in two dimensions, to verify the efficacy and rigor of our approach. We use this method to describe the scale of dominant features in Missouri River cross sections and compare cross sections before and after regulation to demonstrate how a fractal geometry analysis using our methods can quantify changes in physical habitat patterns. Conventional methods are not able to describe theses changes since they cannot identify significant scales or how features of different scales are organized to compose the channel bed form. Improved description of complex shapes should lead to increased understanding of aquatic processes in general, and in particular, the way aquatic organisms relate to physical habitat.

KEY-WORDS: Fractal/ Mandlebrot/ Aquatic/ Scale/ Habitat/ Cantor Set/ Sine wave/ Pattern/ Spatial/ Andrle

INTRODUCTION

Milne (1991) eloquently states that "landscape structure affects the spread and disturbance and regulates the movements of resources, organisms, and energy." However, spatial relationships in aquatic habitat studies are usually based on engineering approximations or methods having restrictive assumptions that inadequately describe the spatial structure within river channels. For example, a finite element approach of discretizing spatial data into cells, while very useful for quantifying bulk flow or material transport, misses information that exists at scales smaller than the cell size. Consequently, it is impossible to quantify physical habitat across the range of scales necessary to describe habitat utilization for different communities in aquatic systems or to completely describe processes that may occur across a range of scales. Large individuals, such as adult salmon, respond to features larger in scale than do caddisflies. Similarly, the scales at which organic matter is stored and transported through an aquatic system may be quite different from the scales required to predict water surface elevations. Relatively new methods to characterize complex geologic features also have restrictive assumptions. Spectral analysis requires that the underlying spatial structures can be described using a finite series of harmonic functions, and the fractal dimension assumes that spatial complexity is constant across a broad range of scales. Figure 1 illustrates spatial information loss during discretization of a Cantor "cross section" (described later) as an example of how information is lost during spatial approximation.

We present a modified method, based on fractal geometry, that characterizes spatial data with minimal use of preconceived assumptions or simplifying approximations. The method incorporates and builds on two important methods of analyzing data, the fractal dimension (Mandlebrot, 1975)and the angle measure technique - AMT (Andrle, 1994). Our approach identifies dominant features in a set of coordinate data, locates the positions of such channel features in the cross section, and quantifies how features of different scales relate to one another. We apply this method to mathematical constructs having known characteristics, such as Cantor Sets arrayed in two dimensions, to verify the efficacy and rigor of our approach. We use this method to describe the scale of dominant features in Missouri River cross sections and compare cross sections before and after regulation to demonstrate how a fractal geometry based analysis using our methods can quantify changes in physical habitat patterns.

SPATIAL ASSESSMENT USING FRACTAL GEOMETRY

An understanding of our method for describing spatial pattern requires a brief discussion of fractal geometry. Complex shapes can be classified into three categories: pure fractals, statistical fractals, and multifractals. A pure, or mathematical, fractal object is constructed by repeating the same shape at increasingly smaller scales. An example of a pure fractal is the Cantor Set (Figure 2). This set has a repeated pattern of removing the middle third of the current line segment that completely determines its complex shape. Few naturally occurring geologic features can be characterized as pure fractals. However, many geologic features and biotic structures can be categorized as statistical fractals and multifractals. A statistical fractal has portions that, upon magnification, look similar, but not *exactly* like a larger portion of the object. For example, a tree appears to exhibit the same pattern of stem and branches at different scales although the pattern is not *identical* over all scales. In contrast, a multifractal feature is characterized by different spatial patterns across the complete range of scales. Multifractal models occur commonly in the earth sciences because geologic data are often characterized by limited spatial correlation. River cross sections are later shown to be multifractal geologic features, probably because erosional and depositional processes that form river channels are not distributed uniformly laterally or longitudinally.

Mathematical and statistical fractals both have a fractal dimension, D_f, which is less than the topological dimension of the object. The fractal dimension identifies the rate at which the complexity of the object changes with scale of observation. Mandlebrot and Richardson developed the M-R plot as a simple means of identifying D_f for a particular object. In its simplest form, an M-R plot is created by measuring the outline of a geologic feature or other shape with a series of rulers of different lengths and plotting the number of times a ruler fits by the ruler's size. When plotted on log paper, the data determines D_f as the negative slope of the best fit line plus 1 (Middleton, 1991). For a mathematical fractal, D_f holds over *all* possible scales. D_f applies to a statistical fractal over a wide, but not infinite, range of scales, and a multifractal object has several fractal dimensions over its range of scales. The M-R plot is easily applied to pure fractals such as the Cantor Set which has $D_f \approx 0.63$ (Figure 2) or statistical fractals such as the coastline of Great Britain which has $D_f \approx 1.52$. For multifractals, the M-R plot is generated over the entire range of scales to identify break points which mark changes in the fractal dimension. These breakpoints determine the subsets of scales for which individual M-R plots can be made.

The fractal dimension, D_f, is a useful metric to describe the general spatial complexity of a fractal object. However, it does not indicate the manner in which features of different scales are related to one another, nor does it provide any information on the location of features of different scales. One encounters difficulty in using algorithms dedicated to calculating fractal dimensions to identify dominant or characteristic scales. Consequently the fractal dimension by itself cannot provide sufficient information to serve as a framework for describing or understanding aquatic processes. In addition, river cross sections are generally multifractal features, so that a single fractal dimension may be inadequate to describe spatial patterns in river cross sections. We provide enhancements to the M-R plot to improve its ability to depict aquatic spatial patterns and demonstrate the use of the Angle Measurement Technique (AMT) as a tool to provide supplemental information to the M-R plot for aquatic spatial analyses.

An alternative method to performing a fractal analysis that lacks some restrictive assumptions of a simple fractal dimension is the AMT of Andrle (1994). Unlike the fractal dimension, the AMT assumes that geological structures may have dominant features that can be described over one scale or a limited range of scales. For example, one of the dominant scales in a gravel bed stream would be approximated by the average diameter of the gravel substrate. The inputs into AMT are the coordinates of a curve digitized from a map and the scales, S, used to calculate the angles. A typical AMT analysis involves the following steps. First the program randomly chooses an initial starting point, A, from which it finds the two digitized points, B and C, that are S distance from A. Let $\theta = (180^\circ - \angle BAC)$, or θ is the supplementary angle to BAC (Figure 3). This process is repeated 500 times for each scale, and the mean angle is calculated for each scale, S. The mean angle and S values are written to a file from which one can graph mean angle by log(S). Significant scales are identified as peaks in the mean angle by log(Scale) plot. Andrle contends that the more complex a curve is, the greater the mean angle measure because the mean angle is a measure of how much the path of a scale along the digitized curve differs from a straight line. Thus, a characteristic or dominant scale of the curve would have the greatest mean angle measure. Significant scales are identified by the smaller peaks in the mean angle plot. Lack of discernible dominant or characteristic scales indicates that the curve is a statistical fractal.

Computation of the M-R plot can be improved by using multiple, randomly selected points to reduce the bias of starting at the leftmost point. Typically we measure forward from a randomly selected starting point and store the number of fits and then reposition the scale at the same randomly chosen starting point and measure backwards to the left of the curve and add the number of fits in this direction to those that had been stored. For this same scale, we repeat this process with 9 more randomly chosen starting points. We determine which of these 10 starting points is best for this scale by comparing the scaleinduced error in measuring the curve from that point.

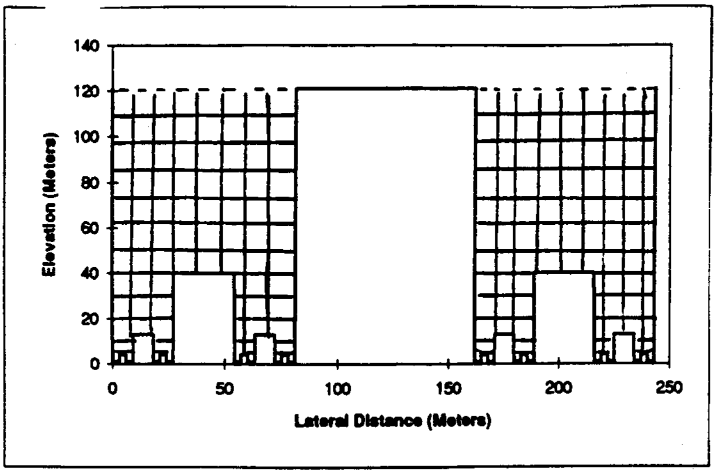


Figure 1: Cantor Section 1 with grid

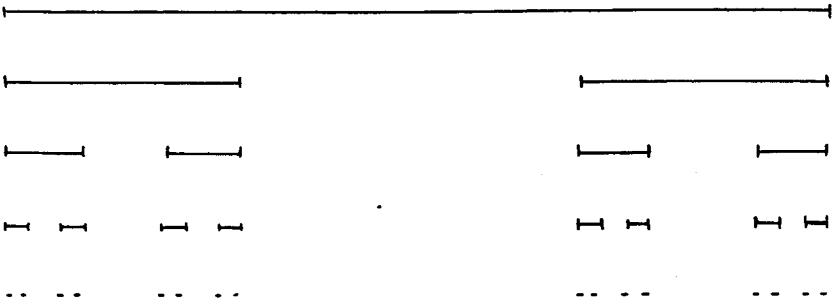


Figure 2: Cantor Set

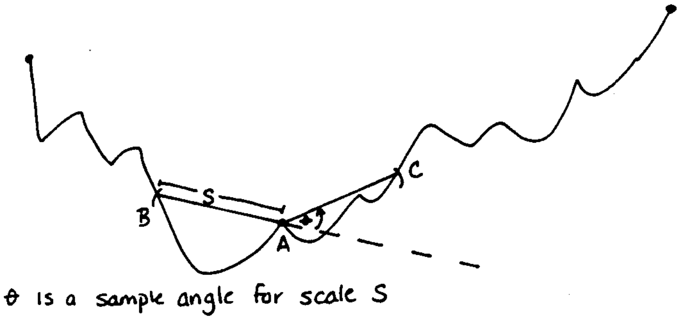


Figure 3: Supplementary angle using AMT

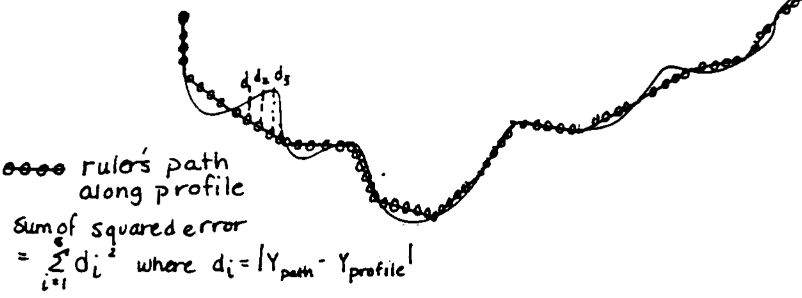


Figure 4: Sum of squared error in measuring algorithm

We define error much like mean squared error in least squares linear regression, i.e., the sum of squared vertical difference between the path the scale makes and the curve (Figure 4). This sum is divided by the number of fits that the scale made across the curve from that position.

A similar error term is used to associate dominant or characteristic scales as identified by the AMT analysis with horizontal position along the curve. We arbitrarily consider a scale to "exist" along a horizontal portion of the curve if its sum of squared error in that intersection is less than 1/3 its length. Existence plots indicate the presence of specific user-defined scales with a horizontal line. Gaps in the line indicate that the scale does not "exist" at that lateral position in the curve. Once the horizontal positions of different scales are identified, then the nesting pattern (how different patterns are imbedded within each other) can be described.

We also improve the AMT by determining the relative vertical and horizontal components associated with the mean angle. For each angle BAC, the supplementary angle provides a measure of the change in its path that a scale must make to fit the curve, The mean horizontal and vertical values result from breaking vector **AC** into its component vectors. We determine the overall significant scales in the curve as well as the scales that are significant in the horizontal and vertical directions. We will illustrate the utility of our approach in three cases--pure fractal, non-fractal, and multifractal.

APPLICATIONS

A Cantor cross section is constructed by taking a line of length 243 units and dividing it into three equal segments. The middle segment is raised to height determined by (1), where k is the current iteration, and vertical lines of length determined by (1) join the three segments together. This process is repeated 5 times and the resulting cross section is shown in Figure 5.

$$\sum_{i=1}^{6-k} 3^{i} \tag{1}$$

We performed a sensitivity analysis by switching the order of the raised segment to the first (Cantor Section 2 - Figure 6) and third positions (Cantor Section 3 - Figure 7). We also analyzed sine waves with different amplitudes but equal period of 50 units to document the performance of the methods on simple, non-fractal shapes . We analyzed river cross sections to illustrate the value of a fractal analysis for evaluating spatial impacts of river regulation. These cross sections depict the Missouri River at river kilometer 1263.3 at two discharges (170 and 1416 cms) and under regulated and unregulated conditions (Figures 9-12).

RESULTS AND DISCUSSION

Cantor Sections

Table 1 summarizes our analyses of these sections. The variables provided are fractal dimension, critical scale, mean angle, significant scales, horizontal change, and vertical change. Mean angle is a measure of complexity of the curve and is the mean angle associated with the critical scale. Significant scales are the scales associated with lesser peaks on the AMT plot. Horizontal change is the average change in the horizontal direction that the critical scale makes in fitting the curve, and vertical change is the average change in the vertical direction of the critical scale. The fractal dimension is inadequate to differentiate among these three different curves because they differ only in the order of the features and not in the size of the features. However, we distinguish between the section with the central feature and the sections with the large feature at the end through the mean angle and significant scales and inspection of the existence plots.

The AMT is able to provide information to quantitatively distinguish among the different Cantor sections. Note that the mean angle is much greater for section 1 than sections 2 and 3 which have the same mean angle. Section 1 has the largest feature in the middle, so the horizontal distance that scales can spread over is much shorter than in Sections 2 and 3. Thus the supplementary angle for these sections is smaller. The critical scales are the same for Sections 2 and 3 because the patterning of the features is the same for these sections. Scales of approximate length 4, 11, and 36 are common to all three, and these are the heights of the shortest and most frequent features. For Section 1, 119 is significant because it is the height of the central feature which determines the depth of both major voids. Scale 119 is not significant in Sections 2 and 3 because it can fit twice in the larger main void. In sections 2 and 3, a scale of 60 is significant because it can hit exactly at the midpoint of the second largest feature (Figure 14). Scales of length near 108 are significant because they mark the end of the two adjacent features having a height of 40 units. A scale of this size will have to go in almost a strictly vertical direction to continue to follow the curve from point (108,40). In all the sections, a scale must change more in the vertical direction rather than the horizontal, but Sections 2 and 3 allow more change in the horizontal direction than section 1 does. Sections 2 and 3 have a wider main void than section 1 does, so this follows. Note that the nesting pattern of these sections which makes them distinct is evident from the existence plots (Figures 14-16).

Table 1: Comparison of Cantor Sections

	Cantor section 1	Cantor Section 2	Cantor Section 3
Fractal Dimension	1.25	1.29	1.28
Critical Scale	116	36	38
Mean Angle	134.13	118.28	119.36
Significant Scales	1, 4, 13, 37, 119	4, 11, 36, 60, 103, 167	4, 12, 38, 60, 103, 168
Horizontal Change	.38	.41	.40
Vertical Change	.89	.85	.86

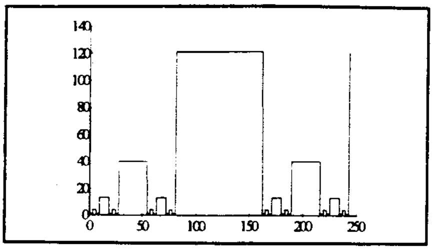


Figure 5: Cantor Section 1

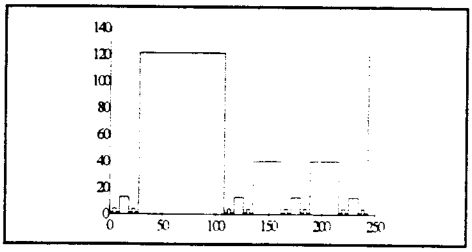


Figure 6: Cantor Section 2

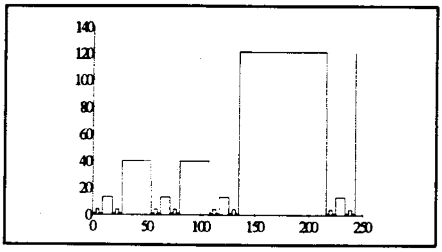


Figure 7: Cantor Section 3

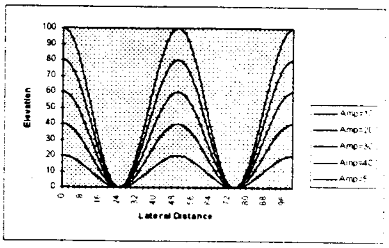


Figure 8: A portion of the sine waves used for analysis

The utility of these methods to describe spatial patterns is reinforced by our analysis of a family of sine waves (Figure 8 and Table 2). The greatest fractal dimension is for the sine wave that is approximately half as deep as it is wide. The dimension is small for those waves that are virtually vertical, such as the one with depth of 1600 units or virtually horizontal, such as the one with depth of 20 units because the fractal dimension measures how much the curve is filling the space in which it resides and not by its excursion. Analysis of the waves also identifies a problem associated with the use of the fractal dimension. The perimeter-ruler relationship used to calculate the fractal dimensions is similar to the error reduction obtained by approximating the perimeter of a circle by a series of polygons that increase in their number of sides (e.g. triangle, square, pentagon, etc.). A plot of perimeter-ruler relationships based on these polygons will generate an M-R plot with a fractal dimension of 1.0 even though the circle is not a fractal object. While this answer is correct in a strictly mathematical sense, it also indicates that the M-R plot not only describes the behavior of the line but also includes perimeter-ruler relationships because the integral of a portion of the M-R line provides an estimate if the area of the polygon for that scale. Consequently, the M-R plot confounds perimeter-area information along with perimeter-ruler information. The "straight-line" behavior of the M-R plot is partially determined by area-perimeter relationships. The AMT deals only with the excursion of the perimeter line and is not directly influenced by perimeter-area relationships and consequently provides a much more sensitive indicator of spatial pattern, Note that the AMT provides considerable detailed information about each of the sine waves. The critical scale corresponds to the wave height which is twice the amplitude. The amplitude length marked a plateau point for the AMT curve for the deeper sine waves because scales larger than the amplitude can fit only once on a decreasing portion of the wave which leads to high supplementary angles. As the curves became more peaked the mean angle increases. Also, the percent change of a ruler in the horizontal direction decreases as amplitude increases.

Table 2: Comparison of Sine Waves

Amplitude	Fractal Dim	Critical Scale	Mean Angle	Horizontal	Vertical
10	1.06	23	5.52	.81	.54
20	1.15	25	102.06	.60	.74
30	1.17	50	130.73	.39	.91
40	1.12	76	145.57	.29	.95
50	1.08	100	158.38	.23	.97
100	1.05	196	166.57	.12	.99
200	1.03	376	173.22	.06	1.0
400	1.02	758	176.61	.03	1.0
800	1.01	1476	178.33	.02	1.0

We present a partial analysis of Missouri River cross section data as an example of how fractal geometry techniques can be employed to gain an understanding of spatial patterns in aquatic systems and how river regulation can impact these patterns. A more complete analysis of changes in the Missouri River can be found in Latka et al. (1994) and Nestler et al. (1995). One location on the Missouri River was gaged (Table 3) under unregulated conditions at 170 cms (W06H) and 1416 cms (W50H) and regulated (1992) conditions at 170 cms (W06E) and 1416 cms (W50E). At the lower discharges, those portions of the cross section that are above water were eliminated for both the plots and the analysis. Regulation has considerably altered the spatial pattern of the Missouri River. First, the present river at low flow has considerably greater depth than under historical flow conditions(compare water surface reference lines in Figure 9 and 10). The reasons for this pattern are unknown but speculations are provided in Latka et al. (1994). The historical conditions are characterized by the presence of more dominant scales in both 170 and 1416 cms, and more incised bottom relief as indicated by increased mean angle between existing and historical conditions. A probable conjecture is that river regulation, in the form of interrupting the downstream transport of sediment has resulted in a simpler bed form in which vertical relief and features of certain scales have been eliminated or modified. Quantifying these changes provides a template upon which a wide array of biotic and physical factors can be explored and explained.

As a first step in applying fractal concepts to better understand aquatic processes, we characterize each cross section condition by scale and discharge (Figures 17-20). Each plot is obtained by summing the cell-by-cell discharges associated with each dominant or characteristic scale as identified in Figure 13-16. Clearly, this relationship is cumulative, since a large scale can include the discharge of smaller scales. The comparison of high flow conditions between historical and existing conditions provides considerable insight into the effects of river regulation. The degradation of the existing channel, probably caused by interruption of sediment transport by upstream dams, has resulted in the loss of significant overbank flooding. Consequently, the historical condition presents a diverse range of nested scales that convey considerable discharge (Figure 17) whereas the existing high flow conditions restrict flow to either the main channel or relatively small features as the water surface elevation rises just enough to cover some overbank areas (Figure 18). As a.result, existing high flow conditions are characterized by flow conditions that are generally concentrated in either numerous small scale features or a few larger features. Many more insights result from this analysis and will be detailed in future papers. We have included here only the most salient points as the purpose of this paper is to demonstrate the efficacy of our tool.

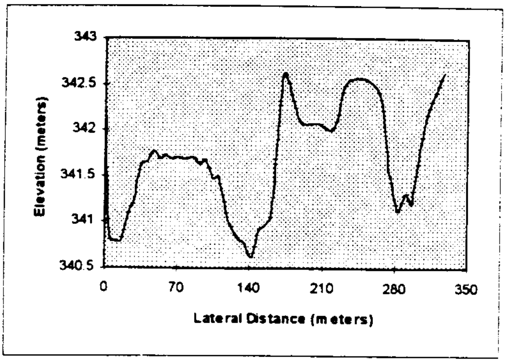


Figure 9: W06E transect
Missouri River kilometer 1263.3

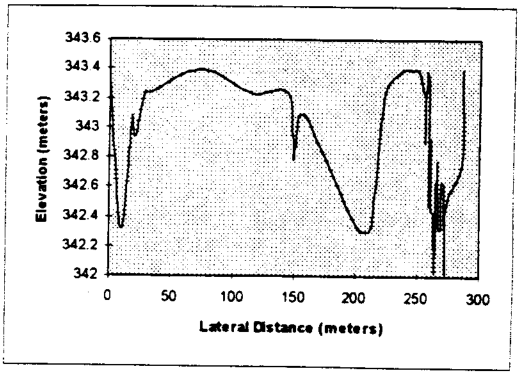


Figure 10: W06H transect
Missouri River kilometer 1263.3

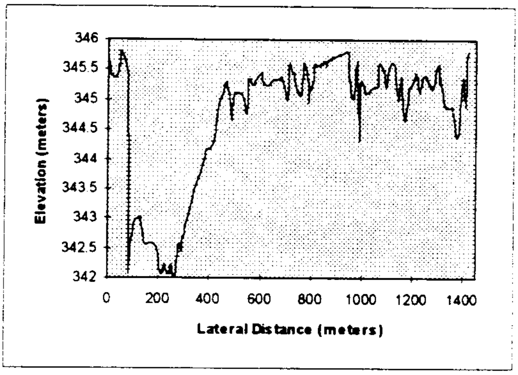


Figure 11: W50E transect
Missouri River kilometer 1263.3

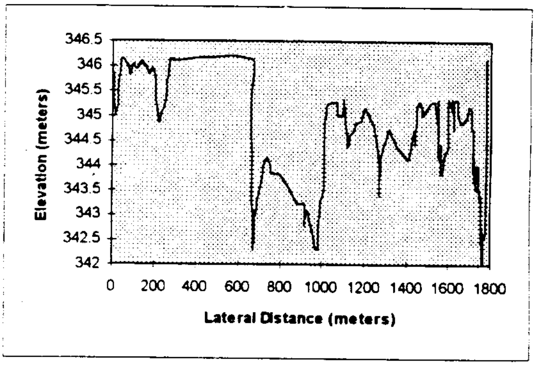


Figure 12: W50H transect
Missouri River kilometer 1263.3

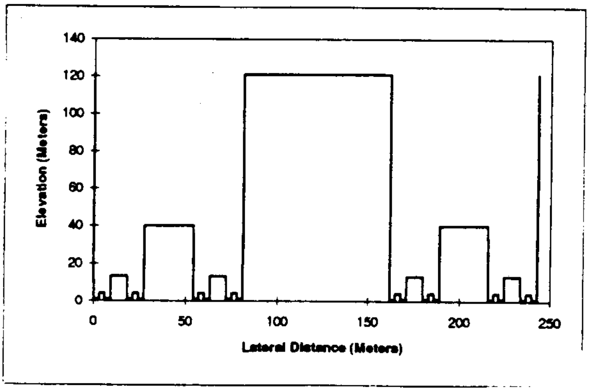


Figure 13: Cantor Section 1 with critical scale illustrated

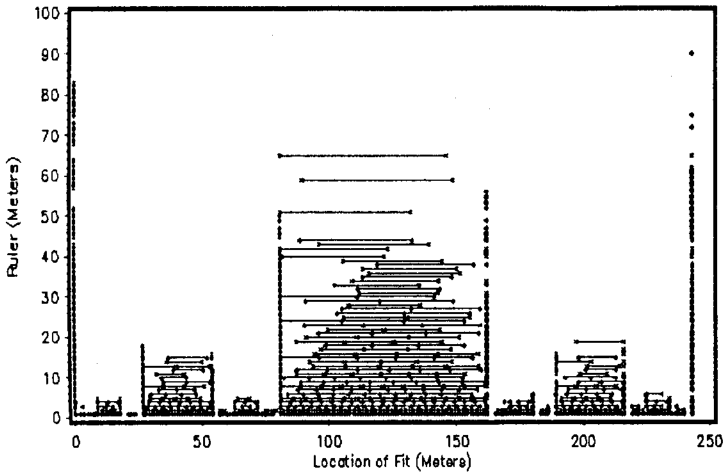


Figure 14: Cantor Section 1 Existence Plot

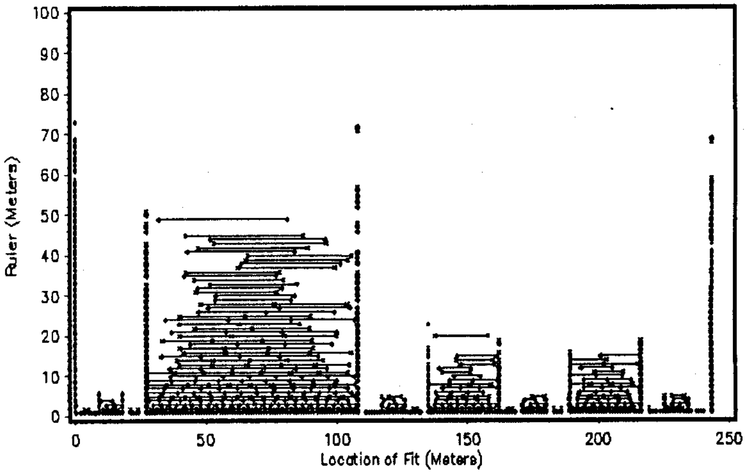


Figure 15: Cantor Section 2 Existence Plot

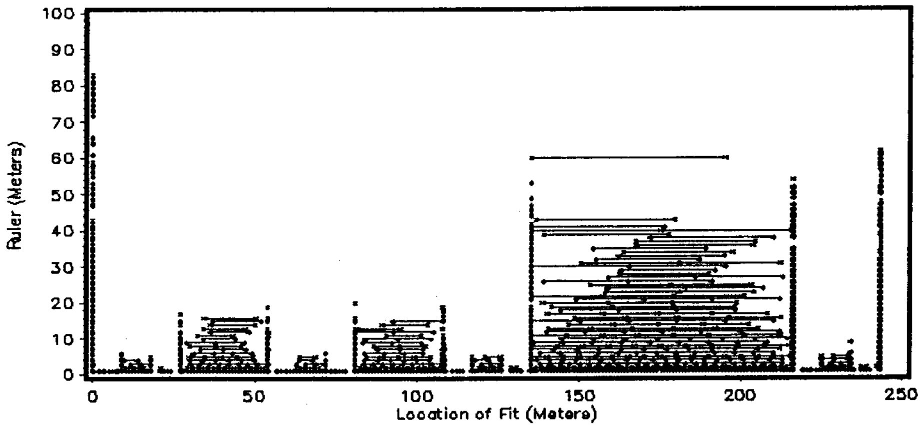


Figure 16: Cantor Section 3 Existence Plot

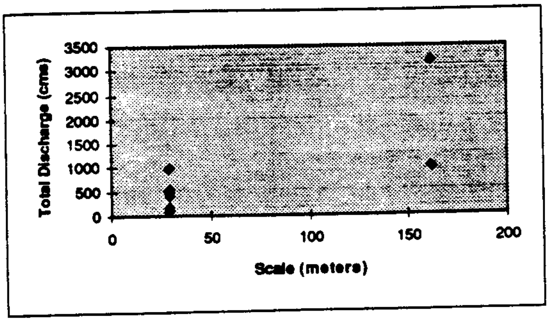


Figure 17: Discharge plot for W06E

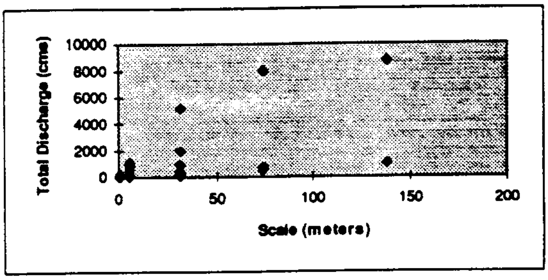


Figure 18: Discharge plot for W06H

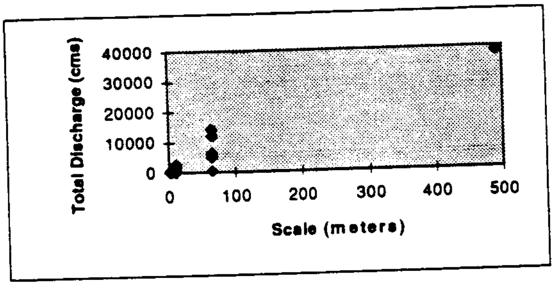


Figure 19: Discharge plot for W50E

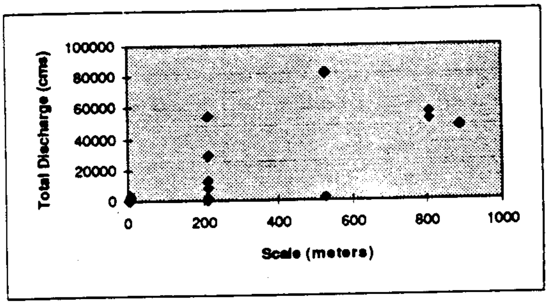


Figure 20: Discharge plot for W50H

Table 3: Comparison of Missouri River Transects

	W06H	W06E	W50H	W50E
Fractal Dim	1.00	1.00	1.00	1.00
Critical Scale	1	29	2	10
Mean Angle	4.42	1.92	2.31	1.46
Horizontal Change	.01	1.0	.99	1.0
Vertical Change	1.0	.02	.05	.02
Significant Scales	1, 6, 32, 74, 138	29, 162	2, 8, 212, 527, 805, 887	2, 10, 65, 490

CONCLUSIONS

We present a selection of fractal geometry tools that can be applied to better understand aquatic processes. These tools are shown to accurately describe spatial patterns in sensitivity studies employing Cantor sections and a family of sine waves. Cursory application of these tools to river cross sections demonstrate their ability to detect and characterize bed form changes that are difficult to detect using more conventional methods. These methods can probably be applied to a variety of aquatic processes to improve both our understanding and management of these systems. We will conduct studies in the future to apply these concepts to achieve greater comprehension of aquatic processes. Major parts of the method presented herein have been included in two patent applications submitted to the U.S. Patent Office.

ACKNOWLEDGEMENTS

The authors gratefully thank the Corps of Engineers Omaha District and Missouri River Division for providing the historical data that was employed in this analysis. The authors thank Ellen Czaiaka, Dr. Roger Glick, Carla LeFlore, Jace Pugh and Toni Schneider of Waterways Experiment Station for graphical and technical support of this project.

The authors would like to especially thank Mr. Doug Latka of the Missouri River Division for sharing his insights and understanding of the Missouri River that led to the development of some of the methods described herein. The tests described and the resulting data presented, unless otherwise noted, were obtained from research conducted under the Environmental Impact Research Program of the U.S. Army Corps of Engineers by the Waterways Experiment Station. Permission was granted by the Chief of Engineers to publish this information.

REFERENCES

Andrle, Robert. (1994). The Angle Measure Technique: A New Method for Characterizing the Complexity of Geomorphic Lines. Mathematical Geology. 26(1), pp.83-97.

Barnsley, Michael. (1993). *Fractals Everywhere*. Academic Press Professional. Boston, p.175.

Feder, J. (1988). *Fractals*. Plenum Press. New York.

Gardener, R. H. and Turner, M. G., eds. (1991). *Quantitative Methods in Landscape Ecology: The Analysis and Interpretation of Landscape Heterogeneity*. Springer-Verlag. New York.

Latka, D., J. M. Nestler, and L. Hesse. 1993. Restoring Physical Habitat in the Missouri River: A Historical Perspective. In Proceedings of the American Fisheries Symposium "Restoration Planning for the Rivers of the Mississippi River Ecosystem". pp. 636-638.

Mandlebrot, Benoit. How Long is the Coast of Britain? Statistical Self-similarity and theFractional Dimension. Science. 156, pp.636-638.

Middleton G. V.,ed. (1991). Nonlinear Dynamics, Chaos, and Fractals with Applications to Geological Systems. Short Course Notes. Vol.1. Geological Association of Canada. Toronto.

Nestler, J. M., D. Latka, T. Schneider, and P. N. Johnson. 1995. Physical Habitat Analysis Using the Riverine Community Habitat Assessment and Restoration Concept (RCHARC): Missouri River Case History. Technical Report EL-95-18.

USING HYDRAULIC AND WATER QUALITY MODELING OUTPUT FOR INSTREAM FLOW STUDIES

L. Toni Schneider
U.S. Army Engineer Waterways Experiment Station
3909 Halls Ferry Rd
Vicksburg, MS 39180
USA
schneil@ex1.wes.army.mil

John M. Nestler
U.S. Army Engineer Waterways Experiment Station
3909 Halls Ferry Rd
Vicksburg, MS 39180
USA
nestlej@ex1.wes.army.mil

ABSTRACT

Many hydraulic and water quality models commonly employed by engineers to simulate environmental conditions in streams and reservoirs can also be employed to describe physical habitat impacts of impoundment or stream regulation. These models can vary in complexity from relatively simple one-dimensional (longitudinal) steady-state models such as HEC-2 or QUAL2E to time-varying one- or two- dimensional models such as UNET, CE-QUAL-RIV1, or CE-QUAL-W2. Hydraulic and water quality output from these models can be directly linked to instream flow models. We present example applications in which hydraulic model or water quality output is indirectly coupled to PHABSIM, a software package often employed by biologists to conduct instream flow studies. Examples include: 1) time-varying water quality model to predict the downstream water quality and physical habitat effects of altered flows associated with peaking hydropower operation on the main stem Missouri River; and 2) time-varying water quality model used to predict the downstream impacts on physical habitat resulting from hydropower upgrade/uprate on dams located on the Cumberland River System. In both these examples, combining hydraulic and water quality model output with the PHABSIM system created a foundation from which engineers and biologists can interact more efficiently to evaluate the effects of reservoir operations on in-pool and downstream aquatic resources.

KEY-WORDS: Instream Flow Needs/ IFIM/ PHABSIM/ Habitat Analyses/ Physical Habitat Modeling/ Stream Impact Assessment/ Flow Requirements/ Maintenance Flows

INTRODUCTION

Impact assessment studies that describe the environmental impacts of major water resources development usually cover several major topic areas including channel stability-capacity and inpool and downstream water quality studies, in addition to instream flow studies. Channel stability studies are commonly conducted downstream of dams or in flood conveyance channels to determine if the flow capacity of a stream has changed as a result of sedimention, erosion, or bank sloughing. Water quality studies are commonly conducted to determine the effects of flow alterations on nutrient dynamics or waste assimilation. Instream flow studies are conducted primarily by biologists to determine the effects of flow alterations on physical and chemical habitat for aquatic biota. From both a planning and implementation perspective, these diverse studies may involve different teams of professionals that may not understand or appreciate how the tools employed for one study objective may support, supplement, or even enhance the efforts in seemingly unrelated study efforts by other teams of professionals. Consequently, efforts may be duplicated across several different studies with a resultant decline in quality across all studies. In this paper we describe how tools that are commonly employed in flood routing, water quality, and water diversion studies can be linked to tools that are commonly employed to conduct instream flow studies determining the impact of flow alterations on aquatic habitat.

Supplementing the data needs for instream flow studies with data from other parallel study efforts requires a basic understanding of the steps involved in conducting instream flow studies. Instream flow studies are usually individually tailored to meet the specific needs and requirements of each application. However, in very general terms, instream flow studies usually mirror concepts familiar to ecological modelers that specialize in impact assessment in that models are used to predict environmental impacts of project atlernatives on key variables and thus facilitate trade-off analysis between environmental and economic impacts. The most commonly employed program library for executing an instream flow study is the Physical Habitat Simulation System (PHABSIM System). A variety of options and pathways that vary considerably in their treatment of biological and hydraulic simluation is available within the system (Milhous et al. 1981, Milhous et al. 1989). However, an instream flow study typically consists of the following seven steps (Nestler et al. 1989):

1. Categorize potential impacts of water resources projects into major components (i.e., water quality, channel morphometry, flow, etc.)
2. Describe the component parts, often using a marriage of concepts found in open-channel hydraulics, hydrology, sediment transport, aquatic ecology, and environmental engineering.
3. Describe the existing state of the system in terms of key driving variables and response variables (flow regime, total habitat, or minimum daily dissolved oxygen level)
4. Modify values of one component (e.g., the shape of the channel or the flow regime) based on field measurement or analytical or simulation techniques
5. Evaluate impacts over a range of values for the component being evaluated (incrementally change a driving variable and observe the incremental response of the system)
6. Repeat the evaluation, changing values for other components of impact on the water resources project.
7. Based on the evaluations, prepare assessments/recommendations for the water resources project.

Note in the above seven steps that steps one through four involve many of the same tools or steps that are commonly employed in engineering studies to simulate channel stability/capacity flood routing studies, and water quality studies. Steps one through four provide the major opportunity to supplement the technological base (primarily the hydraulic/hydrologic information) of an instream flow study with information from other major study elements. Usually the biological base of an instream flow study cannot be supplemented with information from other study elements.

The most flexible approach for conducting the hydraulic portion of an instream flow study divides the problem into two steps (Milhous et al. 1989):

1. Predicting water surface elevation-discharge relationships (e.g., Figure 1)
2. Predicting flow pattern (depths and velocities) across a transect once the water surface elevation-discharge relationship is known (e.g., Figure 2).

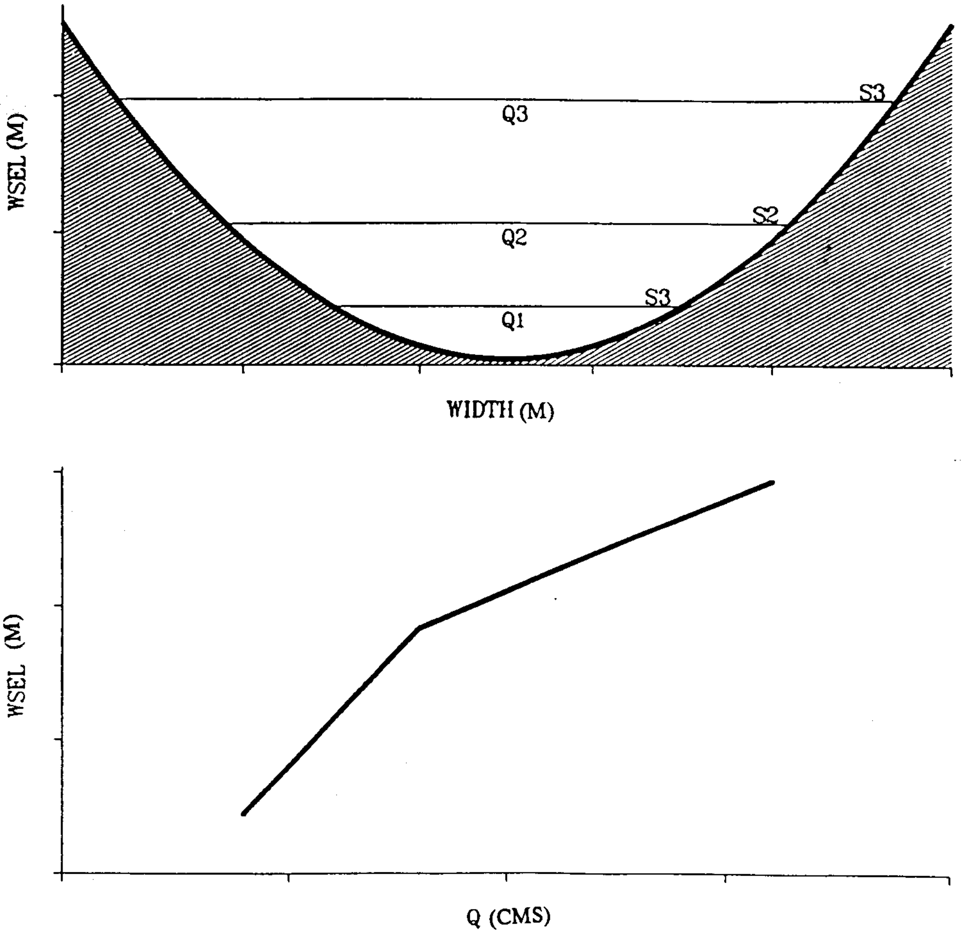


Figure 1. Determining the stage-discharge relationship either by measurement or simulation is the first step in performing the hydraulic portion of an instream flow study. (A) As river flow through this hypothetical cross section increases from discharge Q1 to Q2 and then to Q3, the river water surface elevation (stage) increases from S1 to S2 and then to S3. (B) Graphical representation of the stage-discharge relationship.

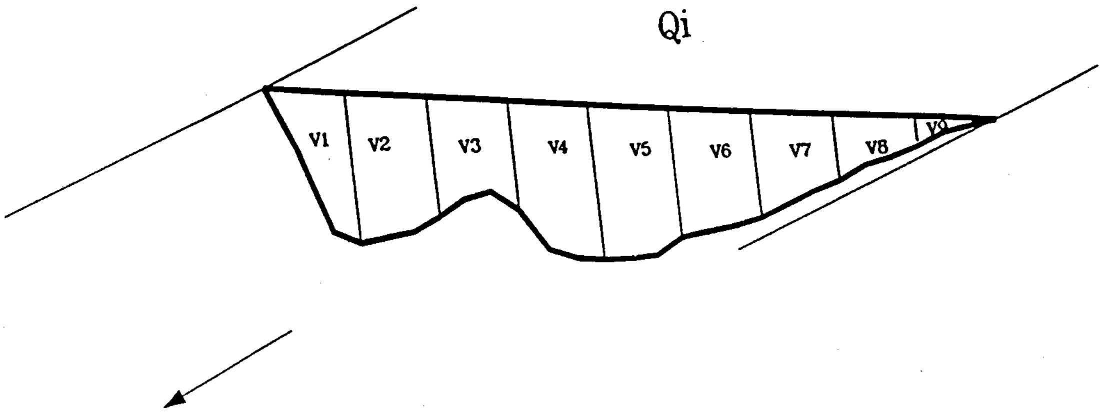


Figure 2. After the stage-discharge relationship has been determined, the next step in performing the hydraulic portion of an instream flow study is to determine the lateral flow pattern; that is, to describe the velocity distribution, V1 to V9, across the transect for a specific discharge, Qi. Depths can be easily determined by subtraction because both the water surface elevation and the bottom elevations are known (from the steps presented in Figure 1).

The most common interface between an instream flow study and other elements of impact assessment involves the first step. Prediction of the stage-discharge relationship is a common element of almost all river impact studies. It is relatively easy to import stage-discharge information into an instream flow study.

METHODS

The following examples illustrates how an instream flow study can benefit from the technological base provided by parallel studies. For this example, the hydrodynamic and water quality model, CE-QUAL-RIV1, was used to predict the extent of downstream water quality changes associated with different operational alternatives for two dams, one on the Missouri River and one on the Cumberland River. In both examples, RIV1 was used to predict time-varying stage-discharge information at nodes that coincided with the locations of transects to describe fish habitat dynamics. The stage-discharge information was passed to the PHABSIM System where it was used to predict a lateral depth-velocity pattern of sufficient detail that it could be used to simulate fish habitat. In both these cases, the addition of the time-varying stage-discharge information considerably enhanced the information that would normally have been available if the hydraulic simulation modules in PHABSIM only were employed.

CE-QUAL-RIV1 is a one-dimensional (longitudinal) hydrodynamic and water quality model for riverine systems. This model was developed for highly unsteady flows, but can be used for steady flow conditions. CE-QUAL-RIV1 consists of two components, RIV1H, the hydrodynamics module, and RIV1Q, the water quality module. RIV1H predicts the hydrodynamics necessary to drive the water quality model. This includes discharge, area, top width, and stage data for each node in the reach for each time-step in the simulation. RIV1Q then takes this information and predicts over time the concentration of twelve different water quality constituents, including temperature. Implementation of CE-QUAL-RIV1 requires the following steps:

1. Channel cross-sectional data
2. Set of boundary conditions and initial conditions.

3. Meteorological data.
4. Set of a number of different model coefficients and exponents.
5. Model calibration.

The CE-QUAL-RIV1 model and PHABSIM system, when coupled together, can be used to determine the effects of peaking operation on tailwater habitat. RIV1H, the hydraulic code of CE-QUAL-RIV1, can be executed to create stage-discharge pairs needed for the IFG4 program which is part of the PHABSIM system. The IFG4 program then generates a distribution of velocities and depths across the cross section, one distribution for each stage-discharge pair at a preselected interval (daily or hourly values). After predicting the velocity and depth distributions across a cross section, the IFG4 program passes this information to the HABTAT program (another PHABSIM module) in which the cell by cell conditions are evaluated relative to the criteria of the target life species. The result of HABTAT is the amount of available weighted useable area for each targeted species for each specific discharge.

RESULTS

Figures 3 and 4 present the information that was available for evaluation after the PHABSIM system was linked to the RIV1 model. Figure 3 shows stage-discharge plots for two cross sections that are generally similar in shape. One cross section is immediately downstream of a dam, and the other is 32 kilometers downstream of the dam. Note in Figure 3 that the RIV1 model was able to capture the attenuation of the peak flow and the increase in the low flow as the power generation wave moved downstream of the dam. Figure 4 represents an example using a dynamic flow model. Note that in simulating habitat under dynamic flow conditions, the axes are different than under a steady-state analysis. Rather than the ordinate representing discharge, it represents discharge at regular time intervals. An instream habitat analysis using a dynamic flow model presents habitat as a function of time and distance and not as a function of discharge only.

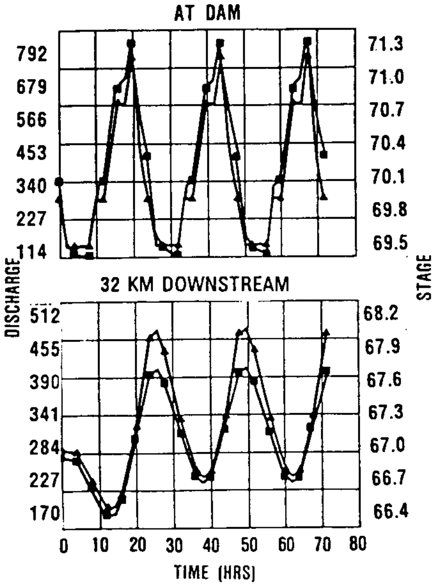


Figure 3. The above two plots demonstrate change in flow characteristics as a peaking wave flows downstream. Flow at the dam (top plot) varies from approximately 113 to 736 cms. However, note that flow conditions 32 km downstream (bottom plot) are considerably different. In the latter case, flow varies from approximately 198 to only 453 cms.

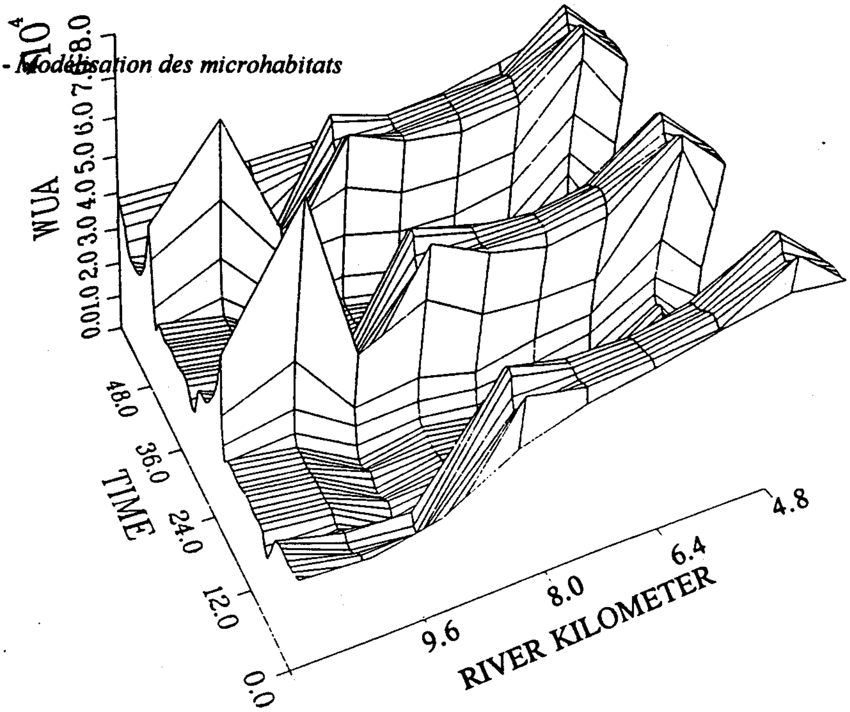


Figure 4. This plot presents adult brown trout habitat over a period of 60 h in a river below a peaking hydropower project. The periods of high habitat value correspond to periods of nongeneration. The habitat minima (troughs centered at hours 24 and 48) occur during periods of maximum generation.

DISCUSSION

In some cases, it may be necessary to link the PHABSIM system to a dynamic flow model to generate hourly habitat values for each transect. Also, in some situations, it is important to know how the habitat varies along a stream as a power wave from a hydroelectric project moves downstream. It may also be necessary to analyze each transect for habitat because flow may change substantially as the releases move downstream of the dam. Furthermore, it may also be necessary to include a temporal component in the habitat analysis because the habitat requirements of some aquatic species exhibit diurnal or seasonal changes. The advantage of the analysis presented in Figure 4 is that it is the most accurate possible description of time-varying habitat available to aquatic organisms under dynamic flow conditions. That is, flow conditions that aquatic biota are subjected to are more accurately portrayed than if steady-state hydraulics were being used. The analysis is sufficient for the effects of different alternatives to be completely assessed. From such an analysis the worker can determine not only habitat values at flow extremes, but also determine how habitat changes over intermediate flow values and the rate of habitat change over different parts of the generation cycle.

ACKNOWLEDGEMENTS

The tests described and the resulting data presented herein, unless otherwise noted, were obtained from research conducted under the EIRP Program of the U. S. Army Corps of Engineers. Permission was granted by the Chief of Engineers to publish this information. We gratefully thank Dr. Patrick Deliman and Ms. Dorothy Tillman for their review of this literature.

REFERENCES

Milhous, R. T., Wegner, D. L., and Waddle, T. J., User's Guide to the Physical Habitat Simulation System, FWS/OBS-81/43, Office of Biological Services, U. S. Fish and Wildlife Service, Washington, D. C., 1981.

Milhous, R. T., Updike, M., and Schneider, D., User's Guide to the Physical Habitat Simulation (PHABSIM) - Version II, Biological Report, U. S. Fish and Wildlife Service, 1989.

Nestler, John M., Milhous, R. T., and Layzer, James B., Alternatives in Regulated River Management, "Instream Habitat Modeling Techniques", pp. 296-315, 1989.

TOWARDS THE NOTION OF TROPHIC VALUE
Concepts and approaches
The brown trout example (*Salmo trutta*)

B. Dumont, G. Suard, Y. Le Coarer, G. Carrel et B. Rivier
Cemagref Aix-en-Provence
U.R.E. Hydrobiologie
BP 31
13612 Aix-en-Provence cedex 1
France
E-mail: bernard.dumont@cemagref.fr.

ABSTRACT

The concept of trophic value for brown trout requires notions of availability and transfer closely related to the predatory behaviour of this species, to the behaviour of potential preys and to flow dynamics. In order to supplement simulation methods for carrying capacity in regulated streams with additional elements, a first study of feeding and prey drift was undertaken in a river section under legal minimum discharge where hydrological stability is strong. The objectives were to determine the mechanisms affecting drift structure and food intake by fish. Improvements were made to field digitalization and to the drift study design. The drift longitudinal gradients observed in two flow series have repercussions on sub-populations densities and conditions. High turbulence is a limiting factor in prey selectivity. The trophic value approach for a given river section for brown trout must necessarily combine an estimate of the flow diversity and an assessment of the biological potential if solid comparisons are to be made.

KEY-WORDS: Trophic value / Brown trout / Feeding / Drift study / Flow Diversity / Reference Discharge / Regulated River.

INTRODUCTION

When defining artificial discharge characteristics, carrying capacity simulations for salmonid populations try to answer the several questions raised in the past 20 years on minimum flow requirements (Bovee and Cochnauer, 1977; Bovee, 1978). More recently, in France, adaptations were brought upon the method to test the potential of multivariate analyses (Souchon *et al*, 1989; Souchon, 1994).

The physical parameters used in the general model of distribution per development stages include other implicit informations (food type and availability, growth, territoriality) for all post-emergence stages. However, as noted by Irvine *et al* (1987), how can we detect that a salmonid population is more limited by trophic factors than by the sum of useable habitats ? Are there comparable biological functions between natural situations and regulated streams ? What can be transfered from natural conditions to regulated conditions? On what basis should we assess, comprehend and compare lotic hydrosystems? And how can we integrate a trophic value in simulations and discussions prior to decision-making?

In a recent review, Orth (1995) highlighted our incomplete knowledge: "a major dilemna in instream flow assessments is our immature capabilities for considering the indirect effects of flow on multiple trophic levels". He concluded that more research is needed, associated with the evaluation of minimum flow requirements, exploring various paths: "1) Research on the efficacity of meso-scale analyses can provide more guidance on predicting the diversity and abundance of habitats. 2) Research is needed to more clearly demonstrate the indirect effects of flow in altering food web. 3) Habitat suitability criteria must be more rigorously tested and linked to some measures of individual fitness."

For these reasons, we have initiated, several years ago, research on the trophic value for the brown trout (*Salmo trutta*). We opted to assess the mechanisms of prey availability to trout populations under regulated discharge conditions. Artificial hydrological conditions are selected according to legal evolution in France (Merle 1996). This option reduces the probability of climate biases and allows an approach specific to given flow characteristics with the following objectives:

- validate the hypothesis of a drift spatial structure related to a specific flow configuration,
- test these hypotheses on fish food intake,
- study the trophic performances related to the distribution of individuals.

Our goal is to provide information complements to currently used salmonid habitat simulations under artificial discharge characteristics. A primary objective is to evaluate the role of flow on predation mechanisms affected by hydrodynamic changes over a river bed reach.

The first part of this paper proposes a global reflexion on the needs of references and the evolutions of habitat modeling, important for each specific case. The second part presents the preliminary results of new approaches adapted to a lotic system and to brown trout.

CONCEPTS

The notion of trophic value

The term trophic in ecology refers to "everything concerning the circulation of food within ecosystems" (Ramade, 1993).

Essentially, the trophic value is a relative notion since the concept may apply at different levels (specific or global) and at various perceptions of the trophic structure (nutrients, primary productivity, benthos or fish productions).

In France, Léger (1937) introduced the concept of biogenic capacity. The objective then was to adapt fish stockings to the estimated food carrying capacity of the river. But, although a minimum biological database has been compiled, this topic has yet to be explored in depth, especially in a global perspective of sound fish management (Maisse and Baglinière, 1991).

The abundant litterature in the last century has demonstrated that brown trout shows a great ecological plasticity (Baglinière and Maisse, 1991; Greenberg, 1994) and very opportunistic feeding habits (Stankovitch, 1992; Neveu, 1991). The trout's basic ethology (feeding, reproduction, migration, territoriality) is well known through the life cycle. Feeding behaviour in a lotic environment is mostly based upon predation on drifting preys, with a varying low rate of benthic feeding (Bachman, 1984).

Applied to brown trout, the trophic value of a site equals the number of preys the species can catch, while the trophic potential corresponds to the total number of preys at the site. Our approach aims at developing an estimation method of the trophic value for brown trout, based upon information on the trophic potential combined with flow organization and diversity.

Three types of information are essential: on-site benthos populations, availability of potential preys (drift and mobility) and diet composition.

The other component of the approach is the relationship between hydrology and biology in a concept of biodiversity, in order to elaborate a functional reference system to compare different situations.

Selecting a functional reference system

The diversity of hydraulic conditions within a river reach, a fundamental factor of trophic value, regulates both the distribution of the various organisms (trophic potential) and the dynamics of exchange (drift) between different meso-habitats in the reach.

One of the problems facing the determination of a regulated discharge is the absence of a functional reference, which would allow the classification of any river reach according to its natural potential, within a global scheme of biocenotic structure acceptable by all.

With this positionning achieved, it would become easier to propose sound management objectives for a given system.

The scientific difficulty is not technical but cultural, because it refers to the first of three fundamental principles in ecology proposed by Thienneman (1920), that of biocenotic diversification related to the variety of habitat conditions: "The greater the diversity of the conditions in a locality the larger is the number of species which make up the biotic community".

The difficulty is not technical because the modem means of data collection and analysis (hydrology, hydraulics and biology) and the modeling methods take into account the time-space basis essential to an ecosystemic overview.

It is cultural because the idea is not to find what is going well or bad, but to position the system in functional terms, relating to life forms which are, or were, present at some stage of an hydrological cycle.

Following the notions of zonation (Huet, 1949; Illies and Botosaneanu, 1963), the notions of ranking of hydrographic networks (Shreve *in* Gregory and Walling, 1973), the concept of biocenotypes (Verneaux, 1977), the various methods for the classification and simulation of lotic systems tend to take into account the structure of their functions.

In accordance with the logics of continuum and stability in a dynamic equilibrium as developed in fluvial geomorphology by Leopold (*in* Gregory and Walling, 1973), and applied to the structural and functional organization of lotic communities (Vannote *et al.*, 1980; Newbold *et al.*, 1981), it appears interesting to supplement the notion by developing a concept of biodiversification combining hydraulics and biology and to test it in an operational and sectorial approach.

The objective is to create a step-by-step characterization method, excluding in the first step general biological organization such as longitudinal zonation and biogeography.

The scientific community agrees that, between the excesses of extreme phenomena, the ecological equilibrium and the biodiversity of lotic systems are the consequence of seasonal fluctuations of river discharges (Ward and Stanford, 1983; Bayley, 1991). The permanent balance between flood discharges and dry-weather discharges has a long-term effect on the morphology of streams and on the associated biocenosis.

In our concept, the first step is to grade the structuring determinisms, beginning with the first determinism which characterizes a river, water flow. Indeed, hydrodynamic behaviour during annual cycles on a river reach determines spatial distributions, temporal sequences, and consequently the existence and balance of the various life forms linked or not to the presence of current.

Figure 1 illustrates the basic concept of biodiversification in running water. It represents the biological reference discharge around which the peak of the biodiversity maintenance theoritical curve is found in the "intermediate disturbance hypothesis" (Connell *in* Ward and Stanford, 1983).

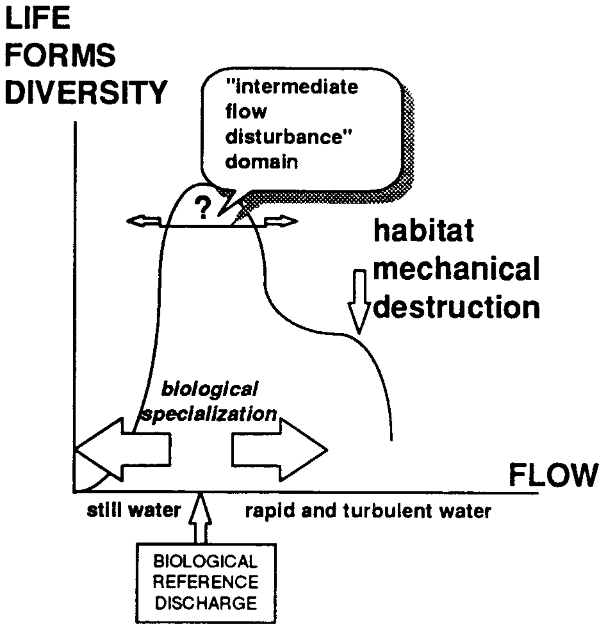


Figure 1: Theoritical diagram of biodiversification as a function of discharge in a river reach.

In this local context, biological potentials depend upon the various aquatic supports, whose presence and survival are the consequences of hydrological events in the local geological context, and the causes of spatial and instantaneous diversification in the reach's flow. These potentials are apparent mostly in terms of density, as a function of other determinisms in the system.

Thus, the determination of the maximum hydraulic diversification is only the first step of other functional declinations (support density, temperature regimen, benthos structure, etc.).

Determination of a Reference Discharge

This concept is not new and was developed in the course of impact assessment studies of hydroelectrical works in the early 1980s, as the "Biological Reference Discharge", (BRD) by Dumont and Rivier (1981).

According to Thienneman's first principle of ecological diversity, we can assume that the discharge value offering the largest biological diversification is the discharge providing, in terms of spatial distribution, the widest range of hydraulic variations within the river reach. The problem resides in defining the standard measure to identify the discharge value which, in a given river reach, offers the greatest diversification of life forms linked to water flow. In our early research, we had opted for a minimum of three velocity ranges of flow facies : F1<0.3 m/s ; F2 = 0.3-0.8 m/s ; F3 > 0.8 m/s. A flow velocity measurement is statistically oriented by the river bed geometry. Hydraulic diversity is greatest when three classes ar equally represented. A similar approach is found in the flow velocity separation method, with polygons of meso-habitat common types (Orth 1995).

When applied to an hydrological cycle, the determination of a BRD reveals the system's potential in generating predominant lotic or lentic phases. The BRD value may also measure the range of diversification phase occurences in terms of duration, frequency, continuity, and according to seasons. Applied directly at a fixed regulated discharge, it locates the biodiversification hydrodynamic potential in the meso-habitat web.

MATERIALS AND METHODS

Two research domains had to be addressed: river morphology modeling and drift study technique.

Numerical Terrain Model (NTM)

In recent years, Le Coarer and Dumont (1995) have developed a method to digitalize the lotic space. The technique combines a spatial grid (triangular base straight prisms), a curvilinear identification and a generation of cross-profiles, perpendicular to the river discharge. It also offers a varied topological management (grid, polygons, transverse, facies) which ultimately allows coupling with a biological model at various perception scales. Coupling with water level calculations has not been attempted, but the method offers potential links with various hydraulic softwares.

Drift study

Statzner *et al* (1984) and Brittain and Eikeland (1988) have reviewed the topic. We also conducted a preliminary assessment in which we selected our tactics and methods for the drift study (Suard and Dumont, 1989). Drift is a complex phenomenon combining daily behaviour diversities, seasonal biocenosis variations and short term climatic influences. The objective is to integrate the proper space and time scales in determining the trophic value.

The drift study used electrical pumps with vortex effect and a two-inch inside diameter integral intake. Each submersible pump has a rigid suction pipe, of equal diameter, approximately 1 meter long. Aerial filtration out of the wetted bed is conducted from a 70 mm diameter flexible force pipe. Flows are continuously recorded by ultrasound rate-of-flow meters (Ultraflux, UFT 321 or Digisonic P) with sensors located on rigid sections of the force flow transit. The pump rate (6 or 10l/s depending on the model) was selected in order that suction speed is always much higher than natural on-site flow velocity.

The immersion of the suction point is calibrated at the upper limit of the mean bottom roughness and is facing the general flow direction. Unless wanted, the immersion point is located away from the surface or from any substrate. A velocity profile is measured vertically to the suction point prior to pumping. When starting pumping operations, a visual control is performed in slow conditions to detect an eventual vortex at the intake and to adjust the position of the suction point.

There are several advantages to this system: no clogging, accurate monitoring of the flow, easy time splits, no needs for frequent wadding in the river after installation, facilitated location of suction point.

Trouts and stomach contents

A Fulton condition coefficient (K. FULTON = $100*W/L^3$) is calculated from individual trout length and weight measurements. Stomach contents are collected by stomach forced washes on anasthetized fish. This technique is more efficient than conventional stomach pumping (Neveu and Thibault, 1977). The pressure, generated by a garden spray, is directed in the stomach by a small rigid cannula. The flow separates the preys and forces them in the mouth.

The stomach content is then collected in a small sieve and preserved in 4% formalin. In the laboratory, stomach contents are dried on a filter by a vaccuum pump and are weighed to the nearest 0.1 mg. The repletion coefficient equals the relative proportion (%) of the stomach content weight on the total weight of the fish.

STUDY AREA

The studied reach is located on the Durance River, immediately below the Serre-Ponçon dam (southern Alps, altitude 650 m). The dam, of a 1260 million m^3 capacity, is the deviation head of one of France largest hydroelectrical works (power production = 1707 MW) which channels over a distance of 200 km the waters of several drainage basins to the Mediterranean Sea. The minimum legal discharge is in the order of 1/40 of the interannual module, or 2.1 m^3/s in the study area where the lowest natural discharge recorded was around 17 m^3/s.

The proximity of the dam (3 km), associated with a reservoir feeding the deviation canal (capacity of 250 m^3/s), provides the river reach with a great hydrological and morphological stability, and induces a low range of temperature variations (average 11°C) caused by the turbining of deep waters. Unretained flood flows are very rare and short-lasted (July 1983, May 1986, October 1993). The result of these overall conditions is a fine sediment accumulation when lateral torrents, some of which are temporary, carry fine amounts of suspended matters; large aquatic vegetation beds may develop over short periods of time (*Chara* sp. and *Potamogeton* spp.).

PRELIMINARY RESULTS

Preliminary results on drift and trout feeding are presented. They represent a first global overview of a data set presented in a stage report by Rogue (1995) and being currently analysed in the course of a third cycle research project. A study is planned for a five-year period (1995-1999) and will compare the micro-habitat simulation predictions with the response of a salmonid population after the implementation of a minimum legal discharge on a pyrenean river (Aude).

Comparison between simulation and observed distribution of trouts

Using the numerical model described above, we compared the micro-habitat simulation with the actual distributions of fish captured by electrical fishing in July 1992. The NTM was derived for the whole river reach, and 54 fishing zones were sampled (Carrel *et al.*, 1992). A fishing zone corresponds to a flow facies sub-unit. The mean sampling surface is 116 m^2 and the mean time fishing effort is 9.2 mn.

For each zone, a habitat value using preference curves (Souchon *et al.*,1989) was calculated for each development stage (alevins, juveniles, adults). Trout densities per zone were compared for three size groups (TR1<100 mm, TR2 = 100 to 200 mm, TR3>200 mm) based upon the global population structure.

Wilcoxon comparison tests were conducted after weighing resulting values by the maximum value in each variable (development stages and TR classes). Each of the three series present a highly significant difference ($p < 0.01$). Therefore, the hypothesis of a global distribution model to simulate a carrying capacity cannot be used under the current conditions at the studied river reach. Several explanations are possible, including a trophic limitation caused by an important substrate clogging and a weak minimum legal discharge.

In order to address the trophic aspects, we have examined the following points.

Drift and flow facies

Several experiments were conducted during April 1994 to verify the spatial structure of the drift and to combine trout feeding data (zones A, B, C, D, E) at two locations (1 and 2) separated by 500 m and having different flow characteristic sequences (Figure 2).

- Location 1: a longitudinal set of 4 pumps installed in a pool-riffle sequence were operated at nightfall (8:30 to 10:30 PM, coded L1DN) and at sunrise (5:30 to 8:00 AM, coded L1ND).

At the same location, a transversal set of two pairs of pumps in the upstream portion of the riffle was used to compare the drift near riverbanks and at mid-channel (10:00 to 12:00 AM, coded T1D). Pumps 1 and 4 were located nearshore in a straight line with pumps 2 and 3, respectively.

- Location 2: a longitudinal set of 3 pumps in a riffle end-lotic run was operated at nightfall (8:30 to 10:00 PM, coded L2DN).

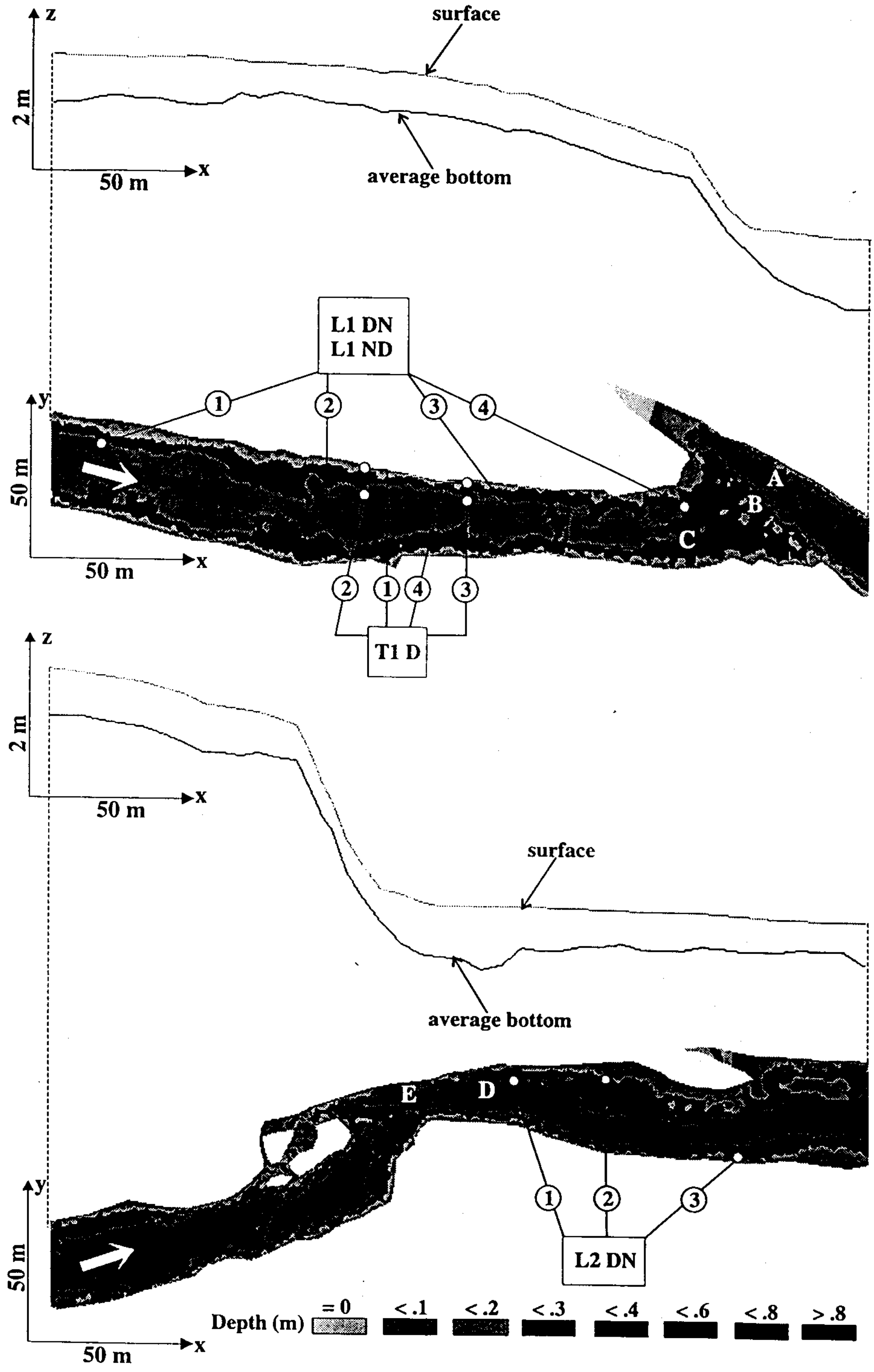


Figure 2 : Fish sampling sites A, B, C, D, E and drift pumping series, L1 ND, L1 DN, T1 D, L2 DN

Each data set is compiled from a continuous monitoring; the cumulative results are presented in Table 1 as:

- drift density: number of individuals per unit of volume filtered. This measurement compares the concentrations of potential preys.
- Drift flow: number of individuals per unit of surface and per unit of time. The area filtered is estimated by the ratio between the mean pump flow and the front velocity at suction point. This measure assesses the frequency of potential prey passages in a given field of vision for a fixed observer.

Table 1: Global drift data

	front velocity m/s	m^3 filtered	drift density ind/m^3	drift flow ind/mn/100cm^2
Site 1 - L1DN				
pump 1	0.08	35.2	34	2
pump 2	0.55	30.4	45	15
pump 3	0.55	28.1	64	21
pump 4	0.93	20.3	65	36
Site 1 - L1ND				
pump 1	0.08	39.0	6	0.3
pump 4	0.93	37.7	7	4
Site 1 - T1D				
pump 2 (unchanged)	0.55	27.5	62	20
pump 1 near P2 and riverbank	0.20	33.7	33	4
pump 3 (unchanged)	0.55	24.7	70	23
pump 4 near P3 and riverbank	0.24	17.9	51	7
Site 2 - L2DN				
pump 1	0.59	25.9	41	14
pump 2	0.22	14.8	31	4
pump 3	0.30	27.0	23	4

At location 1, the nightfall longitudinal data (L1DN) reveal an increase in drift density in an upstream-downstream gradient. In terms of drift flow, the gradient is amplified by the correction for surface filtered.

At sunrise (L1ND), the gradient is apparent only at the drift flow level; the drift densities, much lower than at nightfall, are similar at the distant pumps (p1 and p4).

In the T1D day time study, a gradually increasing longitudinal drift gradient is also recorded between pumps p2 and p3. The gradient also occurs near the riverbank between p1 and p4. An increasing transversal gradient is clearly showing from the riverbank towards mid-channel for each pair of pumps (p1 and p2, p4 and p3). However this transverse gradient is less apparent in the downstream portion of the riffle.

At location 2, there is a decreasing upstream-downstream gradient in drift density. In terms of drift flow, the front velocity, higher at p3 than p2, compensates a lower drift density.

Distribution of fish and feeding pattern

Fishing zones were selected on the basis of their relative locations in the local geomorphology (Table 2).

Table 2: Physical parameters in zones A, B, C, D, E.

	Zone A	Zone B	Zone C	Zone D	Zone E
Geomorphologic position	transition riffle-pool	riffle	riffle	transition riffle-run	transition riffle-run
Mesohabitat at 2.6 m3/s	shallow pool	slow riffle	slow riffle	medium pool	shallow pool
Surface (m2)	221	200	320	199	162
DEPTH (m)					
mean	0.58	0.23	0.38	0.76	0.65
median	0.59	0.21	0.35	0.76	0.64
VELOCITY (m/s)					
mean	0.30	0.23	0.48	0.35	0.47
median	0.27	0.51	0.48	0.34	0.47
lower quartile	0.22	0.25	0.36	0.15	0.28
upper quartile	0.37	0.70	0.60	0.52	0.59
ROUGHNESS (mm)					
mean size	150	120	210	260	140
range	100-1070	80-340	140-410	170-550	90-450

According to the drift study, zones A, B and C are located in an upstream-downstream increasing drift gradient while zones D and E are in an upstream-downstream decreasing gradient. Zones A, B and C have significatively different depth and velocity. Zone B is the most heterogenous in terms of velocity and is the shallowest. Zone D and E are the deepest and are located in a transition between riffle and run.

Table 3 summarizes the trout biological parameters per zone. Because of technical difficulties, repletion data from zone C were not simultaneous to data from zones A and B.

Table 3: Biological data in zones A, B, C, D, E

	Zone A	Zone B	Zone C	Zone D	Zone E
TROUT (*Salmo trutta*)					
Trout nb	26	17	40	13	34
nb/100 m2	12	8.5	12.5	6.5	21
g/100 m2	872	274	576	291	1164
LENGTH (mm)					
mean	178	141	164	160	160
median	183	126	155	168	132
WEIGHT (g)					
mean	74	31	47	47	57
median	60	16	34	44	28
K.FULTON					
mean	1.02	0.81	0.92	0.97	1.08
median	1.04	0.83	0.90	0.97	1.04
REFLETION	(a) comparable data			(b) comparable data	
mean	0.69^a	0.40^a	0.32	0.62^b	0.81^b
median	0.55	0.40	0.18	0.27	0.53

Biomass and density are greatest in zone E. Zone A is second in biomass and is equal to zone C in density. Zones B and D have similarly low biomass and density. These observations suggest a higher trophic value in zones E and A, which is also apparent in condition coefficients of individuals.

Globally, the Spearman Rank Correlation (Table 4) between variables indicate that there are no significant correlation between repletion rate and weight or length, nor between the condition coefficient and weight or length, while there is a weak significant correlation between the condition coefficient and the repletion rate.

Table 4: Spearman Rank Correlations (coefficient, *significance level*) in biological data.

	Length	Weight	K.Fulton	Repletion
Length	1	*<0.0001*	*0.07*	*0.24*
Weight	0.9768	1	*0.74*	*0.46*
K.Fulton	-0.1610	0.0294	1	*0.009*
Repletion	-0.1054	-0.0664	0.2317	1

Repletion and condition may be considered as good indicators of the state of individual fish within the population. The significant correlation between repletion and condition is more surprising and, at the same time, more interesting because it shows that fish with the best condition factors ate more. It is surprising because this correlation is observed on only one meal, and because the simultaneous sampling allows to compare repletion between zones A and B, and between zones D and E. Figure 3 illustrates these trends.

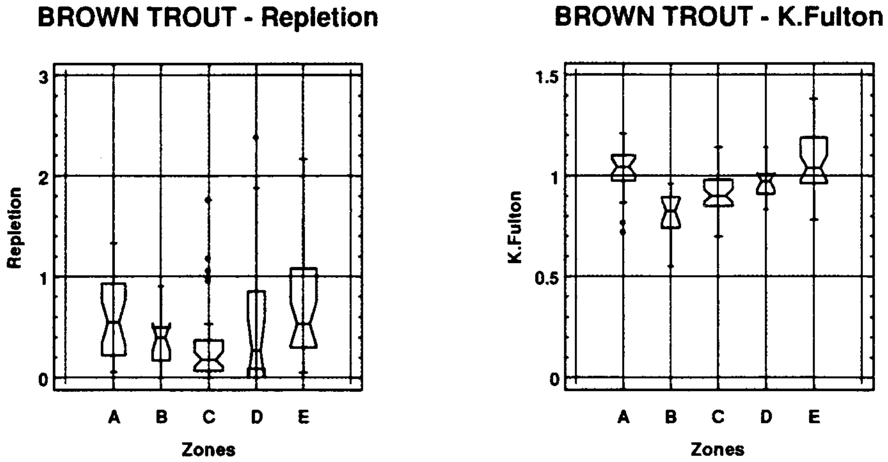


Figure 3. Notched Box-and-Whisher Plot for Repletion and Condition (K. Fulton) Data.

The weaker trophic performances in zone B are interpreted as an evidence of lower efficiency in catching preys. Contiguity with zone A suggests that drift density conditions are very similar. Turbulence becomes a limiting factor for trout food selection.

DISCUSSION

Preliminary results of an integrated project combining a drift study and a trout feeding study in a regulated river reach indicate that the drift is strongly related to the topography of the river bed and to flow sequences, and that individual trouts integrate the physical parameters and the trophic conditions of their distribution sites. The integration of environmental conditions is reflected in both fish densities (abundance and biomass) and individual condition factors. It is likely that these results are associated with the territoriality of the species.

The hydraulic transition immediately below riffles (zones A and E) has the highest trophic value in the studied regulated discharge conditions. Conversely, the downstream portion of the riffle (zone B), where velocity is most heterogenous and turbulence is greatest, has the lowest trophic value.

Riffles act as production and injection sites towards the lower reaches of a fraction of drifting benthos. Longitudinal gradients of drift densities and drift flows can be very important according to the time of day, a normal situation in any drift event, and according to the nature of transitions in meso-habitat sequences. Therefore, determining the trophic value for trout necessarily implies a simultaneous assessment of hydrodynamic diversification and of benthic stock structure and availability.

Data is not yet sufficient to draw firm conclusion on the trophic functionning of a trout population subjected to strong artificial discharge constraints. However, the meso-habitat approach, combined with an assessment of the hydrodynamics of the studied river reach, seems to be an interesting approach for the determination heterogeneity of the trophic value. A strong spatial heterogeneity of individual condition factors may induce a non-saturation trophic index. Assessing the trophic value of a site for brown trout must necessarily combine functions of benthos availability and prey selectivity to the determination of hydraulic characteristics, because these notions depend jointly on the nature and quantity of potential preys, and on the quality of flow diversity.

In the mid-term, it appears feasible to assign, for given regulated discharge conditions, a trophic value to a river reach based on data on meso-habitat sequences and hydrodynamic diversity within these sequences (transfer velocities, turbulence level, substrate densities). Current knowledge, combining deterministic and stochastic modeling (grid elements, velocity domain) is sufficient to initiate this research. Assigning an absolute trophic value is a more complex initiative. Site-specific biological data will be needed to assess food availability in terms of potential prey vulnerability to brown trout.

REFERENCES

Bachman R.A. (1984). Foraging behaviour of free-ranging wild and hatchery brown trout in a stream. *Trans. Am. Fish. Soc.*, 113: 1-32.

Baglinière J.L. & G. Maisse (1991) (Eds.). La truite, biologie et écologie. INRA éditions. 303 p.

Bayley P.B. (1991). The flood pulse advantage and the restoration of river-floodplain systems. *Regulated Rivers* 6: 75-86.

Bovee K.D. (1978). General Management Concept. The Incremental Method of Assesssing Habitat Potential for Coolwater Species, with Management Implication. *Am. Fish. Soc. Spec. Publ.* 11: 340-346.

Bovee K.D. & T. Cochnauer (1977). Development of weighted criteria, probability-of-use curves for instream flow assessments: fisheries. *Instream Flow Information Paper n° 3.* FWS/OBS-77/63. U.S. Fish and Wildlife Service, Ft. Collins, Colorado. 38 p.

Brittain J. E. & T.J. Eikeland (1988). Invertebrate drift - A review. *Hydrobiologia* 166: 77-93

Carrel G., B.Dumont, Y. Le Coarer & B. Rivier (1992). Morphodynamique et biologie piscicole de la Durance dans le secteur Espinasses - Rochebrune (Aval de Serre-Ponçon). Rapp. Tech. Cemagref - Min. Environ. 103 p.

Dumont B. & B. Rivier (1981). Le débit de référence biologique. *Doc. tech. Cemagref Aix-en-Provence, n°19.* 30 p.

Greenberg L.A. (1994). Effects of predation, trout density and discharge on habitat use by brown trout, Salmo trutta, in artificial streams. *Freshw. Biol.* 32, 1-11.

Huet M. (1949). Aperçu des relations entre la pente et les populations piscicoles des eaux courantes. *Trav. stn. rech. Groenendaal.* 11 (3/4): 332-351.

Illies J. & L. Botosaneanu (1963). Problèmes et méthodes de la classification et de la zonation écologique des eaux courantes considérées surtout du point de vue faunistique. *Mitt. int. Ver. theor. angew. Limnol.* 12: 1-57.

Irvine J.R., Jowett I. G. & D. Scott (1987). A test of the instream flow incremental methodology for underyearling rainbow trout, *Salmo gairdnerii*, in experimental New Zealand streams. *N. Z. J. Mar. Freshwat. Res.* 21: 35-40

Le Coarer Y. & B. Dumont (1995). Modélisation de la morphodynamique fluviale pour la recherche des relations habitat / faune aquatique. Colloque "Habitat-Poissons". *Bull. Fr. Pêche Piscic.* 337/338/339: 309-316.

Léger L. (1937). Economie biologique générale des cours d'eau alpins. *Bull. Fr. Pêche Piscic.* 109:1-13.

Maisse G. & J.L. Baglinière (1991). Connaître le bases biologiques de la gestion, une idée toujours d'actualité, pour la truite (Salmo trutta L.). *in* Baglinière J.L. & G. Maisse (Eds.), La truite, biologie et écologie, pp.297-302. INRA Editions.

Merle G. (1996).Submitted.

Neveu A. (1991). Stratégie alimentaire de la truite commune (*Salmo trutta* L.) en eaux courantes. *in* Baglinière J.L. & G. Maisse (Eds.), "La truite, biologie et écologie". pp.97-120. INRA Editions.

Neveu A.. & M. Thibault (1977). Comportement alimentaire d'une population sauvage de truites fario (*Salmo trutta* L.) dans un ruisseau des Pyrénées Atlantiques, Le Lissuraga. *Ann. Hydrobiol.* 8: 111-128.

Newbold, J.D., J.W. Elwood, R.V. O'Neill & W. Van Winkle. (1981). Measuring nutrient spiraling in streams. *Can. J. Fish. Aquat. Sci.* 38: 860-863

Orth D.J. (1995). Influence du compartiment trophique dans les réponses des populations de poissons aux variations artificielles de débit. Colloque "Habitat-Poissons". *Bull. Fr. Pêche Piscic.* 337/338/339: 317-328.

Ramade F. (1993). Dictionnaire encyclopédique de l'écologie et des sciences de l'environnement. EDISCIENCE international. 822 p.

Rogue O. (1995). Approche de la notion de valeur trophique des faciès d'écoulement pour la truite fario (*Salmo trutta*) en situation de débit artificiel. *Dipl. Ing. ENGEES, Strasbourg.* 80 p.

Souchon Y. (1994). Etat d'avancement des recherches sur la modélisation de l'habitat des poissons des cours d'eau en France. *Bull. Fr. Pêche Piscic.* 332: 57-71.

Souchon Y., F. Trocherie, E. Fragnoud & C. Lacombe (1989). Les modèles numériques des microhabitats de poissons, applications et nouveaux développements. *Sciences de l'eau,* 2: 807-830.

Stankovitch S. (1922). Alimentation naturelle de la truite (Trutta fario, Linné) dans les cours d'eau alpins. *Trav. Lab. Piscic. Univ. Grenoble.* pp. 115-191.

Statzner B., C. Dejoux & J.M. Elouard (1984). Field experiments on the relationship between drift and benthic densities of aquatic insects in tropical sreams (Ivory coast). I. Introduction: review of drift literature, methods and experimental conditions. *Rev. Hydrobiol. Trop.* 17: 319-334.

Suard G. & B. Dumont (1989). La recherche d'un indice de saturation trophique à l'égard des ichtyocénoses. Application à la truite fario (*Salmo trutta*). Etat de la connaissance et protocole technique adopté. *Doc. tech., convention EDF-DER/Cemagref E 30/L 13-2F9754.* 50 p.

Thienemann A. (1920). Die Grundlagen der Biozönotik und Monards faunistische Prinzipien. *Festchrift für Zschokke. Basel.* 4: 1-14.

Vannote R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell & C.E. Cushing (1980). The River Continuum Concept. *Can. J. Fish. Aquat. Sci.* 37: 130-137.

Verneaux J. (1977). Biotypologie de l'écosystème "eau courante". Détermination approchée de l'appartenance typologique d'un peuplement icthyologique. *C.R. Acad. Sc. Paris.* 284 D: 675-678.

Ward J.V. & J.A. Stanford (1983). The intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. *in* Fontaine III, T.D. & S.M. Bartell, Dynamics of lotic ecosystems. pp.347-356.

The Relationships Between River Channel Morphometry, Complexity and Aquatic Habitat

Chad P. Gubala
Associate Director, Institute of Natural Systems Engineering
Utah State University, Logan, Utah, USA 84322-4110
(chad@neptune.cee.usu.edu)

Joseph M. Eilers and Joseph A. Bernert
E+S Environmental Chemistry
PO Box 609, Corvallis, Oregon, USA 97339
(74131.1574@compuserve.com)

ABSTRACT

Increasing complexity of physical habitat typically results in increasing biological diversity. However, measuring habitat complexity and resultant biological potential in freshwater aquatic systems has been limited by practical constraints. Recent technological advances have provided the means to measure and assess the relationships between aquatic habitat complexity and biological potential.

Using an innovative, high-technology method, finely-resolved morphometric data was collected for 12 1-km sections of the Willamette River, Oregon, USA. Electronically linking an advanced hydroacoustic array with Different Global Position System (DGPS) survey devices allowed for the efficient collection of precise position and depth data over a broad spatial domain. Raw position and depth data obtained for each reach were transformed into regular finite-difference grids through interpolating kriging techniques. The complexities of each reach were assessed by developing spatial metrics, such as fractal dimensions and coefficients of variation. Results from the 12 study reaches of the Willamette River reveal a relationship between complexity (derived from the coefficient of variation) and distance upstream. Additionally, reach complexity indices relate positively with biological indices obtained through independent investigations.

KEY-WORDS: SONAR / GPS / Bathymetry / River / Habitat / Oregon /

INTRODUCTION:

In response to an April 1990 Mandate from the Oregon (USA) Joint Legeslative Emergency Board, the Oregon Department of Environmental Quality (ODEQ) formed the Willamette River Technical Advisory Steering Committee (WRTASC). The WRTASC was charged with developing a comprehensive study to provide the necessary technical and regulatory understanding required to protect and enhance habitat and water quality of the Willamette River. One phase of the resultant Willamette River Basin Water Quality Study (WRBWQS) included assessment of the physical habitat defined by channel morphometry.

Biological communities in rivers develop in response to both water quality and the physical attributes of the system. We attempted to quantify the physical habitat of the river to develop to better understand the linkages between fish assemblages and its physical components. By characterizing the physical habitat, a better understanding of the factors that influence community composition and population abundance among different reaches is developed.

Evaluation of habitat quality is also an important component for assessment of ecological integrity. Physical habitat quality was qualitatively assessed in test reaches of the Willamette River during field sampling activities in 1992-93 (Tetra Tech, 1992; 1994). This qualitative assessment was used to evaluate ecological integrity of benthic and fish community samples. However, the qualitative nature of the habitat assessment limited the interpretation of these data. Given the changes in the physical habitat that have occurred in the Willamette River basin and the importance of the riparian habitat to the integrity of aquatic and terrestrial biota, development of a more quantitative habitat assessment protocol was deemed necessary. Development of a quantitative protocol will increase the comparability of data collected from different sources and will reduce uncertainties associated with evaluating the relationship between biological assessment metrics and habitat quality.

Habitat assessment is the evaluation of the structure of the surrounding habitat that influences the quality of the water resource and the condition of the aquatic community. An assessment of the habitat is the intended to asses human-induced perturbation to the physical structure of the habitat. The ultimate goals of habitat assessments are varied, but the immediate objective is typically to document the relative quality and/or quantity of habitat available to fish within a given reach (Simonson, et al., 1994). A "macrohabitat" assessment of physical habitat, as is used herein, uses several parameters from a site to arrive at an overall assessment of habitat conditions or availability for that site. Individual parameters are selected that represent specific components of the habitat structure. The integration of the information derived from those parameters allows for a holistic assessment. Because some parameters are likely to be interrelated and may be highly correlated, it is necessary to test the degree of correlation and, thus redundancy, in the parameters.

Channel morphology is determined by the flow regime of the river, local geology, land surface form, soil and human activities. Meandering channels tend to have a greater variety of habitats than artificially straightened channels, which generally have uniform conditions (Simonson, et al., 1994). Diverse habitat generally supports more species, a greater variety of lifestages and higher abundance of fish. River reaches made up of a series of connected geomorphic channel units (macrohabitat units) that can generally be described by visual determination. Habitat units include bends, riffles, runs, pools, islands, dams, log jams, etc. The pool/riffle or run/bend ratio are measurements intended to be made by dividing the average distance between riffles or bends by the average river width. These parameters assume that a stream with riffles or bends provide more diverse habitat that a straight or uniform depth stream. These parameters could not be assessed within the site lengths examined in the Willamette River as all of the sites could be described as a single geomorphic unit, a run. Due to this uniform nature of the macro habitat study units of the Willamette, complexity analyses were based upon detailed bathymetric data. This manuscript focuses upon the innovative methods and analysis techniques developed to compare the physical habitat of 13 reaches of the mainstem and primary tributary of the Willamette River.

METHODS:

Twelve mainstem and one tributary segment of the Willamette River were scheduled for detailed bathymetric analyses, the basis for physical habitat comparisons (Table 1.). The bathymetric data were collected at the same 13 sites as another suite of habitat components, collected over several years. These data were collected in February and March of 1995 when river stage was at it's highest. This made it possible to map sections of the river that would have been to shallow for operation of a motorized craft during the Summer or Fall.

Table 1.) Site locations for bathymetric study sections. Location coordinates are provided in degree decimal-minutes and represent the downstream starting point of a 1.0km reach.

SAMPLING DATE	STATION ID	RIVER MILE	LATITUDE, N	LONGITUDE, W
02/22/95	W06	6	45d34.72m	122d44.72m
02/22/95	W17	17	45d26.93m	122d39.26m
02/22/95	W25	25	45d21.51m	122d36.50m
02/22/95	W49	49	45d16.89m	122d57.52m
02/25/95	W58	58	45d12.24m	123d02.85m
02/25/95	W77	77	45d01.56m	123d04.46m
02/25/95	W93	93	44d51.92m	123d08.85m
02/25/95	W113	113	44d41.03m	123d07.15m
02/25/95	W130	130	44d34.69m	123d14.55m
03/06/95	W145	145	44d25.46m	123d13.45m
03/06/95	W150	150	44d22.39m	123d14.21m
03/07/95	W190	190	44d01.44m	122d57.65m
03/07/95	Mc27	M27	44d03.37m	122d49.75m

A Global Position System (GPS) base station was established in the vicinity of each of the study sections. A second "rover" GPS was fixed in a motorboat and linked to a data logger, connected directly to a hydroacoustic array (SONAR; Gubala, et al., 1994). The sonar used a narrow beam, high frequency transducer, adjusted to work optimally within each river segment. The combined sonar/GPS system provided simultaneous position and depth information precise to within 0.5m (x,y) and 0.05m (z). A study reach was transected using a diagonal search pattern extending from bank to bank. Each river reach was passed over a second time in order to better define the thalweg. Generally, well over 1,000 individual depth/position measurements were made on each reach.

The position and depth data were downloaded on-site and the position data were immediately "differentially corrected" to enhance x,y precision. The raw bathymetric data were then reviewed for completeness and accuracy. Due to the high resolution of the instrumentation, spurious data was periodically encountered (less that 2%) due to "second echo" phantoms. These false signals were easily censored through used of forward and backward auto-regressive techniques. Computer aided continuous surfaces were generated for the study sites using a finite difference grid density of 20-50m (depending upon the domain size).

Several measures of spatial variability associated with depth were calculated for each site (Table 2.). The data were corrected for difference in water depth during different sampling times. In addition to the standard suite of statistical metrics, fractals were calculated at each site to further evaluate spatial complexity. Fractals are mathematical/spatial measures of similarity in geometric patterns (Mandelbrot, 1983; Burough, 1986).

RESULTS:

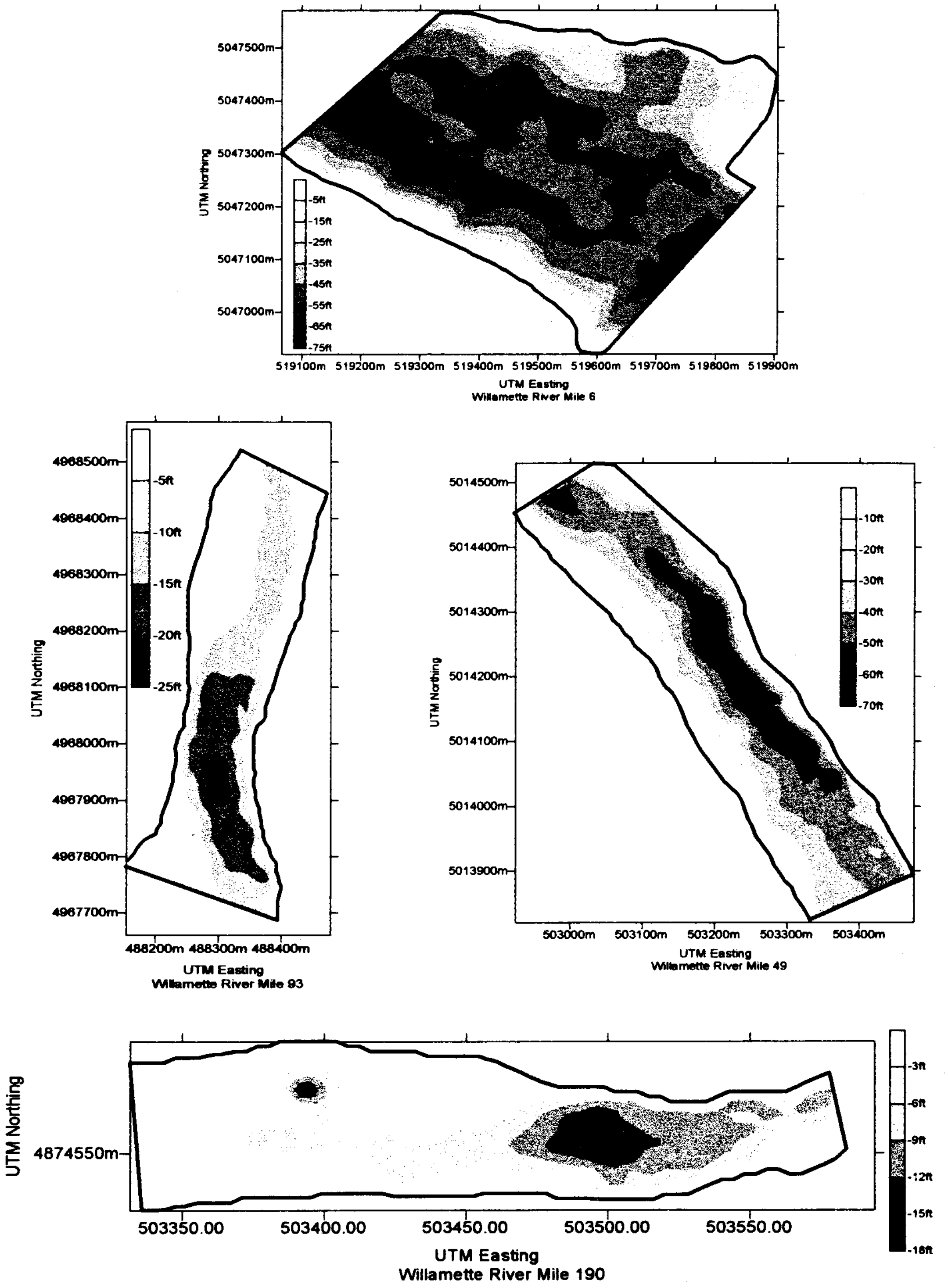
Bathymetric maps for four of the thirteen study reaches are presented in Figures 1a-d. Depths and size have been scaled according to the range in each reach to provide the best visual effect. Horizontal position information is georeferenced by the Universal Transverse Mercator (UTM) system (Muehrcke, 1978). Therefore, the position and orientation of each plot is consistent with the actual river orientation. The study reaches generally decrease in depth as one progresses upstream (Table 2.).

Table 2.) Statistical measures of river depth (in feet) associated with each sample reach on the Willamette River.

RIVER MILE	MEAN	MAXIMUM	STD DEV	CO. OF VAR.	FRACT. DIM.
6	45.8	74.6	14.7	32	0.59
17	59.2	101.1	27.3	46	0.31
25	33.9	63	10.9	32	0.31
49	34.6	65.4	15.3	44	0.45
58	18.1	40.1	9	50	0.59
77	8	15.7	2.9	36	0.87
93	10	23.4	5.1	51	0.62
113	7.3	11.3	2.2	30	0.47
130	12	22.5	4.2	35	0.5
145	4.3	14.7	3.4	77	0.5
150	6.8	19.7	3.5	51	0.2
190	5.6	16.3	3.4	60	0.54
M27	4.7	19.6	3.5	74	0.75

To quantify the physical complexity of river segments, we computed varioius measures of spatial variability associated with depth (Table 2.). Presumably, greater complexity of physical habitat results in more niches/refuges for fish and other aquatic biota to grow and reproduce. Since only single measures of aquatic habitat were compile in previous studies, it was necessary to reduce the

Figures 1a-d: Willamette River Sections 6, 49,93,190



bathymetric data into a single measurement or index. The statistics of physical complexity condense the vast amount of digital information on river depth into values that can be more easily compared among sites. Qualitative assessments of habitat can also be performed by visual examination of the bathymetric maps.

The major statistics were generated from the digital bathymetric data files. A small subset of these parameters are presented in Table 2. Two key parameters, indicative of river complexity, are the coefficient of variation (CV) and the fractal dimension (FD). The CV ((std dev/mean)*100) presents a unitless measure of dispersion and can be easily compared between sites. The FD also presents a dimensionless parameter that indicates the degree of smoothness of a natural surface. Typically these indices vary from 0-2, indicating greater landscape complexity with increasing values. A description of the means of calculating this parameter is found in Mandelbrot, 1983.

The two basic indices (CV and FD) representing reach complexity provide conflicting results. The CV indicates that complexity generally increases as one progresses upstream. However the FD indicates maximum complexity at sites W77 and Mc27 with low correlation to river mile. However, the FD statistics can be heavily influenced by the contouring intervals for these sections. Efforts are in progress to provide more consistent FD indices for these reaches.

DISCUSSION AND CONCLUSIONS:

Willamette River complexity, represented most reliably by the coefficient of variation for each reach, generally increases with the distance from the mouth. Considered with other habitat parameters, listed in Table 3., reach complexity comprised about 10% of an overall predictive habitat score, derived from an independent study of the same reaches (Tetra Tech, 1995).

Individual ranking and scoring of habitat parameters was derived from a stepwise multiple regression analysis. The total scores in Table 3. also show a general trend increasing with distance upstream. This is no surprise as reach complexity, also increasing upstream, was the single best predictor of total habitat score ($r^2 = 0.977$; total model $r^2 = 0.9992$). Most of the predictive variables included in the total habitat score were also correlated tightly with complexity, defined by the coefficient of variation. The addition parameters enable the habitat score to classify a wider range of river conditions with increased sensitivity.

Analysis of habitat scoring and assessment by reach requires considerably more work to refine the predictive capability of this overall approach. However, the method by which the bathymetric data was obtained, presented and manipulated exemplifies an effective new procedure for analysis of large rivers: SONAR/GPS. Visualizing the precise physical habitat of individual reaches or entire systems allows for a much more thorough understanding of the system attributes potentially controlling biological diversity. The precision and cost-effectiveness of this mapping procedure permits it's deployment over a wide range of aquatic systems.

Table 3.) Habitat parameters used in determining total habitat scores for the Willamette River. (Refer to Tetra Tech, 1995 for parameter details).

River Mile	Bed Substrate	% Fines	Embed-dedness	% Along Shore	% Channel Width	% Canopy
6	5	1	1	3	0	10
17	16	1	1	0	0	6
25	10	1	1	1	0	16
49	11	1	1	7	16	12
58	13	7	5	6	6	16
77	13	7	7	13	6	5
93	15	5	3	7	5	17
113	11	5	5	11	0	13
130	11	2	2	2	5	16
145	17	7	11	7	6	5
150	13	5	7	9	18	12
190	19	20	13	4	5	7
M27	20	9	11	12	16	17
	Reach Complexity	% Bank Vegetation	Bank Erosion	% Disturbed	% Ground Cover	**Total Score**
6	**6**	4	17	5	6	**58**
17	**8**	3	8	9	11	**63**
25	**8**	6	17	5	17	**82**
49	**9**	14	14	11	19	**115**
58	**11**	13	1	12	13	**103**
77	**13**	20	18	20	19	**141**
93	**12**	20	10	12	17	**123**
113	**10**	20	18	20	18	**131**
130	**8**	13	13	20	15	**107**
145	**12**	13	15	6	9	**108**
150	**11**	14	18	5	9	**121**
190	**14**	14	10	17	11	**135**
M27	**20**	20	20	14	12	**170**

REFERENCES:

Burough, P.A. 1986. Principals of Geographic Information Systems for Land Resource Assessment. Clarenden Press, Oxford.

Gubala, C.P., Branch, C., Roundy, N. and D.H. Landers. 1994. Automated Global Positioning System charting of environmental attributes: A limnologic case study. Science of the Total Environment. 148:83-92.

Mandelbrot, B.B. 1983. The Fractal Geometry of Nature. W.H. Freeman and Co., NY.

Muehrcke, P.C. 1978. Map Use. Reading, Analysis and Interpretation. J.P. Publications, Madison, WI.

Simonson, T.D. J. Lyons and P.D. Kanehl. 1994. Guidelines for evaluating fish habitat in Wisconsin streams. North Central Forest Experiment Station, Forest Service, U.S. Department of Agriculture, St. Paul, MN. General Technical Report NC-164..

Tetra Tech, 1992. Willamette River Basin Water Quality Study. Field Sampling Plan: Water Quality Model Calibration and Ecological Assessment. Final Report. Prepared for the Oregon Department of Environmental Quality. Tetra Tech, Inc., Bellevue, WA.

Tetra Tech, 1994. Willamette River Basin Water Quality Study - Phase II. Biological Sampling Data Report. Final Report. Prepared for the Oregon Department of Environmental Quality. Tetra Tech, Inc., Bellevue, WA.

Field sampling design and spatial scale in habitat-hydraulic modelling: Comparison of three models

David Scruton
Fisheries and Oceans, Science Branch, P.O. Box 5667, St. John's, Newfoundland, A1C 5X1 Canada

Jan Heggenes
Department of Biology and Nature Conservation, Agricultural University of Norway, P.O. Box 5014, 1432 Ås, Norway

Sylvie Valentin
Division Biologie des Ecosystèmes Aquatiques, Laboratoire d'Hydroécologie Quantitative, CEMAGREF, 3 bis quai Chauveau, CP 220, 69336 Lyon Cedex 09, France

Atle Harby and Tor H. Bakken
SINTEF-Norwegian Hydrotechnical Laboratory, 7038 Trondheim, Norway

ABSTRACT

Habitat-hydraulic models simulating habitat productive capacity for fish have met with limited success. A variety of reasons related to habitat structure have been invoked to explain this, but the limitations of hydraulic modelling have attracted little attention. An important aspect is what spatial scale is relevant for hydraulic data collection in the field, and the model simulations. We tested the hypothesis that hydraulic field sampling procedures and modelling scale *per se* affect results, even if the same fish habitat data and similar basic hydraulic models are used.

We compared the IFIM, the EVHA and the HABITAT habitat-hydraulic models on the same 5.56 km long and 12 to 35 m wide river segment in Newfoundland, Canada. The segment was mapped longitudinally for meso-habitat types, and 54 habitat reaches consisting of riffle (30%), run (25%), glide (20%), flat (9%) and pocket water (16%) were identified. All three models are based on the same habitat variables for simulating fish habitat. Data for fish habitat use were collected by direct underwater observation of fish, and simultaneous registration of available habitat. Data are for 127 brook trout and 193 Atlantic salmon fry and 186 parr.

Hydraulic field sampling procedures and thus spatial modelling scales are different between the models. Hydraulic data for the IFIM model were collected from, on average, 40 verticals in 14 transects stratified across and along the entire river segment, and at one flow (high; 10 m^{3}s^{-1}). Stratified transect selection was based on habitat type, and only one transect was used per section to represent selected habitat reaches. A partial data set from 3 transects were collected at moderate (6-7 m^{3}s^{-1}) and low (1.5m^{3}s^{-1}) flow. Water surface elevations were collected at all transects at all flows. Data for the EVHA model were converted into EVHA format from data surveyed for the HABITAT model, at 1.5^{3}s^{-1}. Data for the HABITAT were collected from 4 sub-segments selected as representative. Each segment included one habitat type. In each segment from 12 to 16 stratified transects were spaced, and hydraulic data recorded from, on average, 30 stratified verticals on each transect. Complete data sets were collected on two (1.5 and 7.5 m^{3}s^{-1}) contrasting water flows. This allows for a higher spatial resolution in the EVHA and HABITAT models on selected sub-segments, while the IFIM models covers the entire segment studied, but with lower resolution.

Habitat-hydraulic modelling results were in general similar between the IFIM and EVHA with respect to curves of suitable habitat versus flow, but there were differences between tha HABITAT and IFIM/EVHA. This is attributed mainly to different biological models, while hydraulic models performed similarly. WUA curves gave less information than separate suitability curves for habitat variables.

INTRODUCTION

Reasonably predictive models have been developed for microposition choice by fish in streams (e.g. Hughes 1992, Hill & Grossman 1993). However, habitat-hydraulic models simulating habitat productive capacity have met with limited success. A variety of reasons have been invoked to explain this, notably that biotic factors or all relevant hydraulic factors are not included, interactions between variables are not or only poorly simulated, fish may select habitats differently depending on spatial and temporal scales, or limiting temporal events may reduce production below capacity (e.g. Orth 1987, Heggenes 1996). Surprisingly, the limitations of hydraulic modelling have attracted less attention. An important aspect of ecology that is often overlooked, is the relevance of spatial scaling (Wiens 1989). Spatial scale may be particularly relevant for hydraulic data collection in the field, and model simulations, as suggested by the greater success of microhabitat models. If different field sampling designs or modelling scales produce different results with respect to productive capacity, more focus on hydraulic models is warranted.

The objective of the present study was, therefore, to test the hypothesis that hydraulic field sampling procedures and modelling scale *per se* may affect results. To do this, we compared habitat production capacities simulated by three habitat-hydraulic models, the IFIM, the EVHA, and the HABITAT. The hydraulic models are basically similar, and the same fish habitat selection data were used.

STUDY AREA

Pinchgut Brook is a major tributary of the Harry's River on the west coast of the island of Newfoundland, Canada (Fig.1). Harry's River has a drainage basin of 640 km^2 and a mean annual flow (MAF) of 26.5 m^{3}s^{-1} while Pinchgut Brook has a drainage area of 119.0 km^2 with a MAF of 3.9 m^{3}s^{-1}.We compared simulations from the three habitat hydraulic models on the same 5.56 km long, 12 to 35 m wide (mean 16.9 ± SD4.1) river reach (Fig. 2). The segment was mapped longitudinally for meso-habitat types, and 54 habitat reaches consisting of riffle (30%), run (25%), glide (20%), flat

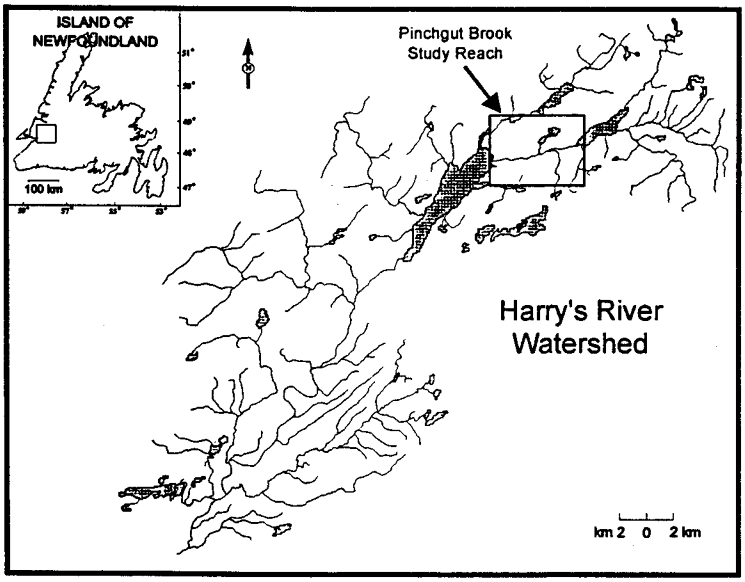


Figure 1. The study area and location in Canada.

(9%) and pocket water (16%) were identified (Fig. 2). Anadromous Atlantic salmon (*Salmo salar*) and anadromous and resident brook trout (*Salvelinus fontinalis*)are the dominant species in the stream.

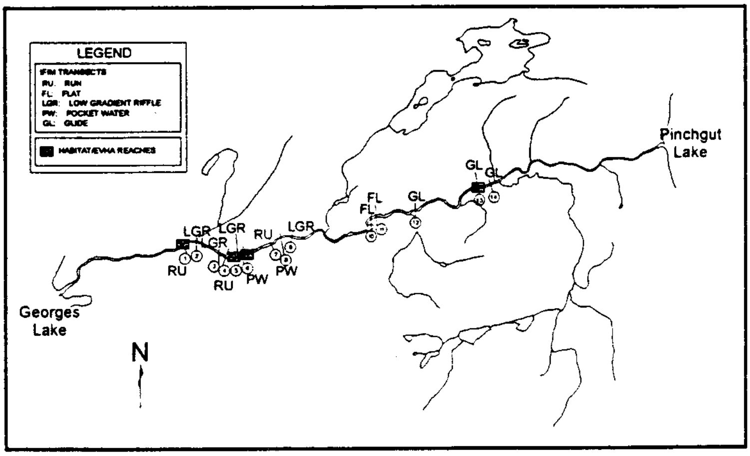


Figure 2. The study reach in Pinchgut R. with habitat reaches and transects.

MATERIALS AND METHODS

Fish habitat selection

Fish habitat selection data were collected by equal-area sampling within reach and habitat type strata. Separate reaches were delineated based on changes in gradient and distribution of habitat types. An upper reach, of lower gradient and consisting of predominantly glide and flat habitats and with preponderance of spawning gravels, was distinct from the lower reach which was higher in gradient and consisted of riffle, run, and pocket water habitats. Equal area sampling of the five habitat types ensured that available microhabitat combinations are equally represented in the development of suitability and preference criteria (Modde and Hardy 1992).

All three models were based on the variables mean water (column) velocities, water depths, and subtratum particle size for simulating fish habitat. Data for fish habitat use were collected by direct underwater observation (by snorkeling) of fish across transects. Transects were established across the width of the river perpendicular to the flow. Surface area covered was approximately 325 m^2 in each habitat type. Dives were completed by the same two snorkelers, 22 in all during the period August 22-29, 1995. Each dive lasted approximately 25 minutes and were completed during daylight hours. Underwater visibility was about 5 m. Water temperatures were always higher than 14 ^{0}C (14 to 21 ^{0}C) and flows ranged from 0.81 to 1.33 m^3s^{-1}. Additional snorkeling was completed to provide additional data on brook trout for suitability criteria. This involved establishing 20 m long transects parallell to and at distances of 1 to 5 m from the shore with effort distributed equally among habitats.

On encounter, the diver observed the fish for up to 5 minutes to confirm that it was holding a position of its own choice. Data registered by the snorkeler include fish species, length (cm), position (height off substrate), and activity (yes/no). Fish position was marked with a lead flag, and measurements were subsequently collected for mean water column and nose velocity (cms^{-1} ; Marsh-McBirney FloMate 2000 electromagnetic flow meter), total water depth (cm), habitat type (above), dominant and subdominant (percentage) substratum particle size (organic, mud, silt, sand (0.06-2mm), gravel (3-59mm), cobble (60-250mm), boulder (>260mm), and bedrock within a 20 cm radius of the fish holding position, and the presence/absence of cover was noted. If present, cover was identified and classified as instream (substrate type, vegetation or organic debris, turbulence), surface turbulence, or stream side overhanging vegetation. Habitat availability data were also collected at all transects. Based on simultaneous registration of available habitat (total water depth, dominant and subdominant substratum particle size, mean water column and bottom water velocity, and cover) in stratified verticals at 2 m intervals along each transect, habitat preferences could be calculated (Jacobs' formula; Jacobs 1974).

Habitat suitability curves, based on polynomial curve fitting to frequency histograms of fish habitat use data, were developed for IFIM and EVHA for the variables depth, mean water column velocity, and substrate for two size classes of Atlantic salmon ($\leq$ 5 cm, fry/underyearlings, n = 191; and > 5 cm, parr 1+ through 3+, n = 185). The selection of polynomial model was evaluated by goodness of fit (r^2), the relative reduction in error for each order, residual errors, and by subjective 'fit' to the data distribution. The polynomial curves were smooted at the tails of the distributions and normalized. Habitat preferences, classified as suitable, indifferent and unsuitable based on relative differences in availability/usage histograms, were developed for HABITAT for the same habitat variables, but for size classes $\leq$ 6 cm and > 6 cm for Atlantic salmon and < 9 cm and $\geq$ 9 cm for brook trout as identified by regression anlysis (below).

Habitat-hydraulic modelling

The Instream Flow Incremental Methodology (IFIM) was developed by the U.S. Fish and Wildlife Service (USFWS) in the late 1970's and have enjoyed widespread application in the United States (Stalnaker et al. 1994). The IFIM process is managed by the PHABSIM (Physical Habitat SIMulation) system which is a series of computer program modules which link stream channel hydraulics with aquatic habitat utilization of fish (and other organisms). In this study, modelling was completed using RHABSIM, a PC-DOS computer software application that was developed to reduce complexity of the PHABSIM and widen the application of the methodology. There are a number of hydraulic models within IFIM including WSP (a step-backwater method), MANSQ (uses Manning's equation to predict water surface elevations), and IFG4 (uses rating curves to develop water surface profile and accepts output from other models). In this project, hydraulic simulation was accomplished using the IFG4 module. This model requires a complete set of hydraulic measurements at one flow (usually the highest), and 2 additional sets of water surface elevations at contrasting flows.

Habitat models within PHABSIM normally incorporate the variables depth, mean velocity, substrate, and (occasionally) cover and are usually expressed as habitat suitability functions (curves). A composite habitat suitability, determined from the univariate curves using a variety of functions available within the RHABSIM software (e.g. straight multiplication, geometric mean, lowest limit, weighted average), are then used to calculate cell by cell values in each hydraulic transect. Weighing factors for each transect are then used to determine weighted usable area (WUA) for the study reach at a given flow by combining the results from all transects. The output from PHABSIM is an incremental relationship between flow and useable habitat (WUA) generated for various species and life stages of importance. The RHABSIM software also allows for spatial habitat maps of suitability (if the critical reach approach is used) and habitat time series can also be developed.

The EVHA model was developed since 1985 by Cemagref, Laboratoire d'Hydroécologie Quantitative, based on the PHABSIM procedure of IFIM, and in a PC-Windows version (Ginot and Souchon 1995) as used in this study. The hydraulic model, a stepbackwater one-dimensional model with lateral calculation of velocities, is modified from FLUVIA (Baume and Poirson 1984) with Limerinos equations (Pouilly et al. 1995), shown to be more accurate for high-to-medium streambed roughness (Trocherie 1987). The model requires one set of hydraulic and topographic field measurements. The surveyed flow (SF), usually within SF/5 to 10xSF of aimed flow, should be relatively low (Ginot and Souchon 1995). The habitat model is similar to the RHABSIM option with straight multiplication of suitability coefficients (depth, mean velocity, substrate), which is multiplied by wetted area of cell to obtain cell WUA. Normal output are spatial habitat values for cells of one to all transects (maps), longitudinal and lateral profiles of habitat variables, curves for WUA, wetted area, mean depths and velocities versus flow, and habitat time series (Capra et al. 1995, Valentin et al. 1996).

The HABITAT (Harby and Heggenes 1995) in the River System Simulator was jointly developed by SINTEF Norwegian Hydrotechnical Laboratory and Norwegian Freshwater Ecology and Inland Fisheries Laboratory. It needs input from registrations of substrate, water depth and water velocities preferrably on three contrasting water flows as well as input from the hydraulic model HEC-2 (US Army Corps of Engineers 1982). Mean water velocities from HEC-2 calculations are converted to water velocities for a cell of the river transect using measured values and linear correlation. The HEC-2 describes stationary streamflow and is used to calculate the friction described as Mannings n. Simultaneous observed values of discharge, water velocity and water level at each cross-section are required for calibrating the HEC-2 model. The deviation between observed and calculate values is minimized by adjusting the Mannings coefficient (French 1986), which in HEC2 includes all energy losses. In this study, deviation in depth was on average ±1.7cm and $\pm 3.5 cms^{-1}$ for mean velocity. Biological models within HABITAT are usually expressed as habitat preference functions based on usage/availability and separately for each habitat variable. Spatial habitat suitability maps, habitat-suitability-versus-flow-curves and timeseries of total available habitat suitabilities are normal output from the model. Calculation of WUA using real weights for the habitat variables is also possible.

Hydraulic field sampling procedures and thus spatial modelling scales are different between the three models. Hydraulic data for the IFIM model were collected from, on average, 40 verticals in 14 transects stratified across and along the entire river segment, and at one flow (high; 10 m^3s^{-1}). Stratified transect selection was based on habitat type (Fig. 2, Tab. 1), and only one transect was used per section to represent selected habitat reaches. A partial data set from 3 transects were collected at moderate (6-7 m^3s^{-1}) and low ($1.5m^3s^{-1}$) flow. Water surface elevations were collected at all transects at all flows.

Table 1. Study segments selected for the three models, with cross references.

Habitat type	Reach number (length, m)	IFIM transect number	No. of HABITAT transects	EVHA transect number
Run	15 (47)	1	12	12
Riffle	22 (66)	5	16	16
Pocket water	23 (55)	6	16	16
Glide	51 (60)	13	14	15

Data for the EVHA model were converted into EVHA format from data surveyed for the HABITAT model, with surface elevations collected at 1.5^3s^{-1} in order to aim at modelling from 0.5 to 10^3s^{-1}. Hydraulic data from all transects contained lateral distance, depth, mean velocity and substrate. Topographic data were constructed after digitization of maps of the transects, with upstream/downstream limits of representativity located mid-between two consecutive transects. WUA's were first obtained for the four reaches (Tab. 1), and for the entire reach weighted by the length of each habitat reach.

Data for the HABITAT were collected from 4 sub-segments selected as representative for the run, riffle, pocket water and glide habitat types, and all segments were associated with IFIM transects being studied (Tab. 1). Each segment included only one habitat type. In each segment from 12 to 16 stratified transects were spaced, and hydraulic data recorded from, on average, 30 stratified verticals on each transect. Complete data sets were collected on two (1.5 and 7.5 m^3s^{-1}) contrasting water flows (11-16 September and 16-21 October 1995, respectively). This allows for a higher spatial resolution in the HABITAT model on selected sub-segments, while the IFIM models covers the entire segment studied, but with lower resolution.

In each segment we collected longitudinal profile (elevation) of transects relative to the benchmark, bottom profile (elevation) of each transect, water surface elevation (minimum 2) for each transect on either side of the river or across river if necessary, and in each stratified vertical we collected total water depth, mean water column water velocity, substrate particle size, and cover type.

RESULTS

Fish habitat selection

Data were collected for 76 and 52 individual brook trout (mean total length 8cm±SD4, range 2-20cm) and 370 Atlantic salmon parr (mean TL 5.4cm±SD3.0, range 2-15cm). There were significant differences in habitat use between species (Kolmogorov-Smirnov and Kruskal-Wallis tests, $p<0.05$). Atlantic salmon occupied deeper (mean 30cm±SD16) and more fast flowing (mean $28cms^{-1}$±SD18) habitats compared to brook trout (depth: mean 24cm±SD11, mean water velocity: mean $18cms^{1}$±SD 13). Brook trout used finer substrates than Atlantic salmon (Fig. 3).

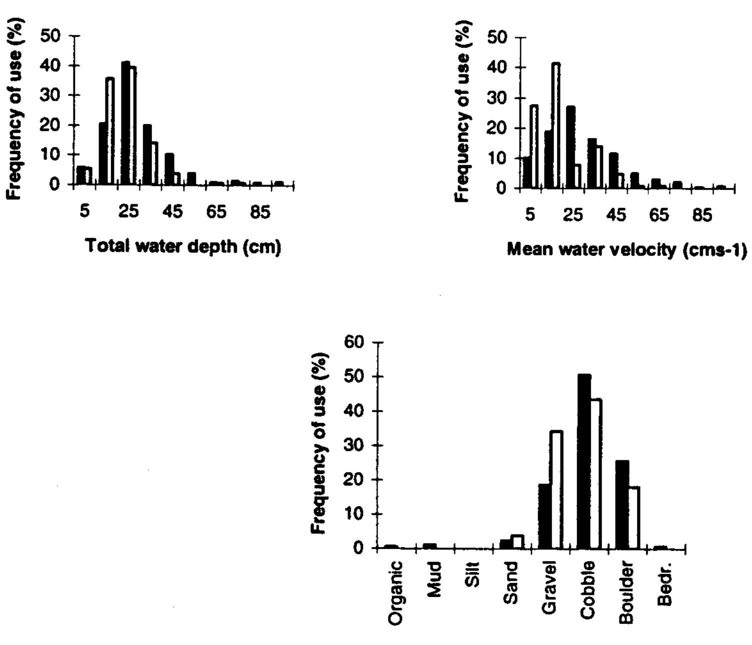


Figure 3. Differences in habitat use between Atlantic salmon (solid bars) and brook trout (open bars) for water depth, mean water velocity and dominant substrate in Pinchgut Brook, fall 1995.

Fish length significantly affected habitat use within species (stepwise linear regressions, $p<0.05$). Smaller fish were primarily associated with the slow water habitats (glide and flat) while larger fish were more in the faster water habitats (riffle, run, and pocket water).

Suitability curves (based on use) for Atlantic salmon demonstrated similar use between fry and parr for depth, velocity and substrate, but with parr tending to utilize slightly deeper, faster water over coarser substrates (Fig. 4). The difference in habitat use between age classes were less than demonstrated in some other studies (e.g. deGraaf and Bain 1986, Morantz et al. 1987, Heggenes 1990), and may reflect the fact that changes in habitat use is a gradual and continuous process. A similar trend in usage of deeper, faster water over coarser substrate was also apparent witn increasing size of brook trout. Habitat use by brook trout were generally narrower than for comparable size groups of juvenile Atlantic salmon.

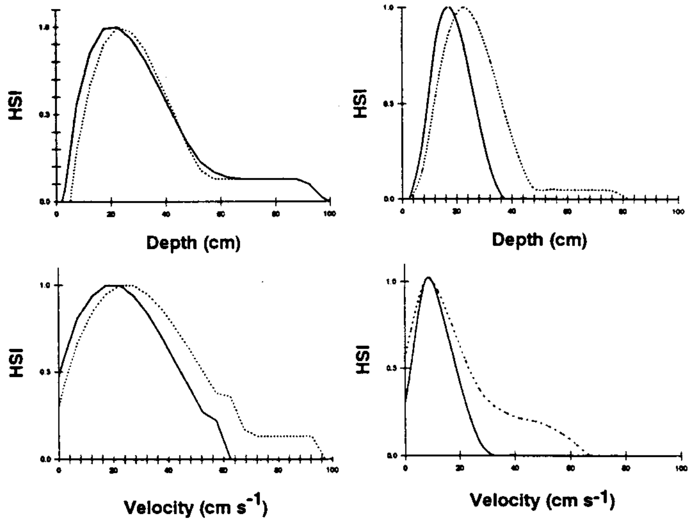
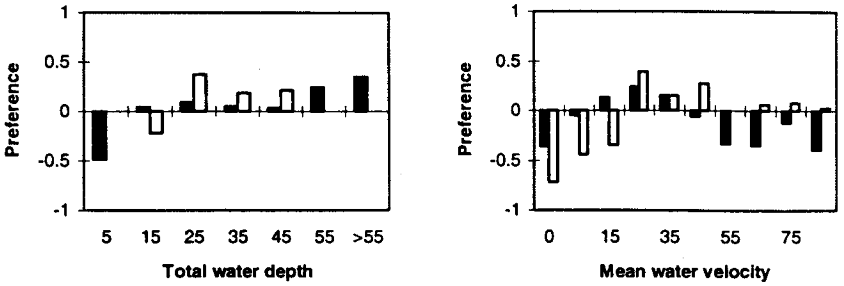


Figure 4. Habitat suitabilities for young-of-the-year (solid line) and larger parr (dotted line) of Atlantic salmon (left) and for small (solid line) and larger (dotted line) brook trout (right) for water depth and mean water velocity in Pinchgut Brook, fall 1995.

For calculation of habitat preferences (Fig. 5), the in situ fish data were divided into small (≤ 6 cm, young-of-the-year, n=262) and larger parr (> 6cm, n=108) for Atlantic salmon, and brook trout 2-9cm (n=51) and larger (n=25). Because number of fish observations was limited, data were pooled to make preference values more robust by using relatively large unit intervals for depth and mean velocity (Fig. 5), and by categorizing them into unsuitable, indifferent and suitable.



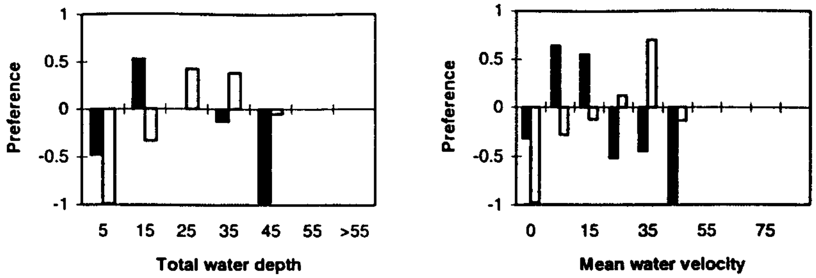


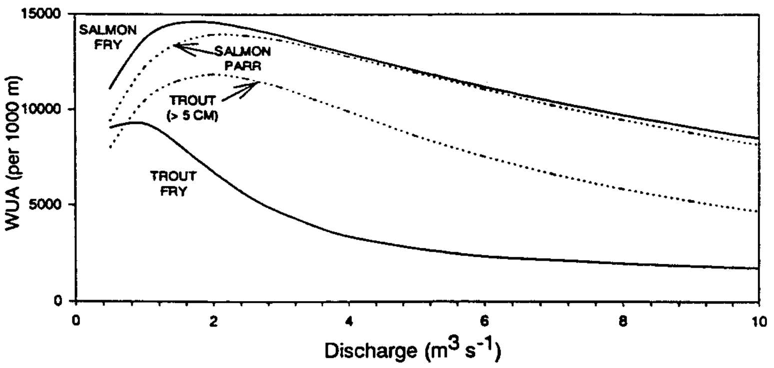
Figure 5. Habitat preferences for young-of-the-year (solid bars) and larger parr (open bars) of Atlantic salmon (upper) and for small (solid bars) and larger (open bars) brook trout (lower) for water depth and mean water velocity in Pinchgut Brook, fall 1995.

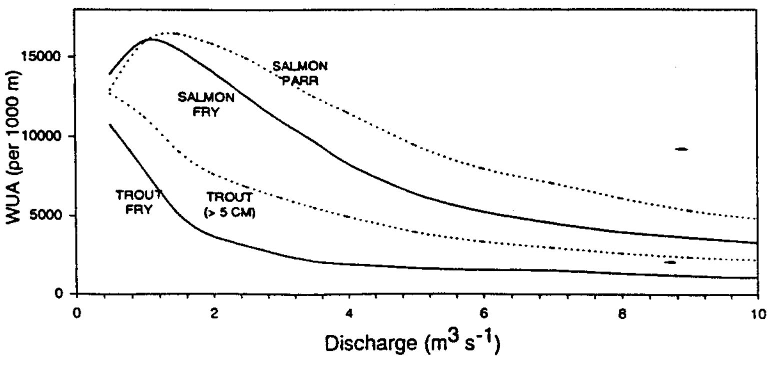
Suitable-habitat-versus-flow curves in three models

All the models produce a high number of different outputs and in different formats. For model comparisons, we made an effort to standardize as much as possible ouput on variables which appeared to be sensitive to changes in model assumptions; water depth and mean water velocity, and for riffle habitat and the entire study reach.

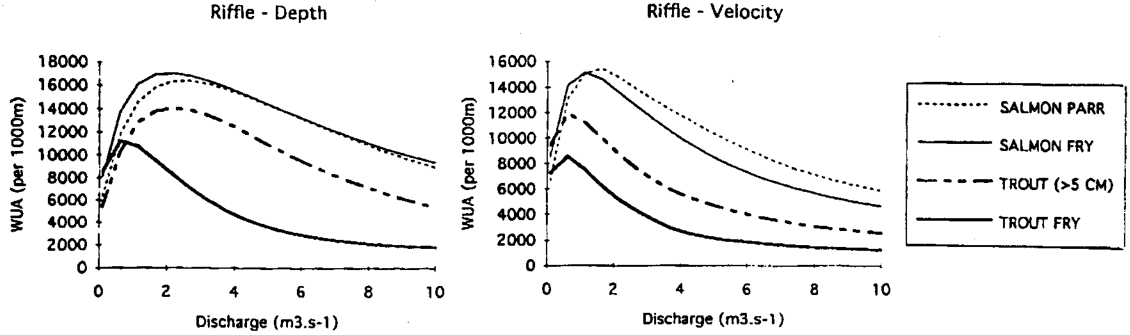
Riffle habitat

a: IFIM


Riffle - Depth


Riffle - Velocity

b: EVHA



c: HABITAT

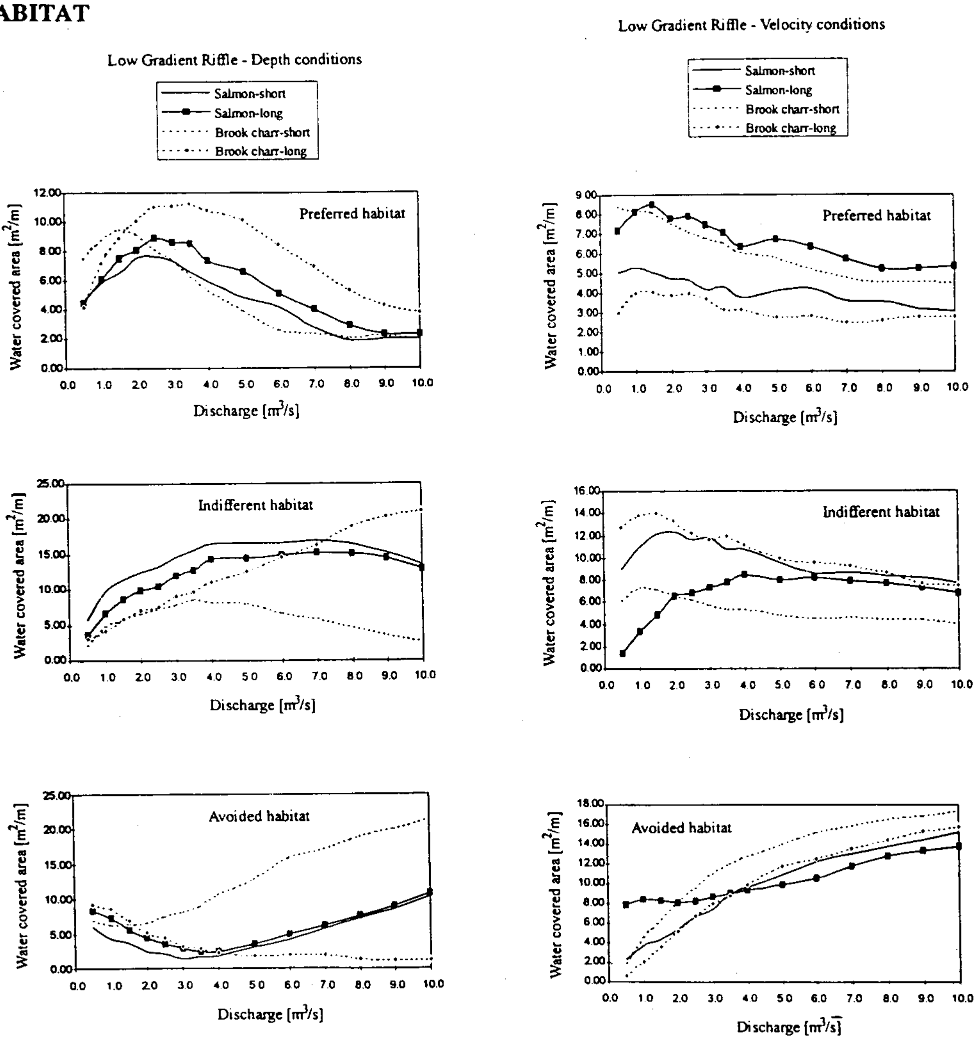
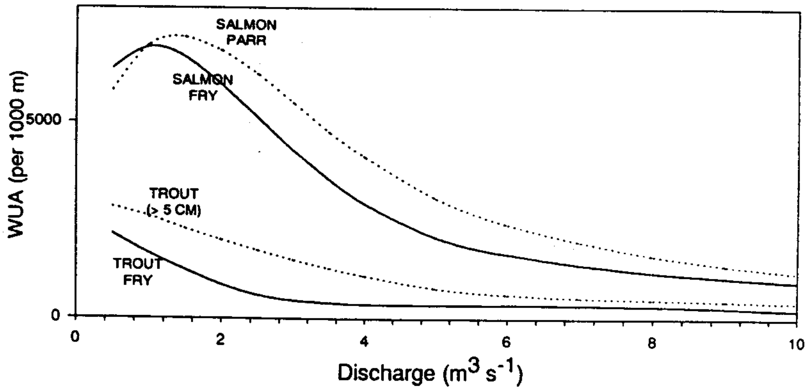
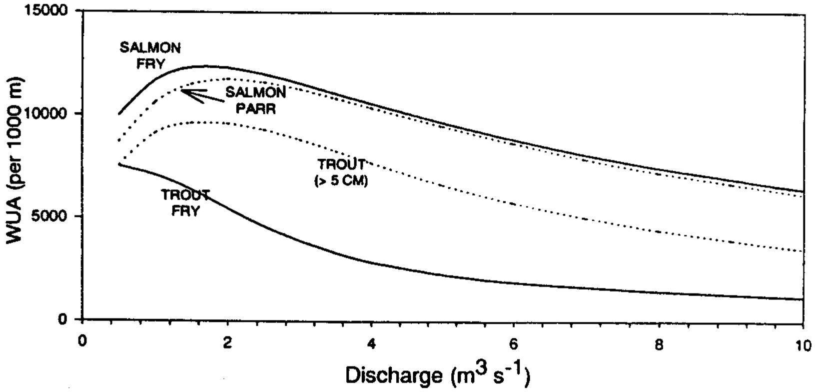


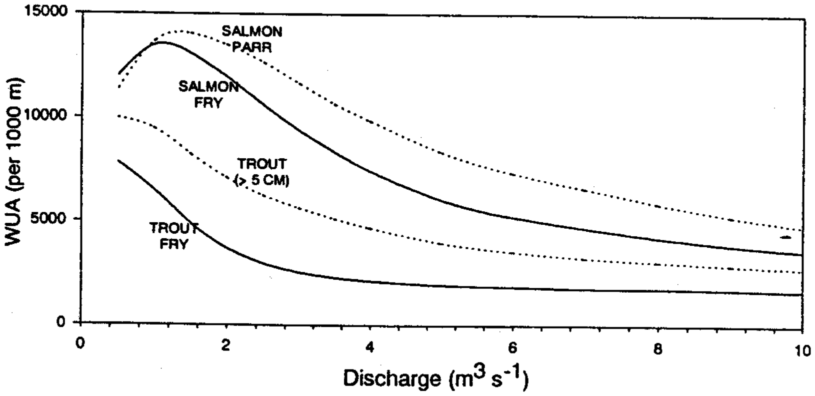
Figure 6. Modelled water depth and mean water velocity suitabilities in riffle habitat for two size-classes of Atlantic salmon and brook trout, and with increasing discharge in Pinchgut Brook fall 1995; a) IFIM, b) EVHA, and c) HABITAT.

For entire study reach

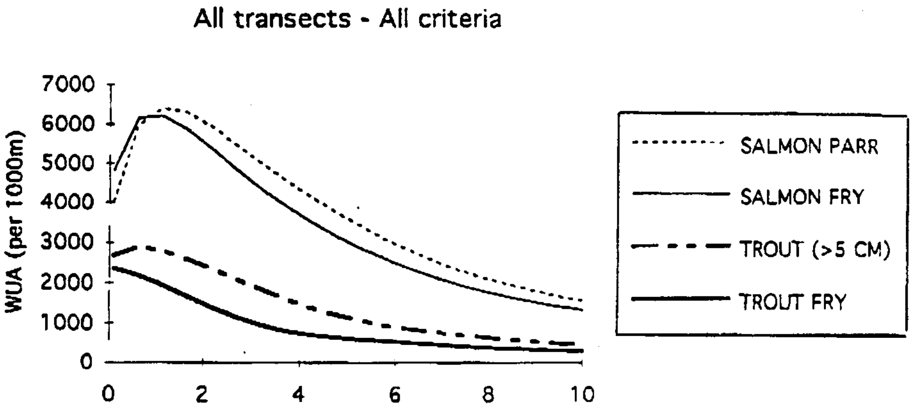
a: IFIM

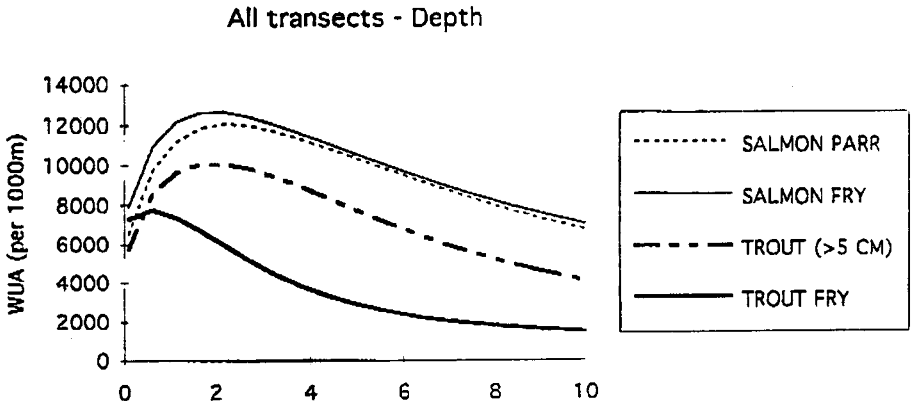

All Transects - All Criteria

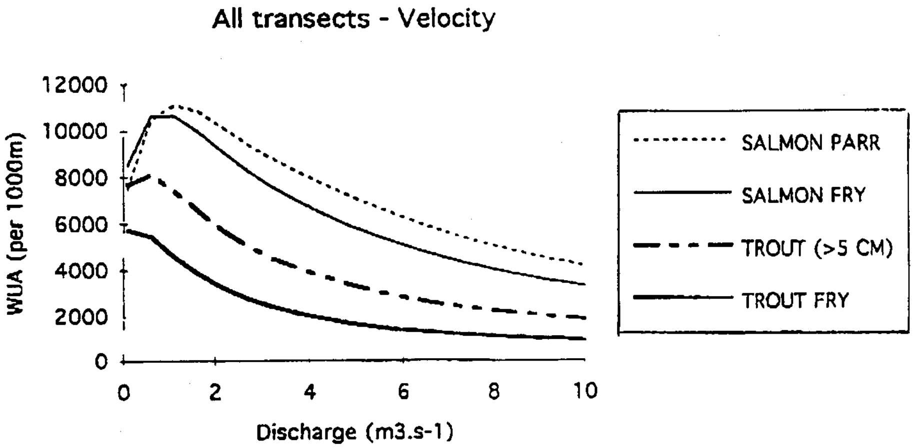

All Transects - Depth


All Transects - Velocity

b: EVHA


All transects - All criteria
7000
6000
5000
4000
3000
2000
1000
0
WUA (per 1000m)
0
2
4
6
8
10
SALMON PARR
SALMON FRY
TROUT (>5 CM)
TROUT FRY


All transects - Depth
14000
12000
10000
8000
6000
4000
2000
0
WUA (per 1000m)
0
2
4
6
8
10
SALMON PARR
SALMON FRY
TROUT (>5 CM)
TROUT FRY


All transects - Velocity
12000
10000
8000
6000
4000
2000
0
WUA (per 1000m)
0
2
4
6
8
10
Discharge (m3.s-1)
SALMON PARR
SALMON FRY
TROUT (>5 CM)
TROUT FRY

c: HABITAT

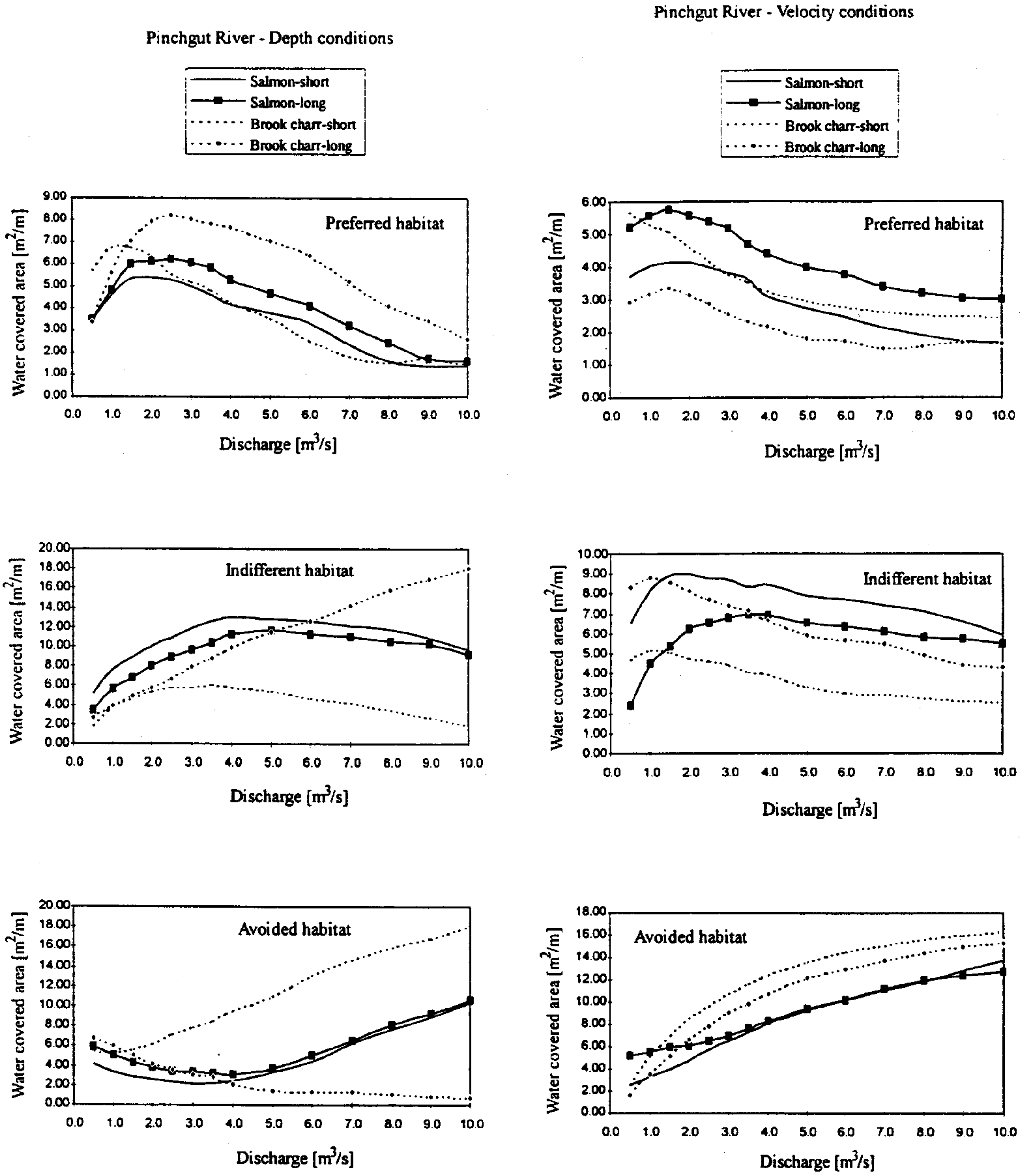


Figure 7. Modelled water depth and mean water velocity suitabilities for entire study reach, for two size-classes of Atlantic salmon and brook trout, and with increasing discharge in Pinchgut Brook fall 1995; a) IFIM, b) EVHA , and c) HABITAT.

DISCUSSION

The IFIM and EVHA may be directly compared as the biological components are the same and the hydraulic components similar. There is however an important difference in the spatial scale and field sampling design in that the IFIM results are based on 2-4 transects per habitat type in different reaches along the entire study reach, and EVHA on 12-15 transects in each of a few selected representative reaches. The trends in WUA for both species and size groups are similar between the two models, indicating considerable robustness in the hydraulic models. Different spatial scales, field sampling design or hydraulic models (IFG4 or stepbackwater model) were not responsible for great changes in WUAs. Therefore, the results suggest that in this study on this type of rivers (quite small, straight, and structurally simple, i.e. suitable for one-dimensional modelling) the results of the two approaches are similar. This may not be true for more complex morphologies (Leclerc et al. 1995). The result may also suggest limited sensivity in the one-dimensional hydraulic models to small-scale morphological structures relevant to fish habitat use.

The 'representative reach' in EVHA is similar to HABITAT and is also one of several field sampling methods that can be employed in an IFIM study. An important caveat to the above conclusion relates to the choice of 'representative reach'. Because of the magnitude of results from our study, we have necessarily limited the comparison to 'combined' results (all transects) and riffle habitats. In all models there were substantial differences in WUA between habitat types. This suggests that (i) it is important to consider the quantity and distribution of habitat types in any practical habitat-hydraulic study and (ii) the selection of a 'representative reach' to model a reach containing different habitats may not be desirable.

The results for the HABITAT model were different. The biological input was based on fish habitat preferences (and not use only as in IFIM and EVHA) and used real weights. This allows for the use of multivariate models and 3-D suitability curves in the biological modelling. Also the presentation of results is different and more complex. However, it provides more information and more comprehensive interpretation of results. In terms of preferred depth and velocity habitat, the HABITAT indicated optimal flow at higher discharges than the WUA-curves from IFIM and EVHA. WUA curves produced by HABITAT were in general similar to those from IFIM and EVHA, but indicated optimum at somewhat higher flows than did IFIM/EVHA. There was, however, an important exception for large salmon. The HABITAT indicated no clear optimum flow for large salmon, it increased and stabilized with increasing discharges. This is most likely to be explained by the differences in biological models. Therefore, biological models seem to be more important in generating differences between IFIM, EVHA, and HABITAT, than hydraulic models.

In rivers with more complex morphologies, two-dimensional hydraulic models have been suggested as more suitable (Leclerc et al. 1995). However, while the required fieldwork and logistics increase considerably from 1D to 2D models, there is little difference between 2D and 3D models in this respect, except for computational time. Three-dimensional models will give much more information using only a little bit more resources than 2D models. We therefore suggest that future improvements in predictive hydraulic modelling will be with three-dimensional models.

Space and time limited the inclusion of results on (i) spatial habitat maps, (ii) time series analysis, and (iii) between habitat comparisons. This will be pursued in future collaborations.

REFERENCES

Baume, J.P and Poirson, M. (1984). Modélisation numérique d'un écoulement permanent dans un réseau hydraulique maillé a surface libre, en régime fluvial. *La houille blanche* 1, pp. 95-100.

Capra, H., Breil, P. and Souchon, Y. (1995). A new tool to interpret magnitude and duration of fish habitat variations. *Regulated Rivers* 10, pp. 281-289.

deGraaf, D.A. and Bain (1986). Habitat use and preferences of juvenile Atlantic salmon in two Newfoundland rivers. *Transactions of the American Fisheries Society* 115, pp. 671-681.

French, R.H. (1986). Open-channel hydraulics. McGraw-Hill Int. Ed.

Ginot, V. and Souchon, Y. (1995). *Logiciel EVHA. Evaluation de l'habitat physiques des poissons en rivière (version 1.0, beta test). Vol. 1. Guide méthodologique. Vol. 2. Guide de l'utilisateur.* Cemagref Lyon BEA/LHQ, France, et Ministère de l'Environment, Dir. de l'Eau, Paris, France.

Harby, A. and Heggenes, J. (1995). Habitat User's Manual. In Alfredsen, K., Bakken, T.H. and Killingtveit, A. (eds.) *The River System Simulator. User's Manual.* SINTEF NHL, report 1995.

Heggenes, J. (1990). Habitat utilization and preferences in juvenile Atlantic salmon in streams. *Regulated Rivers* 5, pp. 341-354.

Heggenes, J., Saltveit, S.J., Vaskinn, K.A. and Lingaas, O. (1994). Predicting fish habitat use responses to changes in waterflow regime: Modelling critical minimum flows for Atlantic salmon (Salmo salar) and brown trout (S. trutta) in a heterogeneous stream. *Proceedings 1st International Symposium Habitat Hydraulics*, Norwegian Institute of Technology, Trondheim, pp. 124-142.

Heggenes, J. (1996). Habitat selection by brown trout (*Salmo trutta*) and young Atlantic salmon (*S. salar*) in streams: Static and dynamic for hydraulic modelling. *Regulated Rivers* (In print).

Hill, J. and Grossman, G.D. (1993). An energetic model of microhabitat use for rainbow trout and rosyside dace. *Ecology* 74, pp. 685-698.

Hughes, N.F. (1992). Selection of positions by drift-feeding salmonids in dominance hierarchies: Model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. *Canadian Journal of Fisheries and Aquatics Sciences* 49, pp. 1999-2008.

Jacobs, J. (1974). Quantitative measurement of food selection: a modification of the forage ratio and Ivlevs electivity index. *Oecologia* 14, pp. 413-417.

Leclerc, M., Boudreault, A., Bechara, J.A. and Corfa, G. (1995). Two-dimensional hydrodynamic modelling: a neglected tool in the instream flow incremental methodology. *Transactions of the American Fisheries Society* 124, pp. 645-662.

Morantz, D.L., Sweeney, R.K., Shirvell, C.S. and Longard, D.A. (1987). Selection of microhabitat in summer by juvenile Atlantic salmon (*Salmo salar*). *Canadian Journal of Fisheries and Aquatic Sciences* 44, pp. 120-129.

Modde, T. and Hardy, T.B. (1992). Influence of different microhabitat criteria on salmonid habitat simulation. *Rivers* 3, pp. 37-44.

Olsen, N.R. and Stokseth, S. (1995). Three-dimensional numerical modelling of water flow in a river with large bed roughness. *IAHR Journal of Hydraulic Research* 33.

Orth, D.J. (1987). Ecological considerations in the development and application of instream flow-habitat models. *Regulated Rivers* 1, pp. 171-181.

Pouilly, M., Valentin, S., Capra, H., Ginot, V. and Souchon, Y. (1995). Note technique: Methode des microhabitats, principes et protocoles d'application. *Bulletin Francais de Pêche et Pisciculture* 336, pp. 41-54.

Stalnaker, C.B., Lamb, B.L., Henrickson, J., Bovee, K.D. and Batholomew, J. (1994). The Instream Flow Incremental Methodology: A Primer for IFIM. National Ecology Reserach Centre, National Biological Survey, Fort Collins, Colorado. 99 p.

Trocherie, F. (1987). *Mise en application de l'article 410 du Code Rural. Elément d'une démarche de détermination des débits réservés.* CEMAGREF-DQEPP/DPN 85-15.

US Army Corps of Engineers (1991). HEC-2. Water Surface Profiles, User's Manual. Hydraulic Engineering Centre, US Army Corps of Engineers, Davis, California, USA.

Valentin, S., Lauters, F., Sabaton, C., Breil, P. and Souchon, Y. (1996). Modelling temporal variations of physical habitat fro brown trout (Salmo trutta) in hydropeaking cinditions. *Regulated Rivers* 12 (In press).

Wiens, J.A. (1989). Spatial scaling in ecology. *Functional Ecology* 3, pp. 385-397.

METHODOLOGY FOR FISH ASSEMBLAGES HABITAT ASSESSMENT IN LARGE RIVERS. APPLICATION IN THE GARONNE RIVER (FRANCE).

Marc POUILLY (1)
Yves SOUCHON (2)
Yann LE COARER (3)
Daniel JOUVE (4)

(1) ORSTOM - Université LYON 1, Bât 403, 43 Bd du 11 Novembre 1918, 69622 VILLEURBANNE - France ; pouilly@lyon.cemagref.fr
(2) CEMAGREF 3bis, quai Chauveau CP 220 69336 LYON Cedex 09 - France ; yves.souchon@cemagref.fr
(3) CEMAGREF Le Tholonet BP 31 13612 Aix-en-Provence Cedex 1- France ; yann.lecoarer@cemagref.fr
(4) CNR DXHY 2, rue André Bonin 69316 LYON Cedex 4 - France

ABSTRACT

The purpose of this paper was to present a concrete example of habitat simulations for the fish assemblage in the Garonne river (average discharge 200 m3/s) near Toulouse (France). This example reveals how the technical problems of adaptating habitat simulation methodologies in large rivers has been solved. The first of three points discussed is a description of a new tool for acquiring hydraulic data on a non-wadable river, that is compatible with previous data calculation used in classical fish habitat simulation (Phabsim, EVHA). In this case, an Acoustic Current Doppler Profiler ™ was successfully used to measure depths and velocities on cross-sections ; it was limited to a minimum water depth of 0.8 m, due to the 1200 kHz frequency used in this study. Secondly, this study illustrates the linkage of physical data at different flows with multivariate fish habitat models. Global habitat models were available for 13 lifestages from 5 species (representing 70 % of density of the total assemblage) : barbel (*Barbus barbus*), gudgeon (*Gobio gobio*), chub (*Leuciscus cephalus*), bleak (*Alburnus alburnus*) and roach (*Rutilus rutilus*). Thirdly, the authors propose an interpretation for habitat at two levels : a classical separate target species interpretation or a new procedure mixing several stages and species, that compares the difference between predicted fish assemblage with reference fish assemblage. We show that it is possible to simulate the potential habitat for the whole fish assemblage with individual species models. The relative weak number of species in french rivers, due to historical biogeography, makes it possible. We also propose an approach that simulates the physical behaviour of the stream on vestiges of morphological units, which thus provides potential rehabilitation. The habitat simulation agrees qualitatively with the known habitat use of the species. The range of flows which seems to structure fish assemblage on an annual scale approximates the interannual average minimum flow. We also discuss the different objectives for river preservation or rehabilitation, depending on the actual ecological status of the hydrosystems.

KEY-WORDS : large river, extension of Phabsim methodology, Garonne river (France), fish assemblage, ecological status, fish habitat, Acoustic Current Doppler Profiler ™, EVHA software.

INTRODUCTION

During the last quarter of this century, a greater awareness developed of the necessity to manage riverine ecosystem in a more holistic manner (Petts *et al.*, 1989 ; Rabeni, 1992). The dominant reductionist philosophy, which promotes a case by case management of lotic hydrosystems, is contring the holistic management. However, the application of integrated management remains problematic for several reasons : 1) there is insufficient knowledge on the dynamic functioning of hydrosystems ; 2) integrated management these evaluates ecological impacts of water development with less precise quantification than in more traditional engineering ; 3) policy makers lack a sound comprehension of integrating ecological requirements. There is a need for more comprehensive methodologies dealing with symbolic biological targets like fish, which could become a measurable "witness" of the efficiency of the management options.

The simulation of the change in fish habitat as a result of flow changes are example of how integrated management can be achieved. This became a major challenge for ecohydraulics research starting in 1975, when permanent conflicts on quantitative water use and relicensing hydroelectric powerplants arose in several countries. Tools which combine hydraulic simulation and fish habitat requirements have preceded safe theory and knowledge on fish habitat and population dynamics (Gaudin et al., 1995). Nevertheless, people working on minimum flow assessment greatly appreciate methodologies widely used (Reiser *et al.*, 1989 ; Armour and Taylor, 1992) like IFIM including PHABSIM (Stalnaker, 1979 ; Bovee, 1982 and 1986) and clones (RSS, Killingtveit and Fossdal 1994 ; EVHA, Ginot 1995). They accepted temporally the coarsness of the tool and their helped by their concrete questions to improve several points of the methodology. As pointed out by Stalnaker (1993) and others (Orth, 1987 and 1995; Pouilly and Souchon, 1995), more effort must be made towards including biological explanations and the validity of fish habitat simulation. In this field, the prediction of the spatial well being of complex fish assemblages in large rivers appeared to be specially difficult. This is complicated by inadequate physical description of rivers, where cost effective field data collection (depth, velocity and substrate) are difficult such as in non-wadable rivers.

To deal with diverse fish assemblages, different solutions have been proposed :

- selection of limited target fish species (Bovee, 1986) in a wide range of habitat needs, representative of major habitat-use guilds (Leonard and Orth, 1988). Minimum flow assessments could be based on the intermediate guild between edge-dwelling and fast-water guilds (Jowett and Richardson, 1995),
- selection of species on the basis of economical (salmon, brown trout) or symbolic consideration (Hippopotamus amphibius) (Gore *et al.* 1992).

The difficulty with numerous species and life stages is to objectively choose the weighting criteria between them. One needs choose a good compromise between the needs of all species ? Leonard *et al.* (1987) proposed for rivers the flow that provides the maximum habitat for the most critical limited habitat of all the species lifestages throughout the year. This could be acceptable for a case expertise, but we still lack of validation criteria, in particular what habitat threshold or duration act as bottlenecks responsible for temporal building of the fish assemblages ?.

This paper presents a concrete example of habitat simulations for the fish assemblage in the Garonne river near Toulouse (France), which addresses the problems of adaptation of Phabsim types of methodologies in large rivers. Three points were presented :

(1) description of a new tool for acquiring hydraulic data on a non-wadable river, that is compatible with previous data calculation used in classical fish habitat simulation (Phabsim, EVHA),

(2) illustration of the linkage of physical data at different flows with multivariate fish habitat models,

(3) proposals an interpretation for habitat at two levels : a classical separate target species interpretation or a new procedure mixing several stages and species, that compares the difference between predicted fish assemblage with reference fish assemblage.

METHODS

Site description :

Garonne is the major river in South West of France. The head of the basin takes place in Pyrénées mountains. Upstream from Toulouse, a large tributary called Ariège joined Garonne. After this confluence, the mean annual discharge is 200 m3/s (Sormail 1989), and the river exhibits a nival hydrological regime. Several upstream dams regulate the discharge and provide irrigation waters. As a result, the low flow during the summer for 10 consecutive days declines to 40 m3/s (Sormail 1989). There is only one channel in this area.. Downstream from Toulouse, dredging activities during the last 30 years lowered the river level from its historical less of 2 meters. Most of the alluvial substrate has desappeared, and the natural sequence of morphological units has been modified. The river bed is now dominated by molassic paving, and the typical riffle units have been largely eliminated. It is in therefore remaining areas that we choose to analyze. This allows us to infer what could be the effect of susbtrate and discharge restoration.

Habitat description survey

Two replicate samples sites were established in the first 10 km downstream from Toulouse. These site locations were chosen to represent a maximum of habitat conditions and substrate in the remaining natural areas. The general procedure is the same as the transect description procedure described in Pouilly *et al.* (1995, adapted from Bovee 1982). The mean water depth was more than 0.8 m, which permited us to use of the Acoustic Current Doppler Profiler (ADCP, RD Instruments compagny ™, San Diego California-USA) based on water depth and current velocity. Six transects were chosen in site 1 and seven in site 2. Each transect was performed perpendicular to the major direction of the flow. They were described by ADCP during 4 surveys. For an average of 100 m width, it took approximatively five minutes for one boat survey.The margin of the transect was localised precisely by a laser theodolite, and ADCP was able to determine the boat trajectory with an integrate compass and an estimation of the boat speed. When the

ADCP could not measure the margin of the transect because of too shallow water depth, then we used the propeler method for determining velocity and depth (OTT-C2 propeler ™). Each transect was described by several cells defined by upstream and downstream representative limits and by a mean width of 1 meter (100 ± 10 cells for a width between 100 and 120 meters). Each cell gave the local hydraulic conditions corresponding to the measurements taken by the current propeler or the mean of the four survey done by the ADCP. This procedure gave the topographic and hydraulic information necessary to use EVHA software to evaluate fish habitat (Ginot and Souchon 1995).

How the ADCP works (Gordon 1989)

The gear is composed of a cylinder (diameter 22 cm, height 60 cm, weight 23 kg) which supports four transducers, and by an electronic interface (deckbox) compacted to a computer. The four transducers were oriented at an angle of 30° relative to the vertical and in 90° azimuth increments. The transducer cylinder is installed vertically on the side of a boat and kept enough (around 15 cm) to ensure that bubbles do not get under the transducer faces when the boat is underway.

ADCPs transmit short acoustic pulses along narrow beams on one frequency. Several gears exist, each with one frequency adapted to one water depth range. Lower frequencies produce greater range whereas higher frequencies produce finer resolution. We used the frequency of 1200 kHz which is adapted to a water depth of 0.8 to 20 meters. The ADCP processes and records the echoes from successive volumes along the beams to determine how much the frequency has changed. The difference in frequency between transmitted and reflected sound is proportional to the relative velocity between the ADCP and the small particules in the water that reflect sound (plankton, suspended sediment). This frequency shift results from the Doppler effect. Each transducer provides a measure of water depth and computes velocity in a series of contiguous cells (one every 25 cm depth). Using four transducers produces a three dimensional velocity vector for each cell and cross check the data quality. The software associated with the deckbox produces real time results which allowed us to determine that the ADCP was working correctly.

The hydraulic model

Calculations of depth and velocity for a wide range of flows was made using an hydraulic model based on Limerinos formula (Ginot and Souchon 1995 ; see also Pouilly *et al.* 1995 for a detailed description of the model). This model is available on EVHA software developped at Cemagref Lyon (Ginot and Souchon 1995).

Estimating species lifestages habitat

Habitat models were available for 13 lifestages from 5 species of the assemblage, all of which were examined (Table 1). They represent 70 % of density of the total assemblage (Pouilly and Souchon 1995), and correspond to the available fish habitat models. Barbel (*Barbus barbus*) and gudgeon (*Gobio gobio*) are benthical and rheophilic species. Chub (*Leuciscus cephalus*) and bleak (*Alburnus alburnus*) are ubiquitous for velocity. Roach (*Rutilus rutilus*) is a lenitophilic species. The species were divided in two or three lifestages based on the natural breaks of size frequency histograms.

The fish habitat models were developed first in the Rhône basin (667 microzones, Pouilly, 1995). One hundred and twenty seven electroshocked microzones from the Garonne river were also available (Pouilly and Souchon, 1995). In order to offer a broader range of habitat conditions sampled, we have decided that it was better to pool the two data bases and to work with general habitat models for the 794 microzones,

Habitat simulation for one discharge and one species lifestage was computed using two types of information :

- the local values of current velocity and water depth, derived from the hydraulic model, and substrate description,
- coefficients from the multiple regression between species lifestage density and values of current velocity, water depth and substrate. This biological model is based on multivariate coefficients and has been developped following the procedure described by Pouilly (1995).

Habitat expressed as Potential Usable Area, was then estimated by the formula :

$$PUA(j) = \Sigma\ Ai * (\ C(j,d)*Di + C(j,v)*Vi + C(j,s)*Si + Cste(j)\ )$$

with :

PUA(j) = Potentially Usable Area for the species lifestage j

Ai = Area of cell i of a transect of a station

Di, Vi, Si = value of depth (d), current velocity (v) and substrate (s) in cell i

C(j,d), C(j,v), C(j,s) = regression coefficients of depth (d), current velocity (v) and susbtrate (s) for the species j

Cste(j) = regression intercept

PUA(j) could be summed by transect or by station. Because of standardized densities in the regression, PUA corresponded to density coefficient rather than to real densities. The PUA were estimated for different discharges, furnishing an evolution of the potential habitat for the lifestage species versus discharge.

General interpretation at the assemblage level

The evolution of the potential habitat versus discharge could be estimated for each of all the lifestages of the 5 principal species of the assemblage. It resulted many different shapes of habitat evolutions stage by stage, which compromised a simple interpretation of all these informations.

An actual reference of the fish assemblage was established in a more natural reach of the river 10 km upstream from Toulouse (Pouilly and Souchon 1995). One possible interpretation on potential habitat was to search which discharge in the two replicate sites produced the closest estimated assemblage to the reference assemblage. By doing this, we hypothesized that habitat is a major determinant for shaping fish assemblages in flowing waters (Horwitz, 1978 ; Jowett, 1992 ; Souchon, 1994).

The proposed interpretation was estimated by the sum of the square of the difference between the estimated and reference proportion of each lifestage species in the assemblage (Table 1) :

$$Prox(Q) = \Sigma\ (Pe(Q,j)-Pr(j))^2$$

with :

Prox(Q) = proximity between the estimated and reference assemblage for the discharge Q

Pe(Q,j) = relative proportion of the species lifestage j in the estimated assemblage for the discharge Q

Pr(j) = relative proportion of the species lifestage j in the reference fish assemblage of the Garonne.

RESULTS

Physical description versus discharge for each site

Figure 1 gives an example of depths and current velocities from the four ADCP surveys on a transect. The results were consistant and it was possible to average data for each elementary cell.

The hydraulic modeling was performed between 20 and 300 m3/s. For the two sites, this range of discharge included severe low flow and the mean annual discharge (Sormail 1989). Table 2 shows the principal characteristics of the stations for these discharges.

Site 1 was 415 meters long and was analysed by six transects. The mean channel slope was 0,0001. Transects 1, 2, 5 and 6 correspond to fast, shallow channel, whereas transects 3 and 4 were deeper and slower. This station was straight and this corresponded to a symetrical transverse section for all the transects.The water depth and the wetted area increased continuously with increased discharge, with a slow and down after reaching 85 m3/s.

Site 2 was 610 meters long,, with a mean slope was 0.0005. It was analysed by 7 transects. Transect 7 described a long downstream riffle, with a slope of 0,0011. This riffle constituted a sill which made the upstream deeper and slower. Transects 1 and 6 represented a fast flowing channel, whereas transects 2 to 5 were deeper and slower. Station 2 corresponds to a river curve with a symetrical transects. The water depth and the wetted area increased quickly from 20 to 100 m3/s, but remained stable because of a steepside transversal section.The slope and the steeps of station 2 had faster velocities than in station 1 (Table 2).

Species lifestages' potential habitat

The shape of the curves of PUA versus discharge was quite similar in the two stations for the 13 lifestages (fig. 2a and b). The habitat conditions for small fish, lifestages 1 (except for roach), decreased with increasing discharge. The maximum PUA for these lifestages were between 20 and 50 m3/s. The habitat conditions for lifestages 2 and 3 of bleack and barbel, and the lifestage 3 of chub and roach, increased continuously with increasing discharge and sometines decreased round about 100 m3/s. Their maximum PUA corresponds to 300 m3/s (maximum simulated discharge). The habitat conditions for lifestages 2 of chub were the same as the lifestages 1 results. The habitat conditions of lifestage 1 and 2 increased continuously with discharge at station 1, and decreased (after 100 m3/s for lifestage 1 and after 60m3/s for lifestage 2) in station 2, which was faster than station 1 (Table 2).

Higher discharge rates in station one favored large individuals of chub and roach, whereas higher discharge in station 2 favored lifestage two of barbel.

Species lifestages' potential habitat for a remaining riffle

Lifestages 1 of all species and lifestages 2 of roach and chub had a maximum PUA at discharges between 40 and 55 m3/s, which decreased quickly (fig. 3). This favored lifestages 2 and 3 of barbel (the more rheophilic species), which agrees with what is known about the habitat behaviour of the species, especially for the feeding areas (Baras, 1982).

General interpretation at the assemblage level

The difference between the estimated and the reference assemblage was very typical for the riffle of station 2. It shows a large decreased (closer to the reference assemblage) between 35 and 60 m3/s (fig. 4). This difference was similar, but unpronounced for all of station 2. This contrasts with station 1 where the difference increased continuously with increased discharge.

DISCUSSION

Hydraulic data

This study provides an example of how the hydraulic description of large river may be described with the Acoustic Doppler Current Profiler. This method provides fast and practical way to obtain sound data (fig. 1). The major problem met with this technique is the choice of the site locations. The water depth limitation (min. 80 cm) makes it difficult to represent all habitat conditions. As a result, riffles and shallow areas could be underestimated. Only one riffle could be described (transect 7 station 2), which represented fast velocity conditions and a natural morphological unit. Current research tries to improve shallow water measurements in this process. The ADCP is usefull to produce the elementary geometry for basic 1D hydraulic simulation in Phabsim family softwares. The ADCP better represents the longitudinal dimension because of the rapidity of the measurements. It appearss also compatible with new 2D hydraulic modeling (Leclerc *et al.* 1995) and offers the possibility to check physical parameter simulations at different discharges. This method buttnecises Stalnaker *et al.* (1989) optimism, that physical sampling is feasable in non-wadable rivers.

Fish habitat models and simulations

Poor data quality and non availability of fish habitat models could limit the application of Phabsim methodologies. For example, Gore and Nestler (1988) recommended working exclusively with regional based fish habitat developed on a regional basis . Therefore in the Garonne river, we used general models with aggregated data from two different geographical basins. For practical management, it is preferable to quickly mobilise the best available knowledge. In this case, we used a general habitat model that produces interpretable and useful results in terms of potential habitats' relationships to flow. However, it is not possible to develop fish habitat model in each Phabsim application. Nevertheles, general habitat models can also yield coherent predictions of fish population in a wide range of hydroecoregions (Brown trout, *Salmo trutta fario*, France ; Souchon, 1994).

We used multivariate models to counter criticisms about the non-independance of key variables (Mathur *et al.*, 1986). In our research, the models consider average physical conditions for velocity, depth and substrate in sampled habitat zones. We believe that the models could be further improved by introducing (1) the diversity of local adjacent habitat, (2) such parameters like cover, which can be treated with multivariate approaches. It is also obvious that the progressive incorporation of complexity and marginal habitat use by fish (Conder and Annear 1987) will improve the models' accuracy.

The lifestages' evolutions of potential habitat were similar for all the small fish, while the larger fish had different evolutions corresponding to their own habitat preferences. However in their early development, all these species have the same habitat preference for shallow and slow water (Copp, 1989 ; Poizat, 1993 ; Pouilly, 1994). Larger adult fish, such as barbel, more rheophilic of these species, preferedhad habitat conditions in the faster locations and especially in the riffles. This contrasts with the more lenitophilic species such as roach, which favours in the slower locations. The others two species (chub and bleack) are more ubiquitous. Habitat conditions for the bleack are the same for both stations and are in the middle scale between roach and barbel. Habitat condition for lifestage 3 of the chub had the same evolution as the roach. The habitat simulation agrees qualitatively with the known habitat behaviour of the species.

Definition of river management goals and associate habitat simulation strategy

The more precise and explicit the management objectives, the easier it is for scientist to provide sound reseach data to resource managers. The ecological status of the river is the major criteria to consider, given the growing inclusion of environmental goals in national legislation. More precedence is given to purely fisheries considerations. A usefull typology can be described in four categories :

(1)- Excellent ecological status, where a preserved hydrosystem is in a quasi pristine state with presence of unique biota. Protection rules would need to be very strict. Total perservation is the goal ;

(2)- Reversible ecological status, where the river can still be restored to approximate original conditios. This would require restoration to recover an original situation ;

(3)- Degraded ecological conditions, where original conditions cannot be restored. The river could be rehabilitated to a better ecological status ;

(4)- Severely degraded ecological status, where the most that can be done is prevent the system from further critical degradation.

For this typology, it is important to have a good knowledge on the fish assemblage reference sensu National (USA) Research Council (1992) to evaluate the potential of restoration and to provide a point of comparison to judge the effectiveness of the management plan. This reference could be historical (unfortunately in very few situations), or existing preserved reaches in a river or a similar regional stream.

In this study no historical fish assemblage reference exists. The comparison was made with an actual reference to an existing reach with active morphodynamic and unaltered substrate. A more degraded dowstream reach of the Garonne river was impacted by a project of flow regime modification, therefore we selected replicate habitat study stations in the less degraded reach in order to offer two possible interpretations : the potential of rehabilitation through morphological improvements (i.e. substrate refill) and the potential of rehabilitation through flow regulations, which will not degrade the actual state. This does not allow usto compare observed fish assemblage with simulated potential habitat.

For the choice of the target fish species, we propose an interpretation at the total assemblage level. Nevertheless, others biological models have to be developed to complete the actual possibility with 13 lifestages and species, for instance 70 % of the total average fish density. We think that it is both possible and desiderable to develop future fish habitat models for a maximum number of species, which is therefore considered as an utopia by Bayley & Li (1992) for tropical area.s. In temperate areas such as Europe, the relative low number number of fish species, makes perhaps this an achievable goal. If a model focusses on fish species with known habitat requirements, such as salmonids, the importance of the rare, endangered, ecological key species (Thyus and Karp, 1989), could be underestimated. In assemblages with numerous species, the fish habitat guild procedures appear to offer a good compromise (Leonard and Orth, 1987 ; Bain and Boltz, 1989).

The fish habitat interpretation at the assemblage level

More biology (Stalnaker, 1993) or a better ecological view (Orth, 1987 and 1995 ; Pouilly and Souchon, 1995) in interpretation of fish habitat requires developments of new weightings factors between the different lifestages and species of fish along with an integrated representation of river response versus flow. Because biological models were not developed on a seasonal pattern and because we did not analyze the hydrologic cycle, we compared the results to a global observed result (i.e. the mean assemblage composition). Leonard *et al.* (1986) used an optimization matrice to identify the flow that provides the maximum habitat for the most critically habitat limited lifestage. This procedure assigns the same weight for each species/lifestages, regardless of dominance. As the result, we used the reference assemblage for comparisons and we searched for the discharge which gave the most closed estimated assemblage. This procedure compared the results of all the species lifestages. The most interesting results showed that (1) only the remaining morphological unit produced a predicted assemblage closest to the reference one and (2) the pluriannual average monthly low flow produced the best approximation of the global structure of the fish assemblage. This record for the Garonne cyprinids fish assemblage, is similar to previous observations on salmonids communities, especially brown trout (Souchon *et al.*, 1989 ; Jowett, 1992).

Our procedure presuppose that all the the biological models have a similar accuracy. If one of them was less accurate, the estimation error made on one species lifestage would affect the estimation on the assemblage. Table 1 showed the accuracy of the models used. Additionally some portions of the stream may have physical conditions at high flows very different from the conditions where the habitat models were validated. According to Gore and Nestler (1988), error is introduced when habitat changes are projected for discharges that are different from those used when the suitability curve was developed. A sensitivity analysis could help to avoid missinterpretation, by checking different stages during the simulation procedure (i.e. depths or velocities ranges) ; this is possible when using some convivial softwares (EVHA ; Ginot, 1995).

It is possible to describe the physical conditions of fish habitat in large non-wadable rivers. The future challenge is to interprete habitat simulation in a broader ecological perspective. We have shown that need to be addressed questions based on definition of management objectives, as well as how to choose a reference that provides a sound interpretation of the potential fish habitat versus flow. Progress must be made in validating biological models for dynamic comprehension (i.e. habitat times series) (Capra *et al.* 1995 ;Pouilly and Souchon, 1995), interpretating habitat results, and validating the biological use of habitat simulation methodologies.

ACKNOWLEDGEMENTS

We thank Professor Belaud and all his staff from Ecole Nationale Supérieure Agronomique de Toulouse and P. Roger from Cemagref for helping with the collection of biological data.. Doug Mc Lain and Sylvie Valentin help us to deeply revise the english langage of a previous draft. This study was partially funded by the Agence de l'eau Adour- Garonne. We thank particularly F. Simonet, M.Roux and R. Lohou for encouraging an experimental methodology. The study is a contribution to a scientific program of Cemagref titled " hydrodynamic and ecology in running waters ".

REFERENCES

Bain M.B., Boltz J.M. (1989). *Regulated streamflow and warmwater stream fish : a general hypothesis ana research agenda.* U.S. Fish and Wildlife Service, Alabama : Biological report 89 (18).

Bayley P.B., Li H.W. (1992). Riverine fishes. *In* : Calow P. & Petts G.E. (eds), *The rivers handbook.* Blackwell Scientific Publications, Oxford pp 251-281.

Bovee K.D. (1982). *A guide to stream habitat analysis using the Instream Flow Incremental Methodology.* U.S.D.I. Fish and Wildlife Service, Office of Biological Services, Fort Collins, Colorado. : Instream Flow Information Paper n°12, FWS/OBS 82/86 : 248 p.

Bovee K.D. (1986). *Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology.* Fish and Wildlife Service, Office of Biological Service, Ft Collins : Instream Flow Information Paper n° 21, FWS/OBS 86/7 :188p.

Capra H., Valentin S., Breil P. (1995). Chroniques d'habitat et dynamique des populations de truite. *Bull. Fr. Pêche et Piscic.* 337-338-339 pp. 337-344.

Conder A.L., Annear T.C. (1987). Test of weighted usable area estimates derived from a PHABSIM model foi instream flow studies on trout streams. *North American Journal of Fisheries Management.* 7 pp. 339-350.

Copp G.H. (1989). The habitat diversity and fish reproductive function of floodplain ecosystems. *Environmental Biology of Fishes.* 26 pp. 1-27.

Gaudin P., Souchon Y., Orth D.J., Vigneux E. (1995). *Habitat poissons. In* : Colloque "Habitat poissons", Villeurbanne. *Bull. Fr. Pêche et Piscic.* 337-338-339 pp 1-418.

Ginot V. (1995). EVHA : Un logiciel d'évaluation de l'habitat du poisson sous Windows. *Bull. Fr. Pêche ei Piscic.* 337-338-339 pp. 303-308.

Ginot V., Souchon Y. (1995). *Logiciel EVHA. Evaluation de l'habitat physique des poissons en rivière (version 1.0, beta test).Vol. 1. Guide méthodologique. Vol. 2. Guide de l'utilisateur.* Cemagref BEA/LHQ Lyon, France, et Ministère de l'Environnement, Dir. de l'Eau, Paris, France.56 p et 109 p. + annexes.

Gordon R.L. (1989). Acoustic measurement of river discharge. *Journal of Hydraulic Engineering.*115 (7) pp. 925-936.

Gore J.A., Layzer J.B., Russel I.A. (1992). Non traditional applications of instream flow techniques foi conserving habitat of biota in the Sabie River of southern Africa. *River conservation and Managemeni* 11 pp. 162-177.

Gore J.A., Nestler J.M. (1988). Instream flow studies in perspective. *Regulated Rivers: Research & Management.* 2 pp. 93-101.

Gore J.A., Nestler J.M., Layzer J.B. (1989). Instream flow predictions and management options for biota affected by peaking-power hydroelectric operations. *Regulated Rivers: Research & Management.* 3 pp. 35-48.

Horwitz R.J. (1978). Temporal variability patterns and the distributional patterns of stream fishes. *Ecological Monographs*. 48 pp. 307-321.

Jowett I.G. (1992). Models of the abundance of large brown trout in New Zealand rivers. *North American Journal of Fisheries Management*. 12 pp. 417-432.

Jowett I.G., Richardson J. (1995). Habitat preferences of common, riverine New Zealand native fishes and implications for flow management. *New Zealand Journal of Marine and Freshwater Research*. 29 pp. 13-23.

Killingtveit A., Fossdal M.L. (1994). *The River System Simulator - An integrated model system for water resources planning and operation. In* : Hydrosoft 94, 21 - 23 september, Porto Carras, Greece : pp 1-8.

Leclerc M., Boudreault A., Bechara J.A., Corfa G. (1995). Two-dimensional hydrodynamic modeling : a neglected tool in the instream flow incremental methodology. *Transactions of the American Fisheries Society*. 124 (5) pp. 645-662.

Leonard P.M., Orth D.J. (1988). Use of habitat guilds of fishes to determine Instream Flow requirements. *North American Journal of Fisheries Management*. 8 pp. 399-409.

Leonard P.M., Orth D.J., Goudreau C.J. (1986). *Development of a method for recommending instream flows for fishes in the upper James River, Virginia.* 152, Blacksburg : 122 p.

Mathur D., Bason W.H., Purdy E.J., Silver C.A. (1985). A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences*. 42 pp. 825-831.

National Research Council (1992). *Restoration of aquatic ecosystems*. National Academic of Science, National Academaic Press, : 552 p.

Orth D.J. (1987). Ecological considerations in the development and application of instream flow habitat models. *Regulated Rivers: Research & Management*. 1 pp 171-181.

Orth D.J. (1995). Food web influences on fish population responses to instream flow. *Bull. Fr. Pêche Piscic.* 337/338/339 pp. 317-328.

Poizat G. (1993). *Echelle d'observation et variabilité des abondances de juvéniles de poissons dans un secteur aval du Rhône*. Thèse de doctorat, Université LYON I : 155p.

Pouilly M. (1994). *Relations entre l'habitat physique et les poissons des zones à cyprinidés rhéophiles dans trois cours d'eau du bassin rhodanien : vers une simulation de la capacité d'accueil pour les peuplements.* Doctorat, Université Claude Bernard-Lyon I - BEA/LHQ CEMAGREF : 256 p.

Pouilly M., Valentin S., Capra H., Ginot V., Souchon Y. (1995). Note technique : Méthode des microhabitats, principes et protocoles d'application. *Bulletin Français de Pêche et de Pisciculture*. 336 pp 41-54.

Pouilly S., Souchon Y. (1995). Méthode des microhabitats : validation et perspectives. *Bull. Fr. Pêche Piscic.* 337/338/339 pp. 329-336.

Rabeni C.F. (1992). Habitat evaluation in a watershed context. *American Fisheries Society Symposium*. 13 pp. 57-67.

Reiser D.W., Wesche T.A., Estes C. (1989). Status of instream flow legislation and practices in North America. *Fisheries*. 14 (2) pp. 22-29.

Sormail L. (1989). Besoins en eau et politique de soutien d'étiage dans la vallée de la Garonne. *Revue Géographique des Pyrénées et du Sud-Ouest*. 60 (4) pp. 535-548.

Souchon Y. (1994). Etat d'avancement des recherches sur la modélisation de l'habitat des poissons en France. *Bull. Fr. Pêche Piscic.* 332 pp. 57-71.

Souchon Y., Trocherie F., Fragnoud E., Lacombe C. (1989). Les modèles numériques des microhabitats des poissons : application et nouveaux développements. *Revue des Sciences de l'Eau*. 2 pp. 807-830.

Stalnaker C.B. (1979). The use of habitat structure preferenda for establishing flow regimes necessary for maintenance of fish habitat. *In* : Ward, J.V. &Stanford, J.A. (eds), *The ecology of regulated streams*. New York : pp 321-337.

Stalnaker C.B. (1993). Fish habitat evaluation models in environmental assessments. *In* : Hildebrand, S.G. & Cannon, J.B. (eds), *Environmental analysis. The NEPA experience.* Lewis, Boca Raton, Florida : pp 140-162.

Stalnaker C.B., Milhous R.T., Bovee K.D. (1989). Hydrology and hydraulics applied to fishery management in large rivers. *Canadian Special Publication Fisheries Aquatic Sciences*. 106 pp 13-30.

Tyus H.M., Karp C.A. (1989). *Habitat use and streamflow needs of rare and endangered fishes, Yampa River, Colorado*. U.S. Department of the Interior, Fish and Wildlife service : 27p.

Waters B.F. (1976). A methodology for evaluating the effects of differents streamflows on salmonid habitat. *In* : Orsborn, J.F. &Allman, C.H. (eds), *Instream Flow Needs*. American Fisheries Society, Western Division, Bethesda, Maryland : pp 254-266.

Table 1 : Species life-stages relative proportion in Garonne reference assemblage (Pouilly and Souchon 1995) and multiple regression coefficients and characteristics

Species life-stage	size range (mm)	relative proportion	depth coefficient	velocity coefficient	substrate coefficient	regression intercept	correlation coefficient	regression slope
Bleack 1	[30 - 80]	15.4	-0.00025	-0.00235	-0.02775	0.26509	0.866	0.746
Bleack 2	]80 - 120]	0.1	0.00215	0.0007	-0.01912	-0.038	0.956	0.918
Bleack 3	]120 -	0.2	0.00287	0.00016	-0.0152	-0.09196	0.924	0.724
Barbel 1	[30 - 110]	28.6	-0.00351	0.00031	0.05318	-0.12227	0.917	.838
Barbel 2	]110 - 185]	2.5	-0.00162	0.00571	0.0573	-0.43816	0.859	.915
Barbel 3	]185 -	7.6	0.00132	0.00391	0.05603	-0.56108	0.799	.986
Chub 1	[30 - 110]	16.1	-0.00462	-0.00977	-0.07365	1.06653	0.973	.733
Chub 2	]110 - 170]	0.2	-0.00034	-0.00579	-0.05463	0.54911	0.611	.785
Chub 3	]170 -	2.0	0.00272	-0.00327	-0.06669	0.35059	0.887	.972
Roach 1	[30 - 90]	1.8	0.00015	-0.00401	-0.0814	0.63145	0.937	0.78
Roach 2	]90 - 150]	0.9	0.00127	-0.00339	0.00019	0.02405	0.999	0.929
Roach 3	]150 -	0.2	0.00338	-0.00313	-0.0646	0.29157	0.955	0.926
Gudgeon	[30 - 80]	24.5	-0.00476	-0.00646	-0.03661	0.7375	0.893	1.08

Table 2 : Hydrodynamics characteristics of two stations and two typical transects described in river Garonne. These characteristics were estimated for two discharges (20 and 300 m3/s) by the hydraulic models of EVHA software (Ginot and Souchon 1995) and from data which comes from ADCP survey (Gordon 1989).

	mean width (m)		wetted area (*100 m2)		max. depth (m)		mean depth (m)		max. velocity (m3/s)		mean velocity (m3/s)	
Discharge (m3/s)	20	300	20	300	20	300	20	300	20	300	20	300
Station 1	70	110	40.8	62.1	3.8	6.0	1.5	2.9	0.78	1.58	0.24	0.98
Station 2	100	130	81.2	105.6	4.0	5.5	1.0	2.0	0.96	2.09	0.25	1.14
Riffle Tr. 7 - sta. 2	95	120	13.2	23.4	0.8	1.9	0.4	1.2	0.96	2.10	0.58	1.46
Deep channel Tr. 4 - sta. 1	75	135	6.9	9.7	3.1	5.3	2.0	3.4	0.16	1.12	0.12	0.77

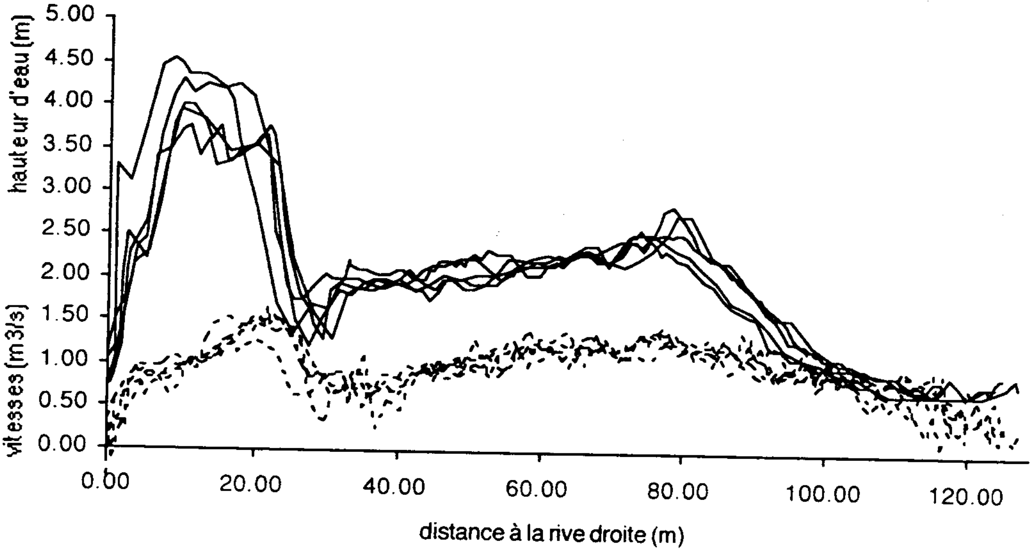
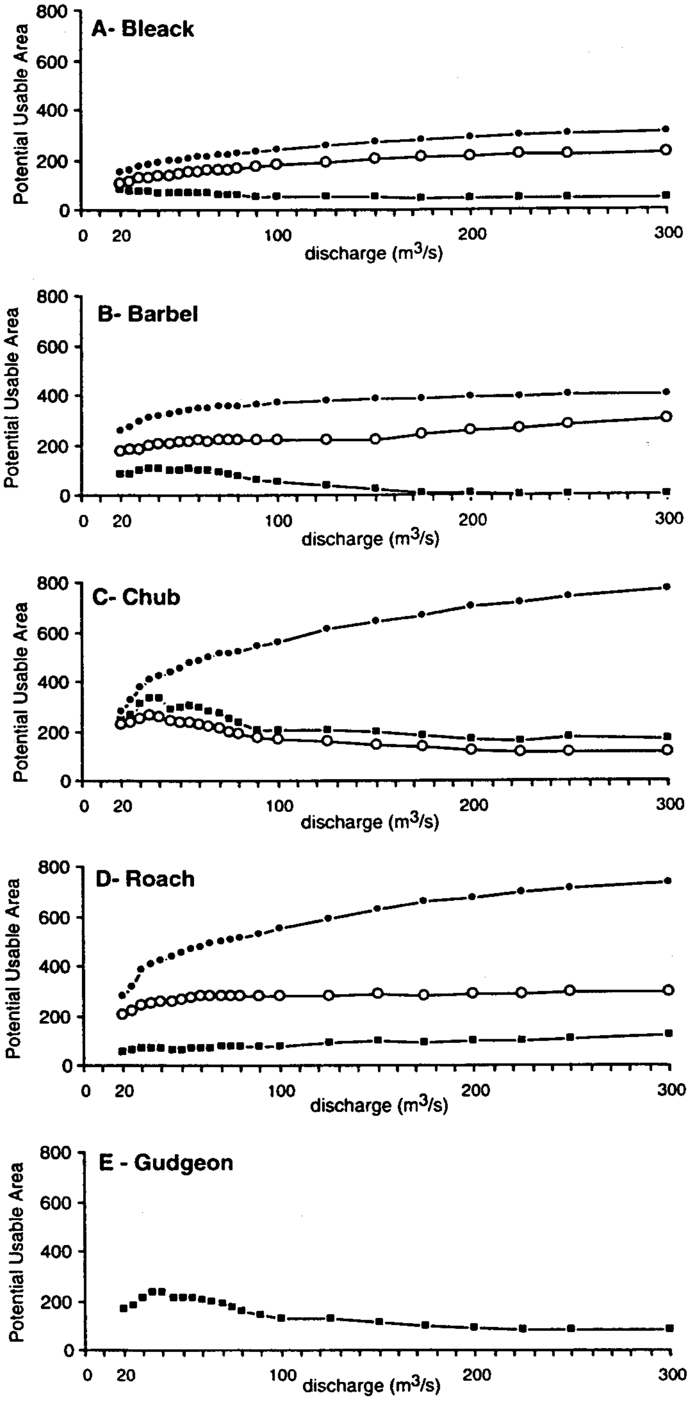
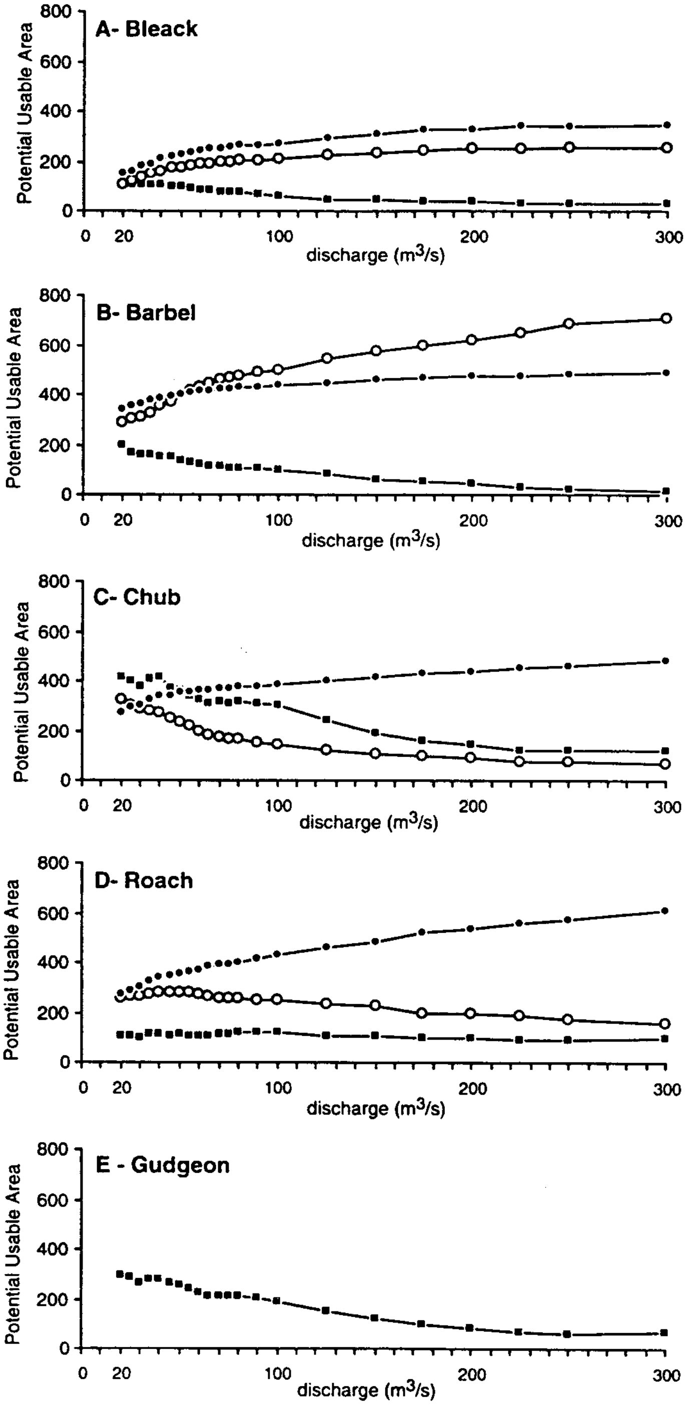


Figure 1 : Examples of depths and current velocities measured during four ADCP surveys on a transect on Garonne river near Toulouse for a discharge of 210 m3/s.

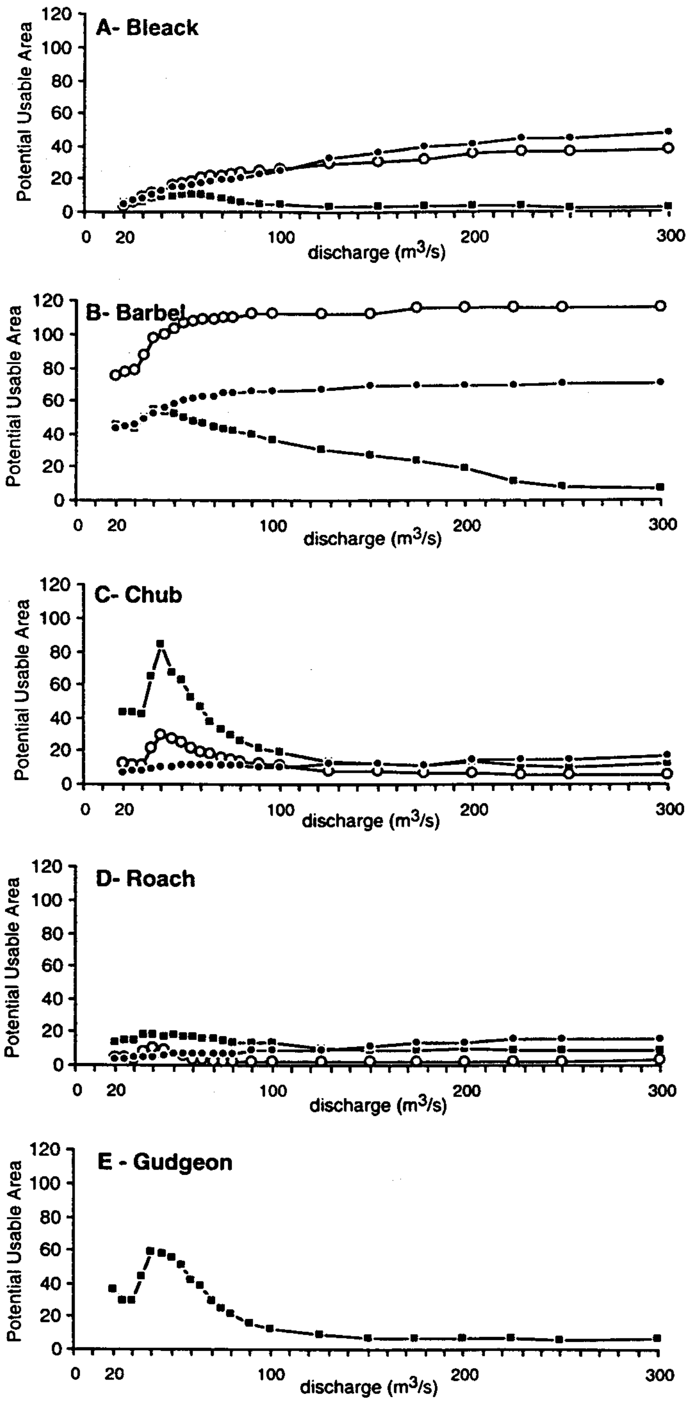


PUA station 1

Figure 2 : Evolution of potential usable area for 13 life-stages from 5 species in (a) station 1 and (b) station 2 of Garonne near Toulouse. Square = life-stage 1 ; circle = life-stage 2 ; disc = life-stage 3 (see Table 1 for size range of each life-stage).



PUA station 2



PUA station 2 - transect 7

Figure 3 : Evolution of potential usable area for 13 life-stages from 5 species in a riffle of station 2 of Garonne near Toulouse. Square = life-stage 1 ; circle = life-stage 2 ; disc = life-stage 3 (see Table 1 for size range of each life-stage).

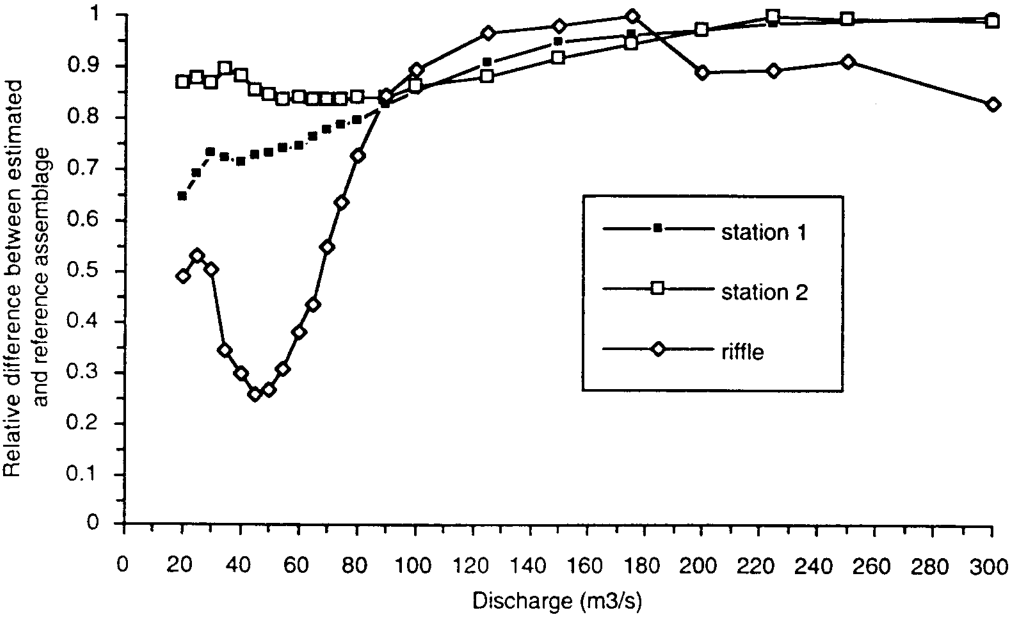


Figure 4 : Evolution, function of discharge, of relative differences (sum of square difference) between estimated fish assemblage for two stations and a riffle and reference fish assemblage of Garonne river near Toulouse.

SPATIOTEMPORAL USE OF THE RIFFLE/POOL UNIT BY FISH IN A BRITTANY BROOK

Jean-Marc Roussel

Agnès Bardonnet

NRA, Laboratoire d'Ecologie Aquatique
65 rue de Saint-Brieuc, 35042 Rennes cedex
France
roussel@roazhon.inra.fr

ABSTRACT

Electrofishing surveys as well as direct observations were conducted in Kerlegan brook, Brittany. Pool and riffle fish communities were sampled by day and by night in October. The only species exhibiting a diel pattern of macrohabitat use within the riffle/pool unit was brown trout (*Salmo trutta* L.). Trout used riffle and pool during daylight and aggregated in pool at night. Minnows (*Phoxinus phoxinus* L.) selected pool and sculpins (*Cottus gobio* L.) riffle when loach (*Nemacheilus barbatulus* L.) were present in both macrohabitat types.

Direct observations on riffle indicated a peak of activity at dusk or during dusk and night for sculpins all along the year, but no diel habitat change could have been recorded for this benthic species.

Results on trouts have pointed out a daylight or day and night activity all throughout the season, and significantly aggregated in the pool at night from May to October. Spatial distribution in the pool differed between active and resting trout, mainly in relation to cover, back-eddy and main water current.

KEY-WORDS: *Salmo trutta* / *Cottus gobio* / Fish Community / Riffle / Pool / Diel Shifts / Seasonal Shifts / Behavior / Functional Habitat.

Habitat use by fish is one of the topics in freshwater ecology which has experienced the greatest development for the last two decades. For Blondel (1979), "habitat" denote 1- the set of each environmental condition that is needed and used by a species, and 2- by extension, the favourable areas where the species is living. This second level refers to large spatial scales where geographic limits of population and species distribution are drawn. Physiology, adaptative and colonisation abilities are preponderant factors to explain this distribution. On the contrary, surrounding habitat conditions which lead the fish to choose a specific place in the channel refer to smaller spatial scales. Even inside an hydrologically stable habitat, fish will not always keep the same position, and their habitat choices will mainly depend on their activity. Salmonid streams are characterized by highly heterogeneous flow conditions, from macrohabitat (such as the riffle/pool unit) to microhabitat (focal point conditions). So even with short displacements, fish can easily experienced very different habitat conditions. If at the beginning most of the studies on carrying capacities of stream were deconnected from studies on diel or seasonal activity of fish (except for spawning), more and more results are now published on the subject (Harris *et al.*, 1992; Kwak *et al.* 1992; Nickelson *et al.* 1992; Milner *et al.* 1993). However, there is still a lack of knowledge about the behavioral cues governing the relationships between fish and their habitat.

The aim of this study was to describe diel fish habitat change inside a riffle/pool unit in a Brittany brook, and to follow diel and seasonal variations in riffle and pool use by the two predominant species, brown trout (*Salmo trutta* L.) and sculpin (*Cottus gobio* L.).

STUDY AREA

Kerlegan brook is a tributary of the River Scorff, Brittany, France (Figure 1). Its total length is 7.2 km, and it drains an area of 20.4 km^2. Gradient is 13 m.km^{-1} and the substrate is a mixture of gravel and sand, with pebble and small blocks in rapid areas. Brittany has a temperate oceanic climate, and mean maximum and minimum water temperature reach 17°C in july and 4°C in December.

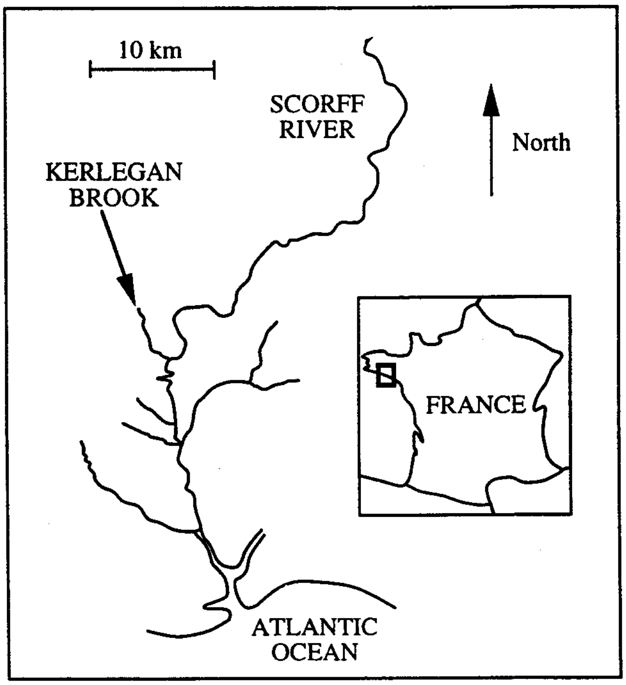


Figure 1: Study site

METHODS

In the last kilometre before the confluence with Scorff river, three pool/riffle units (see Table 1 for macrohabitat description) were electrofished with a Héron (direct current electrofisher, 200 watt continuous output). A special design of prepositioned nets allowed the partition of each unit in riffle and pool macrohabitats just before electrofishing. In October, two of these units were electrofished during daylight (at least 3 hours after dawn) and one by night (at least 3 hours after dusk). Two weeks later the opposite sampling (2 units by night, 1 unit by day) was realised. At night, the sampling area was lighted with a 100 watt halogen spot. Two removals were made to assess electrofishing efficiency by Carle and Strub's method (Carle & Strub, 1978). All fish were anesthesied, mesured and trout were individually marked (pit-tag). For trout, 3 life stages were assessed from body length (Young-Of-the-Year (YOY), One-Year-Old (OYO), and older trouts). They were confirmed by scalimetric analysis.

Just upstream these three units, direct observations on fish use of space and behaviour were made during a complete 24 hours cycle in a pool for trout and in a riffle for sculpin (Table 1). Thirteen observations were made, 9 during true night or day (about every 2.5 hours) and 4 during twilight (2 at dawn and 2 at dusk spaced by half an hour). The day after, the experimenter confirmed the results by 3 checking chosen in order to confirm the tendencies observed. A special attention was paid to conduct all the observations during moonless night.

Table 1: Main characteristics of the study site.

	electrofishing (October)		observations	diving
	riffle	pool	riffle	pool
Total length (m)	24	26	6	11
Mean width (m)	2.4	2.7	2.6	2.9
Mean depth (cm)	7	24	from 13 (July) to 22 (January)	from 30 (July) to 35 (January)
Mean velocity (cm/s)	14	4	from 20 (July) to 39 (January)	from 3 (July) to 16 (January)
Mean substrate	pebble, gravel and stones	sand, silt and stones	pebble, gravel and stones	sand, silt and stones

In the pool, the diver equiped with a 20 watt waterproof lamp, progressed from the downstream to the upstream end in 20 to 30 min depending on the number of trout seen. The three life stages (YOY, OYO, and older trouts) were visually assessed by the diver. Fish were counted, their activity (feeding or resting) and associations with cover (underbanks, stones, macrophytes), back-eddy or main current were noted.

In the riffle, the observer lied down on a board set across the channel above a 2 x 2 m metal square. This square was divided into 25 quadrants which were checked successively from bank to bank beginning at the upstream end. Using a 20 watt halogen lamp directed throught a clear plastic sled (size: 0.5 x 0.5 x 0.1 m), the observer located all sculpins seen during 2 successive checking taking 15 to 20 min each.

Pool and riffle observations were conducted at the same time in April (not in riffle), May, July, August, October and January.
Hypothesis about the use of macrohabitat (riffle or pool) by different life stages and species by day and night, were tested using non-parametric Mann-Whitney or Wilcoxon tests.

RESULTS

Electrofishing Surveys

Comparing fish densities in the 3 units (pool + riffle), the first electrofishing had no effect on the other two weeks later (p>0.05, Mann-Whitney test). Similarly, differences between day and night densities on an entire unit (riffle + pool) were not significant (p>0.05, Mann-Whitney test).

A total of 8 different species were caught, but only densities of sculpin (*Cottus gobio* L.), minnow (*Phoxinus phoxinus*, L.), trout (*Salmo trutta* L.) and stone loach (*Nemacheilus barbatulus* L.) were big enough to be considered. For these 4 species and 3 life stages of trout, differences in densities between riffle and pool by day, riffle and pool by night, and between day and night in riffle, day and night in pool, were all tested (Table 2).

Table 2: Mean densities caught by day and night in riffle and in pool for each species. Arrows are added when differences are significant (Mann-Whitney test, p<0,05).

			riffle		pool
Sculpins	day		6.63	↔	0.97
	night		5.57	↔	1.57
Minnows	day		0	↔	1.99
	night		0.02	↔	2.03
Loaches	day		0.36		0.42
	night		0.62		0.66
YOY trouts	day		0.02		0.08
	night		0		0.05
OYO trouts	day	↕	0.11		0.12
	night		0		0.18
Older trouts	day		0.04		0.04
	night		0	↔	0.08
All trouts	day	↕	0.17		0.24
	night		0	↔	0.31

Comparing day with night densities for each macrohabitat type (vertical arrows, Table 2), there were no significant differences for sculpins, minnows and loaches. Significant differences between day and night densities in riffle were found for all trouts summed. If at a global level, this tendency existed for all life stages, it was significant only for One-Year-Old trouts. Comparing now riffle with pool densities for day and for night (horizontal arrows), sculpin densities were always significantly higher in riffle. On the contrary, minnow densities were always significantly higher in pool, whereas no significant preference appeared for loach between riffle or pool. During daylight, all life stages of trout used both riffle and pool, and there were no significant differences in densities between the two macrohabitat types. At night, trout left the riffles, and densities significantly increased in pools (except for YOY).

Results from marking survey showed that trouts never changed from one unit to another, and an average of 66% of marked trouts were still present in the same unit two weeks later (Table 3). Inside one unit, all the fish caught in riffle by day have been first caught or will be recaptured in the pool of the same unit by night, exept for one fish in the second unit.

Table 3: Number of individuals marked during the first sample (beginning of the arrows) and the number of these ones recaptured 2 weeks later (end of the arrows).

		day		night
Unit I	pool	7	3 →	6
	riffle	3	3 ↗	0
Unit II	pool	2	1 →	1
	riffle	1		0
Unit III	pool	6	← 6	11
	riffle	3	↙ 3	0

Pool Diving Observations On Trouts

OYO trouts were the most numerous, older touts and YOY being 3 to 4 times scarcer (Figure 2). Number of OYO was maximum in July, August and October and then dropped in January. From May to October, significantly more OYO were seen at night. Changes in number of older trouts were similar to those for OYO all along the year, exept in April when a maximum of older trouts were seen at night. Differences between day and night mean numbers of trout seen were significant in April and August. YOY were observed in pool from July to January, but there were no significant difference between day and night (p>0.05, Mann-Whitney test).

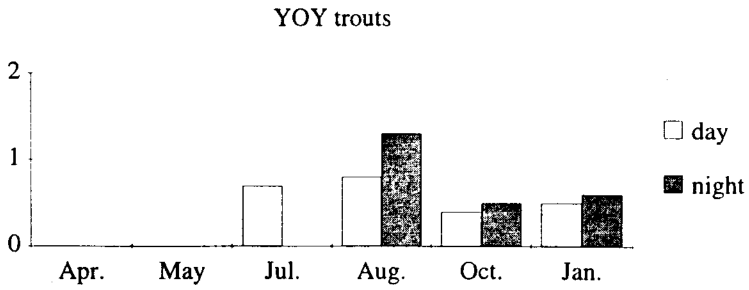


Figure 2: Mean numbers of Young-Of-the-Year trout (YOY) seen in the diving pool by day and by night at each sampling date

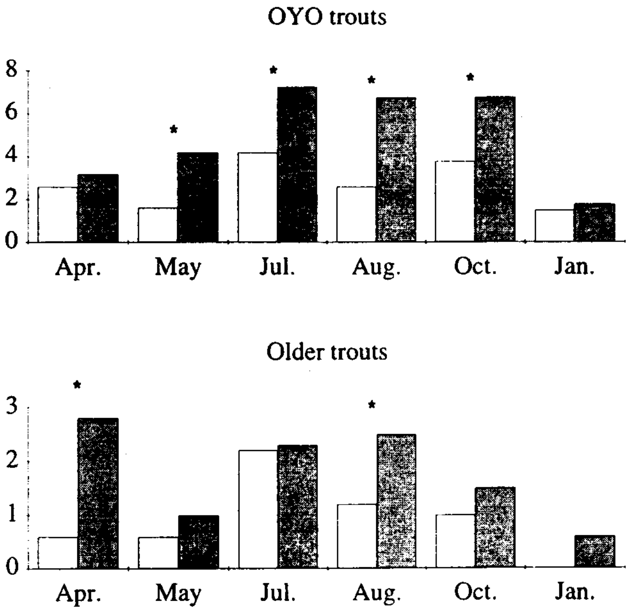


Figure 2: Mean numbers of One-Year-Old (OYO) and older trouts seen in the diving pool by day and night at each sampling date. * : significant differences between mean numbers seen by day and by night ($p<0.05$, Mann-Whitney test).

In average on the 24 h cycle, lower proportions of active trout were recorded in August and October (differences between life stages behaviour being small, all life stages were summed, Figure 3). Percentages of active trouts reached 25-30 % in April, May, July and January by day. Nocturnal activity was only observed from April to July, and maximum activity was recorded in April. No specific increase of activity was recorded during twilight.

When resting (Figure 4a), trouts were always lying on the bottom of the pool, keeping still in low velocities, either positive or negative (back-eddy). On a 24h cycle, 60% of resting fish were under cover, with minimum

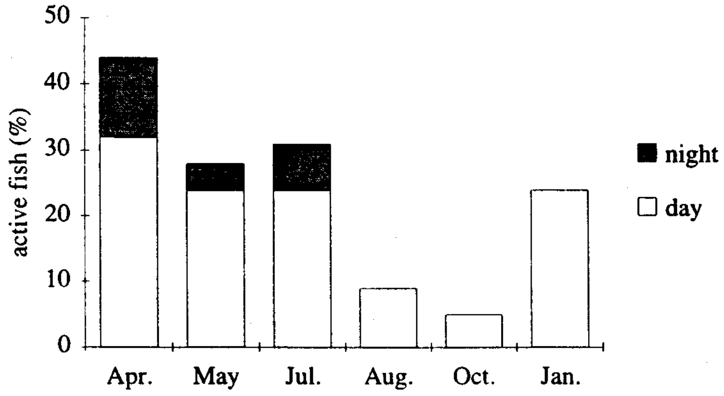
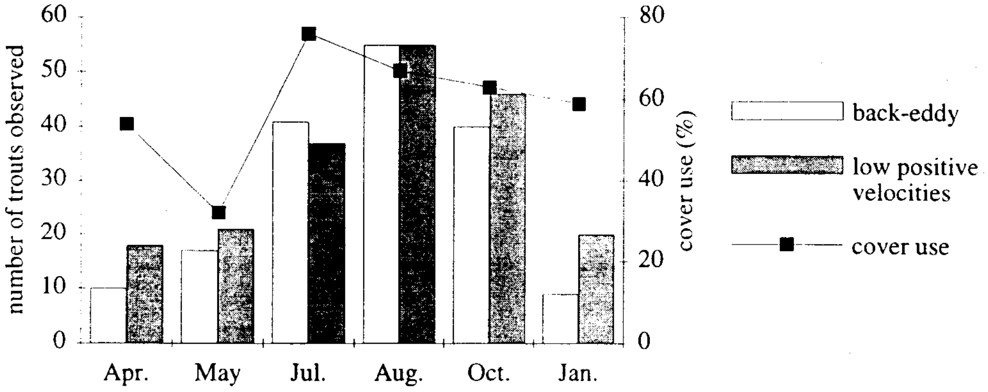


Figure 3: Proportions of fish in activity in the pool by day and by night at each sampling dates.

cover use in May (32%) and maximum in July (76%). When active (Figure 4b), trouts were free-swimming and feeding on drift in the main current of the pool. Only one trout having a foraging behavior (swimming around in the pool) was observed in the pool in August. Back-eddy was sometimes used for drift-feeding in April, May and July. Percentage of cover use when feeding by day was maximum in August and October. Trouts were then mainly under macrophytes (*Ranunculus sp.*) and underbanks. All along the year, underbanks were the most cover type used (55%) by resting fish, whereas they used both macrophytes (56%) and underbank (44%) when they were feeding on drift under cover (Figure 5).

a: Resting Trout



b: Active Trout

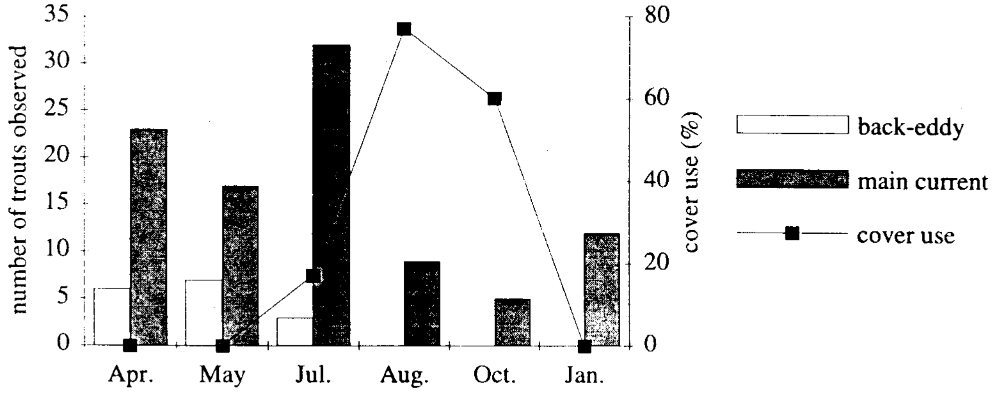


Figure 4: Percentages of trout using cover when resting (a) or feeding (b). White bars represent the mean numbers of trouts using back-eddy Shaded bars represent the mean numbers of trouts in positive current either in low (resting trout) or higher (active trout) velocities.

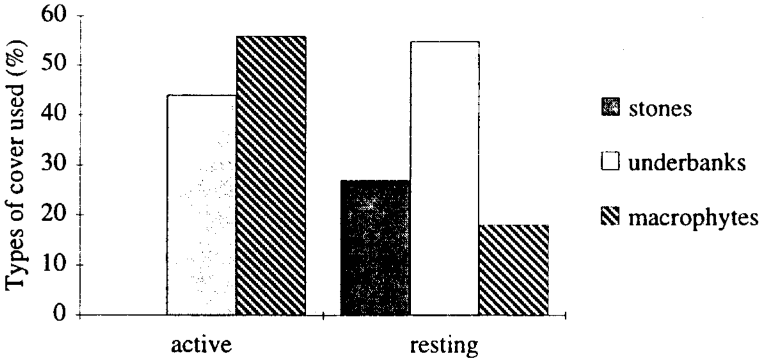


Figure 5: Trout dispaching among the different cover types in the pool (day+ night and sampling dates all summed).

Riffle Observations

Sculpins are very closely associated with the substratum and with their stone shelter. Direct observations in the wild supposed a supplementary illumination which obviously disturbed the fish when it was flashed over one or two min. Under these conditions activity could not be assessed at an individual level. Because sculpins spend most of their time under shelter (Gaudin, 1981), we assumed that the number of sculpins entirely visible during observations corresponded to active fish. Inter-seasonal comparisons cannot be made since we ignore how many fish were actually present on the riffle. At a nycthemeral scale (Figure 6), significantly

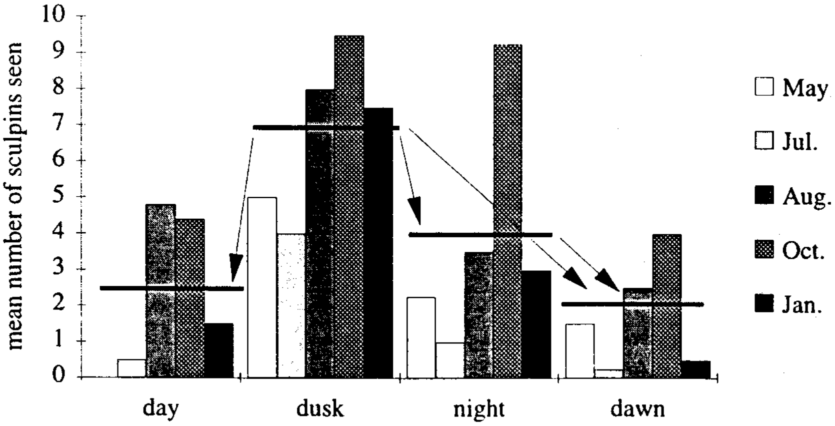


Figure 6: Mean numbers of sculpins seen in the 24 h cycle at the different sampling dates. The horizontal lines indicate the mean (for all the sampling dates) and the arrows indicate the series which significantly differ (p<0.05, Wilcoxon test)

more active fish were seen in average at dusk all along the year ($p<0.05$, Wilcoxon test). Variations in the number of fish seen (Figure 7) between day, dusk, night and dawn were respectively the same all along the seasons except for October when night reached the level of activity measured at dusk. However, no diel variation in sculpin distribution within the sampling area was found. The only tendency concerned the lateral distribution throughout the quadrants all along the year. The right part of the riffle was almost never used (4% in average) for an area covering 20% of the sampling surface, whereas a high proportion of sculpins (38%) were seen in the left part of the riffle (20% of the sampled area). The only habitat variable found to explain this lateral distribution was the higher proportion of loose pebbles and stones in the left part of the riffle. Concurrently, pebbles and stones were more embeded in sand in the right part of the riffle than anywhere else.

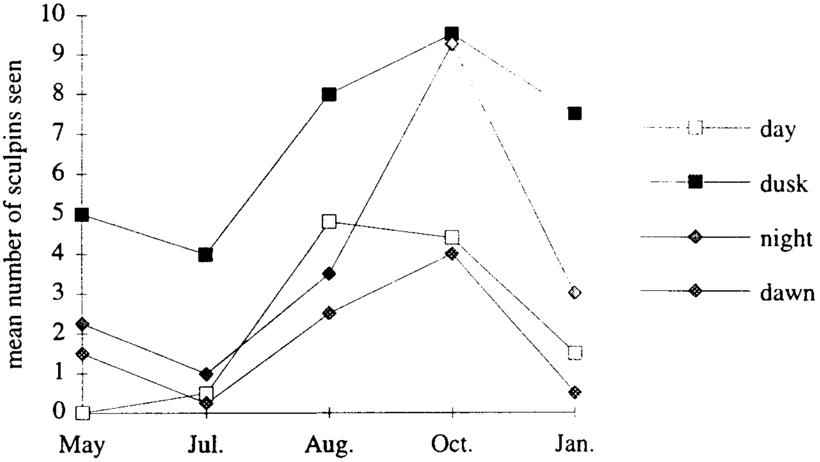


Figure 7: Mean numbers of sculpins seen during the 24 h cycle at each sampling dates.

DISCUSSION

Results of electrofishing surveys in October have shown a spatial preference between riffle and pool in minnows and sculpins. Minnows were only present in pool whereas sculpins densities were higher in riffle. For this reason, there was a sharing of the space between the two species, already observed in larger stream by Jones (1975). However, densities of sculpins in pools were not negligible (1-1.5 ind/m^2). Even if they are known to select areas with coarse substrate (Greenberg and Stile, 1993), numbers of studies outline the wide range of depth and water velocities used by this species (Gaudin and Caillère, 1990; Greenberg, 1991; Copp, 1992). In our study, sculpin preferences for riffle could have been emphazised in October by the presence of small individuals (< 40 mm), which were 1.8 times more numerous in riffle than in pool. Because of favourable coarse substrate (Greenberg and Stile, op. cit.), or strong predation on larval in pool, riffles might be better nursery areas for sculpins. The spatial segregation within the riffle/pool unit did not include stone loach. This species would prefer silt substratum and low velocities (Baglinière and Arribe-Moutounet, 1985). Even if the area concerned by these conditions were much larger in pool compared with riffle, densities were not higher in pool in this study. However, another parameter could have played a role in loach distribution. According to Welton *et al.* (1983), loach strongly select weed against gravels or pebbles. Yet, in our study aquatic macrophytes (mainly *Ranunculus sp.*) covered 40% of the surface in riffle when it covered only 5% in pool. The two factors (silt in pool and weed in riffle) might have compensated each other.

At a nycthemeral scale, there were no macrohabitat shifts between day and night for loach, sculpin and minnow. This result confirms previous snorkling observations on macro and microhabitat used by sculpins during day and night (Roussel and Bardonnet, in press). According to riffle observations, most of the sculpins entirely visible, were counted at dusk. This agrees with previous results on a dark-activity released by light reduction at sunset for sculpin (Andreasson, 1969). However, if locomotor activity was restricted to dusk in May in Andreasson's study, it generally kept on going at a high level throughout the night for the other seasons, which happened only in October in our study. Other studies on *Cottus sp.* confirmed a night activity (Gaudin, 1981; Greenberg and Holtzman, 1987), but underlined the possible change in rhythmic activity according to density or trophic conditions. No specific areas were chosen throughout the quadrants during dusk compared with the other time of the day. Sculpins are known to have a small home range and to spend most of their time under cover (Downhower *et al.* 1988; Greenberg, 1991). Experimental (Gaudin, op.cit.) as well as field results (Greenberg, op.cit.) show that sculpins have a high residency rate spending most of their time under or close to their stone shelter. Then, even at the small spatial scale we worked (0,16 m^2) no habitat change was revealed.

Water depth is considered as one of the most important factor affecting spatial distribution of salmonids in stream in autumn (Egglishaw and Shackley, 1982). In natural stream, spatial segregation exists between Young-Of-the-Year in shalow habitats and older trouts in deepest ones (Baglinière and Arribe-Moutounet, 1985; Kennedy and Strange, 1986). Our observations showed that YOY appeared early in pool (July). Besides, in October, no significant difference was found for YOY densities in riffle and in pool. Then, the progressive spreading from shalow to deepest areas by YOY would have began sooner in this study compared with Elliott (1986) who observed YOY in pools only in winter.

Distribution of One-Year-Old and older trouts within the riffle/pool unit changed between day and night. In October, both riffle and pool were used by day, whereas most of trouts were in pool at night. This tendency has been already observed for OYO in an experimental study (Roussel and Bardonnet, 1995), but only 25% of individuals in average were in riffle during daylight. Differences in riffle use between the two studies could be explained by the absence of shelter in the experimental study.

Diving observations in pool confirmed that the number of trouts in the pool was significantly higher at night, and this was true from May to October. Trout used the pool almost exclusively as a resting area in August and October, whereas 25-45% of active trouts were observed in the pool the rest of the year. There are few studies on seasonal variations of brown trout feeding activity in the wild under our latitudes: Bachman (1984) observed a summer decrease of feeding rate. However, there were changes in density at the pool scale with a large decrease in winter which could be an explanation for the difference in the level of activity in winter. At a nycthemeral scale, feeding activity was essentially diurnal, even if some individuals were observed free-swimming at night in spring and July. Nocturnal activity has been already reported for brown trout (Clapp *et al.*, 1991), which have a special ability to feed under very low light level (Robinson and Tash, 1979). Obviously, our observations never showed a clear inversion in diel activity pattern as recorded between summer and winter in northern streams (Eriksson, 1973; Müller, 1978; Cunjak and Power, 1986; Heggenes *et al.*, 1993).

Use of cover in the pool concerned mainly resting trout, with a maximum of cover use in July. However, active trouts also use cover but only during low waterflows (July, August and October). Brown and Mackay (1995) have also reported a higher cover use in summer and fall for cutthroat trout (*Oncorhynchus clarki).* Use of underbanks was high for active and active trouts. Aquatic macrophytes also provided a good shelter for active trouts in summer.

Within the pool, spatial distribution of trout depended on their activity. Most of drift-feeding trouts were in the main current of the pool, whereas half of the resting ones lied on the bottom of the back-eddy. Then, a diel use of an infra-facies habitat (like back-eddy or main current), already observed under experimental conditions (Roussel and Bardonnet, 1995) was confirmed by this field study. Within the entire riffle/pool unit, diel movements from pool to riffle have to be related to feeding activity (Bridcut and Giller, 1993; Roussel and Bardonnet, 1995). Then, the diel habitat change between feeding and resting position is perceptible at different scales: the macro-scale which refer to the habitat type (riffle or pool in this study) and the meso scale which refer to the main hydraulic events (vein of water current, back-eddies). The micro-scale (microhabitat) was not studied here. It has been largely investigated since almost 20 years. A previous study has emphasized how the variables of these different scales interacted on the diel habitat choice of one-year-old trout (Roussel and Bardonnet, in press).

Spatial heterogeneity between riffle and pool, and inside the pool provide a set of habitat conditions, which authorized the achievement of diel activities without large displacement. For this reason, riffle/pool unit in brook can be considered as a "functional unit" since trout found in it all the habitat conditions to realize the whole of their daily behavior.

REFERENCES

Andreasson, S. (1969). Locomotory activity of *Cottus poecilopus* Heckel and *Cottus gobio* L. (Pisces). *Oikos.* 20 : 79-94.

Bachman, R.A. (1984). Foraging behavior of free-ranging wild and hatchery brown trout in a stream. *Trans. Am. Fish. Soc.* 113 : 1-32.

Baglinière, J.L. and D. Arribe Moutounet (1985). Micro-répartition des populations de Truite Commune (*Salmo trutta*), de juvéniles de Saumon Atlantique (*Salmo salar*) et des autres espèces présentes dans la partie haute du Scorff (Bretagne). *Hydobiologia.* 120 : 229-239.

Blondel, J. (1979). Biogéographie et écologie. Masson, Collection d'écologie 15, Paris, 173 p.

Britcut, E.E. and P. S. Giller (1993). Diet variability in relation to season and habitat utilisation in brown trout, *Salmo trutta* L. In southern Irish stream. In R.J. Gibson and R.E. Cutting, Production of juvenil atlantic salmo*n, Salmo salar*, in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118

Brown, R.S. and W.C. Mackay (1995). Fall and winter movements of and habitat use by Cutthroat trout in the Ram River, Alberta. *Trans. Am. Fish. Soc.* 124 : 873-885.

Carle, F.L. and M.R. Strub (1978). A new method for estimating population size from removal data. *Biometrics.* 34 : 621-630.

Clapp, D.F., R.D. Clark and J.S Diana (1990). Range, activity, and habitat of large, free-ranging brown trout in Michigan stream. *Trans. Am. Fish. Soc.* 119 : 102-1034.

Copp, G.H. (1992). An empirical model for predicting microhabitat of 0+ juvenile Fishes in a lowland river catchment. *Oecologia.* 91 : 338-345

Cunjak, R.A. and G. Power (1986). Winter habitat utilization by stream resident brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*). *Can. J. Fish Aquat. Sci.* 43 : 1970-1981.

Downhower, J., P. Lejeune, P. Gaudin and L. Brown (1990). Movements of the chabot (Cottus gobio) in a small stream. *Pol. Arch. Hydrobiol.*, 37 : 119-126.

Egglishaw, H.J. and P.E. Shackley (1982). Influence of water depth on dispersion of juvenile salmonids, *Salmo salar* L and *Salmo trutta* L, in a Scottish stream. *J. Fish Biol.* 21 : 141-155.

Elliott, J.M. (1986). Spatial distribution and behavioural movements of migratory trout *Salmo trutta* in a Lake District stream. *J. Anim. Ecol.* 55 : 907-922.

Eriksson, L.O. (1973). Spring inversion of the diel rhythm of locomotor activity in young sea-going brown trout, *Salmo trutta trutta* L., and atlantic salmon, *Salmo salar* L. *Aquilo Ser. Zool.* 14 : 69-79.

Gaudin, P. (1981). Eco-éthologie d'un poisson benthique, le chabot, *Cottus gobio* L (*Cottidae*): Distribution, alimentation et rapports avec la truite, *Salmo trutta* L. Ph.D thesis, University of Lyon: 178 p.

Gaudin, P. and L. Caillère (1990). Microdistribution of *Cottus gobio* L. and fry of *Salmo trutta* L. in a first order stream. *Pol. Arch. Hydrobiol.* 37 : 81-93

Greenberg, L.A. and R.A. Stiles (1993). A descriptive and experimental study of microhabitat use by young-of-the-year benthic stream fishes. *Ecol. Freshwater Fish*, 2 : 40-49.

Greenberg, L.A. and D.A. Holtzman (1987). Microhabitat utilisation, feeding periodicity, home range and population size of banded sculpin, *Cottus carolinae.Copeia.* 1 : 19-25.

Greenberg, L.A. (1991). Habitat use and feeding behavior of thirteen species of benthic stream fishes. *Env. Biol. Fish.* 31 : 389-401.

Harris, D.D., W.A. Hubert and T.A. Wesche (1992). Habitat use by young-of-the-year brown trout and effects of Weigth Usable Area. *Rivers.* 3 : 99-105.

Heggenes, J., O.M. Krog, O.R. Lindas, J.G. Dokk and T. Bremnes (1993). Homeostatic behavioural responses in a changing environment : brown trout (*Salmo trutta*) become nocturnal during winter. *J. An. Ecol.* 62, 295-308.

Jones, A.N. (1975). A preliminary study of fish segregation in salmon spawning streams. *J. Fish. Biol.* 7 : 95-104.

Kennedy, G.J.A. and C.D. Strange (1986). The effects of inta- and inter-specific competition on the survival and growth of stocked juvenile atlantic salmon, *Salmo salar*, and resident rainbow trout, *Salmo trutta* L., in an upland stream. *J. Fish. Biol.* 28 : 479-489.

Kwak, T.J., M.J. Wiley, L.L. Osborne and R.W. Larimore (1992). Application of diel feeding chronology to habitat suitability analysis of warmwater stream fishes. *Can. J. Fish. Aquat. Sci.* 49 : 1417-1430.

Milner N.J., R.J. Wyatt and H.D. Scott (1993). Variability in the distribution and abundance of stream salmonids, and the associated use of habitat models. *J. Fish Biol.*, 43 : 103-119.

Müller K. (1978). Locomotor activity of fish and environmental oscillations. *in* Thorpe J.E., Rhythmic activity of fishes, 1-19 , Academic Press, London.

Nickelson T.E., J.D. Rodgers, S.L. Johnson and M.F. Solazzi (1992). Seasonal changes in habitat use by juvenile coho salmon (Oncorhynchus kisutch) in Oregon coastal streams. *Can. J. Fish. Aquat. Sci.* 49 : 783-789.

Robinson, F.W. and J.C. Tash (1979). Feeding by arizona trout (*Salmo apache*) and brown trout (*Salmo trutta*) at different light intensities. *Env. Biol. Fish.* 4 : 363-368.

Roussel, J.M. and A. Bardonnet (1995). Activité nycthémérale et utilisation de la séquence radier/profond par les truitelles d'un an *(Salmo trutta* L.). *Bull. Fr. Pêche Piscic.* 337/338/339 : 221-230.

Roussel, J.M. and A. Bardonnet, (in press). Changements d'habitat en fonction de l'activité et du nycthémère: exemple de la truite (Salmo trutta L.) et du chabot (Cottus gobio L.). *Cybium.*

Welton, J.S., C.A. Mills and E.L. Rendel (1983). Food and habitat partitioning in two small benthic fishes, *Nemacheilus barbatulus* L. and *Cottus gobio* L. *Arch. Hydrobiol.* 97 : 434-454.

Microposition choice in stream-living Atlantic salmon (*Salmo salar*) parr and brown trout (*Salmo trutta*): Habitat-hydraulic 3-dimensional model and test

Jan Heggenes
Department of Biology and nature Conservation, Agricultural University of Norway, P.O. Box 5014, 1432 Ås, Norway

Atle Harby
SINTEF-Civil and Environmental Engineering, 7038 Trondheim, Norway

Tammo Bult
Ocean Sciences Centre, Memorial University of Newfoundland, St. John's, Newfoundland, Canada
A1C 5S7

ABSTRACT

Microhabitat selection data from spatio-temporally heterogeneous streams must be interpreted in the light of site and time, and particularly the spatial scale, range and dimensions studied. Salmonids in streams select holding stations based on 3-dimensional microgradients (cm^3) in relevant habitat variables. Traditional hydraulic models are one or 2-dimensional and simulate average hydraulic conditions within large areas relative to the fish. Tests have indicated limited generality for habitat-hydraulic models in predicting fish habitat selection and/or production capacity. The critical focus has often been on a presumed inability of the habitat models to incorporate and model relevant variables in habitat choice by fish. However, spatial dimensions and scale in hydraulic modelling may also be important. It may be too coarse-grained to incorporate important ecological information derived from fish microposition choice, which was the problem investigated here.

Micropositions occupied by young Atlantic salmon, *Salmo salar*, and brown trout, *S. trutta*, in streams are characterized by relatively low snout water velocities, but close to a fast current. To test consistency in and modelling of microposition choice, behaviour and position of 725 fish were recorded by direct underwater observation at a particular location and moment in time, at particular spatial scales and ranges. Fish positions were located to the nearest cm^3 within a heterogeneous 122 m long river section representing about 2000 - 3600 m^2 or 355 -3745 m^3 depending on waterflow within the studied range of 3 - 4 m^3s^{-1}. To reduce environmental noise in the data, observations were implemented over an 11-day period with stable flows and weather conditions. A microposition selection model was developed. A numerical model for 3-dimensional simulation of water flow in rivers with large roughness elements was developed for the same river section. The geometry of the section was measured by random stratified sampling in the field with 2061 data entries.

Water velocity measurements were used to validate the model performance. We compared our 3D micro-model with a conventional habitat-hydraulic model, by simulating habitat suitabilities at water flows of 2 - 5 m^3s^{-1}. The results indicated differences between models based on simulations of water velocities and depth. Optimal flow was about 3.5 m^3s^{-1} for the conventional 2D model, while it varied little in the studied flow range for the 3D model. This is attributed both to differences in hydraulic and biological models, e.g. depth dimension and snout velocity. The results are discussed in the context of spatial dimension and scale problems associated with conventional hydraulic modelling.

INTRODUCTION

Optimal foraging theory suggests that foraging behaviour in animals maximize potential net energy gain, although such behaviour may be modified by risk (e.g. Hart 1993). Streamliving salmonids are ideal examples of such behaviour. Because their foraging behaviour translate into spatial requirements (e.g. Hill & Grossman 1993), this is of relevance in habitat-hydraulic modelling. In streams juvenile Atlantic salmon (*Salm. salar*) and brown trout (*Salmo trutta*) mainly feed by holding station against the current and dart out to catch food items drifting past in the current. Consequently, potential net energy gain of a holding station is determined through micro water velocities (energy gain = drifting food items; energy cost = holding position in the current and intercepting food items), i.e. spatial requirements at or close to the holding station. Thus, studies of trout and salmon habitat selection in streams have consistently demonstrated that of all studied habitat variables, the fish are most selective for low snout water velocities, presumably to reduce energy cost of holding position in the current (e. g. Heggenes et al. 1996).

This is the ecological basis for reasonably predictive models of microposition choice by fish in streams (e.g. Hughes 1992, Hill & Grossman 1993). On the other hand, habitat-hydraulic models simulating habitat productive capacity have experienced less success. One obvious reason may be the lack of congruence in spatial dimensions and scale between fish habitat selection and hydraulic models. Fish observations are made on a certain location and time, and on certain spatial and temporal scales. Conventional hydraulic models use relatively large spatial scales and are one or 2-dimensional. They simulate average hydraulic conditions within 'cells' of several m^2 or more. Fish, however, select holding stations also based on 3-dimensional microgradients (cm^3). Thus, inability of hydraulic models to simulate conditions in three dimensions and on a smaller scale known to be relevant to fish, may explain part of the hitherto limited success. Based on ecological theory, incorporating 3-dimensional modelling on small spatial scales would be appropriate.

Therefore, the objectives of our study was, first to develop a 3-dimensional model able to simulated hydraulic conditions on small spatial scales known to be relevant to fish. A second objective was to test the ecologically based hypothesis that there would be discrepancies between 3-dimensional habitat-hydraulic modelling on a small scale and conventional 2-dimensional modelling on a large scale, leading to different descriptions of habitat suitabilities. We did this by investigating how habitat suitability changed with water flow, by conventional hydraulic modelling and compared to 3D microposition modelling. If discrepancies were found, this would pose a serious problem to habitat modelling, if not, this would be of theoretical interest, but not call for adjustments in the conventional habitat modelling approach.

MATERIALS AND METHODS

The 19 km long River Gjengedalselva, a third order soft-water stream, is located on the central west coast of Norway. It has a drainage area of 171 km^2. Mean annual discharge is 12 m^3s^{-1}, but the water flow regime is highly variable due to snowmelt and precipitation, with mean low flows in late winter (January-May) and during dry summer periods around 1-2 m^3s^{-1}, and high floods in spring and fall often up to 100 m^3s^{-1} (maximum 240 m^3s^{-1}). Atlantic salmon and brown trout were the only fish species observed on the studied river section. One 122m long river section (Fig. 1), consisiting of a pool, run, riffle, and rapid representing a range of habitat-hydraulic conditions available to the fish, was chosen for study.

Fish microhabitat selection

Behaviour and microhabitat selection by young Atlantic salmon and brown trout were recorded by direct underwater observation using a dry suit, mask and snorkel. Daytime observations were during an 11-day period (4-15 September 1995), between 09.00 and 18.00 hours, when underwater visibility was 7 meters or more, and weather conditions and waterflow (3-4 m^3s^{-1}) were stable. This was done to reduce possible effects of uncontrolled environmental effects on fish behaviour (waterflow, light, temperatures). All snorkeling work was done by the same diver. Each dive started at the downstream end of the section, and the diver moved slowly upstream in a zigzag fashion to randomize any bias associated with nearshore or midstream observations. On encounter, a fish holding station was observed for up to 6 minutes to establish fish species, study its behaviour and make sure it was holding a microposition of its own choice. If so, species, behaviour, total fish length (±1cm) and height above bottom (±1cm) were determined by the diver. Snout position was marked with a

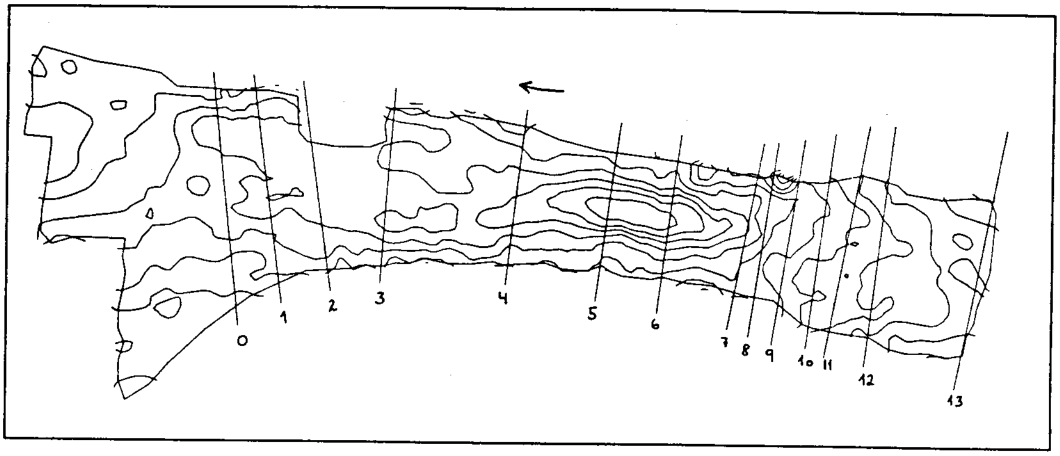


Figure 1. Depth contour map of the section in R. Gjengedalselva with numbered transects for conventional modelling. Flow from right to left.

coloured lead weight with a flag. Later, distance from river bank and nearest transect (±1cm), snout water velocity (±1cms-^{1}), surface water velocity (cms-^{1}), mean water velocity (cms-^{1}), and total water depth (±1cm) were determined. A direct reading Höntzsch Flowtherm sensor FT25GFA fitted with a 1.8-cm diameter vane wheel was used to measure velocities.

Hydro-physical modelling

Data on actual waterflows were provided from two gauging stations in the river (22 and 11 years of data, respectively). Flows are greater than or equal to annual mean flows less than 40% of the year (<30% in winter), indicating short time periods with very high runoff and correspondingly long periods with low. Data on topography were collected by in situ surveying of 2061 verticals, and were used to describe the river bed in the Sediment Simulation In Intakes with Multiblock option (SSIIM) 3-dimensional hydraulic model (Olsen and Stokseth 1995). The data were also used to describe the topography of 10 transects for conventional habitat modelling with the HABITAT (Harby and Heggenes 1995) in the River System Simulator (RSS). Additional topographic data were collected in 4 transects for conventional modelling (Fig.1). Also water velocities were measured on two contrasting water flows, 2 and 5 m^{3}s^{-1}, in these 4 additional transects. The velocity measurements served as input in the conventional HABITAT model, as well as for validation of the SSIIM model. Water velocities (cms^{-1}) were measured 0.05 and 0.10 m above the river bed, and at 0.6 x total depth. We used an OTT flow meter with vane wheel diameters of 5 and 10 cm, and over a 40 sec. period. Water surface elevations were measured at 2 and 5 m^{3}s^{-1} for calibrations of the HEC-2 (US Army Corps of Engineers 1991) in the HABITAT and calibration of the initial water surface calculations in the SSIIM.

In the conventional habitat-hydraulic model, HEC-2 simulates steady flow. The results from HEC-2 serve as input for the HABITAT model, together with transect data for water depth, mean water velocity and substrate, and fish preference data.

The 3-D SSIIM model was used to simulate water depths and velocities with high spatial resolution. The program solves the Navier-Stokes equations with the k-epsilon model on a three-dimensional almost general non-orthogonal grid. The grid is structured. A control volume method is used for the discretization, together with the power-law scheme or the second order upwind scheme. The SIMPLE method is used for the pressure coupling. The solution is implicit, also over the boundary of the different blocks. This gives the velocity field in the geometry. The velocities are used when solving the convection-diffusion equations for different sediment sizes (Olsen 1995, Olsen and Stokseth

1995). The biological model for suitable fish habitat is based on the same variables as for HABITAT, in addition to snout water velocity.

We used a grid of 50x30x6 grid-lines in the i- (parallel to the flow), j- (perpendicular to the flow) and k-directions (depth). This corresponded to 7105 cells for habitat-hydraulic modelling. To investigate possible influence of spatial resolution, we also used a grid of 99x29x5 = 14355 cells.

RESULTS AND DISCUSSION

Model calibration

Calibration of the HEC-2 normally means adjusting the friction, described by Mannings formula (French 1986), so as to fit calculated water surface elevation with observed values. Calibration results (Fig. 2) indicated best fit in the area between transects 0 to 6 and 12-13, while fit was poorer in the high-gradient and more hydrodynamically complex area 7 to 11. The SSIIM calculations were closer to the observed values (Fig. 2).

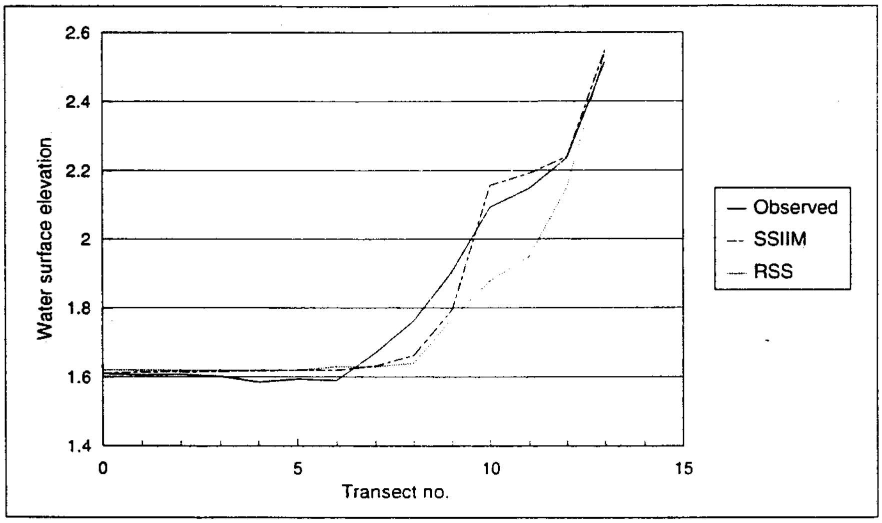


Figure 2. Calibration results for the HABITAT and validation of the SSIIM compared to observed values.

Due to preliminary technical and practical problems, we were only able to compare mean water velocities for two transects and at a flow of 2 m^3s^{-1}. In the low-gradient (pool) transect 4 observed mean velocity was 0.113 ms^{-1}, which was close to 0.11 ms^{-1} calculated by HABITAT and 0.119 ms^{-1} by SSIIM. In the more high-gradient (riffle/rapid) transect 8 corresponding values were 0.341 ms^{-1} for observed, and 0.38 ms^{-1} and 0.342 ms^{-1} for the HABITAT and SSIIM, respectively. Thus the SSIIM calculations were closer to observed values than were the HABITAT calculations particularly at higher gradients, and suggesting more realistic hydraulic modelling in more complex situations.

Fish habitat selection

During 11 days of intensive observation, microposition and habitat data were collected for 602 individual young Atlantic salmon (mean total length 8.6cm±SD2.4, range 3.5-16cm) and 123 brown trout (mean TL 7.1cm±SD3.2, range 3.5-20cm) (Fig. 3). There were significant differences in habitat use between species (Kolmogorov-Smirnov and Kruskal-Wallis tests, $p<0.05$). Relative to brown trout, Atlantic salmon selected holding stations with higher snout water velocities (median 2 and 5 cms^{-1}; respectively), closer to the bottom (median 8 and 2 cm; respectively) and in deeper water (median 37 and 65 cm), i.e. more in the midstream areas (Fig. 3). Trout occupied areas closer to the river banks.

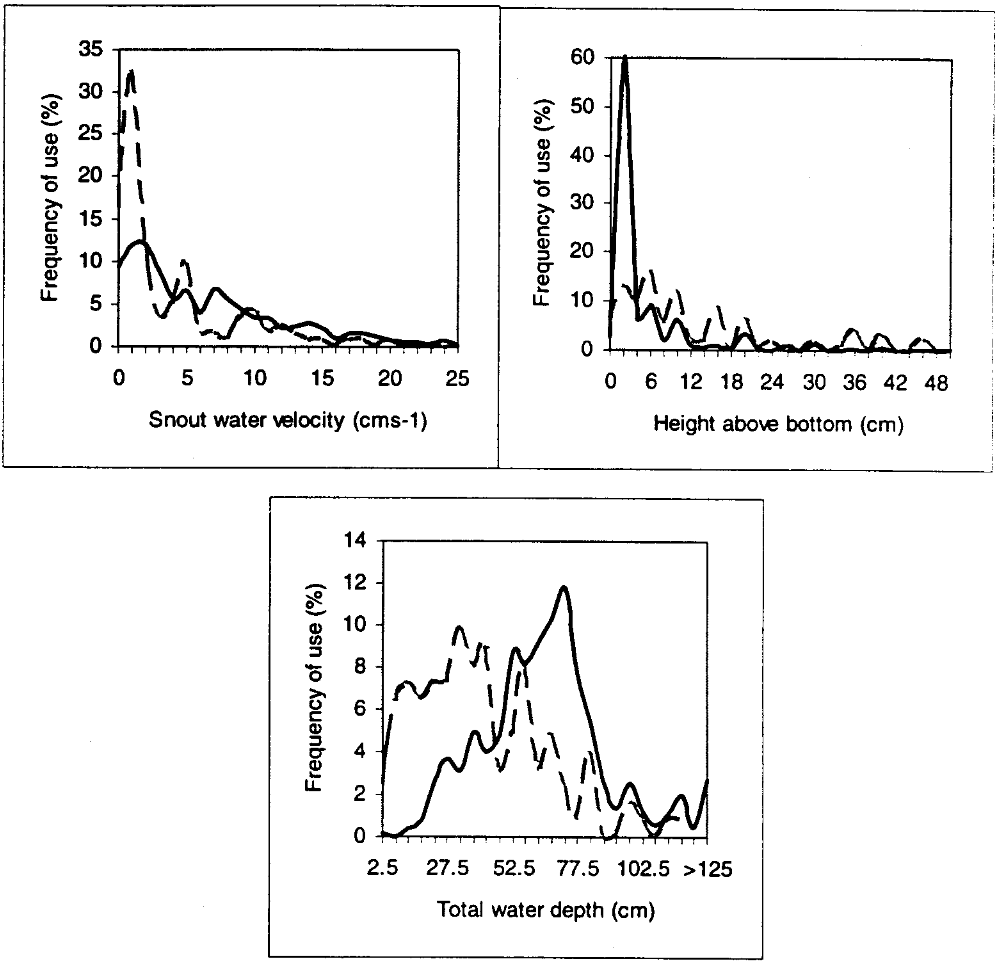


Figure 3. Microposition selection by 602 Atlantic salmon (solid line) and 123 brown trout (darted line) for snout water velocity, height above bottom and water depth on a 125m section in Gjengedalselva R.

In calculation of preferences for habitat-hydraulic model simulations we focused on Atlantic salmon which provided more data, and pooled data to make them more robust by using relatively large unit intervals for depth and mean water velocity, and by categorizing (micro)habitat variables into preferred, indifferent and avoided (Tab. 1).

Table 1. Preferences for (micro)habitat variables for Atlantic salmon in R. Gjengedalselva

(Micro)habitat variable	**Preferred**	**Indifferent**	**Avoided**
Snout water velocity	0 - 9	10 - 15	>15
Mean water velocity	15 - 22	1 - 14, 23 - 26	0, >26
Total water depth	50 - 85	25 - 49, 86 - 90	1 - 24, >90

Comparison of 3-dimensional and 2-dimensional models

While the conventional 2D modelling could be easily done on a computer, the 3D modelling proved to be demanding and timeconsuming with respect to computer resources. Thus results are reported for a limited range of environmental conditions.

Mean water velocity

The small-scale 3D and the conventional larger scale 2D models were compared by simulating suitable (micro)habitat versus flow, i.e. for the variables snout and mean water velocity and depth (Fig. 4). It is important to point out that trends are more easily comparable, while the 2D model quantifies habitat as area and the 3D as volume. Simulations of suitable mean water velocities were different between the two models. The conventional model showed a clear optimum at 3.5 m^3s^{-1} with about 45% of total river section area as preferred, while the SSIIM indicated much less (about 20% of total volume) and also little variation in preferred mean velocities. Similarly, with respect to indifferent mean water velocity habitat (Fig. 4), the conventional model indicated a low of about 20% of total area at 3.5 m^3s^{-1}. In contrast, the SSIIM model suggested decreasing volumes representing indifferent mean velocities with increasing flows (from 60 to 35%). Only with respect to avoided mean water velocity habitat was there reasonable agreement between the two models. Because the biological models are identical with respect to mean water velocity preferences (Tab. 1), this striking discrepancy between the predictions of the two models may be attributed to differences in their respective spatial scales and hydraulic models.

Simulations of water velocities in the SSIIM are derived theoretically. The small-scale field data (including velocity measurements from more points than the HABITAT) are only used to validate model performance. The conventional HABITAT only simulates one mean velocity per transect, and assign velocities to cells based on proportional linearity with field measurements. Which model is more realistic remains to be demonstrated, but theoretically the SSIIM is because of its smaller scale. From a management point of view, the two models suggest very different solutions to the same problem.

The investigated flow range of 2-5 m^3s^{-1} may seem narrow. However, the greatest changes in habitat conditions tend to take place at the lower flows which we therefore chose to investigate.

Depth

We found considerable differences also with respect to both quantity and trends in simulated suitable water depth habitat versus flow (Fig. 4). The conventional 2D model show decreasing preferred depths with increasing flow, from 40% of total area at 2 m^3s^{-1} to 25% at 5 m^3s^{-1}. On the other hand, the 3D model indicated a stable volume of about 45% within the simulated flow range. Indifferent depths were more similar, yet the 2D model suggested a slight increase and the 3D a decrease. Again trends in avoided habitat were the same, an increase with increasing flows. As with mean water velocities, the biological inputs of depth preferences are identical. Again, this suggets that the differences are caused by diffent spatial scales and calculation procedures in the hydraulic models. The water surface levels in the two models are almost identical. However, simulations in the SSIIM include more small-scale and detailed data on the bottom topography, while the conventional HABITAT have no data from between the transects. Therefore, it is likely that the SSIIM simulations are more realistic. The difference in scale amounts to differences in trends and quantified suitable depth habitat.

Snout water velocities

As might be expected, the SSIIM indicated a relative reduction in suitable snout velocity volumes with increasing flow (Fig. 4). As flow increased, the suitable habitat contracted to a thin layer close to the bottom and smaller volumes close to the river banks. This is consistent with field observations (e.g. Heggenes et al. 1996), with increasing velocities salmon are found in closer association with the river bed. As flow decreased, the relative velocity differences in the vertical dimension increased. We speculate that this suggests greater bias for habitat-hydraulic models, which are based on calculation

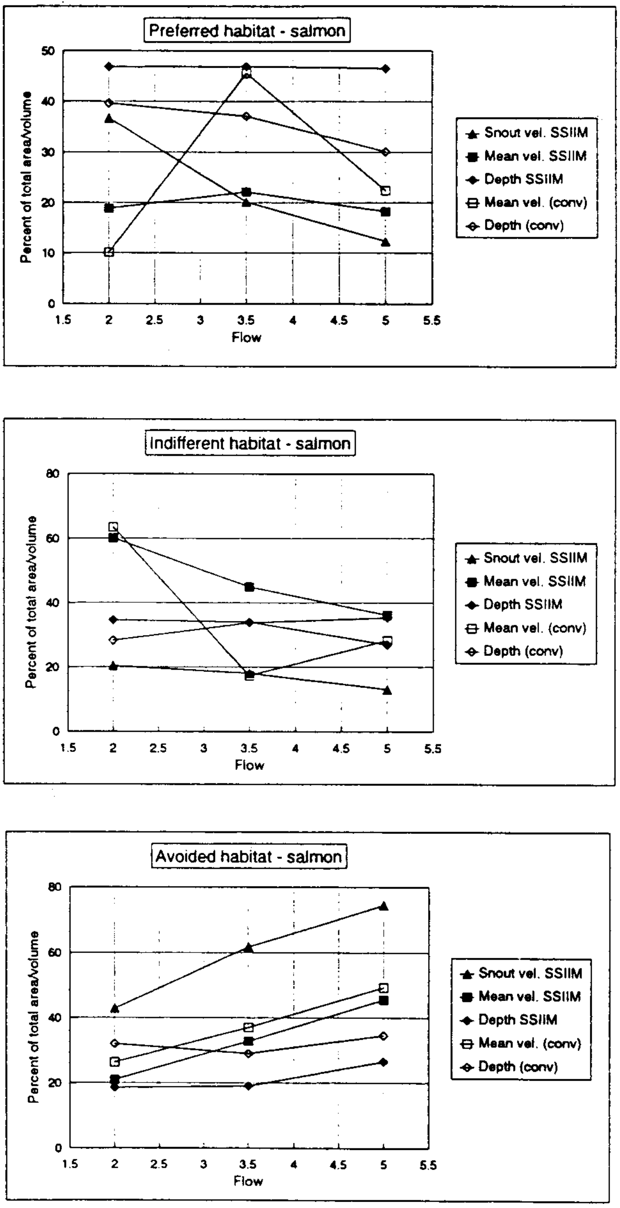


Figure 4. Modelled snout water velocity, mean water velocity and water depth suitabilities in a 122m river section for Atlantic salmon parr, and with increasing flow in R. Gjengedalselva september 1995. Simulations for 3D small-scale SSIIM model are compared with conventional 2D HABITAT model.

of mean water velocities, at lower flows. Small-scale 3D models incorporating snout velocities are likely to become more realistic when the relative velocity differences in the vertical direction increases, usually at low flows and/or when stream morphology is complex. An assessment of suitable minimum flow is often the problem we try to address by using habitat-hydraulic modelling.

We did an exploratory test of the influence of spatial resolution *per se* by doubling the number of grid cells in the flow direction in SSIIM simulations for the 3.5 m^3s^{-1} flow. The results (Fig. 5) suggested no important changes in predicted habitat suitability for snout water velocities, which presumably in the most scale sensitive habitat variable. This may suggest that the SSIIM simulations close to the river bed (and river banks) are not precise enough for microposition modelling. This can be improved by using more topographic data and a porosity option (Olsen 1995). Olsen and Stokseth (1995) found reliable SSIIM results also close to the river bed using this approach. Also, in this modelling we did not include biological modelling of a nearby fast current (providing food, above)

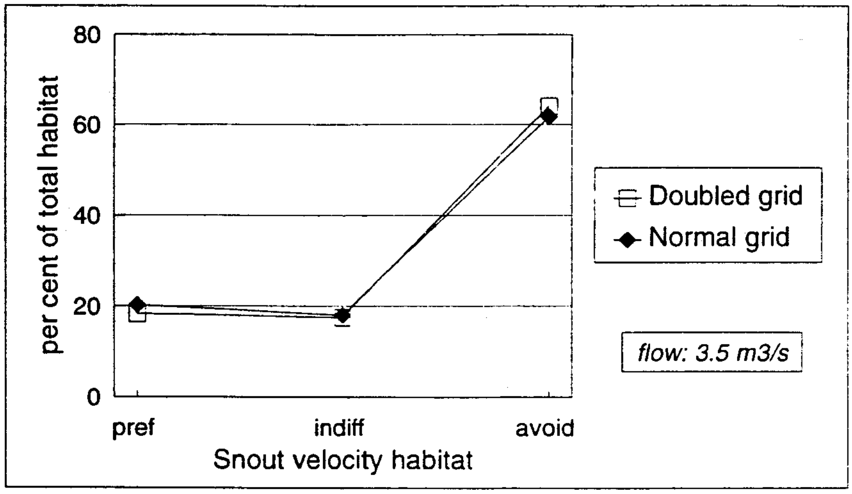


Figure 4. Modelled snout water velocity suitabilities for Atlantic salmon parr at 3.5 m^3s^{-1} waterflow in R. Gjengedalselva september 1995. Simulations for 3D small-scale SSIIM model at normal and doubled grid are compared.

If the two model approaches were used independently to find an optimal minimum flow with respect to Atlantic salmon habitat and for the investigated river section, the conclusions would have been different. The 2D larger scale HABITAT model clearly indicated 3.5 m^3s^{-1} as an optimum flow, both considering mean water velocities and depths. In contrast, the 3D small-scale SSIIM indicated little variation is preferred mean velocities and depths within the investigated range of flows (2-5 m^3s^{-1}). Also considering indifferent habitat, a lower optimum flow appears to be the better solution. Simulations of preferred snout water velocities would support this conclusion. Which solution is the 'correct' solution, i.e. which model is more realistic? It seems obvious that 3D small-scale modelling has greater potential for realistic modelling, but it is yet in its infancy. One step is to refine modelling procedures, e.g. with respect to porosity and topography. A particularly interesting aspect is the potential for more sophisticated biological models, e.g. incorporating scale and time. Another step is of course to test the model predictions with in situ field data.

REFERENCES

French, R.H. (1986). Open-channel hydraulics. McGraw-Hill Int. Ed.

Harby, A. and Heggenes, J. (1995). Habitat User's Manual. In Alfredsen, K., Bakken, T.H. and Killingtveit, A. (eds.) *The River System Simulator. User's Manual.* SINTEF NHL, report 1995.

Hart, P.J.B. (1993). Teleost foraging: facts and theories. In Pitcher,T.J. (ed.): Behaviour of teleost fishes. Chapman and Hall, Fish and Fisheries, series 7, pp. 253-284.

Heggenes, J., Bagliniere, J-L. and Cunjak, R. (1996). Environmental variability in spatial niche relationships for young Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) in hetergeneous streams. Submitted.

Hill, J. and Grossman, G.D. (1993). An energetic model of microhabitat use for rainbow trout and rosyside dace. *Ecology* 74, pp. 685-698.

Hughes, N.F. 1992. Selection of positions by drift-feeding salmonids in dominance hierarchies: Model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. *Canadian Journal of Fisheries and Aquatic Sciences* 49, pp. 1999-2008.

Olsen, N.R.B. (1995). A three-dimensional numerical modelling of sediment movements in water intakes with multiblock option. Version 1.4 beta. User's Manual. University of Trondheim, Trondheim.

Olsen, N.R.B. and Stokseth, S. (1995). Three-dimensional numerical modelling of water flow in a river with large bed roughness. *IAHR Journal of Hydraulic Research* 33.

US Army Corps of Engineers (1991). HEC-2. Water Surface Profiles, User's Manual. Hydraulic Engineering Centre, US Army Corps of Engineers, Davis, California, USA.

Validation strategies of habitat models

Stratégies de validation des modèles d'habitat

TWO-DIMENSIONAL HABITAT MODEL VALIDATION BASED ON SPATIAL FISH DISTRIBUTION : APPLICATION TO JUVENILE ATLANTIC SALMON OF MOISIE RIVER (QUÉBEC, CANADA)

Paul Boudreau[1], Gilles Bourgeois[2], Michel Leclerc[1], André Boudreault[2] and Louis Belzile[2]

[1] Institut national de la Recherche scientifique - Eau
2800 Rue Einstein, C.P. 7500, Sainte-Foy (Qc),
Canada G1V 4C7
paul_boudreau..., michel_leclerc@inrs-eau.uquebec.ca

[2] Groupe-Conseil GÉNIVAR Inc.
5355 Des Gradins, Québec
Canada G2J 1C8
gbourgeois..., aboudreault..., lbelzile@genivar.com

ABSTRACT

A two-dimensional (2-D) application of juvenile salmon habitat modeling is presented. A reference study site on the Moisie River, where a projected water diversion has been proposed, was chosen to implement the application. Validation of the model and implications for instream flow requirements recommendations represented the key issues of this research. The retained validation procedure which had to be adapted to a large river context is based on the aptitude of the habitat suitability indexes (HSI models) to reliably represent the habitat selection behaviour by fish. It also focusses on the use of distinct biological data sets to implement the comparative analysis between observations and predictions, and on the use of reliable physical data provided by a 2-D hydrodynamic model based on a Triangular Irregular Network (TIN). A brief description and a discussion on the use of this particular "finite element" modeling tool are given. Comparison with the 1-D approach and their appropriate use among different flow management or habitat enhancement contexts is also discussed. Apparent inconsistencies in HSI established for different years and hydrological conditions are presented and discussed in relation to instream flow recommendations. Results of the validation procedure are included together with a description of the consequences of the use of this particular validation strategy on instream flow requirements recommendations. The expected output of these recommendations would be an "attenuated habitat regime" with a more sustained availability of juvenile fish habitat.

KEY-WORDS: Two-Dimensional model / One-Dimensional model / Model validation / Juvenile habitat / Atlantic salmon / *Salmo salar* / Habitat suitability indexes / HSI / Triangular Irregular Network / TIN / Hydrodynamic model / Physical habitat / Finite element method / Spatial validation strategy / Instream flow requirements / Habitat selection

INTRODUCTION

Context

The Moisie River flows into the gulf of St. Lawrence on its northern shore at approximately 20 km west of Sept-Îles, Quebec (50° 12' N, 66° 05' W; Figure 1). The Moisie River watershed drains an area of 19 248 km^2, which is characterized by a dominant Precambrian bedrock covered by acid soils and a boreal forest dominated by black spruce *Picea mariana*, balsam fir *Abies balsamea* and white birch *Betula papyrifera*. The mean annual discharge of the river at its mouth is 436 m^3/s, reaching maximum in spring and minimum in late winter. The most common fish species is the Atlantic salmon *(Salmo salar)*, followed by the brook trout *(Salvelinus fontinalis)* (Leclerc *et al*, 1995; Boudreault *et* al, 1989).

Hydro-Québec, a public hydroelectricity utility, has undertaken studies on the energy potential of the drainage basin of the Sainte-Marguerite River, a neighbour of the Moisie River. The preliminary planning scheme includes the diversion into the Sainte-Marguerite River of 74% of the discharge of the Aux Pékans and Carheil Rivers, two upstream tributaries (Figure 1). This would represent a 42% reduction of the annual average discharge of the Moisie River at the confluence with the Aux Pékans and 13.4% at the mouth of the river.

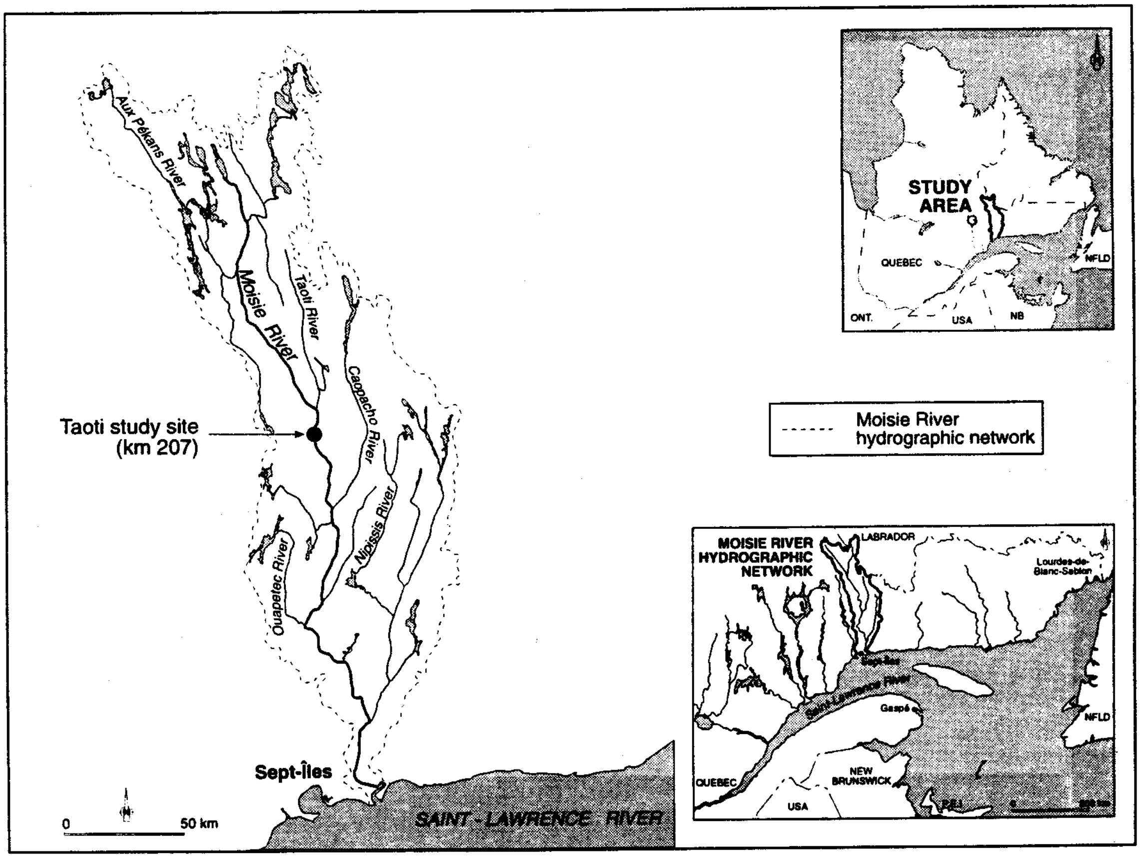


Figure 1 : Moisie River watershed and location of Taoti study site

A major concern of these studies was to determine a minimum recommended flow for the preservation of Atlantic salmon habitats, which motivated the development and application of two-dimensional rearing and spawning habitat models (Boudreault *et* al, 1989; Leclerc *et al*, 1995). More recently, complementary habitat model validation studies appeared necessary to increase confidence in the tools, in particular, for juvenile habitat modeling. Different aspects of the methodology, in particular, the sensitivity of the habitat model to an increase in the precision of the physical model, and the representativeness of habitat preference curves with regard to the spatial distribution of juvenile fish population were identified as important concerns. The first concern was more methodological as it did not represent a key element of decision for the project. The second aspect related to model validation has been identified as a strategic issue by regional pressure groups which participated in the public hearings on the project.

The validation issue

The most common validation strategy for habitat model relies on the functional relation between two data sets: simulated habitat availability - *vs* - fish biomass. The *suitable fish habitat availability* (Weighted Usable Area: *WUA*) is established for limitative phases of the life cycle of fish and for critical flow discharges, usually low flows. The biomass is usually estimated for the corresponding phase and spatial scale, i.e., the river reach under study, to insure the homogeneity of the two data sets. However, this procedure is very costly to undertake as it requires an intensive monitoring of total biomass over several years. In fact, the relatively high degree of uncertainty related to both data sets leads to large intervals of confidence of correlation computations and the only way to reduce the uncertainties is to increase the sample size. It is also possible to raise the precision of each data set by either increasing the field effort or by using a better physical habitat model. When successfull, this classical strategy offers the highest degree of confidence with regard to the habitat - flow discharge relationship as it is based on an explicit relation with biomass. However, due to the cost of such a validation effort, it is only affordable on small rivers.

Despite the difficulty of applying this methodology, some authors successfully found such habitat - discharge - vs - biomass relationships using the PHABSIM modeling approach: Capra (1995), Nehring and Anderson (1993), Jowett (1992), Souchon *et al* (1989), Bovee *et al* (1988), Gowan (1984), and Orth and Maughan (1982), for instance. Some others did not succeed in finding such relationship: Conder and Annear (1987), Scott and Shirvell (1987) and Shirvell and Morantz (1983),. This does not invalidate the methodology because the failure may be attributed to several factors such as the intrinsic uncertainties of the different data sets, insufficient sampling effort and corresponding large confidence intervals. One can also invoke the failures to satisfy underlying model hypothesis, e.g. optimal occupation by fish, territorial behaviour, or representativity of Habitat Suitability Indexes (HSI). The non-limitative role of velocity, depth and substrate variables in habitat definition is proposed as an explanation for failure by Conder and Annear (1987). Morhardt and Altouney (1988) and Fausch *et al* (1988) consider that the plant cover on banks, a variable that is not usually considered by classical models, plays a significant role on habitat quality. Obviously, one can argue that this factor is mostly significant for streams. Again, this statement does not invalidate the methodology, it simply points out the weakness of a model neglecting a significant variable. Baltz *et al* (1987) consider that temperature should be included as a basic variable to improve the model. Again, they point out the representativity of retained variables. Finally, Bourgeois (1992) and Bourgeois *et al* (1996a) consider that some model validation failures might be attributed to the intrinsic limitations of PHABSIM paradigm to represent spatial heterogeneity of physical variables, and consequently, the physical habitat.

A validation strategy adapted to a large river

For juvenile habitat model validation on the Moisie River, we were confronted to major constraints while choosing a proper strategy. The river gabarit (discharge usually >150 m^3/s; depths often >1.5 m, velocity often > 1.5 m/s, etc...) restricted the use of electro-fishing to determine either the biomass or absolute densities. Usual validation strategies based on *WUA* for a critical flow - vs - biomass could not be undertaken efficiently and affordably. An alternate strategy was then developed in order to validate the fish habitat suitability model (HSI) by correlating directly the habitat value and a relative abundance determined by divers observations.

This approach does not determine a critical flow for fish populations, but at least, it increases the confidence in the *WUA*-discharge relationship as estimated with the retained HSI, and it predicts a probable fish distribution for a given flow range. In absence of a clearly identified "critical minimum flow" for juvenile fish populations during the rearing period, the determination of a guaranteed minimum flow will then be based on the maintenance of close-to- optimum conditions as demonstrated by the *WUA*-discharge relationship. Such a rational would then consequently lead to a decrease of summer flow discharges during wet years (as a result of the diversion) and a corresponding increase during dry years (release from the storage reservoir via the diversion dam). This approach to flow regulation can be interpreted as a "low peak attenuation strategy" resulting into a higher and more sustained habitat availability.

METHODOLOGY

Moisie River habitats : rational of reference site selection

The river reach which has been selected for habitat studies is called Taoti and it is located 207 km upstream from the Moisie River mouth and 90 km below the proposed diversion. It is only accessible by air. The complete rational for selecting this site has been presented in Boudreault *et al* (1989) and in Leclerc *et al* (1995). The main criterias were : presence in great abundance of spawning areas and of juvenile habitats, best overall quality of habitats within the river, sensitivity of the site to the flow reduction by the proposed diversion. The retained site has the following dimensions : 1020 m lenght and ~300 m average width. Mean summer discharge stands around 168 m^3/s and the projected diversion would reduce it by 33% without flow release. Geomorphologically, the site is a typical shallow-water breaded reach. Figures 2a and 2 b show site maps for bathymetry and substrate classification.

Sampling : physical data

Two field sampling campaigns were conducted in 1994 and 1995 for physical characterization. The first one allowed to record the bathymetry along transverse sections with a total amount of 12 000 measurement points which provide a very good estimate of the morphology. Positioning was obtained by using a transit while depths were measured by echosounder (Hydro-Québec, 1995). Substrate size and composition were characterized by visual interpretation over 576 patches (9 m^2 area each) as described by Bourgeois *et al* (1996b). Local percentage of coverage by each of 5 characteristic substrate sizes (boulders, cobbles, pebbles, gravel, sand) was estimated over the patches. Extrapolation over the entire river reach was obtained by a meso-scale visual interpretation procedure (delineation of homogeneous areas). A DGPS (Differential Global Positioning System) was employed successfully in 1995 (Trimble ProXL instrument) for positioning.

Among physical variables that need to be established in the field for hydrodynamic model implementation, one can mention stage-discharge relationship at the upstream and downstream sections of the study reach (used for driving boundary conditions within the two-dimensional model); position of the dry-wet limits (wetted-area) for two distinct flows (135 and 160 m^3/s) to validate the drying-wetting capabilities of the hydrodynamic model; velocity measurements (vertically integrated) for discharges of 86 and 197 m^3/s ; and water level corresponding to 81 and 197 m^3/s flows (both data sets used for hydrodynamic model validation purposes).

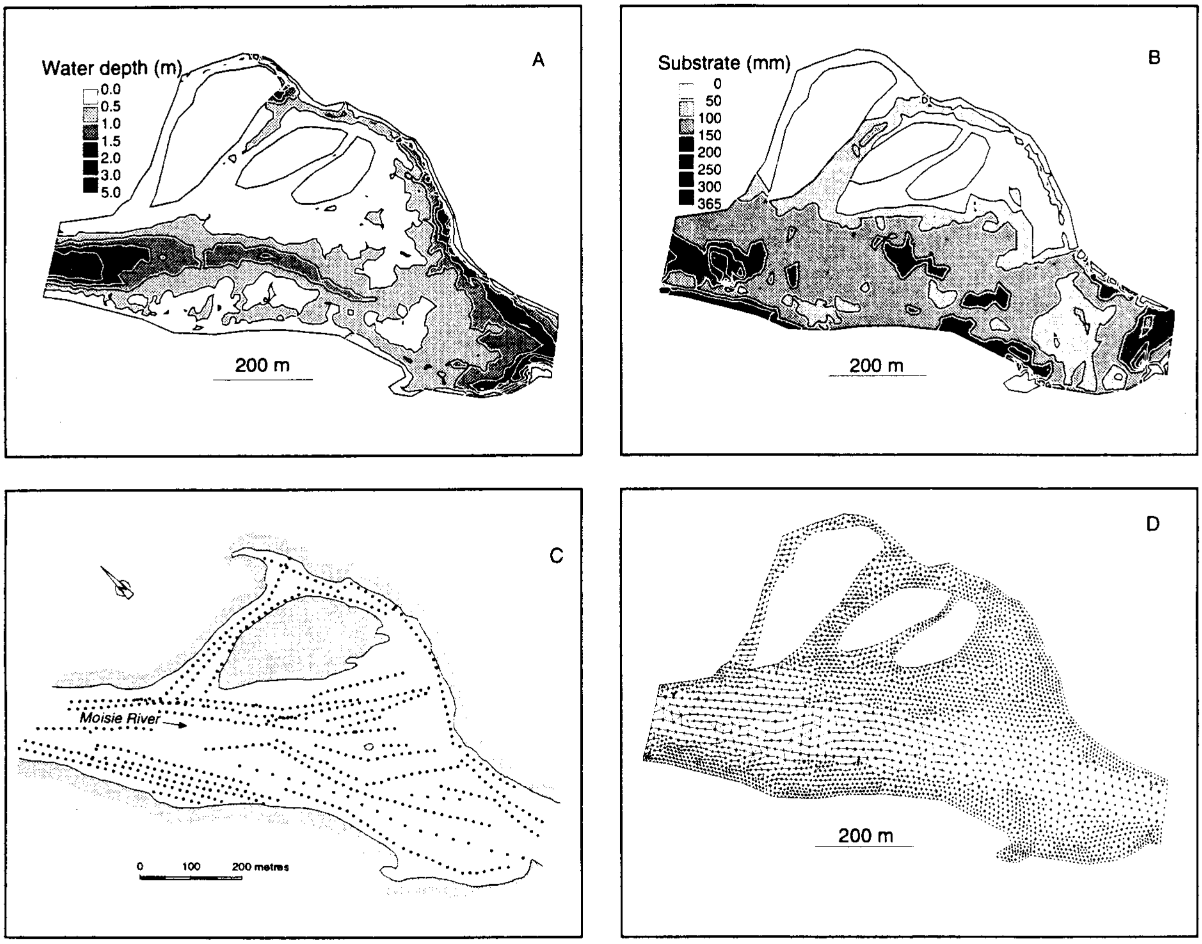


Figure 2 : Taoti study site: a) depth for a dischage of 197 m^3/s ; b) substrate classification (mean grain size); c) biological sampling stations; d) finite element mesh (FEM)

Sampling : biological data

Observation of the presence-absence and number of juvenile Atlantic salmon (0^+, 1^+, 2^+) was realized by snorkeling identification on patches of 9 m^2. The observed number of juveniles in each patch was later translated into a "relative" density for the validation procedure. The main goal was to acquire a data set representing the relative fish distribution over the study site (Bourgeois *et al*, 1996b). Figure 2c depicts the biological sampling effort by showing the patch distribution on the site. Local velocity and depth were also measured at each patch. These values were not used in the physical modeling because of the high heterogeneity of the data set with respect to flow discharge. Measurements were done at each fish location (for presence of fish), or at the patches center (absence).

The summer of 1995 was characterized by of a very severe drought possibly critical for fish population (probability of once every 50 years; discharge of 50-60 m^3/s typical of the end of August and relatively high water and air temperatures). These particular conditions provided a very contrasting flow state with regard to fish habitat distribution under more usual discharge conditions. Therefore, this rare situation in 1995 is susceptible to increase the reliability of the habitat model validation exercice due to the fact that the fish's preferendum was established in 1988 during a previous study while flow discharge stood around 130-160 m^3/s (Leclerc *et al*, 1995; Boudreault *et al*, 1989).

Two-dimensional (2-D) drying-wetting hydrodynamic model

A complete description of the 2-D hydrodynamic model used for this study is available in Leclerc *et al* (1990a,b) and Boudreau *et al* (1994). A more specific description of the tool for habitat modeling purposes is provided in Leclerc *et al* (1994, 1995, 1996). Briefly, this model provides dynamic drying-wetting capabilities, an essential feature to deal with natural watercourse processes where the lateral flow boundaries move continuously with flow discharge or tides. As in every 2-D horizontal model, velocities are vertically integrated (mean value). The model uses a so-called "non-conservative" mathematical formulation (velocities and water levels as primitive state variables) based on St-Venant, or shallow water equations. It also uses a finite element (often called TIN for Triangular Irregular Network) spatial and numerical discretization scheme which is reputed to provide the best interpolation method to reproduce adequately the field data within the so-called "Numerical Terrain Model" (NTM). Finally, the program offers the possibility to deal with transient flow processes (e.g., tides) but this feature is not currently used for rivers which usually have a quasi steady-state behaviour.

Physically, when compared to one-dimensional (1-D) solution tools, this category of models offers more reliable results for use in habitat models. In fact, complete 2-D models cover the entire range of forces and acceleration that occur in the natural flow equilibrium, i.e., *local and convective accelerations*, *gravity* (slope), *bottom, plant and ice resistances, wind stresses, turbulent shear stresses and compressions, and Coriolis force*. The calibration procedure is also simplified as it only relies on a local functional relationship between substrate or macro-rugosity sizes and roughness coefficients (Boudreau *et al*, 1994; Leclerc *et al*, 1995). Therefore, this parameter does not depend on flow discharge as in 1-D models which set this parameter as a global flow dependent transverse section property instead of a local (x, y) property.

Figure 2d shows the discretization mesh of the Taoti site which comprises 5 841 triangular elements and 12 227 computation nodes. Such a mesh size is necessary to represent adequatly the field data variability, especially, bathymetry. But the NTM is still a model and consequently, it needs to be validated. However, as the mesh construction has to be performed by using automated meshing tools (like the MODELER developed at INRS-Eau) to achieve the job readily, one should be carefull with high level of automation. In fact, the tool should offer the users a minimum of visual control over the field data transfer process. Figure 3 depicts a validated computed flow field for a discharge of 86 m^3/s, a low flow state. Notice the dry areas identified automatically by the model during simulation. The most usual range of summer hydrological conditions (55-300 m^3/s) represented by fourteen different flow-events was simulated for habitat modeling purposes.

Habitat suitability models

Different sets of HSI curves were developed for salmon parr rearing habitats. Each one takes into account the availability of physical conditions within the river to identify fish's *preferenda* instead of simple utilization curves (Bovee, 1986). Two biological models were established, each of them comprising the suitability curves (basic indexes) for velocity, substrate and depth, and a global index computed by a geometric mean of basic indexes, as follows:

$$IG = I_V^a \cdot I_H^b \cdot I_S^c \qquad (1)$$

I_V, I_H and I_S are respectively basic indexes for velocity *(V)*, depth *(H)* and substrate size *(S)*; *a, b and c* represent weighting factors, the sum of them equals 1.0. These factors (Table 1) were obtained from a principal component analysis (PCA) and were determined from the relative contribution of each variable to the explained variance.

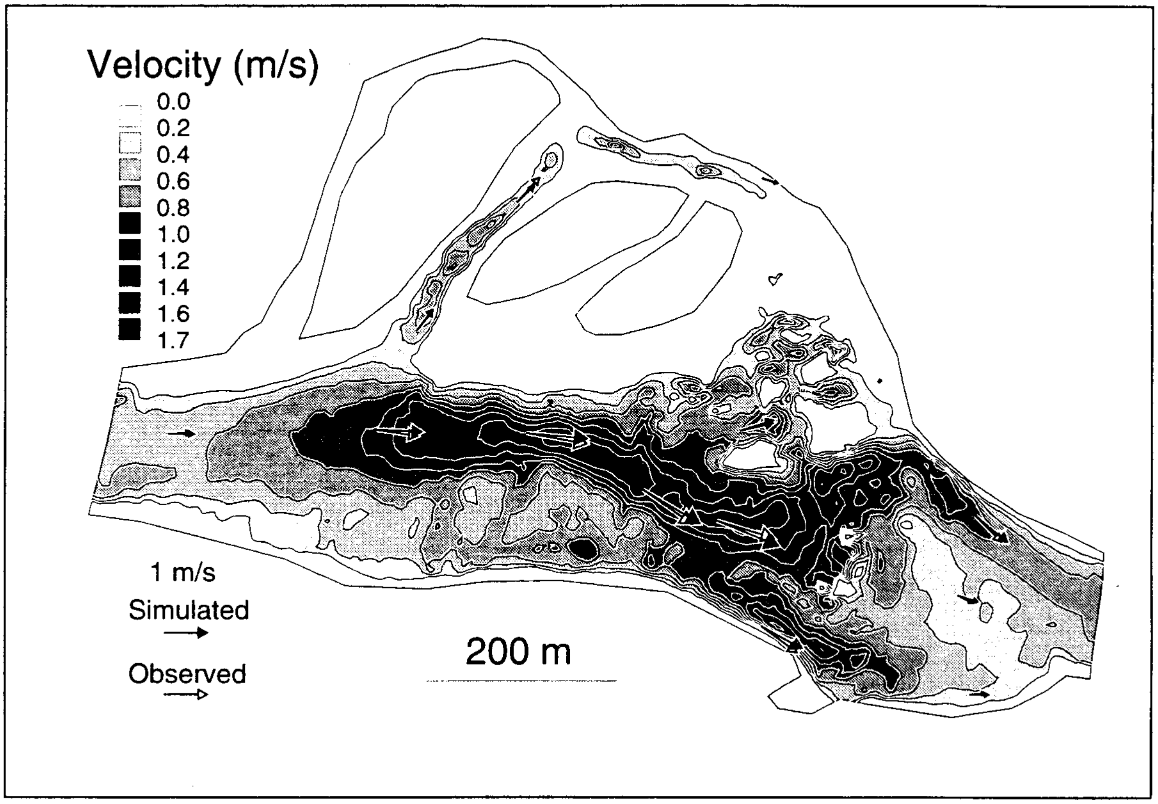


Figure 3 : Velocity module for a discharge of 86 m³/s and measurement-prediction comparisons

The first model (M-88), targetted for biological validation purposes, was based on 1988 biological data obtained from several study reaches in the Moisie River (Boudreault *et al*, 1989), and on more usual hydrological conditions than in 1995. Following recommendations by an independent scientific steering committee which controlled the final parametrization and use of these curves in 1989, some adjustments were made to them, especially at the extremes of variable ranges in order to give back some habitat value to physical conditions pertaining to these ranges. This approach was implemented to obtain a more conservative habitat model for all conditions. The adjusted parts of the curves are indicated on Figure 4. The physical variables that were considered then were mean velocity (4 a), depth (4 b) and mean grain size (4 c). The second model (M-1995) was set up using the same procedure with 1995 data (Figure 4). As opposed to M-88, no adjustements were made on these curves. One can readily note that not all the HSI curves are consistent over the years. This result is probably explained by the significant difference between the hydrological conditions in 1988 and 1995. The discrepancies between the two models and their consequences on habitat availability will be discussed later.

Table 1 shows the different sets of weighting factors used for computing the habitat value through weighted geometric mean. Again, one can observe some discrepencies between the different sets of parameters. When compared, M-88 and M-1995 point out to an increasing importance devoted to velocity and substrate by fishes in 1995 when hydrological conditions were more severe.

Table 1: Weighting factor for global index *(IG)* computation

Habitat model	Data set for PCA	Velocity factor *a*	Depth factor *b*	Substrate factor *c*
M-88	1988	0.39	0.4	0.21
M-95	1995	0.53	0.09	0.38

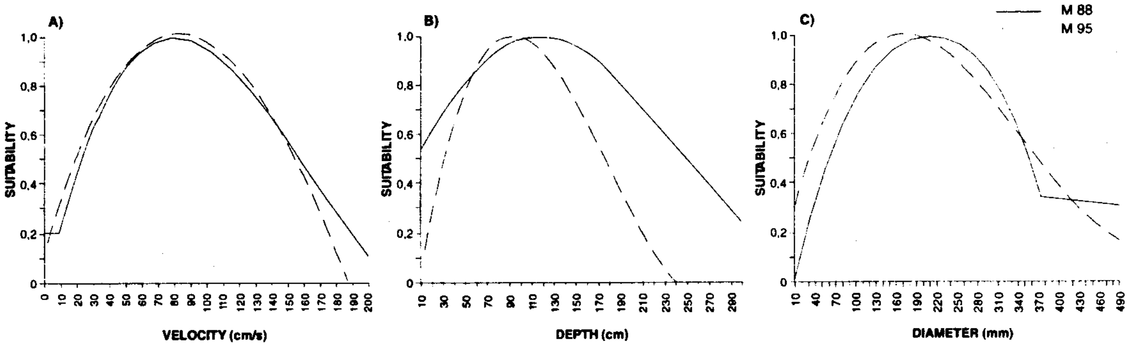


Figure 4 : Salmon parr Habitat Suitability Indexes for a) velocity; b) depth; c) substrate mean size

Concepts of the validation method

Relative abundances of juvenile fishes were correlated to corresponding local habitat value taking into account the particular daily discharge during the biological survey. Local values of physical conditions were obtained from the 2-D hydrodynamic model by interpolating the velocity-depth results between the available simulated flow fields.

Strictly, the highest confidence in the validation procedure is obtained by using the 1995 biological data set to validate M-88. This is highly suggested by a basic rule of model validation which states that a model developed (calibrated) with a given data set (set A, 1988 survey) can not be validated with the same data. This implies the use of distinct data sets (B, 1995 survey) obtained preferably under different conditions, to avoid a potential "totology". That is why, herein, the best validation exercice is provided by comparing 1995 relative fish abundance to habitat value computed from suitability curves determined from the 1988 survey.

A second rule that has to be considered in validating models is related to the *homogeneity* of data sets to be correlated. Ideally, the two data sets should share 1) the same units, 2) the same spatial characteristics (scale and location) and 3) the same temporal characteristics (duration period, time of occurence). When the model aims to establish a causal relation between an independent variable (habitat value) and a dependent one (fish distribution) the units may be distinct. In the particular case of the Moisie River, the two latter principles were strictly observed. Moreover, in order to increase the independence of the data sets under comparison, physical values of fish habitat were established from the hydrodynamic model instead of from direct observations within the patches.

A similar comparison exercice were conducted by using M-95 to evaluate the sensitivity of distinct parametrizations within the habitat model. This sensitivity analysis was based on the habitat availability *(WUA)* response to different models with respect to discharge. In 2-D finite element models, functional relationships between *WUA* and discharge can be defined from spatial integration of the flow domain with respect to local habitat value *(IG)* for each discharge.

$$WUA = \int_{\text{Flow domain}} IG(x,y)dA \equiv \sum_{NE} A^E \cdot \overline{IG^E} \tag{2}$$

Where A^E is the considered triangular element, NE the total number of elements within the mesh.

RESULTS

Physical model validation

The validation procedure for hydrodynamic models compares observation - vs - predictions for different variables, usually velocities and water levels. This procedure is based on the verification of the agreement of two unit-homogeneous data sets. Our experience of validating hydrodynamic models has led us to consider also the discharges (imposed to the model - vs - realized through the resolution process, and mass conservation within the model) and the lateral flow boundaries agreement (observed - vs - simulated). Herein, only the global agreement metrics will be reported.

For the fourteen different flows under consideration (ranging from 55 to 300 m^3/s), disagreement between imposed and computed discharges stands at around 2.4% (0.9% upstream limit; 4% downstream limit). The mass conservation, i.e., the difference between computed discharges at the upstream and downstream limits, is then achieved with a 4.3% mean discrepancy. The better agreement for upstream limit is related to the fact that it corresponds to the upstream discharge boundary condition. The stage-type boundary condition imposed at the downstream limit is a necessary scheme to ensure "unicity" of the solution. The discharge calculated therein produces poorer results. Nevertheless, such an agreement is considered very good.

As for the lateral position of flow limits, the validation procedure is performed by direct comparison of predicted and observed locations. In general, a good agreement reveals that bathymetry has been well accounted for within the NTM, especially along the banks, and also that the water surface level was well predicted. Again, the validation is event-dependent. Two discharges were considered for this comparison: 135 and 160 m^3/s. No metrics are available to compare the data sets except visual interpretation. Generally, very good agreement was achieved, especially when the bank slopes were steep. When horizontal flat zones are considered, only a small disagreement between measured bathymetry and NTM, or between water level data sets can lead to rapidly increasing discrepancies. Moreover, it is very difficult, both in the model and in the field, to evaluate homogeneously this information. This happened in a single area (less than 5% of the total boundaries) within the simulation domain where the habitat is not considered good for juveniles due to small grain size, low velocities and sahallow water (Gibson, 1993). The consequences are then negligible for fish habitat estimates .

For velocities, two discharges (86 and 197 m^3/s) were considered for a total amount of thirty local observations (Figure 3 for 86 m^3/s). Vertical homogeneity of data sets is assured by vertical integration of velocity, both in the field (average velocity at 0.8 and 0.2 x depth), and in the model (vertical mean as state variable). Horizontal scale of homogeneity is more difficult to achieve as the velocity is measured very locally while the simulation results represent more of a regional value. Nevertheless, a mean absolute deviation of 8,8% was achieved as a result which is considered a very good agreement. Moreover, 83% of comparison points stood within a 15% deviation limit while only two points showed poorer results.

A similar exercice was conducted for water levels. Usually, the agreement is very good in 2-D models and this statement was again confirmed by discrepancies exceeding rarely ±10 cm, even at the upstream limit where the imposition of boundary conditions is achieved through discharge alone.

Two-dimensional habitat distribution

Application of 2-D habitat modeling gives as a basic output a precise spatial distribution of habitat value for each life stage. As a result of the use of accurate 2-D hydrodynamic model and of the finite element method to represent distributed variables, very contrasted and nuanced habitat maps can be drawn readily. Figure 5 depicts a typical map of salmon parr habitat (M-88 model) for a close to optimum discharge of 130 m^3/s. One can readily observe that, for this discharge, most of the Taoti study site is formed of very good parr habitats (*IG* >0.7), the best values being located close to upstream and downstream limits of the flow domain. One can also note that the habitats situated in the middle zone are characterized by a patchy structure.

Sensitivity of habitat availability (*WUA*) to HSI model

To illustrate the sensitivity of *WUA* to fish *preferenda*, Figure 6 depicts results for parr when using M88and M95. The typical behaviour of these curves is classical : a quick increase of habitat availability with discharge below an optimum value, an optimum flow value and a slight decrease with discharge within the flood range. One can notice also that the *WUA* estimated from M88 is significantly higher (2-5%) for the most usual flow range (>125 m^3/s). This behaviour is related to the fact that this model attributes a higher value to the higher range of physical conditions as a result of both, the analysis of the 1988 data set, and the adjustements made by the scientific steering committee. Nevertheless, the global aspect of these curves is very similar as they remain almost parallel. In fact, the optimum *WUA* values stand around the same discharge, and increase or decrease of habitat availability with discharge occur within the same flow ranges. Therefore, once converted into Instream Flow Needs, discrepancies between the different models in use for the study would not lead to contradictory conclusions. The recommendation for minimum garanteed flow at Taoti site was 135 m^3/s (Boudreault *et al,* 1989).

Habitat model validation

Figure 7 shows the ultimate result of the validation exercice of juvenile salmon habitat model on the Moisie River. On this histogram, one can see that habitat value *(IG),* as computed by M88 model and predicted values of physical conditions, is represented as the independent (explicative) variable, and the apparent densities observed in 1995 (number of parrs / area of patch) play the role of dependent variable. One can observe readily that juvenile salmon express a growing preference for better and better habitats. One can also observe that the relation seems to adopt a parabolic behaviour, i.e., the preference level for an increasing value of habitat does not progress linearly with this variable. This suggest that when integrating linearly the flow domain with respect to habitat value (equation 2) to compute *WUA*, small values of *IG* are likely to provide an overestimate of habitat availability. Other explanations may also be adopted: either the habitat can be under-utilized, or the observer under estimates the fish presence especially when the fish distribution is much diluted within the observed patch. Nevertheless, none of these interpretations contredicts the fact that the fish chooses the best habitats and that the M88 model, which was parametrized with a data set (1988) corresponding to very different physical conditions, succeeded in predicting habitat selection by fish for the 1995 survey.

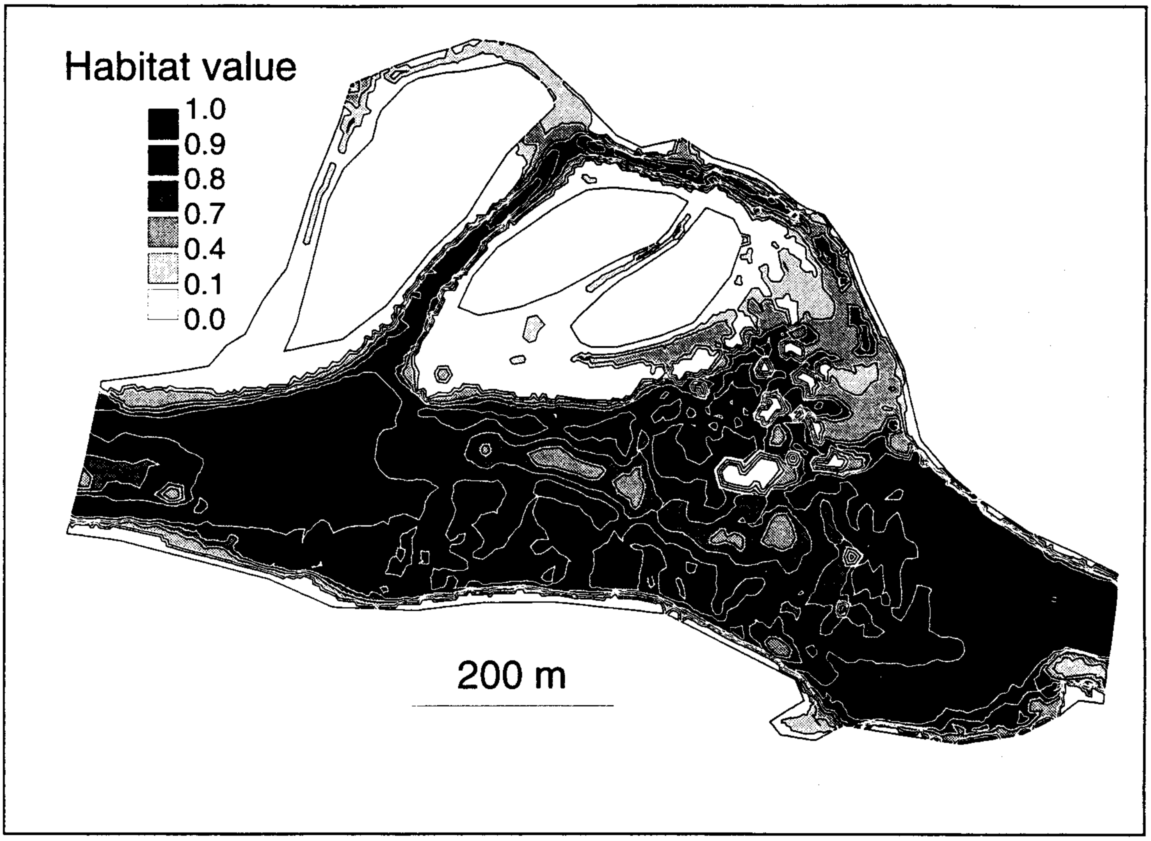


Figure 5 : Parr habitat map for the M88 model and a close-to-optimum discharge of 130 m^3/s

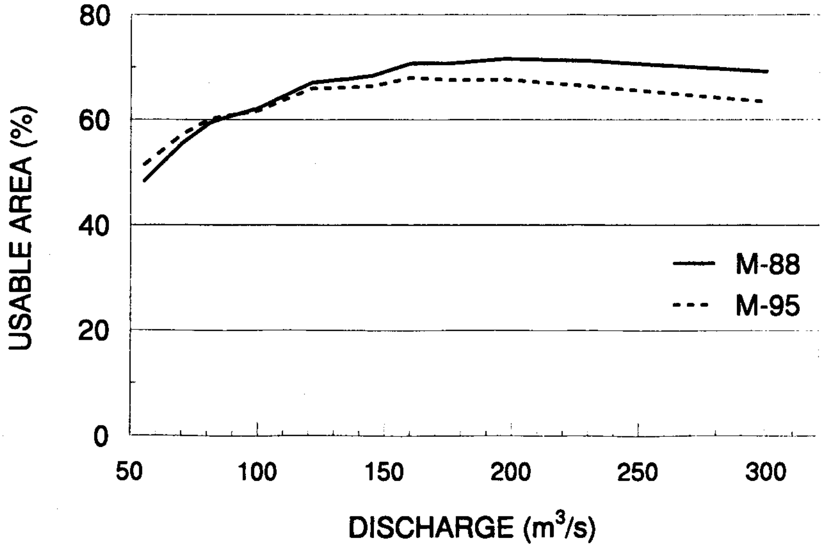


Figure 6 : Sensitivity of salmon parr habitat availability to two HSI models (M-88 and M-95)

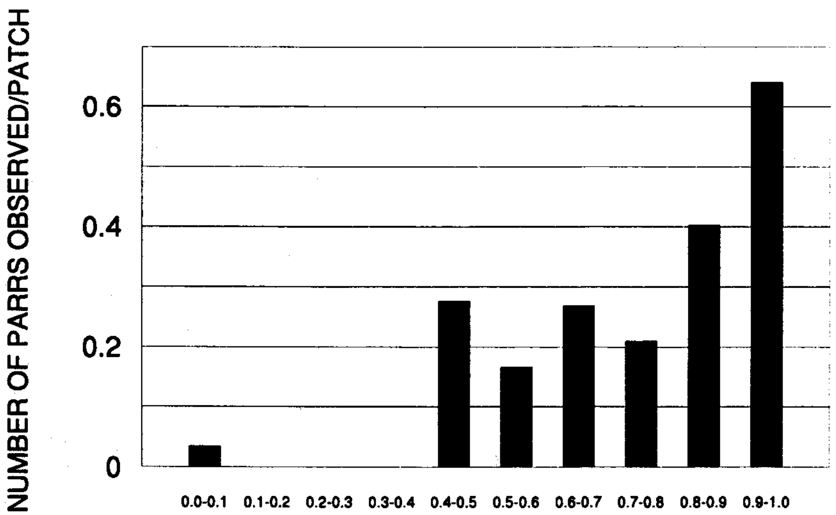


Figure 7 : Validation results of M88 model by using 1995 data of juvenile salmon habitat selection behaviour

DISCUSSION

On the validation strategy and consequences on the decision rational of instream flow requirements

The validation strategy adopted for this large river raises the following questions : what exactly has been validated, and what are the consequences on the decision rational ? As pointed out at the beginning of the paper, the adopted procedure does not lead to an explicit relationship between a "critical flow" (as represented by its corresponding

WUA) and the fish biomass. Such a relation is ideal to set appropriate recommendations of minimum flow requirements, because it reveals the limiting role of habitat on fish populations. In fact, it constitutes the strategic component of the decision rational by anchoring the recommendation in "hard facts".

In our view, the procedure proposed herein permits to validate the *HSI* model itself (together with the weighting factors) which represents the main component of the habitat model. Even if, by doing so, a relation between habitat and biomass is not defined, that does not mean that it does not exist. The question is how to set appropriate and conservative recommendations for flow management ? Our attitude has been to recommend the maintenance of "close to optimal" hydrological conditions in the river as "minimum flow requirement" rational. On the one hand, this approach maximises the habitat availability, especially during naturally dry summers. On the other hand, the diversion itself contributes to the habitat maximization by reducing the extreme summer floods during wet years, thus maintaining habitat conditions closer to the optimum. The result of this approach would then be a habitat low peak attenuation and a sustained high level of habitat availability.

Obviously, additional recommendations have to be proposed for the other critical life stages of fish, e.g., maintenance flow for spring clean up fine material of riverbed, minimum flow requirements for spawning (fall), egg incubation (end of winter) and upstream migration periods. This was done carefully and reported in Boudreault *et al* (1989). However, these subjects were beyond the scope of this paper.

On the two-dimensional modeling approach

There would be a lot to mention on the use of a two-dimensional hydrodynamic modeling approach to spatial habitat analysis. We will only note that such models seem necessary to estimate precise values of physical variables in time and space and as a function of flow. In order to apply the proposed validation strategy, such models appear essential to provide independent local (fishwise positionned) and flow dependent physical data sets to estimate habitat value. In fact, the hydrodynamic model is not only a full part of the habitat model, it is a component of the decision tool and, in this sense, it has to be validated together with the habitat model.

Wherever it is possible to define a "critical *WUA*" - vs - biomass, a classical PHABSIM 1-D model is probably sufficient to set flow recommendations as this approach succeeds in representing the general habitat trends with respect to flow (Bovee, 1996; these proceedings). But our belief is that the 1-D approach, even deterministic with respect to physical conditions, achieve this goal statistically through spatial representativity. However, it can hardly provide accurate, local sensitive habitat values, or be extrapolated to deeply modified physical conditions that would result from habitat enhancements or highly reduced (or increased) flow conditions. This interpretation limits the use of 1-D models to *reach-scale* and *observable* flows dependent state variables, e.g. *WUA*.

In more complex situations, use of 2-D modeling will provide consequently natural analyzing capabilities and features to planners to set more appropriate recommendations for fish habitat protection. Moreover, our experience of 2-D modeling - vs - 1D demonstrates that the cost of 2-D applications is probably comparable, if not cheaper than 1-D in relation to field effort and validation procedure :

- no need to characterize intensively flow velocities, only some validation results are necessary;
- bathymetry can be measured randomly instead of systematically on cross sections;
- consistent roughness coefficients with local value of substrate size, resulting into a single step calibration procedure.

On the reproductibility of suitability curves : consequences on the habitat availability and IFN rational

It has been stated that suitability curves may vary from one year to another within the same river reach . Nickelson *et al* (1992) and Baltz *et al* (1991) concluded to similar statements. One can speculate on the causes of this behaviour: failure to satisfy underlying hypothesis of optimal occupation, intrinsic uncertainties related to the methodology of suitability curves determination, etc... Again the question that can be raised is related to the consequences on the recommendations of flow requirements for fish. It has been observed that even if the curves do not seem consistent over the years, some characteristics remain the same, especially the optimal value of each single variable. In fact, even if extreme ranges of physical conditions are given different habitat values, this register does not play a significant role in the *WUA* estimation over a river section characterized by rich habitats. We also observed that even with distinct habitat suitability models, the behaviour of habitat-flow curves does not seem very much sensitive, remaining parallel (same slope and *WUA* optimum). This results in the same interpretation, relatively, for gain or lost of habitat with respect to flow alteration. Consequently, flow recommendations should not suffer from apparent inconsistencies.

CONCLUSION

In conclusion, the size of the Moisie River imposed the choice of an alternative strategy for validating the juvenile salmon habitat model. This strategy was based on the demonstration of the appropriateness of using 1988 M88 suitability curves to estimate habitat availability with respect to flow. Technically, the use of a two-dimensional hydrodynamic-habitat model was required not only to apply habitat modeling to a huge river but also to achieve the validation strategy itself, by providing accurate fishwise located and flow dependent physical values to habitat computation steps. It has also been demonstrated that, even with apparent inconsistencies in suitability curves over years and different hydrological conditions, basic characteristics of the curves (e.g. optimum value) lead to a successful demonstration of the appropriateness of using the M88 model for defining juvenile salmon habitat.

As for the decision rational, the difficulty in defining a "critical" flow for population of juvenile salmon imposed the modification of the usual approach to recommend minimum flow requirements. It has been decided to focus on the maintenance of optimal conditions, especially during dry years when the modified flow in the river would be maintained even higher than natural conditions. During the wet years, the result of upstream flow diversion would also increase the habitat availability by reducing the high velocities and depths without a significant reduction of the wetted area. This approcah can be interpreted as a "habitat low peaks attenuation strategy".

ACKNOWLEDGEMENTS

We wish to acknowledge all those who were involved in the field surveys but could not be mentioned as co-authors. Thanks also to Mrs. Geneviève Corfa who acted as the research project director for Hydro-Québec, the promoter of the hydropower project. We would like to mention the precious role played by our informatician and numerician collegues of INRS-Eau, especially Mr. Yves Secretan in developping the computer tools mobilized for this study. We wish to express our gratitude to all actual members of the federal-provincial scientific committee, in particular, its experienced chairman Mr. François Caron for his very precious recommendations to conceptualize and implement the validation procedure. Finally, thanks to Hervé Capra, for his judicious criticisms when reading the present paper, and to Ronald Greendale for the English language proofing.

REFERENCES

Baltz, D., B. Vondracek, L. Brown and P. Moyle (1987). Influence of Temperature on Microhabitat Choice by Fishes in a California Stream. *Transactions of the American Fisheries Society*. **116**:12-20.

Baltz, D., B. Vondracek, L. Brown and P. Moyle (1991). Seasonal Changes in Micro-habitat Selection by Raibow Trout in a Small Stream. *Trans. Am. Fish. Soc.* **120:** 166-176.

Boudreau, P., M. Leclerc and G. Fortin. 1994. Modélisation hydrodynamique du lac Saint-Pierre, fleuve Saint-Laurent: l'influence de la végétation aquatique. *Revue canadienne de génie civil*, **21**: 471-489.

Boudreault, A., M. Leclerc, J. F. Bellemare, and G. Shooner (1989). Étude des répercussions du détournement de la rivière Aux Pékans sur les habitats salmonicoles de la rivière Moisie. Report of Gilles Shooner & Ass. to the Direction de l'Environnement, Hydro-Quebec, Quebec City, Canada, 120 pp. and 4 appendices.

Bourgeois, G. (1992). Modélisation de l'habitat physique du saumon de l'atlantique avec PHABSIM: cas du ruisseau Catamaran. Master's thesis. University of Moncton, New-Brunswick, Canada. 124 p.

Bourgeois, G., R. Cunjak, D. Caisie, and N. El-Jabi (1996a). A Spatial and Temporal Evaluation of PHABSIM in Relation to Measured Densities of Juvenile Atlantic Salmon in a Small Stream. *North American Journal of Fisheries Management*, **16**: 154-166.

Bourgeois, G., P. Boudreau, A. Boudreault, M. Leclerc (1996b). Modélisation de l'habitat salmonicole au site Taoti sur la rivière Moisie. Rapport préliminaire présenté par le Groupe-conseil Génivar Inc.et l'INRS-Eau à la Vice-présidence Environnement Hydro-Québec (In preparation).

Bovee, K.D. (1986). Development and Evaluation of Habitat Suitability Criteria for Use in the Instream Flow Incremental Methodology. U.S. Fish and Wildlife Service, *Instream Flow Information Paper* **No. 21**, *Biological Report* **86**(7).

Bovee, K.D. (1988). Use of the Instream Flow Incremental Methodology to Evaluate Influences of Microhabitat Variability on a Trout Population in Four Colorado Streams. In: 68th Annual Conference of the Western Association of Fish and Wildlife Agencies. Albuquerque, USA. 31 pp.

Bovee, K. (1996). Perspectives on two-Dimensional River Habitat Models: The PHABSIM Experience. *In*: Leclerc *et al* (1996). *Proceedings of the Second International Symposium on Habitats Hydraulics - Ecohydraulics 2000*. Québec City, June. Volume B, pp. 149-162

Capra, H. (1995). Amélioration des modèles prédictifs d'habitat de la truite *fario* : échelle d'échantillonnage et intégration des chroniques hydrologiques. Thèse de doctorat de l'Université Lyon 1. 281pp.

Conder, A.L. and T.C. Annear (1987). Test of WUA Estimates Derived from a PHABSIM Model for Instream Flow Studies on Trout Streams. *North American Journal of Fisheries Management*. **7**:339-350.

Fausch, K., C. Hawkes, and M. Parsons (1988). Models that Predict Standing Crop of Fish from Habitat Variables: 1950-1985. *General Technical Report PNWT-GTR-213, USDA*, Portland, USA.

Gowan, C. (1984). The Impacts of Irrigation Water Withdrawals on Brown Trout *(Salmo trutta)* and two Species of Benthic Macro-invertebrates in a Typical Southern Michigan Stream. Master's thesis. Michigan State University, East Lansing.

Gibson, R. J. (1993). The Atlantic Salmon in Fresh Water: Spawning, Rearing and Production. *Reviews in Fish Biology and Fisheries*. **3**: 39-73

Hydro-Québec (1995). Projet Sainte-Marguerite, Rivière Moisie. Rapport final des relevés hydrométriques de l'automne 1994 au site Taoti. Direction Ouvrages civils, Service hydraulique, division Hydrométrie.

Jowett, I.G. (1992). Models of the Abundance of Large Brown Trout in New-Zealand Rivers. *North American Journal of Fisheries Management* **12**: 417-432

Leclerc, M., J. F. Bellemare, G. Dumas, and G. Dhatt. (1990). A finite element model of estuarian and river flows with moving boundaries. *Advances in Water Resources* **4**:158-168.

Leclerc, M., J.F. Bellemare, and S. Trussart (1990b) Simulation hydrodynamique de l'estuaire supérieur du fleuve Saint-Laurent (Canada) avec un modèle hydrodynamique aux éléments finis couvrant-découvrant. *Revue canadienne de Génie civil* **17**(5) : 739-751.

Leclerc, M., Boudreault, A., Bechara, J. and G. Corfa (1995). Two-dimensionnal Hydrodynamic Modeling: A Neglected Tool in the Instream Flow Incremental Methodology. *Transactions of the American Fisheries Society*, **124**: 645-662.

Leclerc, M., P. Boudreau, J. Bechara, and G. Corfa (1994). Modélisation de la dynamique de l'habitat des ouananiches *(Salmo salar)* juvéniles de la rivière Ashuapmushuan (Québec, Canada). *Bulletin Français de la Pêche et de la Pisciculture* **332**:11-32.

Leclerc, M. , J. Bechara, P. Boudreau and L. Belzile (1996). A Numerical Method for Modeling the Spawning Habitat of Landlocked Salmon. *Regulated Rivers: Research and Management* (In Press).

Morhardt, E. and E. Altouney (1988). Instream Flow Methodologies and Requirements. Pages 710-718 in *Proceedings of Waterpower 1987*, American Society of Civil Engineering, New-York.

Nehring R.B. and R.M. Anderson (1993). Determination of Population Limiting Critical Salmonids Habitats in Colorado Streams Using Physical Habitat Simulation System. *Rivers*. **4**:1-19

Nickelson, T. E. J. D. Rodgers, S.L. Johnson and M. F. Solazzi (1992). Seasonal Changes in Habitat Use by Juvenile Coho Salmon (*Onchorhynchus kisutch)* in Oregon Coastal Streams. *Can. J. Fish Aquat. Sc.* **49:** 783-789.

Orth, D.J. et O.E. Maughan (1982). Evaluation of the Incremental Methodology for Recommending Instream Flows for Fishes. *Transactions of the American Fisheries Society* **111**:413-445.

Scott, D. and C.S. Shirvell (1987). A Critique of IFIM and Observations on Flow Determination in New Zealand. Pages 27-43 in J.F. Craig and J.B. Kemper, editors. *Regulated Streams, Advances in Ecology*. Plenum Publishing, New York.

Shirvell, C.S. and D.L. Morantz. (1983). Assessment of the Instream Flow Incremental Methodology for Atlantic salmon in Nova Scotia. Transactions Canadian Electrical Association, Engineering and Operation Division, Vol. 22, 83-H-108, Montreal.

Souchon, Y., F. Trocherie, E. Fragnoud and C. Lacombe (1989). Les modèles numériques des microhabitats des poissons: application et nouveaux développements. *Revue des Sciences de l'Eau 2:*807-830.

SENSITIVITY ANALYSIS OF PHABSIM IN A SMALL ATLANTIC SALMON STREAM[1]

Gilles Bourgeois
Groupe-conseil Génivar inc.
5355 Boulevard des Gradins
Québec, Québec, G2J 1C8
gbourgeois@genivar.com

Daniel Caissie
Fisheries and Oceans Canada
P.O. Box 5030
Moncton, New Brunswick, E1C 9B6

and
Nassir El-Jabi
Université de Moncton
Moncton, New Brunswick, E1A 3E9

ABSTRACT

PHABSIM is a widely used habitat model in North America and an important tool for fish habitat management. The model was applied to establish a relation between Weighed Usable Area (WUA) and discharge for juvenile Atlantic salmon (Salmo salar) in Catamaran Brook, New Brunswick.

The objectives of our study were to assess the model's sensitivity to the input of physical parameters (water depth, velocity, and substrate), as well as to bias introduced by the simulation model and the variability in the location of transects. The uniqueness of our study design lies in an approach by habitat-type, and by the use of certain tests such as the bootstrap technique, in order to better quantify the sensitivity of PHABSIM for Atlantic salmon.

Results of the sensitivity analysis showed that the model was not sensitive to random errors introduced in water depth and velocity, or for substrate measurement done by different observers. However, the number of transects and their location were responsible for most of the variability in predicted habitat with variability increasing as discharge increased.

In general, PHABSIM is a robust model. This can be attributed to the considerable number of points used in the model to establish a relation between WUA and streamflow. Just like for any other model, it is still necessary to be aware of PHABSIM's limits and to have sufficient knowledge of the area where it is being applied.

Key Words: habitat, modeling, Atlantic salmon, PHABSIM, sensitivity analysis.

[1] Contribution No. 29 of the Catamaran Brook Habitat Research Project.

INTRODUCTION

With an increasing number of conflicts between water and aquatic resources users, it becomes more and more important to establish instream flows to protect the level of life of these aquatic ecosystems. To this end, the Instream Flow Group of the U.S. Fish and Wildlife Service developed the Instream Flow Incremental Methodology (IFIM: Bovee, 1982). This method is among the most sophisticated for simulation of fish habitat and has widespread applications in the field of environmental impact studies (Beecher, 1990). The major component of IFIM is the PHABSIM model (Milhous *et al.*, 1989), which is used by engineers, hydrologist and biologist to establish a relation between discharge and the physical habitat at different life stages of various species of fish.

The PHABSIM model is most often used to make recommendations regarding instream flow requirements (Reiser *et al.*, 1989; Armour and Taylor, 1991), however, it has also been used in modeling physical habitat to link potential available habitat with standing stocks of fish (Fausch *et al.*, 1988). Despite the widespread use of PHABSIM in instream flow issues and habitat modeling, a considerable number of authors have criticized or questioned certain aspects of this model (Mathur *et al.*, 1985; Morhardt, 1986; Scott and Shirvell, 1987; Gore and Nestler, 1988).

Fausch *et al.* (1988) indicated that most habitat models used thus far, including PHABSIM, did not allow for parameter sensitivity or confidence intervals. For instance, only a few studies have reported results on sensitivity analysis pertaining to PHABSIM. One such studies tested the choice of different aggregation techniques of input parameters and the biological data transformation in the development of preference curves (Morhardt, 1986). In fact, the use of different aggregation techniques can have a profound influence the WUA-discharge while the type of biological data transformation will effect the shape of the suitability index (SI) curve. Morhardt (1986) also tested the model's sensitivity to the location of transects and to the number of data points along them. Given the considerable number of data points for each transect, PHABSIM was not found to be to sensitive to the variability introduced by random errors of depths and velocities. This particular test consisted of a Monte Carlo simulation technique using ±30% errors in velocities and ±10% errors in depth, which indicated that the model was quite robust to these parameters. However, the model seemed more sensitive to the location of transects although not quantified by Morhardt (1986).

Bartz (1990) verified PHABSIM's sensitivity to several possible routines for the hydraulic simulation (e.g. IFG4, WSP or MANSQ), the choice of aggregation techniques and the choice of habitat preference curves. The tests were performed on different size streams; large (30 m^3/s), average (3 m^3/s) and small (0.3 m^3/s). As Morhardt (1986), he noted that the model was sensitive to the choice of aggregation techniques and suggested the use of site specific habitat preference curves.

In a study by Gan and McMahon (1990) very different quantitative estimates of WUA were obtained by the permutation of the various options in the hydraulic and habitat simulation modules of PHABSIM. If results from hydraulic simulations can be verified by field measurement, techniques for validating habitat results are few, hence the acceptance of these results must rest on the realism of the biological assumptions that are made.

Another important aspect of habitat modeling is the impact of mesohabitats and the different sequences of these mesohabitats which can influence fish productivity (Kershner, 1992). In the present study we will refer to these mesohabitats as habitat types which consist of a short segment of stream with a relatively homogenous habitat such as riffles, runs and flats.

The objective of the present study was; a) to calculate the differences between the actual habitat using field data for water depth, velocity, and substrate (d, v, s) compared to a calibrated simulation model; b) to test the sensitivity of PHABSIM to its input parameters (d, v, s); c) to determined the variability of the WUA curve within each habitat types using a resampling technique; and d) to determined the effects of pooling together different types of habitats.

Although many studies dealt with the sensitivity of PHABSIM (Bartz, 1990; Morhardt, 1986; Gan and McMahon, 1990), few have used the approach and comparison by habitat types as well as pooling different habitats together in the analysis. Comparisons were made using certain tests which better quantifies the sensitivity of PHABSIM for Atlantic salmon (*Salmo salar*).

METHODOLOGY

Study area

This study was conducted within the Catamaran Brook Habitat Research Project. Initiated in 1990, the Catamaran Brook project is a long-term (15 yrs) multi-disciplinary research study of a small stream catchment of 50 km^2 (Figure 1) in the Miramichi River system which is well known for its Atlantic salmon population. With a primary focus on the aquatic biota (particularly salmonid fishes) and their habitats, the broad objective of the study is to quantify the effects, if any, of timber harvest. The Atlantic salmon is the most common fish species in Catamaran Brook, with anadromous adults entering the brook to spawn in late October and early November. Brook trout are the most common fish species in the headwaters streams and in Catamaran Lake (Cunjak *et al.*, 1990).

To monitor discharge two hydrometric gauge are in operation at Catamaran Brook, one at mid-basin (Middle Reach) with a drainage are of approximately 25 km^2 and the other gauge is at the mouth (Lower Reach, 50 km^2). The mean annual flow (MAF), at the Middle Reach was estimated at 0.686 m^3 /s or 754 mm of runoff, while the MAF of the whole basin is approximately 1.2 m^3/s. Data on high and low flows show that the 2-year flood is in the order of 6 m^3/s, whereas the 2-year low flow is at 0.06 m^3/s in the Middle Reach.

In the present study, our sites were located in the Middle Reach, and consisted of 30 transects (10 in flat, 10 in run and 10 in riffle habitat). Of these 30 transects, 6 sets of 5 adjacent transects were located in each habitat types for comparative purposes. The six sites consisted of 2 flats, 2 riffles and 2 runs with each 5 adjacent transects. It should be noted that the length of each habitat types was different as the transects were selected to include the full habitat units (e.g. from the start of the riffle to the end). The total length of stream when habitat types were pooled was 90.8 m. Figure 1 shows the location of the study sites, as well as a plan view of the Middle Reach of Catamaran Brook.

PHABSIM Simulation

Three different field surveys were conducted during a high (between 2.6 m^3/s to 3.6 m^3/s), a medium (0.4 m^3/s) and a low flow period (0.17 m^3/s) for which depths and velocities were measured. The characteristics of these parameters as measured during the surveys are shown in Table 1 (Bourgeois, 1992). Substrate information was collected during a separate survey using three different observers and two identification techniques; the pebble-count technique (Kondolf and Li, 1992) and a visual estimation (Bovee, 1982).

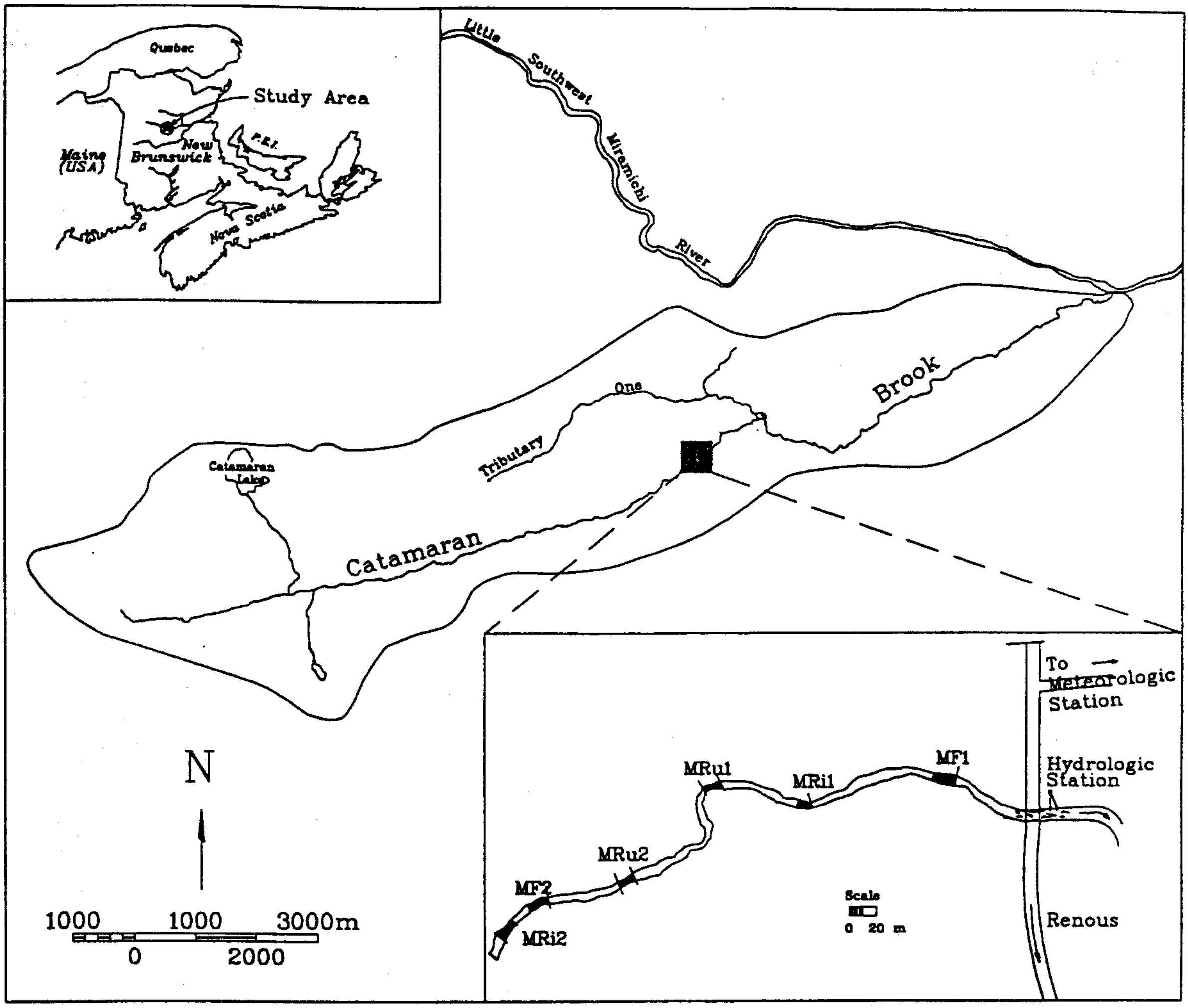


Figure 1: Map of Catamaran Brook basin showing the study reach and sampling sites for the application of PHABSIM.

To establish the relation between habitat and flow for each of the six habitat types, a habitat simulation using PHABSIM was carried out for 13 different flows ranging from 0.1 to 3.5 m^3/s, including three calibration flows. This provided the weighted usable area (WUA) vs. discharge curve for each site.

The habitat-discharge relation was calculated for two life stage of Atlantic salmon young-of-the-year (YOY) and parr based on HSI curves developed by Morantz et al. (1987). A verification study was conducted to test the applicability of these curves to Catamaran Brook (Bourgeois *et al.*, 1996).

Experimental Design

Since direct measurements of depth, velocity and substrate were available for the three calibration flows, it was possible to calculate the WUA directly without the hydraulic simulation of PHABSIM. These calculations were carried out to determined the bias introduced by PHABSIM during the simulation at the calibration flows.

Table 1: Characteristics of water depth and velocity as measured in Catamaran Brook.

		Qhigh (3.0 m³/s)		Qavg (0.5 m³/s)		Qlow (0.15 m³/s)	
Habitat-type	Parameter	max	avg	max	avg	max	avg
Flats	depth (m)	1.07	0.52	0.83	0.35	0.75	0.31
	velocity (m/s)	1.19	0.42	0.88	0.18	0.54	0.07
Riffles	depth (m)	0.85	0.37	0.51	0.20	0.42	0.14
	velocity (m/s)	1.87	0.76	1.17	0.42	0.95	0.20
Runs	depth (m)	1.07	0.36	0.79	0.24	0.66	0.20
	velocity (m/s)	1.58	0.70	1.09	0.35	0.63	0.16

To determine the WUA directly, a computer program (CALCWUA) was written in QBASIC (Microsoft, 1988) which uses the measured values (v, d, s) in the field, and the habitat preference curve was used to calculate the WUA-discharge relation for the calibration flows, knowing the area for each cell.

The importance of a sensitivity analyses in habitat modeling has been discussed by previous researchers (e.g. Reiser *et al.*, 1989, Morhardt, 1986). The first analysis dealt with the bias introduced by the hydraulic simulation when applying the PHABSIM model. The second analysis consisted of studying the sensitivity of the WUA-discharge curve to random errors introduced in velocity and water depth measurements. Three different sites were selected for this analysis to represent the three different types of habitats mentioned above and these included; Middle Flat 2 (MF2), Middle Riffle 2 (MRI2) and Middle Run 2 (MRU2). A Monte Carlo simulation technique was used to generate random errors of up to ± 10% for water depths and ± 30% for velocity measurements. According to Morhardt (1986), these values represent typical extreme case of measurement errors for such physical parameters.

To effectively study the sensitivity of PHABSIM to errors introduced in water depths and velocities, different scenarios were considered. The first consisted of introducing random errors to depths and velocities independently, while the second approach consisted at introducing errors to both parameters. The above mentioned CALCWUA program was capable of simulating random errors on depth and velocities when calculating the WUA-discharge curve and 100 such simulations were carried out.

The sensitivity of PHABSIM to substrate was also tested in the present study. Since the substrate data used by PHABSIM was measured visually and codified in discrete values according to Bovee (1982), the habitat simulation was run using measurements from three different observers, and then compared. In addition, the pebble count technique, as described by Kondolf and Li (1992), was used to quantify substrate by the same three different observers. This analysis will permit a comparison of substrate identification technique and the possible variability introduced by different observers.

The visual substrate measurement technique consists of identifying the predominant substrate of a cell by a code which corresponds to a habitat preference (from 0 to 1) for studied species at different life stage (Bovee, 1982). This code may differ according to the judgment and experience of the observers. The pebble count technique consisted of randomly selecting a number of rocks within each habitat types and measuring their size (Kondolf and Li, 1992). The habitat preference for substrate was calculated by knowing the size of rocks randomly selected in each cell, and compared to preference curves from Morantz *et al.* (1987).

The last sensitivity test performed on PHABSIM was to determine the effect of transects within each habitat types (e.g. in a riffle) on the WUA-discharge results. This analysis provides information on the variability of the WUA curve given the variability between transects. The calculations were carried out using a bootstrap technique approach (Efron and Gong, 1983) which consisted of resampling transects within each habitat types. The bootstrap technique consist of randomly selecting WUA values (with replacement so that the same transects could be selected more than once) for the 5 studied transects of each habitat types. The resampling technique was carried out for the range of flows studied with PHABSIM. By calculating the total habitat (WUA) for each sites for 500 simulations, variability was analyzed in the form of a coefficient of variation (%).

RESULTS

WUA - discharge relation

The WUA-discharge curve was similar between salmon parr and YOY when all habitat types (riffles, flats, and runs) were pooled together (Figure 2). The maximum WUA for YOY was found at a discharge of approximately 0.5 m^3/s with a value of 1760 $m^2/1000m$, while parr have maximum WUA at a somewhat higher flow of approximately 0.75 m^3/s at 1947 $m^2/1000m$ (All, Figure 2). The flow with maximum WUA will be referred to as the optimal habitat flow. For low flows (<0.25 m^3/s), and high flows (>1.0 m^3/s) the available habitat declines very rapidly with more WUA available for YOY at lower flows, in contrast to more WUA for parr at higher flows. The difference in WUA between YOY and parr is small at low flows, however, the maximum WUA difference between parr and YOY was found at flows of approximately 200% of MAF (i.e. 1.5 m^3/s) with a difference of 30% less available WUA for YOY.

It was observed from the analysis by habitat types (Figure 2) that riffles have more available habitat for YOY at optimum habitat flows (0.5 m^3/s, Figure 2a), while runs showed the least WUA. For YOY, the flats showed similar results to all habitats pooled together. Although riffles produced maximum WUA for YOY, they also have the least WUA during low flows (Figure 2a), therefore riffle becoming the limiting habitat during low flow. Runs on the other hand have more WUA for YOY at higher flows than other types of habitat, although the habitat is quite limited at those flows. During a low flow situation, flats showed the highest values of WUA (Figure 2a).

In the case of parr, the results are different as most WUA-discharge curves show similar shape regardless of habitat-types (Figure 2b). In fact, the preferred habitat for parr was the flats with 40% more WUA at optimal flow compared with the riffles, the least preferred habitat. Moreover, at optimum habitat flow, WUA for parr in riffles is only 1420 $m^2/1000$ m (Figure 2b) compared with 1987 $m^2/1000$ m for YOY salmon (Figure 2a). As was the case of flats for YOY, the runs showed WUA for salmon parr similar to all habitat types pooled together (Figure 2b). The results suggest that run habitat provides more average flow and depth conditions compared to other habitat types. This diversity of flow and depth conditions was also observed in a greater diversity of fish species (R. Cunjak, DFO, pers. comm.).

When WUA-discharge curves were plotted for each sites, more variability between same habitat type was observed and the riffles showed the most homogeneity between sites. In general, flats and runs have more WUA for parr than for YOY while the riffles have preferred habitat for YOY and showed less variability between sites.

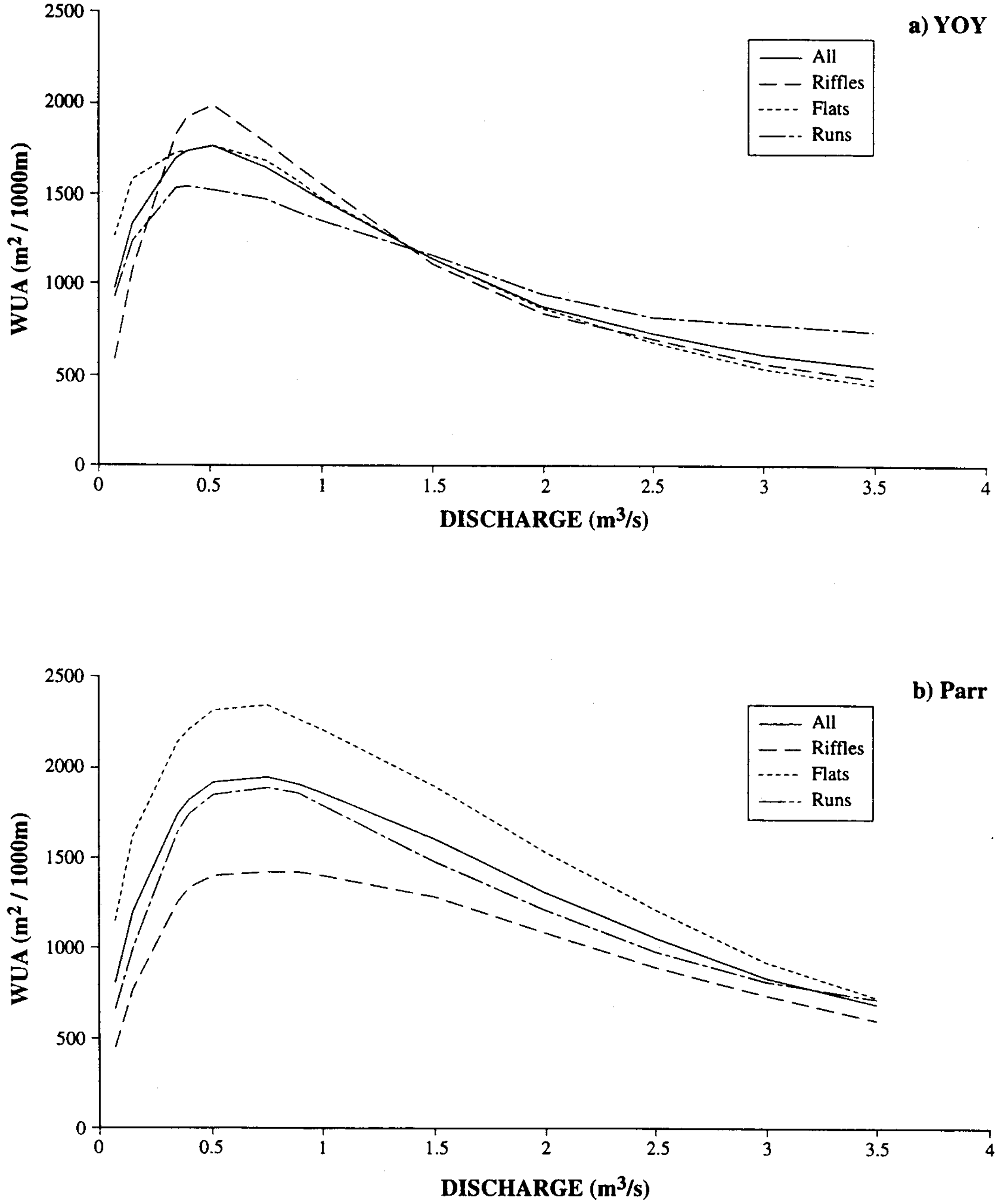


Figure 2. Results of PHABSIM simulations for Atlantic salmon YOY and parr in the Middle Reach of Catamaran Brook.

Sensitivity of PHABSIM to bias in modeling

WUA values were obtained for all calibration flows using direct on-site measurements of water depth, velocity and substrate and subsequently compared with WUA obtained by PHABSIM for the same flows. The WUA values using on-site measurements were calculated using CALCWUA and the differences in percentage (or bias) are shown in Table 2. A positive bias indicates more WUA predicted by PHABSIM than by on-site measurements. More than half of calibrated flows (24 of 36) showed a positive bias (Table 2). The overall absolute bias was calculated at 7.5% with a minimum value of 0.0% and a maximum difference of -34.3%. It is worth noting that the most important bias was observed for medium to higher flows, and only positive bias was observed for medium flows (Table 2). For low flows the overall bias was calculated at only 4.8%.

Table 2: Calculated difference (%) in predicted salmon YOY and parr habitat between PHABSIM and actual on-site measurements of depths, velocities and substrate using CALCWUA for calibration flows; low (0.15-0.17 m^3/s), medium (0.44-0.87 m^3/s) and high (2.6-3.6 m^3/s).

		Parr			YOY	
Site	Low	Medium	High	Low	Medium	High
MF1	9.4	5.1	3.3	9.9	1.5	8.1
MRI1	-13.3	11.4	-18.6	-7.4	15.1	-34.3
MRU1	1.5	20.1	6.0	-4.3	18.8	10.4
MF2	0.4	2.2	-2.2	-2.6	1.6	-10.1
MRI2	-6.4	12.7	0.0	0.2	8.3	-1.4
MRU2	-0.5	9.3	0.5	-1.4	8.2	1.7

Sensitivity of PHABSIM to errors in water depth and velocity measurements

A second sensitivity analysis was carried out to determined the variability introduced in the WUA-discharge curve as a result of possible errors introduced in water depth and velocity measurements. Typically, errors in such parameters can be in the range of ± 10% for depth, and ±30 % for velocity. Table 3 shows, for the three sites MF2, MRI2 and MRU2, the calculated variability in percentage given errors introduced for three different cases: a ± 10% error in water depth only, a ± 30% error in velocity only, and a combination of ±10% error in water depth with ± 30% error in velocity.

The results indicated that the overall variability in WUA as a result of errors introduced in velocities and depths were low (Table 3). The calculations were carried out using a Monte Carlo simulation technique with 100 simulation. The variability was calculated in percentage using quantiles of 16 and 84 % of WUA to estimate the coefficient of variation.

Most of the variability were found to be lower than 8%, and generally decreased with an increase in stream flow (Table 3). The overall variability with the different scenario in errors introduced in both water depth and velocity showed a low variability of approximately 3.9 %.

In general, the variability due to errors in water depth is less than for errors in velocities (Table 3). When considering low flow, most of the variability in WUA was found to be a result of velocities rather than water depth range from 4.8 to 7.5 % (Table 3; velocity ±30 % and low flow). Thus, the PHABSIM model is not very sensitive to water depth and velocities, probably due to the number of sampling points (over 20 points for each transect).

Table 3: YOY habitat sensitivity analysis of PHABSIM to error introduced in velocities and depth measurements using a Monte Carlo technique (100 simulations). Variability is expressed using a coefficient of variation in percentage.

SITE	Discharge Low	Medium	High
		Velocities (±30%)	
MRI2	5.6	4.6	1.5
MF2	4.8	1.2	0.9
MRU2	7.5	3.0	1.1
		Depth (±10%)	
MRI2	4.1	1.0	2.0
MF2	2.3	1.7	1.7
MRU2	3.1	1.6	0.9
		Velocities (±30%) and depth (±10%)	
MRI2	6.5	5.1	2.3
MF2	5.6	2.2	2.0
MRU2	6.7	3.2	1.3

Sensitivity of PHABSIM to substrate measurements techniques

The sensitivity of PHABSIM to substrate was carried out by calculating WUA-discharge relations for two substrate identification techniques: visual measurements, and the pebble count technique. These two substrate identification techniques were applied by three independent observers for three different sites: MF2, MRI2 and MRU2 (Figure 3).

No significant differences were observed between techniques except for site MRU2 (YOY) where the results by the pebble count technique showed more consistent results than the visual estimation (Figure 3). Most of the variability of WUA was found for YOY rather than parr, which could be a function of habitat suitability curves. The observed difference lies within the YOY substrate habitat preference curve which differs significantly between gravel and cobble, while for parr both gravel and cobble have the same habitat preference at unity (see Morantz *et al.*, 1987).

It can be concluded that PHABSIM is not very sensitive to substrate measurement techniques specially for lower flows in Catamaran Brook. Most of the variability in WUA results were found for YOY (MRU2 and MF2), while the best results for both YOY and parr were found in riffles.

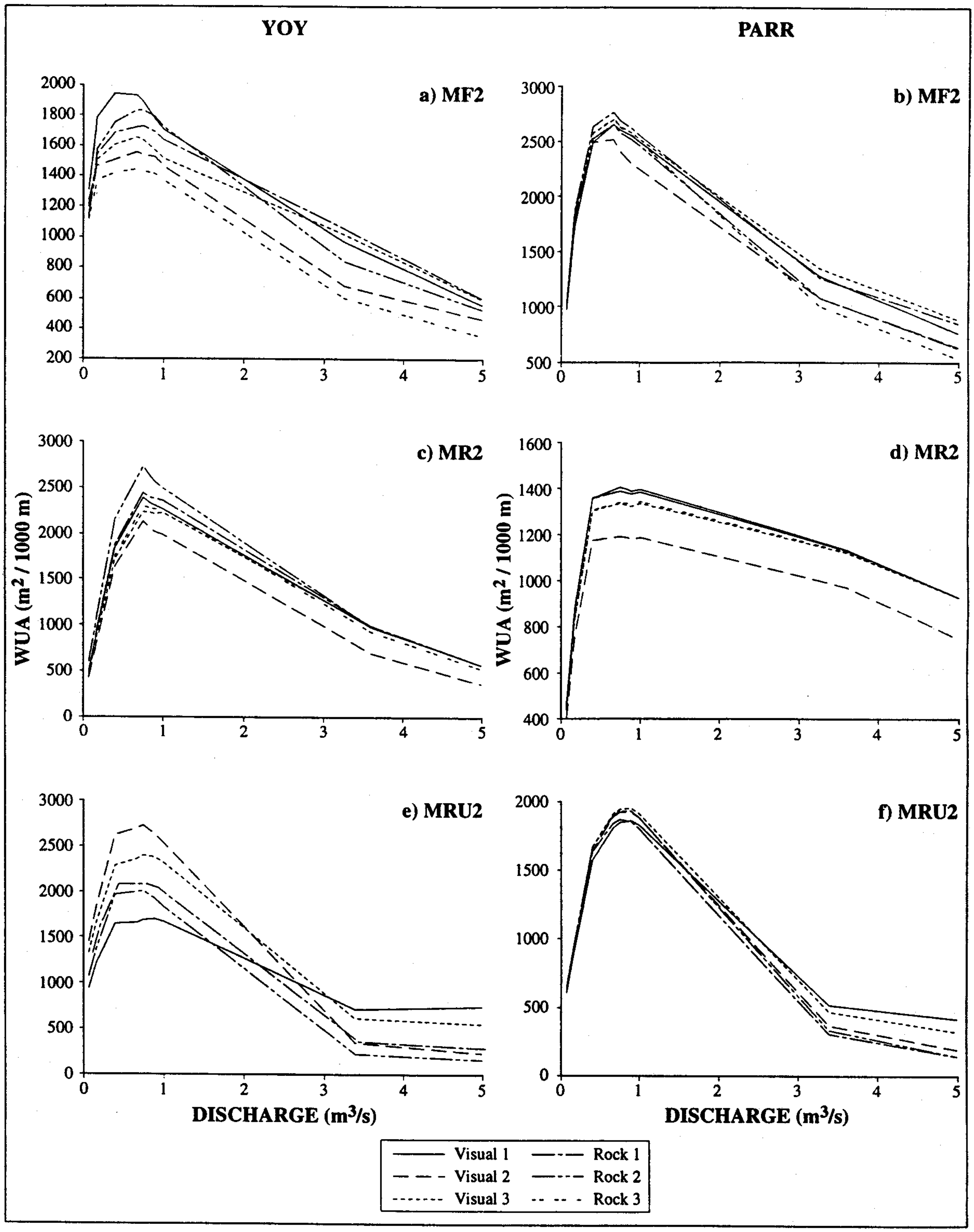


Figure 3. **Results of PHABSIM simulations for various substrate identification techniques in Catamaran Brook.**

Sensitivity of PHABSIM to transects selection

A resampling technique (bootstrap) was used to observe the sensitivity of PHABSIM to transects selection. In many application of PHABSIM, only one transect is chosen within a specific habitat type. In our study, five transects were used and a resampling technique provided information on the variability in habitat within each site. Using the WUA values calculated by PHABSIM, each transect was randomly resampled with replacement to reconstitute the total habitat of each site. This resampling was repeated for 500 simulations at each of the six sites, by habitat type (flats, runs, and riffles) and for all habitat types pooled together in the case of YOY salmon (Table 4). The variability was calculated using the 5th and 95th quantile around the median value (assuming a normal distribution) and was expressed as a coefficient of variation.

The overall variability of WUA per sites (resampling with five transects) was calculated at 15.6 %, ranging from 8.4 % (MRU2) to 23.6 % (MRI1, Table 4). It was interesting to note that the riffles showed more variability in WUA, with mean values of 23.6 % and 17.3 % for MRI1 and MRI2, respectively, and 14,5 % when pooled by habitat type. In contrast, runs showed the least variability in WUA with values of 9.5 % and 8.4 % for MRU1 and MRU2, and 7.0 % when pooled together (Table 4). In general, the variability in WUA as a function of discharge increased at higher flows. Also the variability decreased when sites were pooled. In fact, the mean variability between sites was in the range of 6.4 % to 8.2 % for flows less than 0.51 m^3/s (last column in Table 4). For flows greater than 1.5 m^3/s (or 200 % of mean annual flow), the variability in WUA was found to be greater than 8%. This shows the advantage of pooling sites together in order to lower the overall variability in WUA. It was noted that the variability in transects location was slightly higher than those observed by random error introduced in water depth and velocity.

Table 4: **Sensitivity analysis of PHABSIM to transects location using resampling technique (bootstrap) with 500 simulations, by site, by pooling habitat types, and for the whole studied reach. Variability is expressed using a coefficient of variation in percentage.**

Q	SITE						HABITAT-TYPE			
(m^3/s)	MF1	MF2	MRI1	MRI2	MRU1	MRU2	Flats	Riffles	Runs	REACH
0.07	9.8	9.7	16.8	9.8	7.4	13.0	7.4	9.3	8.0	8.2
0.15	9.3	4.7	17.3	12.5	4.0	10.4	5.7	10.1	5.7	6.6
0.35	17.1	7.6	16.6	12.0	8.3	5.4	9.1	9.7	4.8	6.4
0.40	18.9	8.0	17.7	11.8	9.0	4.1	10.4	9.9	4.9	6.9
0.51	21.9	8.0	18.2	11.5	9.8	---	11.4	10.5	4.9	7.1
0.75	26.5	9.1	19.8	11.6	8.7	1.6	13.3	12.1	4.8	7.1
0.89	26.7	9.5	21.5	10.6	8.7	1.2	13.4	11.6	5.5	7.0
1.00	25.0	8.9	22.5	11.6	8.4	0.8	13.4	11.9	5.6	7.5
1.50	26.1	10.6	23.6	16.3	11.4	5.6	12.9	15.3	6.6	8.4
2.00	24.5	10.5	31.1	22.5	13.9	9.1	14.2	18.4	8.9	8.9
2.50	26.9	13.8	35.3	27.4	14.4	13.8	16.1	21.6	9.5	9.3
3.00	30.5	19.6	36.0	32.8	12.5	17.1	20.2	23.5	10.6	10.1
3.50	28.7	24.7	31.0	34.7	12.8	19.0	21.7	24.7	10.6	11.6
Mean	22.4	11.1	23.6	17.3	9.5	8.4	13.0	14.5	7.0	8.1

DISCUSSION

This study enabled us to assess the physical habitat of Atlantic salmon in the Catamaran Brook using the PHABSIM model. Application of the model provided us with a relation between the Weighted Useable Area (WUA) and discharge. The sensitivity of the model parameters was investigated, with tests and analyses being done on three habitat types found in Catamaran Brook (i.e. riffles, runs and flats).

The results showed that as a general rule, WUA was at its maximum when the flow is near the mean annual flow (0.6 m^3/s) regardless of habitat type. This is similar to recommendations made by Tennant (1976), who suggested between 60 and 100 % of mean annual flow as optimal for fish habitat.

In addition, the discharge at maximum WUA (optimal habitat flow) was found at a higher flow for parr than for YOY, which is to be expected as parr needs greater water depth and are able to tolerate faster water velocities. It was also noted that runs and flats were more favorable habitats for parr, while riffles are preferable for YOY.

The results from PHABSIM were compared to actual habitat (WUA from on-site data) to estimate the bias introduced by the modeling. The overall absolute bias was calculated at 7.5% and more than 65 % were of positive bias, indicating that the PHABSIM model provides a higher habitat value than actual measurements. The maximum biases were also observed at medium flows.

The reason for the maximum bias occurring at medium flow lies within the calibration of velocities for which more weight is given to the lower and higher value leaving the middle velocity with a greater positive bias. Generally, when applying PHABSIM for instream flow requirements studies, the important portion of the WUA-discharge curve is at low flows which showed low biases between CALCWUA and PHABSIM (Table 2).

The sensitivity of PHABSIM to its input parameters demonstrated the model's robustness for measurements errors of 10% in water depth and 30% in velocities. These errors resulted in variations of approximately 4% (expressed as a coefficient of variation) in the WUA-discharge curves. This is comparable with results found by Morhardt (1986), which indicated approximately 8% variability for the mid portion of the WUA-discharge curve.

The analysis of the model's sensitivity to substrate identification demonstrated that when using two different techniques (visual measurements and pebble counts) with three different observers, the variability of the WUA curve was small. Differences in WUA was smaller for low discharge which is probably due to a greater homogeneity of the substrate in the central portion of the stream. Fines and vegetation were mainly found at the edges of the stream which contributed to some variability in WUA values for higher discharges. Of the different habitat type, the riffles provided the best results for both YOY and parr, again probably due to a better homogeneity in substrate which is often observed in riffles.

PHABSIM was also tested for the sensitivity to the location (or selection) of transects. Based on a resampling technique it was found that the variability increased with discharge. Also the variability in transect location was slightly higher than those observed by introduction of random errors to water depth and velocity.

The present study showed by resampling transects that the variability can be reduced when sites of similar habitat types were pooled together. Further reduction in the variability was also realized by pooling different habitat types at the reach level. In general, more variability was observed in riffles while the least variability could probably be achieved by including more transects with less cross-sectional data or by considering a two-dimensional modeling approach such as suggested by Leclerc *et al.* (1995).

In conclusion, the PHABSIM model is robust. This was noted by Morhardt (1986) who attributed this to the considerable number of points used by the model to establish a relation between WUA and streamflow. Despite all the criticisms it has drawn, the PHABSIM model remains a powerful tool. But like any other model, we must identify its limits, continue to test it and improve it and especially, to have a good knowledge of the system it is applied to. In such a context the PHABSIM model can be very useful in helping to resolve potential conflicts involving the management of water resources and fish habitat.

REFERENCES

Armour, C. and J. Taylor. (1991). Evaluation of the Instream Flow Incremental Methodology by U.S. Fish and Wildlife Service field users. *Fisheries*. 16(5):36-43.

Baltz, D., B. Vondracek, L. Brown and P. Moyle. (1987). Influence of temperature on microhabitat choice by fishes in a California stream. *Transactions of the American Fisheries Society*. 116:12-20.

Bartz, B. (1990). Sources of uncertainty and effect on interpretation of results in the development of instream flows for fisheries habitat. M.Sc. Thesis, Utah State University, Logan, Utah, 142 p.

Beecher, H. (1990). Standards for instream flows. *Rivers*. 1(2): 97-109.

Bourgeois, G.E., R. Cunjak, D Caissie, and N. El-Jabi. (1996). A spatial and temporal evaluation of PHABSIM in relation to measured density of Atlantic salmon in a small stream. *North American Journal of Fisheries Management*, 16:154-166.

Bourgeois, G.E. (1992). Modélisation de l'habitat physique du saumon de l'atlantique avec PHABSIM: Cas du ruisseau Catamaran. Master's thesis. Université de Moncton, Moncton, New Brunswick, 124 p.

Bovee, K.D. (1982). A guide to stream habitat analysis using the instream flow incremental methodology. U.S. Fish and Wildlife Service, Instream Flow Information Paper No. 12, FWS/OBS - 82/26.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. U.S. Fish and Wildlife Service, Instream Flow Information Paper No. 21, Biological Report 86(7).

Conder, A.L. and T.C. Annear. (1987). Test of WUA estimates derived from a PHABSIM model for instream flow studies on trout streams. *North American Journal of Fisheries Management*. 7:339-350.

Cunjak, R.A., D. Caissie, N. El-Jabi, P. Hardie, J.H. Conlon, T.L Pollock, D.J. Giberson and S. Komadima-Douthwright. (1993). The Catamaran Brook (New Brunswick) Habitat Research Project: Biological, physical and chemical conditions (1990-1992). *Canadian Technical Report of Fisheries and Aquatic Sciences*. No: 1914.

Cunjak, R.A., D. Caissie and N. El-Jabi. (1990). The Catamaran Brook Habitat Research Project: Description and general design of study. *Canadian Technical Report of Fisheries and Aquatic Sciences*. No:1751.

Efron, B., and G. Gong. (1983). A leisurely look at the Bootstrap, the Jacknife and Cross-Validation. *The American Statistician*. 37(1): 36-48.

Fausch, K., C. Hawkes, and M. Parsons. (1988). Models that predict standing crop of fish from habitat variables: 1950-1985. General Technical Report PNWT-GTR-213, USDA, Portland.

Gan, K. and T. McMahon. (1990). Variability of results from the use of PHABSIM in estimating habitat area. *Regulated Rivers: Research and Management*. 5: 233-239.

Gore, J. and J. Nestler. (1988). Instream flow studies in perspective. *Regulated Rivers: Research and Management*. 2: 93-101.

Kershner, J.L., W.M. Snider, D.M. Turner, and P.B. Moyle. (1992). Distribution and sequencing of mesohabitats: Are there differences at the reach scale? *Rivers*. 3(3): 179-190.

Kondolf, G.M. and S. Li. (1992). The pebble count technique for quantifying surface bed material size in instream flow studies. *Rivers*. 3(2): 80-87.

Leclerc, M., A. Boudreault, J. Bechara, and G. Corfa. (1995). Two-dimensional hydrodynamic modeling: A neglected tool in the Instream Flolw Incremental Methodology. *Transactions of the American Fisheries Society*. 124: 645-662.

Mathur, D., W.H. Bason, E.J. Purdy, and C.A. Silver. (1985). A critique of the instream flow incremental methodology. *Canadian Journal of Fisheries and Aquatic Sciences*. 42: 825-831.

Milhous, R., M. Updike, and D. Snyder. (1989). PHABSIM system reference manual: version II. US Fish and Wildlife Service. Instream flow Information paper #26, FWS/OBS-89/16.

Microsoft QuickBASIC. (1988). Programming in BASIC. Microsoft Corporation, Redmond, WA, 458 p.

Morantz, D.L., R.K. Sweeney, C.S. Shirvell, and D.A. Longard. (1987). Selection of microhabitat in summer by juvenile Atlantic salmon (Salmo Salar). *Canadian Journal of Fisheries and Aquatic Sciences*. 44: 120-129.

Morhardt, E. (1986). Instream flow methodologies. Research Project 2194-2. Prepared for Electrical Power Research Institute, by EA Engineering, Science, and Technology, Inc., Lafayette.

Reiser, D.W., T.A. Wesche, and C. Estes. (1989). Status of instream flow legislation and practices in North America. *Fisheries*. 14 (2): 22-29.

Scott, D., and C.S. Shirvell. (1987). A critique of IFIM and observations on flow determination in New Zealand. Regulated Streams, Craig and Kemper, editors, pp. 27-43.

Shirvell, C.S. (1989). Ability of PHABSIM to predict Chinook salmon spawning habitat. *Regulated Rivers: Research and Management*. 3: 277-289.

Tennant, D.L. 1976. Instream flow regimens for fish, wildlife, recreation, and related environmental resources. Pages 359-373 in: J.F. Orsborn and C.H. Allman, editors. Proceeding from Symposium and Spec. Conference on Instream Flow Needs, Vol. II, American Fisheries Society, Bethesda, Maryland.

EVALUATION OF ONE-DIMENSIONAL AND TWO-DIMENSIONAL HYDRAULIC MODELING IN A NATURAL RIVER AND IMPLICATIONS IN INSTREAM FLOW ASSESSMENT METHODS

Karl Tarbet
Research Assistant, Institute for Natural Systems Engineering
Department of Civil and Environmental Engineering
Utah State University, Logan, Utah 84322-4110
United States of America
tarbet@aaron.cee.usu.edu

Thomas B. Hardy
Director, Institute for Natural Systems Engineering
Department of Civil and Environmental Engineering
Utah State University, Logan, Utah 84322-4110
United States of America
hardy@aaron.cee.usu.edu

ABSTRACT

Spatially intensive channel morphology, water surface elevations, and velocity distributions at multiple discharges were collected over a 0.5 kilometer reach of the Logan River. A 2-dimensional hydraulic simulation model (RMA-2) was calibrated to a flow rate of 7.7 m^3/s and used to simulate a range of flows. Spatially dependant depth and velocity errors were examined at two observed discharges. Depth error magnitudes ranged between 0.0 and 0.4 meters and were highest in areas of complex channel geometries. This was attributed to differences between the finite element mesh and measured channel elevations. Velocity errors ranged in magnitude between 0.0 and 0.9 m/s with the greatest errors associated spatially with complex channel geometries. A variable number of regularly spaced cross sections were used to represent 1-dimensional river hydraulics typical of PHABSIM applications. Study results showed that regularly spaced cross sections, 24 meters, apart closely approximated the two-dimensional hydraulic simulations. Increasing the number of cross sections (up to 180) did not substantially improve model agreement. Reducing the number of cross sections however, resulted in increased differences. Adult brown trout habitat was compared for the one-dimensional cross section analyses and the 2-dimensional hydraulic simulations. Predicted relationships in Weighted Usable Area (WUA) were very similar using both approaches and suggest that differences between using 1- and 2-dimensional hydraulic simulations diminish rapidly with adequate characterization of the stream channel using cross sections. Our study results also indicate that proper characterization of the spatial domain in the finite element mesh is critical for proper hydraulic simulation results using 2-dimensional hydraulic models. These results are discussed in light of data requirement, hydraulic modeling constraints, and predictions of changes in the quantity and quality of fisheries habitat.

KEY-WORDS: Open Channel Flow/ Instream Flows/ One-dimensional Hydraulics/ two-dimensional Hydraulics/ Spatial Error Analysis/ Comparative Evaluation/ Brown Trout Habitat Modeling.

INTRODUCTION

The most used, accepted, and debated instream flow assessment tool is the Instream Flow Incremental Methodology (IFIM) (Reiser, 1985; Lamb, 1989; Gore and Nestler, 1988; Orth and Maughan, 1982; Mathur et al., 1985; Shirvell, 1986; Scott and Shirvell, 1987). Aside from this debate, an integral part of the IFIM is a set of hydraulic and habitat models referred to as the Physical Habitat Simulation System (PHABSIM) and is utilized to evaluate the quantity and quality of fisheries habitat as a function of discharge (Milhous et al., 1989; Bovee, 1995). A central feature of PHABSIM is the use of a variety of 1-dimensional hydraulic simulation routines that rely on cross section data of the river geometry, water surface elevations at different discharges, and observed velocities to calibrate these models. The calibrated hydraulic models are then utilized to simulate the hydraulic attributes of depth and velocity over a user specified range of discharges. However, the application of the one-dimensional hydraulic models in PHABSIM obtains a simplified picture of the actual hydraulics, which is not always considered sufficient (Ghanem, 1995). In particular, PHABSIM calculates velocities at a cross section by dividing the river into independent cells and then solves Mannings equation in terms of velocity for each cell for a specified discharge. The calibrated Mannings n value at each cell (or vertical) is determined from a single set of measured velocities across each cross section. When modeling a river with cross sections that may be anywhere from ten meters to many hundreds of meters apart, detailed velocity information throughout the spatial domain within the channel cannot be obtained. Although selecting a large number of cross sections over very small areas (i.e. a few meters) will likely improve velocity prediction capabilities, use of one-dimensional hydraulics is still limited in its ability to accurately predict flow about complex channel geometries which have significant two and three dimensional characteristics. Additionally, the cost and time constraints in operational instream flow studies typically preclude collection of cross sections at this level of measurement scale. Recently, because of the advances in the capabilities of computers, models for two-dimensional hydraulics are becoming more widely available and application of 2-dimensional hydraulics within the area of instream flow assessments are now occurring (Leclerc, 1995; companion articles in these proceedings). However, the benefits of these models need to be addressed in terms of the data requirements and estimation of anticipated hydraulic simulation errors within natural channels compared to applications using more traditional 1-dimensional hydraulics. In this paper, we explore hydraulic simulation errors using 2-dimensional hydraulics within a natural river channel and compare these results with more conventional 1-dimensional hydraulic simulations using cross section data. We also evaluate the study results in light of operational instream flow studies.

Material and Methods

Study Area

A 0.5 km reach within the Logan River in northeastern Utah, USA was chosen for the comparison studies. The reach starts 30 meters below First Dam, upstream of the Utah State University Water Research Lab (UWRL) and continues downstream for approximately 500 meters below the dam. The width of the Logan River throughout the reach ranged from approximately 10 to 20 m and has a gradient of approximately 1 percent. The majority of the substrate is made up of cobble and boulder with some small areas of gravel. The study reach has four classes of habitat: run (57%), riffle (14%), turbulent (16%), and pool (13%). The three dominant species of fish include mountain whitefish (*Prosopium williamsoni*), albino rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*).

Data Collection Methods

Field data was collected within the Logan River during the summer and fall of 1995. Channel geometry data was collected roughly in the manner outlined by Ghanem (1995), such that the river can be represented by a finite element mesh which interpolates geometry between collection points. The spatial domain within the Logan River was measured over approximately a 1x1 meter grid within four 25 meter long sections corresponding to key fisheries habitat areas. In addition, cross section profiles were placed roughly every 20 meters down the longitudinal axis of the river channel between each of the intensively sampled spatial grids. Additional sampling occurred throughout the river using an irregular systematic point sampling procedure where channel geometries varied substantially between cross sections. At each spatial domain data point, bed elevation, mean column velocity, and substrate characteristics were recorded. The water surface profile within the reach was also surveyed at the time the field measurements were collected. These measurements were made over flow rates corresponding to high (summer) and medium (fall) flows.

2-Dimensional Hydraulic Modeling

The Logan River was modeled with a single finite element mesh rather than smaller portions since the RMA-2 model used in the study is designed for far-field problems (Thomas and McAnally 1985). Additionally, the finite element mesh was extended upstream and downstream of the measured spatial domain in order to place the boundary conditions away from the area of study as suggested by Ghanem (1995). This extension of the mesh also helped in the stability of the solution, in addition to keeping the boundary conditions some distance from the study area. The process of calibration and selection of boundary conditions was simplified by computing water surface profiles using a one-dimensional water surface profile model HEC-RAS. The HEC-RAS computed water surface profiles were used as a guide for calibrating the two-dimensional water surfaces in RMA-2 for the range of flow rates represented by the collection data. The RMA-2 model was calibrated at a flow rate of 7.7 m^3/s by adjusting the Manning roughness values based on delimited substrate characteristics and by variation in the spatial magnitudes of the turbulence exchange coefficients. The calibrated model was then used to simulate hydraulic conditions at an observed flow rate of 4.2 m^3/s.

Habitat Modeling

Suitability index curves for adult brown trout currently in use by the Utah Division of Wildlife Resources for conducting PHABSIM analyses within the State of Utah were used to compute the combined suitability at each spatial coordinate. Only depth and velocity curves were utilized in the analyses. The combined suitability index values were used to compute the associated Weighted Usable Area (WUA) over a range of simulated discharges using the 2-dimensional hydraulic simulations.

For this study, different densities of 1-dimensional hydraulic cross sections from the finite element grid at longitudinal spacings which ranged between 2.4 to 24 meters. The 1-dimensional cross sections were derived from the finite element mesh by spacing 15 points across the mesh perpendicular to the direction of flow. The hydraulic properties at each cross section were derived from the solution of the 2-dimensional model output at a flow rate of 4.2 m^3/s . The corresponding hydraulic cell attributes (i.e. depth and velocity) were extended over the spatial domain by replacing all finite element grid point values corresponding to the area of the mesh one-half the distance between adjacent cross sections. This approach was used to approximate the application of PHABSIM habitat modeling techniques within the study reach. The subsequent 1-dimensional hydraulic properties for the reach for each set of cross section densities were used to compute the WUA for comparison with the 2-dimensional model output at this same flow rate.

RESULTS AND DISCUSSION

2-Dimensional Hydraulic Model Calibrations

The slope (0.01) of the Logan River, in this reach, was considered to be at the upper limits for application of the RMA-2 model. Several of the sections were found to have high instabilities in convergence of the solution at the high flow rates which we related to conditions which were approaching critical flow. The RMA-2 model used in this study is not designed for critical flow conditions. However, adjustments of the turbulence exchange coefficients at the calibration and simulated lower discharges was sufficient to obtain convergence in the solutions. It is our belief that application of a different hydraulic model that is capable of handling critical of super critical flow conditions such as the Federal Highways implementation of RMA-2 may be necessary under similar or more extreme field conditions. Overall however, calibration of the model and delineation of the boundary conditions based on output from the HEC-RAS model was considered adequate in this application.

Measured Depths and Velocities Compared to the 2-Dimensional Model Output

Given the space limitations, representative results will be provided from a section of the Logan River in order to illustrate study findings. Figure 1 provides an example of the depth error contours over this section of the study area at the calibration flow of 7.7 m^3/s and simulated low flow of 4.2 m^3/s. Figure 2 provides the associated velocity error contours at these same discharges. Depth and velocity errors were computed by comparison of the spatially dependant difference between measured field values at both the calibration and simulated flows. A total of 136 depth and velocity comparisons were made within this reach at a flow rate of 7.7 m^3/s while 150 points were used at a flow rate 4.2 m^3/s. Figure 3 provides the frequencies associated with the magnitudes of the depth and velocity errors at these two flows for this same reach of river. These results indicate, that for this study, the overall frequency in magnitudes of the errors in both depth and velocity varied marginally between the two flow rates. The spatial distribution of these errors however, varied substantially between the two simulations (compare Figures 1 and 2). The magnitude of the errors were also greater than previous work obtained by Leclerc (1995) in an application of 2-dimensional hydraulics using a different model and may be a function of the higher gradient of this river and/or associated with problems in the use of the RMA-2 model under near critical flow conditions.

A close examination of the spatial distribution of the depth and velocity errors between the two modeled flows shows that the greatest errors between observed and simulated velocities is associated with areas having the greatest depth errors. The depth errors are a direct result of compromising the spatial representation of complex spatial geometries within the channel in the construction of the finite element mesh. We found that during trial calibration runs, the size and spatial representation of mesh elements in highly heterogenous channel topographies significantly affected model stability and resulting model velocity predictions. In this application, we often sacrificed the spatial heterogeneity of the finite element grid and to reduce mesh size for processing and improved model stability in obtaining a solution. Stability issues were related to elimination of significant differences in mesh element size between adjacent spatial areas which decreased model stability. We therefore implicitly sacrificed depth accuracy in highly heterogenous regions of the channel. The net effect, was to inflate the frequency of higher magnitude velocity errors within these specific regions of the spatial domain as illustrated in Figures 1 and 2. Clearly, this type of compromise should be carefully considered in future applications.

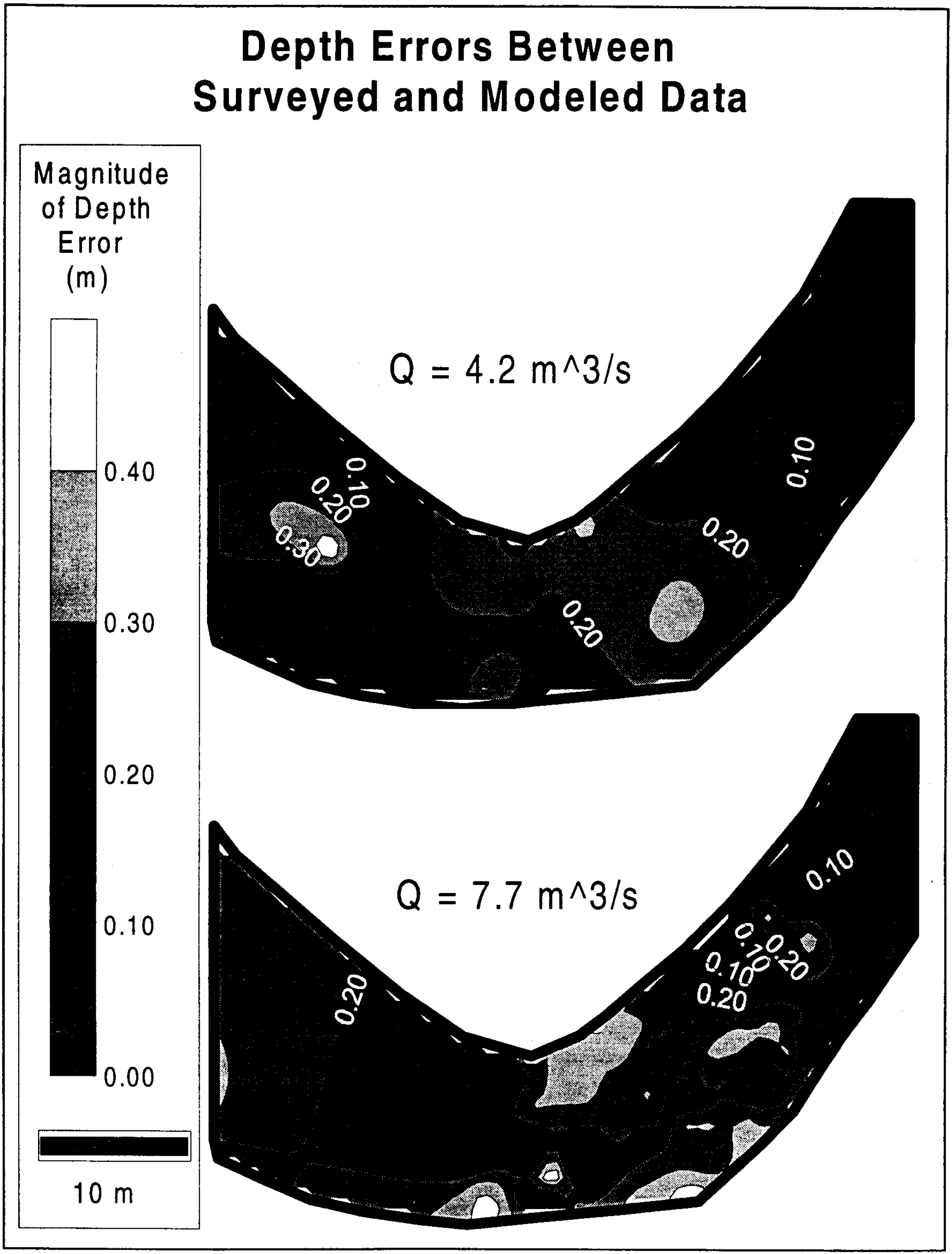


Figure 1. Depth Errors at Flowrates of 4.2 and 7.7 m^3/s

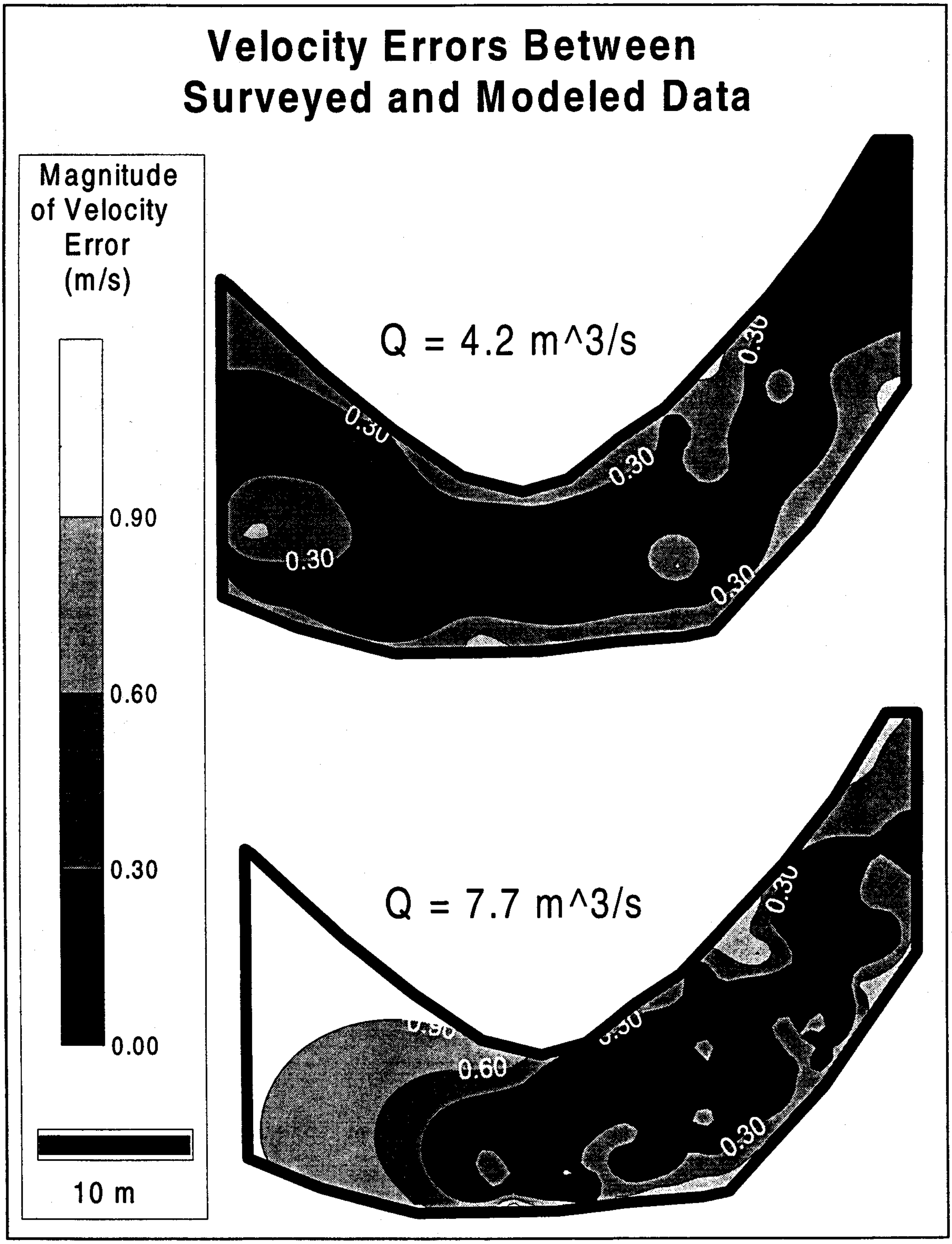


Figure 2. Velocity Errors at Flowrates of 4.2 and 7.7 m^3/s

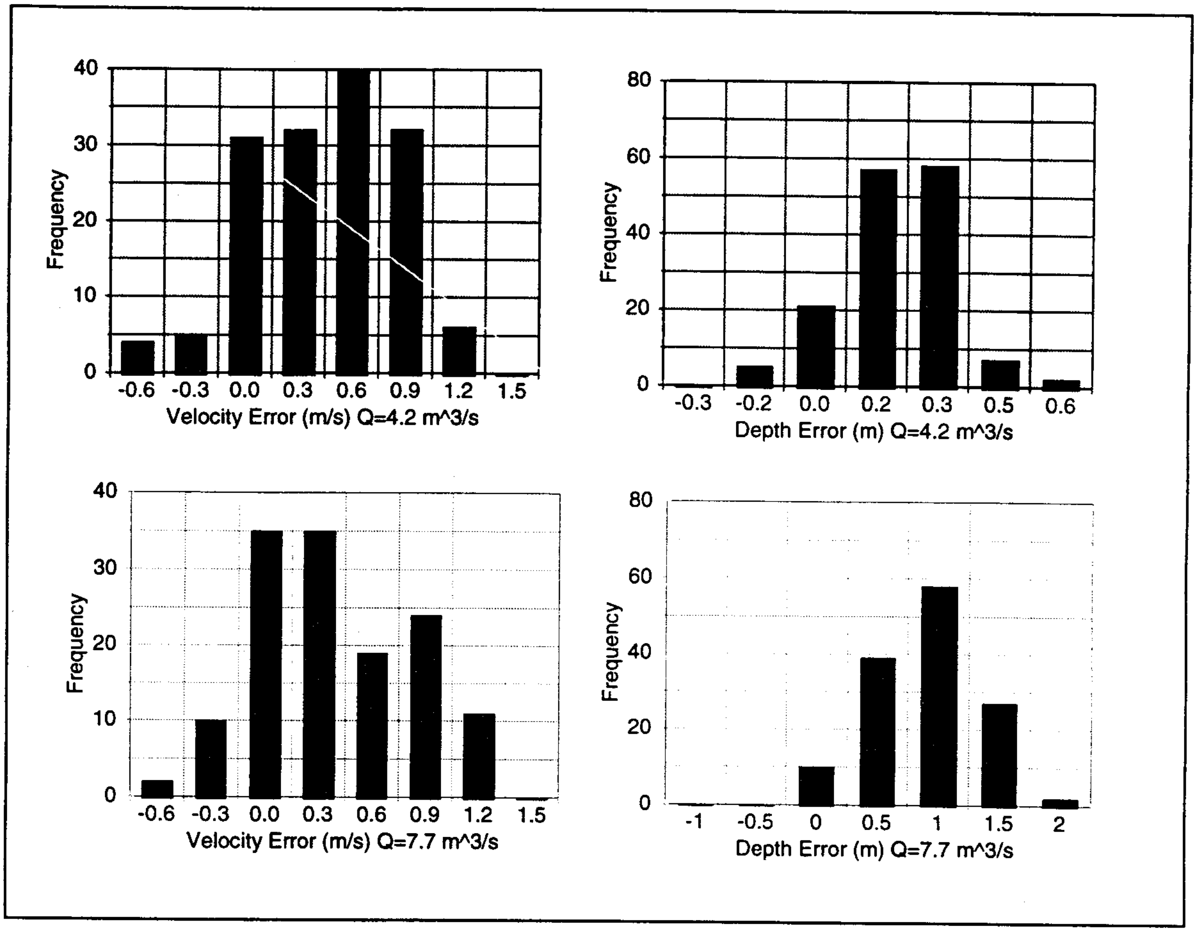


Figure 3: Frequency of Depth and Velocity Errors

Comparison of Habitat Modeling

The depth and velocity contours resulting from the solution of the 2-dimensional hydraulic model for the entire study reach at a flow rate of 4.2 m^3/s are shown in Figures 4 and 5. The corresponding spatial representation of depth and velocities using 19 cross sections at evenly spaced intervals of 24 meters are also shown in these Figures. Although a typical field application of PHABSIM would not normally employ such a high density of cross sections, these results are presented to make several key points. First, the overall pattern of depth and velocity heterogeneity within the entire study reach is generally preserved given this spacing of cross sections within the channel at this flow rate. Placement of fewer cross sections within the reach substantially reduces the agreement between the two approaches while increasing the number of cross sections only marginally improved the agreement with the 2-dimensional model output. For example, we found that increasing the number of cross sections to over 180 did not substantially improve the comparisons between the 2-dimensional hydraulic simulations compared to the representation using the spatially extended 1-dimensional values using 19 cross sections. This result is potentially misleading however, given the use of a single flow rate in these comparisons, since the flow dependant changes in the velocity magnitudes and their spatial distributions are marked (see Figures 2 and 3).

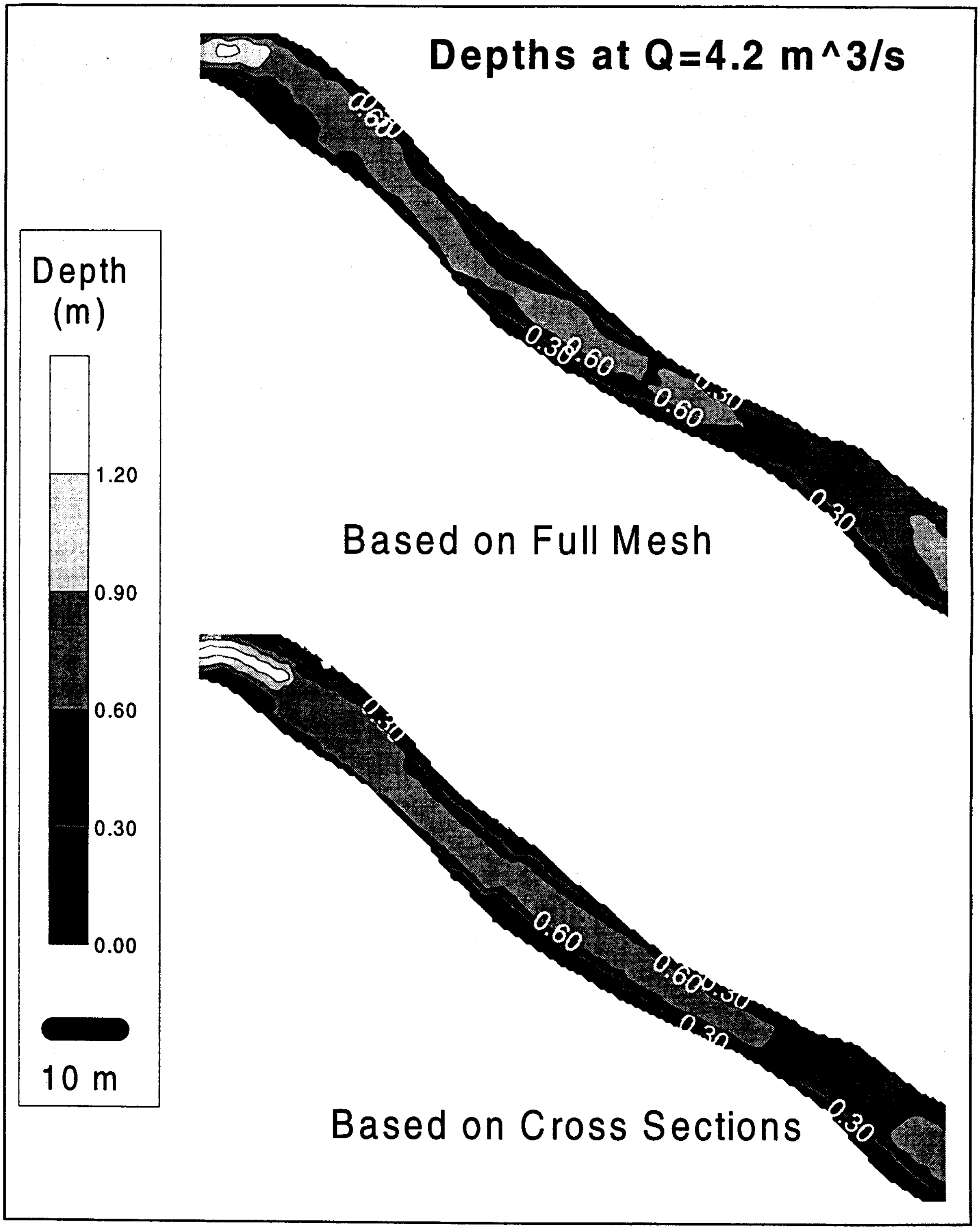


Figure 4: Water Depth Contours

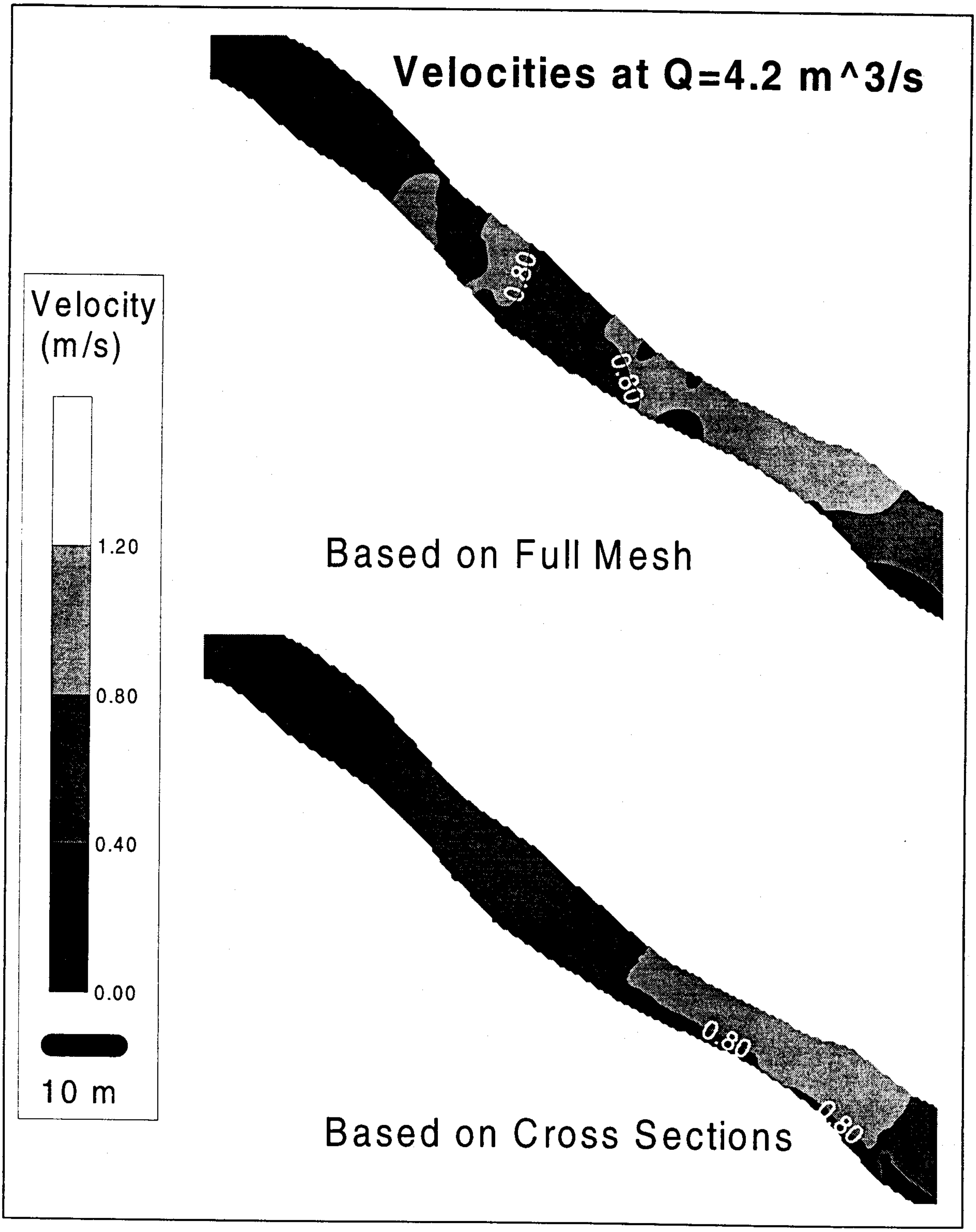


Figure 5: Water Depth Contours

We expected that more significant differences between these two approaches would exist if the flow dependant changes in velocity were considered in these comparisons between the 1-dimensional and 2-dimensional hydraulic simulations but was not borne out by our study results. The simulation of WUA over a range of discharges using the 2-dimensional hydraulic model output and the corresponding spatial extension of the associated 1-dimensional hydraulic properties for increasing density of cross sections is provided in Figure 6. It is not surprising, given the close agreement between the depths and velocities in Figures 4 and 5 that the resulting predictions of WUA from these comparisons are in close agreement (Figure 6) for all cross section densities. We again note, that with fewer numbers of cross sections (i.e. 6-10) these results would begin to depart significantly. These results would support the concept that the differences between applications of 1-dimensional and 2-dimensional hydraulic modeling of habitat based on suitability curves are relatively insensitive between the two approaches given adequate coverage of cross sections. An opposing view of differences between the use of 2-dimensional hydraulics in habitat modeling using both PHABSIM based suitability curves and a mechanistic based bioenergetics model can be found in Ludlow and Hardy (these proceedings). A more detailed treatment of these comparisons are in preparation (Hardy and Tarbet, manuscript) which compares 1-dimensional modeling of habitat using PHABSIM, based on stratified random sampling and systematic sampling approaches compared to the 2-dimensional model results.

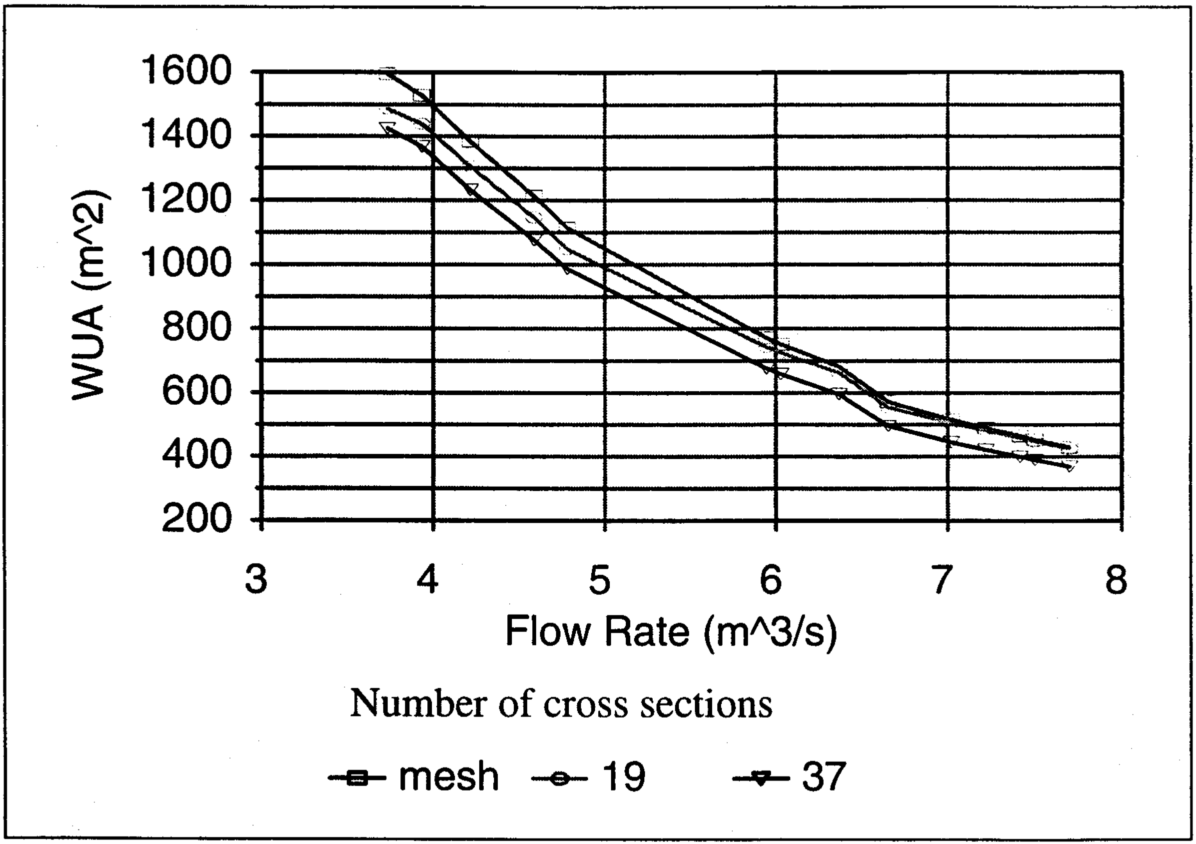


Figure 6: Weighted Usable Area For mesh and different numbers of cross sections

Conclusions

We found that calibration of the RMA-2 model was problematic in the Logan River given the high gradient and spatially complex changes in channel geometries. Spatial accuracy in the representation of the channel geometries to facilitate computational limitations as well as model instabilities directly affected both the magnitude and spatial distribution of the depth and velocity errors. Our study results highlight the importance of careful delineation of complex geometries in terms of meeting a desired data quality objective in the hydraulic simulations. Although stable solutions of the hydraulic model were obtained which generated complex flow patterns over the spatial domain, the range and magnitudes of the velocity errors may not be acceptable in light of applied instream flow assessments. Choice of an alternative hydraulic model (e.g. Ghanem, 1995; Leclerc, 1995) which can accommodate high channel gradients with critical or super critical flow conditions has the potential to reduce the observed depth and velocity errors. We felt that it was encouraging in some respects, that the magnitude of the depth and velocity errors remained fairly constant (i.e. consistent bias) between the two evaluated discharges. This further suggests that improving model calibrations using a better defined finite element mesh in complex areas of the channel in conjunction with the application of a more robust hydraulic model is likely to reduce the observed errors. Our study further suggests, that with an adequate characterization of the spatial domain using cross sections and more conventional 1-dimensional hydraulic simulations, differences between 1-dimensional and 2-dimensional hydraulic modeling of habitat converge based on use of PHABSIM type suitability curves. This particular study result does not address the efficacy of using this type of habitat modeling approach, and linkages of the 2-dimensional hydraulic modeling with alternative habitat modeling approaches suggest more pronounced differences can be found (e.g. see Ludlow and Hardy, these proceedings). We feel, that under typical PHABSIM applications using limited number of cross sections, that "improved" modeling based on 2-dimensional hydraulic simulations are possible. However ,based on our research in the Logan River, we suggest that this "improvement" will be contingent on accurate delineation of the spatial domain and that application of these modeling tools to sparsely interpolated spatial data will likely result in large depth and velocity errors.

REFERENCES

Bovee, K.D. (1995). A comprehensive overview of the instream flow incremental methodology. National Biological Service, Fort Collins, CO., 322pp.

Ghanem A. H. (1995) Two-Dimensional Finite Element Modeling of Flow in Aquatic Habitats. A thesis submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy in Water Resources Engineering.

Gore, J.A. and J.M. Nestler. (1988). Instream flow studies in perspective. Regulated Rivers Research and Management 2:93-102Hill, J. and G.D. Grossman. 1993. An Energetic Model of Microhabitat use for the Rainbow Trout and Rosyside Dace. *Ecology.* 74(3) pp. 685-698.

Lamb, B.L. (1989). Quantifying Instream Flows: Matching Policy and Technology. Instream Flow Protection in the West.

Leclerc M, Boudreault, A., Bechara J.A., Corfa, G. (1995). Two-Dimensional Hydrodynamic Modeling: A Neglected Tool in the Instream Flow Incremental Methodology. *Transactions of the American Fisheries Society.* 124(5) pp.645-662

Mathur, D. W.H. Bason, E.J. Purdy, and C.A. Silver. (1985). A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences* 42 pp. 825-831.

Milhous, R., Updike M, and Schneider, D. (1989). Physical Habitat Simulation Systems Manual - Version II, Instream Flow Information Paper No. 26. U.S. Department of the Interior Fish and Wildlife Service Research and Development, Washington DC 20240

Orth, D.J., and O.E. Maughan. 1982. Evaluation of the incremental methodology for recommending instream flows for fishes. *Trans. Amer. Fish. Soc.* 111 pp. 413-445.

Reiser, D.W. (1985) Status of Instream Flow Legislation and Practices in North America. *Fisheries*, 14(2)

Scott, D., and C.S. Shirvell. (1987). A critique of the Instream Flow Incremental Methodology and observation on flow determination in New Zealand. Pages 27-43 *in* J.F. Craig and J.B. Kemper, editors. Regulated streams: advances in ecology. Plenum Press, New York.

Shirvell, C.S. (1986). Pitfalls of physical habitat simulation in the Instream Flow Incremental Methodology. Canadian Fisheries and Aquatic Sciences Technical Report 1460. 68 pp.

Thomas A. W. And Mcanally W. H. Jr. (1985). User's Manual for the generalized computer program system Open-Channel flow and Sedimentation TABS-2 Main Text. Waterways Experiment Station, Corps of Engineers.

PROBLEMS WITH THE USE OF IFIM FOR SALMONIDS AND GUIDELINES FOR FUTURE UK STUDIES

David J. Bird

National Rivers Authority, South Western Region, Manley House, Kestrel Way, Exeter EX2 7LQ, UK

ABSTRACT

The objectives of the Instream Flow Incremental Methodology (IFIM) are appropriate for addressing a wide variety of water resource management problems. NRA South Western Region has used the technique in recent studies and is interested in expanding its use to additional ecological types of river. Concerns have been identified in this review regarding the modelling of habitat selection data by the Physical Habitat Simulation System (PHABSIM) to predict flow requirements. These difficulties are summarized below:

- Situations where physical habitat is not the primary regulating factor for populations
- Inappropriate choice of species/life stages
- Inadequate attention given to habitat scale effects
- Ignorance of the relative importance and interaction of habitat variables
- Sampling error in habitat variable measurements
- Unrealistic interpretation of habitat suitability criteria.
- Inappropriate transferability of habitat suitability criteria
- Inappropriate choice of PHABSIM reach
- Ignorance of temporal habitat variation

Many of these issues can be addressed in properly designed studies. Detailed, field based habitat selection studies should be restricted to critical periods and life stages (e.g late summer juveniles) and should be supported by carefully chosen criteria from the literature for other periods and life stages. These criteria should be transferable to a variety of streams that share the same ecological attributes.

KEY-WORDS: PHABSIM/IFIM/WUA/habitat suitability criteria/salmonids/coarse fish/life stage/habitat variable/transferability/habitat availability

INTRODUCTION

The Instream Flow Incremental Methodology (IFIM) is a technique developed in the US (Bovee, 1982) to resolve the conflict between offstream and instream uses of water. A set of habitat suitability criteria are collected for target species based on the microhabitat variables depth, velocity, substrate and, in some cases, cover. These 'curves' are fed into a suite of computer programmes called Physical Habitat Simulation (PHABSIM) which calculates the amount of physical habitat available at different streamflows.

There are 3 main categories of habitat suitability curve. Category 1 curves are subjectively based on expert judgement of habitat requirements from visual assessment of reaches (e.g. Chaveroche and Sabaton, 1989), other IFIM studies (see Thomas and Bovee, 1993), or knowledge of life history strategies (Armitage and Ladle, 1989). Category 2 curves are derived from field based studies of habitat use (e.g. Cunjack, 1986; Bird *et al.*, 1995). Category 3 curves express habitat preference by factoring out habitat availability from the category 2 function (e.g. De Graaf and Bain, 1986; Heggenes and Saltveit, 1990; Greenberg *et al.*, 1994).

IFIM has been used by the National Rivers Authority (NRA), to investigate the ecological implications of summer low flow conditions in recent years. Applications have initially focused on chalk streams and utilized for the first time in the UK, habitat suitability curves based on direct observation of fish (Bird *et al.*, 1995). In order to expand this work to other ecological types of river the NRA requires a critical review of the limitations of IFIM so that best practice procedures can be devised.

PROBLEMS

Situations where physical habitat is not the primary regulating factor for populations

Habitat selection studies will not be appropriate in reaches where other factors limit fish populations. Food availability (Gibson *et al.*, 1984; Orth, 1987; Ensign *et al.*, 1990), the proximity of spawning areas (Benson, 1953; Solomon and Templeton, 1976; Beard and Carline, 1991), water quality (Egglishaw and Shackley, 1985; Degeman *et al.*, 1986), competition (Hegge *et al.*, 1993) and predation (Mittelbach, 1986; Greenberg, 1992; Gotceitas and Godin, 1993) can all act as regulatory factors. Changes in the abundance of a fish population in a stream reach should also be clearly distinguished from temporary redistribution (Kraft, 1972; Mathur *et al.*, 1984). Habitat suitability curves should instead be developed in unexploited streams (Orth, 1987) at approximate carrying capacity (Bovee, 1982). Models such as Habscore (Wyatt *et al.*, 1995) could be used routinely in planning habitat selection studies to check that habitat is limiting and that carrying capacity conditions exist.

The term used to express quantities of physical habitat in IFIM studies is 'weighted usable area' (WUA). Several investigators have tested WUA predictions from PHABSIM modelling against stock abundance to verify that physical habitat is limiting (Conder and Annear, 1987; Irvine *et al.*, 1987; Gan and McMahon, 1990). The success of predictions has varied widely in different studies (Fausch *et al.*,1988; Shirvell, 1989; Jowett, 1992; Nehring and Anderson, 1993). IFIM is, however, a tool for predicting impacts of flow-changes on available habitat rather than on stock size (Gore and Nestler, 1988). Stock size is influenced by many additional factors (see above) and no holistic model is currently available to account for this in impact assessments. Consequently, WUA is unlikely to consistently relate to stock size in all situations. Indeed, Capra *et al.*, (1995) present a method where stock level information is used to validate model predictions of limiting habitat events. This approach is encouraged in future studies.

Inappropriate choice of species/life stages

Habitat evaluations ideally would account for the precise needs of all life history stages of the species in the stream community. Unfortunately, this degree of accuracy is often precluded in practice by knowledge and resource limitations (Garcia de Jalon, 1995). The sensitivity of the target species/life stage to habitat conditions (Sale *et al.,* 1982) should therefore have an important bearing on the necessary accuracy and precision (and therefore cost) of habitat suitability criteria. For most species, habitat requirements change throughout the life cycle and therefore habitat 'bottlenecks' (i.e. physical factors that act to limit populations) will affect life stages most dependent on the habitat in short supply (Elliott, 1994). Development of accurate, high precision, microhabitat based suitability criteria for fry/juveniles is often more appropriate than for adults for the following reasons:

* distances between resting and feeding areas are shorter (Helm *et al.,* 1982). This ensures that microhabitat measurements are relevant to requirements.
* individuals often exhibit sedentary behaviour. This increases sensitivity by precluding rapid dispersal in response to short-term flow changes (Nehring and Anderson, 1993), reduces fright bias, and facilitates the use of statistically pure random sampling designs (Bovee, 1986; Bird *et al.,* 1995).

By contrast, precise determination of the relative value of each habitat type can be difficult in the case of more mobile adult salmonids (Clapp *et al.*, 1990). It is suggested that lower precision criteria (e.g. Category 1 curves) are used for this life stage.

The development of high quality suitability criteria for spawning is encouraged given its importance as a limiting factor. Such criteria do exist (e.g Shirvell, 1989) but not to the same extent as fry/juvenile criteria.

Inadequate attention given to habitat scale effects

More attention needs to be given to spatial scale in habitat selection studies (Johnson, 1980; Shirvell, 1986; Bozek and Rahel, 1991; Simonson, 1993; Vondracek and Longanecker, 1993). Unfortunately, microhabitat data are often collected over a range of habitat types but then combined for suitability curves without regard to the habitat related selection differences (Heggenes, 1991; Vondracek and Longanecker, 1993). The restriction of microhabitat measurements to the focal points occupied by fish is a major weakness of IFIM. The location of energetically profitable positions are very important for salmonids (Fausch, 1984; Shirvell, 1989; Rincon and Lobon-Cervia, 1993). Microhabitat gradients should therefore be measured at fish stations. Bain *et al.* (1985) provide a useful methodology where mean substrate (index of coarseness) and the standard deviation (index of heterogeneity) were computed from dominant categories recorded along a sectioned rope. There is scope for extending this method to other variables, particularly water velocity.

Ignorance of the relative importance and interaction of habitat variables

Disagreement exists over the relative importance of the different variables and there is often no regard to interactions between variables despite the fact that it may invalidate the simplistic limiting factors approach (Bowlby and Roff, 1986; Gibson, 1993; Elliott, 1994). Cover, for example, is important but is often excluded from IFIM studies due to modelling difficulties created by its many functions and interactions with other variables (Hartman, 1965; Dolloff, 1986; Bugert *et al.*, 1991).

Multivariate statistical techniques should be used more widely to discriminate between selected variables and associated ones (Capen, 1981; Kessler and Thorpe, 1993). In addition, development of multivariate suitability functions expressing interactions (Orth and Maughan, 1982; Voos, 1981; Bullock *et al.*, 1991) should receive more attention.

Important biological interactions can be accounted for in modelling by creating habitat quality strata of one independent variable (often suitable or unsuitable) and using other independent variables as continuous value functions. Intervals of a continuous variable can also be grouped and treated as discrete variables. These conditional criteria are particularly useful in describing behavioural interactions concerning cover and substrate (Bovee, 1986; Williamson, 1994).

Sampling error in habitat variable measurements

Sampling error may be created by deliberately sampling where the quarry are expected to occur (e.g. Morantz *et al.*, 1987), or where sampling efficiencies vary significantly over the range of each habitat variable (Orth *et al.*, 1982). This disproportionate sampling effort is virtually impossible to detect (Bovee, 1986) but could be alleviated by developing mesohabitat specific curves. Inaccuracies are also introduced where fight bias or the capture method impedes the determination of focal points. This will usually be higher for more mobile fish (e.g. adult life stages) and for more active sampling techniques (e.g. electric fishing) (Williamson, 1994).

Sampling techniques should be carefully matched to mesohabitats. For example, although diving observations are generally suited to pool and deep glide mesohabitats, random point electric fishing will be more appropriate in many riffles, chutes and shallow glides (Heggenes *et al.*, 1990).

An adequate sample size is important to obtain adequate precision levels for criteria and to facilitate fitting a function to the observed frequency distribution. The required sample size can be estimated from the variance of samples (Williamson, 1994) and is typically, 150 to 200 observations (Bovee, 1986; Williamson, 1994). Estimates below 150 could reflect restricted habitat availability, suggesting that a more diverse study reach should be used (Williamson, 1994).

Unrealistic interpretation of habitat suitability criteria

The mathematical interpretation of suitability curves by IFIM (Bovee and Cochnauer, 1977; Bovee, 1986) is flawed as it implies that fish are found with certainty at locations exhibiting modal levels of occurrence of a habitat variable (Scott and Shirvell, 1987). The probabilistic interpretation of curves is extended by multiplying weighting factors for two or more habitat variables together to calculate WUA (Bovee and Cochnauer, 1977; Bovee, 1986; Scott and Shirvell, 1987). Using this convention, several combinations of depth, velocity, and substrate can give similar levels of WUA. In reality, however, the quantities of available habitat will probably differ significantly (Mathur *et al.*, 1984) due to variations in the importance of the habitat variables (see above).

The assumption that small areas of optimum habitat have the same productive capacity as large areas of suboptimal habitat (Scott and Shirvell, 1987) ignores habitat fragmentation effects (Baldes and Vincent, 1969; Heggenes *et al.*, 1994). Consequently, in streams with highly variable flow regimes there are few core habitat areas but pools are more stable and provide low flow refuges (Heggenes *et al.*, 1994). IFIM also interprets non-occupied areas or periods of low WUA to have little value for the fish population. Fisheries may, however, benefit indirectly through factors such as food production or provision of flushing flows for spawning (Shirvell, 1989). The method by Capra *et al.*, (1995) is again recommended as an interpretational tool (see above).

Inappropriate transferability of habitat suitability criteria

The importance of transferability of criteria has been shown empirically by comparing PHABSIM outputs using sensitivity analysis. Williamson (1994) re-examined data by Waddle (1992) to test depth and velocity criteria for adult brown trout from two separate studies (Raleigh *et al.*, 1986; Thomas and Bovee, 1993). He found significant differences in the habitat limiting flows and concluded that habitat descriptions were more sensitive to selection of suitability criteria than to hydraulics. Similar results were also obtained by Willis (pers. comm.) in a southern chalk stream. One set of generalised category 1 curves (Armitage and Ladle, 1989) and two sets of chalk stream specific category 2 curves (Johnson *et al.*, 1993; Bird *et al.*, 1995) were tested. Outputs using the chalk stream criteria showed close convergence and correlated with the habitat recognition of sites. By contrast, convergence and correlation were absent when the category 1 curves were tested.

Adequate consideration should be given to habitat availability in the development of suitability criteria (Heggenes *et al.*, 1991; Williamson, 1994). This is because fish will tolerate and adapt to suboptimal habitat when preferred habitat is in short supply (Heggenes and Saltveit, 1990; Taylor, 1991; Heggenes, 1994). An approach that could be taken to account for this is to develop mesohabitat specific habitat suitability criteria (see above). The perceived absence of 'universal' responses to habitat variables (Heggenes and Saltveit, 1990) has prompted the US Fish and Wildlife Service (USFWS) to strongly recommend that the transferability of criteria are validated before use. Nonparametric statistical methods for transferability testing have been developed by Thomas and Bovee (1993) and USFWS suggest that, at the very least, criteria are critically evaluated by fisheries biologists.

Although preference functions match habitat use against availability, they too can be highly stream specific (Heggenes, 1990; Williamson, 1994), and have some undesirable statistical and mathematical properties. When both the use and availability distributions simultaneously enter the limits of their distributions there is a risk of misrepresenting actual preference simply because of the small probability ratios involved (Morhardt and Hanson, 1988; Williamson, 1994). Attempts have been made to alleviate these problems. Williamson (1994) suggests applying nonparametric tolerence limits to recalculate utilization and preference curves using only frequency values falling within the tolerence levels established. Greenberg *et al.* (1994) produced D-value preference criteria using Jabob's (1974) formula but simplified the modelling of habitat quantities by using discrete preference, indifference and avoidance categories.

A major difficulty in the application of IFIM is the seasonal shift in habitat use (Rimmer *et al.*, 1984; Baltz *et al.*, 1991; Maki-Petays *et al.*, 1995). Diurnal variations in habitat use have also been observed in response to shifts in activity (Campbell and Neuner, 1985; Fraser *et al.*, 1993). Despite these considerations, curves based on daytime preferences for feeding during summer (Morantz *et al.*, 1986; Moore and Gregory, 1988) are often extrapolated to recommend stream flows throughout the year (Shirvell, 1986). This erroneously implies that behavioral adaptations to changing microhabitat availability and other factors (e.g. temperature) do not occur (Vondracek and Longanecker, 1993; Heggenes, 1994).

Curve development has been recommended under intermediate seasonal flow conditions when microhabitat diversity is perceived to be greatest (Bovee, 1986; Gore and Nestler, 1988). Unfortunately, the ommission of high and low flows for that period will not account for the behavioural switches that are known to occur (Heggenes, 1994). These range from downstream displacement, movement to pools or stranding at low flows to sheltering behind large substrates at high flows (Heggenes, 1994). This again suggests that projections to discharges substantially different from those for which the suitability curves were developed are invalid (Gore and Nestler, 1988).

Transferability of criteria may be inappropriate because the nature and intensity of competition and/or predation differ between source and target streams (Hindar *et al.*, 1988; Gibson *et al.*, 1993). In sympatric situations it is often difficult to determine whether species are using preferred habitat or are displaced to suboptimal habitat by a more dominant species (Cunjack and Green, 1983; Hearn, 1986; Bird *et al.*, 1995; Heggenes *et al.*, 1995). In non-saturated environments where biotic effects are at low levels (e.g. Heggenes, 1991) habitat criteria are likely to be narrowly defined and will consequently have low transferability (Thomas and Bovee, 1993). To broaden curves it is necessary to choose streams with a sufficiently high standing crop to force individuals into suboptimal locations (Thomas and Bovee, 1993).

Inappropriate choice of PHABSIM reach

A 'critical reach' habitat mapping approach has been used in UK studies with transects placed in close proximity where impacts are perceived to be greatest (e.g. Johnson *et al.*, 1993). There is a need to introduce more extensive habitat mapping approaches (Petts and Maddock, 1994; Williamson, 1994) to account for mesohabitat availability in rivers where affected reaches are less discrete. Bird and Maddock (1996) identified 'representative reaches' on the River Tavy based on the proportional availability of mesohabitat types in overall impacted sectors. This method provides the access advantages of the critical reach approach, is economical on transect requirements, but is more representative of river conditions.

Ignorance of temporal habitat variation

Examination of temporal water flow regimes has received insufficient attention (Heggenes, 1994), despite the fact that habitat bottlenecks do not operate continuously (Orth, 1987). Responses of populations to discharge changes may involve a significant time lag (Conder and Annear, 1987). Without extensive analysis of outputs the potential exists for intentional manipulation of results by users (Gan and McMahon, 1990). Identifying the frequency and duration of habitat bottlenecks for vulnerable life stages should be key tasks in instream flow assessment (Sale *et al.*, 1982; Stalnaker *et al.*, 1994; Capra *et al.*, 1995). Habitat bottlenecks defined in terms of WUA should also be compared with fluctuations in stock abundance (Stalnaker *et al.*, 1994; Capra *et al.*, 1995). For example, the availability of fry habitat in the first months of life was the biggest indicator of year class strength for trout in Colorado streams (Bovee, 1988).

CONCLUSIONS AND RECOMMENDATIONS

Despite the wide range of problems, particularly the difficulty in obtaining accurate suitability criteria, IFIM should still be regarded as a key tool for addressing water resource problems. It should be recognised that there are presently no superior alternatives to IFIM. In addition, IFIM presents biological information in a format suitable for incorporation into the water resources planning process. It is therefore recommended that IFIM is encouraged in UK studies, but with better designed habitat selection and impact assessment studies that address the main problems (see below).

Detailed field based habitat selection studies should be confined to critical periods and sensitive life stages and follow the guidelines below:

* A diverse sample of pristine sites at approximate carrying capacity should be selected by expert judgement and habitat evaluation methods such as Habscore.
* Microhabitat criteria should be stratified into mesohabitats (e.g. riffles, pools, glides) to account for scale effects in habitat selection.
* The full range of flow conditions within critical periods should be sampled to account for behavioral habitat switches.
* Microhabitat gradients at fish locations should be quantified, particularly water velocity shear zones.
* Improving the use of cover as a habitat variable is a challenging objective that should be considered in separate R&D investigations.
* The relative importance and interaction of habitat variables should be assessed by multivariate statistical analysis.
* Statistically pure sampling designs should be used with sample sizes large enough for criteria to meet precision requirements.

Literature based habitat selection criteria will be adequate for non-critical periods and less sensitive life stages **provided they are carefully screened for transferability**.

The selection of reaches for PHABSIM should involve habitat mapping and be appropriate to the investigation. Temporal habitat variation should be analysed to identify habitat bottlenecks and these should be validated by fish abundance data if possible. Sensitivity analysis should be used to compare the different types of habitat criteria and the effects of various flow setting scenarios.

REFERENCES

Armitage, P.D. and Ladle, M. (1989). Habitat preferences of target species for application inPHABSIM testing. *FBA Report* RL/T04053K1/1 to the Institute of Hydrology.

Bain, M.B., Finn, J.T., & Booke, H.E. (1985). Quantifying stream substrate for habitat analysis studies.*N. Am. J. Fish. Manage.* 5, 489-493.

Baker, J.A. and Ross, S.T. (1981). Spatial and temporal resource partitioning by southeastern cyprinids. *Copeia* 1981: 178-184.

Baldes R.J. and Vincent, R.E. (1969). Physical parameters of microhabitats occupied by brown trout in an experimental flume. *Trans. Am. Fish. Soc.* 98: 230-238.

Baltz, D.M., Vondracek, B., Brown, L.R. and Moyle, P. (1991). Seasonal changes in microhabitat selection by rainbow trout in a small stream. *Trans. Am. Fish. Soc.* 120: 166-176.

Beard, T.D Jr. and Carline, R.F. (1991). Influence of spawning and other stream habitat features on spatial variability of wild brown trout. *Trans Am. Fish. Soc.* 120: 711-722.

Benson, N.G. (1953). The importance of groundwater to trout populations in the Pigeon River, Michigan. *Trans. N. Am. Wild. Conf.* 18: 269-281.

Bird, D.J., Lightfoot, G.W. & Strevens, A.P. (1995). Microhabitat use by young salmon and trout in a southern U.K. chalk stream. *Proceedings IFM 25th Annual Study Course,* Lancaster, 99-114.

Bovee, K.D. (1982). A guide to stream habitat analysis using the Instream Flow Incremental Methodology. *Instr. Flow Inf. Pap.* 12, USDI Fish Widl. Serv., FWS/OBS-82126.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. *Instr. Flow Inf. Pap.* 21, USDI Fish Widl. Serv. Biol. Rep. 86(7).

Bovee, K.D. (1988). Use of the Instream Flow Incremental Methodology to evaluate influences of microhabitat variability on trout populations in four Colorado streams. *Proceedings Annual Conference Western Association Fish and Wildlife Agencies* 68: 227-257.

Bovee, K.D. and Cochnauer, T. (1977). Development and evaluation of weighted criteria, probability of use curves for instream flow assessments in fisheries. *Instr. Flow Inf. Pap.* 3, USDI Fish Wild. Serv., FWS/OBS-77163.

Bowlby, J.N. and Roff, J.C. (1986). Trout biomass and habitat relationships in southern Ontario streams. *Trans. Am. Fish. Soc.* 115: 503-514.

Bozek, M.A. and Rahel, F.J. (1991). Assessing habitat requirements of young Colorado River cutthroat trout by the use of macrohabitat and microhabitat analyses. *Trans. Am. Fish. Soc.* 120: 571-582.

Bugert, R.M., Bjornn, T.C. and Meehan, W.R. (1991). Summer habitat use by young salmonids and their responses to cover and predators in a small southeast Alaska stream. *Trans. Am. Fish. Soc.* 120: 474-485.

Bullock, A. Gustard, A. and Grainger, E. (1991). Instream flow requirements of of aquatic species. *Institute of Hydrology Report* No. 115.

Campbell, R.F. and Neuner, J.H. (1985). Seasonal and diurnal shifts in habitat utilized by resident rainbow trout in Western Washington cascade mountain streams. P. 39-48 in Olson, F.W., White, R.G. and Hamre, R.H. (eds). *Symposium on small hydropower and fisheries. Am. Fish. Soc.*, Bethesda, Maryland, USA.

Capen, D.E. (1981). (ed.) The use of multivariate statistics in studies of wildlife habitat. *US For.Serv.Gen.Tech. Rep.* 87: 249 p.

Capra, H., Breil, P. and Souchon, Y. (1995). A new tool to interpret magnitude and duration of fish habitat variations. *Regul. Riv.* 10: 281-289.

Chaveroche, P. and Sabaton, C. (1989). An analysis of brown trout (*Salmo trutta* fario L.) habitat:the role of qualitative data from expert advice in formulating probability-of-use curves. *Regul. Riv.* 3: 305-349.

Clapp, D.F., Clark Jr, R.D. and Diana, J.S. (1990). Range, activity, and habitat of large, free-ranging brown trout in a Michigan stream. *Trans. Am. Fish. Soc.* 119: 1022-1034.

Conder, A.L. and Annear, T.C. (1987). Test of weighted usable area estimates derived from a PHABSIM model for instream flow studies on trout streams. *N. Am. J. Fish. Manage.* 7: 339-350.

Copp, G.H. (1990). Shifts in the microhabitat of larval and juvenile roach, Rutilus rutilus (L.), in a floodplain channel. *J. Fish Biol.* 36: 683-692.

Copp, G.H. and Kovak, V. (1995). Does microhabitat use reflect ecomorphological changes in juvenile roach Rutilus rutilus (L.)? *International Symposium on relations between fish and their habitat: Theories, methods and applications.* In press.

Copp, G.H. and Penaz, M. (1988). Ecology of fish spawning and nursery zones in the floodplain, using a new sampling approach. *Hydrobiologia* 169: 209-224.

Cunjack, R.A. (1986). The winter biology of stream salmonids. *PhD thesis*. University of Waterloo, Canada.

Cunjack, R.A. and Green, J.M. (1983). Habitat utilization by brook charr, *Salvelinus fontinalis* , and rainbow trout, *Salmo gairdneri* , in Newfoundland streams. *Can. J. Zool.* 61: 1214-1219.

Degeman, E., Fogelgren, J.E., Tengelin, B. and Thornelof, E. (1986). Occurrence of salmonid parr and eel in relation to water quality in small streams on the West Coast of Sweden. *Water Air Soil Pollut.* 30: 665-71.

De Graaf, D.A. and Bain, L.H. (1986). Habitat use by and preference of juvenile Atlantic salmon in two Newfoundland rivers. *Trans. Am. Fish. Soc.* 115: 671-681.

Dolloff, C.A. (1986). Effects of stream cleaning on juvenile coho salmon and Dolly Varden in southeast Alaska. *Trans. Am. Fish. Soc.* 115: 743-755.

Egglishaw, H.J. and Shackley, P.E. (1985). Influence of water depth on dispersion of juvenile salmonids, *Salmo salar* and S. trutta L. in a Scottish stream. *J. Fish Biol.* 21:141-156.

Elliott, J.M. (1994). Quantitative ecology and the brown trout. *Oxford University Press*. Oxford. 286 p.

Ensign, W.E., Strange, R.J. and Moore, S.E. (1990). Summer food limitation reduces brook and rainbow trout biomass in a southern Appalachian stream. *Trans. Am. Fish. Soc.* 119: 894-901.

Fausch, K.S. (1984). Profitable stream positions for salmonids: relating specific growth rate to net enegy gain. *Can. J. Zool.* 62: 441-451.

Fausch, K.S., Hawkes, C.L., and Parsons, M.D. (1988). Models that predict the standing crop of stream fish from habitat variables: 1950-85. *US Forest Serv., Gen. Tech. Rep.* PVW-GTR-213.

Fraser, N., Metcalfe, N.B. and Thorpe, J.E. (1993). Temperature-dependent switch between diurnal and nocturnal foraging in salmon. *Proc. R. Soc. Lond.* B 252: 135-139.

Gan, K. and McMahon, T. (1990). Variability of results from the use of PHABSIM in estimating habitat area. *Regul. Riv.* 5: 233-239.

Garcia de Jalon, D. (1995). Management of physical habitat for fish stocks. P. 363-374 in Harper, D.M. and Ferguson, A.J.D. *The ecological basis for river management.* Wiley.

Garner, P. (1995). Suitability indices for juvenile 0+ roach (*Rutilis rutilis* (L.)) using point abundance sampling data. *Regul. Riv.* 10: 99-104.

Gibson, R.J. (1993). The Atlantic salmon in freshwater: spawning, rearing and production. *Rev. Fish Biol. Fish.* 3: 39-73

Gibson, R.J., Whoriskey, F.G., Charette, J.Y. and Winsor, M. (1984). The role of lakes in governing the invertebrate community and food of salmonids during the summer in a Quebec boreal river. *Nat. Can.* 111: 411-427.

Gibson, R.J., Stansbury, D.E., Whalen, R.R. and Hillier, K.G. (1993). Relative habitat use, and interspecific and intraspecific competition of brook trout (*Salvelinus fontinalis*) and juvenile Atlantic salmon (*Salmo salar*) in some Newfoundland rivers. p. 53-59 In Gibson, R.J. and Cutting, R.E. (eds). Production of juvenile Atlantic salmon, *Salmo salar* , in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118.

Gore, J.A. and Nestler, J.M. (1988). Instream flow studies in perspective. *Regul. Riv.* 2: 93- 101.

Gotceitas, V. and Godin J.G. (1993). Effects of aerial and in-stream threat of predation on foraging by juvenile Atlantic salmon (*Salmo salar*). P. 35-41 In Gibson, R.J. and Cutting, R.E. (eds). Production of juvenile Atlantic salmon, *Salmo salar* , in natural waters. *Can. Spec. Publ. Fish. Aquat. Sci.* 118.

Greenberg, L.A. (1992). The effect of discharge and predation on habitat use by wild and hatchery brown trout. *Regul. Riv.* 7: 205-212.

Greenberg, L., Svendsen, P, & Harby, A. (1994). Availability of microhabitats and their use by brown trout and grayling in the River Vojman, Sweden. *Proceedings of International Symposium on Habitat Hydraulics*, Tromso, Norway, in press.

Grossman, G.D. and Freeman, M.C. (1987). Microhabitat use in a stream fish assemblage. *J. Zool.* (Lond.) 212: 151-176.

Hartman, G.F. (1965). The role of behavior in the ecology and interaction of underyearling coho salmon (Oncorhynchus kisutch) and steelhead trout (*Salmo gairdneri*). *J. Fish. Res. Bd. Can.* 22: 1035-1081.

Hawkes, C.L., Millar, D.L. and Layher, W.G. (1986). Fish ecoregions of Kansas: stream fish assemblage patterns and associated environmental correlates. *Env. Biol. Fishes* 17 (4): 267-279.

Hearn, W.E. (1986). Interspecific competition and habitat segregation among stream-dwelling trout and salmon: a review. *Fisheries* 12 (5) 24-31.

Hegge, O., Hesthagen, T., and Skurdal, J. (1993). Juvenile competitive bottlenecks in the production of brown trout in hydroelectric reservoirs due to intraspecific habitat segregation. *Regul. Riv.* 8: 41-48.

Heggenes, J. (1991). Comparisons of habitat availability and habitat use by an allopatric cohort of juvenile Atlantic salmon *Salmo salar* under conditions of low competition in in a Norwegian stream. *Hol. Ecol.* 14: 51-62.

Heggenes, J. (1994). Spatial niches in wild, allopatric brown trout (*Salmo trutta*) in heterogeneous lotic environments. Microhabitat selection during summer. *International Symposium on Habitat Hydraulics*, Trondheim, Norway. In press.

Heggenes, J. and Saltveit, S.J. (1990). Seasonal and spatial microhabitat selection and segregation in young Atlantic salmon (*Salmo salar* L.) and brown trout (*S. trutta* L.) in a Norwegian river. *J. Fish Biol.* 36: 707-720.

Heggenes, J., Northcote, T.G. and Peter, A. (1991). Seasonal habitat selection and preference by cutthroat trout (*Oncorhynicus clarki*) in a small coastal stream. *Can. J. Fish. Aquat. Sci.* 48: 1364-1370.

Heggenes, J., Saltveit, S.J., Vaskinn, K.A. and Lingas, O. (1994). Predicting fish habitat use responses to changes in waterflow regime: Modelling critical minimum flows for young Atlantic salmon, *Salmo salar* , and brown trout, *S.trutta*, in a heterogeneous stream. *International Symposium on Habitat Hydraulics*, Trondheim, Norway. In press.

Heggenes, J., Bagliniere, J. and Cunjack, R. (1995). Spatial niche selection and competition in young Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) in lotic environments. *International Symposium on relations between fish and their habitat: Theories, methods and applications*. In press.

Helm, W.T. (1982). Defining stream fish microhabitat requirements for water project planning. *Utah State University*, Logan. 13 p.

Hindar, K., Jonsson, B., Andrew, J.H. and Northcote, T.G. (1988). Resource utilization of sympatric and experimentally allopatric cutthroat trout and Dolly Varden charr. *Oecologia* 74: 481-491.

Irvine, J.R., Jowett, I.G. and Scott, D. (1987). A rest of the instream flow incremental methodology for underyearling rainbow trout (*Salmo gairdneri* Richardson) in man-made New Zealand streams. *N. Z. J. Mar. Freshw. Res.* 21: 35-40.

Jacobs, J. (1974). Quantitative measurement of food selection: a modification of the forage ratioand Ivlev's electivity index. *Oecologia* 14, 413-417.

Johnson, D.H. (1980). The comparison of usage and availability measurements for evaluating resource preference. *Ecology*, 61: 65-71.

Johnson, I.W. & Elliott, C.R.N. (1993). River Allen Habitat Mapping Survey. *Report to National Rivers Authority Wessex Region*, 12 p.

Johnson, I.W., Elliott, C.R.N., Gustard, A and Clausen, B. (1993). River Allen Instream Flow Requirements. ***Report to National Rivers Authority Wessex Region***, 42 p.

Jowett, I.G. (1992). Models of the abundance of large brown trout in New Zealand rivers. *N. Am. J. Fish. Manage.* 12: 417-432.

Kessler, R.K. and Thorpe, J.H. (1993). Microhabitat selection of the threatened spotted darter (Etheostoma maculatum) and closely related orangefin darter (E. bellum). *Can. J. Fish. Aquat. Sci.* 50: 1084-1091.

Kraft, M.E. (1972). Effects of controlled flow reduction on a trout stream. *J. Fish. Res. Board Can.* 29: 1405-1411.

Leonard, P.M. and Orth, D.J. (1988). Use of habitat guilds of fishes to determine instream flow requirements. *N. Am. J. Fish. Manage.* 8: 399-409.

Maki-Petays, A., Muotka, T., Tikkanen, P., Juusko, A., Kreivi, P. and Kuusela, K. (1995). Seasonal changes in microhabitat use by juvenile brown trout. ***International Symposium on relations between fish and their habitat: Theories, methods and applications.*** In press.

Mathur, D., Bason, W.H., Purdy, E.J. and Silver, C.A. (1984). A critique of the Instream Flow Incremental Methodology. *Can. J. Fish. Aquat. Sci.* 42: 825-831.

McNeely, D.L. (1987). Niche relationships within an Ozark stream cyprinid assemblage. *Environ. Biol. Fishes.* 18 (3): 195-208.

Mittelbach, G. (1986). Predator-mediated habitat use: some consequences for species interactions. *Environ. Biol. Fishes* 16: 159-169.

Moore, K.M.S. and Gregory, S.V. (1988). Response of young-of-the year cutthroat trout to manipulation of habitat structure in a small stream. *Trans. Am. Fish. Soc.* 117: 162-170.

Morantz, D.L., Sweeney, R.K., Shirvell, C.S. and Longard, D.A. (1987). Selection of microhabitat in summer by juvenile Atlantic salmon (*Salmo salar*). *Can. J. Fish. Aquat. Sci.* 44: 120-129.

Morhardt, J.E. and Hanson, D.F. (1988). Habitat availability considerations in the development of suitability criteria. P 392-403 in Bovee, K. and Zuboy, J.R. (eds). Proceedings of a workshop on the development and evaluation of habitat suitability criteria. *US Fish Widl. Serv. Biol. Rep.* 88 (11).

Moyle, P.B. and Baltz, D.M. (1985). Microhabitat use by an assemblage of California stream fishes: developing criteria for instream flow determinations. *Trans. Am. Fish. Soc.* 114: 695-704.

Moyle, P.B. and Vondracek, B. (1985). Structure and persistence of the fish assemblage in a small Californian stream. *Ecology* 66: 1-13.

Nehring, R.B. and Anderson, R.M. (1993). Determination of population-limiting critical salmonid habitats in Colorado streams using the Physical Habitat Simulation System. *Rivers* 4: 1-19.

Orth, D.J. (1987). Ecological considerations in the development and application of instream flow-habitat models. *Regul. Riv.* 1: 171-181.

Orth, D.J. and Maughan, O.E. (1982). Evaluation of the incremental methodology for recommending instream flows for fishes. *Trans. Am. Fish. Soc.* 111: 413-445.

Orth, D.J., Jones, R.N., and Maughan, O.E. (1982). Considerations in the development of curves for habitat suitability criteria. P. 124-133 in Armantrout, N.B. *Aquisition and utilization of aquatic inventory information.* Western Division, Am. Fish. Soc. Bethesda, MD.

Raleigh, R.F., Zuckerman, L.D. and Nelson, P.C. (1986). Habitat suitability index models and instream flow suitability curves: Brown trout. *USDI Fish Wildl. Serv., Biol. Rep.* 82 (10.124)

Rimmer, D.M., Raim, R. and Saunders, R.L. (1984). Changes in the selection of microhabitat by juvenile Atlantic salmon (*Salmo salar*) at the summer-autumn transition in a small river. *Can. J. Fish. Aquat. Sci.* 41: 469-475.

Rincon, P.A. and Lobon-Cervia, J. (1993). Microhabitat use by stream-resident brown trout: Bioenergetic consequences. *Trans. Am. Fish. Soc.* 122: 575-587.

Sale, M.J., Railsback, S.F., and Herricks, E.E. (1982). Frequency analysis of aquatic habitat: A procedure for determining instream flow needs. P. 340-346 in Armantout, N.B. (ed.) *Aquisition and utilization of aquatic inventory information.* Western Division, Am. Fish. Soc. Bethesda, MD.

Scott, D. and Shirvell, C.S. (1987). A critique of the Instream Flow Incremental Methodology and observations of flow determination in New Zealand. In Craig, J.F. and Kember, J.B. (eds), *Regulated streams.* Plenum Press, New York.

Shirvell, C.S. (1986). Pitfalls of physical habitat simulation in the Instream Flow Incremental Methodology. *Can. Tech. Rep. Fish. Aquat. Sci.* 1460: 68 p.

Shirvell, C.S. (1989). Ability of PHABSIM to predict chinook salmon spawning habitat. *Regul. Riv.* 3: 277-289.

Simonson, T.D. (1993). Correspondence and relative precision of stream habitat features estimated at two spatial scales. *J. Freshwat. Ecol.* 8 (4): 363-373.

Smith, R.P. (1989). The distribution and habitat requirements of chub (Leuciscus cephalus L.) in several lowland rivers of eastern England. *PhD thesis*, University of East Anglia, 130 p.

Solomon, D.J. and Templeton, R.G. (1976). Movements of brown trout *Salmo trutta* L. in a chalk stream. *J. Fish Biol.* 9: 411-423.

Stalnaker, C.B., Bovee, K.D., and Waddle, T.J. (1994). The importance of the temporal aspects of habitat hydraulics in fish populations. *International Symposium on Habitat Hydraulics,* Trondheim, Norway. In press.

Taylor, E.B. (1991). A review of local adaptation in Salmonidae, with particular reference to Pacific and Atlantic salmon. *Aquacult.* 98: 185-207.

Thomas, J.A. and Bovee, K. (1993). Application and testing of a procedure to evaluate transferability of habitat suitability criteria. *Regul. Riv.* 8: 285-294.

Voos, K.A. (1981). Simulated use of the exponential polynomial/maximum liklihood technique in developing suitability of use functions for fish habitat. *PhD Dissertation.* Utah State University, Logan, 85 p.

Vondracek, B. and Longanecker, D.R. (1993). Habitat selection by rainbow trout *Oncorhynchus mykiss* in a California stream: implications for the Instream Flow Incremental Methodology. *Ecol. Freshwat. Fish* 2: 173-186.

Waddle, T.J. (1992). Are high and low flow habitat values really the same? P. 374-379 in Karamouz, M. (ed.) *Water resources planning and management: Saving a threatened resource - in search of solutions.*

Williamson, S.C. (1994). Using the computer based Physical Habitat Simulation System (PHABSIM). USFWS manual. 142 p.

Wyatt, R.J., Barnard, S. & Lacey, R.F. (1995). NRA R&D Project 338. HABSCORE.

A FULL-SCALE TEST TO VALIDATE THE CONTRIBUTION OF THE IFIM PROCEDURE IN THE CHOICE OF A GUARANTEED FLOW DOWNSTREAM HYDROSTATIONS

G. MERLE, J. EON
Electricité de France/ Generation & Transmission Division/ Hydro Generation Branch
92068 Paris La Défense Cedex 48
France

ABSTRACT

French law lays down the minimum values required by guaranteed flows downstream hydroelectric plants to ensure that the aquatic environment has favourable conditions to the life, circulation and reproduction of species in rivers.

However, the sites are extremely varied with physical and biological characteristics, and water uses that differ to a great extent. It therefore appeared useful to check that the choice of a guaranteed flow met the objectives sought.

Among the evaluation tools available, the IFIM procedure ("microhabitat" methodology in French) is increasingly used to propose a flow which takes into account the quality of the fish habitat. It appeared necessary to validate the pertinence and limits of the information provided by this procedure and, more generally, of the study protocols used to determine guaranteed flows. To do so, measurements are required following the setting up of the thus determined flow.

A national "guaranteed flows" working group has been set up by Electricité de France to establish this operation feedback for a number of sites which will be subject to guaranteed flow increases, on the occasion of the renewal of their licences.

The document describes the approach adopted by the working group, the choice of eight experimentation sites, the present hydrobiological state of these sites, the results of the IFIM procedure for each of them and, finally, the measurement protocols chosen to ensure monitoring following the establishment of a new guaranteed flow.

KEY-WORDS: Guaranteed flow / Habitat / Salmonids / Hydropower.

I. INTRODUCTION

Since 1994, a great many licences for hydroelectric heads have been in the process of renewal. By the year 2000, 138 facilities will be concerned.

An impact study has been provided for each renewal document. The choice of the guaranteed flow is the focus of attention in these studies. It has to reconcile two requirements: draw the best energy part from the dam and make sure that, for the aquatic environment, the conditions are favourable for the life, circulation and reproduction of the aquatic species, while complying with prevailing regulations. Although the consequences of the setting up of a guaranteed flow have been well grasped from the economic standpoint, the knowledge of the ecological aspect still has to be furthered.

An approach has therefore been proposed by EDF, in partnership with other organizations concerned by these issues. Its purpose is to validate the methods for the selection of the guaranteed flow, by taking advantage of the effective increase of the guaranteed flow on the occasion of the renewal of the licences.

II. APPROACH

Of the methods available for the determination of the guaranteed flow, the Instream Flow Incremental Methodology (IFIM, "microhabitat" methodology in French) is increasing used to propose a flow that takes into account the quality of the fish habitat. It has been validated with regard to its principle (fish habitat preferences) and implementation (information to be collected), for Salmonids.

The first objective of the approach is to validate the IFIM procedure a posteriori, through the ecological monitoring of the effects of the increase in the guaranteed flow on a certain number of sites. This feedback should also make it possible to reinforce and perfect the overall approach and the study protocols conducted to determine the guaranteed flows.

A working group has been set up to:

- propose a study programme to be carried out, while specifying the methodological choices,
- select national sites,
- ensure the validation of scientific work,
- provide a significant contribution to the knowledge of rivers with "regulated" flow.

It is made up of representatives from public organizations (Ministry of the Environment, Ministry of Industry, Water Agencies, Conseil Supérieur de la Pêche, EDF) and scientific institutes (Cemagref, Ecole Nationale Supérieure d'Agronomie de Toulouse).

III. CHOICE OF SITES

The choice of the sites must reflect the diversity of the situations encountered in EDF's hydroelectric generation plants: diversity of regions and hydrological conditions, diversity of annual mean flows, diversity of hydro power schemes, etc.

The criteria selected for the choice of sites are as follows:

- annual mean flow and hydrological regime,
- variability of the flow in the bypassed river stretch,
- characteristics of the bypassed river stretch and particularly accessibility,
- possibility of an indicator station,
- knowledge of fishing pressure (possibly fish reserve sector),
- absence of limiting factors (solid transport, quality of water, pollution, etc.),
- presence of incentive factors (high salmon resource, stability of guaranteed flow, etc.),
- local context,
- existing data.

Of a total of 99 impact documents, a first sorting, carried out on criteria concerning existing data, absence of pollution, accessibility, fish quality and local context, led to the selection of 23 sites. The flow criteria (average mean flow and extent of the increase compared with the present situation) have limited the selection to 12 sites. Following an inspection and detailed evaluation of the sites, the choice of the 8 sites was made (Table 1).

IV. CHOICE OF A STUDY PROTOCOL

The choice of the sites was made in part with the knowledge that had already been acquired previously, on the various factors likely to modify the fish populations: quality of water, hydrological and thermal regime, trophic level, habitats offered to living organisms, etc. These data, most of which were collected during the site impact study, are sometimes fairly dated, carried out according to different methodologies and their accuracy may be insufficient to reach the level of knowledge of the ecosystem required by the objectives of the study.

Before monitoring the fish populations subsequent to the increase of the guaranteed flow, it therefore became necessary to provide a phase for the drawing up of **an initial state of the sites**, which describes, with equal quality for each of them, the biological potentialities of the bypassed river stretch in the present situation, i.e. a guaranteed flow set at 1/40 of the average mean flow for the past few years.

Table 1: Choice of 8 experimental sites

Name of site *River*	**Department**	**Average mean flow in m3/s**	**Present guaranteed flow in m3/s**	**Length of bypassed river strech**
Verney *L'Eau d'Olle*	Isère	4	0.100	7.5 km
Pont-Haut *La Roizonne*	Isère	2.75	0.070	5 km
Rory et St Martin *Le Lignon du Forez*	Loire	2.9 and 3.3	0.075 and 0.085	3.5 km and 2 km
Bionnay *Le Bonnant*	Haute Savoie	6.7	0.170	0.7 km
St Georges *L'Aude*	Aude	6.3	0.192	4.1 km
St Lary *La Neste d'Aure*	Hautes Pyrénées	6.6	0.300	5.7 km
Beyrede *La Neste d'Aure*	Hautes Pyrénées	20	0.5	5 km
Fontan *La Roya*	Alpes Maritimes	6.2	0.2	3.7 km

A basic protocol for the monitoring of a site has been developed. It specifies the measurements needed to describe the present hydrobiological state and to monitor the evolution of the environment after the increase of the guaranteed flow. Its design has been an opportunity to formulate a fairly large number of recommendations and rules for the choice of measuring stations, the nature of the studied parameters, the analysis methods to be implemented and the frequency of the measurements.

Duration of studies

The studies are planned for a period of about 5 years: the first year is devoted to additional studies needed to specify the initial state, the next four years must make it possible to reach an aquatic environment state representative of the new hydrological regime.

Study stations

The choice of study stations is particularly important. The stations located in the bypassed river stretch are selected to be representative of the dominant units shown by the morphodynamic study of the overall bypassed river stretch. Their number varies according to the heterogeneity of the sites studied (from 1 to 3). A station must contain the types of habitat of the sector that it represents and, if possible, proportionally to their importance in the overall bypassed river stretch. Its length is generally 10 to 15 times the width of the low-water bed.

Furthermore, study stations outside the bypassed river stretch were sought, which make it possible to have comparison elements. Of the hydropower schemes chosen for this study, it is not always possible to have a **reference station**, located on the same river but in a natural regime and providing a marker on the natural biological potentialities of the river. When this was not possible, it was nevertheless decided to monitor an **indicator station**, located outside the bypassed river stretch, but subject to the same contingencies, especially of a hydrological nature, as this river stretch.

Hydrology and Physico-Chemistry

A good knowledge of the hydrology of the bypassed river stretch is necessary, particulary inflows and overflows (dates, duration and values). Exceptional hydrological events, flood or extended low water level, are useful in explaining a posteriori the responses of a site in terms of biomass.

Thermal monitoring will be performed on a continuous basis throughout the duration of the study in order, for example, to show the limited growth of trout in comparison with durable cold temperatures. A set of measurements at the low water level is scheduled to check the physico-chemical quality for years n+3 and n+4.

Biology

An inventory will be made of invertebrates according to the IBGN protocol (Afnor, 1992) for years n+3 and n+4 so as to have qualitative data. The sorting and determination of invertebrates will be carried out separately for each habitat sampled. Biomass measurements will make to check whether the trophic value is limiting or not.

A fish inventory is made each year throughout the study in order to monitor the development of the fish population in each station. The inventory of spawnings is carried out each year so as to monitor recruitment and determine whether the number of spawnings is limiting. As no site is a fishing reserve, it was requested to "freeze" the present situation concerning the halieutic management of the sections studied, for the 5 years of experimentation.

Morphodynamic state

The IFIM procedure applications are performed in compliance with the protocols recommended by Cemagref and the Research and Development Division of EDF (cf chapter VI).
A check will be carried out on the same stations as during the initial study, in year n+3 or n+4, through a entire IFIM procedure application with a description of the substrate and riparian vegetation:
- in the bypassed river stretch, for the value of the new guaranteed flow,
- in the indicator and/or reference station, for a flow value that is as close as possible to the one corresponding to the initial study.

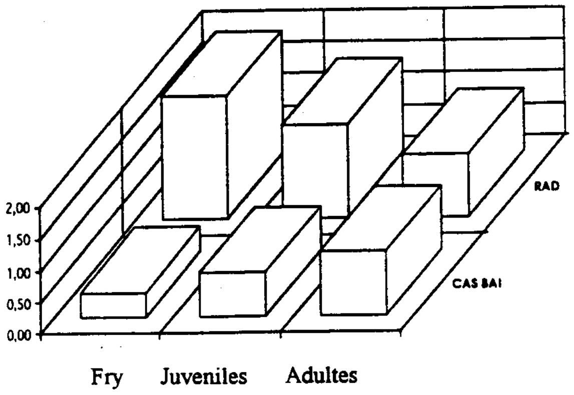
V. PRESENT HYDROBIOLOGICAL STATE OF THE SITES

In order to have initial states of similar quality, two types of additional investigations are performed:
- specific measurements for the purpose of supplementing missing data or checking a point specific to a site, such as the improvement of the physico-chemical quality of water after the installation of a sewage plant,
- systematic measurements, on all the sites, according to standardized protocols, for criteria at the base of the study, i.e. habitat conditions and fish populations.

Fish inventories

For each site, a full exploration of the bypassed river stretch was performed so as to determine the morphodynamic characteristics and establish a sequencing of differents morphodynamic units. This represents the first phase of the implementation of the IFIM procedure since it serves to select the representative morphodynamic units and therefore a station or stations which contain them (see Chapter VI).
In this study, this phase was also a major stage for measurements on biological compartments and more particularly for fish. It was decided to carry out fish inventories on the same stations as those chosen for the measurements of hydrodynamic characteristics.

A specific morphodynamic description of electric fishing stations was made. The lengths and surface areas prospected are a function of the respective size of the various patterns. It is thus possible to express the biomasses measured by homogeneous sections and observe the distribution of the various trout size classes for each type of morphodynamic unit.



Density	Number/ 10 ares
RAD	Riffle
CAS BAI	Cascade - pool

Figure 1: Example of results obtained for differents morphodynamic units in the Verney bypassed river strech

(data from the Lyons Regional Delegation, Conseil Supérieur de la Pêche).

Thus one is able to show the distribution in differents habitats of the three sets: fry (size < 120 mm), juveniles and adults (size > 180 mm) for the present guaranteed flow for the stations of the bypassed river stretch and also for the upstream stations.

For the time being, an icthyological inventory was performed, according to this protocol, on each of the sites during the low water period.

Inventory of spawnings

The presence of reproduction areas in the bypassed river stretch facilitates the establishment of a balanced natural fish population. When one works on study stations which represent a certain linear of the waterway, the selected representative units may not contain identified spawnings, whereas the overall bypassed river stretch provides effective reproduction areas. When the distribution is unknown, which was the case for 5 of 8 sites, it was decided to make an inventory of common trout spawnings (*Salmo trutta L.*).

The common trout reproduction period begins in late autumn and continues into winter. The common trout choose very specific habitats for reproduction. The pool-riffle interfaces are typical areas of trout reproduction. However, in many mountain rivers, there are few pool-riffle alternances. The most used patterns will therefore be flats and riffles (Baglinière, 1991), (Delacoste *et al.*, 1993).

A spawning inventory protocol applicable to all the sites was developed by ENSA Toulouse and the Montpellier Regional Delegation of the Conseil Supérieur de la Pêche. The results obtained on the Pyrenees sites (St Lary, Beyrède) and on the Aude (St Georges) show that the trout in these sectors use reproduction habitats that are quite similar to those seen in other sites in the Pyrenees (Delacoste, 1995) with a priority to flats (depth: 30 cm, velocity: 20 cm/s, gravel from 0.2 to 5 cm in diameter). The gravel surface areas available seem to condition the extent of the reproduction.

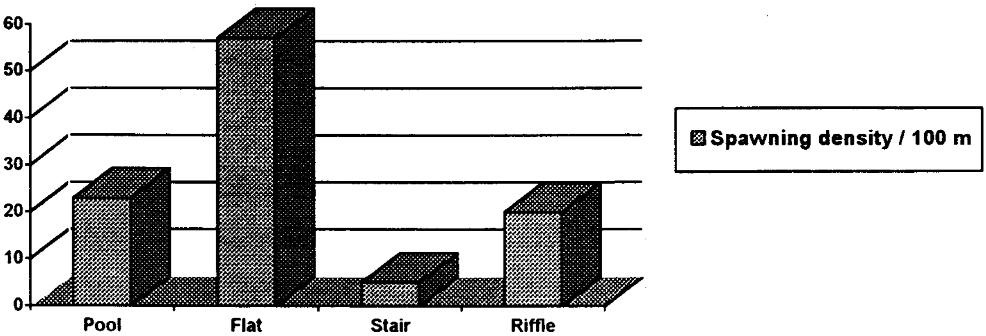


Figure 2: Example of average spawning densities per type of morphodynamic unit on the Beyrède site
(data from ENSA Toulouse)

The present hydrobiological states of 6 of 8 sites have now been completed. Over and above the simple raw analytical results, the special care taken by the many specialists on these sites provides a fine analysis of the functioning of each ecosystem.

VI USING THE IFIM PROCEDURE FOR THE CHOICE OF GUARANTEED FLOWS

Methodology

Many studies have been conducted over the past few years in France to validate and choose a satisfactory implementation of the IFIM procedure. This method consists in describing and simulating the aquatic space available according to the flow and in evaluating a potential habitat capacity. It is based on biological models expressing the relations between the presence or relative density of a fish stage and several variables making up its habitat (Bovee, 1982), (Souchon *et al.*, 1989), (Sabaton and Miquel, 1993).
The application of the IFIM procedure must respect a precise protocol making it possible to obtain the values and graphic supports pertinent for the analysis. It therefore appeared necessary to accurately define the implementation

of this protocol. The application protocols recommended by Cemagref and the Research and Development Division of EDF (Sabaton et al., 1995) were tested and the results prove to be highly compatible. Both protocols are based on major joint guidelines. Each one may be broken down into 5 major stages:

- investigation of the river,
- choice of the representative units which will be specifically studied,
- morphodynamic measurements with one or more flows and hydraulic modelling,
- hydraulic-biology coupling by calculation of grades of fish habitat quality,
- results analysis.

The two approaches mainly differ by the hydraulic modelling used. Cemagref implements a one-dimensional model which requires only single-flow measurements. EDF's Research and Development Division uses a simplified model based on measurements with several flows. This is made when it's possible to control, during data acquisition campaigns, in the sector studied, which is generally the case downstream hydroelectric plants.
Two sites have been studied by applying the Cemagref protocol and four by using the EDF Research and Development Division's protocol.

Implementation

Other than the publication of the two above-mentioned protocols, special instructions have been given so that, regardless of the protocol selected, the present value of the guaranteed flow (1/40 of the annual mean flow) is characterized by in situ measurements. For cases where the EDF Research and Development Division's protocol was applied, measurements for at least a flow value higher than 1/10 of the annual mean flow were performed.
It should also be noted that the habitat was studied as well, for a single flow value (close to the minimum average monthly flow) on the upstream stations (indicator or reference).

Analysis of results obtained and choice of tested guaranteed flows

The results of the IFIM procedure studies carried out on six sites for the moment, have given rise to a series of curves representing the variation, as a function of flow, of the Weighted Usable Area (WUA) per morphodynamic unit, per station and for the whole sector concerned for the three development stages of the fario trout.
Thus we have a first analytical stage making it possible to compare the present hydrobiological state of the site with what it might be with other flows and to make assumptions about the changes that would be provided if the guaranteed flow were to be increased.

However, to choose the guaranteed flows to be tested, it became clear that it was necessary to lay down common principles of assessment which take into account the objectives of the approach, i.e.:

1- Measure the response of the fish populations to a flow increase,

2- Compare this response to the forecast potential developments according to the habitat curves and try to explain any differences,

3- Test the sensitivity of the method.

When the flow increases, the analysis of the curves often shows flow ranges where the WUA variation is high, plateaus and maxima. It is thus possible to propose choices to check the response of the fish populations to flow increases in different WUA development cases (high WUA increase, WUA value stable according to flow).

WUA variations with the flow can therefore be commented upon and analysed according to the other results collected on the same site, such as hydrological regime, fish data, minimum water levels which allow the free circulation of fish throughout the area, available shelter surface areas, etc.

An example of this type of analysis is given below for the Beyrède site:

- All the stations have high fry densities compared with those of adults which therefore seem to be penalized by the present flow conditions. It was consequently agreed that the objective of increasing the guaranteed flow would be to improve the receiving capacity of this stage.

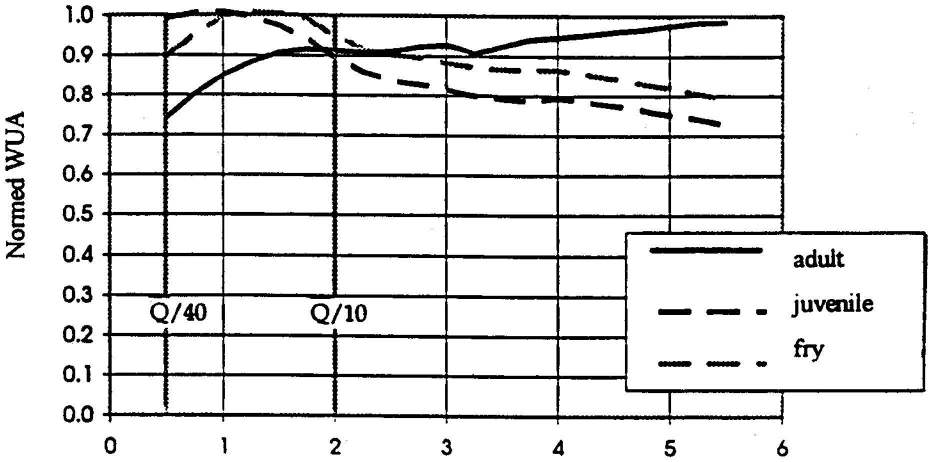


Figure 3: Normed WUA of the full bypassed river stretch according to the guaranteed flow for the three development stages of the fario trout (with intermediate inflows = 0,5 m^3/s)

- For fry and juveniles, the figure 3 shows that the maximum WUA is obtained for a flow of about 1 m3/s. At this flow, the WUA reaches 85% of the maximum value for the adult. However, the value of 1.5 m3/s provides a noteworthy gain of the WUA for the adult stage (which we wish to favour), without excessively penalizing the two other stages. Finally, at 2 m3/s, the WUA would be reduced by 5 to 10% for fry and juveniles with an increase of just 1% for adults.

In conclusion, it would appear worthwhile to test at Beyrède the value of 1.5 m3/s which corresponds to a sizeable improvement of the quality of potential habitat for adults (progressing when the flow increases) likely to bring about an appreciable response of the adult populations while maintaining good conditions for the young stages (regressing when the flow increases).

This assessment common mode will continue to be specified during the study. For the time being, it serves to determine the flow ranges in which the flows to be experimented will be selected for the other sites.

VII MONITORING AFTER INCREASE OF THE GUARANTEED FLOW

As stated in the definition of the basic protocol, the monitoring of the evolution of the aquatic environment will take place over a four-year period, the time required to reach a state of balance after the change of the guaranteed flow, with a particular effort made in the third and fourth years.

The results of the fish inventories confronted with the hydrological evolution of the site should make it possible to show an improvement in the development of the fish population. In this very particular "regulated" flow context, an effort will be made to show the existence of a relationship between the WUA calculated by the IFIM procedure and the fish biomass observed.

The experts who monitor each of the sites will have the possibility of defining more specific investigations which, as the study progresses, will appear to be necessary with intend to obtain a reliable conclusion on the response of fish fauna to the flow increase.

DISCUSSION AND PROSPECTS

The increase of guaranteed flows subsequent to the renewal of a number of licences is an opportunity to make a "full-scale" check of the pertinence and sensitivity of information provided by the IFIM procedure to test the response of the aquatic ecosystem to a variation in the guaranteed flow and help choose a flow to be guaranteed downstream of a hydropower plant.

Many developments are under way to specify the field of use of the IFIM procedure (Pouilly and Souchon, 1995). The purpose of the approach undertaken here is to reinforce the use of this method in the particular field of experimentation concerning river stretches with "regulated" flow. In this context, the necessary conditions seem to be present to obtain a relationship between WUA and the biomass of a fish population.

The criteria taken into account on the experimental sites, to choose the flow to be tested over a four-year period for the purpose of validating the IFIM procedure, concern only the aquatic ecosystem. They can therefore be differed from those used for the final choice of the guaranteed flow of a site and which may satisfy many other objectives: meeting water requirements, economic effects, etc.

The approach accompanied by a working group bringing together all the skills from various fields provides the scientific stringency of this approach and is an opportunity to federate efforts to further the implementation of studies and the use of results. This has already made possible progress in the measurement protocols used and the result assessment mode.

In the long-term, the goal of this work is to provide a pertinent decision-aid tool that will help improve the dialogue between those in charge of the operation of hydroelectric facilities and the entities responsible for overseeing compliance with the general objectives of legislation concerning water.

REFERENCES

Bagliniere, J.L. (1991). La truite commune (*Salmo trutta L.*), son origine, son aire de répartition, ses intérêts économique et scientifique, in Baglinière J.L., Maisse G., La truite, biologie et écologie, *INRA Publ.*, Paris, pp.12-22.

Bovee, K.D. (1982). A guide to stream habitat analysis using the Instream Flow Incremental Methodology. *Instream Flow Information Paper n°12*, FWS/OBS 82/86, 248p.

Delacoste, M. (1995). Analyse de la variabilité spatiale de la reproduction de la truite commune (*Salmo trutta fario*). Etude à l'échelle du micro et du macrohabitat dans six rivières des Pyrénées Centrales Françaises. *Thèse de Doctorat*. INP de Toulouse.

Delacoste, M., P. Baran, F. Dauba and A. Belaud (1993). Etude du macrohabitat de reproduction de la truite commune (*Salmo trutta L.*) dans une rivière pyrénéenne, la Neste du Louron. Evaluation d'un potentiel de l'habitat physique de reproduction. *Bull. Fr. Pêche Piscic.*, 331, pp. 341-346.

Norme AFNOR NF T90-350 (1992). Détermination de l'indice biologique global normalisé (IBGN). *AFNOR* 1-9.

Pouilly, M. and Y. Souchon (1995). Méthode des microhabitats: validation et perspectives. *Bull. Fr.Pêche Piscic.*, 337/338/339 pp.329-336.

Sabaton, C. and J. Miquel (1993). La méthode des microhabitats: un outil d'aide au choix d'un débit réservé à l'aval des ouvrages hydroélectriques. Expérience d'Electricité de France. *Hydroécologie Appliquée*, 5(1) pp.127-163.

Sabaton, C., S. Valentin and Y. Souchon (1995). La méthode des microhabitats - Protocoles d'application. *Rapport EDF/DER - Cemagref* HE-31/95/10.

Souchon, Y., F. Trocherie, E. Fragnoud and C. Lacombe (1989). Les modèles numériques des microhabitats des poissons: application et nouveaux développements. *Rev. Sci. Eau*, 2 pp.807-830.

MODELLING INSTREAM FLOW NEEDS: THE EFFECTS OF A WATER TRANSFER SCHEME ON MACROINVERTEBRATE COMMUNITIES IN THE RECEIVING RIVER WEAR

C.N. Gibbins & M.J. Jeffries
Division of Geography and Environmental Management, University of Northumbria,
Newcastle upon Tyne, UK.

C. Soulsby
Department of Geography, University of Aberdeen, Elphinstone Road,
Aberdeen, UK.

C.R.N. Elliott
Institute of Hydrology, Crowmarsh Gifford, Wallingford,
Oxfordshire, UK.

ABSTRACT

Water transfers into the River Wear, north-east England, are made to support a statutory Minimum Maintained Flow (MMF) of 2 $m^3\ s^{-1}$. The transfer release policy has been based on the need to avoid extreme summer low flow conditions which in the past have resulted in reduced levels of oxygen and fish kills in the river. However, transfers result in sudden and marked increases in flow and may themselves represent hydrological disturbance events which perturb the river ecosystem. This paper describes the use of PHABSIM to assess the relative impact of low flow conditions and water transfer discharges on invertebrate communities at a site on the Wear.

Previous PHABSIM applications have used habitat suitability data for target species to model the effects of flow change. The river management implications of using this species-specific approach are discussed. A method of generating habitat suitability indices based on the requirements of whole invertebrate communities is described. The method uses TWINSPAN and DECORANA to delineate community groups; locality-specific substrate, depth and velocity data are then used with invertebrate community sample information to produce habitat suitability indices.

PHABSIM output for the invertebrate community suggests that transfers do not result in gross alterations to the availability of suitable habitat. Time series analysis of a 43 day transfer release detected no marked increase or decrease in weighted usable habitat at the study site. The augmentation of flows through the use of transfers does however help avoid habitat loss which occurs as discharge drops toward or below the 2 $m^3\ s^{-1}$ MMF.

KEY WORDS: PHABSIM / habitat suitability indices / invertebrate communities / water transfers / usable habitat / River Wear

INTRODUCTION

The need to abstract water for industrial and potable supply without threatening river ecology is a central problem in water resource management. In England and Wales the statutory mechanism for ensuring in-river flow needs are met is the Minimum Maintained Flow (MMF); a specified discharge below which flows must not fall. MMFs have been widely criticised as a mechanism for managing river flows (Petts and Maddock, 1994): many were established by 'rule of thumb' and relate to historical industrial and agricultural demands which may no longer be relevant (Gustard *et al.*, 1987). Specific criticisms arise because MMFs tend to be purely hydrological in derivation, based on a fixed percentage of average flow statistics.

Petts and Maddock (1994) emphasised the need to link the disciplines of hydrology and ecology in prescribing river flows and introduced the idea of Ecologically Acceptable Flows (EAFs). This forms the basis of the IFIM methodology and the associated model Physical Habitat Simulation (PHABSIM). PHABSIM models changes in in-stream habitat in response to flow alterations, the predicted changes allowing assessment of the flows required to achieve a desired ecological goal. PHABSIM is now a legal requirement in many states in USA for establishing ecological in-stream flow needs during environmental impact assessments (Bovee, 1982). It is currently being used to assess the minimum flow requirements of some UK rivers although to date has been applied to only a small number of systems, principally targeting fish species in streams in southern parts of England (Johnson *et al.*, 1993; Newman, 1995).

The River Wear rises at over 600 m a.s.l. in the Pennine uplands in northern England and flows eastward into the North Sea (Figure 1). Flows in the Wear are augmented by inter-basin transfers of water from the Kielder system which allow water abstractions for industrial and potable supply to continue during summer droughts. Transfers are used intermittently to ensure the Wear's 2 $m^3 s^{-1}$ MMF is not violated. Transfer releases can more than double receiving flows and result in tenfold increases in bed velocity in some parts of the channel (Gibbins, 1996). These flow changes are known to result in short-term alterations to the distribution and abundance of some invertebrate species in the river (Archer and Gibbins, 1995). Conversely, the role of a 2 $m^3 s^{-1}$ minimum flow in maintaining in-stream habitats has never been investigated. This paper uses PHABSIM to assess the effects of a transfer made in the summer of 1994 on in-stream habitats using community based invertebrate habitat suitability indices. The method of producing the community-level indices is described and some of the advantages over single species PHABSIM modelling are discussed.

STUDY AREA

The River Wear lies between 54 and 55 ^{o}N and has a catchment area of 1174 km^2 (Figure 1). The geology, topography, climate and hydrology of the catchment have been described in detail elsewhere (Gibbins *et al.*, 1994). The western part of the catchment is upland in character, with landuse dominated by open moorland. Eastward, and with decreasing altitude, the catchment become progressively more urbanised.

Mean daily flows in upper parts of the catchment are around 3.5 $m^3 s^{-1}$ (Stanhope), while in lower parts they increase to 14 $m^3 s^{-1}$ (Chester-le-Street) (Figure 1). The Wear is generally considered 'flashy' due to the dominance of peaty soils and steep channel slopes in the headwaters. Seasonal patterns in evaporation give rise to late summer and early autumn low flows, despite the relatively high rainfall over this period. The lowest recorded flow for Stanhope is 0.235 $m^3 s^{-1}$, recorded in September 1959, and that for Chester-le-Street 2.287 $m^3 s^{-1}$, recorded in July 1984. It is over such periods that transfer releases from the Kielder system are made at Frosterley to support the instantaneous MMF which is measured at Chester-le-Street.

Releases are variable, with their magnitude and duration depending upon prevailing climatic conditions. In some years releases are not necessary; in other years they extend over several months (Table 1).

Table 1: Volumes and rates of transfer from the Kielder system to the River Wear, 1984 to 1995. Volumes are in million m^3, discharges are in m^3 s^{-1}

	1984	1985/8	1989	1990	1991	1992	1993	1994	1995
Total volume released	0.50	Nil	7.83	5.27	1.04	1.20	0.30	2.48	4.01
Number of transfer days	35	Nil	145	105	46	38	18	56	93
Average transfer flow	0.17	-	0.62	0.58	0.26	0.36	0.46	0.48	0.43
Max. transfer flow	0.42	-	0.87	0.84	0.37	0.87	0.46	0.96	0.96
Min. receiving flow	0.43	-	0.48	0.35	0.36	0.50	0.93	0.30	0.27

Mean daily flow at Frosterley: 4.17
Mean daily flow at Chester-le-Street: 14.60

A study site was selected at the nearest gauging station downstream from where transfers enter the river. The site (Witton Park) is 18 km downstream from the Frosterley transfer release point and has a mean daily flow of 7.78 m^3 s^{-1}. It is representative of much of the middle section of the Wear: extensive cobble/pebble bars, a lightly wooded riparian zone and limited marginal plant communities.

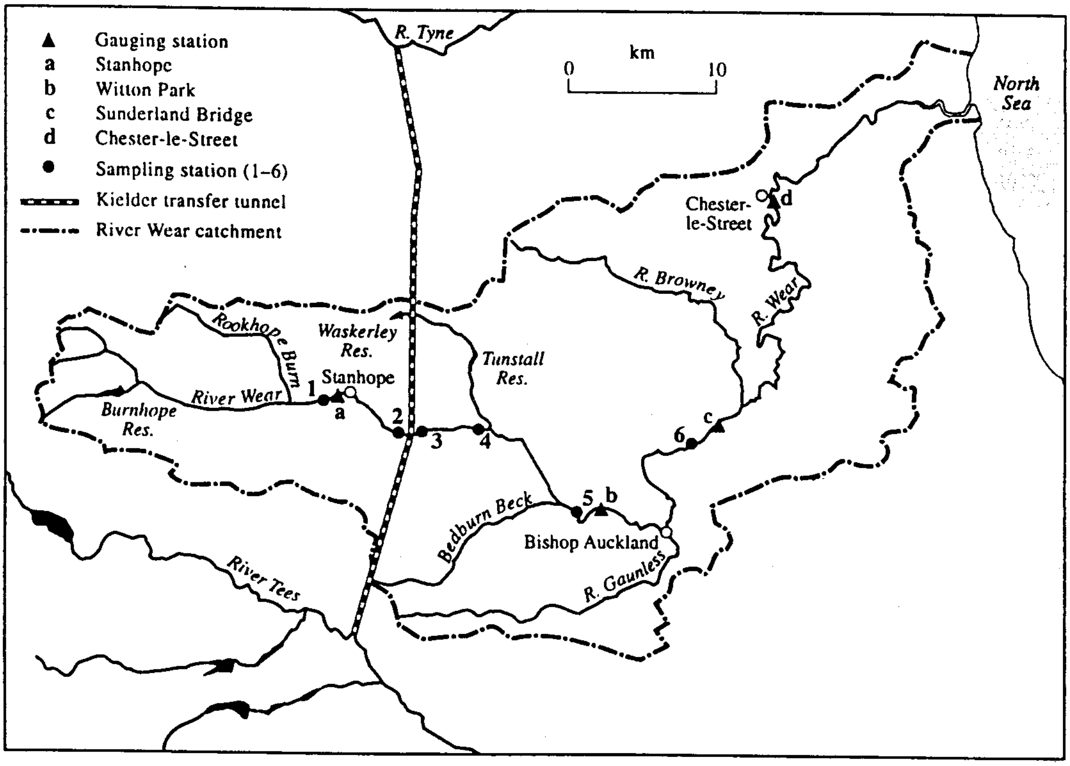


Figure 1: The River Wear catchment showing sampling sites and the Kielder transfer system. Witton Park is site 5

METHODS

Hydraulic Modelling

Since this paper focuses principally upon the possible use of invertebrate community suitability indices, only a summary of hydraulic aspects are given. Hydraulic data collection and modelling components of the work used standard techniques for the application of PHABSIM in the UK (Johnson *et al.*, 1994). Data were collected between June and November 1994 with simulations based upon calibration flows at 1.480 $m^3 s^{-1}$ (Cal 1), 25.988 $m^3 s^{-1}$ (Cal 2) and 20.0 $m^3 s^{-1}$ (Cal 3). Velocity data were taken at Cal 1 as this was nearest to the MMF.

Changes in habitat availability occurring around low flows were of primary interest since this is when Kielder transfers are made. The simulation range was kept at the lower end of the range of historical flows recorded at Witton Park. This was because of the principal interest in low flows but also because of the difficulties of extrapolating well above calibration discharges. Thus, habitat predictions were made for flows between 0.25 and 25 $m^3 s^{-1}$.

Habitat Modelling

The invertebrate community suitability indices were constructed in two stages. First, a number of communities were delineated on the basis of 220 kick samples taken during 1993 and 1994 from six sites along the Wear (Figure 1). Sampling locations at each site were identified by random selection of localities within a 5 m^2 grid placed over each of five separate riffles. For each site, a single sample was collected from each riffle on each of nine sampling occasions. The sites were positioned throughout a 44 km length of the river; they included both upland and lowland sections as well as localities up and downstream of the Kielder release point. Second, a target community was selected from these (see below) and the distribution and abundance of its component species analysed with respect to the depth, velocity and substrate characteristics of sampling points. The physical habitat and invertebrate data for this second stage were collected on 18 July and 3 August 1994 at four of the sites. Immediately prior to each invertebrate sample being taken, velocity, depth and substrate data were collected at each locality. This process generated a total of 40 one minute kick samples (four sites, five replicates per site, two dates), with each sample comprising the whole invertebrate community and physical habitat data required by PHABSIM. The community indices were calculated from this information.

Delineating Community Groupings

The 220 invertebrate samples from six sites over 1993 and 1994 were first classified using TWINSPAN (Hill, 1979a). The two years' data were analysed separately to generate endgroups for all the species records. The endgroups produced by TWINSPAN were then checked to see if any species were classified together in each of the two years. These consistent species groups represented subsets of the original TWINSPAN endgroups.

The second stage involved plotting all of the species data for 1993 and 1994 on DECORANA (Hill 1979b) ordinations. Species subsets identified by TWINSPAN were linked to produce polygons on the DECORANA plots for each year separately. Further species were then added to the each subset, this time based purely on their ordination position. This involved adding species which were part of the original TWINSPAN group in only one of the two years but fell inside the boundaries of the subset when plotted on the ordination in both 1993 and 1994. Finally, species whose ordination position in both years indicated greater affinity with particular subsets than suggested by the initial TWINSPAN classification were included in the community groupings.

The process essentially identified groups of species selected by both TWINSPAN and DECORANA as being associated with each other over a two year period at sites over a 44 km length of the river. Where any differences occurred, the DECORANA results took precedence. This was because DECORANA looks at the distribution of all the species present in the samples and their abundances to identify groupings whereas TWINSPAN is based solely on the distribution of indicator species. The final community groups are listed in Table 2 and illustrated in Figure 2. Species numbers used in the Figure relate to those in Table 2.

The ordination arranged species principally along DECORANA axis 1 (horizontal axes in Figure 2) and suggests site affinities. For example PHABSIM community group 1 comprises many non-insect taxa associated with the most downstream site 6. Groups 3 and 4 are made up predominantly of mayflies and stoneflies found at upstream sampling sites. Group 2 was selected for PHABSIM modelling. It is a taxonomically diverse group with individual taxa primarily found in the mid- and upper-sections of the Wear. Group 1 was dismissed since it is an association of species not present or rare at Witton Park. Groups 3 and 4 comprised small numbers of often scarce species; numbers of individuals available for generating habitat suitability indices would therefore be relatively small if these groups were selected for modelling.

Table 2: Species groups identified for Physical Habitat Simulation (PHABSIM) of a site on the River Wear

Group 1 (species no.)	Group 2 (species no.)	Group 3 (species no.)	Group 4 (species no.)
Pisidium sp.(3)	*Leuctra fusca* (27)	*Rhithrogena semicolorata* (17)	*Caenis rivulorum* (24)
Asellus aquaticus (10)	*Ecdyonurus dispar* (20)	*Perla bipunctata* (30)	*Hydracarina* sp. (12)
Sphaereum sp. (4)	*Dicranota* sp. (55)	*Rhyacophila dorsalis* (40)	*Protonemura meyeri* (25)
Gammarus pulex (11)	Simuliidae (91)	*R. obliterata* (42)	
Prodiamesa olivacea (65)	*Polypedilum laeteum* gp.(82)	*Dinocras cephalotes* (29)	
Erpopdella octoculata (8)	*Sympsiocladius* (85)		
Cricotopus sp. (71)	*Eukieferiellia* sp. (77)		
Microtendipes sp.(86)			
Polycelis felina (1)			
Paratanytarsus sp. (84)			
Leuctra geniculata (26)			
Pottastia longimana gp. (67)			
Ephemerella ignita (21)			
Hydropsyche pellucidula (47)			

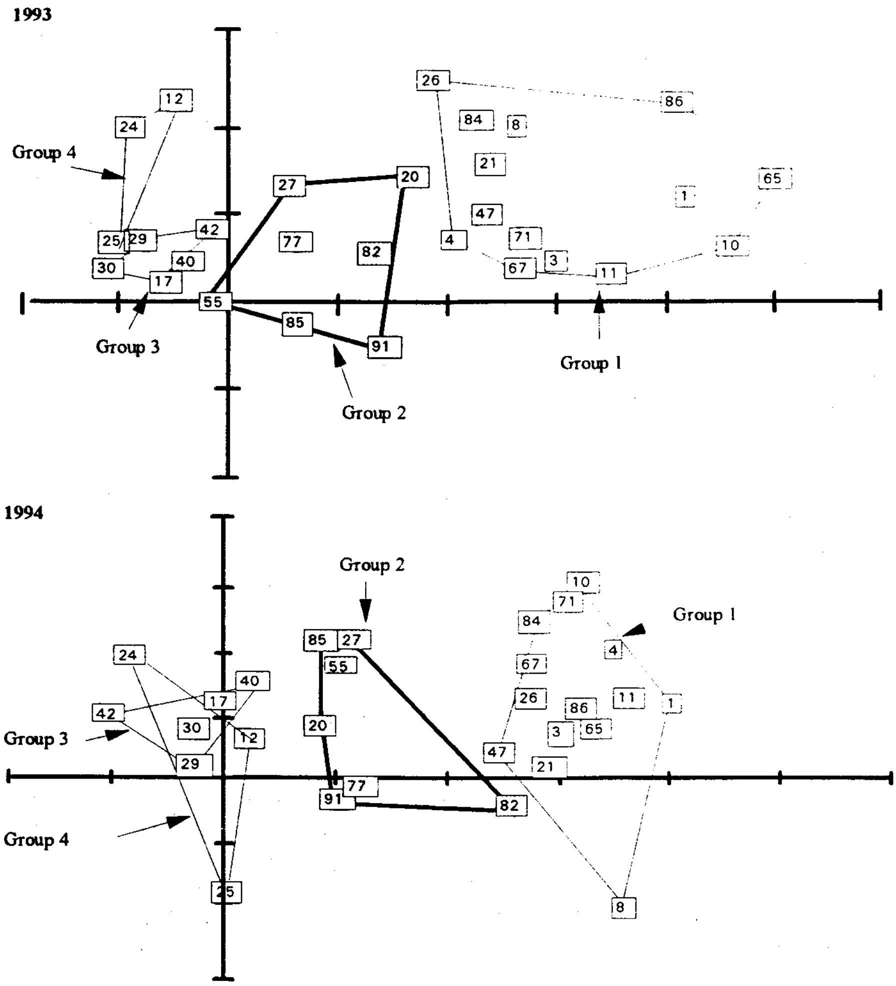


Figure 2: Detrended Correspondence Analysis (Hill, 1979b) ordination of invertebrate species found at six sites on the River Wear, 1993 and 1994. Species numbers are given in Table 2; those species comprising community group 2 are highlighted.

Calculation of the habitat suitability values for community 2 used the 40 invertebrate kick samples with the velocity, depth and substrate information for each. Habitat suitability indices were generated for community 2 in the same way as for individual species described by Bovee (1982). Rather than the abundance of a single species, the combined abundance of the individuals comprising species from community group 2 ('community abundance') formed the basis of the curves. For each habitat variable, suitability takes into account the utilisation of a given value as well as the overall availability of localities (i.e. samples) with that particular value. Thus, suitability is expressed by utilisation divided by availability.

Rather than continuous variables, each habitat variable was broken down into categories and suitability values for community 2 assigned to each. Values were then re-scaled to the range 0-1 to produce the final habitat suitability curves (Figure 3). These values were input to PHABSIM as though representing a single target species.

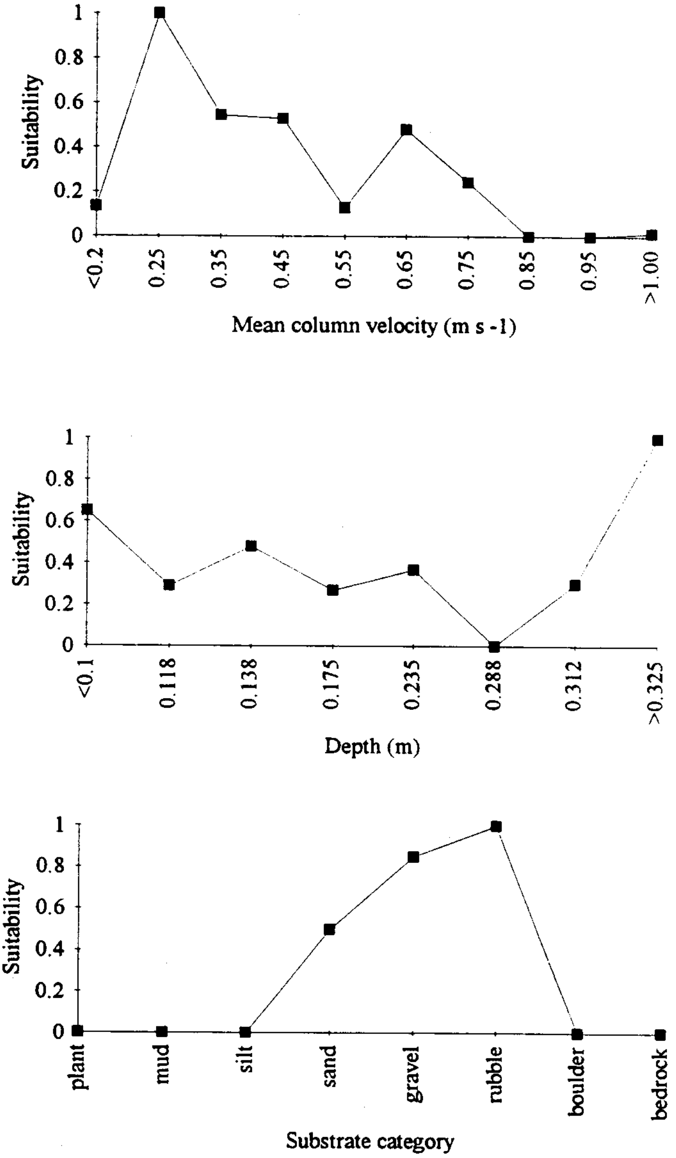


Figure 3: **Habitat suitability indices for a macroinvertebrate community found at sites on the River Wear over 1993 and 1994.**

RESULTS

Figure 4 shows a plot of total available habitat area at Witton Park over the simulated flow range. In terms of this 'potential available habitat', maximum habitat area is provided at the highest flows. Figure 5 shows usable habitat over the simulation flow range for invertebrate community 2. The actual Weighted Usable Area (WUA) values predicted by PHABSIM have been re-scaled as in Petts and Maddock (1994) to show relative changes in 'weighted usable habitat'. Optimum discharge is reached around 15 $m^3 s^{-1}$ with no gross reduction in suitable habitat at the higher end of the simulation range.

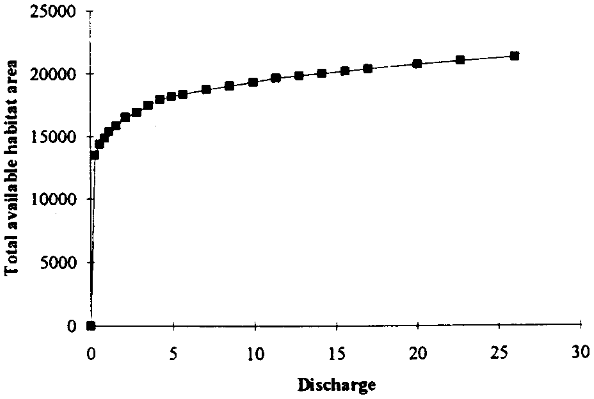


Figure 4: **The relationship between discharge ($m^3 s^{-1}$) and total or 'potential' available in-stream habitat (m^2) at Witton Park on the River Wear**

Can the shape of the curve can be interpreted in the context of the site and the community ? Community 2 is made up predominantly of animals characteristic of riffle habitats: stoneflies, mayflies and simulids. Although Figure 3 shows an optimum velocity of between 0.21 and 0.3 m s^{-1}, these animals were abundant in samples from habitats within the mid and higher velocity categories. This is reflected in a quite even distribution of suitability values up to velocities of 0.8 m s^{-1}. Increasing discharge at Witton Park is accompanied by increasing potentially available habitat as shown in Figure 4. It is also likely to be accompanied by increasing frequency of high velocity areas; a process illustrated by Soulsby *et al.* (1995) who showed how mean and maximum velocities in both riffle and pool habitats on the Wear increase with increasing discharge. So, the tolerance of this group of taxa to relatively high velocities may result in habitats remaining suitable at higher discharges. Only above 15 $m^3 s^{-1}$ does usable habitat begin to diminish.

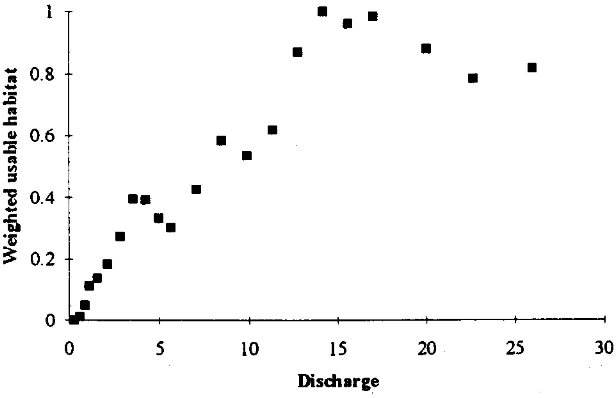


Figure 5: **The relationship between discharge ($m^3 s^{-1}$) and weighted usable habitat for invertebrate community 2 at Witton Park on the River Wear**

Of the three habitat suitability curves for community 2 (Figure 3), that for depth is the least discriminating. It suggests little or no preference for depths within the observed range although the zero value for a mid-range category may influence the weighted usable habitat curve. Because of the re-scaling process, a zero suitability corresponds to the depth with the lowest observed community abundance, rather than a complete absence of animals. A sensitivity analysis was therefore undertaken by substituting a suitability value of one for all depth categories other than zero depth which was assigned a suitability value of zero. This retained the apparent insensitivity of the community to depth but removed the possible influence of the single, mid-range zero value. Results of this are shown in Figure 6. The optimum discharge and overall shape of the curve remain, as do the minor peaks on the rising limb. This suggests that the depth suitability values are not having a major influence on the weighted usable habitat curve and that the hydraulic characteristics of the site may be responsible for the smaller peaks and troughs.

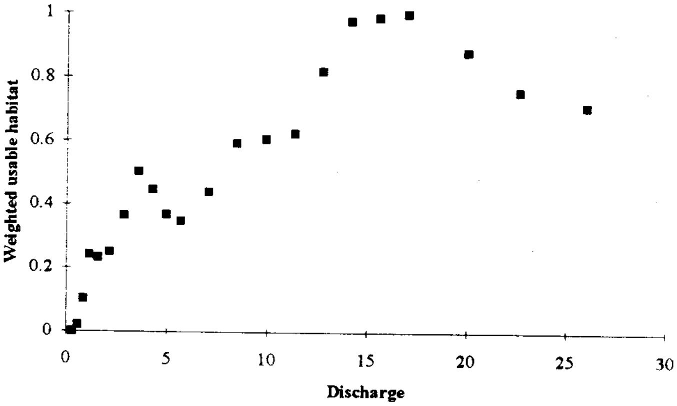


Figure 6: The relationship between discharge (m^3 s^{-1}) and weighted usable habitat for invertebrate community 2 at Witton Park on the River Wear. The curve is based on similar habitat suitability data to Figure 5 except that the community has been given an equal preference for all observed water depths other than zero depth.

Figure 7 shows the result of a time-series analysis of weighted usable habitat during the summer and early autumn of 1994 for invertebrate community 2. This is based upon mean daily flow data from the Witton Park gauging station. Habitat values are all relative to the optimum which occurs at around 15 m^3 s^{-1}. The figure shows relatively low values for most of this period, although usable habitat increased during September as flows increased as a result of rainfall. The period when Kielder water was being transferred to the Wear is also shown. The magnitude of the release was variable, ranging from 0.25 to 0.96 m^3 s^{-1} with an average of 0.65 m^3 s^{-1}. Figure 7 suggests that relative to the optimum, transfers resulted in only a minimal increase in usable habitat at Witton Park, 18 km downstream from the release point. However without transfers habitat values would have progressively fallen during July, reaching a minimum of less than 0.1 in the latter part of the month.

DISCUSSION

This paper did not set out to provide a critique of the whole PHABSIM modelling process; such analyses are readily available elsewhere (Mathur *et al.*, 1985; Scott and Shirvall, 1987). Rather, by looking at a group of invertebrates which are abundant and widespread in the Wear, it has begun to develop a method of improving the value of PHABSIM predictions to river managers. PHABSIM provides an ecological basis for flow management decisions (Petts *et al.*, 1995) and the present research has provided an opportunity to apply the system to low flow issues on the Wear.

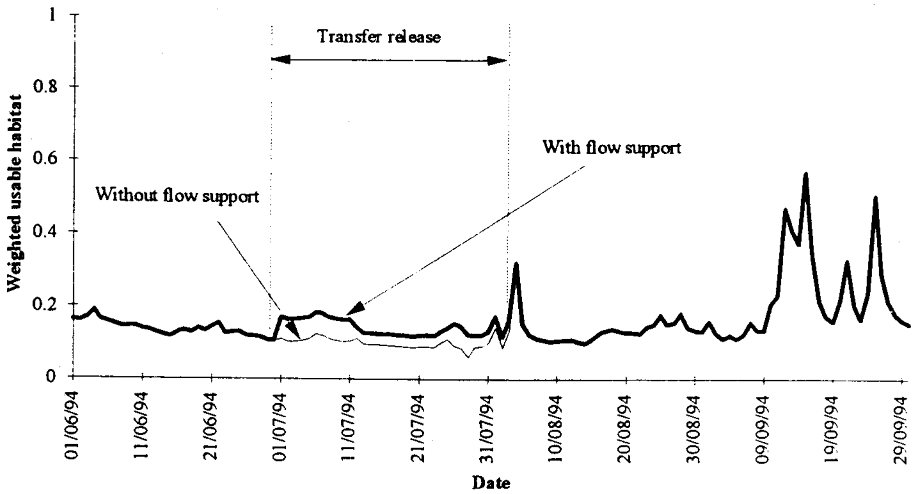


Figure 7: **Changes in weighted usable habitat for invertebrate community 2 at Witton Park, River Wear, between 01/06 and 30/09/94**

Armitage and Ladle (1991) discussed the problems of developing habitat suitability indices for individual invertebrate species. They acknowledged that:

> *'the most accurate estimates of habitat preferences are derived from detailed analysis of distribution patterns with respect to specific variables measured at the point at which a faunal sample is taken'.*
>
> (Armitage and Ladle, 1991. pp. 88-89)

Accepting that such techniques are time consuming and costly, they suggested 'cruder estimates' have to be used. They presented invertebrate suitability curves based upon Institute of Freshwater Ecology data which were used to develop the RIVPACS software. These field data were collected for the purpose of site classification and so the sampling method did not take into account within-site distribution patterns. Also, some of the curves simply conveyed presence and absence information, rather than abundance, and took no account of habitat availability. These shortcomings were acknowledged by Armitage and Ladle whose report represented the first attempt to develop indices for invertebrates in the UK.

Producing suitability indices for invertebrate species, incorporating utilisation and availability data, using detailed point sampling methods remains problematic. At the species level, where abundance values may be low, relatively small differences in abundance between points being sampled can dramatically alter the shape of the curves. At a particular sampling point, a species' abundance may be misrepresented for a number of reasons. First, anomalies may result from differences in sampling efficiency between different habitats for that species. Second, a single species may be slightly more or less abundant at a point for reasons other that the physical habitat; the absence or presence of a predator for example. For a single species, seasonal patterns of larval abundance may result in changes which are independent of the suitability of sampling location. Chance events may also influence the abundance of a particular species in a particular sample. Overall, the less abundant the target species, the greater influence such factors are likely to have on the suitability curves. This issue becomes particularly important given that management is frequently targeted at rare species which, by definition, are less abundant than others in the community.

From a river management point of view, species-specific invertebrate modelling may be of limited value: flow management strategies based upon the requirements of a single invertebrate species are difficult to justify politically. Rare species are preferable in this respect, but the shortcomings outlined above remain problematic. Also, habitat requirements of adult stages may be equally or more important than those of the aquatic larvae; for example some dragonfly (Odonata) species require extensive terrestrial areas over which to forage (Hammond, 1985). For these reasons it may be preferable to attempt to develop suitability curves for groups of co-occurring species. Chance or sampling-related differences in the combined abundance of animals making up a community are likely to be proportionally less influential, while the position of river managers becomes more defensible.

Weighted usable habitat predictions using suitability data for an invertebrate community consisting of stonefly and mayfly species, Simuliidae as a group, two chironomid taxa and the tipulid *Dicranota* sp. were produced for a reach of the Wear at Witton Park. These data were used to illustrate the impacts of Kielder transfers. Only a small increase in weighted usable habitat was evident from the time-series plot over the 1994 transfer period. There was no evidence of the transfer having a negative impact on usable habitat; such impacts may have occurred if, for example, extensive areas of the site became too fast-flowing for the target community. This is consistent with the results of previous work: Gibbins (1996) showed that the 1994 transfer release did not result in alterations to invertebrate community structure along the Wear, including Witton Park. However, Figures 5 and 6 suggest that by augmenting flows, transfers do help avoid the gross reductions in available habitat (i.e. to less than half the maximum) which occur at Witton Park at flows below approximately 3 $m^3 s^{-1}$. Figure 7 illustrates conditions in 1994, where transfers maintained weighted usable habitat at a fairly consistent level over a month period when otherwise it would have fallen to below one tenth of the maximum.

On average, flows at Witton Park are approximately half those at the Chester-le-Street MMF point. Thus even when flows at Chester-le-Street are slightly above 2 $m^3 s^{-1}$, with Kielder releases therefore not necessary, discharges at Witton Park result in weighted usable habitat values which fall below one-quarter of the maximum (Figures 5 and 6). This suggests that the current MMF still results in marked habitat loss in middle sections of the Wear such as Witton Park.

As with all PHABSIM applications, it is important to emphasise that setting flow at an optimum level does not imply that biomass of a target species or community will be maximised. Other factors may be limiting and numerous studies have shown the poor relationship between predictions of optimum habitat and fish biomass (Orth and Maughan, 1982; Shirvall and Morantz, 1983). Moreover, Scott and Shirvall (1987) reviewed PHABSIM studies and found than none were able to demonstrate a relationship between changes in Weighted Usable Area and fish production following hydrological disturbance. Despite their importance to river biota, changes to sediment transport processes which result from flow regime alterations are often overlooked by river managers (Carling, 1995). If marked changes in a river's flow regime are made on the basis of PHABSIM weighted usable habitat predictions, site specific sedimentary conditions on which the predictions were made may change. Therefore it does not necessarily follow that changing flows will optimise habitat availability. None of the community weighted usable habitat curves have been validated by subsequent field observations of community distribution or abundance patterns on the Wear. Such analyses need to be undertaken before flow management decisions based upon PHABSIM are taken. Clearly problems with PHABSIM exist and acceptance of its predictions needs to be tempered by an understanding of its limitations.

Invertebrate communities were identified using classification and ordination techniques widely used in ecological studies to show associations of species or samples (e.g. Weatherley and Ormerod, 1987; Rutt *et al.*, 1990). Simple kick sampling techniques were then used to look at the occurrence of the target community in localities with different physical characteristics and to generate habitat suitability indices. More detailed sampling programmes could improve upon the indices presented in this paper. Also, multivariate methods may allow an assessment of the relative importance of the three physical habitat attributes; this could then be considered in the habitat modelling component of PHABSIM. Heggenes (1994) discussed the relative importance of depth, velocity and substrate to brown trout *Salmo trutta* and atlantic salmon *S. salar* in governing distribution patterns In the absence of alternative data, the three habitat attributes were assumed to be of equal importance for Wear invertebrates. Habitat suitability indices drawn up for several taxa are likely to be less sharply defined than for a single target species since the habitat requirements of each taxa will differ to some extent. The fewer the number of species included, the more discriminating the curves are likely to be. This will affect the WUA curves produced by PHABSIM.

The use of community level data does not negate any of the more fundamental criticisms of PHABSIM. For example, shortcomings related to simplifications in the hydraulic models or the use of just three habitat variables to represent 'niche' remain. Nonetheless, the findings of this preliminary study suggest that a community level approach is feasible and represents a potentially useful contribution to low-flow investigations. Work on the Wear with respect to Kielder transfers and the development of an Ecologically Acceptable Flow regime using PHABSIM is ongoing.

ACKNOWLEDGEMENTS

The research was funded by Northumbria and Yorkshire National Rivers Authority. Views expressed are those of the authors and not necessarily NRA. Rachel Merrix of NRA provided flow data for gauging stations on the Wear.

REFERENCES

Archer, D. and Gibbins, C.N. (1995). Hydrological and species specific responses to water transfers into the River Wear, North-east England. In: Petts, G.E. (Ed.). *Man's Influence on Freshwater Ecosystems and Water Use*. Proceedings of the Boulder Symposium., July, 1995, IAHS Publ. no. 230, Wallingford. pp. 207-218.

Armitage, P.D. and Ladle, M. (1991). Habitat preferences of target species for application in PHABSIM testing. In: Bullock, A., Gustard, A. and Grainger, E.S. *Instream Flow Requirements of Aquatic Ecology in Two British Rivers*. Institute of Hydrology, Wallingford. pp. 87-142.

Bovee, K.D. (1982). *A guide to stream habitat analysis using the Instream Flow Incremental Methodology*. US Fish and Wildlife Service, Instream Flow Information Paper No 12, FWS/OBS 82/26.

Carling, P. (1995). Implications of sediment transport for instream flow modelling of aquatic habitat. In: Harper, D.M. and Ferguson. A.J.D. (Eds.). *The Ecological Basis for River Management*. Wiley and Sons. Chichester. pp. 16-31

Gustard, A., Cole, G., Marshall, D.C.W. and Bayliss, A.C. (1987). *A Study of Compensation Flows in the UK.* Institute of Hydrology, Report no. 99, Wallingford.

Gibbins, C.N. (1996). *The impact of water transfers on the macroinvertebrate fauna of the River Wear.* Unpublished PhD Thesis, University of Northumbria, Newcastle upon Tyne, UK.

Gibbins, C.N., Soulsby, C. and Merrix, R.F. (1994). The impact of inter-basin water transfers on invertebrate communities of the River Wear. In: Kirkby, C. and White, W.R. (Eds.) *Integrated River Basin Development.* Wiley and Sons, Chichester. pp. 165-177.

Hammond, C. O. (1985). *The Dragonflies of Great Britain and Ireland.* Harley Books, Essex.

Heggenes, J. (1994). Physical habitat selection and behaviour by brown trout *(Salmo trutta)* and oung atlantic salmon *(S. salar)* in spatially and temporally heterogeneous streams: implications for hydraulic modelling. In: *Proceedings of the First International Symposium on Habitat Hydraulics,* Trondheim, Norway, August 1994 pp. 12-31.

Hill, M.O. (1979a). *TWINSPAN - a FORTRAN program for arranging multivariate data in an ordered two-way table by classification of the individuals and their attributes.* Section of Ecology and Systematics. Cornell University, Itheca, New York.

Hill, M.O. (1979b) *DECORANA - a FORTRAN program for detrended correspondence analysis and reciprocal averaging.* Section of Ecology and Systematics. Cornell University, Itheca, New York.

Johnson, I.W., Elliott, C.R.N., Gustard, A. and Clausen, B. (1993). *River Allen Instream Flow Requirements.* Report to the National Rivers Authority, Wessex Region. Institute of Hydrology, Wallingford.

Johnson, I.W., Elliott, C.R.N. and Gustard, A. (1994). *Data requirements and collection procedures for application of the Instream Flow Incremental Methodology in the UK.* Institute of Hydrology, Wallingford.

Mathur, D., Bason, U.M., Purdy, C. and Silver, C.A. (1985). A critique of Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences,* 42: pp 242-268.

Newman, A.T. (1995). *Hydro-ecology: Newsletter of the PHABSIM user forum.* May 1995. National Rivers Authority South Western Region,

NRA (1994b). *River Wear Catchment Management Plan Consultation Report.* NRA Northumbria and Yorkshire Region, Newcastle upon Tyne.

Petts, G.E. and Maddock, I. (1994). Linking hydrology and ecology. In: Calow, P. and Petts, G.E. (Eds.) *The Rivers Handbook.* Wiley and Sons, Chichester. pp. 289-308

Petts, G.E., Maddock, I., Bickerton, M. and Ferguson, A.J.D. (1995). Linking hydrology and ecology: the scientific basis for river management. In: Harper, D.M. and Ferguson. A.J.D. (Eds.). *The Ecological Basis for River Management.* Wiley and Sons. Chichester. pp. 1-16

Rutt, G.P., Weatherley, N.S. and Ormerod, S.J. (1990). Relationships between the physicochemistry and macroinvertebrates of upland British streams: the development of modelling and indicator systems for predicting fauna and detecting acidity. *Freshwater Biology,* **24**: pp. 463-480.

Scott, D. and Shirvell, C.S. (1987). A critique of Instream Flow Incremental Methodology and observations on flow determination in New Zealand. In: Craig, J.F. and Kemper, J.B. (Eds.). *Regulated Streams: Advances in Ecology.* Plenum Press, New York. pp. 27-43

Soulsby, C., Gibbins, C.N. and Jeffries, M.J. (1994). The impact of inter-basin water transfers on instream hydraulics and invertebrate communities in the River Wear. In: *Proceedings of the First International Symposium on Habitat Hydraulics,* Trondheim, Norway, August 1994. pp 399-411.

Weatherly, N.S. and Ormerod, S.J. (1987). The impact of acidification on macroinvertebrate assemblages in Welsh streams: towards an empirical model. *Environmental Pollution,* **46**: pp. 223-240.

Alternate modeling approaches

Nouvelles approches de modélisation

The Future of Habitat Modeling

Thomas B. Hardy
Institute for Natural Systems Engineering
Utah Water Research Laboratory
Utah State University, Logan, Utah 84322-4110
United States of America
hardy@aaron.cee.usu.edu

ABSTRACT

This paper examines emerging trends in applied instream flow assessment methods within the context of an ecologically based assessment framework in light of the challenges imposed by the spatial and temporal domains of aquatic ecosystems. I will attempt to highlight what I consider the more promising technologies, modeling, and analysis approaches that represent workable tools to meet the needs of practical applied instream flow assessments. To this end, I will touch on measurement techniques and technologies used to characterize the spatial domain of river systems, analysis tools for characterization of the hydrodynamic elements of rivers in both the spatial and temporal domains, and finally tools and approaches which integrate the biological elements at the individual, population and community levels. Much of my view of the future of habitat modeling remains an abstraction, in that integration of all the pieces has yet to be accomplished, field validation remains to be demonstrated, availability of an integrated analysis framework (i.e. computer software system) is not yet available, and a clear framework for selection and application of specific tools has not been developed. However, in presenting this particular view of the future, I hope to stimulate a broader collaborative effort between biologists, engineers, and resource managers that continues to move the state-of-the-art forward. This effort should not consider the plurality of methods or analytical procedures as competing approaches, but rather as representing a tool rich environment upon which researchers and practitioners can draw to provide scientifically based quantifications in support of management decisions which must protect and enhance our aquatic ecosystems.

KEY-WORDS: Habitat Modeling/Multi-disciplinary Assessments/Instream Flows/Future Trends/N-Dimensional Hydraulics/Bioenergetics Modeling/Remote Sensing/.

INTRODUCTION

From the perspective of an individual aquatic organism (e.g., macro-invertebrate, fish), river ecosystems create a temporally and spatially variable physical, chemical, and biological template within which an individual can exist if it possesses the proper suite of physiological, behavioral, and life history traits (Poff and Ward, 1990; Orth, 1986). This can lead to the view that organisms have a multi-dimensional niche of environmental conditions (e.g., envelopes of velocity, temperature, salinity) and resources (e.g., food, space) within which a viable population can be sustained (May and MacArthur, 1972; Pianka, 1974; Colwell and Futuyma, 1971). The short or long term success of individuals and hence populations or community dynamics can be limited by single and/or combinations of the physical, chemical, and biological components of the river ecosystem which have deterministic, periodic, and stochastic characteristics across both the temporal and spatial domains. This dynamic interplay between factors and responses at the individual, population, and community levels ultimately determines the complexity and therefore to some extent the strategy(s) of assessing impacts or status of resource states. The challenge, is to develop a practical level of understanding of these complex relationships between organisms, populations, communities, and the dynamic interplay over both the spatial and temporal scales of the environment. Once a sufficient understanding of these processes and linkages exist, we can begin to refine or develop tools/methods which can be applied within a rational framework to predict the effects of management actions (Chovanec *et al.*, 1994; Muhar *et al.*, 1995; Beanlands and Duinker, 1984; Orth, 1986).

Efforts directed at the evaluation of instream flow requirements to protect aquatic resources has already resulted in the development and application of a large number of methodologies over the past several decades and continues at an elevated rate today. Excellent reviews of many of the techniques developed and applied within the United States (and elsewhere) can be found in Reiser *et al.* (1989), CDM (1986), EPRI (1986), and Gore (1989). A broad area of research within the "discipline" of instream flow assessments is focused on modification or extension of existing methodologies, while other efforts are being directed at development and application of new tools. This is driven to some extent by the current ecosystem management objectives of resource agencies and a growing consensus among both researchers and practitioners that the disciplinary basis upon which the fundamental science and analytical procedures are developed, validated and applied in instream flow assessments needs a broader ecological perspective (Orth, 1995; Stanford, 1994). This has lead to research focused on the development and application of tools and assessment frameworks aimed at a more quantitative characterization of the factors controlling fisheries resources rather than continued application of tools for evaluation of a single target species from the limited perspective of physical habitat. This includes research on trophic level dynamics, process oriented delineation of flow induced changes in the physical and biological components of the aquatic environment, and in the development of broader based ecological frameworks for the evaluation of impact assessments or restoration efforts in aquatic ecosystems (e.g. Johnson and Law, 1995; Johnson *et al.*, 1995; Hearne *et al.*, 1994; Capra *et al.*, 1995; Leclerc *et al.*, 1995; Addley, 1993; Nehring and Anderson, 1993; Muhar *et al.*, 1995).

The objective of this paper is to highlight key historical research efforts as well as the emerging tools that permit the delineation and modeling of key physical and biological elements of aquatic ecosystems. The research and associated modeling tools discussed in this paper are considered to be "pre-adapted" in that they should allow the integration of results at the individual, population and community levels in habitat modeling efforts in order to assess the spatial and temporal aspects of instream flow requirements and/or impact assessments in river systems. In addition, I have focused on tools and modeling methods which should permit some latitude in terms of whether one approaches the problem of instream flow assessments from a process oriented framework or from a broader scale regional perspective. It is clear that more than one view or approach will be necessary given the nature and issues of instream flow questions facing practitioners.

AN ABBREVIATED REVIEW OF HABITAT MODELING APPROACHES

Many applied instream flow assessment methods that have directly attempted to incorporate the biological requirements of species relied on information obtained at the individual organism level as a basis to model the quality and quantity of habitat (Bovee, 1995). The most widely publicized methodology employing this approach is the Instream Flow Incremental Methodology (IFIM) and in particular one of it's major components the Physical Habitat Simulation System (PHABSIM). This approach primarily focused on the delineation and evaluation of individual "preferences" for depth, velocity, and substrate/cover for a handful of target species and life stages. This type of approach inherently requires that the physical or spatial domain of the river system be measured and/or modeled so that the biological criteria can be used to evaluate the relationship between flow and habitat (e.g. Bovee, 1982; Nestler *et al.*, 1993; Orth, 1995). The incorporation of the temporal variability of habitat was primarily approached from the integration of hydrologic time series and habitat versus discharge relationships to generate habitat time series (Bovee, 1995). This type of approach is not necessarily deficient as long as the target species and life stages have well defined habitat requirements which are known or can be measured and when the physical characteristics of the hydraulic regime in the river can be measured and simulated in a competent manner and only physical habitat is limiting. Although the efficacy of this basic approach for use in applied instream flow assessments has received criticisms (e.g. Orth and Maughan, 1982; Mathur *et al.*, 1985; Shirvell, 1986; Scott and Shirvell, 1987), its utility in a number of applications has clearly been demonstrated (e.g. Nehring and Anderson, 1993; Bovee *et al.*, 1994; Jager *et al.*, 1993; Jowett, 1992). Some of the more interesting recent research aimed at extending this basic methodology has concentrated on the integration of 2-dimensional hydraulics (Leclerc *et al.*, 1995), extensions of habitat time series to incorporate time dependant thresholds (Capra *et al.*, 1995), incorporation of hydraulic and habitat impacts due to seasonal changes in aquatic macrophytes (Hearne *et al.*, 1994), and in the development, field validation, and testing of the transferability of habitat suitability curves (Thomas and Bovee, 1993; Waite and Barnhart, 1992; Kwak *et al.*, 1992; Bozek and Rahel, 1992; Huber and Rahel, 1989; Beecher *et al.*, 1993; Beecher, 1995).

Other pertinent research within the broader arena of instream flows has focused on the delineation of key life history characteristics in terms ontogenetic shifts in habitat use under natural and induced flow variability (Heland *et al.*, 1995; Bardonnet and Gaudin, 1990; Bardonnet *et al.*, 1993; Crisp and Hurley, 1991), the relationship between flow and macro-invertebrate community dynamics (Lancaster and Hildrew, 1993; Gore, 1989; Jowett *et al.*, 1991; Weisberg *et al.*, 1990), and the importance of trophic level dependancies between macro-invertebrates and fish (Filber and Hawkins, 1995; Bevelhimer, 1996; Weisberg and Burton, 1993; Easton and Orth, 1992; Roell and Orth, 1994). Efforts employing mechanistic individual based bioenergetics, physical habitat based approaches utilizing population models, and multi-variate statistical approaches have also produced encouraging results (Addley, 1993; Jager *et al.*, 1993; Bovee *et al.*, 1994; Hill and Grossman, 1993; Jowett, 1992). This has included results based on linking community level distribution and abundance with spatially explicit delineations of the habitat mosaic at the meso-scale (Aadland, 1993; Dibble and Killgore, 1994; Bain, 1995; Jowett, 1992). A broader view of the river corridor as an integrated ecosystem has also provided excellent research on methods and frameworks for delineating the process driven linkages between flow, sediment transport, channel structure, and the riparian community (Hill *et al.*, 1991; Nillson *et al.*, 1991; Rabeni and Jacobson, 1993; Stromberg and Patten, 1991; Stromberg, 1993). Although not covered in this paper in order to save space, other efforts critical to instream flow assessments have included work on the development and integration of water quality and temperature modeling with existing physical habitat modeling programs.

This overview has purposefully been neither exhaustive nor as broad as the literature permits, and is intended to highlight some of what I have considered to represent key elements of the physical, and biological processes and concomitant analytical approaches that are amendable to incorporation into a systematic integrated framework for applied instream flow assessments. It is clear, that more process oriented methods that can be demonstrated to represent critical linkages between these processes and responses at the individual, population and community levels can only improve our existing science. This includes application of techniques that represent more of a community approach based on broader spatial scales. However, this implies that the current focused research within hydrology, hydraulics, and biology needs greater collaborative efforts in order to identify and then quantify the importance of and linkages between the physical, chemical and biological elements of the aquatic ecosystem in light of the need for practical, cost effective instream flow assessments.

THE FUTURE OF HABITAT MODELING

Even a casual reading of the referenced work and the more extended literature available indicates that the direction(s) of the future for habitat modeling will to a large extent be dependant on the perspective one takes at the onset of the journey. That is, can practical instream flow assessment methods be developed from a consideration of community function at the meso-scale habitat level, a broader scale approach using basin wide multi-variate regression models, or an incremental bottom-up approach based on empirical and/or analytical modeling of the component physical, chemical, and biological processes. Since this is a multiple choice question, I would submit that "all of the above" is the correct response and will be dependant on the nature of the specific instream flow question(s) (i.e. regional versus site specific). Of course, other more insidious constraints such as time, money and politics are likely to play a role. In order to provide a rational framework for presentation of my view on the future of habitat modeling, I have opted to loosely use the paradigm of the Instream Flow Incremental Methodology (IFIM) as a basis for discussion. This has the advantage of using a widely familiar framework which employs an incremental approach to problem solving represented by a series of discrete but inter-related steps. I have chosen to focus on three areas broadly represented by the processes of data collection, hydraulic analyses, and habitat/species modeling. Within each of these three broad conceptual elements, I have elucidated specific approaches and modeling techniques, that I feel represent viable near term solutions to the development and application of "habitat modeling" within the framework of applied instream flow assessments. I have purposefully left out other elements of the IFIM such as study design and negotiation in order to focus more on the "tools" rather than on the process or framework.

Data Collection

At the core of this (or perhaps any) approach is the assumption that the aquatic environment can be delineated into meaningful habitat mosaics (or types) which preserve their inherent variation, or lack thereof, with changes in discharge. It also implies that one can relate the importance of these habitat mosaics and their flow dependant characteristics to the biological needs of the individual, population or community. Clearly, the delineation of the spatial domain across scales that may range from less than a meter when concerned with macro-invertebrates to tens of meters or many kilometers when considering fish poses a significant challenge. This is especially true in light of the desire to improve the linkages between field data and application of 2- and 3-dimensional hydraulic models or in the potential use of landscape ecology metrics to describe and relate the importance of the spatial characteristics of the habitat mosaic as discussed below. In this section, I will explore the use of what I have roughly considered "remote sensing" technologies and use of other technological advancements that are rapidly approaching data acquisition capabilities to measure across these widely divergent spatial scales in an integrated fashion.

Delineation of habitat within instream flow assessments has classically relied upon linear habitat mapping using simple habitat descriptors such as run, pool, riffle, glide, etc. (Bovee, 1982), to more complex habitat descriptors dependant of the physical forming processes for each habitat type (Rossgen, 1984; Hankin and Reeves, 1988; Hawkins *et al.*, 1993; Kershner and Snider, 1992). Some techniques have relied on the development and application of more quantitative metrics at the meso- and macro-scales based upon a combination of physical attributes such as slope, channel stability, sinuosity, channel type, hydraulic properties, and habitat formation process which have been shown to be correlated with observed distribution and/or abundance of fish species (e.g. Rossgen, 1984; Hankin, 1984; Hankin and Reeves, 1988; Bain, 1995). Recent work has also attempted to link meso-scale habitat features with reproducible hydraulic characteristics at the micro-habitat level (Danehy, 1994; Jowett, 1993). The approach discussed here, starts at the reach level characterization of the meso- or macro-scale habitat types from a surface perspective and continues with the delineation of the subsurface characteristics of the hydraulic attributes at the micro-habitat level.

Delineation of the Surface Spatial Domain

One of the more likely techniques for use in the delineation of the type and spatial extant of meso-scale habitat features is the application of remote sensing technologies. Key factors to any remote sensing application should include cost, time, and ability to acquire data at appropriate spatial resolutions. Satellite imagery with existing platforms may be well suited where spatial resolutions of between 10 and 30 meters are appropriate such as in large rivers but is not considered practical for most "normal" instream flow studies. Aerial photography and aerial multispectral videography however, can achieve resolutions on the order of 0.5 to 10 meters and are more appropriate for the smaller spatial domains of many rivers and streams. One advantage of aerial photography is its high spatial resolution compared to satellite or videography systems, however it lacks the inherent advantage of digital multispectral attributes as well as taking prolonged periods for receipt of the imagery for use in near real time field applications. Recent advances in digital cameras and camcorder technologies however, may overcome these inherent disadvantages of classical aerial photography. Use of aerial photography for habitat mapping and other integrated applications is well known and therefore will not be discussed further.

Recent advancements in the technology of videography systems and its emerging use in natural resource applications however, deserves some treatment. Use of multispectral videography for the delineation of the spatial extant and type of aquatic habitats in river systems has rapidly increased over the past few years. Recent work, has included the delineation of meso-scale hydraulic features in both turbid and clear water systems (Anderson *et al.*, 1993; Panja and Hardy, 1995; Panja *et al.*, 1993; Hardy *et al.*, 1994;), the empirical determination of changes in meso-scale habitat features as a function of discharge interpreted in light of fish community use within specific habitat types (Panja and Hardy, 1995; Hardy *et al.*, 1994; Gilver *et al.*, 1995; Snider *et al.*, 1993), the prediction of species distributions (Hardy and Shoemaker, 1995), and mapping of sediment distributions important for spawning salmonids (Crowther *et al.*, 1995; Gilver *et al.*, 1995).

Multispectral videography is capable of providing near real time digital imagery at a variety of spatial scales ranging from 0.5 to 10 meters per pixel ground resolution over extensive river reaches of several hundred kilometers. The number of spectral bands can range from 1 to 20 depending on the system and needs of the particular application. This technology also has inherent advantages based on its digital format which permits classification of habitat types using image processing techniques and is directly compatible with existing GIS software systems. Most existing systems are also linked directly to GPS to permit geo-referencing of the imagery to a standardized map base.

As an illustration, Figure 1 provides an example of a typical 3-band false color composite from a multispectral videography image of a river using filter sensitivities in the green, red, and near-infrared bands of the spectrum. The corresponding unsupervised classification for fisheries habitat is also provided in Figure 1. It is apparent that the computer based classification is quite functional in the representation of the spatial distribution of meso-scale features which can be augmented by ground based delineations. As noted, this type of image classification permits the empirical delineation of habitat types over ranges of discharge which can be linked to fisheries collection data by habitat type.

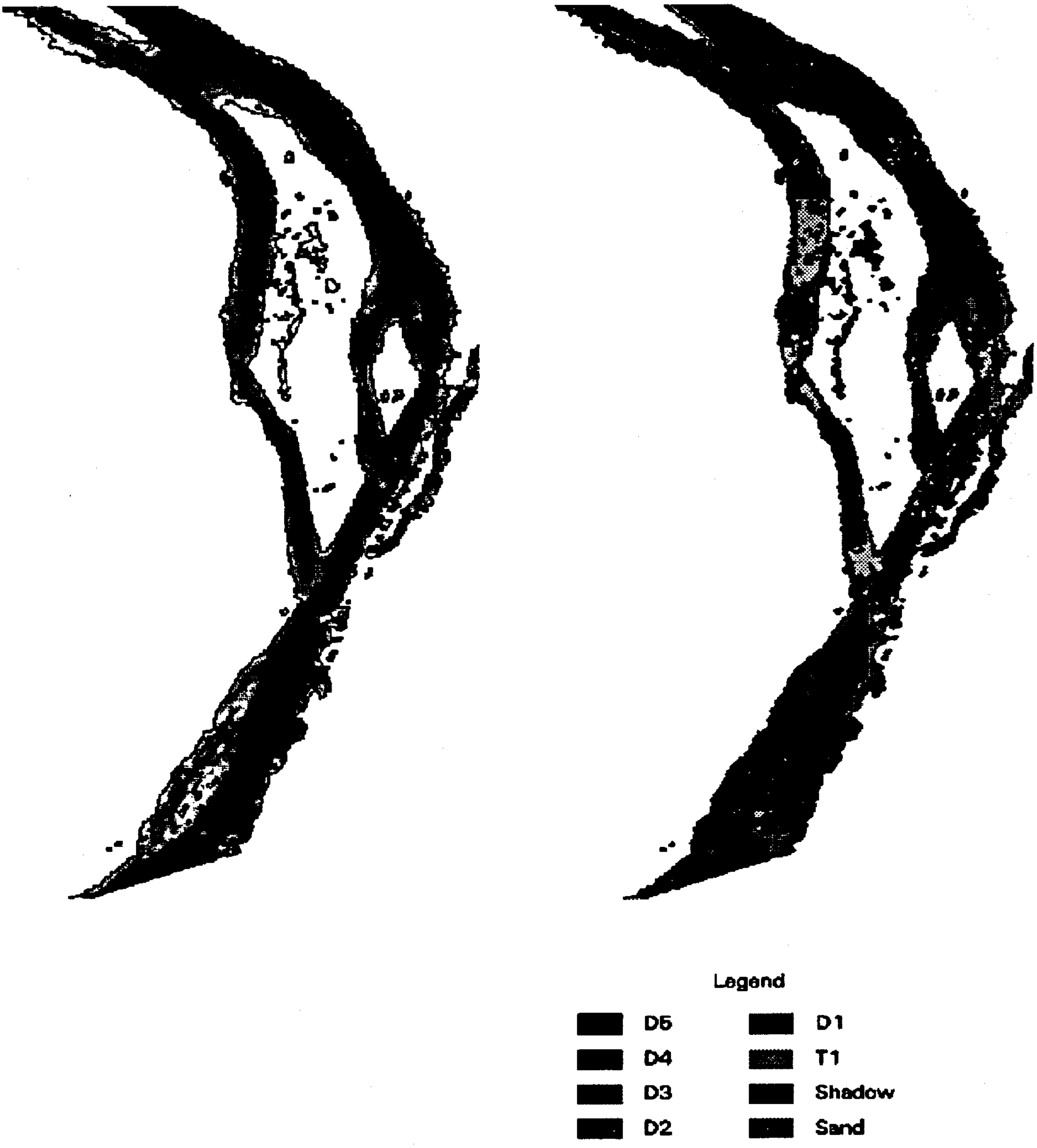


Figure 1. Example of a 3-band false color composite image and corresponding computer based classification of fisheries habitat using multi-spectral videography.

The use of GPS during field sampling in conjunction with printed imagery has also been shown to be of value for field crews in the location of sampling areas for ground based measurements. Another important feature of the use of this type of digital imagery is for long term change detection in support of monitoring programs since temporal data sets can be directly rectified as time dependant overlays using GIS. This can permit both the quantification of changes in surface area of specific habitat types but also in tracking these changes over the longitudinal spatial domain. An added benefit of this type of imagery is the ability to quantitatively assess riparian species composition and distribution along the river corridor using the same imagery (Redd *et al.*, 1993). Furthermore, since this type of imagery is compatible with most commercially available GIS software systems, it can also be utilized to derive a variety of spatial based physical and biotic metrics commonly used in landscape ecology such as fractal dimension, dominance, contagion, habitat diversity, relative evenness, and edge effect. The opportunities for use of these types of analytical metrics derived from the delineation of the spatial domain over relatively large reaches of a river system which preserves the finer scale spatial heterogeneity of meso-habitat characteristics is obvious and has great potential to examine the linkages between functional responses of individuals, populations, or communities with the spatial heterogeneity of the habitat mosaic (e.g. Bain, 1995; Aadland, 1993).

Delineation of the Subsurface Spatial Domain

Another rapidly developing field of "remote sensing" with application potential for instream flow assessments in river systems is the integration of differentially corrected GPS and hydro-acoustic arrays for obtaining channel morphometries, apparent roughness or substrate characteristics, aquatic vegetation distribution and densities, and development of finite element grids for n-dimensional hydraulic and habitat modeling (Gubala *et al.*, 1995; Tetra Tech, Inc., 1995; see Bartsch et al.; Tarbet and Hardy; and Ludlow and Hardy, these proceedings). Real time mapping of the subsurface spatial domain employing these systems are capable of achieving x-y resolutions of 0.15 cm and elevation data of between 15-30 cm given 60 second wait times. Even under typical field applications in medium to large river systems data acquisition can still achieve 1-3 meter spatial accuracy while acquiring data over extensive longitudinal spatial domains covering several kilometers or more per day. The reader is referred to companion articles in these proceedings cited above for examples of this type of data in order to conserve space. Data quality objectives, time, and cost basically dictate the spatial accuracy thresholds rather than a specific limitation of the technology. Often, the application of these technologies at high discharge rates is an advantage over more classical field measurement techniques and the subsequent data can be used in simulating hydraulic characteristics at lower discharges of interest. Another important consideration is that data obtained from these types of systems can also be integrated with either classical measurement techniques using total stations, laser levels and multispectral videography imagery for use in both hydraulic modeling and GIS applications. The integration these type of data using GPS allows for the rectification of imagery and subsurface profiling data to a standardized map base in GIS systems or for navigation by field crews to specific locations for additional ground based measurements such as fisheries collections (e.g. Hardy and Shoemaker, 1995).

Subsurface profiling in shallow water conditions where hydro-acoustic sampling may be limited (i.e. 1 meter or less) has been demonstrated with the use of pressure transducers coupled to velocity probes, total station reflectors, and differentially corrected GPS. These efforts have been demonstrated to provide a valuable tool for applications in streams having restrictive spatial domains and in direct integration with spatial sampling employing hydro-acoustics in large river domains (Harby, personal communication 1996; Hardy, unpublished field data 1996). This type of technology can provide a rapid method for the characterization of the 3-dimensional river geometry while permitting the collection of the micro-habitat velocity profiles and results in data amendable to n-dimensional hydraulic modeling and calculation of spatial metrics as noted above. The use of under water video has also been demonstrated as a valuable tool in the quantification of the particle size distribution of the surface sediments in deep water habitats (IPC, 1995; Hardy, unpublished field data) and have been collected with the use of GPS for direct linkage with the hydro-acoustic profiling technologies. Use of this technique employs a standardized measurement scale which permits the characterization of the particle size distribution while viewing the video imagery on the computer screen.

In essence, the ability to characterize the surface and sub-surface spatial domains over a variety of spatial resolutions in streams and rivers using existing technology has been clearly demonstrated and should permit the development and evaluation of tools or modeling approaches which can better define the linkages between physical and biological processes and responses at the individual, population or community levels.

N-dimensional Hydraulic Simulations

As illustrated above, recent advances in the ability to acquire extensive spatial data on the geometries of rivers has the potential to greatly improve the application of existing one-dimensional and more advanced 2- and 3-dimensional hydraulic simulations. I would suggest that use of existing one-dimensional hydraulic simulation models versus 2- or 3-dimensional models is not a question of one approach representing a superior method, rather that we have a richer set of tools from which to choose. It is also evident, that in context, one still is required to make a conscience and justifiable selection of the appropriate tool(s) to meet a given study objective(s). I believe that more extensive research on defining the tradeoff between data requirements versus obtaining some prescribed level of error in the velocity simulations using 1-, 2-, or 3-dimensional hydraulic models needs more critical assessment. However, my general view is that moving from the existing one-dimensional hydraulic simulations to use of 2-, or 3-dimensional models is intuitively better and allows a more rational linkage of the hydraulic simulations with the 3-dimensional spatial representation of the channel characteristics over micro- to meso-habitat scales. The reader is strongly encouraged to examine Bovee (these proceedings), for a particularly poignant view of 2-dimensional hydraulic modeling within the context of practical instream flow assessment frameworks given his experience in the development of PHABSIM. Given the current trend of research to evaluate 2-dimensional hydraulic simulations as evidenced by articles in these proceedings, some of its anticipated benefits warrant further discussion.

One potential advantage of using two-dimensional hydraulic simulation models is that they do not necessarily require the measurement of velocities throughout the entire spatial domain. These class of models generally require that the boundary conditions be specified at one or both ends of a simulated reach which requires either an estimate of the discharge an initial velocity distribution across the channel and the relationship between stage and discharge (i.e. one-dimensional hydraulics!). The advantages of not requiring extensive velocity measurements is likely to be realized only as long as an accurate delineation of the variation in spatial roughness can be obtained along with an accurate delineation of the channel geometry. This has obvious advantages over existing techniques employing one-dimensional hydraulic simulations where extensive velocity collections and calibrations are required. Our research using 2-dimensional hydraulic models indicate that the accuracy of simulated velocities over the spatial domain is highly dependant on the accuracy of the delineated stream geometry and characteristics of the finite element grids used in the model (e.g. Tarbet and Hardy, these proceedings). As noted in the previous section, the ability to acquire spatially explicit data over large spatial domains is technologically feasible, and on-going research is addressing the issues of computational stability and accuracy in spatial velocity simulations given data density dependant representations of stream geometry and finite element grid characteristics. The trade-off between the amount of data collected or required will undoubtedly be a function of the data quality objectives of a particular application. One of the more important areas of research that must be addressed before the use of 2-dimensional hydraulic modeling can hope to receive more widespread acceptance within existing instream flow programs is the validation of model accuracy, data collection requirements, model calibration guidelines, and practitioner training programs. Setting these issues aside, both the finite element grid representation of the channel and resulting velocity simulations which inherently incorporate the complex flow patterns arising from complex channel geometries can be used for integration of either classical habitat modeling approaches such as in PHABSIM (e.g. Leclerc *et al.*, 1995), mechanistic based bioenergetics approaches (e.g. Ludlow and Hardy, these proceedings), or to compute a variety of spatial and biological metrics which take advantage of the hydraulic properties over the spatial domain as a function of discharge (e.g. Bovee, these proceedings). Although the utilization of these "improved" modeling capabilities holds promise to better define and represent linkages between the hydro-dynamic properties of rivers and resulting responses at the individual, population and community levels, it remains yet to be conclusively demonstrated or validated over a wide array of systems.

Existing use of 3-dimensional hydraulic models in applied instream flow assessments is not so much limited by it's data requirements as by the access to raw computing power by practitioners. Practical field level applications will continue to require access to super computers and high end work stations until desk top work stations or PC based architectures improve substantially. If the current rate of hardware development as evidenced over the past decade continues, then this may be a reality within the next decade. Demonstration of the utility of 3-dimensional hydraulics in the development and application of new modeling approaches however is strongly encouraged, since "pre-established" modeling capabilities and use in assessment frameworks will facilitate their integration into mainstream applications when the hardware becomes more widely available.

Another area of research in hydraulic modeling is the need to provide more quantitative assessments of the predicted changes in substrate characteristics and meso-scale habitats under proposed flow regimes. The ability to make accurate forecasts of the changes in both the sediment size distribution and resulting changes in meso-scale habitat characteristics has important implications in assessing instream flows and habitat restoration efforts. It is one thing to be able to make accurate delineations of the existing river geometry and spatial predictions of velocity distributions and yet another to predict how the river geometry and substrate characteristics are likely to change under an altered flow regime. These changes in both sediment size distributions and larger scale changes in meso-habitat characteristics have profound implications on invertebrate community distribution and abundance as well as in the quantity and quality of fisheries habitat. This aspect of modeling habitat in applied instream flow assessments is in this authors opinion, poorly represented by documented methods, validated analytical tools, and a framework for their integration in instream flow applications. With the emphasis currently being given to shear stress and invertebrate community dynamics (Davis, 1986; Statzner and Higler, 1986; Statzner, 1988) the ability to incorporate this aspect of hydraulic modeling is critical.

Species and Habitat Modeling

The biological importance of accurate delineations of the type, location and extent of habitat features has been recognized by a number of investigators in river systems (Rinne 1991; Freeman and Grossman 1993; Fausch and White 1981; Southall and Hubert 1984; Addley 1993; Stanford 1994; Lobb and Orth 1991; Rabeni and Jacobson 1993; Bovee *et al.*, 1994). In fact, the delineation of the contact zone between two (or more) habitat types as a spatial metric or attribute may constitute a unique habitat type in itself (Ranney et al. 1981, Noss 1983, Lovejoy et al. 1986, Harris 1988, Yahner 1988, Malcolm 1994). Clearly, spatially explicit models can be utilized for the development and testing of habitat metrics derived from the landscape ecology literature that are now receiving more attention in the aquatic community. These metrics describe various aspects of spatial heterogeneity that may be linked to functional responses of various species or community dynamics such as species dispersal, colonization potential, foraging efficiency, predator avoidance and species replacement (Li and Reynolds 1994; Bain, 1995; Aadland, 1993; Jowett, 1992). Some of the more commonly applied metrics that may be of value include fractal dimension (Burrough, 1986; O'Neill *et al.*, 1988), dominance (O'Neill *et al.*, 1988), contagion (O'Neill *et al.*, 1988; Turner *et al.*, 1989; Li and Reynolds, 1994), habitat diversity and relative evenness (Shannon and Weaver, 1962; Pielou, 1969; Romme, 1982), and edge effect (Malcolm, 1994).

Many of these important structural or spatial metrics related to habitat types can be characterized by the amount of contact or edge between habitat types or the length of interface or contact zone between two different habitat types and should be accessible using the integration of the surface and sub-surface spatial delineation techniques and hydraulic modeling tools described in this paper. Use of the extensive spatial data sets obtainable with the technologies described above, permits the application of a variety of statistical techniques that have already been shown to be extremely useful in defining gradients of environmental variables, such as depth and velocity, heterogeneity of habitat patch size, persistence of habitat types over ranges of discharges and other biologically significant response axes (e.g. Jowett, 1992; Bain, 1995; Changeux, 1995; Barnard *et al.*, 1995; Milner *et al.*, 1995). Analyses using these indices can often expose previously unknown relationships between the habitat mosaic, community structure, and its function. A key element in these approaches will be the acquisition of species, population, and/or community level use of spatially delineated habitat types which as described above, should be facilitated by the integration of remote sensing of the surface domain in conjunction with the sub-surface profiling of the river and output from 1-, 2- or 3-dimensional hydraulic simulations. These techniques have some level of appeal in that they explicitly attempt to integrate the physical and biological attributes of systems in terms of the systems potential to support an individual target species, population or community. The ability to link these types of analyses with predictive tools in terms of hydraulics, channel changes due to changes is flow and/or sediment dynamics and the corresponding changes in spatial composition and distribution of meso- and macro-scale habitat mosaics however, has yet to be widely demonstrated. Linkages of this nature using process driven models for key physical and biological elements should be considered as a high priority for on-going research.

Another emerging direction in habitat modeling involves the development and application of individual based bioenergetics models (e.g. Jager *et al.*, 1993; Addley, 1993; Rose and Cowen, 1993; Van Winkle *et al.*, 1993; Hill and Grossman, 1993). One of the more appealing aspects of these efforts is the potential to bypass the site specific requirements of suitability index curves that currently plagues applications using PHABSIM. These models are also appealing in light of the explicit linkages to and integration of trophic level dynamics. The integration of ontogenetic shifts in habitat use, which includes diel shifts between spatial locations within the channel or in flow dependant changes in habitat use will represent important advances for these modeling efforts. This requires an accurate characterization of the spatial domain in conjunction with improved hydraulic simulation of the velocity characteristics. These efforts will also require better tools for the modeling of the seasonal changes in stream temperature and in particular the associated changes in macro-invertebrate densities. This may also include the incorporation of flow dependant changes in macro-invertebrate and larval drift in altered flow environments as well as utilization of decision support systems that would allow the incorporation of linear dominance hierarchies of multi-species assembledges.

Analysis of the Temporal Domain

The most commonly applied temporal domain analysis to date has been the simple use of habitat time series based on the integration of the hydrologic time series and habitat versus discharge relationships of particular species and life stages. A number of metrics based on percent exceedance ranges have been proposed or some variation on this theme (Bovee, 1995) however, the linkage of these metrics and limiting conditions or long term viability of the fish community is generally lacking. Recently, concepts employed in the analysis of hydrologic time series such as run sum lengths have been proposed (Capra *et al.*, 1995) as well as development of more integrative aspects of the limiting population component based on habitat availability over time (Bovee, 1988; Nehring and Anderson, 1993; Bovee et al., 1994). More integrated trophic level considerations have also been proposed and attempt to directly integrate the time dependant interactions between habitat availability and trophic level interactions (see Orth, 1995). To this author, this area of research focus is critically lacking in terms of delineating the appropriate linkages between magnitude, duration, and timing of specific habitat quantity and quality and expected responses at the individual, population or community level.

SUMMARY

Available technologies such as multispectral videography and aerial photography are likely to provide access to surface level spatial delineations of aquatic habitat types that are relevant at the individual, population or community level for use in deriving empirical relationships between habitat availability and discharge. Application of these technologies for similar instream flow questions are currently well represented in the literature. These techniques also provide access to the use of GIS for resource characterizations, impact assessments and long term monitoring data. These techniques also provide an excellent opportunity to acquire data over extensive spatial domains that are likely to provide valuable linkages to land scape ecology metrics which may help understand the dynamics in the habitat mosaic and individual, population and community responses to flow alterations in river systems. Technologies also exist for the characterization of the sub-surface topography of rivers at highly accurate spatial scales using GPS linked hydro-acoustic arrays which can be integrated with depth transducers and velocity meters using more conventional surveying techniques. This technology can be employed to acquire data over many kilometers of river per day and provide a direct linkage to both the computation of key physical and biotic metrics of ecological importance as well as providing the data for use in one-, 2- or 3-dimensional hydraulic simulations. Another key element of these two technologies is that data for use in support of ground based fisheries and habitat measurements can be obtained in near real time. Both of these technologies are also compatible with GIS systems commonly utilized today in resource planning. The practical application of two- and 3-dimensional hydraulic simulations is also an important contribution to existing instream flow assessment methods. Potential advantages are directly related to the spatial data generated from the use of hydro-acoustic array sampling of the bed topography and the lack for extensive collection of velocity profiles within the river. Current trends in the use of trophic level based modeling using bioenergetics is also and important direction in the future of habitat modeling. The incorporation of more quantitative assessments of the temporal domain in habitat time series is also important and future research on the critical evaluation of habitat duration and thresholds for sustaining fisheries populations is likely to be critical in applied instream flow assessments. Clearly, the most important aspect of the future for habitat modeling is the greater collaborative efforts between hydrology, hydraulics and biology in more holistic research frameworks aimed at a better quantification of the important process driven characterization of the stream environment in light of demonstrable flow dependant responses of macro-invertebrates and individual fish, populations, and communities.

LITERATURE CITED

Aadland, L.P. (1993). Stream habitat types: their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* **13**:790-806.

Addley, R.C. (1993). A mechanistic approach to modeling habitat needs of drift-feeding salmonids. Master Thesis, Department of Civil and Environmental Engineering, Utah State University, Logan, Utah.

Anderson, P.C., T.B. Hardy, and C.M.U. Neale. (1993). Application of Multispectral videography for the delineation of riverine depths and mesoscale hydraulic features. In: *Proceedings of the 14th Biennial Workshop on Color Photography and Videography for Resource Monitoring*. Utah State University, Logan, Utah, May 25-28, 1993, Editor C.M.U. Neale.

Bain, M.B. (1995). Habitat at the local scale: multivariate patterns for stream fishes. *Bull. Fr. Peche Piscic* **337/338/339**:165-177.

Bardonnet, A., and P. Gaudin. (1990). Diel pattern of downstream post-emergence displacement of grayling (*Thymallus thymallus* L., 1758). *Journal of Fish Biology* 37:623-627.

Bardonnet, A., P. Gaudin, and J. Thorpe. (1993). Diel rhythm of emergence and of first displacement downstream in trout (*Salmo trutta* L.), Atlantic Salmon (*Salmo salar* L.), and grayling (*Thymallus thymallus* L.). *Journal of Fish Biology* 43:755-762.

Barnard, S., R.J. Wyatt, and N.J. Milner. (1995). The development of habitat models for stream slamonids, and their application to fisheries management. *Bull. Fr. Peche Piscic* 337/338/339:375-385.

Beanlands, G.E. and P.N. Duinker. (1984). An ecological framework for environmental impact assessment. *Journal of Environmental Management* 18:267-277.

Beecher, H.A. (1995). Comparison of preference curves and habitat utilization curves based on simulated habitat conditions. *Rivers* 5(2):109-120.

Beecher, H.A., T.H. Johnson, and J.P. Carleton. (1993). Predicting microdistributions of steelhead (*Oncorhyncus mykiss*) parr from depth and velocity preference criteria: Test of an assumption of the instream flow incremental methodology. *Canadian Journal of Fisheries and Aquatic Sciences* 50:2380-2387.

Bevelhimer, M.S. (1996). Relative importance of temperature, food, and physical structure to habitat choice by smallmouth bass in laboratory experiments. *Transactions of the American Fisheries Society* 125(2):274-283.

Bovee, K.D. (1982). A guide to stream analysis using the Instream Flow Incremental Methodology. Instream Flow Information Paper: No. 12, FWS/OBS-82/26, U.S. FWS, Fort Collins, CO.

Bovee, K.D. (1988). Use of the instream flow incremental methodology to evaluate influences of microhabitat variability on trout populations in four Colorado streams. *Annu. Conf., West. Assoc. Fish and Wildl. Agencies* 6:227-257.

Bovee, K.D., T.J. Newcomb, and T.G. Coon. (1994). Relations between habitat variability and population dynamics of bass in the Huron River, Michigan. National Biological Survey Biological Report 21. 63 pp.

Bovee, K.D. (1995). A comprehensive overview of the Instream Flow Incremental Methodology. National Biological Service, Fort Collins, CO., 322pp.

Bozek, M.A., and F.J. Rahel. (1992). Generality of microhabitat suitability models for young Colorado River cutthroat trout (*Oncorhynchus clarki pleuriticus*) across sites and among years in Wyoming Streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49:552-564.

Burrough, P.A. (1986). Principles of geographical information systems for land resources assessment. Clarendon, Oxford, U.K.

CDM (1986). Minimum Instream Flow Study. Camp, Dresser, and McKee. Final Report. Commonwealth of Virginia State Water Control Board.

Capra, H., P. Breil and Y. Souchon. (1995). A new tool to interpret magnitude and duration of fish habitat variations. *Regulated Rivers: Research and Management* 10:281-289.

Changeux, T. (1995). Fish community structure at the scale of a large European basin: longitudinal organization, slope influence and regional tendencies. *Bull. Fr. Peche Piscic* 337/338/339:63-74.

Chovanec, V.A. H. Heger, V. Koller-Kreimel, O. Moog, T. Spindler and H. Waidbacher. (1994). Requirements for the assessment of the ecological capability of streams - A basis for discussion. *Osterreichische Wasser-Und Abfallwirtschaft* 46(Heft 11/12):257-264.

Colwell, R.K., and D.J. Futuyma. (1971). On the measurement of niche breadth and overlap. *Ecology.* 52(4):567-576.

Crisp, D.T. and M.A. Hurley. (1991). Stream channel experiments on downstream movement of recently emerged trout, *Salmo trutta* L., and salmon, *S. Salar* L.-II. Effects of constant and changing velocities and of day and night upon dispersal rate. *Journal of Fish Biology* 39:363-370.

Crowther, B.E., T.B. Hardy, and C.M.U. Neale. (1995). Application of Multispectral Video for the Classification of

Fisheries Habitat Components in Salmon River, Idaho. In: *Proceedings of the 15th Workshop on Color Photography and Videography for Resource Assessment.* Indiana State University, Indiana. May 2-3, 1995.

Danehy, R.J. (1994). Geomorphic, hydrologic, and hydraulic determinants of fish and macroinvertebrates in a small watershed. Ph.D. Dissertation, SUNY College of Environmental Science and Forestry, Syracuse, NY, 158 pp.

Davis, J.A. (1986). Boundary layers, flow microenvironments and stream benthos. In: *Limonology in Australia* (Eds P. De Deckker and W.D. Williams), pp. 293-312, CISRO, Dr. W. Junk Publishers, Dortecht.

Dibble, E.D. and K.J. Killgore. (1994). A habitat-based approach for studying fish-plant interactions. In: *Proceedings of the 28th Annual Meeting, Aquatic Plant Control Program*, 15-18 November 1994, Baltimore, Maryland.

Easton, R.S. and D.J. Orth. (1992). Ontogenetic diet shifts of age-0 smallmouth bass (*Micropterus dolomieu Lacepede*) in the New River, West Virginia, USA. *Ecology of Freshwater Fish* **1**:86-98.

EPRI (1986). Instream Flow Methodologies. Final Report, EA-4819 Research Project 2194-2. Electric Power Research Institute, Palo Alto, California.

Fausch, K.D., and R.J. White. (1981). Competition between brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) for positions in a Michigan stream. *Canadian Journal of Fisheries and Aquatic Sciences* **38**:1220-1227.

Filbert, R.B., and C.P. Hawkins. (1995). Variation in condition of rainbow trout in relation to food, temperature, and individual length in the Green River, Utah. *Transactions of the American Fisheries Society* **124**(6):824-835.

Freeman, M. C. and G. D. Grossman. (1993). Effects of habitat availability on dispersion of a stream cyprinid. *Environmental Biology of Fishes* **37**:121-130.

Gilver, D.J., T. M. Waters and A.M. Milner. (1995). Image analysis of aerial photography to quantify changes in channel morphology and instream habitat following placer mining in interior Alaska. *Freshwater Biology* **(34)**:000-000.

Gore, J.A. (1989). Models for predicting benthic macroinvertebrate habitat suitability under regulated flows. In: J.A. Gore, J.A. and G.E. Petts, (eds.), *Alternatives in regulated river management*, Pp. 254-265, CRC Press Inc., Boca Raton, Florida.

Gubala, C.P., T.B. Hardy, J. Eilers and N. Roundy. (1995). SONAR-GPS mapping of aquatic ecosystems and GPS assisted survey of small rivers, streams and creeks. Trimble Users Conference and Exposition 1995 Proceedings. Pp: 189-196. Trimble Navigation Ltd.

Hankin, D. G. (1984). "Multistage Sampling Designs in Fisheries Research: Applications in Small Streams." *Canadian Journal of Fisheries and Aquatic Sciences* **41**: 1575-1591.

Hankin, D. G. and G. H. Reeves (1988). "Estimating Total Fish Abundance and Total Habitat Area in Small Streams Based on Visual Estimation Methods." *Canadian Journal of Fisheries and Aquatic Sciences* **45**: 834-844.

Hardy, T.B., P.C. Anderson, C.M.U. Neale, D.K. Stevens. (1994). Application of Multispectral Videography for the Delineation of Riverine Depths and Mesoscale Hydraulic Features. Paper and Presentation at the American Water Resources Association, Effects on Human-Induced Changes on Hydrologic Systems. June 26-29, 1994. Jackson Hole, Wyoming.

Hardy, T.B. and J.A. Shoemaker. (1995). Use of Multispectral Videography for Spatial Extrapolation of Fisheries Habitat Use in the Comal River. In: *Proceedings of the 15th Workshop on Color Photography and Videography for Resource Assessment.* Indiana State University, Indiana. May 2-3, 1995.

Harris, L.D. (1988). Edge effects and conservation of biotic diversity. *Conservation Biology* **2**:330-332.

Hawkins, C.P., J.L. Kershner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A. McCullough, C.K. Overton, G.H. Reevs, R.J. Steedham and M.K. Young. (1993). A hierarchical approach to classifying stream habitat features. *Fisheries* **18**(6):3-12.

Hearne, J., I. Johnshom, and P. Armitage. (1994). Determination of ecologically acceptable flows in rivers with seasonal changes in the density of macrophyte. *Regulated Rivers: Research and Management* **9**:177-184.

Heland, M., P. Gaudin, and A. Bardonnet. (1995). First behavioral adjustments in relation to habitat use after emergence in running water salmonids. *Bull. Fr. Peche Piscic* **337/338/339**:191-197.

Hill, M.R., W.S. Platts, and R.L. Beschta. (1991). Ecological and geomorphological concepts for instream and out-of-channel flow requirements. *Rivers* **2**:198-210.

Hill, J., and G.D. Grossman. (1993). An energetic model of microhabitat use for rainbow trout and rosyside dace. *Ecology* 74:685-698.

Hubert, W.A., and F.J. Rahel. (1989). Relations of physical habitat to abundance of four nongame fishes in high-plains streams: A test of habitat suitability index models. *North American Journal of Fisheries Management* **9**:332-340.

Jager, H.I., D.L. Deangelis, M.J. Sale, W. Van Winkle, D.D. Schmoyer, M.J. Sabo, D.J. Orth, and J.A. Lukas. (1993). An individual-based model for smallmouth bass reproduction and young-of-the-year dynamics in streams. *Rivers* **4**:91-113.

Johnson, I.W., F.M., and F.M. Law. (1995). Computer models for quantifying the hydro-ecology of British Rivers. *Journal of CIWEM* 9:290-297.

Johnson, I.W., C.R.N. Elliot and A. Gustard. (1995). Modeling the effect of groundwater abstraction on salmonid habitat availability in the River Allen, Dorset, England. *Regulated Rivers: Research and Management* **10**:229-238.

Jowett, I.G., J. Richardson, J.F. Biggs, C.W. Hickey, and J.M. Quinn. (1991). Microhabitat preferences of benthic invertebrates and the development of generalized *Deleatidium* spp. Habitat suitability curves, applied to four New Zealand streams. *New Zealand Journal of Marine and Freshwater Research* **25**:187-199.

Jowett, I.G. (1992). Models of the abundance of large brown trout in New Zealand rivers. *North American Journal of Fisheries Management* **12**:417-432.

Jowett, I.G. (1993). A method for objectively identifying pool, run, and riffle habitats from physical measurements. *New Zealand Journal of Marine and Freshwater Research* **27**:241-248.

Kwak, T.J., M.J. Wiley, L.L. Osborne, and R.W. Larimore. (1992). Application of diel feeding chronology to habitat suitability analysis of warmwater stream fishes. *Canadian Journal of Fisheries and Aquatic Sciences* **49**:1417-1430.

Kershner, J. L. and W. M. Snider. (1992). "Importance of a Habitat-level Classification System to Design Instream Flow Studies." *River Conservation and Management* **12**: 179-193.

Lancaster, J. And A.G. Hildrew. (1993). Flow refugia and the microdistribution of lotic macroinvertebrates. *Journal of the North American Benthological Society* **12**(4):385-393.

Leclerc, M., A. Boudreault, J.A. Bechara, and G. Corfa. (1995). Two-dimensional hydrodynamic modeling: a neglected tool in the instream flow incremental methodology. *Transactions of the American Fisheries Society* **124**(5):645-662.

Lobb, M. D. III and D. J. Orth. (1991). Habitat use by an assemblage of fish in a large warmwater stream. *Transactions of the American Fisheries Society* **120**:65-78.

Li, H. and J.F. Reynolds. (1994). A simulation experiment to quantify spatial heterogeneity in categorical maps. *Ecology* **75**(8):2446-2455.

Lovejoy, T.E., R.O. Bierregaard, Jr., A.B. Rylands, J.R. Malcolm, C.E. Quintela, L.H. Harper, K.S. Brown, Jr., A.H. Powell, G.V.N. Powell, H.O.R. Shubart, and M.B. Hays. (1986). Edge and other effects of isolation on Amazon forest fragments. Pages 257-285 In: M.E. Soulé, editor. Conservation Biology: the science of scarcity and diversity. Sinauer Associates, Sunderland, MA.

Malcolm, J.R. (1994). Edge effects in central Amazonian forest fragments. *Ecology* **75**(8):2438-2445.

Mathur, D., W.H. Bason, E.J. Purdy Jr., and C.A. Silver. (1985). A critique of the instream flow incremental methodology, *Canadian Journal of Fisheries and Aquatic Sciences* **42**:825-831.

May, R. M., and R. H. MacArthur. (1972). Niche overlap as a function of environmental variability. *Proceedings of the National Acadamey of Sciences, USA* **69**(5):1109-1113.

Milner, N.J., R.J. Wyatt, S. Barnard, and M.D. Scott. (1995). Variance structuring in stream salmonid populations, effect of geographic scale and implications for habitat models. *Bull. Fr. Peche Piscic.* **337/338/339**:387-398.

Muhar S., S. Schmutz and M. Jungwirth. (1995). River restoration concepts - goals and perspectives. *Hydrobiologia* **303**:183-194.

Nehring, R.B. and R.M. Anderson. (1993). Determination of population-limiting critical salmonid habitats in Colorado streams using IFIM/PHABSIM. *Rivers* **4**:1-19.

Nestler, J. M., L. T. Schneider and D. Latka. (1993). RCHARC: a new method for physical habitat analysis, in Engineering Hydrology, a symposium sponsored by the Hydraulics Division/ASCE, July 25-30, San Francisco, CA, 294-299.

Nillson, C., A. Ekblad, M. Gardfjell, and B. Carlberg. (1991). Long-term effects of river regulation on river margin vegetation. *Journal of Applied Ecology* **28**:963-987.

Noss, R.F. (1983). A regional landscape approach to maintain diversity. *BioScience* **33**:700-706.

O'Neill, R.V., J.R. Krummel, R.H. Gardner, G. Sugihara, B. Jackson, D.L. DeAngelis, B.T. Milne, M.G. Turner, B. Zygmunt, S.W. Christensen, V.H. Dale, and R.L. Graham. (1988). Indices of landscape pattern. *Landscape Ecology* **1**(3):153-162.

Orth, D.J. (1986). Ecological considerations in the development and application of instream flow-habitat models. *Regulated Rivers: Research and Management* **1**:171-181.

Orth, D.J. (1995). Food web influences on fish population responses to instream flow. *Bull. Fr. Peche Piscic.* **337/338/339**:317-328.

Orth, D.J. and O.E. Maughan. (1982). Evaluation of the incremental methodology for recommending instream flows for fish. *Transactions of the American Fisheries Society* **111**:413-445.

Panja, K., T.B. Hardy and C.M.U. Neale. (1993). Comparison of Meso-scale hydraulic features at different discharges in a turbid river system using multispectral videography. In: *Proceedings of the 14th Workshop on Color Photography and Videography for Resource Monitoring.* Utah State University, Logan, UT. 25-29 May 1993.

Panja, K., and T.B. Hardy. (1995). Comparison of Multispectral Videography based Classification of Mesoscale Habitats and Ground Based Mapping under Turbid Riverine Conditions. In: *Proceedings of the 15th Workshop on Color Photography and Videography for Resource Assessment.* Indiana State University, Indiana. May 2-3, 1995.

Pianka, E.R. (1974). Niche overlap and diffuse competition. *Proceedings of the National Academy of Sciences, USA 71*(5):2141-2145.

Pielou, E.C. (1969). An introduction to mathematical ecology. Wiley-Interscience, New York. 286 pp.

Poff, N.L. and J.V. Ward. (1990). Physical habitat template of lotic systems: Recovery in the context of historical pattern of spatiotemporal heterogeneity. *Environmental Management 14*(5):629-645.

Rabeni, C. F. and R. B. Jacobson. (1993). The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Freshwater Biology 29*:211-220.

Ranney, J.W., M.C. Bruner, and J.B. Levenson. (1981). The importance of edge in the structure and dynamics of forest islands. Pages 67-95 In: R.L. Burgess and D.M. Sharpe, editors. Forest island dynamics in man-dominated landscapes. Springer-Verlag, New York.

Redd, T.H., C.M.U. Neale, and T.B. Hardy. (1993). Classification and delineation of riparian vegetation on two western river systems using airborne multispectral video imagery. In: *Proceedings of the 14th Biennial Workshop on Color Photography and Videography for Resource Monitoring.* Utah State University, Logan, Utah, May 25-28, 1993, Editor C.M.U. Neale.

Reiser, D.W., T.A. Wesche and C. Estes. (1989). Status of instream flow litigation and practices in North America. *Fisheries* **14**(2):22-29.

Rinne, J. N. (1991). Habitat use by spikedace, *Meda fulgida* (Pisces: Cyprinidae) in southwestern streams with reference to probable habitat competition by red shiner *Notropis lutrensis* (Pisces: Cyprinidae). *Southwestern Naturalist*

36:7-13.

Roell, M.J. and D.J. Orth. (1994). Trophic basis of production of stream-dwelling smallmouth bass, rock bass, and flathead catfish in relation to invertebrate bait harvest. *Transactions of the American Fisheries Society* 122:46-62.

Romme, W.H. (1982). Fire and landscape diversity in sub-alpine forests of Yellowstone National Park. *Ecological Monographs* 52:199-221.

Rose, K.A., and J.H. Cowan, Jr. (1993). Individual-based model of young-of-the-year striped bass population dynamics. I. Model description and baseline simulations. *Transactions of the American Fisheries Society* 122(3):415-438.

Rossgen, D.L. (1984). A stream classification system. In: *Proceedings of the Symposium on Riparian Ecosystems and their Management: Reconciling Conflicts Uses* (Ed. R. Hamre), pp. 91-95, USDA Forest Service, General Technical Report RM-120, Tucson, Arizona.

Scott, D., and C.S. Shirvell. (1987). A critique of the instream flow incremental methodology and observations on flow determinations in New Zealand. In: J.F. Craig and J.B. Kemper (eds.), *Regulated Streams*, Pp. 27-43, Plenum Publishing Corp.

Shannon, C.E., and W. Weaver. (1962). The mathematical theory of communication. University of Illinois Press, Urbana. 125 pp.

Shirvell, C.S. (1986). Pitfalls of physical habitat simulation in the Instream Flow Incremental Methodology. Canadian Fisheries and Aquatic Sciences Technical Report 1460. 68 pp.

Snider, M.A., J.W. Hayse, I. Hlohowskyj, K.E. LaGory, M.M. Greaney, J.A. Kuiperand, and R.A. Van Lonkhuyzen. (1993). Use of aerial videography to evaluate the effects of Flaming Gorge Dam operations on natural resources of the Green River. In: *Proceedings of the 14th Workshop on Color Photography and Videography for Resource Monitoring*. Utah State University, Logan, UT. 25-29 May 1993.

Southall, P. D. and W. A. Hubert. (1984). Habitat use by adult paddlefish in the upper Mississippi River. *Transactions of the American Fisheries Society* 113:125-131.

Stanford, J.A. (1994). Instream flows to assist the recovery of endangered fishes of the upper Colorado River basin. National Biological Survey Biological Report 24. 47 pp.

Statzner, B. (1988). Growth and reynolds number of lotic macroinvertebrates: a problem for adaptation of shape to drag, *Oikos* 51:84-87.

Statzner, B., and B. Higler. (1986). Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. *Freshwater Biology* 16:127-139.

Stromberg, J.C., and D.T. Patten. (1991). Instream flow requirements for cottonwoods at Bishop Creek, Inyo County, California. *Rivers* 2:1-11.

Stromberg, J.C. (1993). Instream flow models for mixed deciduous riparian vegetation within a semiarid region. *Regulated Rivers: Research and Management* 8:225-235.

Tetra Tech, Inc. (1995). Willamette River Basin Water Quality Study: Phase II. Ecological monitoring component: Habitat assessment report. Final Report to Oregon Department of Environmental Quality, Portland, Oregon 97204. DEQ Contract No. 97-094.

Thomas, J.A., and K.D. Bovee. (1993). Application and testing of a procedure to evaluate transferability of habitat suitability criteria. *Regulated Rivers: Research and Management* 8:285-294.

Turner, M.G., R. Costanza, and F.H. Sklar. (1989). Methods to evaluate the performance of spatial simulation models. *Ecological Modeling* 48:1-18.

Waite, I.R., and R.A. Barnhart. (1992). Habitat criteria for rearing steelhead: A comparison of site-specific and standard curves for use in the instream flow incremental methodology. *North American Journal of Fisheries Management* 12:40-46.

Weisberg, S.B., A.J. Janicki, J. Gerritsen, and H.T. Wilson. (1990). Enhancement of benthic macroinvertebrates by minimum flow from a hydroelectric dam. *Regulated Rivers: Research and Management* 5:265-277.

Weisberg, S.B. and W.H. Burton. (1993). Enhancement of fish feeding and growth following an increase in minimum flow below the Comowingo Dam. *North American Journal of Fisheries Management* 13:103-109.

Van Winkle, W., K.A. Rovse, and R.C. Chambers. (1993). Individual-based approach to fish population dynamics: An overview. *Transactions of the American Fisheries Society* 122(3):397-403.

Yahner, R.H. (1988). Changes in wildlife communities near edges. *Conservation Biology* 2:333-339.

BIOENERGETICS MODEL FOR DRIFT-FEEDING BROWN TROUT

John W. Hayes
Cawthron Institute
Private Bag 2
Nelson, New Zealand

ABSTRACT

A bioenergetic model and EXCEL computer programme has been developed for drift-feeding brown trout. The model can be used to predict trout growth and the energetic profitability of feeding from invertebrate drift rates and water temperature. Drift rates for user-defined time periods are stratified into 3 mm size-classes with accompanying mean dry weight and energy value. The model estimates energy intake and expenditure within a semicircular foraging area the size of which is proportional to fish size. The foraging area includes optional velocity differentials which trout utilise for energetic advantage when feeding. Energy expenditure on swimming and on body processes are estimated from published metabolic equations, mainly from the studies of J.M. Elliott. The bioenergetics model has potential as an environmental impact tool, or as an aid for hypothesis generation in studies of fish growth in rivers by identifying key factors potentially driving growth. It is intended to be used to assess feeding and growth consequences of changes in water temperature and clarity and invertebrate communities which can occur with human impacts on flow, water quality and land-use.

KEY-WORDS: brown trout / bioenergetics / modelling / growth / feeding / invertebrate drift / temperature /water clarity

INTRODUCTION

Changes in land-use, and installation of hydraulic structures, can alter flow, water temperature, chemical and substrate composition and clarity of rivers, and the taxonomic and size composition of invertebrate communities (Ward, 1976; Armitage 1984; Scott 1994). These changes have important consequences for the growth rate of fish and the size structure of fish populations.

Bioenergetics models coupled with foraging models are proving useful for studying fish growth and food consumption and for predicting the effects of changes in environmental and food variables, although most studies have been on lakes (Hansen *et al.*, 1993). Bioenergetics models treat fish growth as a joint function of food consumption and temperature (Hewitt and Johnson 1992; Elliott 1976a). They are essentially mass-balance equations in which the energy in consumed food is partitioned into its various fates: metabolism, waste products and growth (Ricker 1971). Foraging models can estimate food consumption; relating fish foraging behaviour, prey density and size, and, for visual feeders, prey capture dependence on water clarity (Vinyard and O'Brien 1976; Breck 1993).

Trout have flexible feeding behaviours but feeding on invertebrate drift predominates in moderate to swiftly flowing rivers (Bachman 1984; Hayes and Jowett 1994). Invertebrate drift rate is a more direct measure of food available to trout than is benthic invertebrate biomass and is easier to incorporate into a foraging model. Drift-feeding trout maximise net energy intake while minimising energy expenditure by feeding from sheltered positions into faster water where drift rates are higher (Fausch 1984) and by selecting large prey from the drift (Bannon and Ringler 1986). Energetics models have been used to predict selection of drift-feeding positions for brown trout and other salmonids (Fausch 1984; Hughes and Dill 1990; Hughes 1992).

Bachman (1982) proposed a conceptual growth model for drift-feeding salmonids based on optimal foraging theory predicting that growth and size should be limited by the finite energy availability set by the mean drift density for any river. The metabolic rate of salmonids increases exponentially with size, water velocity, and temperature (Elliott 1976a; Brett and Glass 1973), whereas the mean supply of drift increases only linearly with water velocity (Elliott 1967; Fausch 1984). Consequently, growth rate must decline with age and, for a given river, the energy available from drift will limit the maximum size, unless the trout migrate to more food rich habitats or are able to switch to alternative prey with greater energy return. With increasing body size, metabolic, and reproductive demands with age trout must eat larger and larger prey if they are to continue to grow. If large invertebrates are not well represented in the drift then large trout should be energetically disadvantaged, resulting in a population with a truncated size distribution.

The primary objective of this study was to develop a bioenergetics model for the growth of drift-feeding brown trout which incorporated drift size structure and trout foraging across velocity shears. The model was required to estimate growth rate over the lifetime of an average individual and energetic profitability of feeding in relation to trout size. It was developed with the intention that it would aid the understanding of invertebrate - trout growth relationships in rivers and be useful for predicting the effects on growth of changes in drift rate and size composition and on changes in water temperature and clarity.

MODEL DESCRIPTION

Conceptual Framework of the Drift-feeding Bioenergetics Model

The bioenergetics model for drift-feeding brown trout is composed of a foraging model, an energetics model comprising a suite of energetics equations, and an annual growth model (Fig. 1). The foraging model uses invertebrate drift density and size composition (3 mm size-classes), water temperature and clarity for a series of user-defined time periods (weeks - months) to calculate rate of food intake, and apportion time spent foraging versus maintaining position at the focal point. The output from the foraging model, and water temperature, is used by the energetics model to calculate daily energy intake, balance the energy budget, and calculate growth over each time period. The annual growth model then integrates growth over the time periods for each year accounting for reproduction costs after maturity.

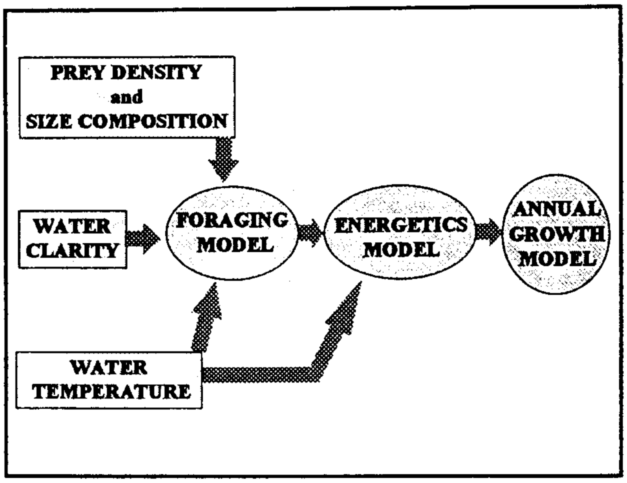


Figure 1: Components of the bioenergetics model for drift-feeding trout

Drift-feeding Foraging Model

Geometry of the Foraging Model

The foraging area of a drift-feeding trout can be approximated by a semicircle around the trout which is positioned close to the bottom of the river at its focal point behind a velocity refuge - such as a rock (Fig. 2). The semicircle should be visualised as a two dimensional cross-section perpendicular to the current, looking upstream. The radius of the semicircle, the foraging radius, is represented by FR. As drifting invertebrates are carried into this semicircle by the current, the trout rises upward and/or sideways to intercept the prey and then returns to its focal point. Lateral and vertical velocity differentials (velocity shears) are optional in the model. To accommodate trout feeding across a lateral velocity differential the semicircle is divided into three strips with the prey capture areas in each strip represented by $CA_{1\text{-}3}$. The mean column water velocities (at 0.4 x depth) of each of the capture areas are represented by $mVel_{1\text{-}3}$, and the focal point velocity (at 0.1 x depth (Hayes and Jowett 1994)) is $FpVel$.

The mean velocity of the water column at the trout's position ($mVel_2$) is calculated using the following equation for the optimal swimming speed of a general salmonid (Stewart (1980).

(1) $$mVel_2 = 0.097 \bullet W^{0.13} \bullet e^{(0.0405T)}$$

where T is water temperature and W is weight (g). Predictions of this equation matched very well with mean column velocities for adult brown trout in New Zealand rivers (Hayes and Jowett 1994) when temperature was fixed at 13°C, the optimum temperature for growth of brown trout (Elliott 1994).

For the lateral velocity differential option, one of the adjacent mean column velocities assumes the same value as the mean column velocity at the trout's position (i.e., $mVel_2$) and the other adjacent velocity is calculated by applying a velocity differential to $mVel_2$ of 0.13 m/s/m operating over the distance 0.66FR (from Hayes and Jowett (1994)). For the vertical velocity differential, $FpVel$ is calculated by applying a velocity differential to $mVel_2$ of -0.6 m/s/m operating over the distance 0.4R - 0.1R where R is twice the fork length (in metres) (from Hayes and Jowett (1994)).

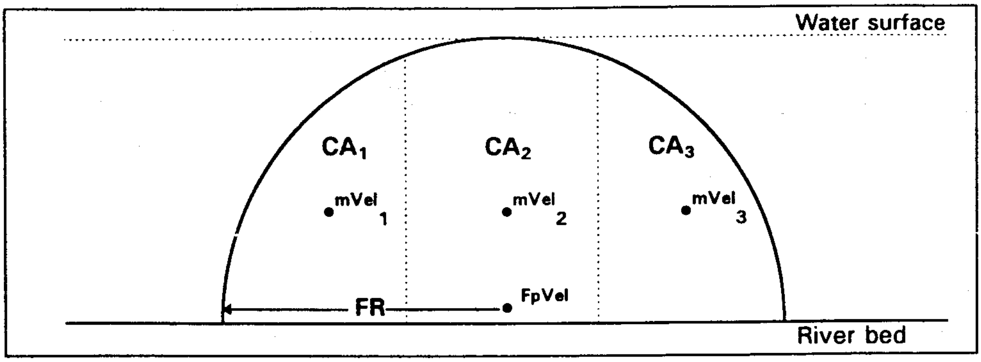


Figure 2: Cross-section of the foraging area, looking upstream, showing how the foraging space is divided to characterise velocities above and adjacent to the trout's focal point. CA_{1-3} = capture areas, $mVel_{1-3}$ = mean column velocities, $FpVel$ = focal point velocity, FR = foraging radius.

To calculate the foraging radius (*FR*), I used a conceptual model of the geometry of prey interception developed by Hughes and Dill (1990) for arctic grayling (Fig. 3). For a drift-feeding fish to capture a drifting prey the fish must see the prey and intercept it before the prey is swept downstream. This model assumes that the fish begins intercepting the prey as soon as it sees it and then travels at its maximum sustainable swimming speed (*VMAX*). The distance at which the fish first sees the prey is the reaction distance (*RD*) and the prey drifts at *V*, the water velocity in the foraging area. The model assumes the fish captures the prey before it crosses a line perpendicular to the focal point (*D* - *E*). The above assumptions mean that the foraging model will return minimum prey capture times and foraging costs.

Using the above model, Hughes and Dill demonstrated that the relationship between *FR* and *RD*, *VMAX*, and *V* can be derived as follows. When line segment $AC = FR$, the time taken for the fish to travel this distance (Forage time = *FT*) will be the same as the time taken by the prey to travel the line segment *BC* (prey time = *PT*), and the fish will catch the prey just as it crosses line *DE*. To travel *AC* relative to the stream-bed, the fish must swim *AB* relative to the water. $AB = RD$ and $BC = V{\cdot}TP$ and, since $PT = FT = RD/VMAX$, by substitution $BC = V{\cdot}RD/VMAX$. Therefore, using Pythagoras' theorem:

(2) $$FR = \sqrt{RD^2 - (V \bullet RD / VMAX)^2}$$ where:

(3) $$RD_i = \left(12 \bullet PL_i\left(1 - e^{(-0.2 \bullet FL)}\right)\right)/100$$ (in m) (from Hughes and Dill 1990)

(4) $$VMAX = \left(36.23 \bullet FL^{0.19}\right)/100$$ (in m.s^{-1}) (from Jones *et al.* 1974)

and where $V = mVel_2$, FL = fork length (cm), and PL = the length midpoint of prey size-class i (mm).

Using Equations 2 -4 the foraging radius can be calculated as a continuous function of fork length, which is compatible with modelling growth where predicted size at the end of a given time period becomes the input for calculating consumption and growth in the next time period. Note that the foraging radius is dependent on water velocity and, through the reaction distance, on prey size and fish size. Therefore, the foraging radius increases with both increasing prey size and fish size (because of the greater eye diameter and retinal acuity of larger fish), and with decreasing water velocity.

The foraging radius will also be influenced by water clarity because the reaction distance of visual foraging fish varies with light intensity and turbidity (Vinyard and O'Brien 1976). I have not yet included water clarity as a continuous function of reaction distance in the bioenergetics model but I plan to do so in the future. In the interim a coarse estimate of the influence of water clarity can be obtained by truncating the reaction distance to the water clarity of interest.

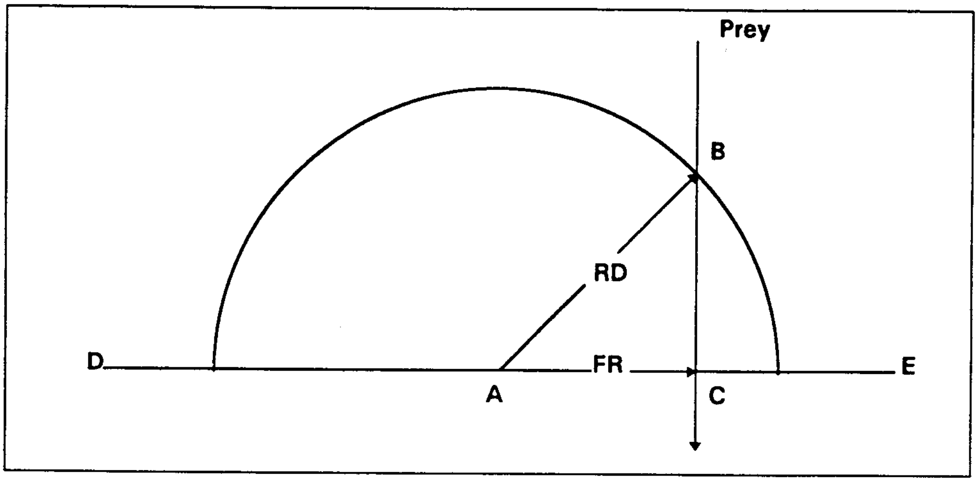


Figure 3: Geometry of prey interception, as seen from above. *A* is the fishes focal point, *AB* is its reaction distance to the approaching prey, and *AC* is the maximum foraging radius. The fish will see the prey at *B* and must intercept it before it crosses the line *DE*. (After Hughes and Dill, 1990)

Rate of food intake

The above foraging model is used to calculate rate of food intake, or consumption (C). Consumption is calculated for each of a series of 3 mm prey size-classes, providing they fall within the minimum and maximum prey sizes selected by trout.

Maximum prey size is limited by mouth gape and minimum prey size is set by foraging cost benefit relationships and gill raker spacing (Wankowski 1979; Bannon and Ringler 1986). Minimum and maximum prey size-classes were calculated as ratios of prey size to fish fork length determined by Wankowski (1979) and used for drift-feeding brown trout by Bannon and Ringler (1986) (minimum prey size (mm) = $0.115 \cdot FL$, maximum prey size = $1.05 \cdot FL \cdot 4.3$; (FL in cm) where the value 4.3 is a prey length:diameter ratio).

Rate of food intake is integrated over the eight prey size-classes in the three prey capture areas ($CA_{1\text{-}3}$.) using the following formula modified from Hughes and Dill (1990):

$$(5) \qquad C = \sum_{i=1}^{8} \sum_{j=1}^{3} CA_j \bullet mVel_j \bullet PC_i \bullet PW_i \bullet 3600 \qquad \text{(in g.s}^{-1}\text{)}.$$

where CA_j is the capture area for the prey size-class i , $mVel_j$ is the mean column velocity (in m.s^{-1}) in CA_j, PC_i is the concentration of prey size-class i passing within CA_j,, and PW_i is the dry weight of prey size-class i.

Apportioning foraging time

Foraging costs, calculated later in the energetics model, are split between total prey interception time and time spent maintaining station at the focal point. The proportion of time spent foraging (*FTsum*) is integrated over eight prey size-classes in the three prey capture areas using the following formula:

$$FTsum = \sum_{i=1}^{8} \sum_{j=1}^{3} CA_j \bullet mVel_j \bullet PC_i \bullet FT \tag{6}$$

where $FT = RD/VMAX$ (Equation 2 proof). The proportion of time spent at the focal point is 1-*FTsum*.

Energetics Model

The energy budget

The energetics model was based on the following energy balance equation (Ricker 1971) solved on a daily time step for the average individual :

$$\Delta B = C - F - U - (R_a + R_s + R_d) \tag{7}$$

where ΔB is the energy accumulated for growth and reproduction, C is consumption (expressed as daily energy intake), F and U are the energy losses in faeces and urine, R_a and R_s are the energy requirements for active and standard metabolism, and R_d is the energy required for digestion (including specific dynamic action).

The primary source for the components of the energy budget (other than C) is the series of studies by Elliott on energetics of brown trout (summarised in Elliott 1976a, 1994). Elliott's energetics equations were derived for 5 - 300 g brown trout in tanks. These equations are extrapolated to larger trout in my bioenergetics model. Extrapolation of bioenergetics equations outside the size-ranges of fish upon which they are developed can introduce errors but is commonly done in bioenergetics modelling (Ney 1993).

The units of the energetic equations are cal/day and cal/h. The components of the bioenergetic equation not obtainable from Elliott's' papers were obtained from other sources including studies on rainbow trout and salmonids in general. Elliott's energetic equations are multiple regression equations utilising power functions of either or both water temperature and weight of the form:

$$X = a \bullet W^{b_1} \bullet e^{b_2 T} \tag{8}$$

where X represents a component of the energy budget and a, b_1, and b_2 are temperature dependent constants.
An equation for R_d (including specific dynamic action) for brown trout could not be found. Instead, R_d was calculated as a constant 15% of C. This value falls between the range for R_d reported for salmonids (9.5 - 19.0%: Jobling 1981; Preide 1985).

Daily energy intake

Consumption, calculated in Equation 5 as the rate of food intake (in $g.s^{-1}$), is used to estimate daily energy intake. The size and number of meals per day and the satiation time per meal are calculated from the rate of food intake (C), applied to the foraging hours (daylight), and the rate of gastric evacuation using equations in Elliott (1972, 1975a,b). Daily consumption is the product of the size and number of meals per day, converted to energy intake ($cal.day^{-1}$) using a average prey energy value.

Foraging costs

Total satiation time (summed for all meals) is split by the proportion of time spent foraging (*FTsum*) and the proportion of time spent maintaining station (1 - *FTsum*). Foraging costs are then calculated from the following equation combining standard (from Elliott (1976a)) and active metabolism (from Stewart (1980)).

$$FC = (4.126 \bullet W^{0.734} \bullet e^{0.192 \bullet T} \bullet e^{2.34 \bullet V}) / 24 \quad (\text{in cal.h}^{-1}) \tag{9}$$

where $V = VMAX$ for the time spent foraging and $V = FpVel$ for the time spent maintaining station. Activity costs are set to zero when the fish is satiated and during the hours of darkness.

Annual Growth Model

The change in energy (ΔB) over each time period estimated by the energetics model is converted to weight using an estimate of energy density for brown trout. For trout greater than 810 g I used 1780 cal.g^{-1} (from Lien 1978) and for smaller trout I used the following equation from Elliott (1976b) for energy density.

$$ED = 751.9 \bullet e^{0.332 \bullet K} \bullet W^{0.072} \quad \text{where } K \text{ (condition factor)} = 100 \cdot W / FL^3 \tag{10}$$

Reproduction costs were obtained from a study of the energy budget of a Norwegian lake population of brown trout (Lien 1978). Lien found that the energy content of mature trout was reduced by an average of 46% (males 44%, females 48%) after spawning, including losses in gonads and body tissue. Body weight only reduced by 18%, with the difference between this and energy content made up by water balance.

The above estimates for energy and weight loss are used in the annual growth model to adjust growth for reproduction losses. Providing weight exceeds the mature weight (defined by the user) at the end of the current growth season, the model reduces ΔB by 46% and W by 18%, placing the hypothetical trout in energy deficit for the next growth season. This energy deficit must then be met before energy is available for growth. Annual growth is integrated from growth over each time period and repeated for 12 years.

MODEL DEMONSTRATION

Demonstration Data

The data used for the modelling demonstration below were collected from the Maruia River, South Island, New Zealand. Drift rates for 3 mm invertebrate size-classes, water temperatures, and foraging hours, are given for five time periods covering approximately a nine month growth season (Table 1 - 3). Mean dry weight for each prey size-class (mid point) was: 0.084 mg (1.5 mm), 0.5196 mg (4.5 mm), 1.437 mg (7.5 mm), 4.190 mg (10.5 mm), 7.556 mg (13.5 mm), 13.800 mg (16.5 mm), 16.960 (19.5 mm). To demonstrate the effect of altering the size structure of prey, drift rates in Table 3 have been modified from those in Table 2. Invertebrates greater than 13.5 mm have been omitted and drift rates for the 1.5 mm and 4.5 mm size-classes have been increased to keep total biomass for each time period the same as in Table 2.

For each modelling run, initial weight of trout was 1 g, both vertical and lateral velocity differentials were chosen, weight at maturity (for reproduction costs) was 1200 g, and prey energy content was 4400 cal. g^{-1} dry weight (approximating a range of drifting invertebrates (Cummins and Wuycheck 1971)). The bioenergetics model is programmed in EXCEL Visual Basic.

Table 1: Physical data for the five time periods used to model trout growth

Time period	No. days	Foraging hours	Temp. °C
1	57	12	9
2	58	15	13
3	28	17	16
4	63	16	15
5	63	13	10

Table 2: Invertebrate drift rates (No. m^3) for the five time periods used to model trout growth

Time period	Invertebrate size class (mid points)						
	1.5	4.5	7.5	10.5	13.5	16.5	19.5
1	1.455	0.603	0.259	0.208	0.025	0.020	0.005
2	1.179	1.202	0.368	0.109	0.046	0.029	0.005
3	3.933	1.281	0.363	0.028	0.003	0.011	0
4	3.691	1.530	0.631	0.094	0.007	0.005	0
5	1.769	0.313	0.117	0.043	0.0124	0.006	0.003

Table 3: Modified invertebrate drift rates (No. m^3) for the five time periods used to model trout growth

Time period	Invertebrate size class (mid points)						
	1.5	4.5	7.5	10.5	13.5	16.5	19.5
1	3.600	0.951	0.259	0.208	0.025	0	0
2	4.081	1.673	0.368	0.109	0.046	0	0
3	4.818	1.424	0.363	0.028	0.003	0	0
4	4.101	1.596	0.631	0.094	0.007	0	0
5	2.558	0.441	0.117	0.043	0.0124	0	0

Modelling scenarios

To demonstrate how the bioenergetics model can be used to investigate the impact of physical and ecological perturbations to rivers, three scenarios are modelled: 1/ the influence of changes to invertebrate size structure, 2/ the influence of water clarity, 3/ the influence of water temperature.

Both the drift size structure and water clarity have a strong influence on the predicted growth trajectory and on maximum size. Growth is depressed mainly in older fish, with growth rates during the first three years relatively unaffected (Fig. 4). This pattern arises because trout need to eat larger prey as they grow, and because both prey size and water clarity play a large role in limiting the reaction distance (and foraging area) of larger trout. Weight declines after about age six because drift rates of large invertebrates are insufficient to meet the combined demands of maintenance and reproduction.

Water temperature is a primary variable driving growth in the bioenergetics model so it is not surprising that changes in temperature can have a profound effect on growth (Fig. 5). A reduction in water temperature of 3°C over all time periods reduced predicted maximum size by 1.6 kg (Fig. 5 A, D). Growth is also sensitive to changes in the length of the growing season, reduced by one month in this example (Fig. 5 A, E). Changes in water temperature and length of the growing season influence the growth at all ages, unlike reductions in the size of prey and water-clarity which have most impact on larger fish.

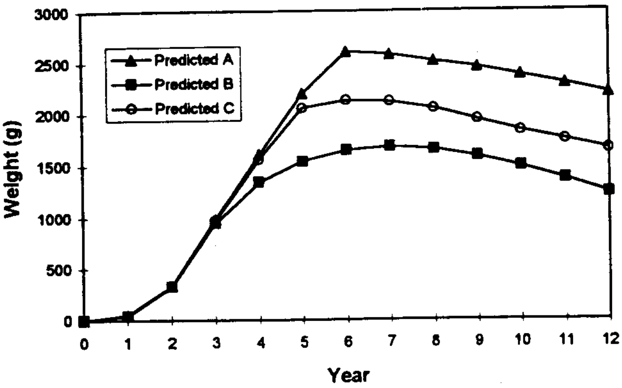


Figure 4: Predicted growth curves for trout feeding on 1.5 - 19.5 mm drifting prey in unlimited water clarity (A) and water clarity = 1.5 m (C) (Tables 1, 2), and on 1.5 - 13.5 mm prey in unlimited water clarity (B) (Tables 1, 3).

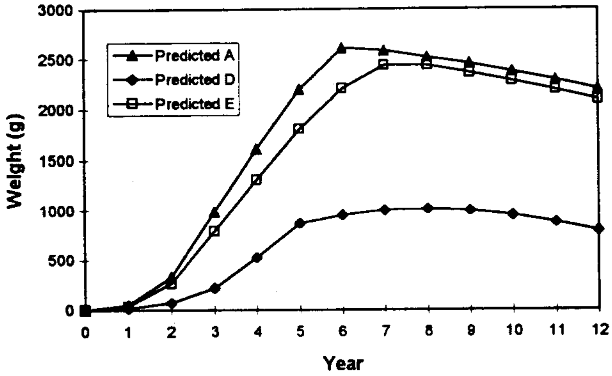


Figure 5: Predicted growth curves for trout feeding on 1.5 - 19.5 mm drifting prey in unlimited water clarity (A), with reduced water temperature (D), and with reduced growing period (E).

DISCUSSION

Bioenergetics models are attractive because they provide a way to link fish growth with underpinning physical variables and lower trophic levels. They allow functional responses of fish to be modelled, promising a better understanding of how and why growth might change under altered habitat and food conditions than is possible with correlative models.

I believe that the strength of the bioenergetics model for drift-feeding brown trout lies in its ability to integrate physical and ecological impacts over the life time of trout through the medium of growth, and that it allows relative comparisons of growth trajectories. It has potential as an environmental impact tool, or as an aid for hypothesis generation in studies of fish growth in rivers, being able to identify key factors potentially driving growth. In addition to predicting growth curves the bioenergetics model can also be used to investigate the energetic profitability of drift feeding for various sizes of trout. The energetic profitability of drift-feeding declines with decreasing mean size of invertebrates and with increasing size of trout (*cf.* Fig. 4).

However, there is still much work to be done before the model can be confidently used in a predictive sense. The model has not yet been tested. Corroboration is needed between measured and predicted growth, ideally in a range of rivers covering the extremes of food and temperature (Hansen *et* al. 1993; Filbert and Hawkins 1995). I am currently undertaking a study comparing predicted with measured growth on a New Zealand river. A sensitivity analysis is yet to be undertaken on the model parameters (Bartell *et* al. 1986; Hartman and Brandt 1993; Ney 1993).

Obtaining an accurate estimate of mean drift rate probably presents the biggest practical hurdle for using the model. Drift rates can be highly variable although a seasonal pattern often dominates the temporal variation (Statzner *et al.* 1984; Sagar and Glova 1992). Nevertheless, in modelling trout growth, drift size structure may dominate over drift rate at least in determining the shape of the growth curve. If this is true then for some applications a rough estimate of drift rate may be sufficient. If estimating accurate drift rates proves too costly then it would useful to consider relationships between drifting and benthic invertebrate density and size composition with a view to substituting benthic invertebrate sampling for drift sampling.

The drift foraging model employed in the bioenergetics model makes the assumption that all prey passing through the trout's foraging area are captured unless the drift rate exceeds handling capacity. This is probably unrealistic, especially for small trout which feed on small invertebrates with high drift rates, and could lead to overestimation of growth. Moreover, water clarity is not adequately modelled at present. A refinement of the model is planned which will incorporate capture efficiencies and functional relationships between these and water clarity and light intensity (*cf.* Ware 1972; Vinyard and O'Brien 1976; Ringler and Brodowski 1983).

It is unclear whether population density needs to be considered when applying the bioenergetics model. High population density has generally been considered to retard trout growth (Backiel and LeCren 1978) but more recently this view has been questioned (Preall and Ringler 1989; Elliott 1994).

The bioenergetics model is intended for drift-feeding brown trout. Although brown trout and other stream trout feed primarily on invertebrate drift (Bachman 1984) their diet can vary widely depending upon the habitat (Fausch 1991). Other foraging models could be constructed to include alternative feeding strategies (*cf.* Werner and Hall 1974; Breck 1993) and coupled with the other components of the present model.

Adjustment of growth for reproduction losses is treated fairly coarsely in the bioenergetics model. With each spawning the trout's energy content is reduced by a constant percentage. This takes no account of potential trade-offs between growth and reproduction and differences between populations (Wooton 1985). Unfortunately, studies estimating complete reproductive losses in salmonids are rare.

Given the assumptions and approximations underlying the bioenergetics model, and the lack of testing at this stage, there is clearly a long way to go before it can be used with confidence in predictive applications. This model therefore should be seen as a first approximation in a process of model testing and refinement.

ACKNOWLEDGMENTS

This study was funded by the Foundation for Research, Science and Technology, and by the New Zealand Fish and Game Council. I was assisted by J. Stark, Y. Stark, K. Shearer, K. Ponikla, A. Quarterman and S. Hayes.

REFERENCES

Armitage, J.D. (1984) Environmental changes induced by stream regulation and their effect on lotic macroinvertebrate communities. In: *Regulated Rivers*, A. Lilehammer and Saltveit, S.J. (Eds.), University press, Oslo, pp. 139-165.

Bachman, R.A. (1982) A growth model for drift-feeding salmonids: a selective pressure for migration. In: E.L. Bannon and Salo, O. (Eds.). *Proceedings of the Salmon and Trout Migratory Behaviour Symposium*. University of Washington, Seattle, Washington. pp. 128-135.

Bachman, R.A. (1984) Foraging behaviour of free-ranging wild and hatchery brown trout in a stream. *Transactions of the American Fisheries Society*. 113 pp. 1-32.

Backiel, T. and E.D. LeCren (1978) Some density relationships for fish population parameters. In: S. Gerking (Ed.). *Ecology of Freshwater Fish Production*. Blackwell Scientific Publications, Oxford. pp. 279-302.

Bannon, E. and N.H. Ringler (1986) Optimal prey size for stream resident brown trout (*Salmo trutta*): tests of predictive models. *Canadian Journal of Zoology*. 64 pp. 704-713.

Bartell, S.M., J.E. Breck, R.H. Gardner and A.L. Brenkert (1986) Individual parameter perturbation and error analysis of fish bioenergetic models. *Canadian Journal of Fisheries and Aquatic Sciences*. 43 pp. 160-168.

Breck, J.E. (1993) Foraging theory and piscivorous fish: are forage fish just big zooplankton? *Transactions of the American Fisheries Society*. 122 (3) pp. 902-911.

Brett, J.R. and N.R. Glass (1973) Metabolic rates and critical swimming speeds of sockeye salmon (*Oncorhynchus nerka*). *Journal of the Fisheries Research Board of Canada*. 22 pp. 1492-1501.

Cummins, K.W., J.C. Wuycheck (1971). Calorific equivalents for investigations in ecological energetics. *Mitt. Int. Verhandlungen der Internationale Vereinigung fur theoretische und angewandte Limnologie*. 18 pp. 1-158.

Elliott, J.M. (1967) Invertebrate drift in a Dartmore Stream. *Archiv für Hydrobiologie*. 63 pp. 202-237.

Elliott, J.M. (1972) Rates of gastric evacuation in brown trout, *Salmo trutta* L. *Freshwater Biology*. 2 pp. 1-18.

Elliott, J.M. (1975a) Weight of food required to satiate brown trout, *Salmo trutta*. *Freshwater Biology*. 5 pp. 51-64.

Elliott, J.M. (1975b) Number of meals in a day, maximum weight of food consumed in a day and maximum rate if feeding for brown trout, *Salmo trutta* L. *Freshwater Biology*. 5 pp. 287-303.

Elliott, J.M. (1976a) The energetics of feeding, metabolism and growth of brown trout (*Salmo trutta* L.) in relation to body weight, water temperature and ration size. *Journal of Animal Ecology*. 45 pp. 923-948.

Elliott, J.M. (1976b) Body composition of brown trout (*Salmo trutta* L.) in relation to temperature and ration size. *Journal of Animal Ecology*. 45 pp. 273-289.

Elliott, J.M. (1994) Quantitative ecology of the brown trout. Oxford University Press, Oxford.

Fausch, K.D. (1984) Profitable stream positions for salmonids relating specific growth rate to net energy gain. *Canadian Journal of Zoology*. 62 pp. 441-451.

Fausch, K.D. (1991) Trout as predator. In: Stolz, J. and J. Schnell (Eds.) *Trout*. The Wildlife Series, Stackpole Books, Harrisburg. pp. 65-72.

Filbert, R.B. and C.P. Hawkins (1995) Variation in condition of rainbow trout in relation to food, temperature, and individual length in the Green River, Utah. *Transactions of the American Fisheries Society*. 124 pp. 824-835.

Hansen, M.J., D. Boisclair, S.B. Brandt, S.W. Hewitt, J.F. Kitchell, M.C. Lucas and J.J.Ney (1993) Applications of bioenergetics models to fish ecology and management: where do we go from here? *Transactions of the American Fisheries Society*. 122 (3) pp. 1019-1030.

Hartman, K.J. and S.B. Brandt (1993) Systematic sources of bias in a bioenergetics model: examples for age 0-striped bass. *Transactions of the American Fisheries Society*. 122 pp. 912-926.

Hayes, J.W. and I.G. Jowett (1994) Microhabitat use by large brown trout in three New Zealand rivers. *North American Journal of Fisheries Management*. 14 pp. 710-725.

Hewitt, S.W. and B.L. Johnson (1992) Fish bioenergetics model 2. University of Wisconsin, Sea Grant Institute, Technical Report WIS-SG-92-250, Madison.

Hughes, N.F. (1992) Ranking of feeding positions by drift-feeding Arctic grayling (*Thymallus arcticus*) in dominance hierarchies. *Canadian Journal of Fisheries and Aquatic Sciences*. 49 pp. 1994-1998.

Hughes, N.F. and L.M. Dill (1990) Position choice by drift-feeding salmonids: model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. *Canadian Journal of Fisheries and Aquatic Sciences*. 47 pp. 2039-2048

Jobling, M. (1981) The influence of feeding on the metabolic rate of fishes: a short review. *Journal of Fish Biology. 18 pp.* 385-400.

Jones, D.R., J.W. Kiceniuk and O.S. Bamford (1974) Evaluation of swimming performance of several fish species from the MacKenzie River. *Journal of the Fisheries Research Board of Canada*. 31 pp. 1641-1647.

Lien, L. (1978) The energy budget of the brown trout population of Øvre Heimdalsvatn. *Holarctic Ecology*. 1 pp. 279-300.

Ney, J. (1993) Bioenergetics modeling today: growing pains on the cutting edge. *Transactions of the American Fisheries Society* 122 pp. 736-748.

Preall, R.J. and N.H. Ringler (1989) Comparison of actual and potential growth rates of brown trout (*Salmo trutta*) in natural streams based on bioenergetic models. *Canadian Journal of Fisheries and Aquatic Sciences*. 46 pp. 1067-1076.

Priede, I. G. (1985) Metabolic scope in fishes. In: Tytler, P., and P. Calow, P. (Eds.). *Fish Energetics: New Perspectives*. Croom Helm, Beckenham. pp. 33-64.

Ricker, W.E. (1971) Methods for assessment of fish production in freshwaters. 2nd ed. In: T. Edmundson (Ed.) *IBP Handbook No. 3*. Blackwell Scientific Publications, Oxford.

Ringler, N.H. and D.F. Brodowski (1983) Functional responses of brown trout (*Salmo trutta* L.) to invertebrate drift. *Journal of Freshwater Ecology*. 2 pp. 45-57.

Sagar, P.M. and G.J. Glova (1992) Invertebrate drift in a large, braided New Zealand river. *Freshwater Biology*. 27 pp. 405-416.

Scott, D., J.W. White and D.S. Rhodes (1994) Invertebrate fauna of three streams in relation to land use in Southland, New Zealand. *New Zealand Journal of Marine and Freshwater Research*. 28 pp. 277-289.

Stewart, D.J. (1980) Salmonid predators and their forage base in Lake Michigan: a bioenergetics-modeling synthesis. Doctoral dissertation. University of Wisconsin, Madison.

Statzner, B., C. Dejoux and J.M. Elouard (1984) Field experiments on the relationship between drift and benthic densities of aquatic insects in tropical streams (Ivory Coast). I. Introduction: a review of drift literature, methods, and experimental conditions. *Revue d'Hydrobiologie Tropicale*. 17 pp. 319-334.

Vinyard, G.L. and W.J. O'Brien (1976) Effects of light and turbidity on the reactive distance of bluegill (*Lepomis macrochirus*). *Journal of the Fisheries Research Board of Canada*. 33 pp. 2845-2849.

Wankowski, J.W.J. (1979) Morphological limitations, prey size selectivity, and growth response of juvenile Atlantic salmon *Salmo salar*. *Journal of Fish Biology*. 14 pp. 89-100.

Ware, D.M. (1972) Predation by rainbow trout (*Salmo gairdneri*): the influence of hunger, prey density, and prey size. *Journal of the Fisheries Research Board of Canada*. 29 pp. 1193-1201.

Ward, J.V. (1976) Effects of flow pattern below large dams on stream benthos: A review. In: *Instream Flow Needs Symposium, Vol. II*. J.F. Osborn and Allman, C.H. (Eds.), *American Fisheries Society*. 99. 235-253.

Werner, E.E. and J.D. Hall (1974) Optimal foraging and size selection of prey by the bluegill sunfish (*Lepomis macrochirus*). *Ecology*. 55 pp. 1042-1052.

Wooton, R.J. (1985) Energetics of reproduction. In: P. Tytler and P. Calow (Eds.) *Fish Energetics: New Perspectives*. Croom and Helm, Beckenham. pp. 231 - 254.

Modeling Atlantic Salmon Population Dynamics in a Small Stream[1]

Richard A. Cunjak
Department of Fisheries & Oceans, P.O. Box 5030, Moncton, N.B., E1C 9B6. Canada
E-mail : cunjakr@gfc.dfo.ca
and
Jean Therrien
Groupe-conseil Genivar inc.,5355, boulevard des Gradins,Québec, Québec. G2J 1C8. Canada
E-mail : jtherrien@genivar.com

A biological model was developed with the main objective of calculating annual survival between life-stages of juvenile Atlantic salmon (*Salmo salar)* in Catamaran Brook, a small stream basin (50 km^2) in the Miramichi River catchment of New Brunswick. Six years of data (1990-1995) were used in the model. Input variables included: daily fish counts and measurements of parr (3-4 age classes), smolts, and adult salmon at a fish-counting fence near the stream mouth; biennial quantification of all habitat-types along the watercourse; fish density estimates by electrofishing at 30 sites; estimates of young-of-the-year emigration via stream drift. Continuous recording of stream discharge at several locations in the basin provided data to assist in interpretation of survival estimates.

Results indicate that annual inter-age survival for juvenile salmon in their first 3 years in the stream averaged between 34% and 37%. Survival from egg deposition (after correction for losses due to predation and retention / non-fertilization) to smolt production was between 0.5% and 0.6% which is low relative to the estimates from many other studies. Potential errors in the computation of the model are discussed such as inaccurate counts of spawning adults during high autumn streamflow, density-dependant factors, and underestimates of fry (0+) abundance by electrofishing in the first year of the project. Another explanation for low survival, especially between egg and 0+ summer densities was the low winter streamflow in 2 years. Survival was five times higher (2.9%) when winters characterized by low streamflow were excluded from the calculation. Apparently, the low winter flows affected survival of parr (range = 23 - 32%) as well as eggs (16 - 25%).

---

[1] Contribution no. 30 of the Catamaran Brook Habitat Research Project

KEY-WORDS : Atlantic salmon / juvenile / population model / survival / winter / emigration / egg / production

INTRODUCTION

The modeling of Atlantic salmon population dynamics has been carried out by researchers for many years and in a variety of locations (Meister, 1962; Elson, 1975; Symons, 1979; Chadwick, 1982; Kennedy and Crozier, 1993). Often, the main objective has been to estimate smolt production (i.e. the number of juveniles migrating seaward). However, even among the more complete models (e.g. Chadwick, 1982; ; Kennedy and Crozier, 1993) with long time series and accurate counts of smolts (output) and egg-laden spawners (input), accounting for inter-annual survival and emigration of parr, and habitat-specific production is rarely attempted.

The main objective of this paper is to evaluate the survival of juvenile Atlantic salmon in Catamaran Brook, the site of a long term (15 year) multi-disciplinary research study of a small stream basin (50 km^2) in the Miramichi River catchment of New Brunswick (Cunjak *et al.*, 1990; 1993). This survival rate is based on counts of salmon moving upstream and downstream in the system, as established from captures in the trap of the fish-counting fence, and from densities derived from mid-summer (July) electrofishing surveys. Specifically, the model evaluates survival (and emigration) separately for each juvenile age-class (0+ to 2+), after accounting for habitat-specific and reach-specific population abundance, to ultimately compute a composite egg to smolt survival for each spawning cohort in Catamaran Brook. A unique feature of the model is that it takes advantage of a greater variety of physical (hydrological) and biological data than are typically available for other streams and population models. These data permit a more detailed interpretation of inter-stage survival estimates as well as providing the basis for explaining discrepancies. Further, it is sensitive enough to permit modifications in the general pattern of inter-stage survival, as deemed necessary based on the identification of new and pertinent information from the project.

STUDY AREA

Catamaran Brook (46^{0}52.7'N, 66^{0}06.0'W) is a third-order tributary of the Little Southwest Miramichi River in north-central New Brunswick, and the site of a long-term fish habitat research project (Cunjak et al. 1993; Cunjak 1995). The length of the main watercourse is 20.5 km with a mean slope of 1.27% (Cunjak et al. 1990). Mean annual discharge is approximately 1 cms ; mean daily water temperatures are generally <20^{0}C in mid-summer (Cunjak 1995); ice cover forms by early December and persists until mid-April.

Atlantic salmon are the most common fish species in the lower 10-12 km of the stream; further upstream salmon distribution is generally limited by numerous beaver dams which persist for many years after construction, and by an impassable barrier to upstream migration (waterfall) at approximately river km 16. Smolts typically emigrate at age 3 (Cunjak 1995).

MATERIALS AND METHODS

The basic steps of the model are shown in Figure 1.

Step 1 - Annual Egg Deposition

The number of eggs deposited in redds (nests) by spawning female Atlantic salmon is the basis for subsequent production of juvenile salmon (fry, parr, and smolts). Annual egg deposition is dependent, therefore, on the number, size, and fecundity of spawning females entering Catamaran Brook.

The number of spawners is determined at the fish-counting fence located near the mouth of the brook. However, the exact number is not possible each year due to problems of high water and avoidance of trapping by some adults while the fence is not operating. Estimates are made of the number of female spawners based on an annual correction for trapping efficiency (Table 1) and the proportion of females already identified in that year's spawning population. Estimates were made separately for the grilse (small, 1-sea-winter salmon) and the large, multi-sea-winter (MSW) salmon. Trapping efficiency values ranged from a low of 50% in 1990 (when the fence was situated in a poor location and often inoperable) to between 90% and 100% thereafter (Table 1).

Table 1. Trapping efficiency factors estimated for different life-stages of Atlantic salmon moving by the fish-counting fence in Catamaran Brook. Estimation for smolts was made using mark-recapture estimates from 1991-1995.

Year	Spawning Adults	Smolts	1+ Parr	2+ Parr
1990	50%	70%	65%	70%
1991	90%	70%	65%	70%
1992	90%	86%	81%	86%
1993	95%	80%	75%	80%
1994	100%	87%	82%	87%
1995	90%	80%	75%	80%

The individual weight of spawners was evaluated using the fork length data (from the counting-fence), and the regression equation from Moore *et al.* (1995) computed for Miramichi River salmon and expressed as follows:

$$\text{Weight (g)} = 0{,}000007 * \text{Fork length (mm)}^{3{,}069}$$

The estimated fecundity of Atlantic salmon spawners in Catamaran Brook was based on the relationship derived from adult females sampled in the Little Southwest Miramichi River (M. Hambrook, DFO, pers. comm., 1994). These values were 1385 eggs/kg for grilse, and 1387 eggs/kg for MSW salmon.

A correction for egg loss from retention in the body cavity and non-fertilization during release at the time of spawning was assumed to represent a 5% reduction in potential egg deposition. A further reduction of potential egg deposition was based on egg predation by juvenile Atlantic salmon parr (mainly precocious parr) during spawning (Table 2). These values were then applied to the estimated number of 1+ and 2+ parr in Catamaran Brook each autumn (see below) and the estimate for eggs removed by predators then subtracted from the total egg deposition.

Table 2. Predation of eggs by Atlantic salmon parr sampled November, 1994, in Catamaran Brook

Parr Age	Number	% with Eggs in Stomach	Average Number of Eggs per Stomach
0+	9	0	0
1+	11	9.1	5.0
2+	50	52.0	6.2

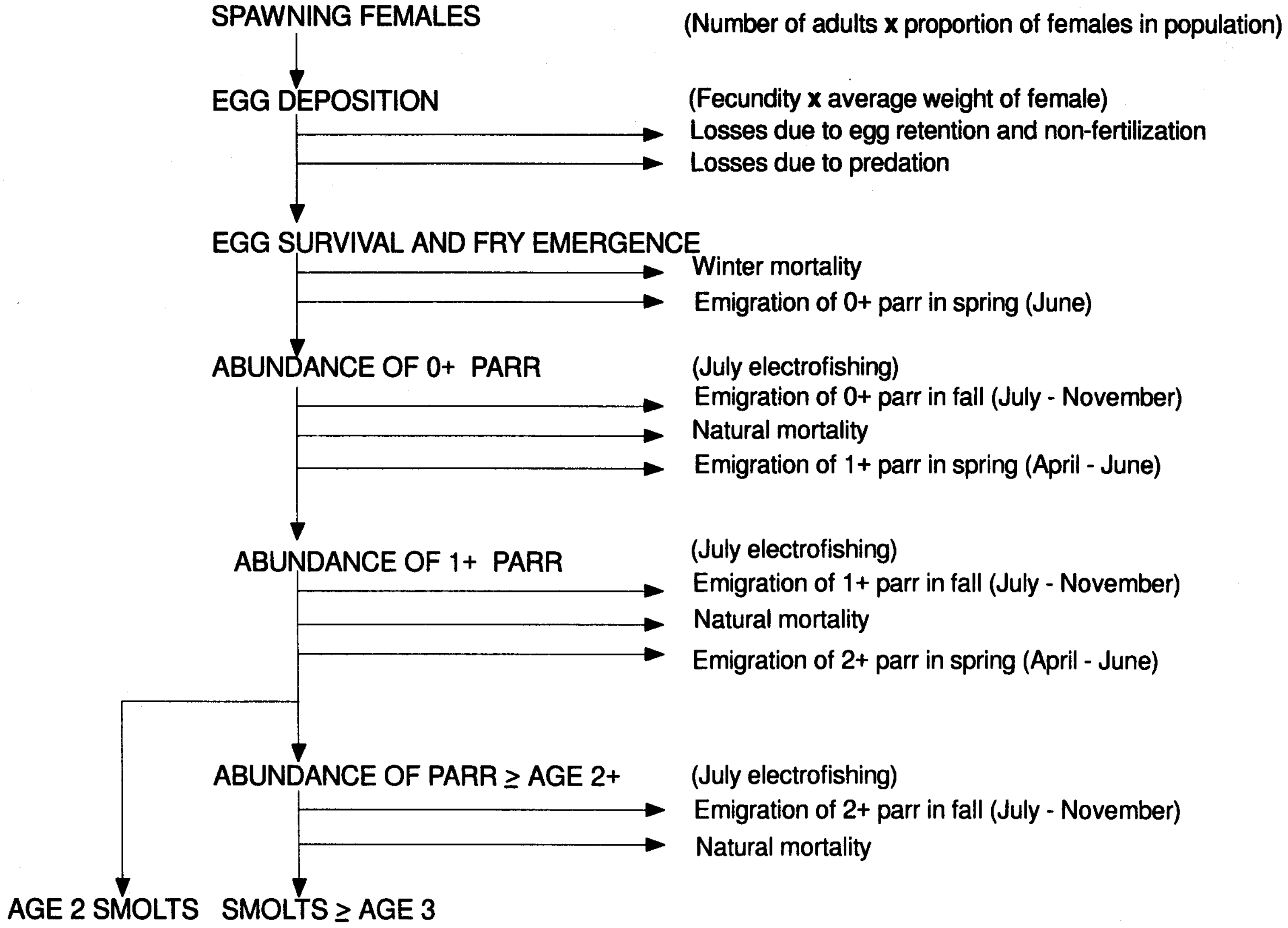


Figure 1. Flow diagram of salmon population model

Step 2 - Estimation of Annual Abundance of Juvenile Salmon, by Age-class

In order to determine inter-stage survival, it was necessary to calculate the annual abundance of each age-class of salmon in Catamaran Brook. This step involved two components :

(i) separate estimates of abundance for 0+, 1+ , and ≥2+ salmon parr based on densities derived from electrofishing in summer (July) in different habitat-types and stream reaches (see Cunjak *et al.*, 1993). Because the population of 3^+ parr wasn't differentiated from the 2^+ parr during the electrofishing surveys, they were pooled (hereafter referred to as 2+ parr). These population estimates were subsequently extrapolated to similar habitat-types within each reach according to the frequency and surface area represented by each habitat-type throughout the length of stream. The cumulative area available for the juveniles was calculated by habitat-type using field data collected during the 1993 habitat surveys for the entire length of Catamaran Brook. Four different habitat-types were considered, (flats, pools, riffles and runs). A total of 120,700 m^2 was surveyed with the proportion of flats, pools, riffles, and runs estimated at 16 %, 8 %, 40 %, and 36 %, respectively;

(ii) corrections were made to the population estimates (above) by accounting for losses from natural mortality (e.g. overwintering, predation) and from emigration (measured as net downstream movement at the counting-fence) for each age-class (Figure 1). First, the annual net downstream migration of parr was estimated.. For each year, we used the months of May to June of the same year and July to November of the preceding year, because electrofishing surveys were always made in July. Distribution within each age-class was obtained using length frequency distributions calculated using summer (July) lengths. In general, the length intervals corresponded to : ≤ 5.0 cm for 0^+ parr; > 5.0 and < 8.8 cm for 1^+ parr; ≥ 8,8 cm for 2^+ parr. For recently emerged salmon fry (<35mm fork length), from mid-June to early July, emigration was based on 1994 and `1995 counts of drifting fry captured near the stream mouth using fine-mesh drift nets. For 0+ parr after July, and for each successive age-class of parr, the number of net downstream emigrants was estimated at the counting fence between each summer's electrofishing surveys (i.e. from July to June of the following year).

Step 3 - Annual Production of Smolts by Age-class

The annual estimate of smolt emigration from Catamaran Brook is based on numbers obtained at the counting-fence (May - June) and corrected for trapping efficiency (Table 1). Efficiency ranged from 70 % to 87 % as determined from mark-recapture experiments in the brook (Cunjak *et al.*, 1993). In 1990, no test was made so efficiency was arbitrarily estimated as 70%, the same as for 1991, before improvements were made to the fence-trapping design. Ages assigned to smolts were based on scale analysis.

Assumptions

There are four basic assumptions for the model :

1. The first is the absence of (significant) migration into or out of Catamaran Brook when the counting fence is not in operation (December to April).

2. The second assumes that densities obtained (by electrofishing) for different age-classes in the study sites were related to a specific habitat-type and reach, and can be extrapolated for a similar combination of habitat-type and reach. The electrofishing efficiency is also assumed to be similar among sites and between years.

3. All adult females will spawn and thus egg deposition is not overestimated.

4. The last assumption refers to the emigration of parr. The fence trapping efficiency for 1+ parr was fixed at 5 % less than for smolts (Table 1) because capture efficiency is reduced for smaller sized fish (Cunjak *et al.*, 1989). Trapping efficiency for 2+ parr was assumed to be similar to smolts because of size overlap. Estimates of movement were inaccurate for 0+ parr because of spacing between conduit (6.5mm) in the fence and trap.

RESULTS

Fence counts of migrating salmon were a major component of the model. These data were especially important to the dependability of the survival estimates because the accuracy of the female spawner count (and egg deposition) is the first-step in the procedure (Figure 1). The average proportion of females among the adult spawners (1990 - 94) was approximately 16 % for the grilse and 75 % for the multi-sea-winter (MSW) salmon. The estimated number of adults moving into Catamaran Brook to spawn (Table 3) ranged from 56-166 grilse and 24-80 MSW for an average of 172 adults for the period, 1990 to 1995. The estimated spawner count for 1990 was the highest during the 6 years of study (Table 3) despite the fence being inoperable for several periods in the autumn due to flood conditions and eventually being washed-out during a major storm event in late October prior to the completion of salmon spawning. The low fence efficiency (Table 1) which accounted for the high number of spawners seems justified based on the abundance of 0+ salmon throughout the river in 1991 (highest population abundance estimated).

Table 3. Annual counts of Atlantic salmon adults (moving upstream) and smolts (moving downstream) at the fish-counting fence in Catamaran Brook, 1990-1995.

	FENCE COUNTS			CORRECTED COUNTS FOR FENCE EFFICIENCY		
YEAR	Smolts	Grilse	MSW	Smolts	Grilse	MSW
1990	760	83	28	1 086	166	56
1991	1 165	79	48	1 664	88	53
1992	2 135	127	67	2 483	141	74
1993	426	107	44	533	113	46
1994	887	56	24	1 020	56	24
1995	933	118	72	1 166	131	80
Average	**1 051**	**95**	**47**	**1 325**	**116**	**56**

The number of smolts counted at the fence annually, corrected for trapping efficiency (Table 1) varied from 533 smolts in 1993 to 2483 smolts in 1992 (Table 3). Except for 1991, 3 years old smolts were the most abundant age-class for this life-stage. On average, for 1990-95, the percent representation by smolts aged 2-4 was 24%, 74%, and 2.0%.

The number of viable eggs in Catamaran Brook, after correcting for predation and losses due to retention or non-fertilization, varied from approximately 100,000 (1994) to 500,000 (1992) with an average of approximately 300,000 eggs (Table 4). The low 1994 value reflected the low number of spawners that year (Table 3). The combination of few spawners and the large number of 2+ parr in the river (from the 1991 spawning cohort, Table 5a) likely resulted in a relatively high egg predation (Table 4) which represented approximately 25% of potential egg deposition compared with a 5-year average closer to 10%.

The population of juvenile salmon present each year in the river was estimated for 0^+, 1^+ and 2^+ salmon parr, using calculated densities by habitat-type and reach in order to calculate age-specific abundance for the entire stream (Table 5a). The summer (July) abundance of juvenile salmon in Catamaran Brook averaged >105,000 individuals per year from 1990 to 1995, and varied from a low of approximately 45,000 in 1995 to a high of approximately 164,000 in 1991 (see Table 5a for age-specific abundance, by cohort). The 1995 estimate also reflects the low number of spawners in 1994 wich resulted in the lowest 0^+parr evaluation for the project. The second lowest 0^+parr estimate was of 56,000 in 1990. That 1990 estimates of 0+, 1+, and 2+ parr abundance were the lowest of all years sampled (Table 5a) suggests that electrofishing efficiency may not have been the same in subsequent years. This might explain some of the high inter-stage survivals calculated (see below). By contrast, electrofishing in 1991-1995 seems to have yielded dependable numbers of parr when comparing river population estimates with independently derived fence counts in subsequent years. For example, 2+ parr were most abundant in 1991 (n=10,360 , Table 5a) and these parr produced the highest smolt count, in 1992 (Table 3). The lowest number of parr in the stream (1992) produced the fewest smolts ever to have emigrated from Catamaran Brook, in 1993 (Tables 3 , 5a).

TABLE 4. Annual egg production in Catamaran Brook

Year	Eggs In Spawning Female ± 95% c.i.	After Spawning[2]	Correction For Egg Predation	Number Of Viable Eggs (± 95% c.i.)[3] At Start Of Winter
1990	304817 ± 13805	289576	23119	266457 ± 12067
1991	415159 ± 12953	394401	44092	350309 ± 10929
1992[1]	539935 ± 9429	512938	31160	481778 ± 8413
1993	304787 ± 6749	289548	37349	252199 ± 5584
1994	156261 ± 5627	148448	40506	107942 ± 3887
Average[1]	344192 ± 8581	326982	35245	291737 ± 7274

[1] Number and confidence limits estimated with stratification weighting based on the female weight estimates.
[2] 5% of the total number of eggs were considered retained in body cavity or unfertilized.
[3] Based on the 95% confidence limits (c.i.) of eggs in spawning females.

Parr emigration, after correction for the efficiency of the counting fence (Table 1), was significant with annual averages of >1000 1+ parr and <300 2+ parr (Table 5a). 0+ salmon demonstrated a strong tendency to emigrate in the spring (June) soon after emergence from the gravel (1990-1995 average = 14,000 fry) with significantly less emigration thereafter. (Table 5a).

Survival was calculated according to the formulae shown beneath Table 5b. For example, the egg → 0+ survival is the sum of 0+ parr in the stream in summer (July) and the 0+ emigrants (June) in relation to egg production (Table 4) from the preceding autumn. This survival estimate varied from between 16.5% and 57.7%, with an average of 35.3% (Table 5b). The 0+ → 1+ parr survival averaged 37% among years; 1+ → 2+ survival averaged 34% (range=26-49%).

Table 5a. **Production and emigration of juvenile Atlantic salmon in Catamaran Brook.**

Cohort (spawning)	Number of Eggs	Parr in River			Emigrating Parr						Smolt Production	
		0+	1+	2+	0+		1+		2+		2+	3+, 4+
					Spring	Fall	Spring	Fall	Spring	Fall		
	(A)	(B)	(C)	(D)	(E)	(F)	(G)	(H)	(I)	(J)	(K)	(L)
1986												784
1987				3938					-10	49	302	730
1988			21083	10360			471	-45	83	76	934	2412
1989		31426	21907	5480	10102	550	1426	343	16	455	71	389
1990	266457	131702	29859	8240	21931	829	415	305	8	350	144	748
1991	350309	74728	23692	9741	12444	470	264	443	49	672	272	1121
1992	481778	68329	23589	6112	11378	430	572	1123	-81	115	45	
1993	252199	122750	24427		20440	772	-128	1095				
1994	107942	14755			7730	424						
Average	291737	73948	24093	7312	14004	579	503	544	11	286	295	1031

Table 5b. Inter-stage survival (%) of Atlantic salmon in Catamaran Brook. 'AVG1' refers to average survival computed from column averages in Table 5a; 'AVG2' refers to Table 5b averages.

	Egg → 0+	0+ → 1+	1+ → 2+	1+ → S2+	2+ → S3+S4+	Egg → 1+	Egg → S
	(M)	(N)	(O)	(P)	(Q)	(R)	(S)
1986							
1987					18.8		
1988			48.4	4.4	23.5		
1989		75.6	25.5	0.3	7.7		
1990	57.7	23.1	27.9	0.5	9.5	13.3	0.418
1991	24.9	32.3	42.1	1.2	12.4	8.0	0.512
1992	16.5	35.6	26.8	0.2		5.9	
1993	56.8	19.9				11.3	
1994	20.8						
AVG1	30.1	33.3	30.4	1.2	14.2	10.0	0.557
AVG2	35.3	37.3	34.4	1.3	14.4	9.6	0.465

0+, 1+, 2+ = parr age-class; S2+, S3+, S4+ = smolt age-class; S = smolt (all age-classes pooled);
M = (B+E)/A; N = (C+G)/(B-F); O = (D+I)/(C-H); P = K/(C-H); Q = L/(D-J); R = MxN; S = (RxP) + (RxOxQ)

The high survival values of 75.6% (0+ → 1+) and 49.4% (1+ → 2+) in Table 5b are believed to be the result of poor electrofishing efficiency in 1990 which led to an underestimate of the denominator terms in the respective calculations.

The egg → 1+ parr survival averaged 10% (range = 6% - 13%). Finally, a global survival from egg to smolt was calculated using a composite equation (Table 5b). For the two cohorts where all the intermediate rates are available (i.e. spawning years 1990 and 1991), the overall survival from egg to smolt was 0.42% - 0.51 % (average = 0.47%).

DISCUSSION

The average egg - smolt survival of 0.47 % calculated with the model was lower than other estimates for Atlantic salmon. In the Bec-Scie, Jupiter, Saint-Jean and Trinité rivers in Quebec, where scientific surveys were conducted for 7-15 years, the average survival rate varied from 1.2 to 5.4 (F. Caron, DEW, Quebec, pers. comm. 1995); in Newfoundland rivers, Chadwick (1982) estimated an average egg - smolt survival of 1.7%; and, Symons (1979), in summarizing the work of Elson (1975) and Meister (1962), estimated egg - smolt survival as 0.6% - 6.2%. Only Crozier and Kennedy (1993), who reviewed the data for two Irish rivers, noted a similar survival of 0.4% but only during the worst years in these rivers.

Factors influencing survival estimates can be related to local environmental conditions, density-dependent factors, adequacy of the model's assumptions, and the reliability of input variables. Environmental factors such as water temperature and stream discharge are known to affect survival of juvenile Atlantic salmon (Frenette et al. 1984; Gibson 1993). For example, Chadwick (1982) noted significant inverse correlations of egg - smolt survival with cold winters and low streamflow in Western Arm Brook.

In Catamaran Brook, the winters of 1991/92 and 1992/93 were significantly colder , with lower streamflow than the winters of 1990/91 or 1993/94 (D. Caissie, DFO, pers. comm.). Coincidentally, these two cold winters had the lowest egg - 0+ survival as well as the lowest inter-age parr survival. Estimation of egg - smolt survival, after excluding data from the winters of 1991/92 and 1992/93 results in an average egg-smolt survival of 2.9%, similar to the survival found in other Atlantic salmon rivers. More years of data are, however, necessary to substantiate the possible 'winter bottleneck' to survival in Catamaran Brook, and the importance of density-independent factors in affecting salmon survival.

Density-dependent factors may have influenced the low egg - 0+ survival of the 1994 cohort in Catamaran Brook because of the high density of spawning in the lower 0.8km of stream below a beaver dam. The combination of greater egg predation risk, super-imposition of redds, and the proportionally greater 0+ emigration (in June) may have combined to reduce the 0+ population estimate in July relative to egg deposition the previous autumn.

Regarding the reliability of input variables, the capture efficiency of electrofishing is likely not constant since factors such as experience of operator and conditions during sampling (i.e. weather, streamflow, etc.) could influence results. This would seem to explain the apparent underestimation of juvenile salmon (all ages) during the 1990 surveys which, in turn , inflated the survival estimates for each subsequent age cohort. Similarly the high water conditions in the autumn of 1990 reduced the efficiency of capture of spawners at the fence which, therefore, affected the accuracy of the egg deposition , a very sensitive variable to the estimation of smolt survival because it is the initial step in the model's computations. A new location for the fence and a more efficient trap design have reduced the problems with data collected after 1990.

In conclusion, salmon survival to the smolt stage appears to be relatively low in Catamaran Brook. This seems to be due to a combination of density-dependent and density-independent (environmental) factors acting in certain years, but particularly influencing winter survival. Inaccuracies in input parameters from the first year of study may also have influenced the estimates. The impact is especially obvious because there are relatively few years of data but should be minimized as more data (years) are incorporated into the model. Finally, the model promises to be a useful tool in understanding the population dynamics of Atlantic salmon in Catamaran Brook over the remaining years of the project because of its unique characteristic of treating inter-age survival and incorporating age-specific emigration rates in the equations. These factors, together with the available physical data (temperature , discharge) should provide a sound basis for monitoring and interpreting population change in the near future.

REFERENCES

Chadwick, E.M.P. 1982. Stock-recruitment relationship for Atlantic salmon (Salmo salar) in Newfoundland rivers. Can. J. Fish. Aquat. Sci. 39: 1496-1501.

Cunjak, R.A. 1995. Addressing forestry impacts in the Catamaran Brook basin: an overview of the pre-logging phase, 1990-1994. pp.191-210, In E.M.P. Chadwick (ed.) Water , science, and the public: the Miramichi ecosystem. Can. Spec. Publ. Fish. Aquat. Sci. 123 : 191-210.

Cunjak, R.A., D. Caissie, and N. El-Jabi. 1990. The Catamaran Brook habitat research project: description and general design of study. Can. Tech. Rep. Fish. Aquat. Sci. 1751: 14p.

Cunjak, R.A., D. Caissie, N. El-Jabi, P. Hardie, J.H. Conlon, T.L. Pollock, D.J. Giberson, and S. Komadina-Douthwright. 1993 The Catamaran Brook (New Brunswick) habitat research project: biological, physical, and chemical conditions (1990-1992). Can. Tech. Rep. Fish. Aquat. Sci. 1914: 81p.

Cunjak, R.A., E.M.P. Chadwick, and M. Shears. 1989. Downstream movements and estuarine residence by Atlantic salmon parr (*Salmo salar*). Can. J. Fish. Aquat. Sci. 46: 1466-1471.

Elson, P.F. 1975. Atlantic salmon rivers, smolt production, and optimal spawning : an overview of natural production. Int. Atl. Salmon Found. Spec. Publ. Ser. 6: 96-119.

Frenette, M., M. Caron, P. Julien, and R.J. Gibson. 1984. Interaction entre le débit et les populations de tacons (*Salmo salar*) de la rivière Matamek, Québec. Can. J. Fish. Aquat. Sci. 41: 954-963.

Gibson, R.J. 1993. The Atlantic salmon in fresh water: spawning, rearing, and production. Reviews in Fish Biology and Fisheries, 3: 39-73.

Kennedy, G.J.A. and W.W. Crozier. 1993. Juvenile Atlantic salmon - production and prediction. p. 179-187. In R.J. Gibson and R.E. Cutting [ed.] Production of juvenile Atlantic salmon, *Salmo salar*, in natural waters. Can. Spec. Publ. Fish. Aquat. Sci. 118.

Meister, A.L. 1962. Atlantic salmon production in Cove Brook, Maine. trans. Am. Fish. Soc. 91: 208-212.

Moore, D.S., G.J. Chaput, and PR. Pickard. 1995. The effect of fisheries on the biological characteristics and survival of mature Atlantic salmon (*Salmo salar*) from the Miramichi River. p. 229-247, In E.M.P. Chadwick (ed.) Water , science, and the public: the Miramichi ecosystem. Can. Spec. Publ. Fish. Aquat. Sci. 123 : 191-210.

Symons, P.E.K. 1979. Estimated escapement of Atlantic salmon (*Salmo salar*) for maximum smolt production in rivers of different productivity. J. Fish Res. Board Can. 36: 132-140.

The effect of water current on foraging behaviour of a rheophilic cyprinid, *Chondrostoma nasus*, during ontogeny: trade-off energetic benefit-swimming costs

Laurence Flore

Hubert Keckeis
University of Vienna, Institute of Zoology, Department of Limnology,
Althanstrasse 14, A - 1090 Vienna, Austria
FLOLO@zoo.univie.ac.at
HUBSI@zoo.univie.ac.at

ABSTRACT

In most European large rivers, populations of formerly dominant rheophilic fish species like *Chondrostoma nasus* have declined drastically. Engineered structural changes have led to a serious decrease of necessary habitats, especially of early fish stages which have very narrow habitat requirements that change over a short period of time. The current velocity appears to be a key abiotic factor in determining habitat selection of rheophilic fish. On one hand, water current acts as a limiting factor for microhabitat use, as fish cannot tolerate velocities above a critical value (Stahlenberg and Peckman, 1986). On the other hand, salmonids can select water currents which optimize their energy balance. Due to the lack of information about critical and energy-optimizing water velocities for 0+ cyprinids during ontogeny, our aim was to investigate 1) if rheophilic cyprinids are able to optimize their energy balance with the water current and 2) which critical water currents they can tolerate during ontogeny. Our laboratory experiments were conducted in a flume where the water velocity as well as the type, size and density of prey were varied. The foraging behaviour of three size classes of nase (15.2 ± 0.98 mm TL, 19.6 ± 0.78 mm TL and 24.5 ± 0.68 mm TL) was filmed and video recordings were analysed with an Image Analysis Software on a PC. Selected parameters of the "optimal foraging models", e.g. consumption rate and capture success of prey were determined. Additionally, critical water currents were measured: 1. nul benefit water velocity, 2. nul consumption water velocity and 3. maximum sustainable water velocity. The capture success was nearly 100% in relatively slow water currents and then drastically declined at high water velocities. No distinct optimum between energy intake and water current was observed within the investigated size range of fish. Nevertheless, we determined a profitable range of water currents which increased in the course of ontogeny, and swimming costs in early larvae were higher in proportion to the maximal energy intake (a positive net energy gain was registered between a range of 0 and 7 cm s^{-1} in 15 mm TL nase and between 0 and 15 cm s˙1 in 20 and 25 mm TL nase). The three defined critical water velocities did not differ from each other in early larvae of nase. In bigger nase, maximal sustainable water velocities were significantly higher than the nul benefit and the nul consumption water velocities. At the latter two last velocities, fish are unable to feed and obtain a positive energetic gain even if they are not submitted to "wash-out". The maximal sustainable water velocities of nase linearly increased with fish size and coincided with the velocities at which C. *nasus* larvae were dislocated (wash-out) in the Danube near Vienna. Based on data from the literature, maximal sustainable water velocities may depend more on the shape of the fish body than on species-specific life history patterns.

KEY-WORDS: young of the year, energy balance, optimal foraging, critical swimming speeds, consumption, capture success, drift feeding, *Chondrostoma nasus*, stream fishes.

INTRODUCTION

Although rheophilic cyprinids were very common in most European rivers, many populations have declined drastically and are severely threatened (Lelek, 1987). One example is the nase, *Chondrostoma nasus* L., a common rheophilic cyprinid species. Several studies have revealed that engineered structural damages led to a drastic decrease of necessary habitats, especially for 0+ fish (Copp and Peňáz, 1988; Spindler, 1988; Copp, 1989; Schiemer and Spindler, 1989; Olivier et al., 1995). Young fish stages are particulary vulnerable and have very narrow habitat requirements that change over a short period of time (Schiemer et al., 1991). In order to reestablish the populations, it is necessary to concentrate on the ecology of early stages which represent a critical period in fish life history (Kamler, 1992).

Changes in river discharge associated with flow regulation can negatively affect fish populations (Bunt and Mundie, 1986; Petts, 1994 in Shirvell, 1994). Higher water velocities, locally or generally produced by engineering measures, can act as a limiting factor for microhabitat (Stahlberg and Peckmann, 1986). Early stages are the most vulnerable to "wash-out" by high water (Heggenes and Traaen, 1988). Several authors during the last decade have devoted interest to investigating water velocity tolerance of fish (Brett, 1964; Houde, 1969; Brett and Glass, 1973; Ottaway and Clarke, 1981; Ottaway and Forrest, 1983; Irvine, 1984; Heggenes and Traaen 1987; Stahlberg and Peckmann, 1987; Facey and Grossman, 1989; Kaufmann, 1990; Meng, 1993; Young and Cech, 1994). Better swimming performance increases feeding efficiency, escape response, and maintenance of station against a current (Beamish, 1978; Webb, 1984, 1986 in Young and Cech 1994). Some studies concerning mainly salmonids indicated an increase in water velocity tolerance of fish with body size (Brett, 1964; Houde, 1969; Ottaway and Forrest, 1983, Heggenes and Traaen, 1988, Meng, 1993). Only few studies have concentrated on critical water currents of cyprinids (Kaufmann, 1990) but most of these have neglected ontogeny (Fricke et al., 1987; Stahlberg and Peckmann, 1987) Many of these previous studies defined critical water velocites as the maximal water velocities that can be sustained by fish. Nevertheless, it is also important to determine critical water velocities with regard to energy intake. Even if water velocities are insufficient to expose fish to "wash-out", they can still be too high to permit feeding and can therefore limit growth rates. Furthermore, some energy models revealed that salmonids can select water currents which optimize their energy balance (Fausch, 1983; Bachman, 1990; Hughes and Dill, 1990; Hill and Grossman, 1993). The selected positions in the water current apparently offer the best trade-off between swimming costs and supply of drifting food, both of which increase with water velocity.

Due to the poor information about critical and energy-optimizing water velocities for 0+ cyprinids during ontogeny, we addressed the following questions: (1) Is the rheophilic cyprinid *Chondrostoma nasus* able to optimize its energy benefit during ontogeny with regard to water current? (2) Which critical water currents can be tolerated by the nase during its ontogeny? We distinguished three types of critical water velocities; in addition to classical maximal sustainable water velocities , we also measured velocities at which feeding ceases (nul consumption water velocity) and at which net energy gain becomes negative (nul benefit water velocity).

MATERIAL AND METHODS

Fish and Prey Cultures

Larval nase were obtained by artificial fertilization of adults caught in a tributary of the Danube River (river km 1905). The eggs were put into well-oxygenated circular black tanks at a temperature of 16 ^{0}C. In our experiments, three mean fish size classes were used: 15.2 ± 0.98 mm TL (3rd larval stage); 19.6 ± 0.78 mm TL (5rd-6rd larval stages); and 24.5 ± 0.68 mm TL (1st juvenile stage). Larval and juvenile stages were defined according to Peñáz (1974). Two types of prey of different size classes were used to test fish foraging behaviour: *Artemia* sp. nauplii (0.7 mm ± 0.08 SD) and *Daphnia magna* (0.99 mm ± 0.08 SD, 1.52 mm ± 0.68 SD). *Artemia* prey were cultured in 5-l glass beakers in a saline solution of 30%o at 25°C. *Daphnia magna* was cultivated in 750 l plastic containers at temperatures between 18 °C and 22 °C. *Daphnia* were fed on yeast and a culture of green algae. Different size classes of prey were produced by sequential sieving through a set of sieves. Prey sizes were checked under binocular microscope.

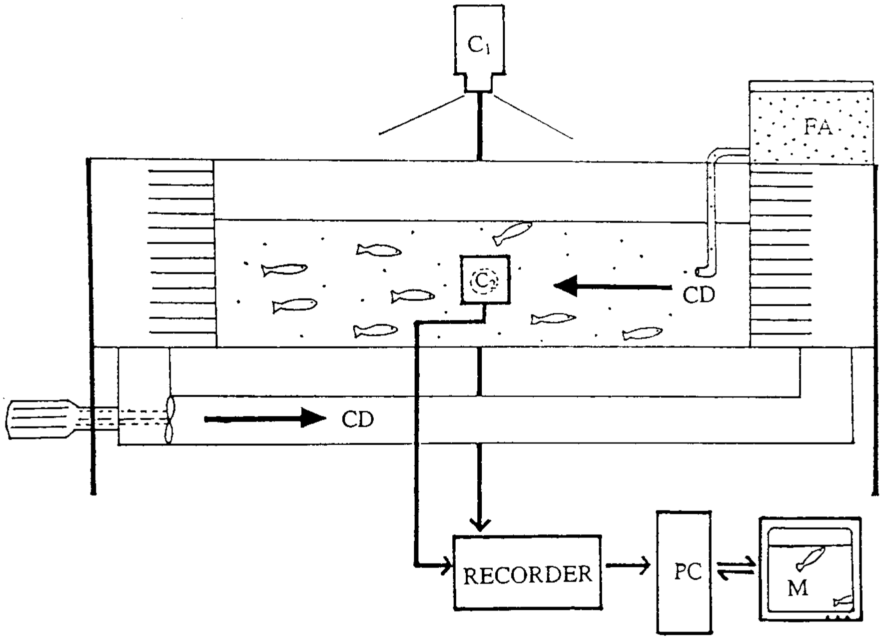


Figure 1: Schematic view of the technical installation. CD: Current Direction; C1: Camera 1; C2: Camera 2; FA: Feeding Apparatus; M: Monitor; PC: Personal Computer.

Experimental Design and Processus

The experiments were conducted in an artificial stream channel measuring: 300 x 50 x 50 cm (Figure 1). It consisted of a closed water circulation system driven by a power motor. Two 500-watt halogen lamps were placed above the channel. Water temperature was maintained at 16 ^{0}C and water depth at 10 cm. The flow rate could be regulated as desired. A calibration engine capacity-water velocity was established in each volume of 100 cm^{3} of the total channel volume. Two different water speed meters were used: a mechanical one (OTT: model C2 from 5cm s^{-1} to 2 m s^{-1}) and an electronic one (Mini Flow S/N 10181, Coastal Leasing, Inc.) to measure slower water velocities (< 5 cm s^{-1}). A smaller volume of the channel was delimited in order to study the foraging behaviour of the young nase. From pilot observations, a volume of 7500 cm^{3} was found to be sufficient to ensure the animals´ freedom of movement. Specific concentrations of prey were introduced into the experimental channel by a 1 cm-diameter tube leading from a 20-l feeding apparatus situated upstream. From this pipe, the debit of water flowing into the channel could be controlled and, ultimately, the number of prey passing through the channel during a certain interval of time. The tests were repeated with three prey numbers per unit of time and vertical surface: *Artemia*: ~200, ~560, and ~2733 $ind.s^{-1} m^{-2}$; *Daphnia*: ~55, ~113, and ~333 $ind.s^{-1} m^{-2}$.

Experimental fish were starved 24 h before the experiments and were introduced in groups of 8 individuals at least half an hour before the beginning of each test (the older the fish, the more time they require to adapt to experimental conditions). These fish groups were changed for each test in order to avoid a learning effect. Feeding behaviour was filmed with semi-professional video equipment. Two cameras were used, one filming in the vertical plane (Sony color video: model DXC-327 P) and one filming in the horizontal plane (Grundig camecor LC 460 SC). The former was connected to a JVC video cassette recorder (model BR-5822 E), while the latter had an incorporated recorder. The two cameras were linked by a synchronisation cable to make each image correspond in the two planes. A light signal was given at the onset of each experiment. The test duration was 7 minutes. The recordings were analysed with an Image Analysis Software (Optimas) on a personnel computer.

Energy Balance

Some former models showed how salmonids can optimize their energy balance with regard to water velocity:
a) model of Fausch for brook trout, brown trout and coho salmon (1983):

$$(1)\ \mathbf{P = D - S}$$

where P = potential profit; D = available drift energy ($D = 1/8 \prod r^2 E$, where r is the feeding radius of two body length and E is the drift energy rate at maximum velocity into the feeding radius; S = swimming cost (focal-point water velocity measured at the fish´s head (Steward, 1980 in Hill and Grossman, 1993)).

b) model of Hughes and Dill for grayling (1990). Following Fausch, these authors proposed another model which takes into account water depth, the shape of the fish´s reaction field, variations in water velocity within the fish´s foraging area and the influence of water velocity on the fish ability to capture prey:

$$(2)\ \mathbf{NEI = GEI - SC}$$

where NEI = net energy intake rate in the foraging space. This space is described with a series of 25 cm strips extending from the water surface to the stream bed and characterized by water depth and velocity. GEI = gross energy intake ($GEI = \sum^{3}_{i=1} \sum^{+3}_{j=-3} CA_{ij} . V_j . PC_i . PE_i$ where CA_{ij} is the defined capture area, V_j is the average water velocity in strip j, PC_i is the concentration of prey size class i and PE_i is the energy content of prey in size class i); SC = swimming cost (Brett and Glass, 1973).

c) model of Hill and Grossman for rainbow trout and rosyside dace (1993). These authors also estimated the potential energy benefit of a microhabitat by measuring food abundance and prey capture success at a given velocity. In addition to Hughes and Dill´s model, they calculated from this potential energy benefit the inorganic components of prey and subtracted the energy associated with the insoluble and indigestible portions:

$$(3)\ \mathbf{E_x = B_x - C_x}$$

where E_x = net energy gained from holding position at velocity x; B_x = benefit obtained at that velocity ($B_x = I_x - (R_{SDA} + F + U)$; $I_x = A_x \cdot S_x$ where A_x is the energetic value of prey available at velocity x and S_x is the capture success for velocity x, R_{SDA} is the specific dynamic action, F is the ingestion rate and U is the excretion rate (Rice et al., 1983 in Hill and Grossman, 1993)); C_x = swimming cost (Facey and Grossman, 1990).

d) The energy balance we determined is based on the last-cited general equation where Ex (net energy gain) is the difference between Bx (gross energy gain) and Cx (swimming cost) at the specific water velocity. In the three above-cited models, fish net energy gains were calculated by the estimated energy content of drift passing in their foraging area predicted from the maximal reactive distances and the fish´s visual field. Following Fausch (1984), Hughes and Dill (1990) integrated prey capture success in their models and Hill and Grossman (1993) the utilisation efficiency (deduction of R_{SDA}, F, U). We obtained the net energy gain from directly observed consumption minus swimming costs. The total prey ingested per fish and minute, accurately counted from the video recordings, was converted into energy content (21 J mg^{-1} dry mass for *Daphnia* sp. (Richman, 1958; Cummins and Wuycheck, 1971) and 24.7 J mg^{-1} nauplii *Artemia* sp. (Paffenhöfer, 1967)). Non-assimilable fractions of energy ingested (30%) were deducted from the total (Ware, 1975; Elliot, 1976; Keckeis and Schiemer,1992). Values for steady-state swimming costs in increasing currents were taken from the literature (Kaufmann, 1990).

Additionally, the following variables were determined:
- Prey capture success : percentage of successful prey strikes in increasing water currents.
- Consumption: number of prey ingested per minute or energy content of prey ingested per minute.
- Maximal consumption: maximal mean value of consumption expressed in energy.
- Critical swimming speeds in the current : 1. **maximal sustainable water velocity:** water velocity at which fish cannot maintain their position in the water. This was measured by progressively increasing water velocity until the animals failed to swim for at least two minutes, which is the time required to reach a steady exercise respiration level (Kaufmann, 1990). Fish were considered as washed out when they were caught on a screen situated at the downstream end (Heggenes and Traaen, 1987). The young fish often lay on the bottom where the current was minimal due to hydrodynamic drag. If they rested more than 5 minutes, this was scored as failed (Meng, 1993). 2. **nul consumption water velocity** : water velocity at which fish no longer fed for at least 10 minutes but were still able to swim. 3. **nul benefit water velocity**: water velocity at which fish net energy gain became negative.

Data analysis

One-way Anova was used to determine significant differences in capture successes, consumption and critical water velocities of the nase in different combinations of factors (water velocity, fish size, prey type, size and density). A normal distribution- and a variance homogeneity-test were carried at prior to this.

RESULTS

Capture Success (CS)

The relationship between capture success (CS) and water velocity for two fish sizes is shown in Figure 2. The data obtained at the three prey densities showed no significant difference and therefore the data from all experiments were pooled. In both size classes, CS declined significantly with increasing water velocity ($p < 0.001$). In small fish (15.2 ± 0.98 mm TL) capture success remained high up to a velocity of 6 cm per second and then decreased drastically to 0 at 7.4 ± 1.28 SD cm per second. In contrast, CS in bigger fish (19.6 ± 0.78 mm TL) diminished more gradually and reached 0 at 14.3 ± 1.56 SD cm per second. This represents a two-fold increase of the range in which prey can be caught from early larval to late larval stages. In still water, CS was not significantly different between the two fish size classes ($p < 0.03$), the values approaching 100. Larger fish had significantly higher CS ($p < 0.03$) under flowing water conditions. A significant effect of prey size on capture success was observed at higher water velocities (Fig 2 b). CS of bigger fish on 1 mm *Daphnia* and 0.6 mm *Artemia* nauplii was similar up to a velocity of 6 cm per second, but CS on smaller sized prey was significantly higher at more rapid current speeds ($p < 0.001$).

Consumption

Consumption, expressed as prey items ingested per unit of time or in terms of energy, was significantly dependent on water velocity (Figures 3 and 4; $p < 0.001$). As an example, Figure 3 illustrates the relationship between consumption at different prey numbers and water velocities for 15.2 ± 0.98 mm TL fish. At the two lower prey densities offered, consumption was almost constant up to 6.0 cm s^{-1} before sharply declining to 0 at a velocity of 7.4 ± 1.28 SD cm s^{-1}. At the highest prey density, maximal consumption was observed at a water current of 2.5 cm s^{-1}, and then declined to levels similar to those at intermediate food concentrations. Prey availability also had a significant effect on the level of consumption. Food intake at the lowest prey density (200 ind. s^{-1} m^{-2}) was significantly lower than consumption at the two other concentrations ($p < 0.001$).

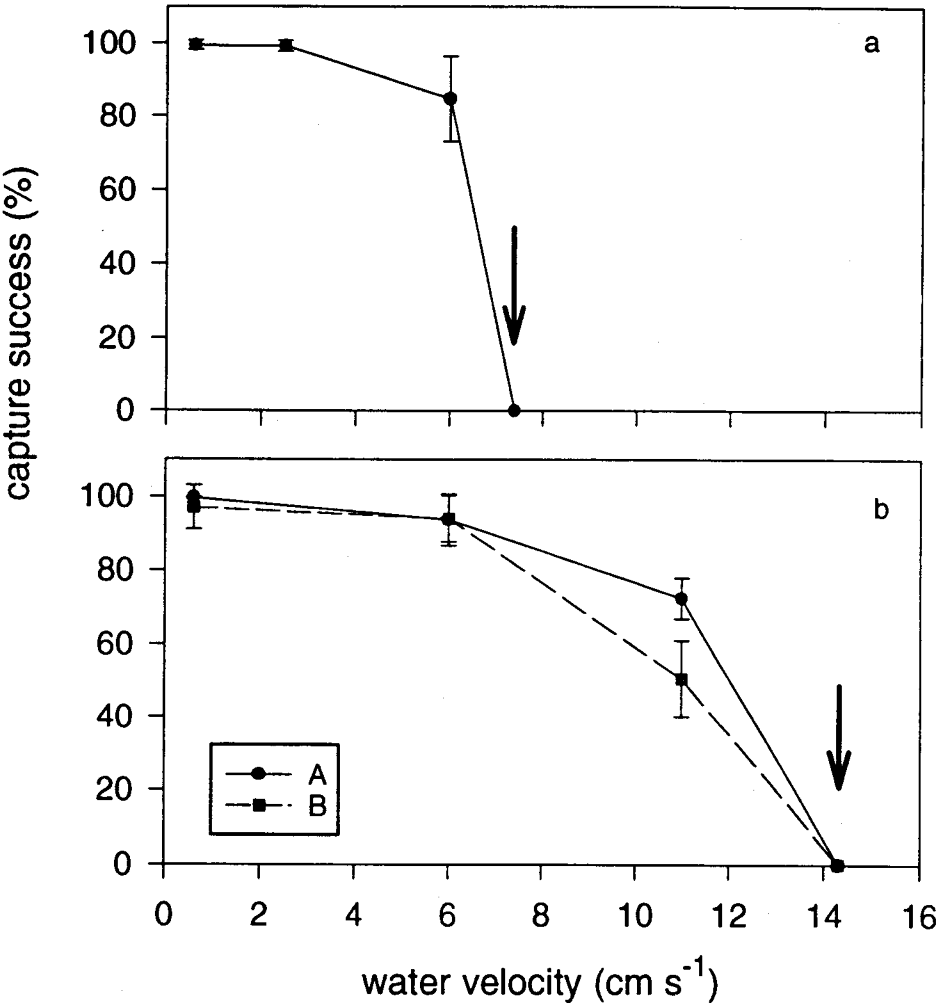


Figure 2: Mean capture success versus water velocity in two size classes of nase. a: 15.2 ± 0.98 mm TL; b: 19.6 ± 0.78 mm TL. Mean Capture Success on different prey types are compared for the second fish size class: A: *Artemia* sp. nauplii of 0.6 mm; B: *Daphnia magna* of 1 mm. The arrows show the water currents at which Capture Success declined to 0. Vertical bars represent ± standard deviations (n = 6).

Figure 4 exemplifies the effect of prey type on consumption of 19.6 ± 0.78 mm TL fish at different water velocities. Consumption of both prey types was high between 0 and 6 cm s^{-1} before decreasing continuously to 0. Although a lower number of *Daphnia* (Fig. 4 b) than *Artemia* nauplii (Fig. 4 a) was ingested by the fish at each water velocity, the energy gained from the bigger *Daphnia* was significantly higher ($p < 0.001$).

Maximal consumption of nase during ontogeny (size ranging from 15 to 55 mm TL) under different current conditions are shown in Figure 5. The values represent consumption on those prey types and sizes where maximal energy uptake could be observed. In each size class of fish the values were relatively constant in a certain range of water currents and then declined drastically to the nul consumption water velocity. Maximal consumption increased significantly with size ($p < 0.001$) and decreased with water velocity within each size class of fish. The velocity at which no consumption was observed generally increased with fish size, although the values for 20 and 25 mm fish were rather similar.

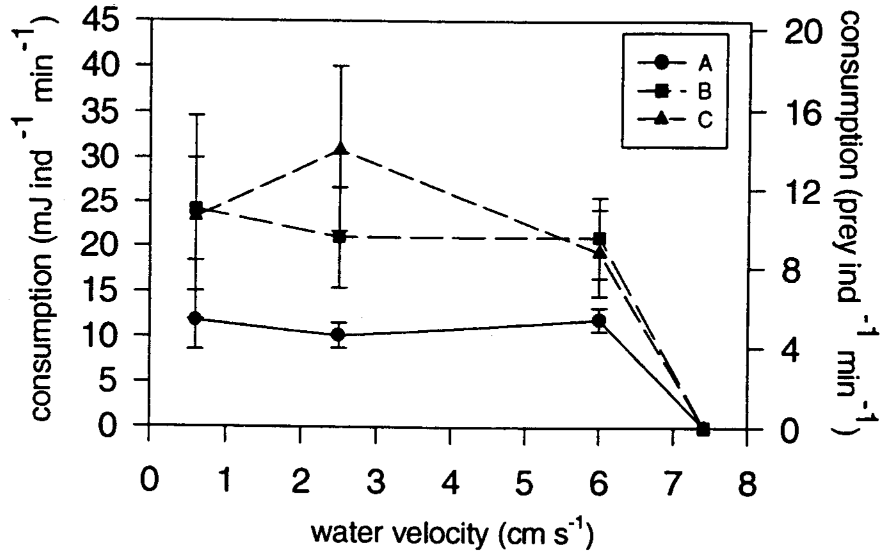


Figure 3: Mean consumption versus water velocity in early larvae of nase (15.2 ± 0.98 mm TL) at three different prey numbers. A: ~200 *Artemia* nauplii min^{-1} m^{-2}; B: ~560 *Artemia* nauplii min^{-1} m^{-2}; C: ~2733 *Artemia* nauplii min^{-1} m^{-2}. Vertical bars represent ± standard deviations (n = 6).

Energy Balance

Table 1 summarizes energy balance with regard to the water current in three fish size classes. Figure 6 illustrates this energy balance for the two size classes of nase. For comparison, consumption in terms of the assimilated fraction (C_{ass}; net energy gain) and swimming costs are expressed as a percentage of maximal energy values obtained at each fish size. The net energy gain showed a maximum at an intermediate water velocity of 2.5 cm s^{-1} in 15 mm fish and of 6 cm s^{-1} in 20 mm fish before decreasing to zero. Late larval stage of fish could feed in a range of water velocities that was about twice as large as early larval fish (0 to 7.2 cm s^{-1} versus 0 to 14.3 cm s^{-1}. The point at which $C_{ass.}$ and swimming costs were equal determined the nul benefit water velocity. This was reached at 6.7 ± 0.15 cm s^{-1} (mean ± standard deviation) by 15 mm nase and at 13.8 ± 0.01 cm s^{-1} by 20 mm nase. At this point, swimming costs corresponded to 25% of the maximum energy gain in the first case and to 15 % in the second case. This indicated that energy lost due to swimming costs represented proportionally more energy for the early larval stage.

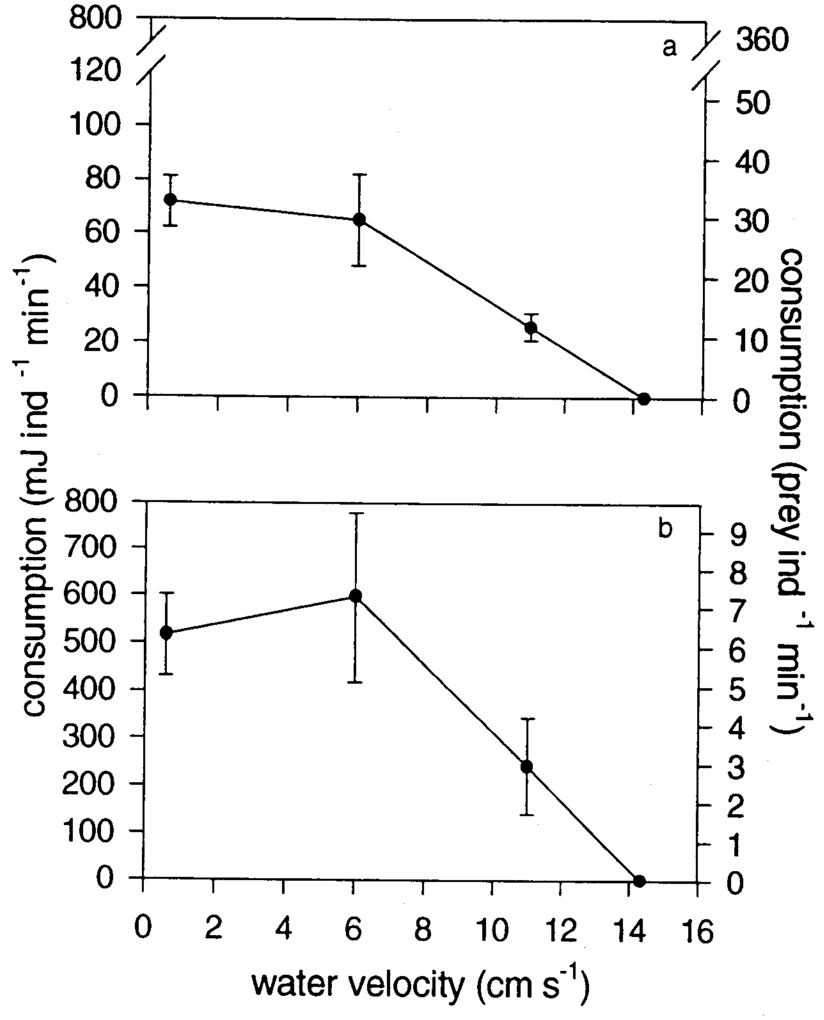


Figure 4: Mean consumption versus water velocity in late larvae of nase (19.6 ± 0.78 mm TL). a: Mean consumption on 0.7 ± 0.08 mm *Artemia* sp. nauplii; b: Mean consumption on 0.99 ± 0.08 mm *Daphnia magna*. Vertical bars represent ± standard deviations (n = 6).

Critical Water Velocities during Ontogeny

Figure 7 and Table 2 summarise the relationship between fish size the three critical water velocities (CWV) measured during this study: nul benefit water velocity, nul consumption water velocity and maximal sustainable water velocity (MSWV). Each of the CWV rose significantly during ontogeny ($p<0.001$). A steep increase between early and late larval stages was observed, whereas the slope diminished between late larval and early juvenile stages. It appeared that the three CWV were not significantly different in the 15-mm-sized nase (p >

0.05). At larger sizes, the nul benefit water velocity and the nul consumption velocity were also not significantly different, but the maximal sustainable water velocities were significantly higher ($p < 0.008$ for 20 mm TL and $p<0.002$ for 25 mm TL).

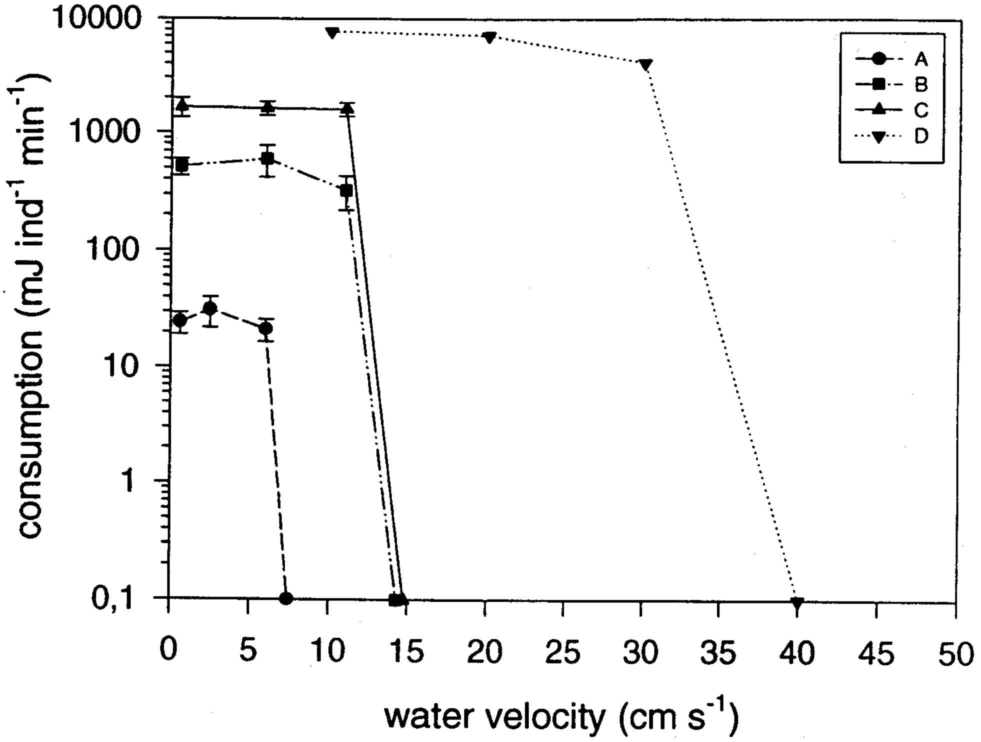


Figure 5: Mean maximal consumption versus water velocity in 4 size classes of nase. A: 15.2 ± 0.98 mm TL; B: 19.6 ± 0.78 mm TL; C: 24.5 ± 0.68 mm TL; D: 55 ± 1.04 TL. Vertical bars represent ± standard deviations (n = 6).

DISCUSSION

Several models have predicted habitat selection of adult salmonids with regard to their energy optimization in the water current (Fausch, 1983; Hughes and Dill, 1990; Addley, 1993; Hill and Grossman, 1993). Nevertheless, the knowledge for early larval and juvenile stages of both cyprinids and salmonids is poor. Addley (1993) attempted to apply a model to young fish but failed to predict optimization in their habitat choice. Defining habitat criteria, especially for young fish, is increasingly important in the field of river management and restoration projects. In order to establish an energy model for 0+ fish of one dominant rheophilic cyprinid species in many European rivers, we analysized and quantified some of the most relevant factors and their changes during ontogeny.

Table 1: Energy balance in three size classes of nase. Consumption (C), gross energy gain (assimilated portion of consumption: $C_{ass.}$), swimming costs (SC; Kaufmann, 1990) and energy benefit at different water velocities during ontogeny. Prey used: *Artemia* sp. nauplii (0.6 mm) in 15.2 ± 0.98 mm TL TL nase, *Daphnia magna* of 1 mm in 19.6 ± 0.78 mm TL nase, *Daphnia magna* of 1.5 mm in 24.5 ± 0.68 mm TL nase. Mean ± SD.

mean TL (mm)	mean w. vel. (cm s^{-1})	C (prey ind.$^{-1}$ min^{-1})	C (mJ ind.$^{-1}$ min^{-1})	$C_{ass.}$ (mJ ind.$^{-1}$ min^{-1})	SC (mJ ind.$^{-1}$ min^{-1})	net en.gain (mJ ind.$^{-1}$ min^{-1})
15.2	**0.6**	10.53 ± 5.13	23.27 ± 11.33	16.29 ± 7.93	3.43	12.86 ± 7.93
15.2	**2.5**	14.00 ± 4.15	30.91 ± 9.16	21.64 ± 6.41	3.59	18.06 ± 6.41
15.2	**6.0**	8.77 ± 2.19	19.36 ± 4.84	13.55 ± 3.39	5.35	8.20 ± 3.39
15.2	**7.4**	0	0	0	6.55	-6.55
19.6	**0.6**	4.40 ± 0.72	516.06 ± 84.57	361.24 ± 59.20	7.94	353.30 ± 59.20
19.6	**6.0**	5.10 ± 1.53	598.16 ± 179.71	418.71 ± 125.80	11.10	407.61 ± 125.80
19.6	**11.0**	2.06 ± 0.86	242.39 ± 101.34	169.67 ± 70.94	19.63	150.04 ± 70.94
19.6	**11.4**	0	0	0	28.53	-28.53
24.5	**0.6**	4.43 ± 0.84	1657.43 ± 315.17	1160.20 ± 220.62	16.25	1143.95 ± 220.62
24.5	**6.0**	2.88 ± 0.39	1076.71 ± 145.76	753.70 ± 102.03	20.70	733.00 ± 102.03
24.5	**11.0**	4.10 ± 0.72	1532.81 ± 270.63	1073.00 ± 189.00	32.73	1040.27 ± 189.00
24.5	**14.7**	0	0	0	47.34	-47.34

Capture Success

Prey capture success (CS) of fish depends on many intrinsic and extrinsic factors. The degree of appetite, for example, determines the motivation to succeed in the prey attack (Dunbrack and Dill, 1983). The fish we tested were starved for about 24 h before the experiments; thus, a high motivation to attack prey could be assumed. The ability to capture prey has been proved to increase with fish size (Mittelback, 1981; Wanzenböck, 1989; Health, 1993). Also a learning effect due to repeated exposures of the same individuals to the prey can significantly improve capture success (Winfield et al., 1983, Scott, 1989), which can be neglected in our investigations because we used independant samples. Improvement in the visual acuity during ontogenesis can be explained by the growth of eye and the development of signal integration, e.g. the optic tectum (Wanzenböck, 1989). The development of the musculature during ontogeny will also allow the fish to swim and reach the prey faster, consequently increasing the probability of capture (Nyberg, 1971). Whereas CS did not differ between late larval (15.2 ± 0.98 mm TL) and early juvenile nase (19.6 ± 0.78 mm TL) in still water, the bigger fish were more successful in their prey attack under flowing water conditions. This can be explained by the fact that detectibility of prey items is likely to be highest at low velocities and lowest at very high velocities (Gendron and Staddon, 1983; Wilzbach, Cummins and Hall, 1986). The nase is apparently a fish species that is relatively successful in its prey capture, even during the first stages of its development. Different CS between fish species has been demonstrated by Wanzenböck (1992) for roach (*Rutilus rutilus*), bleak (*Alburnus alburnus*) and blue bream (*Abramis ballerus*). On the other hand, Hill and Grossman (1993) found that capture success was size- rather than species-dependent as far as trout and dace were concerned.

Furthermore, environmental factors play a major role in fish capture success. Because CS is limited by the rate of muscle contractions, it is probably directly correlated with temperature (Hill and Grossman, 1993). In our investigation, the temperature was maintained constant at 16 ^{0}C, which has been found to be the optimal temperature for nase growth (Kamler et al., 1996). Capture success is highly dependent on prey characteristics. It varies significantly with prey type, size, width and shape (Scott, 1989; Wanzenböck, 1989; Heath, 1993; Hill and Grossman, 1993) as well as with its crypticity (Wilzbach et al., 1986). Locomotor activity is one major factor controlling the initial vulnerability of prey to predation in still water (Scott, 1989; Heath, 1993). In flowing water, Scott (1989) observed that the drift of prey organisms suspended in the water column was sufficient to induce fixation by the fish, and when dead prey were placed in the area, they were fixed with the same efficiency as live prey. We therefore consider that the *Artemia* sp. nauplii and *Daphnia magna* we used as prey presented no difference in escape ability under flowing water conditions. Additionally, maintaining station in the current involves large energy costs and also limits the time available to examine prey; it is therefore unlikely that fish in flowing water will specialize, unless food concentrations are very high (Scott, 1987). The different prey concentrations offered in our investigation did not induce any significant variations in the CS, suggesting that the nase is equally efficient in its prey capture at low and high food availability. We observed that prey size has a relevant effect on CS in nase exceeding 20 mm only at high water velocity. This is probably due to the higher handling times induced by bigger prey, as shown by Shoener (1969), Mittelbach (1981) and Wanzenböck (1992). High water velocities do not allow fish to spend much time for handling rapidly drifting prey. Water velocity is a relevant environmental factor that influences capture success. As shown by Dunbrack and Dill (1983) and Hill and Grossman (1993), we observed CS to be virtually 100% in relatively slow water if the prey are not too small or too large and then to dramatically decline at high water velocities.

Consumption

In contrast to the prediction of the model from Hill and Grossman (1993), consumption and thus net energy gain at different current speeds revealed a different pattern in young nase (< 25 mm total length). The above-cited authors observed a dome-shaped curve (low energy gain at low and high currents, and a maximum gain at intermediate current conditions). Our data showed that larvae and early juveniles achieved high consumption rates over a range of water velocities without significant optima. Above a size-dependent critical velocity, consumption decreased drastically to zero (Figure 5). One reason for this difference may involve the methodology. In the mentioned study, consumption and ultimately the net energy gain were indirectly estimated, i.e. calculated from a combination of observations of capture success in different current conditions and drift densities measured under field conditions. In the present work, consumption was directly obtained from the number of ingested prey times their energetic value. A second potential reason is the low range of water currents that can be sustained by young fish, e.g. from 0 to 7 cm sec^{-1} in early larvae and increasing to 0 to 40 cm sec^{-1} in late juveniles (Figure 5). We assume that early larvae have to be flexible within the very narrow range of water currents which they can tolerate in order to achieve sufficient energy benefit required for their high growth rates. However, a comparison of the maximal energy intake at optimal velocities of young trout and dace (Hill and Grossman, 1993) revealed values similar to our observations. The maximal consumption of 55 mm nase was approx. 7500 mJ ind.$^{-1}$ min^{-1} at a velocity range of 10 to 30 cm s^{-1}; young trout of 66 mm consumed 12072 mJ ind.$^{-1}$ min^{-1} at an optimum velocity range of 8 to 18 cm s^{-1} and dace of 61 mm consumed 9518 mJ ind.$^{-1}$ min^{-1} at an optimum range of 7 to 16 cm s^{-1}.

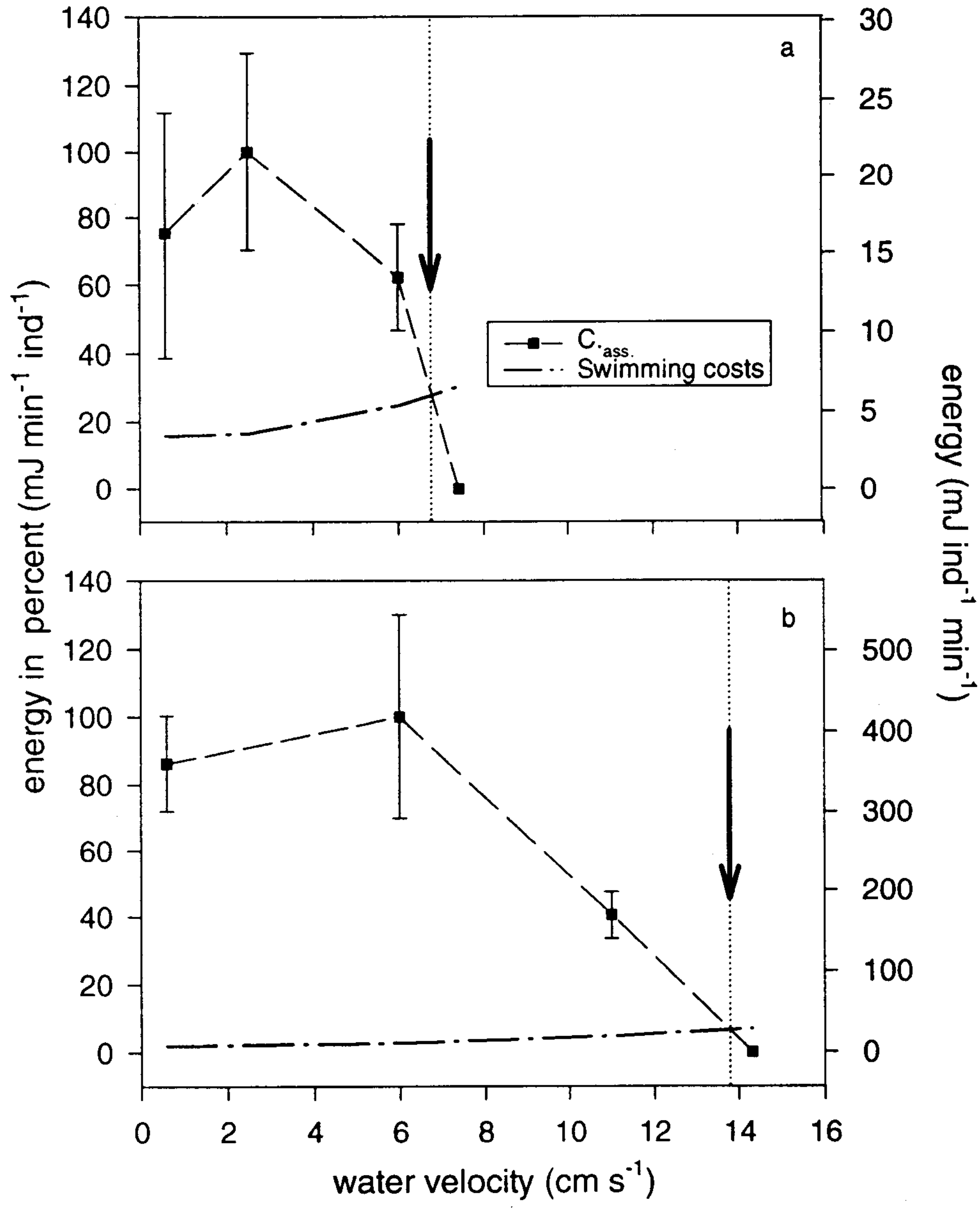


Figure 6: Mean energy balance versus water velocity in two nase size classes. a: 15.2 ± 0.98 mm TL; b: 19.6 ± 0.78 mm TL. The percent values are in relation to the highest observed consumption value. The data for swimming costs were taken from the literature (Kaufmann, 1990). The arrows indicate the nul benefit water currents. Vertical bars represent standard deviations (n = 6).

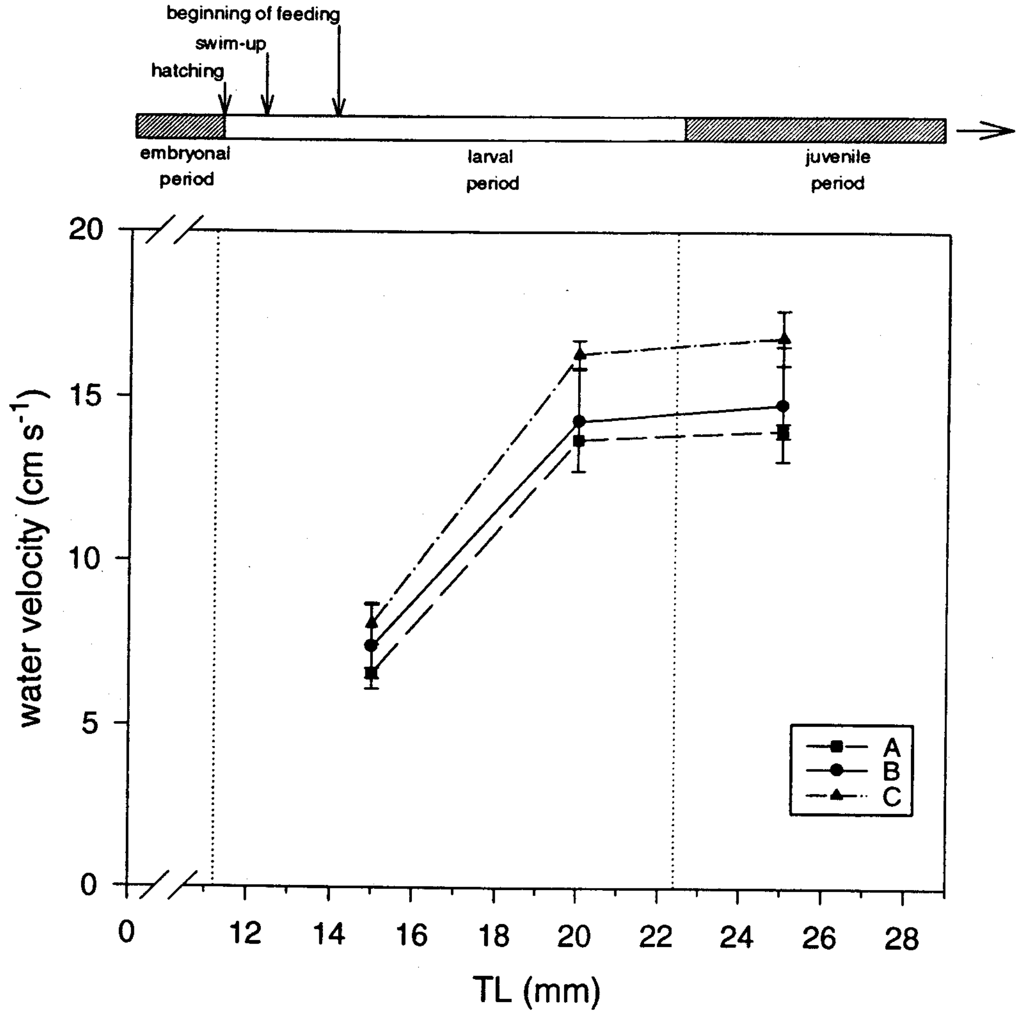


Figure 7: Mean critical water velocities versus fish size. A: nul benefit water velocity; B: nul consumption water velocity; C: maximal sustainable water velocity. Vertical bars represent standard deviations (n = 6).

Energy Balance

The primary objective of this study was to investigate whether 0+ nase are able to optimize their net energy gain with regard to water current. Although no distinct optimum between energy intake and water current was observed in fish < 25 mm, it appeared that the profitable range of water currents increased during ontogeny. This increase is due to morphological and physiological ontogenetic development (e.g. visual, muscular development, increase in swimming performance and metabolic capacity). The relationship between energy intake and water velocity has been discussed above. On the other hand, routine and active respiration rates increase with water velocity and decline with body weight (Kaufmann, 1990). In early larvae of nase, swimming costs significantly influenced the net energy gain. They represented up to 25% of the maximal energy value. The costs of transport in early larvae are higher compared with later stages because of the dramatic changes accompanying the development of early life stages (Kaufmann, 1990). In bigger nase (> 20 mm TL), net energy gain was not significantly influenced by swimming costs. The swimming costs we used from the literature (Kaufmann, 1990) were derived for steady-state swimming conditions and did not account for the costs involved in unsteady state swimming activities such as accelerating and decelerating. Webb (1982 in Addley, 1993) has shown that unsteady swimming is more energetically costly than steady swimming. We should therefore expect swimming costs to be slightly higher when considering unsteady swimming activities. These costs during prey capture can be included based on time budget and reactive distance analyses and will be presented in the near future.

Critical Water Velocities

The second of our main objectives was to determine the critical water velocities (CWV) of the nase and to find out whether significant differences could be detected between the three distinct CWV we previously defined, e.g. the nul benefit water velocity, the nul consumption water velocity and the maximal sustainable water velocity (MSWV). Because early larval nase could tolerate only a very narrow range of water velocities, the three CWV did not significantly differ. On the other hand, the bigger nase (between 20 and 25 mm TL) tolerated a range about twice as large. In this case, maximal sustainable water velocities appeared to be significantly higher than the nul benefit- and the nul consumption- water velocities. It is therefore important to consider the latter two velocities as critical water velocities for late larval and early juvenile stages. Although only a few centimeters per second separate them from the maximal sustainable water currents, they can be relevant physiologically and ecologically. Even if water velocities in available river habitats do not exceed the MSWV and thus do not submit fish to "wash-out", they should also fit in a current range which allows fish to feed and obtain a positive net gain of energy for growth or other activities, e.g. escaping from predators. In addition to the maximal sustainable water velocities, the nul benefit and the nul consumption water velocities should therefore be taken into account as criteria for ecological management from the point of view of fish habitats.

Table 2: Critical water velocities in larval and juvenile nase. Means ± standard deviations (n = 6).

mean TL (mm)	**nul benefit water velocity** (cm s^{-1})	**nul consumption water velocity** (cm s^{-1})	**maximal water velocity** (cm s^{-1})
15	6.6 ± 0.2	7.4 ± 1.3	8.1 ± 0.6
20	13.7 ± 0.0	14.3 ± 1.6	13.3 ± 0.4
25	14.0 ± 0.2	14.8 ± 1.8	16.8 ± 0.8
36	-	-	20.2 ± 5.6
46	-	-	23.2 ± 2.3
56	-	-	27.3 ± 4.2

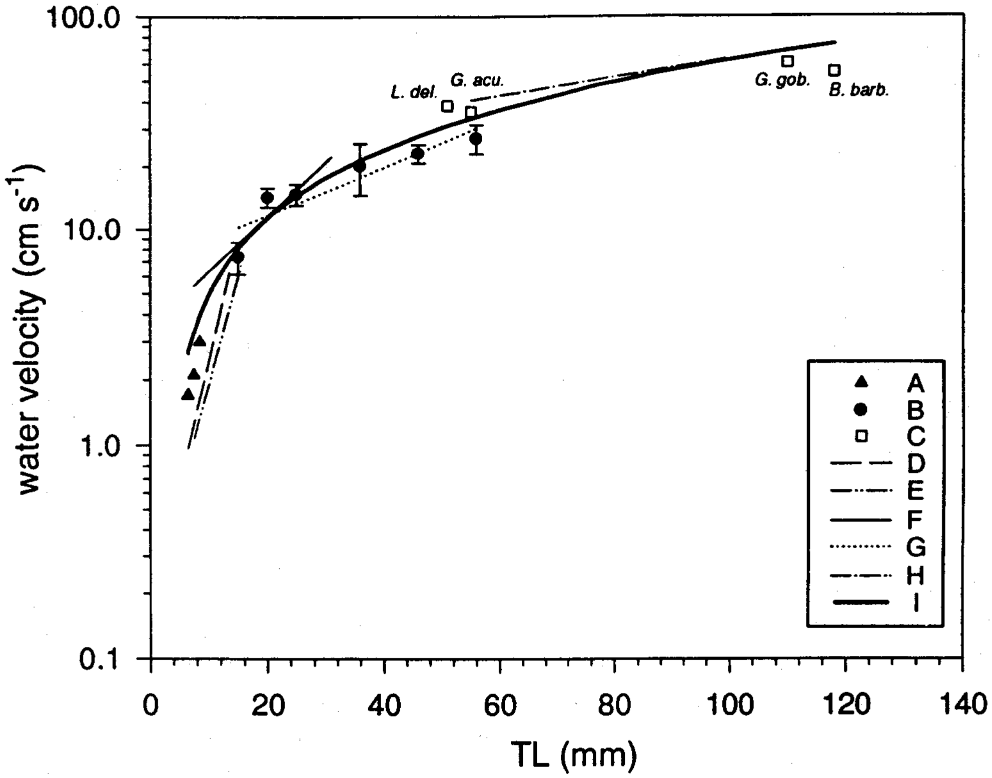


Figure 8: Comparison of the maximal sustainable water velocity of nase versus fish size, with maximal sustainable water velocities taken from the literature. A: *M. saxatilis* at 17 °C (Meng, 1993); B: *C. nasus* at 20 °C (present study; N = 30); C: at 18 °C (Stahlberg and Peckmann, 1987); D: *P. flavescens* at 13 °C ($y = -0.769 + 0.116\ x$, $r^2 = 0.772$) (Houde, 1969); E: *S. vitreum vitreum* at 13 °C ($y = -0.719 + 0.100\ x$, $r^2 = 0.712$) (Houde, 1969); F: *R. rutilus* at 20 °C ($y = 0.540 + 0.026\ x$, $r^2 = 0.964$) (Kaufmann, 1990); G: *C. nasus* ($y = 0.834 + 0.012\ x$, $r^2 = 0.831$); H: *O. nerka* at 15-20 °C ($y = 1.380 + 4.247\ x$, $r^2 = 0.885$) (Brett and Glass, 1973); I: overall regression ($y = -1.478 + 0.642$, $r^2 = 0.96$).

Maximal Sustainable Water Velocities

Better swimming performance increases feeding efficiency, escape response, and maintenance of station against a current (Beamish, 1978; Webb, 1984, 1986 in Young and Cech 1994). Figure 8 synthesises the maximal sustainable water velocities picked from the literature for several species and those we measured for the nase during ontogeny. Although the graph summarizes different species (e.g. salmonids, cyprinids, gasterosteids) with different life history strategies and different temperature preferences, it is surprising that a general trend between sustainable water current and fish size could be observed. Perhaps the shape of the fish body is more relevant in this respect than the species-specific life history patterns. The MSWV increases steeply between_10 and 20 mm fish length, and then the slope of the relationship diminishes. A linear increase of MSWV within each investigated size range of each species was observed. Our data on *Chondrostoma nasus* fit well into this general relationship (Figure 8). As far as smaller fish size classes - between 15 and 30 mm - are concerned, we obtained

]

very similar MSWV for the nase as for the roach (Kaufmann, 1990), although the former one is a rheophilic cyprinid while the latter is a limnophilic cyprinid. The MSWV of the bigger nase (55 mm TL) were slightly lower than those of *O. Nerka* given in Brett and Glass (1973). Maximal sustainable water velocities estimated from the laboratory experiments coincided with the water velocities at which C. *nasus* larvae were dislocated (wash-out) in the Danube downstream from Vienna (Schiemer et al., 1996).

Acknowledgments

We thank the fishery club of Fischamend, especially Mr. Helmuth Belanyecz, for the fishery allowance and the friendly collaboration. We gratefully acknowledge our colleagues who helped collect spawners and supported the technical laboratory installations. The competence of Mr. A. Hain allowed us to improve the quality of filming during our investigations. The electronic speed meter from the Marine Biology Department of the University of Vienna has also been a very important help. The comments on this paper by the reviewers are much appreciated. This study was conducted within the project P 9600 BIO financed by the Austrian Research Council.

REFERENCES

Addley, R. C. (1993). A mechanistic approach to modeling habitat needs of drift-feeding salmonids. Thesis for master of science, Utah State University Logan, 141 pp.

Bachman, R. A. (1982). A growth model for drift-feeding salmonids: a selective pressure for migration. Salmon and trout migratory behavior symposium, E.L. Brannon and E.O. Salo, editors.

Brett, J. R. (1964). The respiratory metabolism and swimming performance of young sockeye salmon. *J. Fish. Res. Bd. Canada* 21(5), pp. 1183-1227.

Brett, J. R. and Glass, N.R. 1973. Metabolic rates and critical swimming speeds of sockeye salmon (*Oncorhynchus nerka*) in relation to size and temperature. *J. Fish. Res. Board Can.* 30, pp. 379-397.

Chesney, E. J. Jr (1989). Estimating the food requirements of striped bass larvae *Morone saxatilis*: effects of light, turbidity and turbulence. *Mar. Ecol. Prog. Ser.* 53, pp. 191-200.

Copp, G. H. (1989). The habitat diversity and fish reproductive function of floodplain ecosystems. *Env. Biol. Fish.* 26, pp. 1-26.

Copp, G. H. and Peňáz, M. (1988). Ecology of fish spawning and nursery zones in the flood plain, using a new sampling approach. *Hydrobiologia* 169, pp. 209-224.

Cummins, K. W. and Wuycheck, J. C. (1971). Caloric Equivalents for Investigations in Ecological Energetics. Internationale Vereinigung für theoretische und angewandte Limnologie. *Mitt. Internat. Verein. Limnol.* 18, pp. 1-8.

Dabrowski, K; Takashima, F. and Law, Y. K. (1988). Bioenergetic model of planktivorous fish feeding, growth and metabolism: theoretical optimum swimming speed of fish larvae. *J. Fish Biol.* 32, pp. 443-458.

Dunbrack, R. L. and Dill, L. M. (1983). A model of size dependent surface feeding in a stream dwelling salmonid. *Environ. Biol. Fishes* 8, pp. 203-216.

Elliot, J. M. 1976.The energetics of feeding, metabolism and growth of brown trout (*Salmo trutta* L.) in relation to body weight, water temperature and ration size. *Journal of Animal Ecology* 45, pp. 923-948.

Facey, D. E. and Grossman, G. D. (1990). The metabolic cost of maintaining position for four North American stream fishes: effect of season and velocity. *Physiological Zoology* 63, pp. 757-776.

Fausch, K. D. (1983). Profitable stream positions for salmonids: relating growth rate to net energy gain. *Can. J. Zool.* 62, pp. 441-451.

Fricke, R.; Handermann, H.; Stahlberg S. and Peckmann, P. (1987). Zool. Jb. Physiol. 91: 101-111.

Gendron, R. P. and Staddon, J. E. R. (1983). Searching for cryptic prey: the effect of search rate. *Am. Nat.* 121(2), pp. 172-186.

Heath, M. R. (1993). The role of escape reactions in determining the size distribution of prey captured by hearring larvae. *Environmental Biology of Fishes* 38, pp. 331-344.

Heggenes, J. and Traaen, T. (1987). Downstream migration and critical water velocities in stream channels for fry of four salmonid species. *J. Fish Biol.* 32: 717-727.

Hill, J. and Grossman, G. (1993). An energetic model of microhabitat use for rainbow trout and rosyside dace. *Ecology* 74, pp. 685-698.

Houde, E. D. (1969). Sustained swimming ability of larvae of walleye (*Stizostedion vitreum vitreum*) and yellow perch (*Perca flavescens*). *J. Fish. Res. Bd. Canada* 26, pp. 1647-1659.

Hughes, N.F. and Dill, L.M. (1990). Position choice by drift-feeding salmonids: model test for Artic grayling (Thymallus arcticus) in subarctic mountain streams, interior Alaska. *Can. J. Fish. Aquat. Sci.* 47, pp. 2039-2048.

Irvine, J. R. (1984). Effects of varying discharge on the downstream movement of salmon fry, *Onchorhyncus tschawytscha* Walbaum. *J. Fish Biol.* 28, pp. 17-28.

Kamler, E (1992). Early life history of fish. An energetics approach. London: *Chapman & Hall*, 267 pp.

Kaufmann, R. (1990). Respiratory cost of swimming in larval and juvenile cyprinids. *J. exp. Biol.* 150, pp. 343-366.

Keckeis, H. and Schiemer,F. (1992). Food consumptions and growth of larvae and juveniles of three cyprinid species at different food levels. *Environmental Biology of Fishes* 33, pp. 33-45.

Keckeis, H.; Frankiewicz, P. and Schiemer, F (1996). The importance of inshore areas for spawning nase *Chondrostoma nasus* (Cyprinidae) in a free-flowing section of a large river (Danube, Austria). *Arch. Hydrobiol.* Suppl. 113 Large rivers 10, pp. 51-64.

Lelek, A. (1987). Threatened fishes of Europe. In: *The Freshwater Fishes of Europe*, vol. 9. Aula-Verlag Wiesbaden, 343 pp.

McLaughlin, R. L. and Grant, J. W. A. (1994). Morphological and behavioural differences among recently-emerged brook charr, *Salvelinus fontinalis*, foraging in slow- vs. fast-running water. *Environmental Biology of Fishes* 39, pp. 289-300.

Meng, L. (1993). Sustainable swimming speeds of striped bass larvae. *Transactions of the American Fisheries Society* 122, pp. 702-708.

Mittelbach, G. G. (1981). Foraging efficiency and body size: a study of optimal diet and habitat use by bluegills. *Ecology* 65(5), pp. 1370-1386.

Moyle, P. B. and Baltz, D. M. (1985). Microhabitat use by an assemblage of californian stream fishes: developing criteria for instream flow determinations 114, pp. 695-704.

Nyberg, D. N. (1971). Prey capture in the largemouth bass. *Amer. Midl. Nat.* 86, pp. 128-144.

Olivier, J.M.; Persat, H. and Boisson, C (1995). Long term variability in fish recruitment in three by-passed sections of the Upper Rhône River (France). First International Symposium"*The Ecology of Large Rivers*", Krems, Low Austria. Poster Presentation.

Ottaway, E. M and Clarke, A. (1981). A preliminary investigation into vulnerability of young trout (*Salmo trutta* L.) and Atlantic Salmon (*Salmo salar* L.). *Can. J. Fish. Aquat. Sci.* 44, pp. 120-129.

Ottaway, E.M. and Forrest, D.R. (1983). The influence of water velocity on the downstream movements of alevins and fry of brown trout. *J. Fish Biol.* 19, pp. 135-145.

Paffenhöfer, G-A (1967). Caloric content of larvae of the brine shrimp *Artemia salina*. *Helgoländer Wiss. Meeresunters* 16, pp. 130-135.

Peňáz, M (1974). Early development of the nase carp, *Chondrostoma nasus* (Linnaeus, 1758).*Zoologické Listy*, 23 (3), pp. 275-288.

Richman, S. (1958). The transformation of energy by *Daphnia pulex*. *Ecological Monographs* 28, pp. 273-291.

Schiemer, F.; Keckeis, H.; Winkler, G. and Flore, L. (in press). Large rivers: ecotonal structure and hydrological properties and their relevance for the fish fauna. *MAB UNESCO*, Partenon Press.

Schiemer, F. and Spindler, T. (1989). Endangered fish species of the Danube river in Austria. *Regulated rivers*: research and management 4, pp. 397-407.

Schiemer, F.; Spindler, T.; Wintersberger, H. Schneider, A. And Chovanec, A. (1991). Fish fry associations: Important indicators for the ecological status of large rivers. *Verh. Internat. Verein. Limnol.* 24, pp. 2497-2500.

Schoener, T. W. (1969). Models of optimal size for solitary predators. American Naturalist 103, pp. 277-313.

Shirvell, C. S. (1994). Effect of Changes in the Streamflow on the Microhabitat Use and Movements of Sympatric Juvenile Coho Salomon (*Oncorhynchus kisutch*) and Chinook Salmon *(O. tshawytscha)* in a Natural Stream. *Can. J. Fish. Aquat. Sci.* 51, pp. 1644-1652.

Stahlberg, S. and Peckmann, P. (1987). The critical swimming speed of small Teleost fish species in a flume. *Arch. Hydrobiol.* 110 (2), pp. 179-193.

Spindler, T. (1998). Ökologie der Brutfische in der Donau bei Wien. Dissertation Univ. Wien., 119 pp.

Ware, D. M. (1975). Growth, metabolism, and optimal swimming speed of a pelagic fish. *J. Fish. Res. Board Can.* 32, pp. 33-41.

Wanzenböck, J. and Schiemer, F. (1989). Prey detection in cyprinids during early development. *Can. J. Fish. Aquat.* Sci. 46, pp. 995-1001.

Wanzenböck, J. (1992). Ontogeny of prey attack behaviour in larvae and juveniles of three European cyprinids. *Environmental Biology of Fishes* 33, pp. 23-32.

Wilzbach, M. A.; Cummins, K. W. and Hall, J. D. (1986). Influence of habitat manipulations on interactions between cutthroat trout and invertebrate drift. *Ecology* 76(4), pp. 898-911.

Winfield, I. J.; Peirson, G.; Cryer, M. and Townsend C. R. (1983). The behavioural basis of prey selection by underlyerling bream (*Abramis brama* (L.)) and roach (*Rutilus rutilus* (L.)). *Freshwater Biology* 13, pp. 139-149.

Young P. S. and Cech J. J, Jr (1994). Optimum exercise conditioning velocity for growth, muscular development, and swimming performance in young-of-the-year striped bass (*Morone saxatilis*). *Can. J. Fish. Aquat. Sci.* 51, pp. 1519-1527.

Zerrath, H. (1996). Sprintleistungen einheimischer Klein- und Jungfische in Sohlgleitenmodellen- Daten zur Bewertung von Fischaufstiegshilfen. *Fischökologie*, pp. 27-48.

A MULTIVARIATE MODEL TO RELATE HYDROLOGICAL, CHEMICAL AND BIOLOGICAL VARIABLES TO SALMONID STANDING CROP IN ITALIAN ALPINE RIVERS

P. Annoni[(1)], I. Saccardo[(2)], G. Gentili[(3)], L.Guzzi[(4)]
(1) CISE S.p.A. - via Reggio Emilia 39, Segrate (MI) - ITALY - E-mail: 1132anno@s1.cise.it
(2) ENEL S.p.A. - DSR CRIS, C.so del Popolo 245, Mestre (VE) - ITALY - E-mail: saccardo@cris.enel.it
(3) GRAIA S.r.l - via Repubblica 2, Varano Borghi (VA) -ITALY - E-mail: graia@interland.it
(4) ENEL S.p.A. - DSR CRAM, via Rubattino 54, Milano - ITALY - E-mail: guzzi@cram.enel.it

ABSTRACT

A multivariate statistical analysis was performed to assess relationships between a group of habitat variables (hydrological, physical, chemical and biological) and the variation of trout biomass in thirteen stations along alpine rivers. We analyzed six different rivers located in the northern and eastern part of Italian Alps.

Various river data were collected both from historical records and experimental surveys carried out over a period of two years in late summer. We collected information about river basin features, channel morphometry, river hydrology, habitat structure, physical, chemical and biological characteristics. Salmonid population was estimated by electrofishing.

Principal component analysis was useful to sort out all relevant variables. After variable selection, we calculated the best regression model using two statistics: Mallows' Cp and R^2 adjusted for degrees of freedom. Selection method consisted in calculating all possible regressions from a subset of input variables. The best model turned out to be the one with six variables: reach elevation, reach mean gradient, channel width, mean annual flow, benthic invertebrate abundance and diversity. The model explained 89% of variation in fish abundance expressed as biomass per unit area of stream (kg/ha).

Two habitat variables, reach elevation and invertebrate abundance, explained 79% of variation in trout biomass (variables transformed in natural logarithms) and invertebrate abundance alone explained 77% of response variance. Various statistical indices were calculated to evaluate model quality and robustness. In addition, for each variable included in the selected multiple regression we calculated the regression coefficient significance level, the variance inflation coefficient (VIF) and the partial correlation coefficient with fish biomass.

In addition to statistical tests we conducted a sensitivity analysis to determine how much the model slope would change due to independent variable measurement errors. Two different analyses were carried out: the former used a theoretical formula to estimate model parameter bias; the latter was based on automatic calculation of thousand different regressions using data matrices, modified according to variable measurement errors. It was then possible to calculate arithmetic mean and standard deviation of model coefficients to obtain an estimation of effects of measurement errors on the model. These effects turned out to be comparable to errors in parameters estimates due to ordinary least squares statistics.

The study demonstrated that it is possible to simplify habitat quality evaluation using a subset of environmental variables and it should be particularly useful in river management issues. The variable mostly correlated to fish population turned out to be benthic invertebrate abundance that, somehow, summarizes many relevant fish habitat characteristics.

An analysis to verify model predictive value is still in progress.

KEY-WORDS: river habitat / stream fish / trout biomass / multivariate statistics / multiple regression / sensitivity analysis

INTRODUCTION

As river habitat protection has been getting more and more important in recent years, we applied a series of models to estimate Italian river habitat value and its modifications due to water resource management. In the last decades many methods have been developed to assess flow requirements at various level of complexity. They range from purely hydrological methods, such as Montana (Tennant, 1975) or Baxter (1961) methods, that use data related only to river discharge, to methods that integrate various river characteristics, such as the Instream Flow Incremental Methodology (Bovee, 1982), that uses both hydraulic and biological data. Furthermore there are models to predict fish abundance that, developed for specific purposes, have been recently employed also to assess the impact of changing flow on fish population.

Fish quantification is a very useful tool to convert a river habitat value into an economical value, that could be helpful in decision making sessions about conflicting water uses. For this reason we developed a model to predict salmonid abundance (biomass per unit area, kg/ha) in Italian mountain rivers starting form measurable characteristics of the environment. In particular, our attention has been focused on getting a precise and simple regression model. In other words, we tried to develop a model that explained as much biomass variation as possible and used, in the meantime, a selection of all relevant variables based on time and money constrains. We followed Habitat Quality Index (HQI) procedure 'philosophy', that was developed by Binns and Eiserman (1979). HQI author's interesting ideas were to use input variables of several type, from basin characteristics to chemical and biological factors, and to apply statistical criteria to obtain a reasonable regression model.

METHODS

Study Sites And Input Variables

Trout habitat was measured in thirteen stream reaches belonging to six different rivers along the Alps (northern Italy): two of them are located in the eastern part of the Alps, the others in the central part of alpine chain.

Selected stream reaches range from an altitude of 150 m a.s.l. up to 1850 m a.s.l. and are subjected to different levels of human exploitation . Some of them are located in almost undisturbed areas (no villages, no hydraulic structures, no fishing activities, limited land use), while others are highly modified by water and land uses. The length of study stations is of the order of magnitude of 100 m, varying from 50 to 300 m depending on the amount of stream needed to adequately sample available habitat.

Input variable selection was carried out at two levels that could be addressed to as qualitative and quantitative approaches. From the qualitative point of view, we used literature and experts indications to identify variables closely linked to fish abundance (Baran *et al.*, 1993; Binns and Eiserman, 1979; Fausch *et al.*, 1988; Jowett, 1992). After this selection, we applied multivariate statistical techniques to choose a subset of variables particularly suitable in describing Italian trout biomass variation.

River habitat variables can be divided into various categories depending on the scale of measurement and river characteristic typology they describe. In particular, it is possible to identify six different variable types (Fausch *et al.*, 1988): 1. river basin variables (such as mean altitude or catchment area); 2. morphological variables (such as water level slope or reach width); 3. hydrological variables (such as mean annual flow or water depth); 4. habitat structure variables (such as cover or eroding banks percentage); 5. biological variables (such as invertebrate abundance or velocity preference factors); 6. physical and chemical variables (such as water temperature or dissolved oxygen concentration).

In our model we wanted to include at least one variable belonging to each of the above categories in order to describe river habitat from the micro to the macro scale. In the meantime we had to limit the number of input variables due to relatively small sample size (thirteen measurement stations), to maintain a reasonable number of degrees of freedom.

Another important consideration we kept in mind in choosing input variables was the ease and the expense of measuring habitat variables, since many river characteristics are generally difficult or costly to evaluate.

All these needs determined the following input variable selection (table 1): 1) stream reach altitude (m a.s.l.); 2) water level slope (%); 3) stream reach width (m); 4) mean annual flow, MAF (m^3/s); 5) discharge that in the average is equalled or exceeded 95% of time, $Q_{95\%}$ (m^3/s); 6) stream flow variation index, calculated as the ratio between peak and low flow, SFV; 7) cover percentage (%); 8) macro-invertebrate abundance (number/m^2); 9) macro-invertebrate diversity; 10) maximum water temperature (°C); 11) nitrate concentration, NO3 (mg/l).

Table 1: Input variable initial set

	SELECTED INPUT VARIABLES												
station index	**S1**	**S2**	**S3**	**S4**	**S5**	**S6**	**S7**	**S8**	**S9**	**S10**	**S11**	**S12**	**S13**
altitude (m a.s.l.)	156	156	156	270	270	270	380	380	340	250	300	1850	1760
slope (%)	0.20	0.40	1.50	0.20	0.40	0.65	0.80	1.00	0.30	0.85	3.90	1.35	4.80
width (m)	45	25	17	11	16	19	10	9	27	18	10	4	7
MAF (m^3/s)	6.9	6.9	6.9	9.0	5.7	5.7	1.9	1.8	4.1	19.2	6.8	0.8	2.7
$Q_{95\%}$ (m^3/s)	1.9	1.9	1.9	1.6	1.6	1.6	0.5	0.5	1.3	5.5	0.6	0.1	0.2
SFV	11	11	11	11	11	11	11	16	12	10	45	17	48
cover (%)	60	30	30	50	10	30	15	50	12	30	45	20	25
invertebrate abundance	229	185	196	77	72	115	94	9	263	570	350	65	1
invertebrate diversity	3.80	6.14	2.78	1.43	2.24	1.86	2.13	0.73	2.75	1.88	1.79	1.20	0.54
maximum water temperature (°C)	18.6	18.6	18.6	15.8	15.8	15.8	13.0	14.0	14.0	11.7	16.0	7.9	10.5
nitrate (mg/l)	0.89	0.89	0.89	0.83	0.83	0.83	0.85	0.84	0.84	3.05	3.70	1.26	0.82
fish biomass (kg/ha)	38	26	31	39	24	29	32	20	22	55	40	19	0

Data Sampling

Experimental campaigns were carried out over a period of two years during summer months only (July and August). Reach altitude was read on topographical maps, whereas mean water level slope and reach width were determined in the field by means of a geodimeter. Hydrological variables (MAF, $Q_{95\%}$ and SFV) were estimated from historical records of river flow. Cover percentage was judged visually by three different expert operators; note that, even if we tried to be as precise as possible, this measuring method is intrinsically affected by large errors (Hogle *et al.*, 1991). Macro-invertebrate population was sampled to get abundance and diversity following the Habitat Quality Index procedure (Binns, 1978). Note that in this case variable diversity is different from Shannon index (Shannon, 1949). Temperature values were either calculated from historical records, where available, or measured during experimental surveys. Nitrate concentration was determined by analysing water samples collected in the field.

Salmonid biomass was estimated by electrofishing, repeating the sampling procedure three times for each station. Collected data were elaborated following Zippin's method (Zippin, 1958). Trout species (*Salmo trutta fario, Salmo trutta marmoratus, Salmo trutta fario x marmoratus,* and *Thymallus thymallus*) and size-class were recorded. Every age classes were considered in the analysis in order to estimate total fish biomass actually present in the stream reach.

Multivariate Statistical Techniques

Small sample size was the first problem we had to cope with. In fact we selected eleven variables for a total of thirteen observations, which resulted in small degrees of freedom. We then utilized principal component analysis to test the possibility of discarding some of initial input variables.

To obtain the best regression model we developed a computer code to calculate all possible models by using all subset regressions and chose the best one using two criterions: R^2 adjusted and Mallows' Cp statistics (Helsel and Hirsch, 1992). The former is the fraction of variance explained by the model (coefficient of determination) adjusted for degrees of freedom in the model (R^2_a). The latter is designed to obtain a regression model that explains as much variance of response variable as possible, by including all relevant variables, and that minimizes the variance of resulting estimates, by keeping degrees of freedom large. The best model was then chosen by maximizing R^2_a and minimizing Mallows' Cp. In case of similar value of R^2_a and Cp, the model with the least number of variables or that includes the less expensive variables should be chosen.

Once we chose the regression model, we tested its quality verifying regression coefficient significance, residual normality and heteroscedasticity. In a multiple linear regression it is also important to measure multi-collinearity of explanatory variables. In fact multi-collinearity results in many undesirable consequences in regression equations, such as instability of slope coefficients and unrealistic signs of coefficients. To measure multi-collinearity we used the variance inflation factor VIF_j that is related to the confidence interval width of j-th regression coefficient (Weisberg, 1985).

Moreover input variables were classified depending on their role in the model and their partial correlation coefficient with salmonid biomass (Steel and Torrie, 1981).

Observed and predicted response values were compared fitting a regression line and testing whether the slope differs significantly from one.

After selecting and testing the regression model, we felt the need to determine how much slope coefficients would change if the habitat variables were changed by an amount reflecting measurement errors. This is a prime issue in models of this kind because river habitat variables can be highly affected by measurement errors.

To conduct the sensitivity analysis we followed two different approaches. The first one was based on theoretical considerations (Weisberg, 1985). As known, one assumption of least squares estimator is that predictors are fixed values measured without errors. When errors do occur in the predictors least square regression is not strictly correct. Yet it is possible to calculate a diagnostic to decide if calculated fitting is seriously different from the 'ideal' one, that is the regression with predictors not affected by measurement errors.

If **X** is the observed matrix of predictors, affected by measurement errors, and **X*** is the 'true value' matrix; then **X=X*+D** where **D** is a matrix of errors. Hodges and Moore (1972) showed that:

$$E(\beta\hat{}) - \beta \propto (\mathbf{X}^T\mathbf{X})^{-1}\mathbf{S}\beta$$

where $E(\beta\hat{})$ is the expected value of the regression coefficient calculated from the **X** matrix, β is the regression coefficient from 'true value' matrix **X*** and **S** is the matrix of measurement error variances. Thus fitting a regression model with input variables affected by measurement errors gives biased estimates of slope coefficients. Clearly $(\mathbf{X}^T\mathbf{X})^{-1}\mathbf{S}\beta$ cannot be precisely calculated because the value of β is unknown. Yet it is possible to give an estimation of its value expanding $\beta\hat{}$ in a Taylor series about **D**=0 and discarding higher order terms. Hence we were able to calculate a first-order estimate of regression coefficient bias due to measurement errors.

The second approach we followed in the sensitivity analysis was a pragmatic one. We calculated one thousand multiple regression using one thousand different data matrixes obtained changing input variable values by a randomly chosen amount within measurement error range. We then calculated arithmetic mean and standard deviation of regression coefficients to quantify the variability of regression results caused by measurement errors.

All statistical calculations were implemented in FORTRAN language using IMSL libraries (IMSL, 1991).

RESULTS

As mentioned above, the first problem we had to cope with was the limited number of degrees of freedom. We then searched for useless variables applying principal component analysis to standardized input data. The first two principal components explained about 66% of variance. From figure 1, that shows first two component weights, it is clear that in this case cover percentage did not contribute in explaining data variability (low weight value for both first and second component); we then discarded it from further analyses.

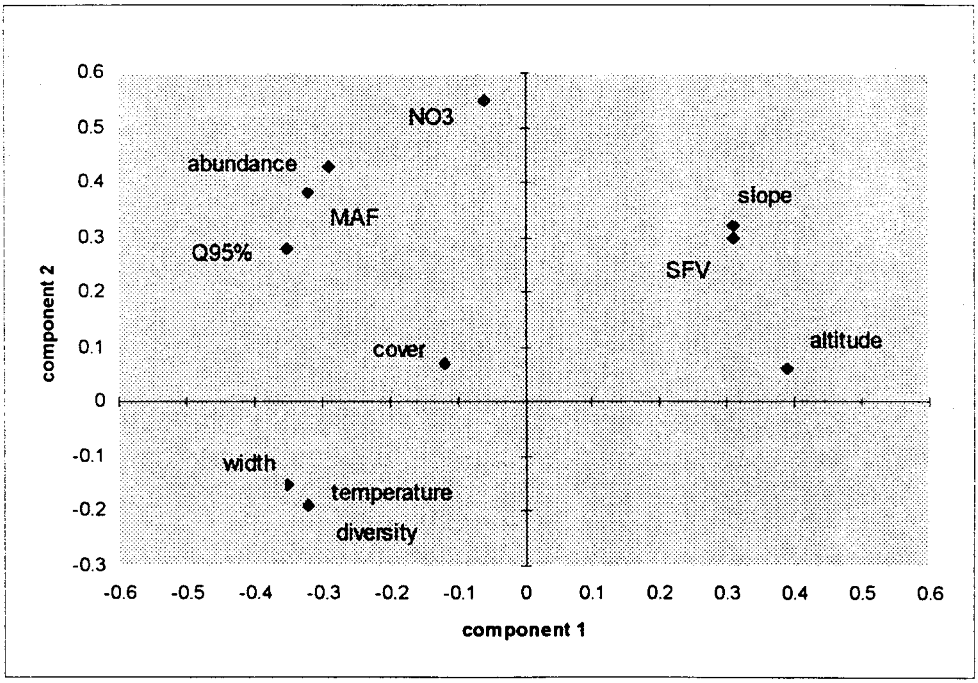


Figure 1: Plot of two first component weights from principal component analysis on untransformed variables

To identify the best multiple regression, we calculated R^2_a and Mallows' Cp indexes for all possible multiple regressions from a subset of ten input variables (altitude, slope, MAF, $Q_{95\%}$, SFV, invertebrate abundance and diversity, NO3). In the overall we then automatically calculated 2^{10} regressions, including the constant model. Results of this analysis are reported in table 2 and figure 2, where it is possible to compare R^2_a and Cp values of the first two best regressions for a given number of input variables.

The best model was regression number 17 with nine variables: altitude, slope, width, MAF, $Q_{95\%}$, SFV, invertebrate abundance, water temperature and nitrates. It had nevertheless several undesirable features: first of all it included a great number of variables, resulting in a model costly to be applied and, in the specific case, with small degrees of freedom. Moreover regression coefficients were not always significantly different from zero and variance inflation factors were generally very high (for all included variables VIF is greater than 10 with a value as high as 59). It resulted in a regression model that was unstable and not statistically significant.

Table 2: R^2_a and Mallows' Cp for several multiple regressions based on ten untransformed variables

reg. index	n. of variables	R^2_a	Cp	altitude	slope	width	MAF	$Q_{95\%}$	SFV	abundance	diversity	temperature	NO3
1	1	56.7	173				X						
2	1	53.6	186							X			
3	2	66.5	121	X			X						
4	2	60.2	145		X		X						
5	3	71	94.8	X			X						X
6	3	68.9	102				X	X	X				
7	4	82	51.9	X			X		X				X
8	4	80.4	56.9	X	X		X						X
9	5	88.3	30.3			X	X	X	X				X
10	5	87.3	33.1	X			X	X	X				X
11	6	89.5	25		X	X	X	X	X				X
12	6	87	30.9	X		X	X	X	X				X
13	7	90.4	21.4		X	X	X	X	X		X		X
14	7	88.1	25.8		X	X	X	X	X			X	X
15	8	94.2	13.9	X	X	X	X	X	X			X	X
16	8	89.1	21.7	X	X	X	X	X	X		X		X
17	9	96.6	10.9	X	X	X	X	X	X	X		X	X
18	9	92.2	15.9	X	X	X	X	X	X		X	X	X
19	10	97.4	11	X	X	X	X	X	X	X	X	X	X

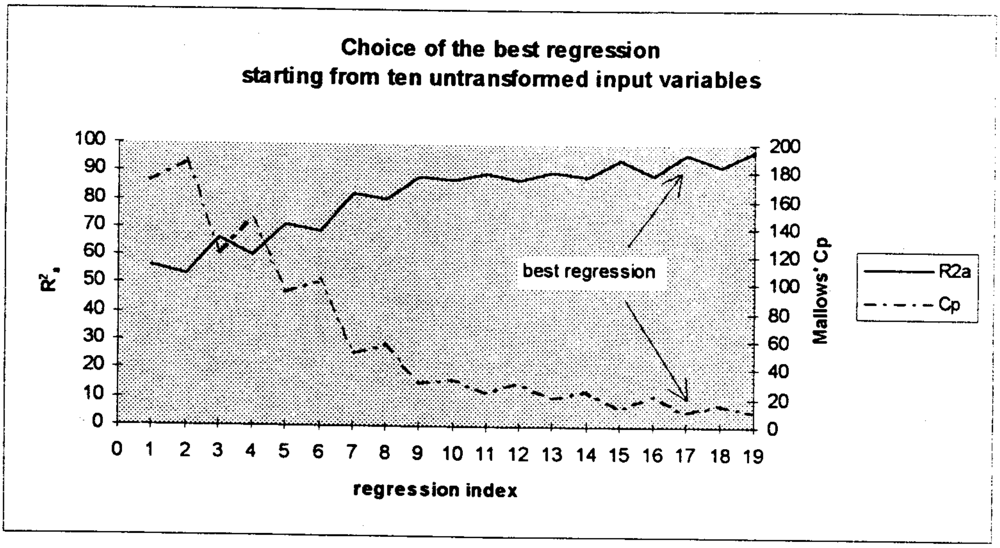


Figure 2: R^2_a and Mallows' Cp trend for several multiple regressions based on ten untransformed variables

To calculate a better regression model, we used log-transformed input variables following various literature indications (Baran *et al.*, 1993; Binns and Eiserman, 1979; Fausch *et al.*, 1988; Jowett, 1992). For fish biomass, we adopted a slightly different transformation to adequately consider stations without fishes. In fact in one of the highest stream reach no fishes were found and so sampled biomass was exactly zero. To avoid mathematical non-sense and according to many literature studies, it was necessary to transform biomass in the following way: log (biomass+1). This transformation had also the desirable advantage to avoid assigning unrealistic weight to small biomass values.

Principal component analysis on transformed data showed two variables, cover percentage and nitrates, that did not contribute to define the first component, that explained almost 55% of total variance (figure 3). Furthermore we applied R^2_a and Mallows' Cp indices to select the best model from the first six principal components and found that the best one was the multiple regression that excluded component number 2. Since cover percentage and nitrate concentration contributed mainly to the second component, we decided to exclude these variables from successive analyses.

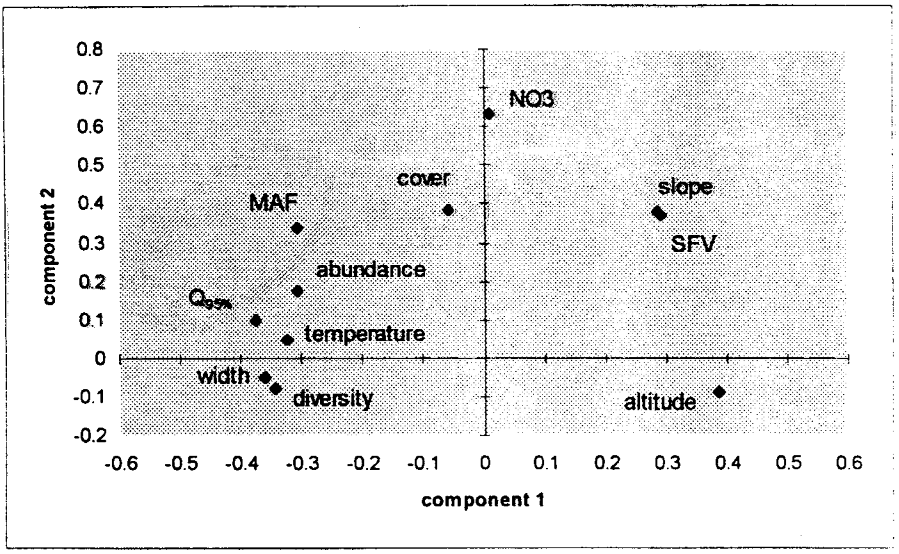


Figure 3: Principal component analysis on log-transformed variables

We then calculated all possible subset regressions using nine log-transformed variables (all initial variables but cover and nitrates) for a total of 2^9 regressions. Table 3 reports R^2_a and Mallows' Cp statistics for the best two regressions calculated for a given number of input variables, from the simplest model that included only one variable to the most complex that included nine variables.

Table 3: R^2_a and Cp statistics for several multiple regressions based on nine log-transformed variables

reg. index	n. of variables	R^2_a	Cp	altitude	slope	width	MAF	$Q_{95\%}$	SFV	abundance	diversity	temperature
1	1	74.4	52.9							X		
2	1	44.7	125	X								
3	2	77.2	43						X	X		
4	2	75.1	47.7		X					X		
5	3	80.6	33.4						X	X	X	
6	3	79.6	35.4			X			X	X		
7	4	83.9	25.3						X	X	X	X
8	4	83.6	25.8	X					X	X	X	
9	5	92.7	10.2	X	X	X				X	X	
10	5	89.1	15.7					X	X	X	X	X
11	6	96.5	5.66	X	X	X	X			X	X	
12	6	94.3	8.54	X	X			X		X	X	X
13	7	97	6.33	X	X	X		X		X	X	X
14	7	96.4	6.92	X	X	X	X			X	X	X
15	8	96.6	8.03	X	X	X		X	X	X	X	X
16	8	96.4	8.18	X	X	X	X	X		X	X	X
17	9	95.5	10	X	X	X	X	X	X	X	X	X

From figure 4 it was easy to select the best model: the eleventh model was clearly the most efficient one both in terms of explained variance and degrees of freedom. This model included six log-transformed variables (seven degrees of freedom) and explained 98% of log-transformed fish biomass. Selected predicting variables were: altitude, slope, reach width, mean annual flow (MAF), macro-invertebrate abundance and diversity. The best multiple regression was then based on a relatively well balanced mixing of river habitat variables since it included one variable related to basin characteristics (reach altitude), two morphological variables (water slope and stream width), one hydrological variable (mean annual discharge) and two biological variables (abundance and diversity of macro-invertebrate).

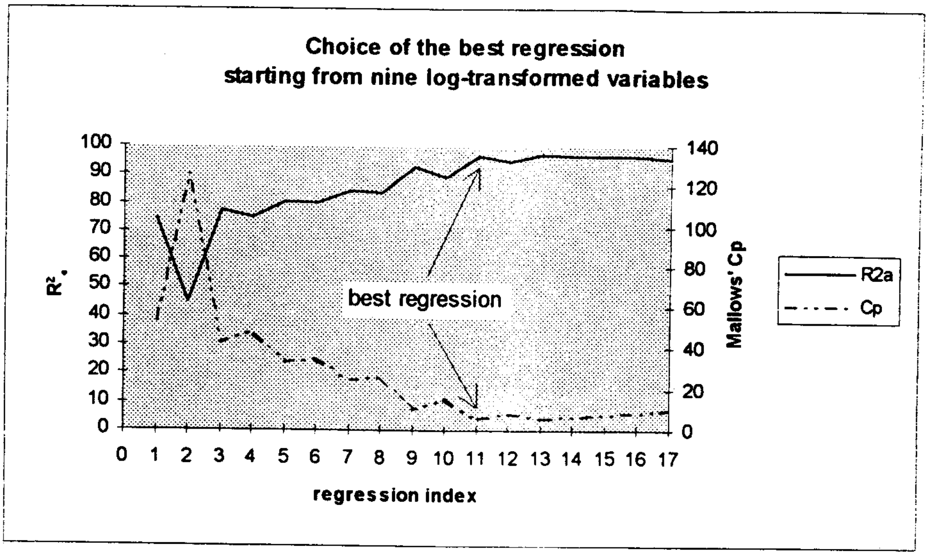


Figure 4: R^2_a and Cp trend for several multiple regressions based on nine log-transformed variables

To verify model quality, we visually checked for residual normality and homoscedasticity; in addition we calculated regression coefficient significance levels and variance inflation factors. In table 4 are reported various statistical indexes to diagnose selected multiple regression. Slope coefficients are all significant at $\alpha=0.05$ and multi-collinearity is not an issue since variance inflation factors were always lower than six. In fact, according to literature indications (Helsel and Hirsch, 1992), serious problems can come up if $VIF_j > 10$. Regression model quality was then rather satisfactory.

Table 4: selected regression diagnostic

input variable	regression coeff.	standard error	t-value	sig.level	VIF
constant	8.478	1.1490	7.3783	0.0003	
altitude	-0.921	0.1405	-6.5557	0.0006	4.348
slope	-0.280	0.0724	-3.8694	0.0083	1.818
width	-0.613	0.1890	-3.2429	0.0176	4.762
MAF	-0.319	0.1097	-2.9043	0.0272	2.941
abundance	0.565	0.0580	9.7542	0.0001	3.226
diversity	-0.680	0.1960	-3.47	0.0133	5.556

From t-values, that indicate input variable significance in explaining fish biomass, it resulted that macro-invertebrate abundance was by far the most important variable. It explained 77% of total fish biomass variability and so macro-invertebrate abundance turned out to be the variable mostly correlated to salmonid population. This variable was actually expected to be strictly related to fish community because it summarizes many relevant fish habitat characteristics that are generally difficult to estimate. We did not analyze in detail the correlation between benthic invertebrate abundance and river habitat, but we suggest that aquatic macrobenthos could be actually related to flow variability, suspended solid concentration and other important factors that characterize fluvial environment. In regulated stream reaches these factors are a prime issue for aquatic population and macro-invertebrate community faithfully reflects the overall conditions of the river habitat.

Table 5 reports input variables ordered by their estimated importance in explaining fish biomass; the ordering method was based on their t-values and partial correlation coefficients with biomass (in this case critical value of partial correlation coefficient at $\alpha=0.05$ is 0.71). Last column of table 5 indicates the percent variation R^2 explained by stepwise inclusion of variables.

Table 5: input variables ordered by their estimated significance in the regression

input variable	significance	partial correlatio	R^2
abundance	9.75	0.97	0.77
altitude	6.55	-0.94	0.79
slope	3.87	-0.85	0.80
diversity	3.47	-0.82	0.88
width	3.24	-0.80	0.96
MAF	2.90	-0.76	0.98

Figure 5 shows the relationship between calculated and experimental biomass values expressed in the original measurement units, kg/ha (no log-transformation). The coefficient of determination calculated for biomass in the original measurement units dropped to 89%, but it was still satisfactory. Calculated slope of the regression line was 0.912 with a standard error of 0.097. Applying t-statistics, it resulted a regression slope not significantly different from one.

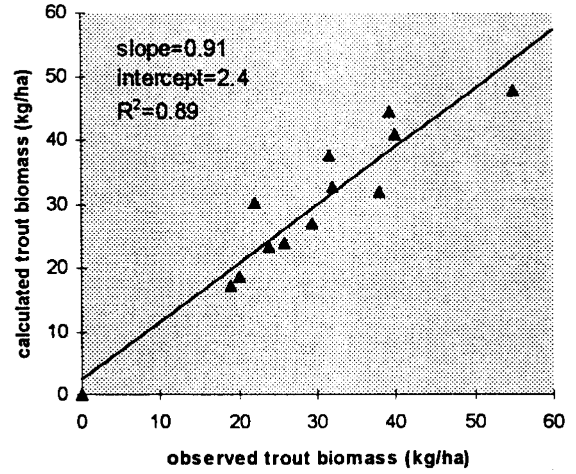


Figure 5: Relationship between observed and calculated trout biomass expressed in kg/ha and fitted regression line

Sensitivity Analysis

A major problem in models of this kind is that least square linear estimators assume that predicting variables are measured without errors (Weisberg, 1985). Since this hypothesis is generally violated when river habitat is measured, it is important to consider variance in input variables when regression coefficients are estimated, because they could be biased (Fausch *et al.*, 1988).

In the present study, input variables were assumed to be affected by measurement errors as a percentage of observed values. Once log-transformed, these errors became additive errors and so we assigned additive measurement errors ξ to log-transformed variables. The choice of error ranges was based both on general considerations and measurement crew indications. In doing that we tried to be conservative, that is we assigned errors possibly greater than actual ones.

We assumed that, for every variable j, each observation x_{ij} was affected by an error within the interval $[-\xi_j , +\xi_j]$ and that all the possible outcomes were equally likely to be the true one. That is equivalent to assume that measurement error of variable j was uniformly distributed on the interval $[-\xi_j , +\xi_j]$. It was so possible to calculate error variances, var($\mathbf{d}_j$) and, consequently, to estimate the regression bias due to measurement errors.

Table 6 reports maximum errors ξ that were assumed to affect input variables and their variances. Note that ξ values approximatly reflect errors on predicting untransformed variables, that is, for example, ξ=0.2 means 20% of measurement error on original predicting variables. Reach altitude error was put equal to zero because its value was read on topographical maps, whereas macro-invertebrate abundance and diversity were assumed to be the most uncertain variables because of macro-invertebrate sampling uncertainties.

Table 6: maximum measurement errors and error variances assigned to log-transformed predicting variables

regression input variable	**maximum measurement error - ξ_j**	**error variance - var($\mathbf{d}_j$)**
altitude	0.0	0.000
slope	0.2	0.013
width	0.1	0.003
MAF	0.2	0.013
abundance	0.3	0.030
diversity	0.3	0.030

Bias estimates (table 7) were calculated both in terms of absolute value and percentage relative to regression coefficients. Bias values were quite small, apart from the one linked to macro-invertebrate diversity that was affected by a bias of almost 23%.

Table 7: regression coefficient bias estimates

regression input variable	**bias - $\|E(\beta\text{^})-\beta\|_j$**	**bias as percentage - $\|(E(\beta\text{^})-\beta)/\beta\text{^}\|_j$ * 100**
altitude	0.0288	3.1
slope	0.0023	0.8
width	0.0738	12.0
MAF	0.0436	13.7
abundance	0.0360	6.4
diversity	0.1559	22.9

To verify these results, we developed a computer code to automatically calculate thousand different regressions using data matrices modified according to variable measurement errors. For every variable j, each observation x_{ij} was randomly sort out from an uniform population centred around the actually observed value with width $2\xi_j$. Table 8 reports arithmetic mean and standard deviation of calculated regression coefficients; to make the comparison easier the same table shows also the statistics (regression coefficient estimate and standard error) of the regression model from the observed data matrix.

Table 8: estimates of regression coefficient variability

	from 1000 different regressions		*from observed data matrix*		
input variable	**arithmetic mean - b_m**	**standard deviation**	**coefficient - b**	**standard error**	**$\lvert(b-b_m)/b\rvert*100$**
constant	8.491	0.9207	8.478	1.1490	0.2
altitude	-0.900	0.1118	-0.921	0.1405	2.3
slope	-0.278	0.0603	-0.280	0.0724	0.7
width	-0.662	0.1611	-0.613	0.1890	7.9
MAF	-0.287	0.0947	-0.319	0.1097	9.9
abundance	0.537	0.0475	0.565	0.0580	5.0
diversity	-0.564	0.1647	-0.680	0.1960	17.1

These results confirmed indications of previous sensitivity analysis because, in this case too, the greatest variability of slope coefficients was the one related to macro-invertebrate diversity. Note that standard deviations of the population of regression coefficients were always lower than standard errors of model coefficients, calculated from least square statistics. That was an encouraging result since it meant that errors due to measurement errors did not sensibly affect the model.

CONCLUSIONS

Starting from a set of river habitat variables we developed a multiple regression model that explained almost 90% of trout biomass variability. Nevertheless we suspect that model precision could be due to the limited sample size and period of time considered. In fact according to Fausch's analysis (1988), among models developed from data sets of more than 20 observations only 46% accounted for 75% or more of fish biomass variation. On the contrary, 68% of models calculated from fewer than 20 samples had coefficients of determination of 75% or more. Fausch's conclusion was that most precise models were related to relatively small geographic areas and/or short period of time, indicating that precise models could often lack generality. Since the model we developed was based on a relatively small sample set (thirteen observations), we intend to verify its predicting value in different seasons and areas. For this reason we are keeping on collecting experimental data along other alpine rivers to assess model predicting value.

Yet the present study was an attempt to apply statistical procedures to reveal the most important habitat variables linked to fish population in Italian alpine rivers. Selected model was based on six habitat variables belonging to different river characteristics. The analysis revealed that macro-invertebrate abundance was a key variable in describing salmonid biomass variability. Actually, macro-invertebrate characteristics (abundance and diversity) are indicative of river habitat overall quality, especially in case of regulated rivers where regulation effects, such as hydro-peaking or suspended sediment concentration variability, can highly influence aquatic community. In these habitats it is generally difficult to describe such limiting conditions using average stream reach characteristics. On the other hand, it is complex to accurately measure and quantify river habitat variability since it often occurs very rapidly. A method to overcome this difficulty is to characterize macro-invertebrate population that faithfully reflects most of past limiting conditions. Including this variable it was possible to explain most of salmonid biomass variance in some Italian alpine rivers.

REFERENCES

Baran, P., M. Delacoste, J.M. Lascaux and A. Belaud (1993). Relations entre les caractéristiques de l'habitat et les populations de truites communes (*Salmo Trutta* L.) de la vallée de la neste d'aure. *Bull. Fr. Pêche Piscic.*. 331 pp. 321-340.

Baxter, G. (1961). River utilization and preservation of migratory fish life. *Proc. Inst. Civ. Eng.* 18-255-44.

Binns, N.A. and F.M. Eiserman (1979). Quantification of Fluvial Trout Habitat in Wyoming. *Transaction of the American Fisheries Society.* 108 pp. 215-228.

Bovee, K.D. (1982). A guide to stream habitat analysis using the Instream Flow Incremental Methodology: Instream Flow Information Paper n° 12. *FWS/OBS 82/86, U.S. Fish and Wildlife Service - Fort Collins - CO.*

Fausch, K.D., C.L. Hawkes and M.G. Parsons (1988). Models That Predict Standing Crop of Stream Fish from Habitat Variables: 1950-85. *Gentile. Tech. Rep. PNW-GTR-213. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.* 52 p.

Helsel, D.R. and R.M. Hirsch (1992). Statistical Methods in Water Resources. *Elsevier Science Publishers B.V.* 522 p.

Hodges, S.D. and P.G. Moore (1972). Data uncertainties and least squares regression. *Appl. Statist.* 21 pp. 185-195.

Hogle, J.S., T.A. Wesche and W.A. Hubert (1991). A Test of the Precision of the Habitat Quality Index Model II. *North American Journal of Fisheries Management.* 12 pp. 417-432.

IMSL (1991). IMSL FORTRAN Libraries Ver. 2.0.

Jowett, I.G. (1992). Models of the Abundance of Large Brown Trout in New Zealand Rivers. *North American Journal of Fisheries Management.* 12 pp. 417-432.

Shannon, C.E. and W. Weaver (1949). The mathematical theory of communication. *University of Illinois Press Urbana. IL.*

Steel, R.G.D. and J.H. Torrie (1981). Principles and Procedures of Statistics A Biometrical Approach. *International Student Edition.* 633 p.

Tennant, D.L. (1975). Instream Flow Regimes for Fishes. *Recreation and Related Environmental Resources. U.S. Fish and Wildlife Service, Montana.* 30 p.

Zippin, C. (1958). The removal method of population estimation. *Journal of Wildlife Management.* 22 82-90.

Weisberg, S. (1985). Applied Linear Regression. *John Wiley & Sons.* 323 p.

COMPARATIVE EVALUATION OF SUITABILITY CURVE BASED HABITAT MODELING AND A MECHANISTIC BASED BIOENERGETIC MODEL USING 2-DIMENSIONAL HYDRAULIC SIMULATIONS IN A NATURAL RIVER SYSTEM

Jennifer A. Ludlow
Research Assistant, Institute for Natural Systems Engineering
Department of Civil and Environmental Engineering
Utah State University, Logan, UT 84322-4110
United States of America
ludlow@aaron.cee.usu.edu

Thomas B. Hardy
Director, Institute for Natural Systems Engineering
Department of Civil and Environmental Engineering
Utah State University, Logan, UT 84322-4110
United States of America
hardy@aaron.cee.usu.edu

ABSTRACT

A 0.5 kilometer reach of the Logan River was intensively sampled for channel geometries, hydraulic properties, invertebrate drift densities, and fish locations at several discharges. The spatial domain data were used to model the 2-dimensional hydraulic characteristics at several discharges and with available invertebrate drift densities used to predict the expected location of drift feeding salmonids within the study reach. Habitat modeling employed a mechanistic bioenergetics model for drift feeding salmonids. These results are compared with predicted habitat quality based on traditional suitability index curves used with the Physical Habitat Simulation System (PHABSIM). Study results indicate that an energetic based evaluation of the spatial domain differs significantly from PHABSIM based analyses, as would be expected. PHABSIM model results indicated that habitat increased in both quantity and quality with flow reductions between summer and winter. Combined suitability of depth and velocity doubled in magnitude between higher summer and lower winter flows and the spatial distribution of suitable habitat was nearly identical. The mechanistic based bioenergetics model results indicated a substantial reduction in habitat quality between summer and winter conditions due to reduced velocities, temperature and food availability. The bioenergetics model results also showed a marked difference in the spatial distribution of habitat quantity and quality between summer and winter conditions. The bioenergetics model results paralleled known habitat shifts between summer and winter conditions consistent with known behavioral changes in habitat use in drift feeding salmonids. Use of 2-dimensional hydraulic simulations were judged to provided a better spatial representation of the hydraulic environment for use with bioenergetics models versus one-dimensional hydraulics based on the explicit incorporation of the spatial domain in the simulation of velocities.

KEY-WORDS: Bioenergetic Modeling /Habitat Modeling/2-Dimensional Hydraulics/PHABSIM/Suitability Curves/Comparative Results/Field Validation.

INTRODUCTION

When water is developed for agriculture, municipal, industrial, or power production uses, the magnitude, timing, and duration of natural flows may be altered, thereby affecting the availability and quality of fisheries habitat (Peters, 1982). Resource managers are faced with determining impacts associated with these altered flow regimes on aquatic resources and prescribing acceptable instream flows. Although a wide array of instream flow techniques have been developed over the past two decades (EA Engineering, Science, and Technology, Inc., 1986; Reiser et al., 1989), the most commonly utilized and debated methodology is the Physical Habitat Simulation System (PHABSIM) of the Instream Flow Incremental Methodology (IFIM) (Milhous et al., 1981; Bovee, 1995; Wilds, 1988). Although this methodology has been generally perceived as one of the most sophisticated and acceptable tools for determining instream flow requirements (Loar and Sale, 1981; Orth, 1987; Gore and Nestler, 1988; Reiser et al., 1989), a perceived lack of quantitative relationship between PHABSIM model predictions (i.e. Weighted Usable Area - WUA) and affected populations of fish and other aquatic resources have lead to critical reviews in the literature (Orth and Maughan, 1982; Mathur et al., 1985; Shirvell, 1986; Scott and Shirvell, 1987). However, recent work has demonstrated that this modeling approach can relate WUA to fish populations of at least a single species or a limiting life stage (e.g. Nehring and Anderson, 1993; Bovee et al., 1994). Current trends in impact assessment methodologies are returning to a more ecosystem-oriented view of the aquatic environment (Orth, 1995) where the importance of flow dependant characteristics of the river corridor (Hill et al., 1991; Stromberg and Patten, 1991; Nilsson et al., 1991), trophic level interactions (Roell and Orth, 1994) and both population and community dynamics are being stressed (Roell and Orth, 1994; Orth, 1995; Bain, 1995). One aspect of this trend has been the development of individual based bioenergetic models which incorporate the spatial elements of the hydraulic environment, invertebrates and fish interactions (Addley, 1993). These types of models can potentially benefit from the developing application of 2-dimensional hydraulic simulations in IFIM type analyses, where spatial representation of the hydraulic environment can be achieved over a wider domain (e.g. Leclerc et al., 1995) This paper examines the potential utility of bioenergetic modeling using 2-dimensional hydraulic simulations in IFIM type analyses and compares these results with more classical PHABSIM habitat modeling approaches.

BACKGROUND

One area of research interest for instream flow techniques has focused on mechanistic based bioenergetic models as an alternative to PHABSIM habitat modeling. Bioenergetic models define the mass balance relationship between the amount of food consumed by a fish and the fish's growth, the energy it expends, and the waste it produces (Brandt and Hartman, 1993). The mass balance equation is founded on the first law of thermodynamics observing the conservation of mass and energy. Bioenergetic models are based on the understanding that all food consumed by a fish is either used as energy, incorporated into body tissue or eliminated from the body as waste (Winberg, 1956). These models incorporate food availability and energy content, swimming and metabolic costs, water temperature, and other factors in calculations of the fish's energy budget. (Brandt and Hartman, 1993; Ney, 1990; Addley, 1993). The most widely used bioenergetic model in North America is the 'Wisconsin' bioenergetic model (Ney, 1990). This model is a generalized fish bioenergetic model that can be applied to any species. It requires an extensive set of input parameters which model the physiology and growth of fish based primarily on the variables of fish size, temperature, and caloric density of the fish and its prey. The appeal of bioenergetic modeling is that the approach links basic fish physiology and behavior to lower level trophic production under varying environmental conditions (Ney, 1990). The field of fish bioenergetics and its application to fish ecology and management, has evolved rapidly due, in part, to the development of easily accessible and versatile computer software (Brandt and Hartman, 1993; Ney, 1993). However, bioenergetic model validation and corroboration is generally lacking

(Brandt and Hartman, 1993; Addley, 1993).

Hughes and Dill (1990) critiqued the earlier work of Fausch's (1984) as over simplifying the calculations of the fish's feeding rate. They argued that the number of prey the fish encounters depends on the fish's reactive distance to prey, water depth, and water velocity. Fausch's model doesn't take into account the water depth, the shape of the fish's reaction field, any variations in water velocity within the foraging area, or the influences of water velocity on the fish's ability to capture prey. Hughes and Dill (1990) developed a model that uses data on the size frequency distribution of prey, the fish's reactive distance to prey, topography of the stream bed, and pattern of water flow in order to estimate the rate at which the fish observes prey of different sizes. Their model results showed an improved relationship between observed grayling locations within the stream compared to the initial model results of Fausch. Hill and Grossman (1993) constructed an energetic model to determine the optimal focal point current velocity for rainbow trout and rosyside dace. They predicted that fish would occupy velocities at which the net energy gain was maximized. They estimated net energy benefit by measuring the potential energetic gains of feeding at a given velocity which incorporated the ability and frequency of the fish to capture prey at different current velocities and included the energy content of drifting prey. Their data suggested that microhabitat use by trout and rosyside dace can be explained by behavior that maximizes net energy intake.

The bioenergetic model used in this study was developed at Utah State University to simulate the direct cause and effect relationship on drift feeding salmonids of the environmental and physiological variables affecting net energy intake (NEI) (Addley, 1993). The mathematical simulation of NEI is based on the mass balance relationship for energy by subtracting the energy costs (swimming and digestion costs, basal metabolism) and losses (excretion and egestion) from the gross energy intake (GEI) obtained as a result of prey capture. The GEI is determined by the rate of prey capture, using the prey capture model of Hughes and Dill (1990) and components of the predation model of Holling (1959). The model was tested in St. Charles Creek, Idaho, on cutthroat trout (*Oncorhynchus clarki*), where excellent agreement between predicted locations and actual stream habitat use occurred (Addley 1993).

METHODS

Study Site

A 0.5 Km reach within the Logan River in northeastern Utah, USA was chosen for the comparison studies. The reach, shown in Figure 1, starts 30 meters below First Dam, upstream of the Utah State University Water Research Lab (UWRL) and continues downstream for approximately 500 meters below the dam. The width of the Logan River throughout the reach ranged from approximately 10 to 20 m. The majority of the substrate is made up of cobble and boulder with some small areas of gravel. The three dominant species of fish include mountain whitefish (*Prosopium williamsoni*), albino rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*).

Data Collection Methods

Hydraulic Measurements

The spatial domain within the Logan River was measured over a 1x1 meter grid within four 25 meter long sections corresponding to key fisheries habitat areas. In addition, cross section profiles were placed roughly every 20 meters down the longitudinal axis of the river channel between each of the intensively sampled spatial grids. Additional sampling

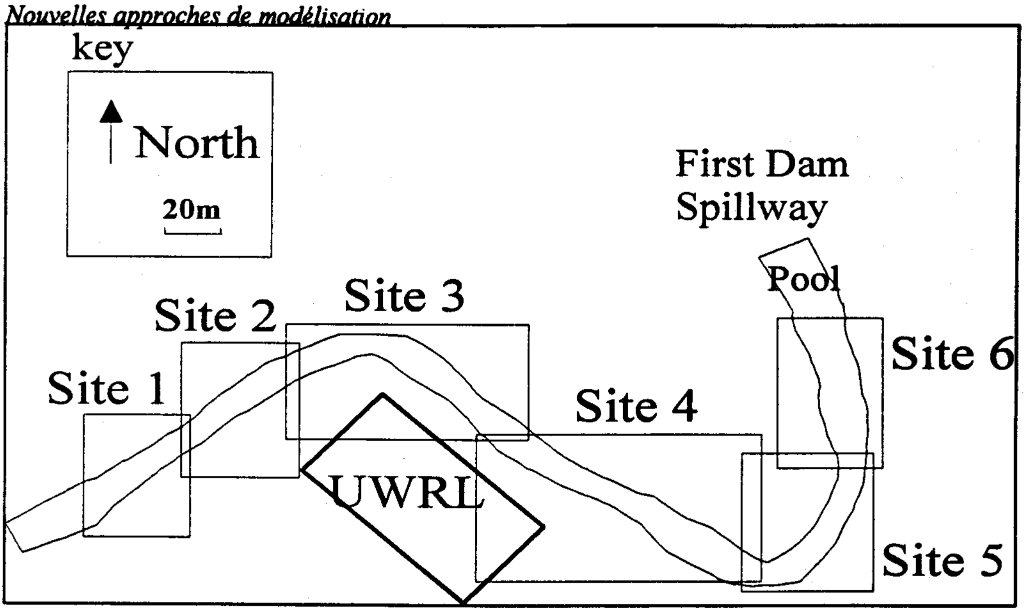


Figure 1: Logan River study reach

occurred throughout the river using an irregular systematic point sampling procedure where channel geometries varied substantially between cross sections. At each spatial domain data point, bed elevation, mean column velocity, and substrate characteristics were recorded. The water surface profile within the reach was also surveyed at the time the field measurements were collected. These measurements were made for two flow rates corresponding to high (summer) and medium (fall) flows. Flows ranged from 3 m^3/s to 6.3 m^3/s over the period of the study. The sampling data was then used to construct a plan view reach level grid system for use in delineation of fish locations and substrate characteristics. In addition, the reach was separated into six meso-scale habitat types represented as deep run/pool, riffle, run, riffle, deep run/pool, run which correspond to sites 1 through 6 in Figure 1.

Fish Measurements

Fish locations within the river were made by underwater observations while snorkeling upstream within each section. The location of each fish was recorded on the plan view spatial grid by determining it's position in the coordinate system using a calibrated total station. The type and length of the fish, the distance the fish was observed from the bottom of the river bed, the total depth at that position, the mean column velocity, and the focal velocity were recorded. The entire river was sampled coincident with the two flow rates measured for the hydraulics and for fish locations only during the winter sampling period. Summer sampling occurred between August 11-23, 1995, fall sampling occurred between October 11-12, 1995 and winter sampling occurred between February 15-21, 1996. Supplemental fish observations were made within selected areas between the summer and fall sampling dates to examine changes in fish position within the reach. The subsampled sections were chosen on the basis of having been previously occupied by fish. Temperature measurements were also taken daily over the duration of the fish observations. Water temperatures ranged from 10°C to 3°C over the duration of the study.

Drifting Invertebrate Measurements

Drifting invertebrates were sampled at each flow rate by placing 3 replicate drift nets at the top and bottom of each of the 6 sections being sampled. Flow rates at each net were computed by measurement of the velocity at the opening of each net. The nets were left in place for approximately 15 to 20 minutes depending on the volume of drifting material and all samples were preserved in a 70% alcohol solution. The drift samples were analyzed for both density and size distribution. Drift size distributions were obtained by sorting each sample according to the following size classes: <3mm, 3-6mm, and >6mm. The energy content of the invertebrates was derived from literature sources (Smock, 1980; Cumming and Wuycheck, 1971) and is represented by Equation 1, where the prey energy (PE) for size *i* has units of Joules and prey length (PL) for a given size interval *i* has units of mm. Drift densities and size classes for winter conditions were taken from literature values (Filbert, 1991).

$$PE_i = 0.3818 * (PL_i)^{2.46} \tag{1}$$

2-Dimensional Hydraulic Simulations

Spatial geometries and the associated hydraulic parameters were unitized to develop a 2-dimensional hydraulic simulation model for predicting depth and velocities within the entire study reach. The simulated data were utilized as input for the bioenergetics modeling. The hydraulic modeling was conducted as part of a collaborative research effort and is reported in Tarbet and Hardy (these proceedings) and therefore will not be discussed within this paper.

Model Evaluation

Bioenergetics Model

The complete form of the bioenergetics model utilized in this study is provided in Equation 2. The reader is referred to Addley (1993) for a complete description of model development.

$$NEI = \frac{\sum_{i=1}^{n} MCA_i * V_{avei} * DD_i * PC_i * (E_i - CC_i) - SC}{1 + \sum_{i=1}^{n} tf_i * MCA_i * V_i * DD_i} \tag{2}$$

The rate of net energy intake (NEI J/hr) is based on the gross energy intake associated with a captured prey of size *i*, minus both energy costs and losses. MCA (m^2) is the maximum capture area at a given location, V_{ave} (m/s) is the mean column velocity in the MCA, DD ($prey/m^3$) is the daytime drift density, PC is the probability of a successful prey capture and assumed to be equal to 1.0, E (J/prey) is the assimilated energy, CC (J) is the cost of capturing prey, SC (J/hr) is the swimming cost, and tf is the handling time and assumed to be 5 seconds.

The invertebrate data, field temperatures, and simulated 2-dimensional hydraulics for the three seasonal datasets were used as input for the energetics model. The model was run for 4 size classes of salmonids corresponding to 200, 250, 300, and

350mm which reflects the size distributions of the observed salmonids within the reach. The resulting model output estimated the net energy intake potential at each spatial grid point (i.e. node point for the finite element mesh). These model results were then compared with fish positions observed in the field.

PHABSIM SI Curves

Suitability index curves for adult and juvenile brown trout currently in use by the Utah Division of Wildlife Resources for conducting PHABSIM analyses within the State of Utah were used to compute the combined suitability at each spatial coordinate. Only depth and velocity curves were utilized in the analyses. The combined suitability index values over the spatial domain were used to compare against the fish positions observed in the field.

RESULTS

During the summer sampling, 42 mountain whitefish, 40 albino rainbow trout, and 40 brown trout were observed. During the fall sampling, 17 mountain whitefish, 3 albino rainbow trout, and 14 brown trout were observed. Finally, during the winter sampling, 28 mountain whitefish, no albino rainbow trout, and 8 brown trout were observed. All fish ranged between 150 and 300mm with some mountain whitefish as large as 380mm. The summer and winter sampling provided observations within all six habitat sites. The fall sampling however, only contains data from sites 1, 2, and 5. Additional observations were not possible due to upstream reservoir drawdowns for dam repairs which resulted in high turbidities. Average invertebrate drift densities at Site 1 ranged from 508 bugs/100 m^3 for summer conditions to 669 bugs/100 m^3 for fall conditions. The fall drift densities were not taken into account in the derivation of the winter drift densities since not all sites were represented due to unexpected reservoir drawdowns and high turbidities. The winter drift densities were derived from Filbert (1991) and assumed to be 85 bugs/100 m^3. The size distribution of the drift for summer conditions was determined to be 50% : 40% : 10% respectively for size classes <3mm, 3-6mm, and >6mm. The size distribution of the drift for winter conditions was assumed to be the same as that measured in the summer samples. Given the limited space and large volume of study results, this paper will only consider a comparison for the habitat modeling results of adult brown trout between summer and winter conditions at Site 1 in order to illustrate overall study findings.

Figure 2 illustrates the depth and velocity distributions over the spatial domain at Site 1 for simulated discharges of 6.3 m^3/s and 3.0 m^3/s corresponding to observed summer and winter flow conditions. These results show that although there was not a substantial reduction in depths between summer and winter, the velocity distributions show a marked decrease throughout this entire reach of the river. It is also interesting to note that at the higher discharge, the variation in the spatial distribution of magnitudes is more evident compared to the lower flow regime. The resulting simulated NEI surface and combined suitabilities under summer and winter conditions are presented in Figures 3 and 4. These results also show observed fish positions during field sampling. A comparison of the simulated NEI response surface between summer and fall conditions versus the response surface of combined suitabilities based on depth and velocity criteria show marked differences. The response surface of the combined suitabilities remain fairly constant over the spatial domain under the two simulated conditions and basically replicates the spatial pattern of velocities. This is attributed to the fact that under both simulated flow conditions, depth suitabilities are relatively constant and velocity suitabilities are dominating model predictions. The lower velocity magnitudes, under the lower simulated flow condition during winter, result in predicted combined suitabilities that are approximately twice the magnitude of the summer values. This implies a more favorable quantity and quality in the available habitat under simulated lower winter flow rates.

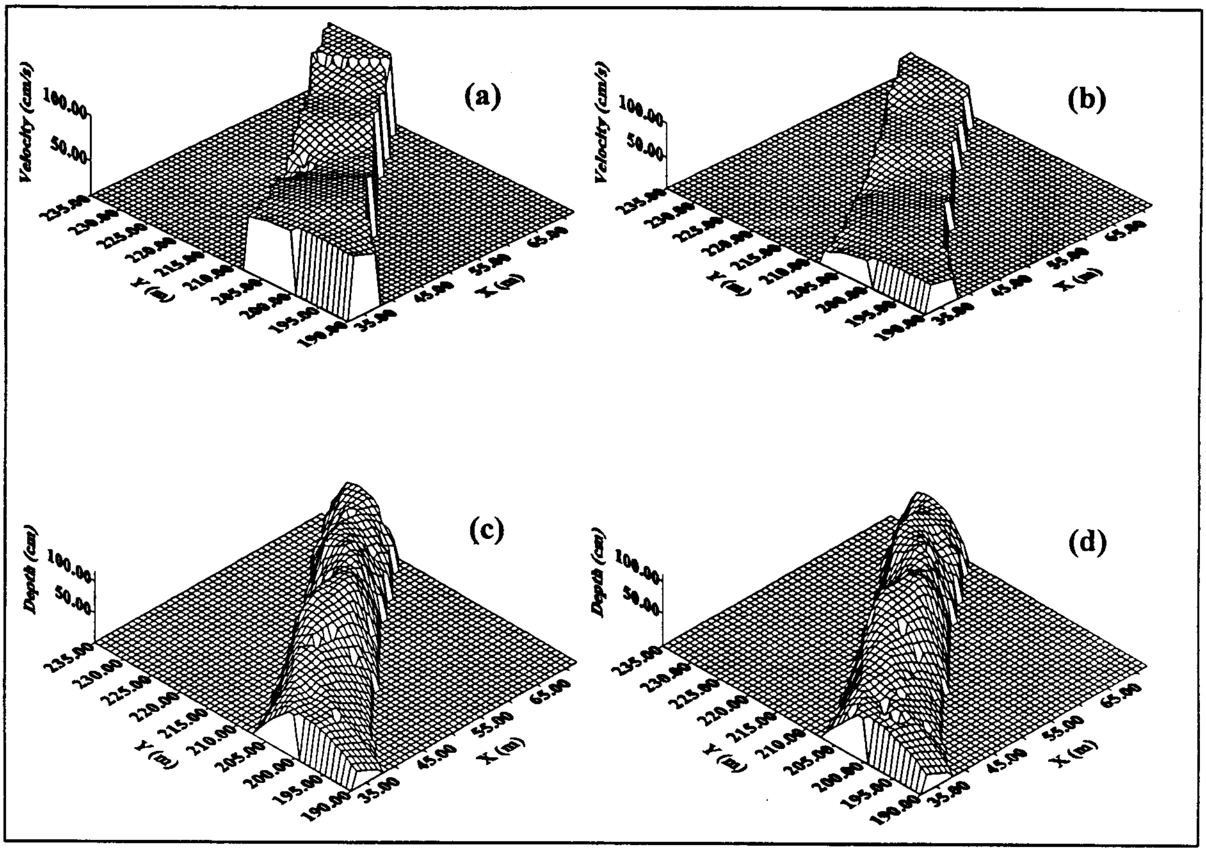


Figure 2: Comparison of summer (a) velocities and (c) depths at 6.3 m^3/s with winter (b) velocities and (d) depths at 3 m^3/s.

In contrast, the NEI response surface shows a three-fold decrease in magnitude between summer and winter conditions with a marked change in the spatial distribution of predicted values. Approximately 20 percent of the stream area under summer conditions is associated with near zero or negative NEI values (i.e. lower right of Figure 3a) and is attributed to instream velocities in excess of the maximum sustained swimming velocity in the NEI model. In contrast, this same region of stream shows usable habitat as indicated by combined suitabilities greater than zero, which is a direct result of the velocity criteria having higher thresholds than the bioenergetic model. The remaining area of the stream has relatively high NEI values at the higher discharge. Location of adult brown trout clearly show an avoidance of the area of the stream associated with negative or near zero NEI values. In comparison, at the lower flow and colder water temperatures during winter, the NEI surface magnitude is reduced to almost half the summer values. Although the velocities have been reduced at the lower discharge, the effect of lower temperature on metabolic rate and reduced invertebrate densities is reflected in the overall reduction of the magnitude of the NEI values. It is interesting to note that the spatial area associated with negative and near zero NEI values during summer now retains positive NEI values. This is attributed to a reduction in the simulated velocities. These results in general, show a trend that is opposite between the two modeling approaches and has important implications from known and observed behavioral responses of salmonids and the use of these models in impact assessments.

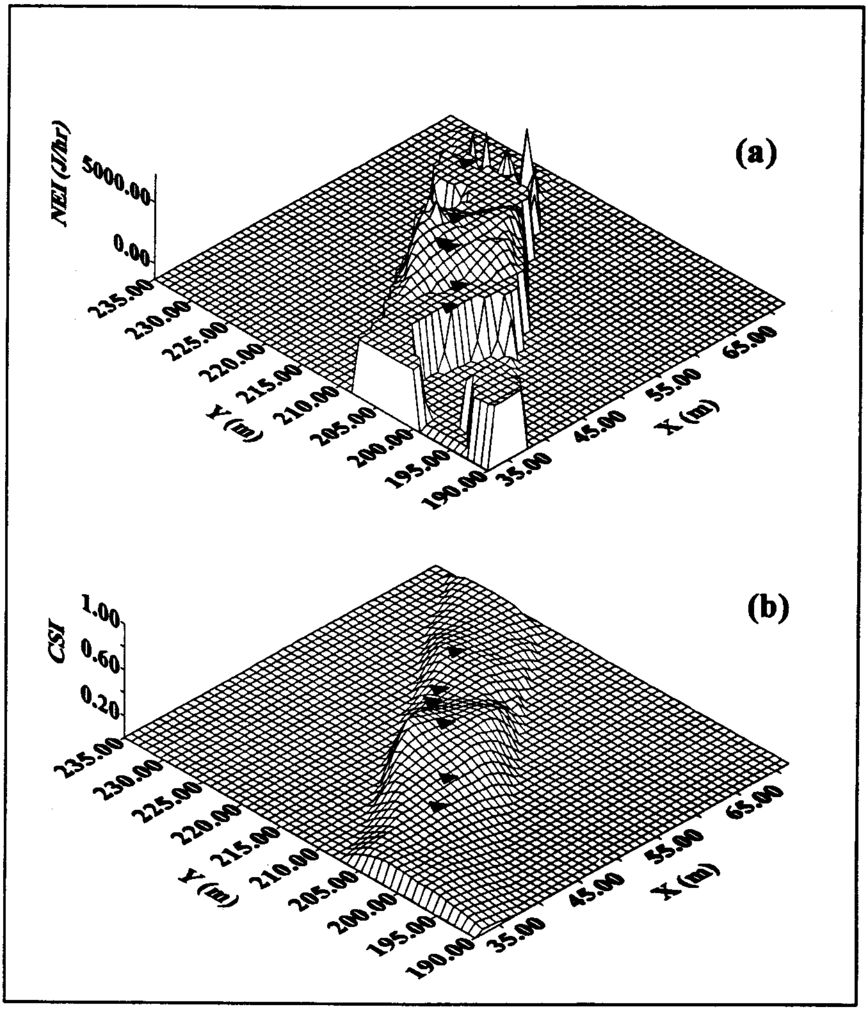


Figure 3: An example comparison of (a) Net Energy Intake (J/hr) and (b) combined suitabilities for adult brown trout in summer conditions for Site 1 at a flow of 6.3 m^3/s and water temperature of 9 °C.

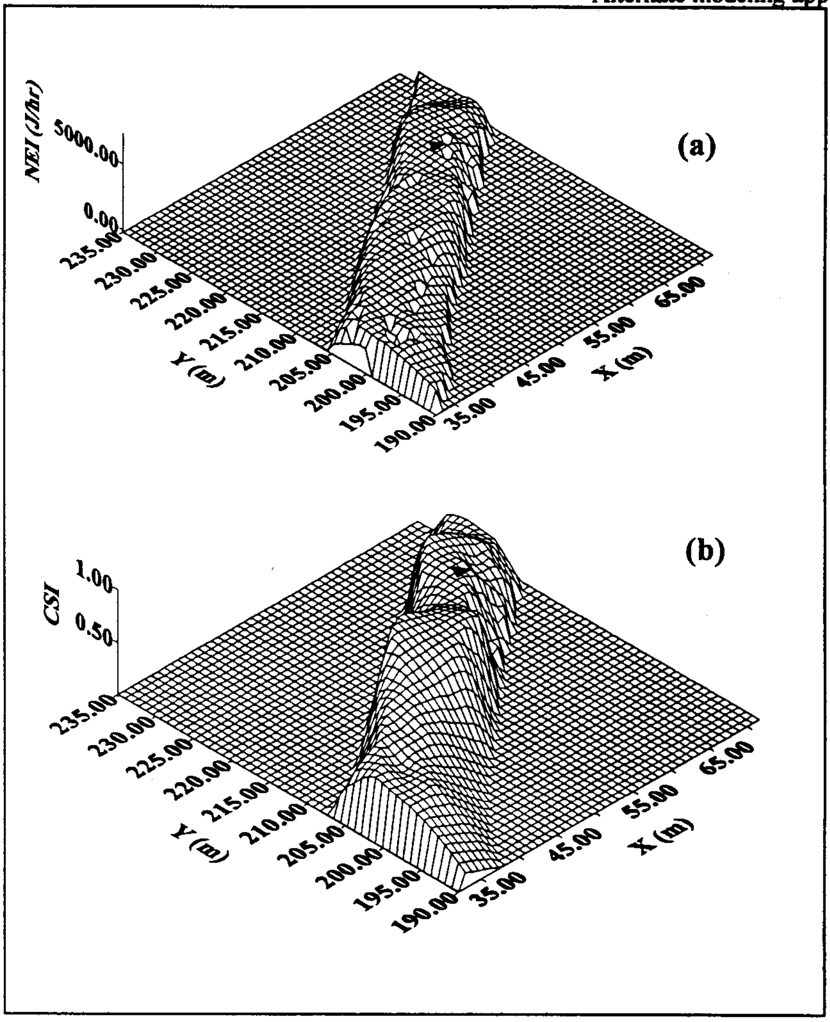


Figure 4: **An example comparison of (a) Net Energy Intake (J/hr) and (b) combined suitabilities for adult brown trout in winter conditions for Site 1 at a flow of 3m^3/s and water temperature of 3°C.**

DISCUSSION

Seasonal shifts in habitat use of salmonids between summer to winter are well documented in the literature (Elliott, 1994; Hill and Grossman, 1993; Calkins, 1989; Chisholm and Hubert, 1987; Cunjak and Power, 1986;). These studies suggest that as the metabolic rate of drift feeding salmonids drop in response to lower winter temperatures, fish move to low velocity areas either associated with the substrate or into low velocity habitat types (i.e. pools). Our study observations of the spatial distribution of salmonids within the study reach showed a clear shift in habitat use between summer and winter, where trout avoided higher velocity areas occupied during summer conditions and relocated in deep slow runs or pool habitats during the winter. The observed shift in habitat use was also reflected in an overall reduction in the winter simulated NEI magnitudes and spatial shifts to areas associated with slow velocity habitats observed within the reach. This is not shown in the combined suitability analyses based on more classical PHABSIM modeling in this study. We feel this is problematic since the depth and velocity dependant analysis involving PHABSIM would suggest that the winter flow reductions represent improved habitat conditions, a result we feel that is not supported by either the bioenergetic modeling or known behavior of winter habitat use by salmonids. These differences are partially dependant on the selection of the specific suitability curves used in our analysis which indicated higher suitable velocities than contained in the bioenergetic model. This particular aspect is not a short coming of PHABSIM per se, since use of seasonal suitability curves could partially overcome this deficiency, especially with the incorporation of temperature based suitability curves as recommended within the broader context of applied IFIM analyses (Bovee, 1986). We maintain however, that use of suitability curves will continue to impose limitations given the wide spread concern regarding transferability issues and cost of site-specific development or validation. These shortcomings are potentially reduced or eliminated using a mechanistic approach as in this study since the underlying response variables should be independent of the stream in which the model is applied.

At present, use of one-dimensional hydraulics is somewhat restrictive in its application to bioenergetic based models since limited spatial dependancies in the simulated velocities are possible. The utilization of 2-dimensional (or 3-dimensional) hydraulic simulations however, can provide expanded opportunities for more spatially explicit evaluations of the hydraulic environment and are more suitable for this type of evaluation technique. Use of 2-dimensional hydraulic simulations in applied field studies will be contingent on the cost-benefits of spatial data requirements for a given data quality objective, which in our opinion has yet to be conclusively demonstrated in the literature although it is one of the focuses of this symposium. Another potential benefit of using a bioenergetic modeling approach relates to the potential to link model output to growth rates and other condition factors within a population (Fausch, 1984; Elliott, 1994) as well as accounting for community dynamics involving linear dominance of selected feeding locations in mixed salmonid populations (Bachman, 1984).

CONCLUSION

Although our study results are encouraging in terms of model output which parallel known behavioral responses of salmonids under seasonal changes in the aquatic environment, we note that application of these techniques within the broader context of an ecologically-based assessment framework still requires considerable research and field level validation. Use of a bioenergetic model based assessment framework at an applied level will require that a temperature model be developed and validated for the river in question as well as the development and testing of seasonal drift density models that should incorporate flow dependant characteristics in invertebrate population dynamics. Although temperature modeling of natural river environments is fairly well developed, the modeling of invertebrate population dynamics may require more effort. Linkages of model outputs over long term temporal domains to observed population or community level dynamics is still lacking and represents a critical evaluation step for these types of models. Use of only depth and velocity criteria in PHABSIM analyses that do not incorporate seasonal shifts in habitat use including temperature may give results which can be miss-leading in light of known salmonid behavior. Clearly, the choice, application and interpretation of models within any applied assessment framework must be consider carefully.

REFERENCES

Addley R.C. 1993. A Mechanistic Approach to Modeling Habitat Needs of Drift Feeding Salmonids. MS Thesis. Utah State University, Logan, Utah.

Armour, C.L. and J.G. Taylor. 1991. Evaluation of the Instream Flow Incremental Methodology by U.S. Fish and Wildlife Service Field Users. *Fisheries* 16(5):36-43.

Bachman, R.A. 1984. Foraging Behavior of Free-Ranging Wild and Hatchery Brown Trout in a Stream. *Transactions of the American Fisheries Society*. 113(1): 1-36.

Bain, M.B. 1995. Habitat at the Local Scale: Multivariate Patterns for Stream Fishes. *Bull. Fr. Peche Piscic.* 337/338/339: 165-177.

Bovee, K.D. 1995. A comprehensive overview of the instream flow incremental methodology. National Biological Service, Fort Collins, CO., 322pp.

Bovee, K.D. 1986. Development and Evaluation of Habitat Suitability Criteria for use in the Instream Flow Incremental Methodology. Instream Flow Information Paper No.21. U.S. Fish Wildl. Serv. Biol. Rep. 86(7).

Bovee, K.D., T.J. Newcomb, and T.G. Coon. 1994. Relations between habitat variability and population dynamics of bass in the Huron River, Michigan. National Biological Service Biological Report 21. 63 pp.

Brandt, S.B. and K.J. Hartman. 1993. Innovative Approaches with Bioenergetics Models: Future Applications to Fish Ecology and Management. *Fisheries* 122(5):731-735.

Camp Dresser & McKee. 1986. Final Report: Minimum Instream Flow Study. Commonwealth of Virginia State Water Control Board. Annandale Virginia.

Carlander, K.D. 1969. Handbook of Freshwater Ecology. Iowa State University Press. Volume 1.

Calkins, D. J. 1989. Winter habitats of Atlantic salmon, brook trout, brown trout, and rainbow trout. CRREL.

Chisholm, I.M. and W.A. Hubert 1987. Winter Stream Conditions and Use of Habitat by Brook Trout in High-Elevation Wyoming Streams. *Transactions of the American Fisheries Society.* 116: 176-184.

Cummins, K.W. and J.C. Wuycheck. 1971. Caloric equivalents for investigations in ecological energetics. *Mitt. Int. Ver. Limnol.* 18:1-158.

Cunjak, R.A. and G. Power 1986. Winter Habitat Utilization by Stream Resident Brook Trout (*Salvenlinus fontinalis*) and Brown Trout (*Salmo trutta*). *Can. J. Fish Aquat. Sci.*, 43: 1970-1981.

EA Engineering, Science, and Technology, Inc. 1986. Instream flow methodologies. Research project 2194-2, Completion report, Electric Power Research Institute, Palo Alto, CA.

Elliott, J.M. 1994. Quantitative Ecology and the Brown Trout. (ed. R.M. May and P.H. Harvey). Oxford University Press Inc., NY.

Fausch, K. D. 1984. Profitable stream positions for salmonids: relating specific growth rate to net energy gain. *Can. J. Zool.* 62: 441-451.

Filbert, R 1991. Is Rainbow Trout Condition Influenced by Invertebrate-Drift Density? MS Thesis. Utah State University, Logan Utah.

Gore, J.A. and J.M. Nestler. 1988. Instream flow studies in perspective. Regulated Rivers Research and Management 2:93-102Hill, J. and G.D. Grossman. 1993. An Energetic Model of Microhabitat use for the Rainbow Trout and Rosyside Dace. *Ecology.* 74(3):685-698.

Hill, J. and G.D. Grossman. 1993. An Energetic Model of Microhabitat Use for Rainbow Trout and Rosyside Dace. *Ecology.* 74(3): 685-698.

Hill, M.T., W.S. Platts, and R.L. Beschta. 1991. Ecological and geomorphological concepts for instream and out-of-channel flow requirements. *Rivers.* 2: 198-210.

Holling, C.S. 1959. Some characteristics of simple types of predation and parasitism. *Canad. Ent.* 91: 385-398

Hughes, N.F. and L.M. Dill. 1990. Position Choice by Drift-Feeding Salmonids: a Model and a Test for Arctic Grayling (*Thymallus arcticus*) in Sub-arctic Mountain Streams, Interior Alaska. *Canadian Journal of Fisheries and Aquatic Sciences.* 47(10):2039-2048.

Johnson, B.J. 1993. Review of Existing Models for Evaluating Fish Population Changes Relative to Environmental Impacts. Report by the U.S. Fish and Wildlife Service, National Fisheries Center, La Crosse, Wisconsin, for the U.S. Fish and Wildlife Service, Environmental Management Technical Center, Onalaska, Wisconsin, July 1993. EMTC 93-S005. 25 pp.

Leclerc, M., A. Boudreault, J.A. Bechara, and G. Corfa. 1995. Two-Dimensional Hydrodynamic Modeling: A Neglected Tool in the Instream Flow Incremental Methodology. *Transactions of the American Fisheries Society.* 124(5): 645-662.

Loar, J.M., and M.J. Sale. 1981. Analysis of environmental issues related to small-scale hydroelectric development, V. Instream flow needs for fishery resources. ORNL/TM-7861, Oak Ridge National Laboratory, Oak Ridge, TN. 105p.

Mathur, D. W.H. Bason, E.J. Purdy, and C.A. Silver. 1985. A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences* 42:825-831.

Milhous, R.T., D.L.Wegner, and T. Waddle. 1981. Users Guide to the Physical Habitat Simulation System (PHABSIM). Instream Flow Information Paper No. 11. U.S. Fish and Wildl. Serv. FWS/OBS-81(43)(Revised).

Nehring, R.B., and R. M. Anderson. 1993. Determination of population-limiting critical salmonid habitats in Colorado streams using the Physical Habitat Simulation System. *Rivers* 4(1):1-19.

Ney, J.J. 1993. Bioenergetics Modeling Today: Growing Pains on the Cutting Edge. *Fisheries* 122(5):736-748.

Ney, J.J. 1990. Trophic Economics in Fisheries: Assessment of Demand-Supply Relationships Between Predators and Prey. Reviews of Aquatic Sciences 2:55-81.

Nilsson, C., A. Ekblad, M. Gardfjell, and B. Carlberg. 1991. Long-term effects of river regulation on river margin vegetation. *J. Appl. Ecol.* 28: 963-987.

Orth, D.J. 1995. Food Web Influences on Fish Populations Responses to Instream Flow. *Bull. Fr. Peche Piscic.* 337/338/339: 317-328.

Orth, D.J. 1987. Ecological considerations in the development and application of instream flow habitat models. *Regulated Rivers Research and Management* 1:171-181.

Orth, D.J., and O.E. Maughan. 1982. Evaluation of the incremental methodology for recommending instream flows for fishes. *Trans. Amer. Fish. Soc.* 111: 413-445.

Peters, J.C. 1982. Effects of River and Streamflow Alteration on Fishery Resources. *Fisheries* (Bethesda) 7(2):20-22.

Reiser, D.W., T.A. Wesche, and C. Ester. 1989. Status of Instream Flow Legislation and Practices in North America. *Fisheries* 14(2):22-29.

Roell, M.J. and D.J. Orth. 1994. The roles of predation, competition, and exploitation in the trophic dynamics of a warmwater stream: a model synthesis, analysis, and application. *Hydrobiologia.* 291: 157-178.

Sale, M.J. and R.G. Otto. 1991. Improving the Assessment of Instream Flow Needs for Fish Populations. Waterpower '91 proceedings pp.76-84.

Scott, D., and C.S. Shirvell. 1987. A critique of the Instream Flow Incremental Methodology and observation on flow determination in New Zealand. Pages 27-43 *in* J.F. Craig and J.B. Kemper, editors. Regulated streams: advances in ecology. Plenum Press, New York.

Shirvell, C.S. 1986. Pitfalls of physical habitat simulation in the Instream Flow Incremental Methodology. Canadian Fisheries and Aquatic Sciences Technical Report 1460. 68 pp.

Smock, L.A. 1980. Relationships between body size and biomass of aquatic insects. *Freshwater Biology* 10:375-383

Stromberg, J.C.and D.T. Patten. 1991. Instream flow requirements for cottonwoods at Bishop Creek, Inyo County, California. *Rivers.* 2: 1-11.

Tennant, D.L. 1979. Instream Flow Regimes for Fish, Wildlife, Recreation and Related Environmental Resources. *Fisheries* 1(4):6-10.

Wilds, L.J. 1988. The Legal-Institutional Analysis Model (LIAM): A Validation Study. Instream Flow Information Paper No. 25. U.S. Fish Wildl. Serv. Biol. Rep. 88(33).

Winberg, G.G. 1956. Rate of Metabolism and Food Requirements of Fishes. Belorussian University, Minsk. Translated from Russian, 1960: Fisheries Research Board of Canada Translation Series 194, Ottawa.

PRESENTATION AND FIRST APPLICATIONS OF A DYNAMIC POPULATION MODEL FOR BROWN TROUT (*Salmo trutta*)

- AID TO RIVER MANAGEMENT -

Catherine Sabaton[1], L. Siegler[2], Véronique Gouraud[1], J.L. Baglinière[3], S. Manné[2]

(1) Electricité de France, Direction des Etudes et Recherches, Département Environnement - 6, quai Watier, 78400 Chatou - FRANCE

(2) Ecole Nationale Agronomique de Rennes, Zoologie - 65 rue de Saint Brieuc 35042 Rennes Cedex, FRANCE

(3) Institut National de la Recherche Agronomique, Laboratoire d'Ecologie Aquatique - 65 rue de Saint Brieuc 35042 Rennes Cedex, FRANCE

ABSTRACT

The modern-day importance of multiple-use strategies for managing water resources necessitates the development of decision-support tools for river administrators. Among the potential uses, fishing and the need to preserve good balance among fish populations require that we be able to monitor the impact of any modification in the environment on the dynamics of fish stocks on the site. To this end, a mathematic model representing the long-term evolution of a trout population under different river management scenarios is currently being developed.

This article presents a synthesized description of the model in question. Based on the Leslie matrix, it describes the structure of a population broken down into age classes; if one knows the population structure for any given year, the model should enable estimating that of the following year. In fact, calculation is on a one-month time step; the passage from one month to the next takes into account various relevant factors: survival rate of individuals in the different age classes, fertility rate of females, linear and weighted growth rates, displacement linked to habitat fluctuations using Weighted Usable Area values.

The model has now been applied to two French rivers. Regular monitoring of trout populations on the Kernec (Brittany) enabled comparing the response of the biological model with no displacement, with actual variations in fish stocks on the first river. In addition, the knowledge of discharge and WUA chronologies on the Echez (Pyrenees) made it possible to carry out initial simulations of the response of a fish population to different river management scenarios on the second site.

The initial results are encouraging. The biological model tested on the Kernec gives a satisfactory representation of the thermal events and the resulting changes in population structure. The hypotheses adopted relating to fish behavior in the face of habitat fluctuations still remain to be confirmed and fine-tuned. The simulations of the Echez, however, clearly show the type of impact that climatic events and river management modes can have on a trout population.

Key words: *Salmo trutta*, population dynamics, modeling, habitats

INTRODUCTION

Present development of a multiple-use water management strategies more than ever requires tools to enable predicting the long-term impact on biocoenoses of artificial modifications in the characteristics of a river. To this end, a number of authors have described models designed to follow evolution in fish populations (Van Winkle *et al,* 1974; Christensen *et al,* 1977; Horst, 1977 and 1978; Travis *et al*, 1980). These models, often based on Leslie's work (1945), have, for example, served in estimating the impact on fish stocks of mortality due to hydroelectric installations. Bovee's work (1978) on modeling habitat suitability opened new paths of research to link hydraulic conditions and evolution in fish populations. In recent years, integration of such quantification of the habitat in the type of models mentioned above has enabled simulation of displacement and mortality of fish as a function of natural or artificial variations in habitat quality (Cheslak and Jacobson, 1990; Williamson *et al,* 1993; Jager *et al*, 1993) and simulation of evolution of a population subject to major climatic events (strong flooding, extreme low-water periods) or to new modes of river management.

This paper presents a structural biological model coupled with a physical model for estimating habitat conditions for the various life stages in a population of brown trout (*Salmo trutta*). It has been applied to two French rivers. On the Kernec in Brittany, regular monitoring of fish populations enabled testing the response of the biological model with no displacement (no habitat chronologies available). On the Echez in the Pyrenees, the simulation tools now available allow for reconstituting discharge and habitat chronologies with which to perform the first simulations of the evolution of a fish population subject to different river management scenarios.

MATERIALS AND METHODS

Description of the model

The model described here concerns a trout population structured in 5 age classes corresponding to five cohorts. The principle behind passage from one age class to the next is identical to that of the Leslie model, although the time step here is monthly.

Leslie model - Reminder and analogy with the model described

The Leslie model (Leslie, 1945) is based on a breakdown of a closed monospecific population into age classes. Each year, the individuals pass from one age class to the next, allowing for mortality; the first age class ("class 0") is supplied by individuals in age classes with a potential for reproduction. The model is based on the following equation:

(1) $$N(t+1) = A*N(t)$$ in which

- the N(t) vector of R^n is composed of the members of the different age classes,
- the terms of matrix A depend on survival and fertility, considered to be constant in each of the different age classes.

The biological parameters defining matrix A correspond here to stable conditions in the environment, under which the structure of the population studied is considered to be in equilibrium, which is to say under which:

(2) $$N(t) = N(t+1) = A*N(t)$$

This equation introduces a relationship among the different biological parameters. If one knows the survival rate of individuals in classes 1 and over, and the mean fertility rate, one can estimate the survival -"in equilibrium"- of individuals in age class 0, which is often poorly known and difficult to calibrate. For any given fish stock, one can then determine the distribution "in equilibrium" of individuals in the different age classes.

The model described here simulates evolution in the population on the basis of environment-related biological parameters (Figure 1); the time step is monthly so as to be able to account for climatic variations and their effect on the population. It is also based on the principle that an equilibrium exists, both in terms of fish stocks and in terms of distribution among the different age classes, when conditions in the environment are stable and in the absence of specific incidents affecting the population.

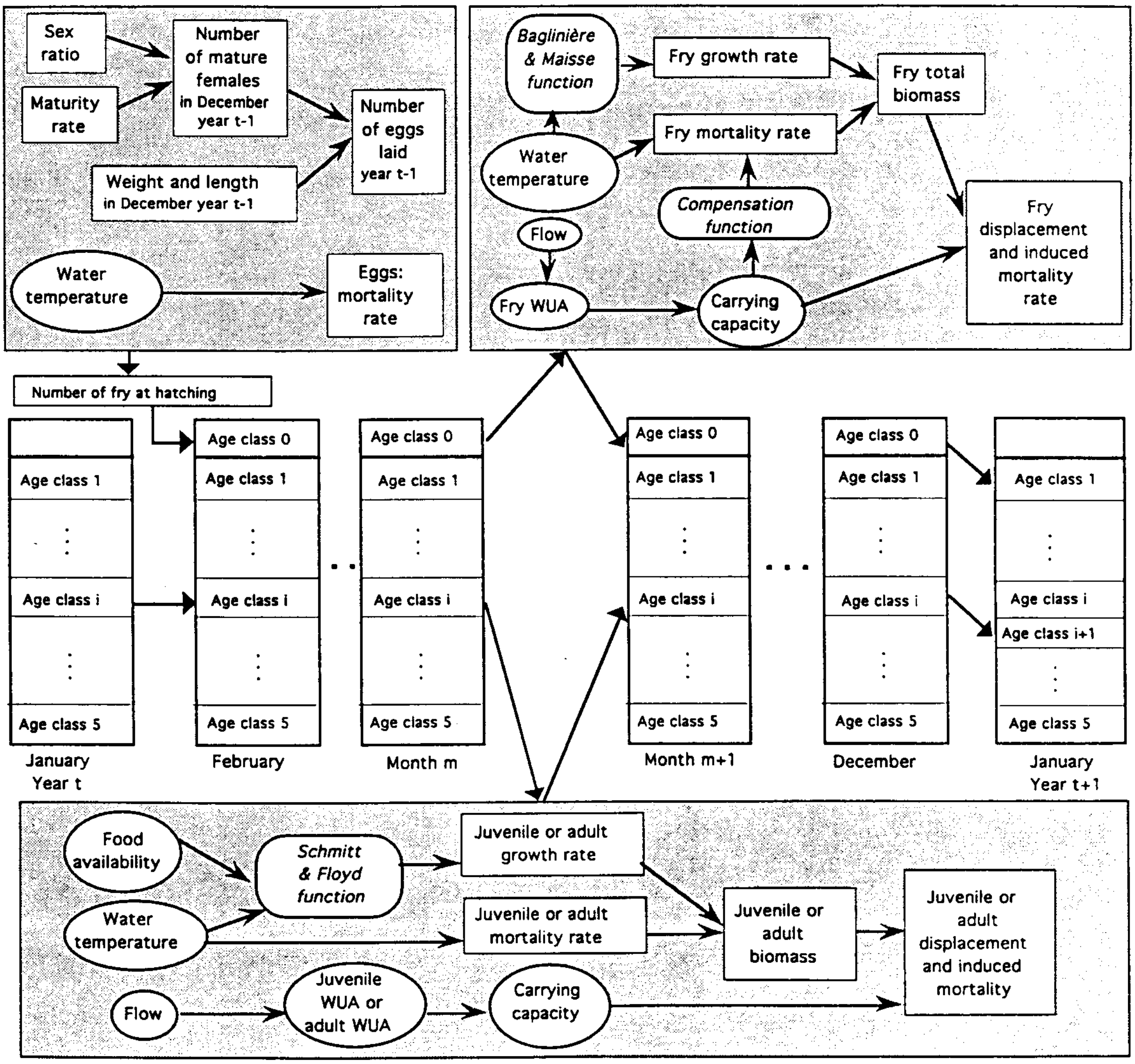


Figure 1: Description of the model - main phenomena modeled

Integration of environmental parameters

Change in a fish population is directly linked to natural or artificial variations in the environment. Among the characteristic environmental parameters likely to modify the biological parameters or the behavior of the trout population, we principally chose the following for our study: temperatures (conditioning the date of spawning, period of incubation, growth, fertility and survival) and habitat quality of the river for the different stages in the trout life cycle (which conditions survival and displacement). The life stages chosen for this study were defined as follows: fry are class 0 individuals, juveniles are class 1 individuals up to the beginning of November (< 18cm on average), adults are larger individuals (class 1 after October and classes 2 and above). The habitat parameter is defined by the carrying capacity of the environment, or the maximum biomass, for each of the three stages, that a segment of the river can hold for a given Weighted Usable Area (Bovee, 1982; Souchon *et al*, 1989; Cheslak and Jacobson, 1990; Williamson *et al*, 1993). Here, our hypothesis is that, for each of the three life stages, carrying capacity per m2 of WUA corresponds, under typical hydrological conditions (taking into account mean limitation periods in terms of maximum rate of occupancy), to a population in which the different age classes are balanced. Under this hypothesis, it is possible to interlink these three carrying capacity values for the corresponding mean monthly WUA chronologies.

Biological parameters

Dates of spawning and emergence depend on thermal conditions, but generally in France, fall between mid-November and mid-January for spawning and between mid-January and mid-March for hatching. Given the monthly scale adopted here, these dates were respectively set in the model in December and February.

Weight increase, for a given month and number of individuals, conditions the biomass of the different age classes.
For fry, we use the growth model developed by Maisse and Baglinière (1990), which links the size of fry (and then the weight, using a size-weight relation) to the sum of the degree-days "lived" since emergence.
For the "one year and over" group, we adopt the relationships of Serchuk *et al* (1980) between weight increase in the fish, water temperature and food availability. If d is the number of days in month m, then for fish in age class i:

$$W(i,m+1) = W(i,m)*(1+cor(i)*g(i,m))^{d} \tag{3}$$

in which g(i,m) and cor(i) are respectively the daily growth rate and the corrective coefficient linked to age, as given by Serchuk (corrective coefficient calibrated for a given trout population).

The absolute fertility rate of a female - or average number of eggs laid - is also directly linked to weight increase. For a trout with weight P, it is given by an equation of the following type:

$$\mathrm{Log}F = a*\mathrm{Log}P + b \qquad \text{(Euzenat and Fournel, 1976)} \tag{4}$$

Survival is broken down according to the different stages in development.

- From spawning to hatching: Hobbs (1940) estimates survival of eggs at 80%. We keep this estimation, but introduce the respectively lethal thresholds of 1° and 15° C for temperature, and 3 mg/l for oxygen (Raleigh *et* *al,* 1986), on the theory that trout embryos are sensitive to the thermal factor and to a lack of dissolved oxygen in the first instants of embryonic life (before closing of the blastopore).

- Mean annual survival of the 0^+ is calculated from the Leslie yearly model for a balanced population. It is in fact broken down in our model to account for two phases in development. From hatching to emergence (February, March), mean survival is set, in accordance with observations on streams in the Pyrenees, at 85% for all individuals in this phase (Lim, personal conversation). From April to the end of December, residual survival ($s_{eq}/0.85$) is evenly distributed over each month (giving $s_{eq,m}$ in month m).
Effective survival of 0^+ individuals is calculated by correcting the mean survival values with a coefficient dependent on conditions in the environment through the year. During the first period, fry are subject to few external threats, with the exception of entrainment episodes; we consider that effective survival is equal to mean survival (corrective coefficient = 1). The highest mortality is actually during the next period (Elliott, 1994; Euzenat and Fournel, 1976). Survival in this period is extremely variable; in addition to mortality due to migration, it is strongly linked to the density of fry present and to thermal conditions.
The corrective factor linked to thermal impact varies from 1 to 0; it is a multiplying factor of mean survival and tends to increase mortality during particularly hot months. The survival rate is considered to decrease above 18.5° C for fry (decline in the thermal preference curves given by Bovee -1978-), reaching 0% at a lethal temperature on the order of 26.5° C (nil thermal preference, according to Bovee).
The phenomenon of density-dependent mortality is represented in the model by a function similar to that described by Faucher *et al* (1983):

$$s_m = s_{eq,m} \frac{1 + kc}{1 + kc * Bm/Bpot} \tag{5}$$

which tends to increase natural survival in month m when habitat conditions are favorable, which is to say when the carrying capacity for fry -Bpot- is higher than the biomass of fry present -Bm-, and to increase mortality in the case of overdensity.

- Mean yearly survival of individuals 1 year and over is, according to numerous authors (Mills, 1971; Maisse and Baglinière, 1991), very stable with age. Under "average" environmental condition, it would appear to be on the order of 40% up to the fifth year. This mean survival rate is evenly distributed over the 12 months of the year; a corrective coefficient is applied each month to take unfavorable thermal conditions into account where necessary, with the survival rate declining from 19° up to 28.5° C (thresholds determined on the basis of thermal preference curves given by Bovee -1978-).

Displacement linked to habitat fluctuations

In addition to the biological phenomena discussed above, we wanted to allow for displacement of fish in accordance with fluctuating habitat conditions (Phillipart, 1977; Chapman, 1966; Bjornn, 1971). To do this, the river segment studied is broken down into reaches in which the WUA chronologies are calculated for each of the three stages during the simulation period. Phenomena of displacement are considered to occur when the fish stock value in one reach at time t exceeds the corresponding carrying capacity, as in the models proposed by Cheslak and Jacobson (1990) and Williamson *et al* (1993).
A certain number of hypotheses were then formulated:

- in a one-month time step, trout are capable of exploring the entire river segment until they find an available habitat or leave the system under study. This search takes the form of both downstream and upstream migration for juveniles and adults, and of downstream migration only for fry;
- individuals leave an overly dense reach and distribute themselves in upstream or downstream reaches depending on habitat availability in these areas. Fish finding no shelter in adjacent reaches pursue their search;
- displacement induces mortality rates which are a function of the age of the fish and the distance from the original habitat (hypothesis also retained by Williamson *et al* -1993- for young *Oncorhynchus tshawytscha* or Cheslak and Jacobson -1990- for *Oncorhynchus mykiss*). For the time being, we have chosen the mortality rates cited by Cheslak and Jacobson (1990) for rainbow trout: 50% adult mortality per reach covered, 80% for juveniles and 90% for fry;
- for fry, such displacement compounds the density-dependent mortality phenomenon in regulating stock levels in the event of insufficient carrying capacity.

Study sites

Two applications of this model are presented here. The first module which does not represent migration, was applied to the Kernec, a Breton river for which we had no discharge and WUA chronologies, but on which trout stocks have been monitored for several years. The complete model, integrating migration, was then tested on the Echez in the Pyrenees, an environment now the subject of a specific integrated water management study, and for which considerable hydrological and morphodynamic data are available

The Kernec is a tributary of the Scorff, a small coastal river in southern Brittany. It is some 6 km long and has a maximum width of 3 meters; it flows through granitic terrain and down steep gradients which vary between 4 and 9.5%. The trout population was studied from 1976 to 1983 by the INRA Aquatic Ecology Laboratory in Rennes, in collaboration with the Higher Fisheries Council. The model was applied using a chronological series of temperatures recorded on the Kernec during the years in which fish were monitored. Knowing the populations in place, it was possible to calibrate the biological functions integrated. Thanks to monitoring of evolution in fish stocks for seven years, we were able to carry out an initial on-site test of the response of the model to thermal conditions, supposing habitat conditions to be stable, which is to say not accounting for phenomena of displacement.

The Echez is a river in the Atlantic Pyrenees region. It is an affluent of the Adour with a fluvial regime characterized by a low-water period from July to September and a high-water period in February-March. Mean annual discharge is on the order of 3 m3/s, and width varies from 10 meters in the upstream reaches to 25 meters downstream. This river is subject to relatively extreme low-water periods which, compounded by considerable withdrawal for irrigation, result in serious drops in throughflow in dry years. Software to simulate the overall dynamics of the river basin - aquatic medium and human activities - was developed (Rietjens *et al,* 1994) to test management alternatives (in particular, compensation for low-water periods by drawing from a nearby river). This latter study provided us with monthly chronologies for discharge, temperature and WUA for the brown trout over a stretch of fifty kilometers, broken down into 39 reaches, under different management scenarios. Fishing by the Higher Fisheries Council and by local fishing associations provided the data with which to calibrate certain of the biological parameters characteristic of the trout population in the area. Our knowledge of the fish stocks is yet fragmentary, but our wish was rather to test response of the complete model to the first management simulations.

Application of the model

The simulation begins on the *Kernec* in 1975, taking the initial number of individuals as equal to the mean number of all fish observed over the next seven years. The large-scale fishing in the area led us to add to natural mortality a value of mortality due to fishing, different depending on the age class, which reduces mean survival rates to 30%, 15% and 0% respectively for the 2+, 3+ and 4+ classes.
As carrying capacities are not known here (no available discharge and WUA chronologies), the simulation supposes that the compensation function applied to fry survival (density-dependent mortality) tends to bring the population toward a state of equilibrium corresponding to the maximum total biomass observed during the monitoring period. Table 1 gives the mean monthly temperatures in summer from 1975 to 1982.

Table 1: Mean monthly temperatures (°C) on the Kernec in summer from 1975 to 1982

Year	1975	1976	1977	1978	1979	1980	1981	1982
June	15,8	19,5	14,2	16,5	16,7	13,0	13,4	16,5
July	17,2	20,4	17,0	18,5	18,4	15,3	15,2	17,8
Aug	19,0	18,2	16,2	15,0	19,2	16,2	15,7	17,4

On the Echez, the stretch studied was divided into three large zones reflecting the changing morphology of the stream from the upstream to downstream reaches. Two simulations of evolution in the population - integrating migration between the different reaches in accordance with fluctuations in local habitat conditions - were carried out for each of the zones from 1969 to 1992. The first uses discharge and temperature chronologies for a so-called "reference" state of the river basin; the second takes into account the systematic addition of 3 m3/s from June to August, to compensate for low water levels during this period corresponding to maximum withdrawals for agricultural purposes. Here, we shall present the results obtained for the zone furthest downstream (15 kilometers long, broken into 8 reaches), the stretch of the river most subject to withdrawal for agricultural needs. Carrying capacities for adults are calculated on the basis of Souchon's estimation of 30g/m2 of WUA (1989). This figure, quite different from that given by Bovee (1982) of 65 g/m2 of WUA, corresponds better to field observations of different French rivers (Souchon *et al*, 1989; Baran, 1995; Lauters, 1995). Carrying capacities per m2 of WUA for the other life stages are 15 g/m2 for juveniles and 10 g/m2 for fry (calculated on the hypothesis of equilibrium under typical hydrological conditions in the Echez).

RESULTS

The Kernec

Figure 2 presents the results given by the model (respectively, in number of 0+ individuals and in number of individuals one year and over - written as "≥1+") in October of each year, which are compared with estimations on the basis of actual autumn inventories. The simulated evolution in numbers reproduces important climatic events (1976 and 1979 for fry, 1976 for the "≥1+") and phenomena of compensation (considerable increase of 0+ individuals in 1977, with repercussions on the "≥1+" in 1978 and 1979 and later on 0+ recruitment in 1980). The model gives a relatively good representation of the observed evolution in numbers from 1976 to 1980, despite a systematically lower level of numbers of fry. We nonetheless note on the one hand that the real population suffered more from the heat episode in summer 1976 than seen in the simulation, and on the other, that its significant decline beginning in 1981 is not reproduced by the model.

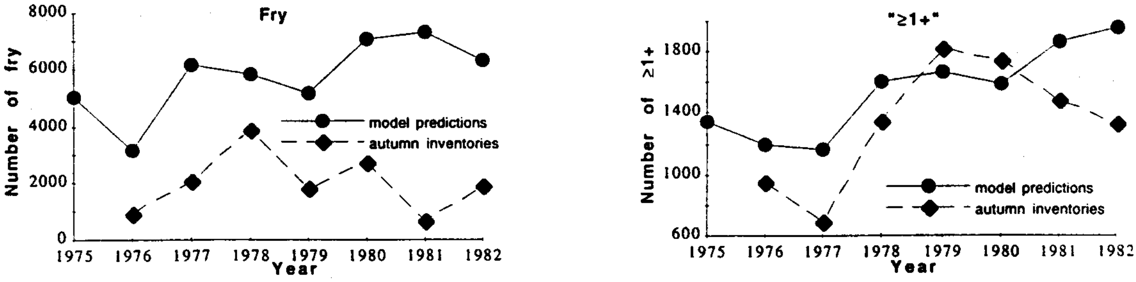


Figure 2: Kernec - model predictions in October compared to autumn inventories , for fry and for "≥1+"

The Echez

Figure 3 shows the evolution in estimated biomasses (fry, juvenile and adult) for the "reference" state in October from 1969 to 1992, compared with those corresponding to added discharge in summer. The first simulation shows how the model translates the impact of limiting episodes on the life cycle of the cohorts affected. For the young stages, these episodes are primarily flooding in autumn-winter (1974, 1982) or late spring (1975, 1991), severe summer low-water periods (1975, 1976, 1985) and periods of high summer temperatures (1973, 1989, 1990, 1991). For adults, limiting episodes correspond to low discharge or high temperatures in summer (1974, 1975, 1976, 1979, 1983, 1985 and 1991), or to low discharge at the end of the year (1988). In the second simulation (added discharge in summer), biomasses are still relatively fluctuating for young stages (limitation primarily during strong discharge episodes). These fluctuations are transmitted to a lesser degree to adults; we do, however, find some strong drops in adult biomass linked to a few autumn low-water periods (in 1985 and 1988). These periods tend simultaneously to limit levels of young individuals and lead to low recruitment of adults in the following year (1986 and 1989).

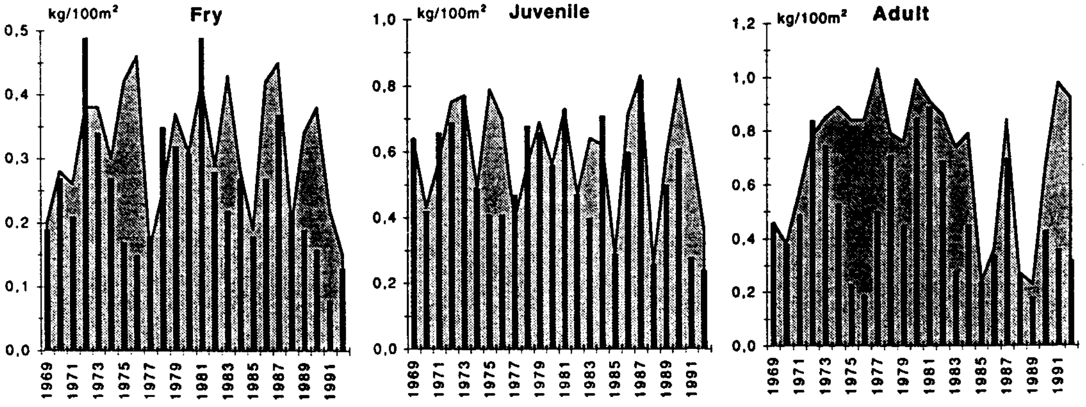


Figure 3: Echez - model predictions of biomasses in October for fry, juveniles and adults - comparison between the simulation for "reference" state and the one with added discharge from June to August-

(■ "reference" state ▨ with added discharge)

The model also enables following, over a one-year period, the monthly evolution in the population in relation to the parallel evolution in total carrying capacities of the study area.

Figure 4 gives the results obtained for a year with no limiting climatic event (1972-Figure 4a) and for a year with a severe summer drought leading to displacement (1975-Figure 4b), in the simulation for the "reference" state. The 1972 curves (Figure 4a) show the extent to which the combined effect of mortality and growth causes the biomasses of the different life stages to fluctuate naturally month by month. The biomass of fry increases up to the point at which mortality is no longer offset by growth rate (which slows down in autumn). That of juveniles increases constantly through the year up to the point of passage to the adult stage (growth rate exceeding mortality); that of adults diminishes up to November (mortality exceeding growth rate) until the massive arrival of juveniles. The impact of the adopted displacement hypotheses on population change shows up clearly in 1975 (Figure 4b). We see the repercussion of displacement -and induced mortality- triggered by the serious summer drought. The increase in biomass of young individuals is blocked in July by the drop in carrying capacity and by a simultaneous increase in mortality related to the poor thermal conditions (still visible in August for fry). Following the regular decrease in adult biomass up to June, we find a sudden drop in July, and the levels attained in October are very low. Recruitment in each of the age classes will be relatively low in 1976.

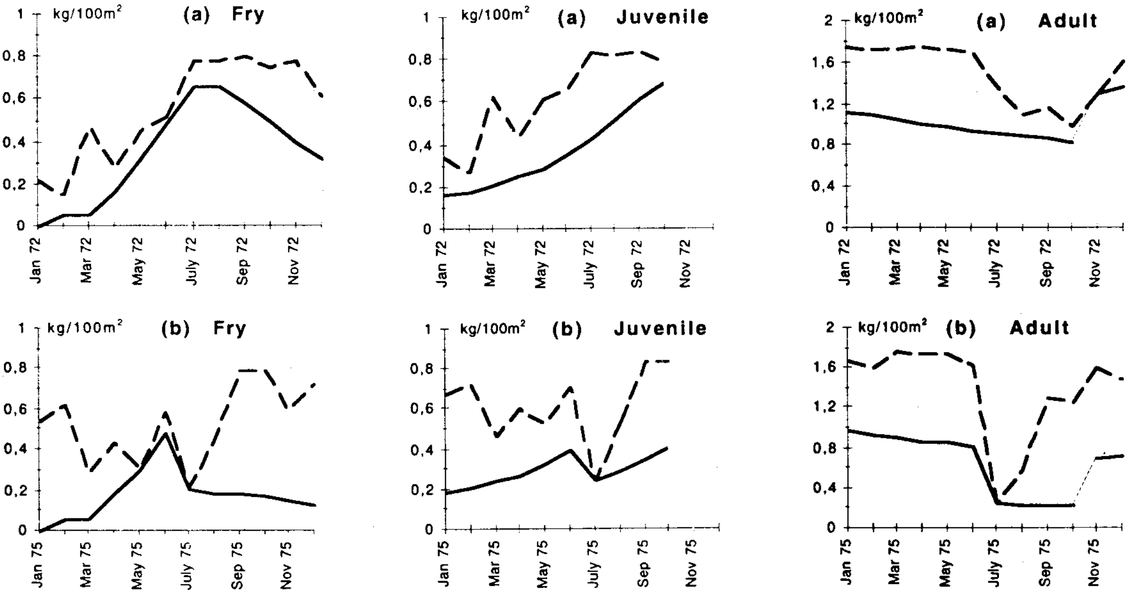


Figure 4: Echez - simulation for "reference" state - monthly evolution of biomasses for fry, juveniles and adultes - 1972 (a) and 1975 (b) (— — — carrying capacity ——— simulated biomass)

Figures 5 shows the evolution in biomass through 1975 for the second simulation (added discharge from June to August). In 1975 (Figure 5), we find changes similar to those presented earlier in 1972 (Figure 4a) to the extent that summer accidents are eliminated. We see, however, a slight limitation of fry in June due to excessive added discharge.

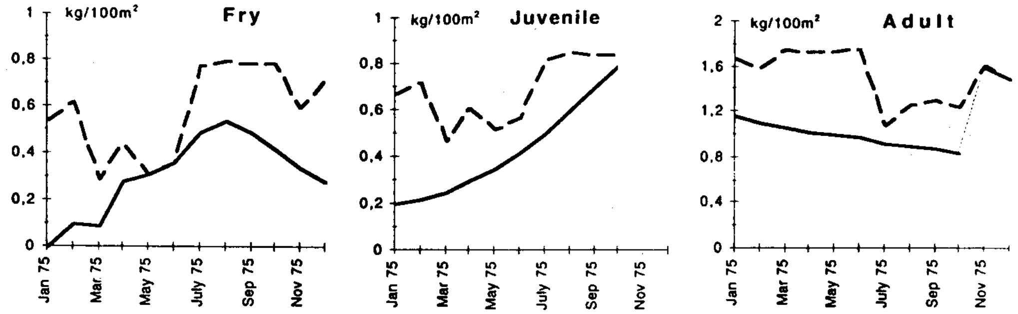


Figure 5: Echez - simulation with added discharge in summer - monthly evolution of biomasses for fry, juveniles and adultes - 1975 (— — — carrying capacity ——— simulated biomass)

DISCUSSION

The simulation results obtained for the Kernec, corresponding to the seven years of monitoring, are encouraging. The model gives a relatively faithful reproduction of evolution in the population over the first 5 years (1976 to 1980), both for "≥1+" and for fry. For the latter, however, there is a discrepancy between simulated levels and real numbers calculated on the basis of fishing. This may be due less to the actual response of the model than to the fact that the numbers observed do not necessarily correspond to those of a balanced population (too few recruits in relation to the spawners present). This apparent imbalance may in fact be explained first by less efficient fishing of small individuals, and second by the fact that the study site is not completely closed, and that the females may have chosen another area for spawning. Whatever the case may be, the biological functions used give consistent results. We must, however, underscore the divergence which begins to be noticeable between estimated and observed numbers as from 1981. This phenomenon may be due to several factors:

- the real population present in the river may not be in equilibrium during the study period and may tend to return to a state of equilibrium different from that used in the model, around which the simulated population is considered to fluctuate;
- the quality of the environment may be changing, and the population would be reacting to this change which is not reproduced in the model.

The characteristic episode linked to the drought in France in 1976 is accounted for in the simulation of the Kernec by the thermal factor only. The associated mortality of "≥1+" and even more of fry (given the low level of recruitment "of ≥1+" actually observed in 1977) appears to be significantly underestimated. It may therefore be that the corrective coefficient of temperature-dependent survival is poorly suited for extreme conditions. We must nonetheless remember that this hot period was without a doubt associated with low discharge, most probably corresponding to limiting habitat conditions for trout which are not accounted for in the simulation of this site. We can clearly see the need here to be able to simulate the impact of hydrological accidents, reproduced in the complete model by migration in relation to habitat quality. The simulations of the Echez were aimed mainly at testing the response of the model over several years, with the integration of the PHABSIM methodology (Bovee, 1982), and attempting to understand what happens, under the displacement hypotheses adopted here, to cohorts subjected to limiting climatic events. It was not possible, with the data now available, to perform a simulation on a scale of less than one month. The one-month time step has the disadvantage of not taking very sporadic events into account, but is, on the other hand, in keeping with the hypothesis that the fish response to hydrological phenomena of a non-negligible duration (noticeable in a one-month span) is displacement. The simulated impact of these climatic events on the population depends on the season in which they occur and the life stage of the fish affected, and the response of trout is relatively clear. We can see however, that, for fry in the very first period of their life, the model gives a poor quantitative estimation of displacement in response to limiting episodes (spring flooding). In the first simulation, the problem is not clearly highlighted, as it is often difficult to determine the exact impact of floods (which most particularly cause WUA to decrease for the youngest stages) on stock levels, given the fact that low-water periods often limit numbers in all age classes. The impact of flooding becomes more visible, on the other hand, in the second simulation representing added discharge in summer. In this simulation, it is clear that the model does not show any effect of strong winter or spring discharge on fry numbers (for example in 1974 and in 1989), though this is certainly an important factor in regulating stock levels in this type of river (Nehring and Anderson, 1993). Bovee (1988) futher noted a clear relationship between flood episodes leading to low WUA values for fry in spring, and the number of recruits in the adult population in the following years. In fact, the mechanism of displacement used in the model cannot allow for a good representation of entrainment of pre- or post-emergence fry during strong flood periods, given that the total biomass they represent is rather small, and that a simple comparison with any carrying capacity is insufficient to represent such incidents (Figure 6 for floods from February to April 1974, and in April 1989). Higher mortality rates and/or more specific entrainment values shall have to be integrated, to represent limiting habitat conditions for this very first period of life.

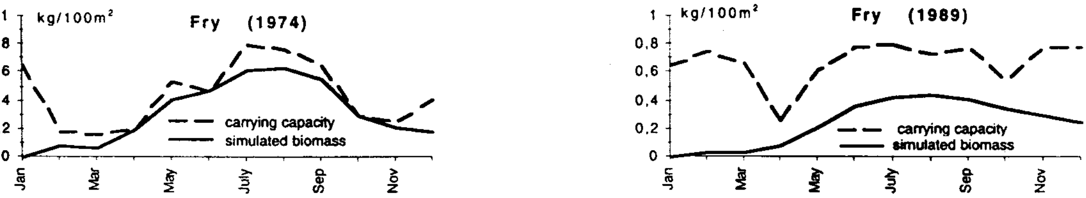


Figure 6: Echez - simulation with added flow - monthly evolution of biomass for fry - 1974 and 1989

The response of fish to fluctuations in discharge is based here on the hypothesis that an overdensity, which might be due to a decline in habitat quality, is offset by the search for available suitable habitats. For adults, this hypothesis is in line with the observations of Baran (1995) on more than 250 river segments in the Pyrenees: trout biomasses observed never exceed the carrying capacities corresponding to the lowest WUA values through the year. For the youngest individuals, studies conducted to link density and WUA have so far proven relatively disappointing (Capra, 1995; Baran, 1995). However, the significant variations in biomass of these age classes through the year and the number of parameters that influence their biological cycle certainly explain in part the difficulty of any simple linkage between densities observed at one point in time and habitat quality.

The phenomena involved in fish displacement are undoubtedly complex, and it would be good to verify that the decision to represent them by comparing biomasses present and momentary carrying capacities (on a scale with the time step) indeed allows us to explain the real fluctuations in population levels. To do this, we shall need simultaneous hydrological chronologies and stock monitoring results. To this end, long-term monitoring of a number of test sites on French rivers, and more specific experiments concerning the behavior of trout in the different life stages, are now being envisaged. We can nonetheless conclude that the model in its present state is a good tool to help in understanding the mechanisms which interlink the different cohorts over time, and that it already allows us to imagine the impact of different river management scenarios on the long-term evolution of a trout population.

REFERENCES

Baran, P. (1995). Analyse de la variabilité des abondances de truites communes (*Salmo trutta* L.) dans les Pyrénées centrales françaises. Thèse 3ème cycle Sciences Agronomiques, Institut National Polytechnique de Toulouse. pp.132.

Bjornn, T.C. (1971). Trout and salmon movements in two Idaho streams as related to temperature, food, stream flow, cover and population density. *Transactions of the American Fisheries Society.* 100(3) pp.423-438.

Bovee, K.D. (1978). Probability of Use Criteria for the family Salmonidae. FWS/OBS-78/07. U.S Wildl.Serv., Coop. Instream Flow Serv. Grp., Fort Collins, CO. pp.80.

Bovee, K.D. (1982). A guide to stream habitat analysis using the instream flow incremental methodology. FWS/OBS-82/26. U.S Wildl.Serv., Coop. Instream Flow Serv. Grp., Fort Collins, CO. pp.248.

Bovee, K.D. (1988) Use of the Instream Flow Incremental Methodology to evaluate influences of microhabitat variability on trout populations in four Colorado streams. Proceedings of the Western Division American Fisheries Society, Albuquerque, NM. July 10-13, 1988.

Chapman, D.W. (1966). Food and space as regulators of salmonids populations in streams. *Am. Nat.*, 100(913) pp.345-357.

Cheslak, E.F. and A.S. Jacobson (1990). Integrating the instream flow incremental methodology with a population response model. *Rivers.* 1(4) pp.264-288.

Christensen, S.W., D.L. DeAngelis and A.G. Clark (1977). Development of a stock-progeny model for assessing power plant effects on fish populations. Pergamon Press, New York, 1977. pp.196-226.

Elliott, J.M. (1994). Quantitative ecology and brown trout. Oxford Series in Ecology and Evolution. Oxford Universty Press. pp.286.

Euzenat, G. and F. Fournel (1976). Recherches sur la truite commune (Salmo trutta L.) dans une rivière de Bretagne, Le Scorff. Thèse 3ème cycle Biologie Animale, Université de Rennes. pp.215.

Faucher, C., C. Sabaton and F. Travade (1983). Utilisation d'un modèle matriciel de dynamique des populations pour la recherche de la réponse d'une population de poissons à diverses perturbations. Application à la sole. Rapport EDF-DER. pp.41.

Hobbs, D.F. (1940). The Horokiwi stream : a study of a trout population. *Fish. Bull. N. Z.* 10 pp. 238.

Horst, T.H. (1977). Use of a Leslie Matrix for assessing environmental impact with an example for fish population. *Trans.Am.Fish.Soc.* 196 pp.253-257.

Horst, T.H.,(1978). Mathematical modeling of power station impacts on fisheries resources in the United States. IFIP Working Conference on modelling and simulation of land air and water resources systems. North Holland Publishin Compagny.

Jager, H.I., D.L. DeAngelis, M.J. Sale, W. Van Winkle, D.D. Schmoyer, M.J. Sabo, D.J. Orth, and J.A. Lukas (1993). An individual-based model for smallmouth bass reproduction and young- of- year dynamics in streams. *Rivers* 4 pp. 91-113

Lauters, F. (1995). Impacts sur l'écosystème aquatique de la gestion par éclusées des ouvrages hydroélecriques. Thèse 3ème cycle Hydrobiologie, Université Paul Sabatier de Toulouse. pp. 189.

Leslie, P.H. (1945). The use of matrices in certains populations mathematics. *Biometrika.* 33 pp.183-212.

Maisse, G. and J.L. Baglinière (1990). The biology of brown trout, *Salmo trutta* L., in the river Scorff, Brittany: a synthesis of studies from 1973 to 1984. *Aquaculture and fisheries Management.* 21 pp.95-106

Maisse, G. and J.L. Baglinière (1991). Biologie de la truite commune (*Salmo trutta L.*) dans les rivières françaises *in* Baglinière J.L., Maisse G., *La truite, biologie et écologie,* INRA Publ., Paris. pp.25-46.

Mills, D. (1971). Salmon and trout : a resource, its ecology, conservation and management. New York, St. Martin's Press pp.351.

Nehring, R.B. and R.M. Anderson (1993). Determination of Population-limiting Critical Salmonids Habitats in Colorado Streams Using the Physical Habitat Simulation System. *Rivers* 4(1) pp.1-19.

Phillipart, J.C. (1977). Contribution à l'hydrologie de l'Ourthe. Dynamique des populations et production de quatre espèces de poissons cyprinidae. *Barbus barbus* (L.), *Leuciscus cephalus* (L.), *Chondrosroma nasus* (L.) et *Leuciscus leuciscus* (L.). Thèse de 3ème cycle. Université de Liège.

Raleigh, R.F., L.D. Zuckerman and P.C. Nelson (1986). Habitat Suitability Index models and Instream Flow Suitability curves : Brown trout, revised. U.S. Fish. Wild. Serv. Bio. Rep. 82 (10-124). pp.65.

Rietjens J., J. Gailhard, Ph. Gosse, K. Malatre, T. Royer, C. Sabaton, F. Travade (1994). Le logiciel AGIRE -Gave-Echez-Version 1.0- Exemples d'application - Rapport EDF- HE-31/94/018

Serchuk, F.M., C.J. Schmitt and B. Floyd (1980). Rainbow trout : a population simulation based on individual responses to varying environmental and demographic parameters. *Env. Bio. Fish.* 5(1) pp.15-26.

Souchon, Y., F. Trocherie , E. Fragnoud and C. Lacombe (1989). Les modèles numériques des micro-habitats des poissons : application et nouveaux développements. *Revue des sciences de l'eau.* 2 pp.807-830.

Travis , C.C, W.M. Post, D.L. DeAngelis and J. Perkowski (1980). Analysis of compensatory Leslie matrix models for competing species. *J. Theoret. Biol.* 18(1) pp.16-30.

Van Winkle, W., B.W. Rust, C.P. Goodyear, S.R. Blum and P. Thall (1974). A striped bass population model and computer programs. ONRL/TM-4578. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Williamson, S.C., J.M. Bartholow and C.B. Stalnaker (1993).Conceptual model for quantifying pre-smolt production from flow-dependent physical habitat and water temperature. *Regulated Rivers : Research & management-1993.* 8 pp.15-28.

Assessing Impacts and Predicting Restoration Benefits of Flow Alterations in Rivers Developed for Hydroelectric Power Production

Mark B. Bain
New York Cooperative Fish and Wildlife Research Unit
Department of Natural Resources, Cornell University
Ithaca, New York 14853 USA
MBB1@Cornell.edu

Vincent T. Travnichek
Alabama Cooperative Fish and Wildlife Research Unit
Department of Fisheries and Allied Aquacultures
Auburn University, Auburn, Alabama 36849 USA
VTravnic@ag.auburn.edu

ABSTRACT

Altered flow regimes downstream of hydroelectric facilities appear to have degraded river habitats and modified fish communities in most regions of the world. Research during the 1980's on fish communities and habitat in flow regulated rivers of the Northeastern United States provided the basis of a general hypothesis of regulated streamflow effects on riverine fishes. This hypothesis predicts that flow regulation would most strongly effect fish restricted to shallow shoreline microhabitats, that species composition would be dominated by habitat generalist in flow-altered reaches, and that a gradient of change in community composition would be found below hydroelectric dams. These predictions were largely confirmed in studies of a large Southeastern USA river with extensive hydroelectric development. The most pronounced finding was a clear pattern of fish faunal recovery in shoreline habitats downstream of the hydroelectric dam. Larval fish nursery habitat in shoreline areas was also found to be sensitive to flow regulation with families dominated by fluvial species being the most responsive. Deep, channel habitats were relatively insensitive to flow regulation with similar species composition in regulated and unregulated river reaches. Evidence from a series of studies on a regulated Southern USA river were used to justify enhanced river flows for the purpose of restoring a diverse riverine fish fauna. Monitoring of the fish fauna following the enhanced flow regime showed that a natural riverine fauna was largely restored. Species richness doubled in sensitive shoreline habitats with restored species being largely those specializing on flowing water habitats. Overall, the original model of regulated flow impacts from the Northeast US was supported by field studies in the Southeastern United States, and predicted faunal restoration was confirmed by field a study after enhancing river flows.

KEY-WORDS: hydroelectric dams / regulated streamflow / fish communities / fluvial species / habitat generalists / riverine fish faunas / community restoration / larval fish / species diversity

INTRODUCTION

The United States has sustained a major loss in the diversity of fish in rivers and streams (Hughes and Noss, 1992) and habitat degradation is a primary cause of this decline (Allan and Flecker, 1993). One of the most pervasive forms of habitat degradation in rivers is the modification of natural flow (Benke, 1990; Ward and Stanford, 1983) by the discontinuous and erratic water releases from hydroelectric dams. There is clear evidence that highly regulated flows alter stream communities (Cushman, 1985; Petts, 1984). Many existing hydroelectric dams in the United States are currently being reevaluated to determine the need for environmental controls and mitigation measures. This process (relicensing) provides an opportunity for environmental agencies to request flow regimes to enhance and restore riverine habitats and fish communities.

Bain et al. (1988) derived a model that reflected a simple pattern between the fish community in a Northeast USA river (West River, Vermont) and the available instream habitat. An abundant and diverse assemblage of small-fish species and size classes were restricted to microhabitat characterized as relatively shallow in depth, slow in current velocity, and concentrated along stream margins in riffles and pools. These shoreline habitats harbored over 90% of all fish and most of the species in the river. This fish community-habitat model conflicted with the traditional view of streams as a linear sequence of riffle and pools habitats by emphasizing a shoreline-midstream orientation.

The fish community-habitat model was applied to a highly flow regulated Northeast USA river (Deerfield River, Massachusetts) that was heavily developed for hydropower production (Bain et al., 1988). The normally abundant and diverse shoreline fish assemblage was reduced in river reaches with highly regulated flows and absent at sites with the greatest extent of flow fluctuations. Fish species and size classes that used either a broad range of habitat, or a microhabitat that was concentrated in midstream areas (deep, fast, or both), were found in elevated densities as a group and peaked in abundance at the most flow regulated sites. These findings indicated that frequent and high flow variability imposed functional habitat homogeneity. The reduction and elimination of the shoreline fishes under fluctuating habitat conditions indicated that this assemblage is not able to effectively persist in their particular microhabitat even though it physically existed at all stream discharges. Without the functional availablity of shallow, slow, shoreline habitats, the stream environment became one general type of usable habitat which was dominated by the few habitat generalists and those species specializing on channel habitats. Further, Bain et al. (1988) noted a downstream gradient of flow regulation impacts on the river fish community that was most pronounced near a hydroelectric dam and diminished well downstream where flow fluctuations had become attenuated.

Bain and Boltz (1989) combined the findings from Northeast USA rivers with a review of literature on altered streamflow effects on fish communities to develop a hypothesis of how regulated flows from hydroelectric dams would change fish communites in large streams and rivers. Their regulated flow impact hypothesis can be concisely stated for fish as follows: fluctuating streamflows change the densities and species composition of fish differently in shoreline and midstream habitats, and the extent of change depends on the severity of flow regime alteration and distance downstream of hydroelectric dams (Figure 1). Based on this regulated flow impact hypothesis, Bain and Boltz (1989) made some specific predictions: (1) fluctuating streamflows reduce the diversity and abundance of fish in shoreline habitats; (2) fluctuating streamflows will not markedly effect the abundance of midstream fish although species

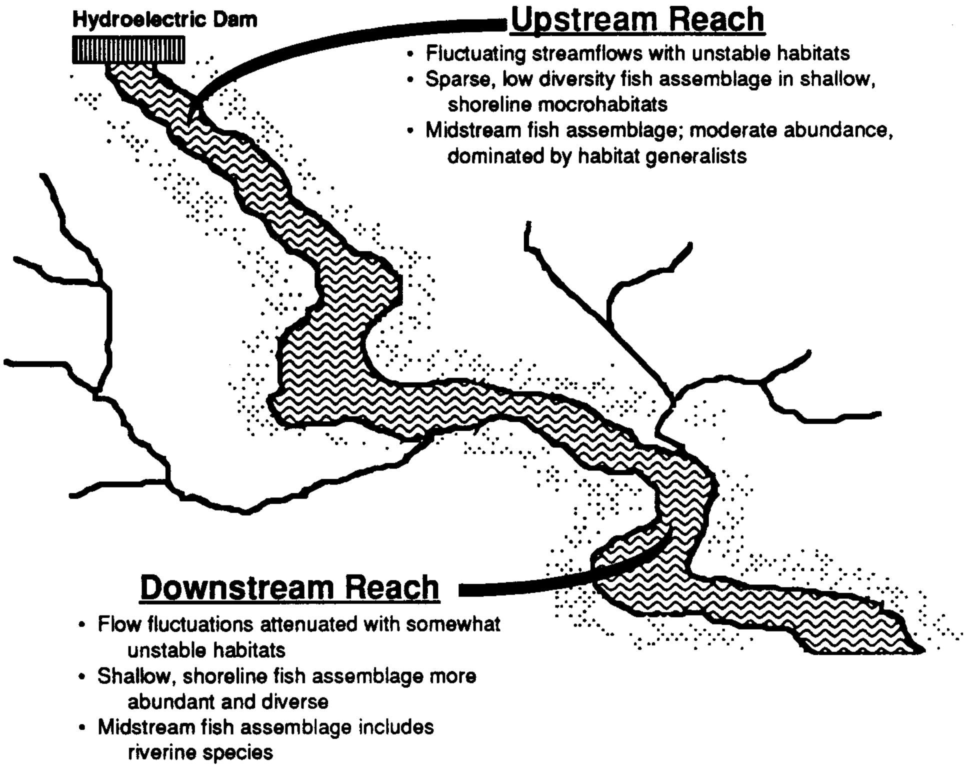


Figure 1. Fish community and habitat relations predicted by the regulated flow impact hypothesis.

composition will be dominated by habitat generalists; and (3) a gradient in species composition and abundance of fish will be evident as the effects of flow regulation diminish downstream.

The regulated flow impact hypothesis was used to design a series of studies on a large Southeast USA river extensively developed for hydroelectric power production and being reviewed for instream flow controls as part of dam relicensing. The highly flow-regulated Tallapoosa River is a tributary of the Alabama River with a mean annual discharge of 135 m^3/s and a watershed largely forested, moderate in relief, and mild in climate. The Tallapoosa River has high fish diversity (125 recorded species) and good water quality characterized as circumneutral (acidity), soft to moderately soft, and low to moderate in specific conductance (Kinsolving and Bain, 1993). Power production by Thurlow Dam (focus of most studies and a relicensing action) occurs during some part of most weekdays and lasts various lengths of time. About 230 m^3/s of water was usually released during periods of power generation (typically a few hours long), and no water was released (leakage flow about 2 m^3/s) during non-generating periods. As a consequence, the Tallapoosa River downstream of Thurlow Dam alternates between lentic conditions and swift, turbulent water with a stage elevated more than 2 m in the channel. Gradually, tributary discharge adds water to the Tallapoosa River and at a point 38-km downstream river discharge does not fall below 5.7 m^3/s and stage variation was about 1 m. Therefore, the Tallapoosa River study reach has a distinct flow regulation gradient downstream from Thurlow Dam. Other large dams iin the basin have similar discharge patterns and flow regulation gradients where unimpounded river reaches occur.

SHORELINE FISH ASSEMBLAGES

Beginning in 1988, Kinsolving and Bain (1993) began testing the first and third predictions of the regulated flow impact hypothesis: fluctuating streamflows reduce the diversity and abundance of fish in shoreline habitats, and a gradient in species composition and abundance of fish will be evident as the effects of flow regulation diminish downstream. An a priori habitat use classification was developed since there was no advance knowledge of which species were specializing on shallow shoreline habitat and which may use these microhabitats as part of a more general instream distribution. All species were categorized as fluvial specialists or macrohabitat generalists using information on habitat use and distribution compiled from ichthyological references such as Scott and Crossman (1973), Pflieger (1975), and Becker (1983). Species classified as fluvial specialists were almost always reported from streams and rivers and were often described as requiring flowing-water habitats throughout life. Macrohabitat generalists included those species that were commonly found in lakes, reservoirs, and streams and were able to complete their life cycle in any of these systems. For this study, the macrohabitat generalist category includes species that require access to streams or flowing-water habitats for a specific life stage but otherwise are commonly found in lakes and reservoirs (two species), An extensive list of species classifications under this system is available from M. Bain.

Fish were collected at 240 randomly located shoreline sampling sites over a 66-km study reach below Thurlow Dam on the Tallapoosa River. Sampling was done with pre-positioned area electrofishers similar to the description in Bain et al. (1985). Electrofishers were positioned parallel to shore and connected to the power supply by a 30-m extension cord so the sampler could be electrified without disturbing fish. The number of fish (by species) collected at each sample site was transformed to an octave scale: <2 fish = 1, ≥ 2 to <4 = 2, ≥ 4 to <8 = 3, ≥ 8 to <16 = 4, ≥ 16 to <32 = 5, ≥ 32 to <64 = 6, etc. This transformation is similar to a logarithmic scale to the base 2, and it is effective for reducing typically high variability in organism counts to an intermediate range of abundance values (e. g., 0-10; Gauch, 1982). The transformation is appropriate because community level studies need to balance the overwhelming numerical influence of dominant species with the information from the occurrence of uncommon species. In addition, fish schooling behavior and variance associated with microhabitat-level samples contribute to great variations in species abundances. Sampling sites were pooled into 10 sections per river reach to develop fish assemblage gradients. The number of sections (10 per river) was chosen, a priori, following the general experience of Gauch (1982) that an intermediate range of observations frequently capture most available information. Longitudinal gradients of change in the characteristics of the fish assemblage was tested with linear regression where river section was the independent variable and mean fish assemblage abundance (transformed numbers by habitat-use group) was the dependent variable. Changes in species diversity were tested using the number of species (species richness) in each river section.

A total of 16,183 fish of 44 species were collected. Almost all were less than 100-mm total length and were either juveniles of large-bodied species (e. g., largemouth bass, *Micropterus salmoides*) or juveniles and adults of small-bodied species (e. g., mosquitofish, *Gambusia affinis*). Similar numbers of macrohabitat generalists (20) and fluvial specialists (24) were recorded in the study. Overall, the patterns of fish abundances and species distributions were highly variable on the Tallapoosa River, with only a few, sparsely collected fluvial species present in low numbers near the dam. The mean abundance of macrohabitat generalists did not change as a function of river section, but there was a significant linear trend of increasing abundance for fluvial specialists along the Tallapoosa River. Regression of the number of generalist species as a function of river section provided no evidence that a linear trend existed along Tallapoosa River. In contrast, the number of fluvial specialists significantly increased along the Tallapoosa River.

The lack of a distinct and statistically significant gradient in the abundance and diversity of macrohabitat generalists along the Tallapoosa River downstream of Thurlow Dam was consistent with the hypothesis that flow fluctuations have little effect on species with broad habitat requirements. Habitat generalists varied in abundance and diversity without any pattern corresponding to a flow regime. Unlike the habitat generalists, a longitudinal gradient was evident for fluvial specialists. The abundance and diversity of fluvial specialists increased as flow fluctuations became attenuated downstream of the Thurlow Dam. Sections with the most variable flows had fluvial fish abundances so low that explanations aside from disturbance effects seem unlikely. The gradient in fluvial specialists on the Tallapoosa River provided evidence that large, daily flow fluctuations cause a disturbance gradient in the Tallapoosa River. The most fluctuating river sections had a fish assemblage with fewer fish and species than less regulated sections downstream. Along the disturbance gradient, habitat generalists provided a base of fish and species that are variable in numbers and composition, but remain a relatively constant assemblage component. Fluvial specialists supplemented this base and gradually diversified and augmented the fish assemblage. Over the 66-km study reach on the Tallapoosa River downstream of Thurlow Dam, the abundance of fish more than doubled and the diversity increased by almost that amount. These increases corresponded with the diminishing severity of flow regulation and they were entirely caused by increasing abundance and diversity of fluvial specialist species.

LARVAL FISH NURSERY HABITAT

Larval fish were not specifically addressed in the regulated flow impact hypothesis, but their nursery habitat is concentrated in shoreline waters in many streams and rivers. The requirements of fish larvae are often quite different from juveniles and adults (Snyder, 1990) and they often respond differently to disturbances (Copp, 1989; Schlosser, 1985). Consequently, Scheidegger and Bain (1995) investigated larval fish assemblage composition in nursery habitat along the Tallapoosa River downstream of Thurlow Dam. Under the regulated flow impact hypothesis, larval fish assemblages should have been responsive to flow regulation in shoreline waters, and the level of response should have been diminished with distance downstream from the hydroelectric dam.

Study sites (250 to 300-m long) was selected 5 and 50-km downstream of Thurlow Dam on the Tallapoosa River. Larval fishes were collected at each site on days of typical seasonal flow every two weeks from late March through September 1988 and late February through August 1989. Frequent flooding reduced sampling at the upstream Tallapoosa River study site in 1989. We assumed that larval fish could have been selectively using specific microhabitats (nursery habitat) when current velocity was less than their maximum sustained swimming speed. Scheidegger and Bain (1995) calculated a maximum current velocity of 8.4 cm/s (i.e., 7 body lengths x 12-mm long larvae) to define the upper velocity limit for potential nursery habitat using length data on larvae of the families likely to be encountered in the Tallapoosa River. A 1-m^2, 500-µm mesh seine was used to collect larval fish in nursery habitat. Data were analyzed at the family level to reduce the possibility of inaccurate conclusions from misidentifications, and to accommodate poor representation of some species.

A total of 2,769 larval fish were collected in 527 seine samples. The mean catch per sample (3.8 larvae/seine sample) was lowest at the upstream Tallapoosa River site, with a much higher (7.0 larvae/sample) catch rate at the downstream site. The two sites were distinct in assemblage composition in both years. In 1988, Centrarchids (*Pomoxis* and *Lepomis*) accounted for 98%of the fish at the upstream Tallapoosa River site, and 25% at the downstream site. The downstream site in 1988 was dominated by Percidae (44%) with a substantial number of Cyprinids (26%). Catostomids were absent from upper site samples in 1988, and were scarce at the downstream site. In 1989, Catostomids and Centrarchids were equally well represented (36% each) at the upstream site, and Percids (61%) and Catostomids (21%) were well represented at the downstream site.

Centrarchids clearly dominated the larval fish collections at the upstream site during 1988 when flow regulation was well established throughout the sampling season. Centrarchids also comprised a substantial portion of the collections at the downstream site. This family is exclusively habitat generalists (species recorded for the river) and they apparently spawned well and persisted as larvae when the Tallapoosa River study reach was most often lentic with high discharges for a few hours a day. Percids dominated the downstream study site in 1988, with Percidae being almost exclusively fluvial specialists. In 1989, river discharge sometimes exceeded the capacity of the Tallapoosa River hydroelectric dams, and abundant river flows regularly resulted in frequent day-long water releases. The Catostomids and Percids appear to have produced abundant larvae that persisted in shoreline habitat when the river had prolonged strong currents throughout their channels. Most of the Catostomid species commonly recorded in the Tallapoosa River are fluvial specialists. Centrarchids declined greatly in representation in the larval fish assemblage during this period. Percids and Catostomids originally comprised a major portion of the river fish fauna in unaltered Southeast USA rivers, while Centrachids were originally a minor component (Swingle, 1953). The shift in family composition between sites and study years corresponds with the availability of flowing water habitat in the river and shoreline waters.

In the Tallapoosa River, there was a substantial increase in shoreline seine catch at the downstream site relative to the upstream site. This pattern of abundances among the Tallapoosa River larval fish study sites is consistent with the first and third predictions of the regulated flow impact hypothesis. Strong flow regulation by Thurlow Dam appears to have impaired the function of shallow, shoreline waters as larval fish nursery habitat or it may have degraded the spawning conditions in the river channel. The larval fish family composition was also consistent with the predictions of the regulated flow impact hypothesis. When flow regulation was large and persistent (1988), the larval fish assemblage at the most flow regulated site was dominated by generalist family Centrarchidae. In contrast, when river discharge was high and more regular, the largely fluvial Catostomid and Percid families dominated the larval fish collections in shoreline waters. Overall, the larval fish abundances and taxa composition indicates that the larval fish assemblage and nursery function of shoreline waters can be altered by flow regulation.

CHANNEL FISH ASSEMBLAGES

Travnichek and Maceina (1994) examined the effects of flow regulation on fish assemblages in deep, channel habitats and shallow, shoreline habitats of regulated and unregulated reaches of the Tallapoosa River, Alabama. Their findings for shoreline waters were consistent with Kinsolving and Bain (1993). However, their investigation of fish in deep water habitats was a direct test of the second prediction of the regulated flow impact hypothesis: fluctuating streamflows will not markedly effect the abundance of midstream fish although species composition will be dominated by habitat generalists.

Deep, channel habitats in the Tallapoosa River was sampled from 1990 through 1992 with a pulsed DC electrofishing boat at random locations within 2-km study sites that were considered either regulated (near hydroelectric dams) or unregulated. Travnichek and Maceina (1994) collected 64 species and 2,715 fish in channel waters. They found little evidence that flow regulation effected species richness or diversity in these habitats because there were negligible differences among sites. However, there were significant differences among regulated and unregulated sites in terms of catch rates and relative abundance of common fluvial species and habitat generalists. The catch rates for fluvial specialist species of the Catostomid family was significantly higher in unregulated sites. Centrarchids, nearly all of which were fluvial specialists, dominated (mean 60%) channel habitat assemblages at regulated sites, but this family had a less pronounced (mean 40%) presence at unregulated sites.

Travnichek and Maceina (1994) concluded that flow regulation had relatively little effect on fish using deep, channel habitats as proposed by the regulated flow impact hypothesis, and that predicted effects were clearly evident in shoreline habitat as documented by Kinsolving and Bain (1993). However, Travnichek and Maceina (1994) showed that small differences between regulated and unregulated sites can be detected in terms of species richness and diversity in deep, channel habitats, and that the abundance of large fluvial species can be impacted by flow regulation.

RESPONSE TO FLOW ENHANCEMENT

The regulated flow impact hypothesis described how river fish communities respond to flow regulation on a small (shoreline versus deep water habitats) and large (upstream-downstream disturbance gradient) scale. Implicit in this pattern is the expectation that enhanced river flows will lessen the effects at both the microhabitat and riverwide scale. However, there is little empirical evidence that increased minimum flows and reduced flow fluctuations below hydroelectric dams results in more abundant and diverse river fish communities in any river system. After the studies reviewed above were completed, a continuous minimum water release of 34 m^3/s from Thurlow Dam was initiated as part of the relicensing agreement of this hydroelectric project. The enhanced flow regime was implemented and evaluated to determine if the expected environmental benefits were realized. The evaluation study was conducted and reported by Travnicheck et al. (1995) on the Tallapoosa River below Thurlow Dam. Travnicheck et al. (1995) specifically tested the predictions that enhanced flows would: (1) increase the diversity and abundance of shoreline fishes; (2) shift assemblage composition from macrohabitat generalists to fluvial specialists; and (3) diminish the gradient of effects downstream from the dam. The study focused on the shallow, shoreline fish assemblage because these habitats were previously found to be most sensitive to regulated flow impacts on fish.

Before 1991, river discharge below Thurlow Dam fluctuated frequently and ranged from near 0 to about 230 m^3/s. The enhanced flow regime that began in 1991 did not eliminate flow fluctuations but it reduced the severity of flow regulation. Minimum river discharge near the dam rarely dropped below 34 m^3/s and flowing water microhabitats were always present throughout the river. The shoreline fish assemblage was depauperate downstream of Thurlow Dam prior to flow enhancement (Kinsolving and Bain, 1993; Travnicheck and Maceina, 1994). After the river flows were increased, species richness more than doubled and the mean abundance of fish per sample increased 500%. Samples without fish dominated (70%) the collections prior to the enhanced flow regime, whereas after it began, samples without fish decreased to 30%. The eight species recorded in the Tallapoosa River downstream of Thurlow Dam were macrohabitat generalists prior to the enhanced flows. One year after increased dam discharges, the shoreline fish assemblage was composed of both generalists and fluvial specialists, with the latter accounting for over 70% of the fish collected. The fish assemblage at the downstream site was much more diverse (26 species) than the upstream site before and after the enhanced flow regime, and the occurrence of empty samples remained constant (13%). Before the enhanced flow regime, macrohabitat generalists accounted for the majority (60%) of the fish collected downstream, whereas one year after the enhanced flows the relative abundance of fluvial specialist species increased to over 80% of all fish collected.

The enhanced flow investigation by Travnicheck et al. (1995) indicated the greatest changes in assemblage composition and gains in fish abundance and diversity were attributable to restoring fluvial species that dominate collections in many streams and rivers. The less pronounced effect of the enhanced flows was on the abundance and diversity of macrohabitat generalists. These findings are consistent with the regulated flow impact hypothesis in terms of fish community improvements (fluvial specialists) and community components receiving marginal benefits (macrohabitat generalists). Thus, the results of Travnicheck et al. (1995) indicate that the enhanced flow regime succeeded in improving habitat conditions and restoring a diverse fish assemblage more reflective of a riverine system.

CONCLUSIONS

The regulated flow impact hypothesis was initially motivated from the detailed study of impacts on a fish community in a small, species-poor river (Deerfield River, Massachusetts) highly developed for hydropower production in the Northeast USA. The hypothesis was later refined with a review of literature on flow regulation effects to provide specific predictions of impacts in other river systems. These predictions were tested in a new region (Tallapoosa River, Southeast USA) where river fish communities are complicated by high species diversity. The series of Tallapoosa River studies largely supported predictions of the hypothesis (Figure 2), and their results justified targeting specific components of a river fish community under flow regulation stress. Thus, the regulated flow impact hypothesis appears useful for designing assessments of impacts from flow regulation. The hypothesis also appears useful in developing justifications for enhanced river flows because the anticipated restoration of a more riverine fish fauna was supported in the study evaluating the instream flow enhancement on the Tallapoosa River.

Much of the hydroelectric generating capacity of the United States was developed prior to knowledgeable consideration of the impacts on fish. With the re-evaluation (relicensing) of many of these projects now underway, the hypothesis is especially useful for justifying and verifying environmental benefits of moderated flow regulation. The goal of the regulated flow impact hypothesis and the research reviewed here was to promote the conservation of riverine fish faunas. Restoration with this aim is by definition a

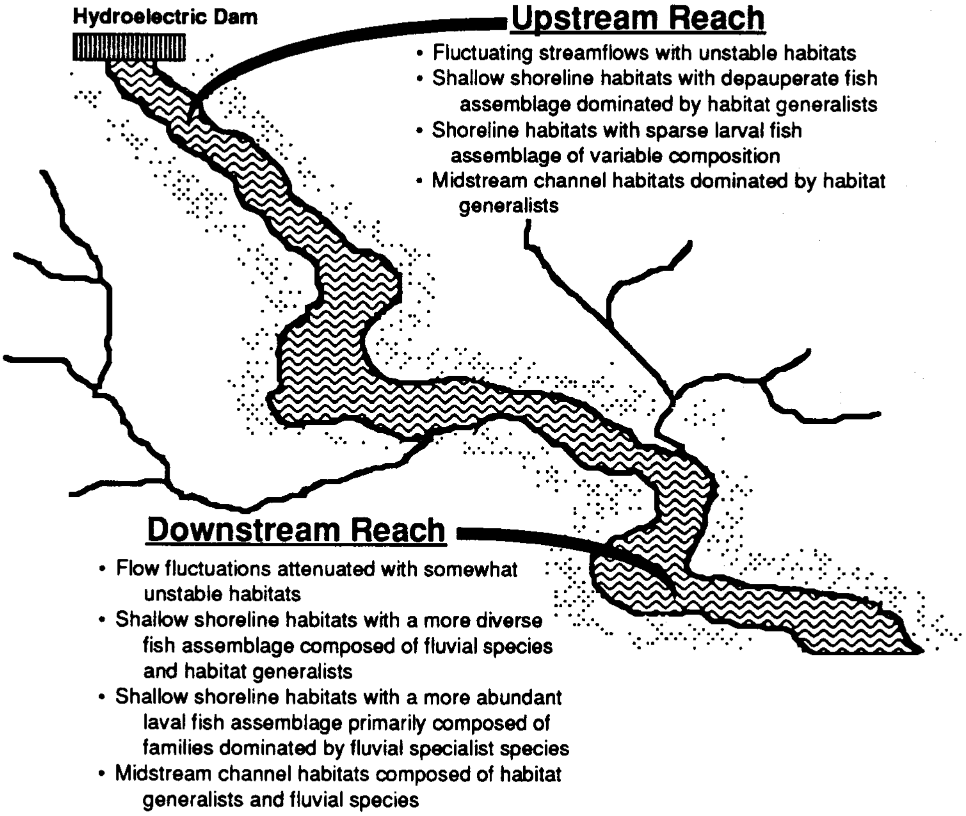


Figure 2. Fish community and habitat relations identified in studies on the Tallapoosa River, Alabama.

community-level endeavor. The most sensitive measure of regulated flow effects appear to be the response of the fluvial specialist component of fish assemblages in shoreline habitats. This fish-habitat unit has been repeatedly found to be the most sensitive to flow alterations. Therefore, maintenance of functioning, shoreline habitats capable of supporting a diverse array of generalist and specialist species covering all life stages appears adequate as a measure of conditions needed to maintain diverse riverine fish faunas.

REFERENCES

Allan, J. D., and A. S. Flecker. 1993. Biodiversity conservation in running waters. *BioScience* 43:32-43.

Bain, M. B., and J. M. Boltz. 1989. Regulated streamflow and warmwater stream fish: a general hypothesis and research agenda. U. S. Fish and Wildlife Service Biological Report 89(18).

Bain, M. B., J. T. Finn and H. E. Booke. 1985. A quantitative method for sampling riverine microhabitats by electrofishing. *North American Journal of Fisheries Management* 5:489-493.

Bain, M. B., J. T. Finn and H. E. Booke. 1988. Streamflow regulation and fish community structure. *Ecology* 69:382-392.

Becker, G. C. 1983. Fishes of Wisconsin. University of Wisconsin Press, Madison, Wisconsin, USA.

Benke, A. C. 1990. A perspective on America's vanishing streams. *Journal of the North American Benthological Society* 9:77-88.

Copp, G. H. 1989. The habitat diversity and fish reproductive function of floodplain ecosystems. *Environmental Biology of Fishes* 26:1-27.

Cushman, R. M. 1985. Review of ecological effects of rapidly varying flows downsteam of hydroelectric facilities. *North American Journal of Fisheries Management* 5:330-339.

Gauch, H.G. 1982. Multivariate Analysis in Community Ecology. Cambridge University Press, Cambridge, England.

Hughes, R. M., and R. F. Noss. 1992. Biological diversity and biological integrity: current concerns for lakes and streams. *Fisheries* 17:11-19.

Kinsolving, A. D., and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. *Ecological Applications* 3:531-544.

Petts, G. E. 1984. Impounded rivers. John Wiley and Sons, New York, New York, USA.

Pflieger, W. C. 1975. Fishes of Missouri. Missouri Department of Conservation, Jefferson City, Missouri, USA.

Scheidegger , K. J., and M. B. Bain. 1995. Larval fish distribution and microhabitat use in free-flowing and regulated rivers. *Copeia* 1995:125-135.

Schlosser, I. J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. *Ecology* 66:1484-1490.

Scott, W. B., and E. J. Crossman. 1973. Freshwater fishes of Canada. *Fisheries Research Board of Canada Bulletin* 184.

Snyder, D. E. 1990. Fish larvae – ecologically distinct organisms. Pp. 20-23 In M. B. Bain (Editor). Ecology and assessment of warmwater streams: workshop synposis. U. S. Fish and Wildlife Service Biological Report 90(5).

Swingle, H. S. 1953. Fish populations in Alabama rivers and impoundments. *Transactions of the American Fisheries Society* 83:47-57.

Travnicheck, V. H., M. B. Bain, and M. J. Maceina. 1995. Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* 124:836-844.

Travnicheck, V. H., and M. J. Maceina. 1994. Comparison of flow regulation effects on fish assemblages in shallow and deep water habitats in the Tallapoosa River, Alabama. *Journal of Freshwater Ecology* 9:207-216.

Ward, J.V. and J.A. Stanford 1983. The intermediate-disturbance hypothesis: an explanation for biotic diversity patterns in lotic ecosystems. Pp. 347-356 In T.D. Fontaine, III and S.M. Bartell [Editors]. Dynamics of lotic ecosystems. Ann Arbor Press, Ann Arbor, Michigan, USA.

Habitat creation and restoration

Création et restauration d'habitat

ECOHYDRAULICS: CHALLENGES AND OPPORTUNITIES WITH FISH AND FISH HABITAT

Christos Katopodis
Freshwater Institute
Department of Fisheries and Oceans
501 University Crescent
Winnipeg, Manitoba, R3T 2N6
CANADA
Chris.Katopodis@ C-A.DFO.DFO-MPO.X400.GC.CA

ABSTRACT

Rivers and streams are integrated hydraulic systems that provide a variety of suitable and accessible habitats for various fish species. Different species and life stages have adapted to and thrive in rivers and streams characterized by various biological, hydrological, hydraulic and geomorphological features and conditions. An holistic approach however is rarely taken in projects intended to provide access to suitable hydraulic features and habitat, to restore and create habitat, or prevent fish from entering hazardous areas. Habitat restoration and mitigation efforts rely primarily on trial and error approaches, emphasize salmonid requirements, mostly neglect ice conditions, occasionally include elementary geomorphological concepts, and in most cases do not incorporate fisheries and engineering knowledge and judgement concurrently. Upstream and downstream (amphidromic) fish migrations in rivers and streams often are sustained with various fish passage facilities. In many cases though, design criteria reflect only the requirements of one or just a few of the migratory species of a river system. Stream restoration and fish passage can be influenced greatly by the development philosophy adopted, which ranges from a tendecy of taming nature (physiodamastic approach), to a tendecy of worshipping nature (physiolatric approach). These two often polarized philosophies are many times applied to fish habitat mitigation and restoration projects without full regard for scientific objectivity. In most cases difficulties arise from the strict application of either of these approaches. The range between these two benchmark approaches also provides a wealth of other options, which in most cases may be preferred. Maintenance of species diversity (from forage fish to sturgeon) is challenging traditional ways of accommodating fish movements and demands a more holistic approach to fishway design. Better mathematical models for simulating river hydraulics (e.g. two-dimensional vs one-dimensional approaches), and better technology in tracking fish movements and levels of activity (electromyographic telemetry) challenge traditional instream flow models and provide opportunities for more realistic habitat simulations. The fact that empirical efforts for providing suitable and accessible habitats rarely apply a high level of biological and physical information and knowledge to project design and construction, provides hope that projects which take full advantage of both biological and physical principles on an ecosystem basis and integrate site specific knowledge may be more effective. The knowledge of biological and physical processes in a river system, the design philosophy adopted, the materials and construction techniques used, all have a direct effect on effectiveness of fish habitat projects, as well as on the level of harmony with nature achieved.

KEY-WORDS: fish / habitat / stream / restoration / instream flow / fishway / fish passage / telemetry

INTRODUCTION

Habitat restoration and mitigation has become more essential because societies, particularly industrialized ones, are approaching a state of diminishing natural resources and need to recycle, conserve or use resources more wisely. Many impacts of water resource development projects on the ecosystem have been documented (e.g. Gore and Petts, 1989). Effects on riverine fish centre on habitat requirements for a) migration - movements to and from various habitats, b) reproduction - adult spawning and egg incubation, c) juvenile rearing, d) adult growth and maturation, and e) shelter - protection from predators or harsh conditions (pollution, winter under ice, etc.). Habitat restoration and mitigation efforts for riverine fish attempt to address one or more of these life cycle requirements by providing habitat access (for upstream and downstream migrations) and habitat suitability (for reproduction, rearing and shelter). The latter includes habitat morphology - space, cover and substrate, and habitat water - running water, water quality, and food transport. Diverse requirements for different species and life stages of each species need particular attention.

Pristine streams, evolved naturally, provide one set of benchmarks for hydraulic conditions suitable for migration and fish production; extensively modified rivers provide another which in many cases limit migration, fish production or change species composition. Studies of both systems have been used to guide mitigation and restoration projects as well as applied research to better understand relationships between variables describing fish habitat, stream hydraulics, biology, hydrology and geomorphology. Fish habitat restoration and mitigation efforts however, have relied primarily on trial and error approaches. Such efforts have emphasized salmonid requirements, mostly neglected under ice conditions, and only occasionally have included elementary geomorphological concepts. In most cases, fisheries and engineering knowledge and judgement are not incorporated concurrently (Frissell and Nawa, 1992; Gore and Petts, 1989; McPhail et. al., 1992; Meehan, 1991; Power et. al., 1993). Although these empirical approaches in many cases are not successful, they have provided valuable information and experience on successes and failures. A common characteristic of such efforts is that they rarely apply a high level of biological and physical knowledge to habitat mitigation and restoration projects.

Fish habitat projects often meet with low levels of biological effectiveness and structural integrity, two factors that tend to be closely related. Typically 40% - 60% of projects are judged successful from biological and engineering perspectives (Frissell and Nawa, 1992; M. Miles & Associates Ltd., 1995; R.L.& L. Environmental Services Ltd. et. al., 1994). Success rates appear to depend on structure type and age, site specific experience, and the degree of change from average natural values. Biological, geomorphological, hydrological, sediment transport, and hydraulic parameters may affect sucess rates. Recently, design approaches have moved from high capital-low maintenance durable or hard materials (e.g. concrete, metal, heavy rip-rap), shaped in uniform patterns and built to withstand extreme flood, sediment, ice and debris events, to low capital-high maintenance soft or natural materials (e.g. rock, gravel, wood, plants), shaped in more natural stream forms and able to maintain structural integrity at moderate or small flood, ice and debris events (e.g. Newbury and Gaboury, 1993). The survival rate of such structures in large floods and high energy streams is expected to be low. Preliminary results from an inspection of habitat enhancement structures in southwestern Alberta, after the large June 1995 flood (frequency estimates presently range widely anywhere from 1:50 to 1:500), reveal that about 77% of the structures were destroyed or severely damaged (M. Miles, personal communication, April 1996).

Fishways for upstream migrants are considered well-developed for anadromous salmonids and have been used by several freshwater fish (Christensen, 1994; Katopodis, 1995; 1992; 1990; Katopodis et. al., 1991; Office of Technology Assessment, 1995). Failures or inefficiencies tend to result from lack of experience and data for specific species (e.g. sturgeon), suboptimal design criteria, or inadequate facility operation and maintenance. Fish passage systems for downstream migrants include physical barrier devices (various fish screens), structural guidance devices (e.g. louvers), and behavioural exclusion devices (e.g. acoustic array, strobe and mercury lights). Fish screens are considered well-developed for several species, and design criteria and guidelines are available in some cases (Department of Fisheries and Oceans, 1995; Pearce and Lee, 1991). Structural guidance devices are not well understood, yet at a few sites high levels of performance have been reported. Low cost behavioural devices elicit avoidance responses, but tend to be site- and species- specific, have demonstrated uncertain reliability, may result in fish habituation, and are difficult to adapt in flowing waters (KGS Group and North/South Consultants Inc., 1992; Office of Technology Assessment, 1995).

APPROCHES TO FISH HABITAT PROJECTS

Stream restoration can be influenced greatly by the development philosophy adopted. Natural streams provide one set of benchmarks and suggest a design philosophy of mimicking nature (physiomimesis). Extesively modified rivers provide another set of benchmarks and suggest a design philosophy of taming or controlling nature (physiodamasis). These two often polarized approaches are many times applied to fish habitat mitigation and restoration projects without full regard for scientific objectivity. Fig. 1 provides a conceptual outline of these two approaches. The taming nature philosophy (physiodamasis), which has characterized many traditional design and construction methods, fixed structures of "permanent" construction, uniformity, and accommodation of extreme hydrological events, with high capital expenditures and low mantenance, may result in a low level of harmony with nature. The mimicking nature philosophy (physiomimesis), which characterizes more contemporary thinking, flexibility, diversity, higher risk of structural failure with low capital expenditures and higher maintenance, is expected to result in a high level of harmony with nature (Marsden, 1995; Shrubsole, 1994).

In most cases difficulties arise from the doctrinal application of either these approaches. The range between these two benchmark approaches also provides a wealth of other options, which in most cases may be preferred. Fig. 1 also illustrates the fact that different paths may lead to a range of possible and feasible options when applied to the design and construction of restoration projects. These paths begin with the level of existing knowledge and full use of existing information and end in a degree of harmony or a desired level of synchrony with nature. The paths may follow different routes but at critical points along the way they are shaped by the classical Hellenic golden rule: ΠΑΝ ΜΕΤΡΟΝ ΑΡΙΣΤΟΝ (PAN METRON ARISTON) or everything is best in good measure (moderation). This is analogous to the use of ecological engineering and ecotechnology as envisaged by Mitsch and Jorgensen (1989). Under certain conditions either the nature taming or the nature mimicking approach tempered by moderation would be the only possible choices, but more commonly in-between solutions tempered by moderation would be more harmonious. For example, it may not be economically or even environmentally beneficial to remove some existing massive dams, but it may be feasible to restore to near natural conditions channelized streams whose banks were shaped with soil materials.

A high level of harmony with nature, sustainable resource development, and environmentally acceptable river engineering works depend on full integration of relevant engineering and scientific disciplines. A research and development cycle to assist in achieving these objectives includes the following: a) physical hydraulic models provide an understanding of the mechanics of flow in the vicinity of various fish habitat structures and quantify key parameters; b) geomorphological approaches reveal the fluvial habitat framework; c) hydrological analyses establish fish responses to flow, sediment, debris and ice events; d) mathematical modeling of stream flow helps to better simulate hydraulic conditions important to fish; e) field studies and objective biological assessment techniques are used to confirm laboratory or mathematical model results, to establish relationships between fish preferences and hydrological, hydraulic or geomorphological variables, to validate habitat and hydraulic simulation models, and to verify their predictions.

Ecohydraulics is at the centre of holistic research efforts outlined above. Ecohydraulics provide a vital link between fish habitat and river engineering, as well as a mechanism for geomorphological responses to hydrological events. The structure of flow at macro and micro scales is basic to the understanding of fish preferences as well as to the capability of mitigating, restoring, or creating suitable habitats. A study program,

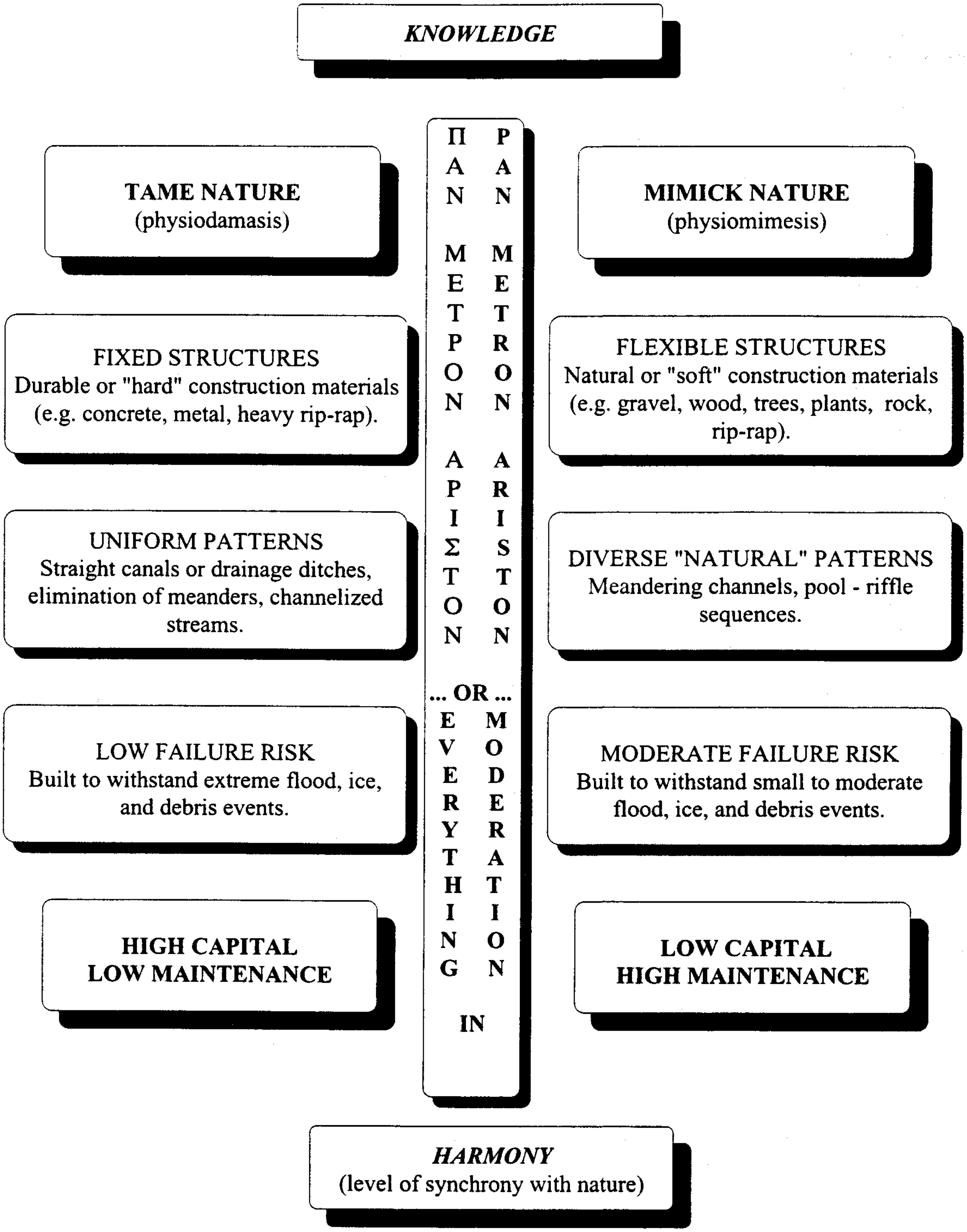


Fig. 1. Approaches to fish habitat projects

spearheaded by the author over a number of years, provides a paradigm of the above ecohydraulic approach and contains components on habitat morphology, habitat water (instream flows), and habitat access for several fish species. Initial efforts on habitat access, particularly fishways for dams (Katopodis, 1995; 1992; 1990; Katopodis et. al., 1995; 1991), were followed by analyses of fish swimming performance or ichthyomechanics (Colavecchia et. al., 1996; Katopodis, 1994; 1992; 1990; Peake et. al., 1995), while more recently studies on habitat structures (Dwivedi et. al., 1993; McPhail et. al., 1992; R.L.& L. Environmental Services Ltd. et. al., 1994; Shamloo et. al., 1995; Trillium Engineering and Hydrographics Inc., 1996; 1995) and instream flows (Ghanem et. al., in press; 1995a; 1995b; Steffler et. al., in press; Waddle et. al., 1996) have been added. Fig. 2 is a map of Canada showing locations of projects outlined in this paper.

HABITAT SUITABILITY

Habitat Morphology

Key natural stream components are defined in Fig. 3 and are used as broad templates in restoring streams as shown in schematics of natural and reconstructed stream segments (Fig. 4 and 5). A study program was initiated to assess the performance of several types of habitat structures for different fish species, hydrologic regions, and hydraulic conditions. The goal of this program is to develop design criteria for habitat structures used by different species and placed in high to low gradient streams. The short record with mostly empirical approaches and general lack of experience with the performance of riverine habitat structures, do not allow for well developed design criteria and standardized construction practices to emerge without significant research efforts. Field observations on several high gradient trout streams, as well as low gradient walleye streams, where habitat structures were constructed in recent years, have led to the selection of field sites for more detailed surveys (R.L.& L. Environmental Services Ltd. et. al., 1994; Trillium Engineering and Hydrographics Inc., 1995; 1996). In parallel, studies on the characteristics of flow near artificial habitat structures, which are critical for biological suitability, were initiated in the laboratory. These studies provide methods to visualize, observe and quantify flow and scour conditions around habitat structures. A systematic approach progressing from simple to more complex structures and testing a variety of designs, rock sizes, flood magnitudes and stream gradients was adopted.

Initial laboratory experiments at the T. Blench Hydraulics Laboratory of the University of Alberta (Dwivedi et. al., 1993) were performed with a single idealized habitat structure located centrally in a rectangular flume. Hemispheres of three different sizes were tested with the ratio of height (h=radius) to depth of approaching flow (d) varying from 0.1 to 3 (Fig. 6). The approach flow Froude number (F) varied from 0.06 to 2.7. Water surface profiles as well as wake characteristics in the vicinity of the hemisphere were studied. For relatively small values of h/d and F, the flow was similar to that of a deeply submerged body. As h/d and F increased, the pattern of flow near the hemisphere and the characteristics of the wake changed significantly. In this case, wakes were observed to extend through the whole depth of flow. Correlations were found for the characteristics of the wake for a range of the flow variables studied. Wake length, normalized to the depth of flow, varied mainly with h/d, increasing as h/d increased. Flow separation occurred at an angle of about 120^{o} from the upstream stagnation point of the body. Water surface profiles as well as wake characteristics in the vicinity of the simple structure were studied.

The above study was expanded to examine a full range of variables, consider scour and deposition, model more realistic habitat structures, collect field data and analyse factors affecting their stability (Shamloo et. al., 1995; Trillium Engineering and Hydrographics Inc., 1996). Experiments conducted in parallel include single objects and multiple objects placed in regular patterns (e.g. a Δ pattern with various distances between the objects). Flow visualization using dye plumes and observations made for different values of the relative depth of flow (h/d), show the existence of two main types of vortices around the obstacles which affect wake characteristics (Fig. 7, 8, 9). A horseshoe vortex system, was dominant mainly in front of the hemisphere and its effects were

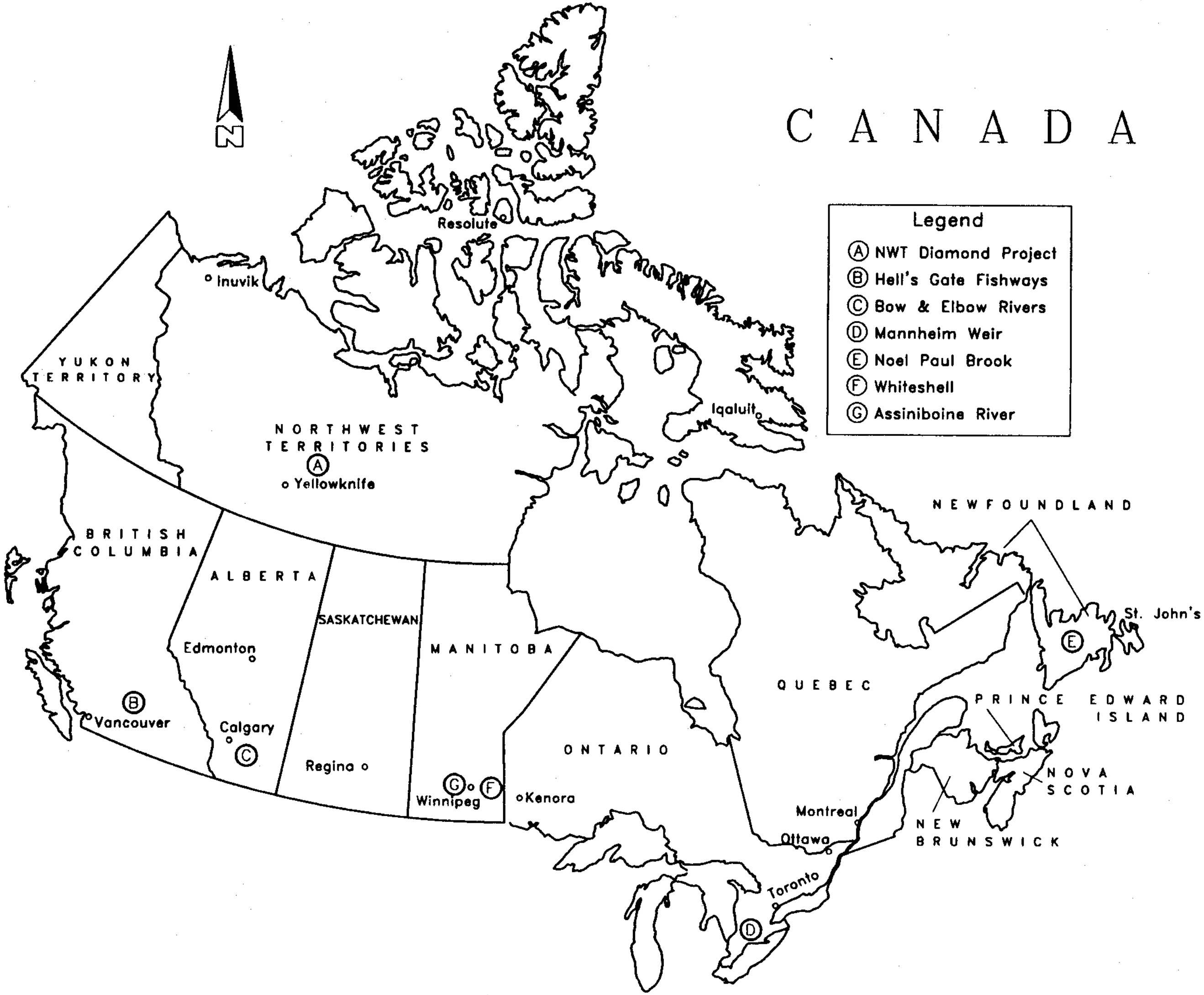


Fig.2 Fish habitat project and study locations.

found to be present only for a short distance downstream. An arch vortex system, which forms as a result of flow separation from the hemisphere, covers the recirculation region behind the obstacle. The velocity fields in the vertical plane of symmetry, measured with a pitch probe, and the velocity field in the wake region, measured with a yaw probe, show the velocity defect in the wake. Preliminary measurements show fast recovery of the velocity in the wake of the hemisphere (Shamloo et. al., 1995).

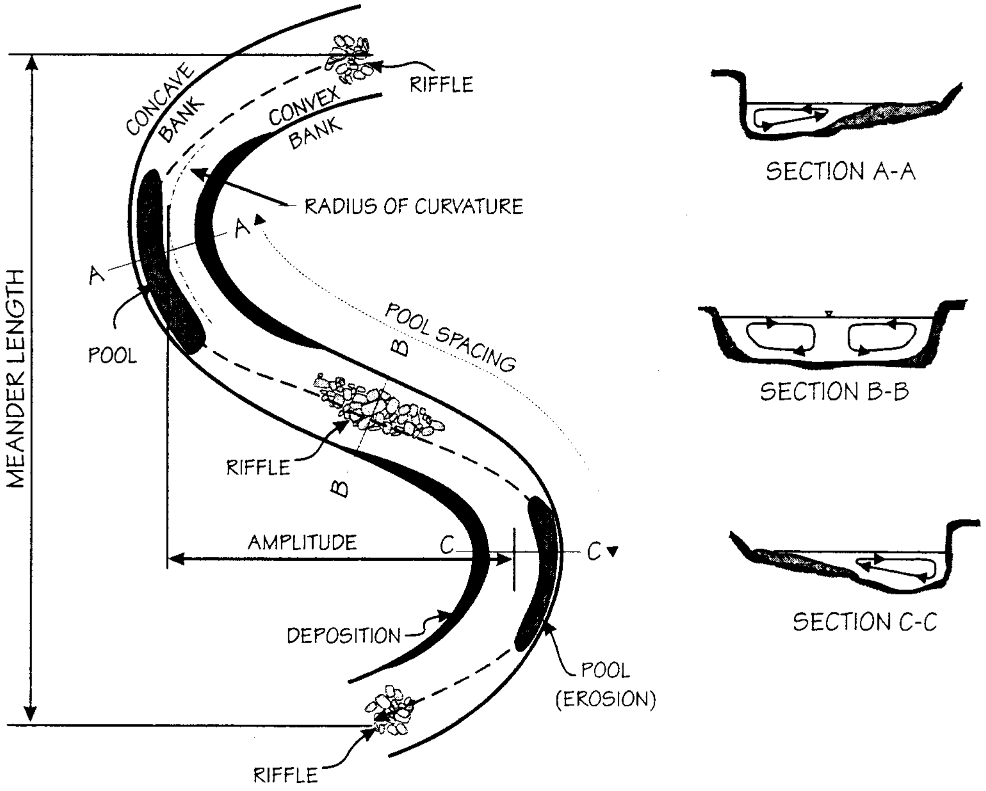


Fig. 3. Typical physical relationships in natural streams: a) pool spacing of 5 to 7 channel widths; b) meander length of 7 to 10 channel widths; c) radius of curvature of 2 to 3 channel widths.

In general, four characteristic flow regimes were observed in the vicinity of the object. For deep submergence (regime 1; d>>h; Fig. 6a and 7), the top layer of flow was not affected by the presence of the object, the lower layer mixed with the recirculation zone, and the flow regime was independent of Froude number. For moderate submergence (regime 2; d>h; Fig. 6b and 7) although the top layer did not mix with the recirculation zone, the object affected the free surface and waves were observed. An increase in the Froude number resulted in more surface waves, for higher relative depths, and in fewer waves for lower relative depths. Usually, wave heights were larger and wave lengths smaller for higher Froude numbers. For depths approximating the height of the object (regime 3; Fig. 8), surface waves were present and mixing occured between the top and lower layers of flow in the recirculation zone. Froude number increases were reflected with fewer, higher and shorter waves. Finally for shallow depths (regime 4; d<h; Fig. 6c and 9), strong backwater flow was observed in the vortex generation zone without the arch-vortex. The vortex generation zone decreased as Froude number increased. The results of these experiments are also used to develop and test a three-dimensional mathematical model to simulate flow around such objects.

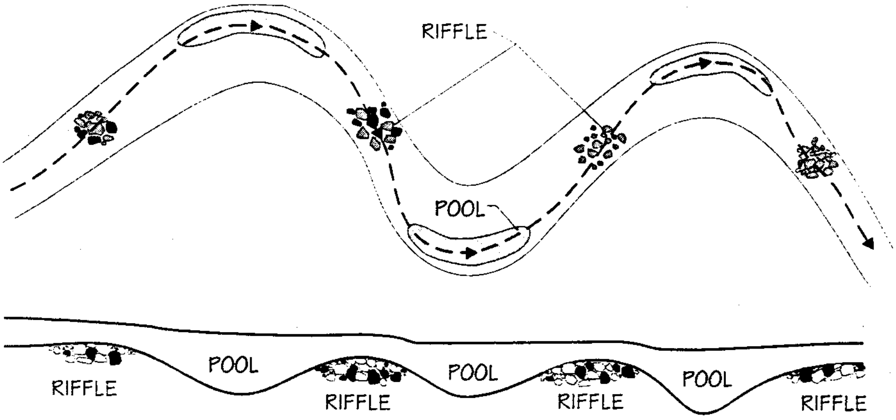


Fig. 4. Schematic of natural meandering watercourse (plan and profile views)

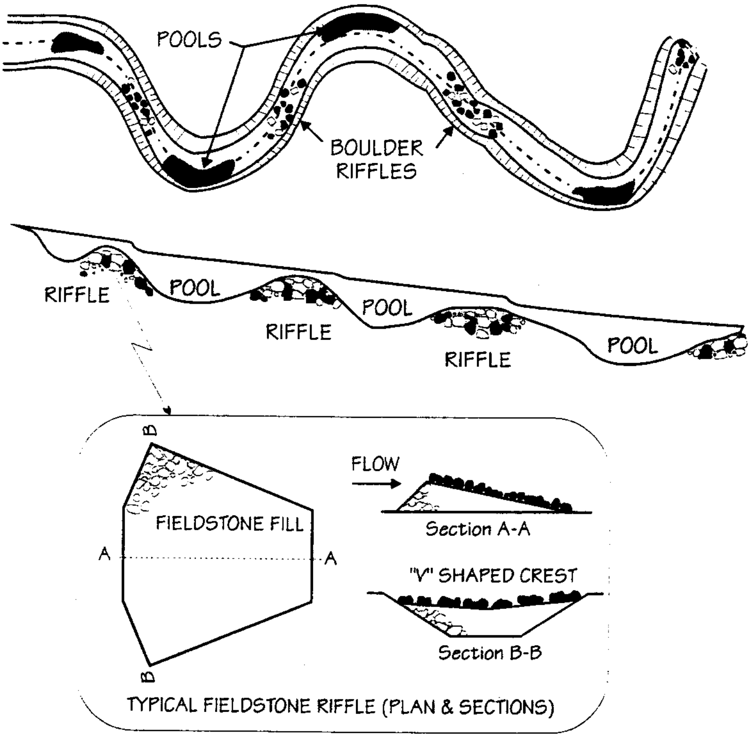


Fig. 5. Schematic of reconstructed watercourse with spawning riffles (adapted from Newbury and Gaboury 1993)

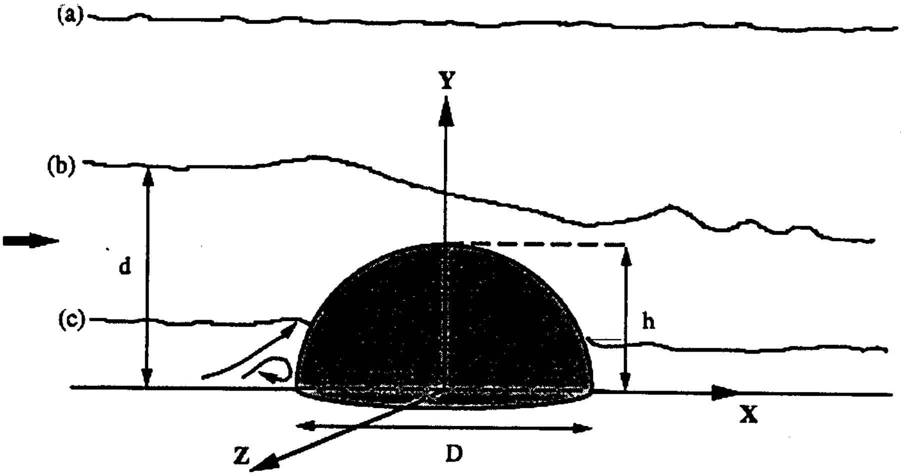


Fig. 6. Typical water surface profiles for flow around a hemisphere:
(a) deep submergence (d>>h; regime 1), (b) moderate submergence (d>h; regime 2) when d close to h; regime 3, and (c) shallow depth (d<h; regime 4).

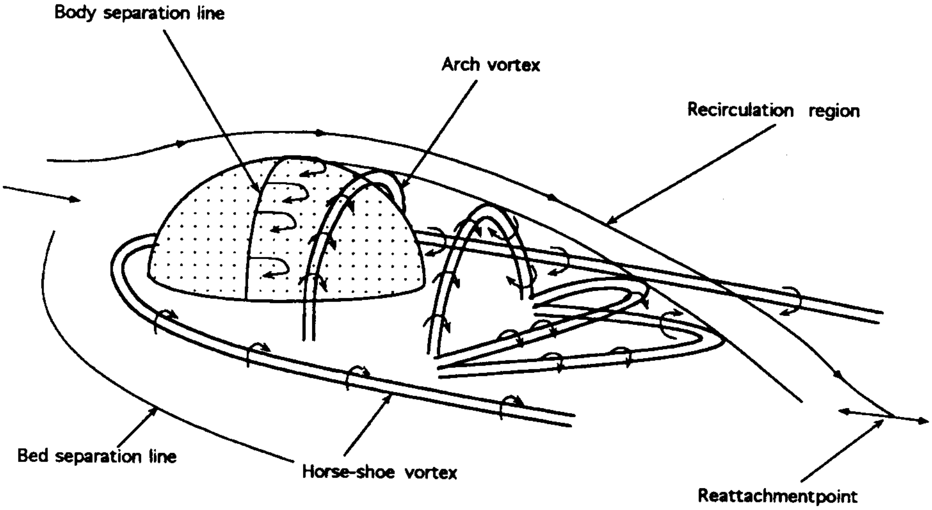


Fig. 7. Vortex system around a hemisphere (regimes 1 and 2).

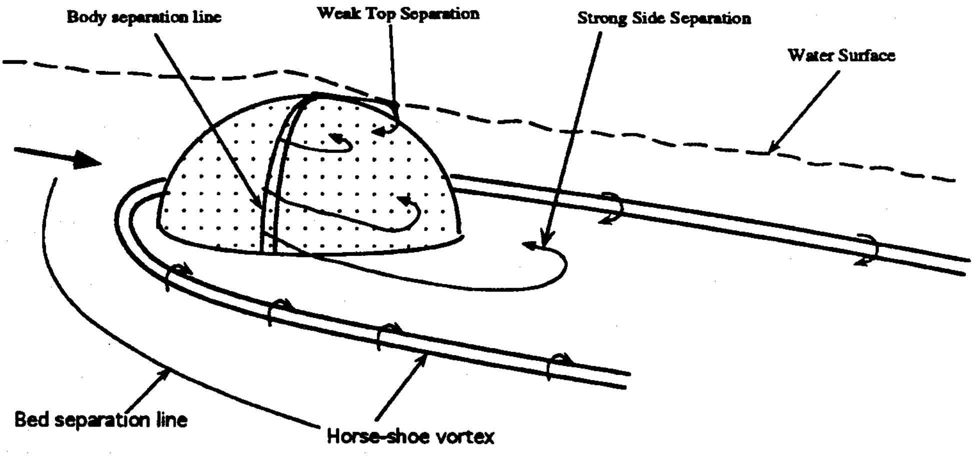


Fig. 8. Vortex system around a hemisphere (regime 3).

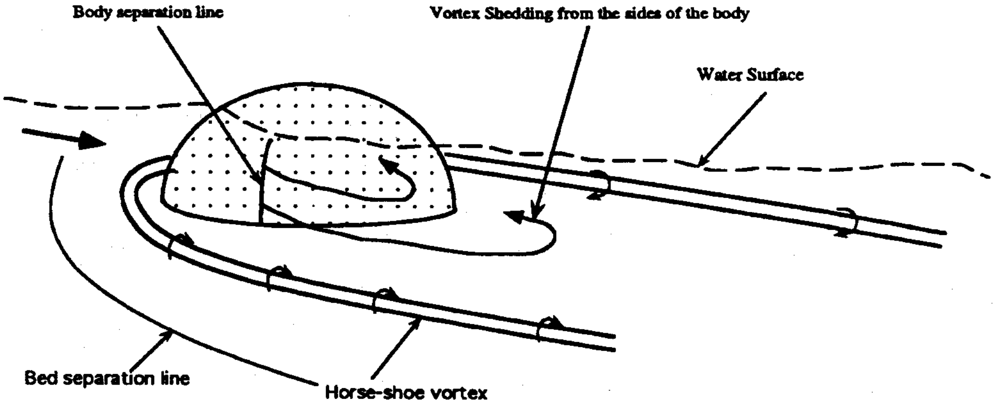


Fig. 9. Vortex system around a hemisphere (regime 4).

The laboratory experiments with objects on rigid beds were followed with similar ones on mobile beds consisting of sand. The process of scour and deposition was observed for fixed objects (hemispheres, cube, rock) and three sizes of sand. Fig. 10 illustrates the scour and deposition process for a hemisphere and a cube. For the hemisphere and in regimes 1 and 2 (higher relative depths) the scour pattern in the middle propagated downstream while the scour pattern on the sides did not. In regimes 3 and 4 (lower relative depths), the scour in the middle stopped and the scour on the sides took on the shape of a fan. The scour and deposition pattern was affected by the size of the hemisphere. As Fig. 10 shows for the cube, the shape and characteristics of the object (streamlining and point of flow separation) strongly influence the scour and deposition pattern. Experiments are continuing with multiple objects and detailed field measurements with typical habitat structures are planned for the summer of 1996.

Habitat Water

Another study relating to habitat suitability, models instream flow needs or habitat water for fish. The hydraulic techniques employed with instream flow models currently in use, including the widely used Instream Flow Incremental Methodology (IFIM), were reviewed (Ghanem et. al., in press; Ghanem et. al., 1995a; McPhail et. al., 1992). Existing models rely on hydraulic parameters and observed fish preferences to describe habitat. These models rely heavily on extensive field measurements and empirical approaches to estimate habitat quality and quantity and hydraulic characteristics and in some cases oversimplify the problem. The one-dimensional (1-D) mathematical models, presently used to simulate fish habitat (e.g. PHABSIM of the IFIM), were compared with two-dimensional (2-D) flow models. The more sophisticated 2-D models are required for projects which include river channel modifications, instream structures, or river engineering works. The performance of three 2-D finite-element models were compared to each other and to the 1-D hydraulic model incorporated in PHABSIM for an idealized case of flow over a side bar. The three two-dimensional models gave almost identical results when compared with each other; however all three differed significantly from the results obtained from the one-dimensional approach. This is not surprising since velocity distribution across the channel and through the depth is a key variable defining habitat, yet the 1-D model assumes steady uniform and gradually varied flow.

Leclerc et. al. (1995) discuss the advantages of 2-D modeling more fully and point out how many of the limitations of existing pseudo 1-D or 1-D models are overcome with this approach. Essentially 2-D models can readily be used in instream flow studies and offer spatial resolution that could define individual fish habitat units and more realistically represent spatial variability in rivers and streams. Depending on the model used, 2-D models can simulate the drying-wetting cycle from discharge variability and can represent flow resistance more accurately as a function of topographical features of the river bed. Some of these models also can deal with critical and subcritical flow changes. In general 2-D models are expected to be more reliable in calibration and validation, to reduce errors in depths and velocities, and to provide improved accuracy and resolution in predicting the effects of altering physical habitat variables. The challenge of improving habitat simulation models and fish preference curves to take full advantage of 2-D hydraulics remains.

A two-dimensional hydraulic model has been developed at the University of Alberta based on the Characteristic Dissipative Galerkin finite element technique (CDG-2D) which offers all the above mentioned advantages. This is a stable and accurate method, and test cases showed that it is suitable for simulating complex flow features such as standing waves, dambreak problems, as well as changes from subcritical to supercritical flows (Fig. 11). Existing data on high gradient rivers (Fig. 12) and field data collected in a low gradient river (Fig. 12) were used to compare the one and two-dimensional models (Ghanem et. al., in press). A physically based computational technique for the wetting/drying problem was developed by coupling the depth averaged shallow flow equations in the surface water areas with vertically averaged groundwater flow equations in the dry areas (Ghanem et. al., 1995b; Fig. 11). The CDG-2D model can serve as a tool to analyze

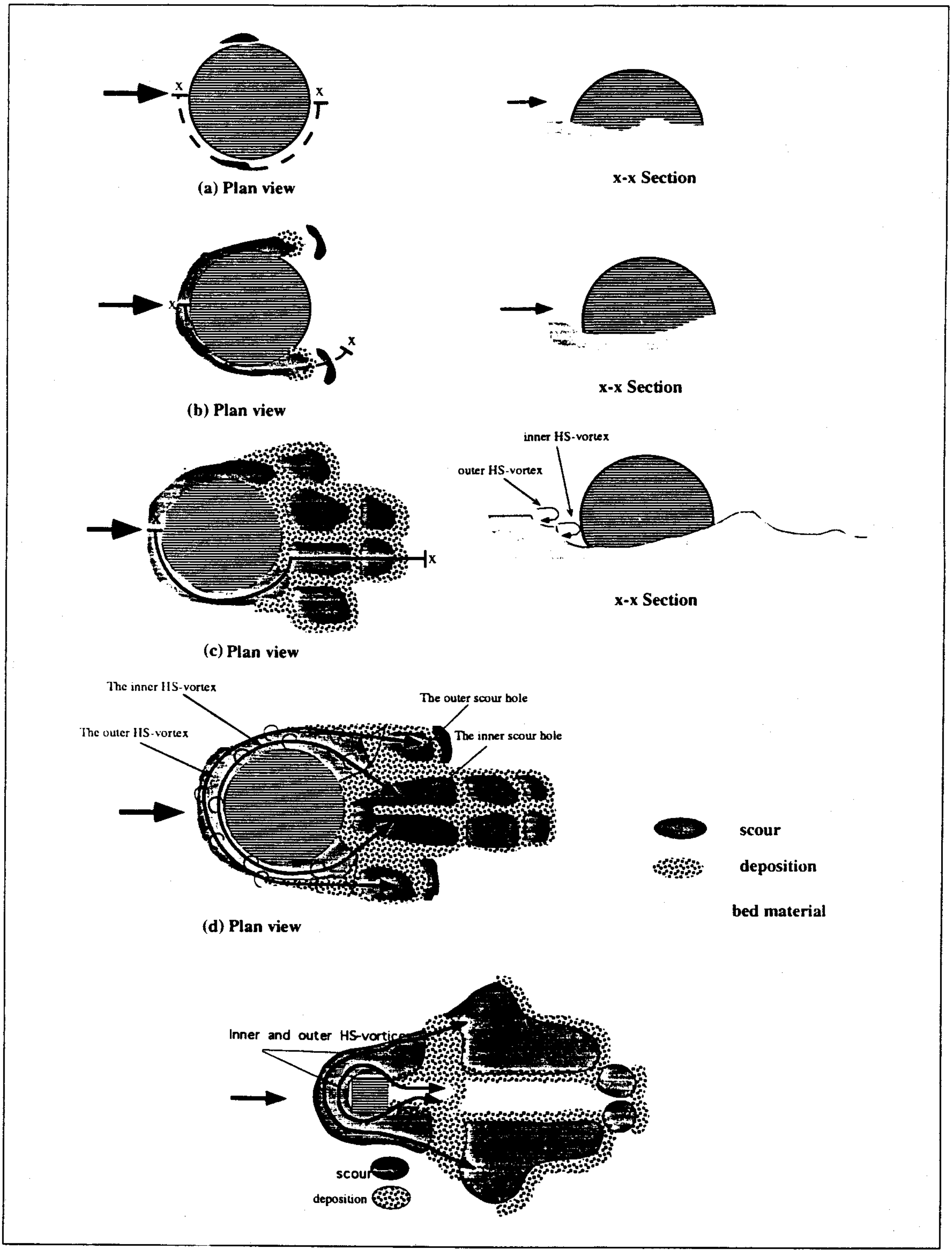


Fig. 10. Process of scour and deposition with a hemisphere (a, b, c, d) and a cube (regime 2).

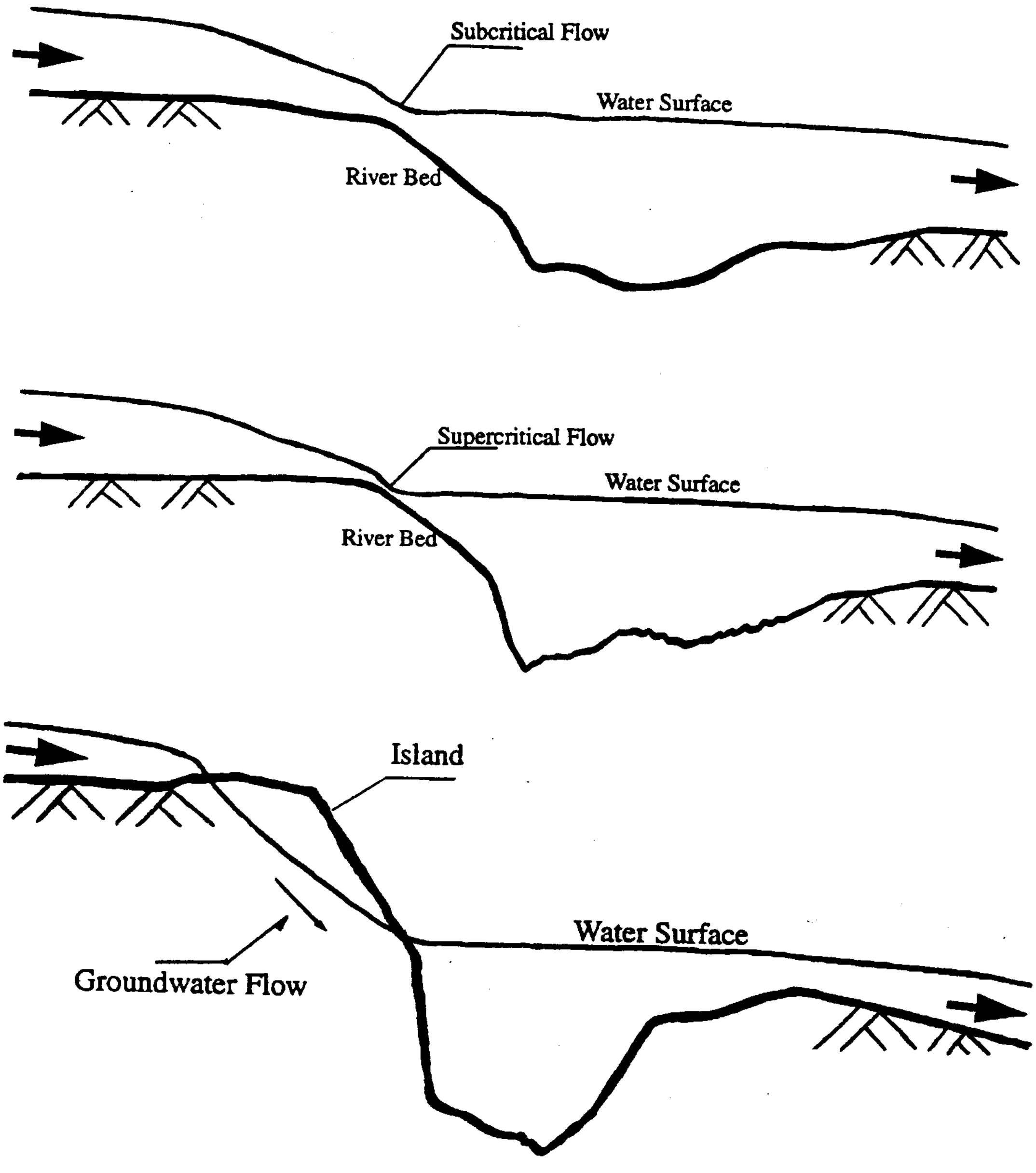


Fig. 11. The CDG-2D mathematical model simulates subcritical flow, supercritical flow, and wet-dry river bed cycles (groundwater flow).

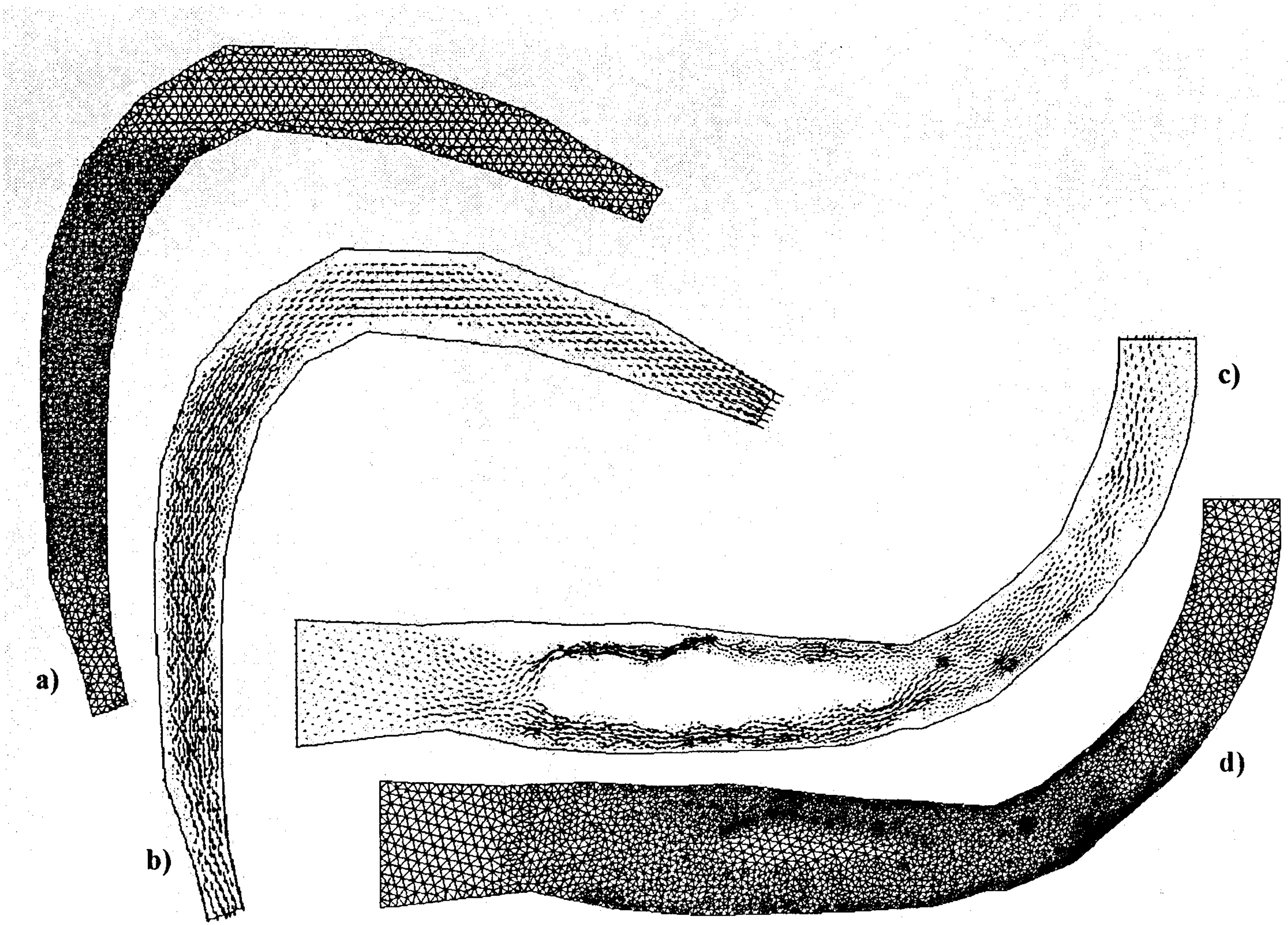


Fig. 12. Mesh generation (a & d) and velocities (b & c) estimated with the CDG-2D for sites on the Assiniboine River (a & b) and the Elbow River (c & d).

the complex flows in natural or artificial fish habitat reaches. The 2-D model gives a better representation of the flow, as it accounts for flow features resulting from bars, islands, meanders, etc. More flexibility in data collection by relaxing the 1-D rigid pattern of transects, allows for better representation of habitat features, and is another advantage of the 2-D approach.

Field studies are underway on the Assiniboine River in Manitoba (Fig. 2, location G), a low slope, prairie river, with several freshwater species, as well as gravel rivers in southern Alberta (e.g. Elbow River near Calgary; Fig. 2, location C), with trout species. These studies use the CDG-2D model to simulate flow and Fig. 12 presents typical mesh generation and velocity field results from these two river sites. Waddle et. al. (1996) compare depths and velocities obtained from simulation of the Elbow River using the IFG-4 used in IFIM and the CDG-2D model. The strength of the 2D approach is demonstrated particularly for split flow and where significant lateral mass transfers occur. Further developments with the CDG-2D model are planned, including a more user friendly interface and use of GIS for presentation data input and output and modeling river topography (Steffler et. al., in press).

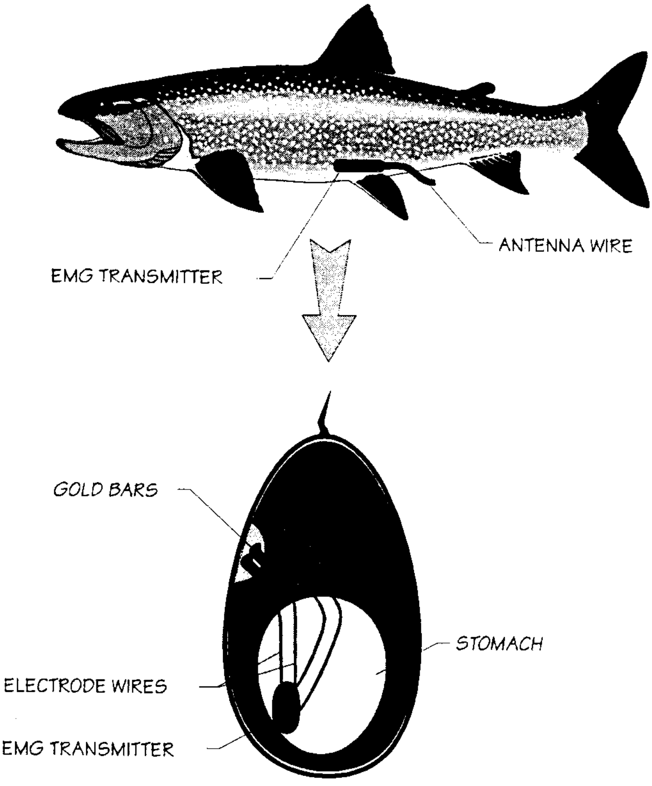


Fig. 13. Placement of an electromyographic or EMG tag for telemetry studies.

Fig. 13 illustrates placement of an electromyographic or EMG tag which can transmit fish muscle activity as well as fish location. This allows direct observation of fish habitat preferences, behaviour and responses to changes in the abiotic variables in the river. It also allows measurement of swimming performance and energetic cost when muscle activity is calibrated. Insream flow studies which incorporate radiotelemetry and EMG tags offer the potential of testing the habitat preference indices used presently and leading to improvements in simulating biotic variables and habitat preference indices. Such studies could take full advantage of the spatial resolution, and improved hydraulic simulation offered by 2-D models, and should lead to better habitat models. A field study is planned for the Kananaskis River in southwestern Alberta with a telemetry component to test this technology. EMG tags also could be used in assessing the biological effectiveness of various habitat structures, fishways, fish screens and fish exclusion systems.

HABITAT ACCESS

Channel fishways, using natural streams as analogs, modified fish guidance systems and fish screens for downstream migrants, modified culvert, pool-weir, vertical slot and Denil fishways provide a diversity of designs for various fish species (forage fish to sturgeon). Of the thousands of fishways built around the world, few are designed for non-salmonids larger than 1000 mm long. Concern over dwindling populations of non-salmonids and large migratory species, like the various sturgeons, has focused attention on passage requirements and design problems unique to these fish. Freshwater fish like Lake sturgeon *(Acipencer fulvescens)* are primary species of concern in the Canadian Provinces of Ontario, Manitoba, and Saskatchewan, as well as in the midwest States in the U.S.A. The most significant fish passage concerns in this area of North America involve existing hydroelectric stations. Although fishway design needs to consider the biology, life cycle, behaviour, swimming ability, space, and hydraulic conditions best suited for species of concern, in most cases very little is known about these aspects. In addition an ecosystem approach to fishway design, where the needs of all rather than a single fish species are considered, entails a multitude of design criteria and greater complexity.

Field studies evaluating the effectiveness of fishways are a significant component of the study program. Fishways in Alberta, Saskatchewan and Manitoba and have been assessed (Christensen 1994; Katopodis, 1992; 1990; Katopodis et. al. 1991) by trapping fish upstream after successfully passing through a fishway. Physical variables such as water levels and temperatures also were measured and water velocities were estimated. These studies have demonstrated, in some cases for the first time, that many freshwater fish, can use several fishway types. A more comprehensive study is underway on the Grand River in Ontario, assessing two fishways on the same dam for passage of smallmouth bass and other species.

Ichthyomechanics

Swimming performance studies also have been utilized to assess water velocity criteria for habitat suitability, fishways, fish screens, fish guidance and exclusion systems which are compatible with different species (Katopodis 1994; 1992; 1990). A database on swimming speeds for several species was generated from literature sources in 1989. This database was recently updated to 1995 and is presently used for a more comprehensive analysis. Swimming performance studies on Lake sturgeon, Atlantic salmon *(Salmo salar)*, brook *(Salvelinus fontinalis)* and brown *(Salmo trutta)* trout recently were completed in Newfoundland using Blazka respirometers. Similar work with walleye (*Stizostedion viteum)* and whitefish *(Coregonus clupeaformis)* is planned for the spring of 1996 in Manitoba. Studies with swimming chambers have produced performance data for several species, sizes and endurance times (Katopodis 1994; 1990). These tests primarily provide prolonged and critical speeds for these species. A fish raceway was designed to observe fish behaviour and measure burst speeds, swimming distances, endurance times and water velocities under ambient conditions and in more direct ways than provided by respirometers (Fig. 14). Fish enter the raceway of their own volition from a holding tank and their position in terms of time and distance from the flume outlet are recorded at regular intervals using coded wire tags. The raceway concept was tested in the Noel Paul Brook site in Newfoundland with adult Atlantic salmon (Colavecchia et. al. 1996). It is expected that swimming performance data using this method would apply more directly to fishway design than swimming chamber data.

Swimming performance is a key element for fish passage and exclusion systems and it varies with body morphology (swimming mode), fish length, water temperature and other variables (Katopodis 1994; 1992). A study on the swimming ability of Lake sturgeon was conducted in the laboratory using swimming chambers appropriate to fish size. Fish had minimum cross-sectional areas less than 10% of their respective swimming chamber. A total of 63 Lake sturgeon were tested, 58 between 120 to 550 mm in length, and 5 between 1060 and 1320 mm in length. Small fish (shorter than 150 mm) were tested in a 3.2 L chamber at water temperatures of 7 and 14°C, intermediate fish (230 to 550 mm) were tested in a 200 L chamber at 7, 14, and 21°C, while large fish (longer than 1000 mm) were tested at 14°C in a PVC pipe approximately 2.5 m long with a diameter of 560 mm, submerged and secured within a large flume (3m by 5m; Peake et al. 1995).

Sturgeon critical speeds were measured by gradually increasing water velocity in 5 cm/s steps every 10 minutes, until fish were fatigued. A 10 minute critical speed was considered more relevant to passage times through fishways than the more common 30 or 60 minute criticals. The 10 minute critical speed for Lake sturgeon was found to increase with fish length and water temperature, following trends similar to other species. Specific critical speeds, expressed in body lengths per second, decreased with fish length, a trend

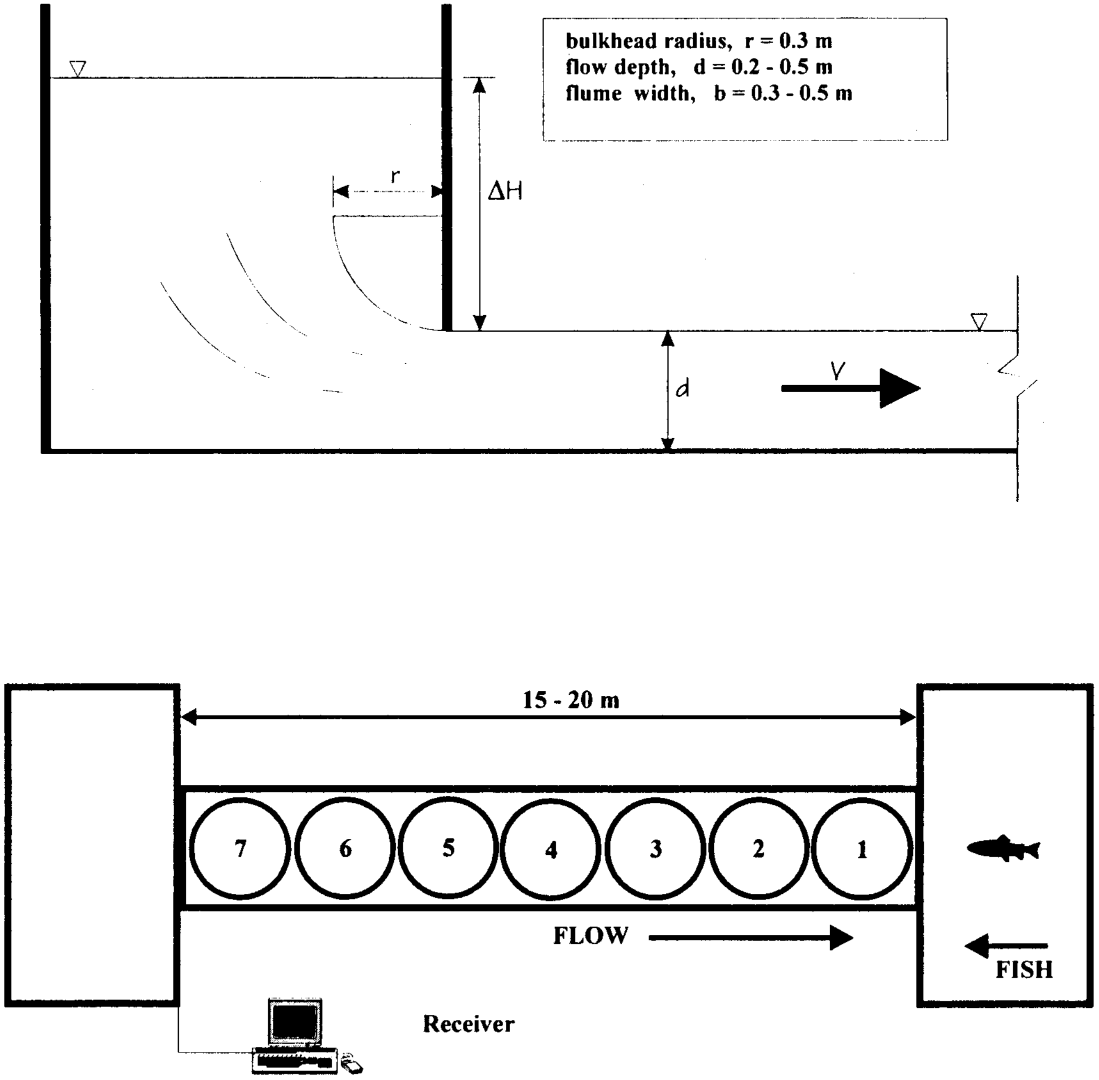


Fig. 14. Burst swimming flume with radiotelemetry equipment and ranges for the 7 antennae; fish tag implant.

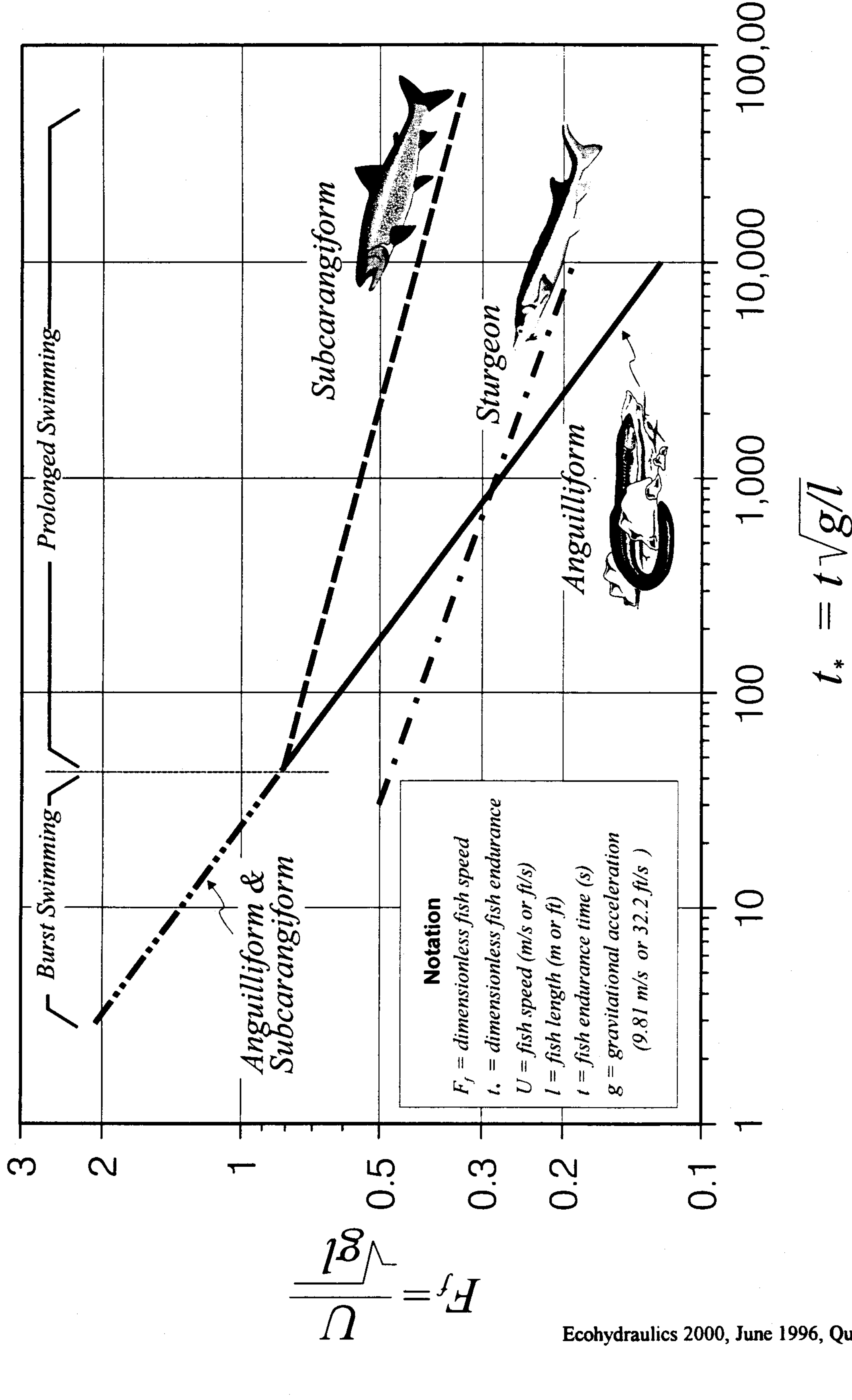
typical of other species as well. Critical speeds for Lake sturgeon however, were significantly lower than for sockeye salmon of similar size. Endurance also was measured at 0.2, 0.4, 0.45,and 0.5 m/s for small sturgeon, at velocities between 0.3 and 0.9 m/s for the intermediate sturgeon, and at 0.9, 1.2, 1.5 and 1.8 m/s for the large sturgeon. Lake sturgeon swimming endurance (time to fatigue at a given water velocity), increased with fish length. For a given fish length, endurance decreased as water velocity or specific swimming speed increased. Endurance also increased with water temperature, but this trend diminished to a negligible effect at water velocities near burst speeds. Although these trends are similar to other species, it was noted that the slope of the Lake sturgeon fatigue curve (i.e. endurance vs swimming speed) did not change in the burst range as it does for sockeye salmon. The sturgeon fatigue curve plotted lower than the corresponding curve for sockeye salmon of a similar size, indicating lower swimming performance (Peake et. al. 1995).

Fig. 15 illustrates mean endurance curves for anguilliform (i.e. eel shaped fish) and subcarangiform (i.e. trout shaped fish) swimmers (Katopodis 1994; 1992; 1990) using normalized (dimensionless) parameters. It is based on the 1989 database of swimming speeds and provides mean regression lines from a preliminary analysis. The regression line for Lake surgeon was derived from the data by Peake et. al. 1995. The Lake sturgeon mean endurance curve is almost parallel to and lower than the subcarangiform curve in the prolonged range, but does not seem to follow the steeper slope of the anguilliform and subcarangiform fish in the burst range. Fig. 15 demonstrates that Lake sturgeon performance is lower than subcarangiform fish of the same length, throughout the burst and prolonged ranges, and lower than the anguilliform fish for dimensionless endurance (t_*) of less than 1,000 or for dimensionless fish speed (F_f) of less than 0.3. However, it is important to remember that sturgeon grow to much larger lengths than other freshwater subcarangiform fish and are usually larger than freshwater anguilliform fish. Note that in Fig. 15 as fish length increases, both relative endurance and fish speed decrease. This means that longer fish are represented to the left of shorter fish in these curves. So, depending on the length of sturgeon and the length of other fish, velocity criteria for fishway design may be similar, as the absolute swimming speed of a large spawning sturgeon may be similar to the absolute swimming speed of a much smaller spawning trout. In other words, the lower performance of sturgeon may be offset by its larger size.

Hydraulic Considerations

The size of upstream migrating fish dictates space and passage opening width in fishways. Adult sturgeon require significantly larger space than other fish since they may be 3 to 4 times longer. Geometric similarity with existing fishway designs (Katopodis, 1992; 1990) needs to be maintained if fishway designs already studied hydraulically and tested in the field are to be used. Depending on site conditions, this may lead to much larger and significantly more expensive fishways. Distorting geometric similarity, to accommodate wider fish passage openings and smaller pools, may reduce fishway size. As a result supplementary hydraulic modeling studies were sponsored by the Department of Fisheries and Oceans (Government of Canada). Studies on pool-weir, vertical slot, Denil, and culvert fishways, as well as resting pools were completed at the T. Blench Hydraulics Laboratory of the University of Alberta. A study to provide fish passage using rock riffles was sponsored by Manitoba Hydro and was completed at the hydraulics laboratory of the University of Manitoba. These studies provide discharge, depth and velocity relationships and assist in designing suitable passage facilities to provide fish with access to habitat interupted by dams, weirs, highway crossings or habitat structures (Katopodis et. al., 1995; Katopodis, 1992; 1990; others pending publication). In general, fishway dimensions and design details affect circulation, flow patterns and velocity distributions considerably, but depth-disharge relationships in a limited way.

Plain Denil fishways have a low velocity layer near the bottom and a high velocity stream near the surface. Water depth and baffle spacing have a large effect on the low velocity layer, but a small effect on the surface stream. Increasing the baffle spacing beyond 3 times the standard spacing weakens the equivalent uniform flow analysis and suggests a greater influence from the fishway slope. Katopodis et al. 1995 present analysis and generalization of these results. Steeppass Denil fishways display higher velocities near the bottom and



F_f = dimensionless fish speed
t_* = dimensionless fish endurance
U = fish speed (m/s or ft/s)
l = fish length (m or ft)
t = fish endurance time (s)
g = gravitational acceleration (9.81 m/s or 32.2 ft/s)

Fig. 15. Normalized endurance (horizontal axis) and speed (vertical axis) for groups of fish from preliminary analysis.

lower velocities near the surface when water depths are relatively low. With deep flows, maximum velocities are at mid-depth, with symmetrically lower velocities above and below (Rajaratnam and Katopodis, 1991). The complex nature of flow in resting pools between Denil fishway segments also was investigated (Rajaratnam et. al., in press).

Laboratory studies on circulation patterns and flow characteristics in vertical slot fishway pools were initiated. Analysis of the path and the diffusion of the slot jet as it travels through the pool and the circulation patterns created are essentially three dimensional complex flow phenomena. Systematic hydraulic modeling of weir fishways continues, with emphasis on fishway slope, weir spacing and height. Plunging, streaming, and the in between unstable transitional regime are investigated. The hydraulics of several culvert fishways were studied (Katopodis, 1992; 1990) and a study to map velocity contours in plain round corrugated culverts is underway. The channel fishway study using rock riffles indicates that velocities could be controlled for passage of several fish species, including sturgeon. One may also consider providing fish passage through a channel which contains habitat features suitable not only for migration but for spawning or other life cycle requirements. An approach to designing such channels is provided by mimicking natural streams which support Lake sturgeon and other species (Katopodis 1995).

In many cases channels are constructed around low head dams which may provide fish with a passage path as well as habitat for spawning, rearing, or holding. These channels need to meet both habitat suitability and fish passage criteria, and in most cases inadequate design information exists. An example is the proposed diversion channel in the NWT Diamonds Project. BHP Diamond Inc. is proposing to develop Canada's first diamond mine, in the headwaters of the Coppermine River watershed (Fig. 2, location A). The project is located near Lac des Gras, a remote area in the permafrost zone, with very little hydrologic, hydraulic, geomorphologic or biologic data. A component of the open pit mining operation would include the dewatering of six small lakes and streams connecting them, and the use of one lake for mine tailings disposal. The connecting streams typically are ephemeral with snow melt water in the spring and intermittent flow during late summer and fall. These streams are used by spring migrating Arctic grayling and Longnose sucker for spawning and rearing (Rescan Environmental Services, 1995). Prior to dewatering in the Koala Lake watershed, the company proposes to by-pass the flow from Panda Lake to Kodiak Lake through a diversion. The company's intention is to build this diversion channel so it conveys the water flow, provides a migration route for fish, and creates suitable habitat to replace the dewatered lotic habitat. Under the Arctic conditions described, and with the limited data and experience available at this site, mimicking of the natural streams in the area, flowing through shallow boulder gardens and with poorly defined channels, may be the best guide in attempting to provide for fish passage and create fish habitat within the diversion channel. A monitoring program to evaluate the effectiveness of this approach and possibly suggest subsequent modifications to the habitat channel is recommended.

Modifications to Existing Facilities

Many challenges await the designer in modifying existing fish passage or fish exclusion facilities, particularly in cases where concrete and metal materials have been used. The two examples that follow illustrate some of these challenges. At Hell's Gate Canyon on the Fraser River (Fig. 2, location B), several vertical slot fishways were built in the 1940's to relieve the obstruction partly created during railway construction. From 1944 to 1947, two fishways were built on the right bank (looking downstream). The main low level fishway has an invert at gauge 4.6 m (15 feet), while the auxiliary high level fishway has a downstream sill invert at gauge 17.4 m (57 feet). A 3.7 m wide by 12.8 m high (12 by 42 feet), and 40 m long tunnel, with the same invert as the main fishway (at gauge 4.6 m or 15 feet), also was constructed. The tunnel was intended to provide a path for the fish to bypass the highly turbulent river flow along the right bank and reach the main fishway. River hydraulics are such that at flows below about 5,000 m^3/s, the flow through the tunnel reverses allowing suspended bedload in the range of 2.5 - 40 mm (D_{10}=10 mm, D_{90}=25 mm) to deposit in the tunnel (B.C. Hydro 1994). Gravel accumulation of more than 4 m deep is not uncommon, effectively raising the tunnel invert by the same amount and reducing its usefulness for fish passage. Observations indicate that Pacific salmon consistently cannot use the blocked tunnel during their migration between August and October. Gravel removal is only possible from November to April. Deposition invariably occurs during the May to July freshets. Physical and mathematical modelling could point to feasible solutions, but such studies are hindered primarily by poor data on river topography and extreme difficulty and expense in obtaining such data.

The Carseland Weir on the Bow River near Calgary (Fig. 2, location C) was completed in 1973 to divert water into the canal of the Bow River Irrigation District. A vertical slot fishway was incorporated, opposite the canal intake, but a fish exclusion system was not built at the water intake. The fishway was designed to operate effectively at river flows between 142 and 566 m^3/s, with optimal operation at 283 m^3/s. Post-construction assessment of the fishway, indicated that: a) in most years, Bow River flow during Rainbow trout, Brown trout and sometimes Mountain whitefish upstream migration is below 142 m^3/s; b) silt and aquatic vegetation tend to clog the fishway exit. Recently, increased interest in overcoming these problems and addressing concerns about fish losses at the irrigation canal intake, have led to a review of the biological and engineering issues involved and study of feasible modifications is underway.

Field Studies

The Regional Municipality of Waterloo , Ontario, in 1989 completed the construction of a weir across the Grand River, a water intake, and pumping station at Freeport (Fig. 2, location D). These structures are part of the Mannheim water supply project, which is intended to meet the rapid growth in water demand in the Kitchener-Waterloo area. Upstream fish passage was required, and two Denil type fishways were installed, one on each side of the weir (Fig. 16). The West Bank Fishway was constructed with less slope (10%) than the East Bank Fishway (20%). The West Bank Fishway is longer and incorporates two resting pools, while the East Bank Fishway is shorter and has no resting pools. Both fishways have the same width and baffle details. From these design features, it is expected that fish using the West Bank Fishway will face lower water velocities, travel a longer distance, with opportunities to rest, while fish using the East Bank Fishway will face higher velocities over a shorter distance with little opportunity to rest before reaching the exit pool. This project provides a unique opportunity to compare the performance of these two different Denil designs.

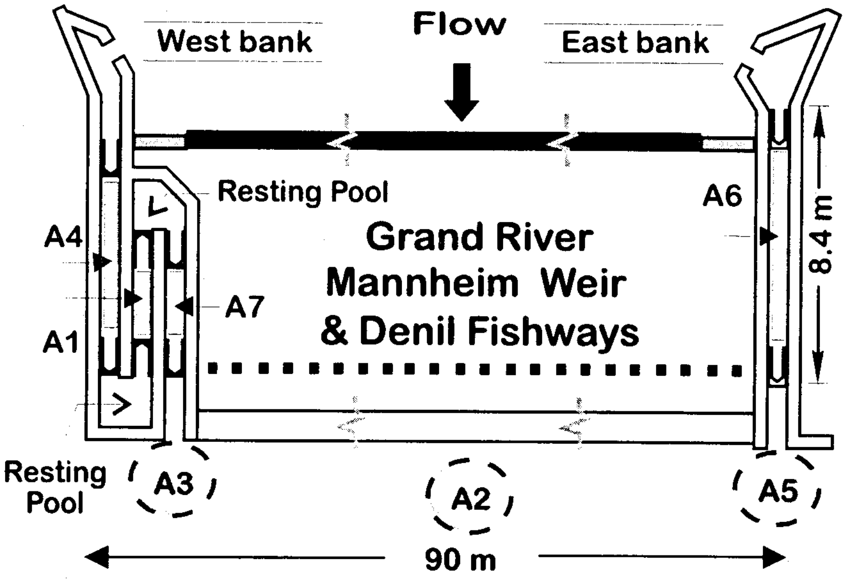


Fig. 16. Schematic of fishways at Mannheim showing telemetry antennae A1 to A7.

Comparisons of the operating performance of the two fishways over five migratory seasons (1990-94) indicated that 2107 fish of 15 fish species used both fishways, with 1719 fish (81.5%) passing through the West Bank Fishway, and 388 (18.5%) passing through the East Bank Fishway (Aqua Terra Fish and Wildlife Management 1994). Fish sampled at the West Bank Fishway ranged in size from a Pumpkinseed with a fork length of 96 mm to a Carp with a fork length of 645 mm, while fish sampled at the East Bank Fishway ranged in size from a White sucker with a fork length of 115 mm to a Carp with a fork length of 551 mm. Most of the species caught in high numbers (e.g. Rock bass, Common shiners, White suckers) showed a clear preference for the West Bank Fishway. Smallmouth bass used both fishways in approximately equal but low numbers, while samples for other species were small.

The above studies were limited in scope and could not determine if the difference in fish passage numbers was due to the difference in hydraulic conditions between the two fishways or reflected a greater abundance of fish along the west bank than the east bank. Fish were sampled during daylight hours and over a few weeks in the spring each year. These studies also could not determine how fish used the fishways (e.g. passage time, location within a fishway or explain differences in fishway use). A study was initiated in 1994 to address these questions in a more comprehensive way by radiotelemetry. Telemetry and hydraulics are combined to track fish movements and provide data on fish passage effectiveness. Fish with coded pulse radio transmitters combined with antennae in the river downstream of the dam and inside the fishways and a radio receiver/data logger, are used to track several fish simultaneously (Fig. 16). Water levels and temperatures upstream and downstream of the Mannheim Weir and within each fishway and resting pool also are recorded. Radio-tagged and floy-tagged fish are collected on a 24 hour basis at the upstream trap of each fishway. This study may provide insights as to behaviour and preferences of several species and assist in refining fishway designs. These sampling techniques have the potential can be used at many fishway sites to assist with entrance location, and more refined fishway design.

CONCLUSIONS

Interdisciplinary research is key to providing appropriate approaches, an ecosystem perspective, and colaborative efforts, to arrive at suitable riverine habitat projects that benefit fish resources. A high level of harmony with nature, sustainable development and environmentally acceptable river engineering works depend on full integration of all relevant engineering and biological disciplines. Significant research efforts are needed to provide more reliable methods for conserving, restoring and enhancing fish habitat. Several professional societies, particularly those representing engineers and biologists, as well as individual professionals need to achieve a greater degree of collaboration in this field. In conducting such studies international collaboration in these areas of work should be encouraged.

Although renewed interest in fishways has provided opportunities for research and development, and several strides have been made in the last couple of decades, many challenges remain, particularly for further work with multispecies facilities. It is important to fully utilize the best information that exists worldwide. Further studies are needed to provide better guidance in fishway entrance location, attempt to quantify fish behaviour and swimmimg ability, as well as develop better fish exclusion systems. Enough is now known to direct the studies needed to develop fishways which accommodate several species including sturgeon. Fish passage or exclusion facilities combined with habitat restoration works hold considerable promise for meeting resource sustainability needs.

ACKNOWLEDGEMENTS

The author would like to express sincere thanks and deep appreciation for the collaboration and contributions to this work of Dr. N. Rajaratnam of the University of Alberta for physical hydraulic modeling studies, Dr. P. Steffler of the University of Alberta for mathematical hydraulic modeling studies, and Dr. R. S. McKinley of the University of Waterloo for radiotelemetry and swimming performance studies. Special thanks to research assistant Dr. S. Wu, University of Alberta graduate students H. Shamloo, C. Albers. G. Sikora, and B. Bai, S. Lovell for building the scale models, and University of Waterloo graduate students C. Bunt, S. Peake, M. Covalecchia, and R. Booth. R. Gervais assisted with data analysis and computer graphics, and Drs. W. Franzin and J. Mathias of the Freshwater Institute reviewed the manuscript. Funding for the studies mentioned was provided by the Department of Fisheries and Oceans. Supplementary funding for the fishway evaluation at Mannheim was supplied by the California Department of Water Resources.

REFERENCES

Aqua Terra Fish and Wildlife Management. 1994. A summary of the Grand River Mannheim Weir Fishway Operations (1990-1994). Produced for the Regional Municipality of Waterloo, Ontario, Canada: 36 p.+5 Appendices.

B.C. Hydro International Limited. 1994. Hell's Gate Hydraulic Study, Fraser River. Report No. BCHIL.029, B.C. Hydro, Vancouver, B.C..

Colavecchia, M.V., C. Katopodis, R.S. McKinley, R. Goosney, D.A. Scrutton. 1996. Measurement of burst swimmimg performance in wild Atlantic salmon *(Salmo salar)* using coded radio transmitted signals. Proc. Ecohydraulics 2000, 2nd IAHR International Symposium on Hydraulics and Habitats.

Christensen, B. 1994. Passage of north temperate fish through the Cowan Dam Denil fishway. M.Sc. Thesis, Department of Zoology, University of Manitoba, Winnipeg, Manitoba, Canada: x+137 p.

Department of Fisheries and Oceans. 1995. Freshwater intake end-of-pipe fish screen guideline. Communications Directorate, Ottawa, Ontario, Canada: iii+27p.

Dwivedi, V.K., N. Rajaratnam, and C. Katopodis. 1993. An exploratory study of fish habitat structures in rivers. Proc. 1993 Annual Conference of the Canadian Society for Civil Engineering and 11th Hydrotechnical Conference, Fredericton, New Brunswick, Canada:p. 573-581.

Frissell, C.A. and R.K. Nawa. 1992. Incidence and causes of physical failure of artificial habitat structures in streams of western Oregon and Washington. North American Journal of Fisheries Management 12: 182-197.

Ghanem, A., P. Steffler, F. Hicks and C. Katopodis. (in press). Two-dimensional hydraulic simulation of physical fish habitat conditions in flowing streams. Regulated Rivers Research & Management.

Ghanem, A., P. Steffler, F. Hicks and C. Katopodis. 1995a. Two-dimensional finite element modeling of flow in aquatic habitats. Water Resources Egineering Report No. 95-S1, Department of Civil Engineering, University of Alberta, Edmonton, Alberta, Canada, 122 p.

Ghanem, A., P. Steffler, F. Hicks and C. Katopodis. 1995b. Dry area treatment for two-dimensional finite element shallow flow modeling. Proceedings, Annual Conference of the Canadian Society for Civil Engineering, June 1-3, 1995, Ottawa, Ontario, Canada, addendum: 10 p.

Gore, J.A. and G.E. Petts (ed.). 1989. Alternatives in regulated river management. CRC Press, Inc., Boca Raton Florida, USA. 344 p.

Katopodis, C. 1995. Recent fishway design problems in Canadian Rivers. Invited lecture, Proceedings of International Symposium on Fishways in Gifu '95, Japan.

Katopodis, C., N. Rajaratnam, S. Wu, and D.G.M. Tovell. 1995. A study of Denil fishways of varying geometry. Technical Report No. 95-R1, Department of Civil Engineering, University of Alberta, Edmonton, Alberta, Canada, 100 p.

Katopodis, C. 1994. Analysis of ichthyomechanical data for fish passage or exclusion system design. Proceedings of an International Fish Physiology Symposium, American Fisheries Society and Fish Physiology Association, Vancouver, B.C., Canada, July 16-21, p. 318-323.

Katopodis, C. 1992. Introduction to fishway design. Working Document, Freshwater Institute, Winnipeg, Manitoba, Canada, 67 p.

Katopodis, C., A.J. Derksen and B.L. Christensen. 1991. Assessment of two Denil fishways for passage of freshwater species. American Fisheries Society Symposium 10: 306-324.

Katopodis, C. 1990. Advancing the art of engineering fishways for upstream migrants. Invited lecture, Proceedings of the International Symposium on Fishways '90 in Gifu, Japan, p.19-28.

KGS Group and North/South Consultants Inc. 1992. Protection and restoration of fish habitat. Department of Fisheries and Oceans, Central and Arctic Region, Winnipeg, Manitoba, Canada: x+297p.+two appendices.

Leclerc, M., A. Boudreault, J.A. Bechara and G. Corfa. 1995. Two-dimensional hydrodynamic modeling: a neglected tool in the instream flow incremental methodology. Trans. American Fisheries Society, 124:645-662

Marsden, B.M. 1995. Danish watercourses: ten years with the new Watercourse Act. Ministry of Environment and Energy, Danish Environmental Protection Agency, Denmark. 208 p.

McPhail, G.D., D.B. MacMillan and C. Katopodis. 1992. Fish habitat mitigation measures for hydrotechnical projects. Canadian Society for Civil Engineering, Proc. Annual Conference, May 27-29, Quebec City, Quebec, Canada, Vol. II: 1-10.

Meehan, W.R. (ed.). 1991. Influences of forest and rangeland Management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19, Bethesda, Maryland, USA. 751 p.

Mitsch, W. J. and S.E. Jorgensen. 1989. Ecological engineering: an introduction to ecotechnology. John Wiley & Sons, Toronto 472 p.

M. Miles & Associates Ltd. 1995. Evaluation of fish habitat improvement projects in B.C. and recommendations on the development of guidelines for future work. Prepared for the Fisheries Branch, B.C. Ministry of Environment, Lands and Parks, Victoria, B.C, Canada. x+40 p.+11 app.

Newbury, R.W. and M.N. Gaboury. 1993. Stream analysis and fish habitat design - A field manual. Newbury Hydraulics Ltd., Box 1173, Gibsons, B.C., Canada. vi+256 p.

Office of Technology Assessment. 1995. Fish passage technologies: protection at hydropower facilities. Congress of the United States, OTA-ENV-641, U.S. Government Printing Office, Washington, DC: x+167 p.

Peake, S., F.W.H. Beamish, R.S. McKinley, C. Katopodis, and D.A. Scruton. 1995. Swimming Performance of Lake Sturgeon, *Acipenser fulvescens*. Canadian Technical Report of Fisheries and Aquatic Sciences No. 2063, iv + 26 pp.

Pearce R.O. and R.L. Lee. 1991. Some design considerations for approach velocities at juvenile salmonid screening facilities. American Fisheries Society Symposium 10: 237-248.

Power, G., R. Cunjak, J. Flannagan and C. Katopodis. 1993. Biological effects of river ice. In Prowse and Gridley, Environmental Aspects of River Ice, NHRI Science Report No. 5, Sakatoon, Saskatchewan, Canada p. 97-119.

Rajaratnam, N., C. Katopodis and M.A. Sabur. (in press). Hydraulics of resting pools for Denil fishways. Journal of Hydraulic Engineering.

Rajaratnam, N. and C. Katopodis. 1991. Hydraulics of steeppass fishways. Canadian Journal of Civil Engineering, 18:1024-1032.

Rescan Environmental Services Ltd. and AGRA Earth & Environmental. 1995. Fisheries Mitigation Strategy for "No Net Loss Policy" April 1995, Prepared for BHP Diamonds Inc. Yellowknife, NWT, Canada.

R.L.& L. Environmental Services Ltd., M. Miles and Associates Ltd., Alberta Research Council. 1994. Analyses of the performance of stream habitat structures in southwest Alberta - Data Report. Prepared for the Department of Fisheries and Oceans, Sustainable Fisheries Program in association with Alberta Environmental Protection, Fish and Wildlife Services, Winnipeg, Manitoba, Canada: 108 p. + App.

Shamloo, H., N. Rajaratnam, P.M. Steffler, and C. Katopodis. 1995. Wakes of bodies in shallow open channel flows. Proc. 1995 Annual Conference of the Canadian Society for Civil Engineering and 12th Hydrotechnical Conference, Ottawa, Ontario, Canada, June 1-3, Vol I:409 - 418.

Shrubsole, D (ed.). 1994. "Natural" channel design: perspectives and practice. Proceedings of the first International Conference on Guidelines for "Natural" Channel Systems, Niagara Falls, Ontario, Canada, Mar. 1994. 465 p.

Steffler, P.M., B. Bai and C. Katopodis. (in press). Application of GIS in modelling river characteristics. Proceedings, Annual Conference of the Canadian Society for Civil Engineering, May 29 - June 1, 1996, Edmonton, Alberta, Canada.

Trillium Engineering and Hydrographics Inc. 1996. Stability of fish habitat structures in the Castle River. Prepared for the Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, Manitoba, Canada: v + 30 p. + appendix.

Trillium Engineering and Hydrographics Inc. 1995. Fish habitat enhancement structures inventory: an inventory of available information in Saskatchewan, Manitoba, and Northern Ontario. Prepared for the Department of Fisheries and Oceans, Freshwater Institute, Winnipeg, Manitoba, Canada: vi+30p. + appendices.

Waddle,T., P. Steffler, A. Ghanem, C. Katopodis and A. Locke. 1996. Comparison of one and two-dimensional hydrodynamic models for a small habitat stream. Proc. Ecohydraulics 2000, 2nd IAHR International Symposium on Hydraulics and Habitats.

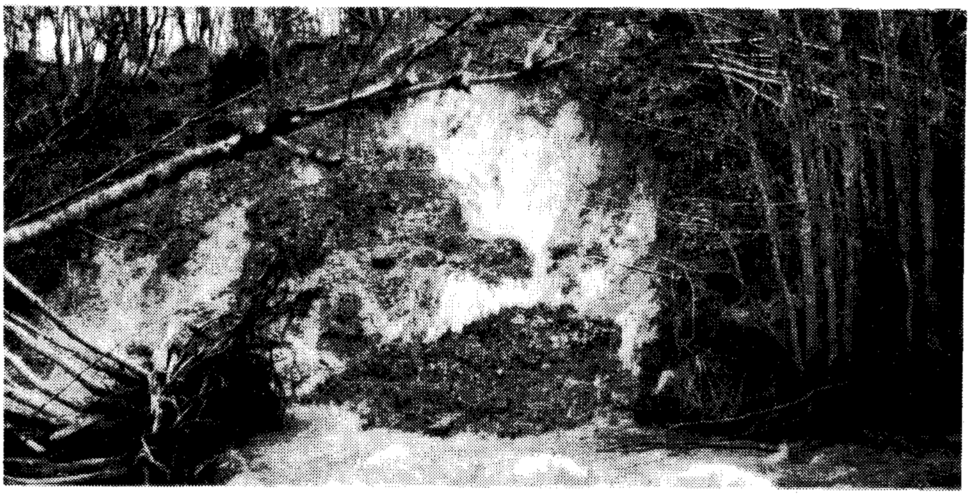
GRÅELVA, A RIVER IN NORWAY THAT HAS BEEN "RAISED" AND RECONSTRUCTED ON A NEW ROCK BED

Authors:
Mads Johnsen, Senior Engineer
and
Tore Olav Sandnæs, Environment and Landscape Adviser

Norwegian Water Resources and Energy Administration
NVE, Region Midt-Norge, V. Rosten 81, N-7505 Tiller, Norway
E-mail: tos@vrm.nve.no

ABSTRACT

The Gråelva is a tributary to the Stjørdal River in central Norway. Large parts of the watercourse are below the elevation of the marine deposition zone, and characterized by massive deposits of clay. The combination of quick clay and riverine erosion has engendered a growing danger of landslides. Following preliminary studies, the Norwegian Water Resources and Energy Administration (NVE) planned and implemented extensive measures to prevent landslides and erosion. Altogether, the plans address a 20-km stretch of river. In several places, the entire river bed will be raised as much as two metres for relatively lengthy stretches. The measures will reduce the risk of losing human life, farms, houses, power lines, roads, etc. The ravine has flourishing vegetation and, all things considered, the river has an abundance of fish. Almost since the outset, the goal has been to reconstruct the Gråelva and its tributaries and to fashion new river beds that resemble the old ones as closely as possible. Importance was attached to creating good conditions for fish and terrestrial species whose existence largely depends on the river. Roughly three km of the most dangerous part of the river has now been secured and reconstructed. Vegetation is beginning to take root and the fish and benthos appear to be doing well in the reconstructed river bed. The load of sediment carried by the river has been reduced significantly. The Norwegian word "Gråelva" means "grey river" in English. Time will tell whether or not the river will eventually need a new name. A number of studies and surveys of a geotechnical, sedimentological and biological nature have been conducted. Some of them will be followed up once the valley has been completely secured. Only then we will know whether our efforts have been successful in terms of the environmental characteristics of the watercourse.



Picture caption: A typical situation in the Gråelva Valley.

INTRODUCTION

The Gråelva is a tributary of the Stjørdal River located in central Norway. In connection with a systematic survey of potential landslide sites in Norway, it was discovered that large areas along the Gråelva consist of hazardous quick clay. This particular type of clay has a rather special structure due to its submersion under the sea during the last Glacial Age. After the land rose, the salts gradually washed out of the clay, leaving it with an unstable structure. Should the quick clay be disturbed, enormous areas could be set off, causing a landslide in just minutes. Norway has experienced several such explosive landslides, accompanied by extensive collateral damage. The worst took place in Verdal, central Norway in 1893, killing 112 people. The potential hazards along the Gråelva are exacerbated by the fact that the river is digging deeper into the landscape. Human lives, buildings and infrastructure are at risk.

The Norwegian Water Resources and Energy Administration (NVE) bears primary responsibility for watercourse management, and receives allocations from the State for planning and implementing safety measures related to Norwegian watercourses. It was therefore natural for NVE to head this safety enhancement project. The project is unique by virtue of its scale, the scope of the proposed measures and the attention being paid to nature and the environment. In addition, great importance has been attached to studying a number of biological conditions prior to initiating the planned re-building project. It is anticipated that the scheduled post-project studies will indicate whether or not we have performed our jobs in an acceptable manner. Although construction and landscaping are well underway, they are not yet complete.

DESCRIPTION OF THE AREA

The Gråelva has a 49 km^2 catchment area called Skjølstadmarka. Roughly 12 km^2 of that area is partly made up of dangerous quick clays. The area has a population of approximately 300 people, and is the site of 40 farms, 25 other houses, roads, major power lines, etc. The area has seen several landslides earlier and several lives have been lost, most recently in 1965. The total length of the Gråelva is approximately 12 km. The marine deposition zone is 180 metres above sea level here. Mean water flow is approximately 0,5 - 2 m^3/s. Mean flood is approximately 10 - 15 m^3/s. The maximum observed flood in January 1992 was almost 40 m^3/s.

Erosion has created deep ravines in the terrain, with slopes of up to 50 m. In the course of time, small landslides have widened the valley floor, and the river meanders through small alluvial deposits. The unstable valley walls are covered by forest, mainly grey alder (Alnus incana). More stable areas are dominated by spruce (Picea abies). The areas near the ravine consist of some forest, but are mainly cultivated fields (Fremstad, 1992).

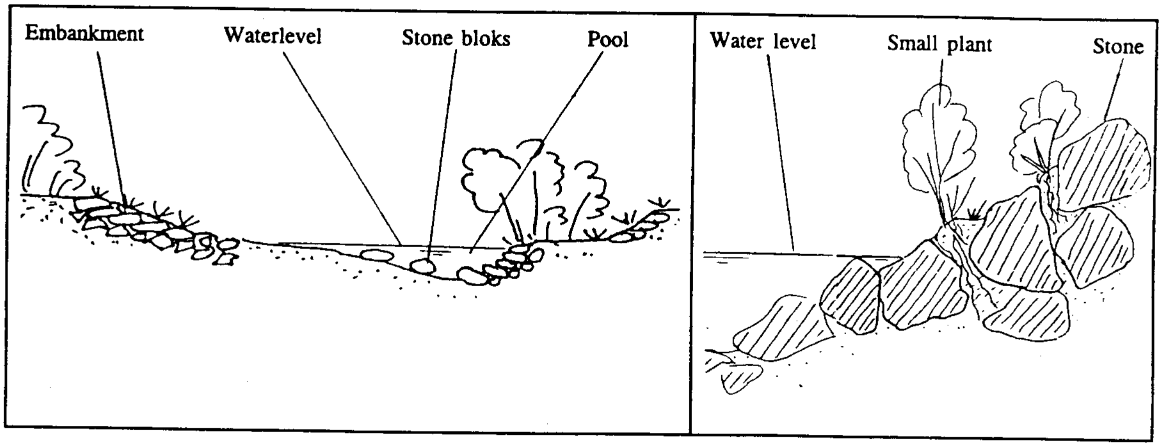
The rich and dense vegetation is home to a wide variety of animal species. Among the members of the deer family, the roe-deer (Capreolus capreolus) and the moose (Alces alces) dominate. In direct association with the watercourse, there is the otter (Lutra lutra L.). Five species of fish have been registered as living in the river, which is dominated by the salmon (Salmo salar) and the anadromous brown trout (Salmo trutta L). The other species are the European flounder (Platichthys flesus, L), the threespine stickleback (Gasterosteus Aculeatus) and the eel (Anguilla anguilla). The dominant benthic species are May flies (Ephemeroptera) and Chironomidae (Berger et al., 1994).

PLANNING

Planning began with an extensive geotechnical survey of the ground in 1990. A general solution for reducing the danger of landslides was presented in a **preliminary project** published by NVE in 1991. It was proposed that the riverbed be raised by about two metres in the most dangerous areas in order to provide support and counterbalance the quick clay. It was also important to protect the area from further erosion by channelling the river water onto a stable river bed built of crushed rock. The plan encompasses roughly 20 km of river, including about 7 km along the main river and the rest along tributaries. The cost of securing the area was estimated at NOK 31 million. The urgency of the situation made it necessary to allocate funding for the first stage, approximately NOK 10 million, rather quickly, so construction could begin already in 1992. 75 per cent of the safety measures were financed by NVE, while the municipality of Stjørdal covered the remaining 25 per cent.

Owing to budgetary constraints, it will take many years to secure the area completely. During these years, erosion and minor landslides will continue to alter small details in the landscape. Consequently, it is important to finalize the finer details of the plans as close to the date of implementation as possible. Thus far, the detailed plans have comprised two parts, each describing measures to be introduced in a given stretch of the watercourse. In addition, it is essential that planners be accessible during implementation. The detailed plans describe the general principles that apply to the project as well as specific notes about surveyed sections.

The principles mainly involve ensuring the work is done in a way that allows the river to retain its original shape. This implies reconstruction of the meanders, the distance across, holes and stretches of rapids (water depths and water flows) in the new river bed. The aim is to ensure that fishing is at least as good as it was prior to reconstruction. Further, the detailed plans attach importance to saving as much as possible of the rich vegetation growing in the river valley and to paving the way for the advent of indigenous vegetation in the areas directly affected by the safety measures. The siting of service roads and gravel tips is also given a central position in these plans. With a view to improving aquatic biotopes, plans have been made to deploy groups of stone blocks, and to build weirs and deep holes. In some rapids, gravel from another river will be used to serve as a breeding substrate for salmon and brown trout. Riparian vegetation will be planted close to the water level. That is of major importanse for biological production in the river. To protect the new vegetation against flood damages, the small plants were pinched between stones or stones were placed over the rootsystem.



The figure to the left shows how a deep pool is shaped.
The example to the right shows the establishment of a forest along the banks of the river.

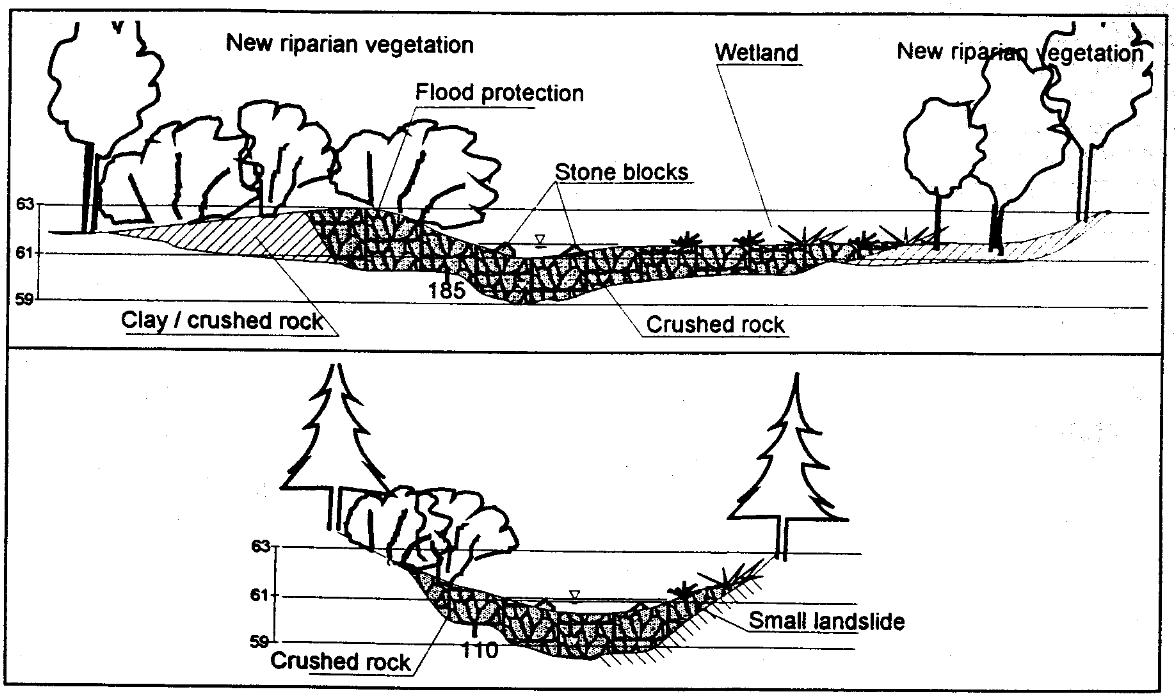
Affected, vegetation-free areas consisting of soil, clay, etc. will be sown with grass seeds to prevent surface erosion and promote biomass production. Combined with machine or manual planting of indigenous species of trees, the grass will be the first stage in the succession of natural riverbank vegetation and towards "natural" flood plain vegetation.

Otters have been observed by NINA (the Norwegian Institute for Nature Research) which has suggested a number of implementation instructions that have been included in the plans to reduce the impact on this species (Rosendal et al., 1992). Among other things, NINA suggested that the project deploy scree containing large blocks of stone, keep some water open during cold spells and reconstruct the natural patterns that occur at the mouths of the small tributaries.

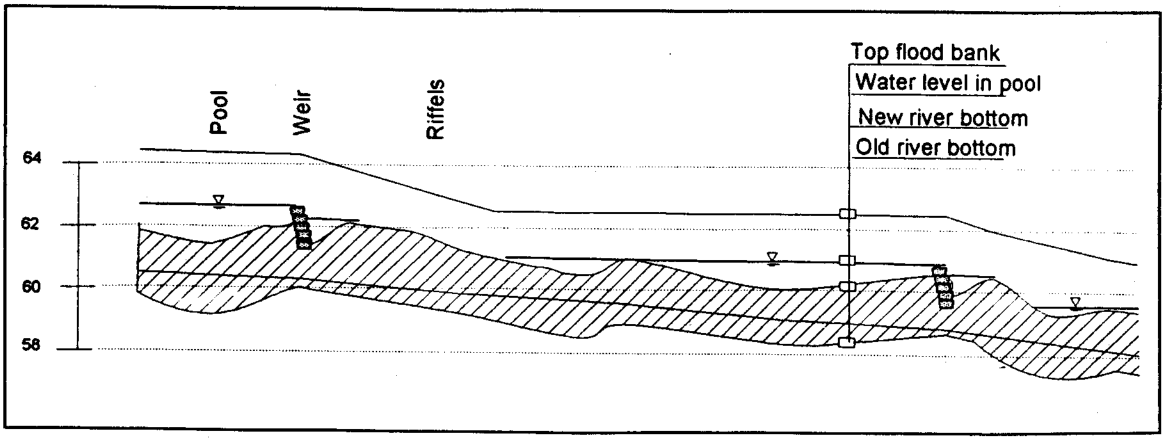
The safeguarding of this area is the largest single project of its kind in Norway. This is why there was a strong consensus that the situation should be used to learn more about how watercourse encroachments on such a scale would influence local conditions. Ably assisted by the country's foremost research institutions, NVE initiated a number of preliminary studies to survey the situation prior to intervention. The Gråelva will later be followed up by post-project studies. The titles of completed and current studies comprise the references on page 6.

Through "The Norwegian Biotope Adjustment Programme" which was initiated and headed by NVE, we have gained access to a broad range of experience regarding a variety of ways to improve and adjust conditions for fish and other species whose biotopes are dominated by the river (Eie et al., 1995). We have benefitted from this experience in connection with the Gråelva Project.

The specific section of the plan is based on detailed surveys of the river and on calculations performed in connection with water lines and rock sizes. This section proposes that measures consistent with the principles in the general section be worked out in detail. In this connection, the plan calls for details of the elevation and shape of the river, and the width, depth and length of each individual section. It also provides a detailed outline for establishment of a new vegetation.



An example of a cross-section of a river plain, and a cross-section of a narrow valley in Gråelva.



An example of a lengthwise section.

PERFORMANCE

Construction began in 1992. Thus far approximately 220,000 m^3 of gravel and clay have been brought in to raise the river bed and make it erosion-proof. It is estimated that a total of roughly 450,000 m^3 of gravel will eventually be needed. Thus far work has been completed on about 3 km of the river bed, while another 1 km has been affected by the construction. Budgetary constraints do not permit work to be done on a continuous basis. Generally speaking, gravel is driven out to the site in the winter, while spreading and landscaping must be performed in the summer. Completion cannot be expected before the year 2000 at the earliest.

Experiences to be included in further planning efforts

To achieve a good filtering effect through the clay and big rocks, it is important that the gravel be well sorted (graduated by size). It is especially advantageous to use bulldozers to spread and shape the gravel. Bulldozers crush some of the gravel under their belts, achieving good coverage of the top layer so the water does not disappear into the rock tips when the water level is low. The disadvantage is that the river bed was even and smooth. Accordingly, bigger rocks had to be added to create more resting and hiding places for fish.

Sowing with ordinary grass seeds on sterile clay soil was not particularly successful. The results were far better with a seed mix that had been tested and used on sites used in connection with the Olympic Games in Lillehammer 1994.

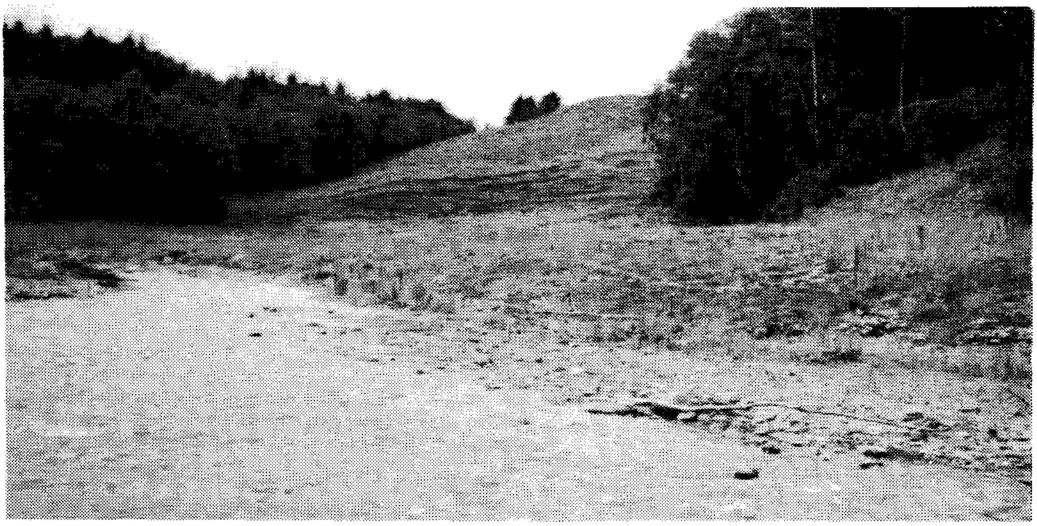
The gravel was transported during the winter when the ground was frozen and covered with snow. However, conditions made it difficult to remove the vegetation cover before the rock was tipped. Consequently, soil and other matter containing vegetation had to be taken from another site.

EFFECTS OF THE EFFORTS

The volume of suspended fine particles in the water near the mouth of the Gråelva indicates reductions from 445 tonnes/day in 1992 to 305 tonnes/day in 1993, and then to 210 tonnes/day in 1994 (Husebye, scheduled for publication in 1996).

As regards fish and benthic organisms, studies made up until 1995 indicate that population densities are lower in the clay-affected, non-secured stretches than in the rock-lined stretches of the river. The production of insects and young fish, especially fry, appears to have increased considerably in the rock-lined areas of the watercourse (Berger et al. Scheduled for publication in 1996).

A typical section of the river prior to institution of the safety measures.



A cross-section following the implementation of safety measures, at an early stage of vegetation succession.

REFERENCES

Berger, H.M., Breistein, J.B., Nøst, T.H. and Mejdell Larsen, B. (1994). "The effects of reduced volumes of silt on water quality, the benthos and fish in the Gråelva. Preliminary studies 1990 - 1992." *NINA, Project Report No. 291.* (Abstract in English)

Eie, J.Ar., Eie J.At. and Brittain, J. (1995). "The Norwegian Biotope Adjustment Programme." *NVE. Power and the Environment, No. 21.* (Abstract in English)

Fremstad, E. (1992) "Vegetation and flora along the Gråelva in Stjørdal, Nord-Trøndelag" *NINA, Project Report No. 155.*(Abstract in English)

Gregersen, O. (1990) "Assessment of the safety of the Gråelva in Stjørdal, Nord-Trøndelag", *NGI Report, 900053-1.* (Norwegian)

Gregersen, O. (1991) "Ground studies and proposals for safety measures along the Gråelva", *NGI Report, 900053-2.* (Norwegian)

Gregersen, O. (1992) "Supplementary ground studies for the assessment of safety measures", *NGI Report, 900053-3.* (Norwegian)

Gregersen, O. (1994) "Supplementary ground studies for the assessment of safety along the Gråelva", *NGI Report, 900053-4.* (Norwegian)

Husebye, S. (1994) "Sediment transport in the Gråelva". *NVE Publication No. 3, 1995.* (Norwegian)

Johnsen, M. (1991) "Securing against erosion and landslides in the Gråelva watercourse in Skjølstadmarka, Preliminary Project. *NVE, Plan No. 8341-1.* (Norwegian)

Johnsen, M. and Sandnæs, T.O. (1993) "Securing against erosion and landslides in the main course of the Gråelva in Skjølstadmarka, Detailed Plan, Part 1. *NVE, Plan No. 8341-2.* (Norwegian)

Johnsen, M. and Sandnæs, T.O. (1995) "Securing against erosion and landslides in the main course of the Gråelva in Skjølstadmarka, Detailed Plan, Part 2. *NVE, Plan No. 8341-3.* (Norwegian)

Meidell Larsen, B. (to be started and scheduled for completion in 1996). "The distribution and population status of riverine pearl oysters in the Hofstad River, Nord-Trøndelag. *NINA, Project Report*

Rosendal, E. and Heggberget, T.M. (1992) "Registration of the incidence of otter in the Gråelva in Stjørdal, Nord-Trøndelag. *NINA, Project Report No. 174.* (Abstract in English)

Sandersen, F. (1992) "The Gråelva, Stjørdal. Survey of sediment origins and sediment budget estimates". *NGI Report, 924037-1.* (Norwegian)

Thingstad, P.G. (1993) "Preliminary ornithological studies regarding the safety measures to combat erosion and landslides in the Gråelva River in the municipality of Stjørdal". *University of Trondheim, Zoology Dept. Memo 1993-11.* (Norwegian)

IMPROVEMENT OF THE RETENTION QUALITIES OF POOLS IN SALMON RIVERS

M. CASTONGUAY
PESCA Conseillers en biologie Inc.
596, boul. Perron, C.P. 11
Maria, Quebec, Canada
G0C 1Y0

C. MARCHE
École Polytechnique de Montréal
C.P. 6079, succ. "Centre-ville"
Montréal, Quebec, Canada
H3C 3A7

ABSTRACT

With the purpose of keeping track of the numbers and distribution of Atlantic Salmon (Salmo salar) in holding pools during their return upriver in the spring, a method of estimating the holding potential of any pool in a river was developed. This method was re-examined in 1994 during an evaluation of the feasibility of restoring to an acceptable standard the holding pools in the Bonaventure River. The methodology involves five stages and results in two indices: IPRM (index of potential of retention at mean flow) and IPR (index of potential of retention at some specific rate of flow) measured on a scale of 0 to 20. Applied to a river reach the proposed methodology allows us, via a simple comparison of IPRM indices, to obtain a global picture of the pools throughout that reach and of how they are affected by water levels in the river. It has highlighted the importance of maintaining the optimal state of turbulence peculiar to each pool. Thus the method allows us to identify those holding pools which can be restored with the most chance of success and for each, those characteristics which have to be improved.

KEY-WORDS: Retention Potentiel/ Holding salmon pool/ Numerical Simulation method/ Habitat Management.

INTRODUCTION

Salmon anglers spend a considerable amount of money for access to natural wilderness where there is a good chance of catching salmon and where anglers are few (Ministère de l'Environnement et de la faune, 1995). When the fishing season opens, and water levels are high, salmon are distributed throughout all the pools in the river (Groupe SALAR, 1992; PESCA, unpublished data). However, as water levels recede in July and August, the salmon congregate in a smaller number of pools known to experienced anglers as holding pools. The anglers frequent only those few pools known to be of good holding quality. This results in reduced global economic productivity. There is therefore good reason for salmon river managers to improve pool quality under low water conditions, particularly in sectors where the river is easily accessible, since angling is at its height in July when water levels are apt to be low.

However, our understanding is not yet complete enough to allow certain identification of those pools likely to be chosen by salmon regardless of water level conditions (Chabot, 1994; Tremblay, 1994; Hydrotech, 1987; Frenette *et al.*, 1970). This is why we have tried, by means of a comparative study of several pools in the Bonaventure River (Gaspé Peninsula, Quebec) to: 1- determine the environmental factors governing the retention of salmon in a pool, 2- define a concrete quantitative methodology for evaluating a pool's retention properties, 3- use this methodology for classifying pools and then to compare the results with anglers' reports about each pool, 4- identify the pools that would be suitable for habitat modification, and 5- specify the modifications most likely to improve those pools' retention properties.

The comparative study of pool quality was performed on the middle reach of the Bonaventure River (sectors C and D). This section is particularly interesting because it includes not only pools renowned for their salmon holding capacity in all water level conditions but also those which are of mediocre capacity throughout the year.

CHOICE OF THE DETERMINANT FACTORS FOR THE RETENTION PROPERTIES OF A POOL

A pool can be defined as a depression in the bed of a river or stream resulting in a local slowing of the current thus giving salmon a place to congregate and rest during their return upstream in the spring. A pool's quality therefore depends on its ability to attract salmon, and is proportional to the length of time salmon stay in the pool, and of course to the total number of salmon swimming upstream to spawn. In order to refer to the attractiveness of a pool independently of the number of fish actually involved, we use the term "retention potential". The potential of retention of a pool may be fully or only partially achieved depending on the year or on the different periods of the summer.

To establish a method of evaluating the retention potential of pools, we have to consider two postulates:

1- The retention potential of a pool essentially depends on local factors and characteristics of the river referred to as macrohabitat (Bovee, 1982; Gras et al. 1994).

To describe a pool 33 variables have been considered. Some are dimensional and others refer to the general environment of the pool (e.g. the arrival of fresh water). The methodology should be constructed from adimensional values in accordance with the principles of dimensional analysis (Langhaar, 1951) so the list of dependent variables was enhanced by the addition of 10 dimensionless variables.

2- The retention potential of a pool is influenced by certain macrohabitat variables that may be considered independent, so that their effects on each other, even where they exist, are minor by an order of magnitude (Kline, 1965).

As a result, the value of the retention potential of a pool may be expressed by a relation of the type:

$$IPRM = \Sigma k_i CPR_i (V_i) / \Sigma k_i$$

where IPRM represents the index of retention potential of the pool at mean flow
CPR_i is the contribution of the variable V_i to the IPRM
and k_i is the relative weight assigned to this contribution.

CHOICE OF REPRESENTATIVE POOLS

To establish the mathematical nature of the various CPR_i functions, a detailed study of nine well-known pools was realized. The salmon frequentation in a pool served as a biological indicator of its retention potential. This information was acquired for each pool by means of a poll of river management authorities, as well as of guides and anglers who regulary visit the Bonaventure River.

As a second source of information, the results of the salmon count carried out since 1984 also allowed an evaluation of the retention capacity of each pool as a function of water level. However we had to presume that the distribution of salmon observed during these counts of spawners is representative of what prevails throughout the fishing season for the water flows recorded. Water levels were established from hydrological data (R. Couture, ministère de l'Environnement et de la Faune, comm. pers.). The analyses were performed on monthly data from the last ten years, with three water flow rates as outcome: summer flood discharge flow (45 m^3/s), mean summer flow (27 m^3/s) and dry season flow (17 m^3/s).

The pools harboring a large number of salmon regardless of the water level were classified as retention pools (pools #52, #59 and #80); those containing salmon only during periods of high water were classed as high-water resting pools (pools #60 and #75) while those containing salmon only during periods of low water were classed as low-water resting pools (pools #64 et #84). Pools holding very few salmon no matter what the water level were classified as being poor. (pools #74 and #96). The level of frequentation by salmon as a function of water flow was graded on a scale of 0 to 20, with "20" being excellent and "0" very weak. In this manner we obtain numerical indices of quality for each of the nine pools.

CHARACTERISATION OF THE REPRESENTATIVE POOLS

The nine typical pools chosen in the study were very methodically analysed during field trips to the Bonaventure River on August 11 and 12 and September 3, 1994. Velocities and depths of water were measured across representative transverse sections of each pool. The slowest water flows were measured using a current meter. Using the measured depths and velocities on the river, the flow rate and the slope of the water surface profile, a good approximation to the hydraulic conveyance could be constructed.

A two-dimensional numerical hydraulic model was constructed for the sample pools to permit a water surface profile calculation for the three flow rates at high, mean and low water levels. Thus the readings and the calculations performed during this stage of the study yielded a portrait of each representative pool at the three different flow rates, taking into account all of the factors affecting retention.

FINE TUNING THE METHOD OF EVALUATION OF POOL RETENTION POTENTIAL

Identification Of The Significant Factors Governing Retention

A correlation (by linear regression) was established among the quality grades of each sample pool and the values for each of the pools' retention factors estimated by using the model. It became obvious that the assigned quality was not dependent on all variables to the same degree (r varying between 0.01 and 0.7). Those which stand out as having a high correlation (r varying between 0.7 and 0.3) are:

VPAV: Variability in the minimum downstream depth in relation to the maximum depth of the pool,
VVAV: Variability in the minimum speed of flow in the pool in relation to the maximum downstream flow rate,
VPAM: Variability in the minimum upstream depth in relation to the maximum depth of the pool,
VVAM: Variability in the minimum speed of flow in relation to the maximum upstream flow rate,
HMAX: Maximum depth of the pool.

Some simple laws were determined to take this influence into account. They are displayed below:

VPAV	V1=	20 - 12 x VPAV	$0 < VPAV < 1$
VVAV	V2=	8 + 12 x VVAV	$0 < VVAV < 1$
VPAM	V3=	16	$VPAM < 0.2$
	V3=	28 - 40 x VPAM	$VPAM < 0.5$
	V3=	8	$VPAM > 0.5$
VVAM	V4=	53 x VVAM	$0 < VVAM < 0.3$
	V4=	16	$VVAM > 0.3$
HMAX	V5=	6	$HMAX < 1.3$
	V5=	(5 x HMAX) - 1	$1.3 < HMAX < 3.4$
	V5=	16	$HMAX > 3.4$

Calculation Of The Retention Potential Index

The introduction of these laws into the formula:

$$IPRM= \Sigma k_i CPR_i (V_i)/ \Sigma k_i$$

yields the calculation for IPRM if we apply a weight k_i to each variable.

A number of calculations were performed in which different k_i were assigned in order to evaluate how the calculated indices of retention potential (IPRM) for the nine sample pools compared as against the quality grades ascribed to each pool. Then the absolute value of the differences between them was totaled for each. It appears that the resulting cumulative error is at a minimum for a value of k_i everywhere simply equal to 1.0. in other words giving equal importance to all variables. This choice has been retained for the present. in the absence of more copious data allowing further minimization of the error.

Table 1 shows the values of the calculated IPRM as well as the assigned quality grade for each of the nine sample pools constituting the source of data on which the method is based. The index is valid only under conditions of mean flow. and provides a comparison of the level of comfort offered by each of the pools only under such conditions.

Table 1: Mean Retention Potential Index (IPRM) for the pools calculated using the mean discharge characteristics and evaluated on a scale of 0 to 20.

Pool	% of annual salmon	Quality grade	VPAV	VVAV	VPAM	VVAM	HMAX	IPRM
52	4.9	16	16.7	18.0	15.3	16.0	10.0	15
59	3.6	15	14.7	13.1	18.0	14.4	15.0	15
60	1.3	12	16.7	14.0	17.0	16.0	13.5	15
64	1.1	12	14.0	15.0	8.0	16.0	6.0	12
74	0.3	6	4.3	9.1	9.5	4.6	5.5	7
75	0.9	10	10.1	8.7	8.0	2.3	7.5	7
80	5.8	16	16.2	12.2	13.4	16.0	10.0	14
84	0.4	8	11.2	16.8	8.0	16.0	12.0	13
96	1.1	12	15.7	9.6	13.8	7.6	16.0	13

VPAV: Variability in the minimum downstream depth in relation to the maximum depth of the pool.
VVAV: Variability in the minimum speed of flow in relation to the maximum downstream flow rate.
VPAM: Variability in the minimum upstream depth in relation to the maximum depth of the pool.
VVAM: Variability in the minimum speed of flow in relation to the maximum upstream flow rate.
HMAX: Maximum depth of the pool.

In parallel with this IPRM. an IPR peculiar to each flow rate (IPR) can also be calculated by considering each variable at its corresponding flow rate. However the order of magnitude of the variations among the nine pools being much greater than that of the varations in the variables as a function of flow rate in each pool. the sensitivity of IPR to variations in flow calculated this way was weak. and required a second analysis.

Variability Of The Index Of Retention Potential AS A Function Of Flow Rate.

Certain pools are permanent (retention potential constant and less susceptible to water level conditions) while the quality of others is highly dependent on these conditions.

As a result. the actual IPR of a pool at a specific flow rate can be expressed by

$$IPR = IPRM + \Sigma IPR \times FP$$

in which

IPR	represents the index of retention potential at a specific flow rate.
IPRM	represents the index calculated for mean rate of flow.
ΣIPR	is the correction to the index assigned to the effect of flow.
FP	is an adjustable weighting factor.

IPRM is known. Determination of any IPR therefore depends on calculating ΣIPR and FP.

For the nine pools selected. the various retention factors able to affect the IPR were analysed once again as a function of flow rate. The correlations. recalculated for each pool. among the quality grades they were given at different flow rates and the retention factors affected by the flow. turned out to be very weak. with the exception of one variable: the upstream depth. PAM. For this variable every pool exhibited the same behavior. illustrated by figure 1. where it is evident that each pool has an optimum upstream depth (PAM_{opt}) which maximizes the IPR. This value PAM_{opt} is different for each pool.

A re-analysis of the measured and calculated values for the different flows therefore highlighted the dependence of this optimum upstream depth and on the turbulent agitation within the pool (figure 2). The relation found. for the nine pools at the three flow rates used in analysis. was of the form:
$PAM_{opt} = a + b\ FMAX$ where the best correlation was obtained for a= 1.87 and b= - 2.90

Substituting for PAM_{opt}. we can redraw figure 2 in its relative form:

$$\Delta(IPR) = \text{function of } \Delta(PAM).$$

ΔPAM being equal to $PAM - PAM_{opt}$. we can establish figure 3 yielding the correction affecting the IPR as the upstream depth varies as a function of flow rate. This correction is given quite simply by the equations:

When $\Delta PAM < -0.3$	$\Delta IPR = -2$
When $-0.3 < \Delta PAM < 0$	$\Delta IPR = 26.6\Delta PAM + 6$
When $0 < \Delta PAM < 0.6$	$\Delta IPR = -13.3\Delta PAM + 6$
When $0.6 < \Delta PAM$	$\Delta IPR = -2$

It remains then to choose the weighting factor FP to be applied against the flow rate correction. New regressions performed on the quality grades acquired via the poll as compared to the IPR values calculated at the two extremes of flow rate allowed us to fix the value of the weighting factor at about 0.8. achieving the maximal coefficient of regression (r = 0.85) (Table 2). All of the relevant elements having been defined. the proposed evaluation method can be used.

Table 2: Retention Potential Index (IPR) adjusted as a function of the influence of discharge on each pool investigated.

Pool	Quality grade	Flow	IPRM	FMAX	PAMopt.	PAM	DIFF PAM	DIFF IPR	FP	IPR
52	14	High	15	0.46	0.54	1.00	0.46	-0.2	0.8	14.9
	16	Low	15	0.46	0.54	0.40	-0.14	2.4	0.8	16.9
59	16	High	15	0.18	1.35	1.10	-0.25	-0.6	0.8	14.5
	16	Low	15	0.18	1.35	0.60	-0.75	-2.0	0.8	13.4
60	16	High	15	0.35	0.86	1.04	0.19	3.5	0.8	17.8
	9	Low	15	0.35	0.86	0.63	-0.23	0.0	0.8	15.0
64	8	High	12	0.50	0.42	1.10	0.68	-2.0	0.8	10.4
	16	Low	12	0.50	0.42	0.70	0.28	2.3	0.8	13.8
74	6	High	7	0.40	0.71	0.80	0.09	4.8	0.8	10.8
	6	Low	7	0.40	0.71	0.45	-0.26	-0.9	0.8	6.3
75	12	High	7	0.20	1.29	1.70	0.41	0.5	0.8	7.4
	6	Low	7	0.20	1.29	1.05	-0.24	-0.4	0.8	6.7
80	16	High	14	0.35	0.86	1.10	0.25	2.7	0.8	16.2
	16	Low	14	0.35	0.86	0.50	-0.36	-2.0	0.8	12.4
84	8	High	13	0.20	1.29	2.00	0.71	-2.0	0.8	11.4
	14	Low	13	0.20	1.29	1.30	0.01	5.9	0.8	17.7
96	12	High	13	0.31	0.97	1.70	0.73	-2.0	0.8	11.4
	13	Low	13	0.31	0.97	1.40	0.43	0.3	0.8	13.2

FMAX: Turbulent agitation
PAM: Upstream depth
DIFF IPR: See laws in the text
IPR: IPRM + (DIFF IPR . FP)

PAMopt: Optimum upstream depth
DIFF PAM: PAM - PAMopt
FP: Weighting factor

DEMONSTRATION OF THE METHOD

The methodology developed herein can be divided into five steps leading successively to two indices: IPRM and IPR taking values on a scale from 0 to 20. The five steps are:

Step 1- Bathymetry of the pool
Step 2- Modeling of the pool's drainage
Step 3- Establishment of the index of potential of retention at mean flow
Step 4- Determination of the optimum upstream depth
Step 5- Qualification of the pool at different flow rates

In order to demonstrate the application of the method. we will evaluate the Second East Pool (#94). This pool is situated in sector D of the Bonaventure River in its confluence with the Reboul River. It stretches approximately 40 meters along a section where the river is from 25 to 30 m wide. The current is fast at the right and slow at the left side of the pool. Anglers find that this pool is poor at all water levels.

Step 1- Bathymetry Of The Pool

The first step consists in performing a bathymetric evaluation of the pool including both its upstream and downstream approaches. at any flow as long as it is known. The bathymetric readings of the Second East pool yielded knowledge about three significant transverse sections and the limits of its banks.

Step 2- Modeling The Drainage

Readings there allowed us to establish a predictive mathematical model of the flow conditions at the three flow rates. Following the validation of the model. drainage calculations were performed for the flow rates typical at high summer waters (41 m^3/s). mean waters (25 m^3/s) and low waters (16 m^3/s). The values for the pool's retention factors at mean flow rate calculated from the model are:

VPAV:	13.3	VVAV:	9.3	VPAM:	8.0
VVAM:	4.2	HMAX:	8.0		

Step 3- Establishment Of The Index Of Retention Potential At Mean Flow

A calculation based on the laws pertaining to each of the retention factors selected. also taking into account the measured values. yields a value for the index of potential of retention at mean flow rate (IPRM) of pool #94.

$$IPRM= 9$$

Step 4- Determination Of The Optimum Upstream Depth

The relationship $PAM_{opt} = 1.87 - 2.9 \times FMAX$ permits us to calculate the upstream depth for optimal retention in Second East Pool. It is a function of the optimal agitation in the pool (FMAX= 0.40) determined by means of the numerical model. The calculations yield:

$$PAM_{opt} = 0.71 \text{ m}$$

Step 5- Qualification of the Pool at Different Flow Rates

The corrections to the index for the various flow rates are calculated by:

$$IPR = IPRM + \Delta IPR \times FP$$

for high summer water flow $IPRH = 9 + (-2.0 \times 0.8)$ $IPRH = 7.4$

for low water flow $IPRB = 9 + (2.1 \times 0.8)$ $IPRB = 10.7$

The various stages of the characterisation have shown that Second East Pool manifested a weak retention potential. tending to increase slightly in times of low water. This calculated quality grade for the Second East pool agrees with the evaluation made by the anglers who called this pool poor. Only 16 salmon were found in this pool during the salmon count carried out every summer since five years ago.

As applied to a segment of a river. the proposed methodology yields. through a simple comparison of IPRM's. a global picture of the pools throughout the segment. and an understanding of the sensitivity of the retention potential of each pool to the varying hydraulic behavior of the river.

PRECISION OF THE IPR EVALUATION METHOD

The proposed methodology. through a simple comparison of their indices of retention potential at mean flow (IPRM). yields a global portrait of the pools within a river reach. A pool has a weak potential of retention if its index of retention potential at mean flow (IPRM) is less than 10. Further. it can be considered a good pool if the index is greater than 13. Finally. it can be said to be sensitive to the river's hydraulicity if the IPR fluctuates around the IPRM by more than 3 as water flows vary between their extremes of high and low. Retention factors with a value less than 10 are those factors which limit a pool's retention potential even when the calculated IPRM is above 13. A good pool will be characterized by five good retention factors with their values exceeding 10.

The adjustment of the IPR evaluation method to variations in flow rate by considering PAM_{opt} and FMAX achieves a precision of 74% as opposed to 17% obtained by the estimation of the retention factors at each flow rate (Table 3). This level of precision allows us to consider the method adequate for identifying those pools amenable to successful management.

Table 3: Precision of the Retention Potential Index adjusted as a function of mean discharge

	Difference (High - Low water)				Variation		
No	**Note 1**	**IPR 2**	**adjusted IPR 3**	**Best IPR 4**	**Best method (1 X 4)**	**IPR (1 X 2)**	**adjusted IPR (1 X 3)**
52	-2.0	-0.4	-2.0	-2.0	4.0	0.8	4.0
59	0.0	-0.5	1.1	0.0	0.0	0.0	0.0
60	7.0	0.6	2.8	7.0	49.0	4.2	19.6
64	-8.0	-0.1	-3.4	-8.0	64.0	0.8	27.2
74	0.0	0.2	4.5	0.0	0.0	0.0	0.0
75	6.0	0.7	0.7	6.0	36.0	4.2	4.2
80	0.0	-0.6	3.8	0.0	0.0	0.0	0.0
84	-6.0	1.0	-6.3	-6.0	36.0	-6.0	37.8
96	-1.0	-1.4	-1.8	-1.0	1.0	1.4	1.8
				TOTAL	190.0	5.4	94.6
				Confidence(%)	100.0	2.8	50.0
				Precision obtained at each flow rate (%)	100.0	17.0	70.0

EXPERIENCED ANGLERS' PERSPECTIVE

As stated during a meeting with several experienced anglers. the choice of these retention factors of a physical nature. as a means of evaluating the retention potential under average flow conditions. seemed logical to them. For them. a good pool presents:

- a generous flow of water from upstream. expressed here by VVAM and PAM;
- an escarpment at the downstream end of the pool. portrayed by VPAM;
- an ample basin allowing the pool to harbour salmon even at low water. here taking the value HMAX;
- zones allowing the salmon to rest immobile. represented by the value of FMAX;
- a rapid rise in the bottom of the pool at its dowstream end. indicated by VPAV;
- an acceleration of the current downstream favoring the formation of a run. e.g. a rapid flow with little agitation. depicted by VVAV.

Four of these factors come out of a comparison between the minimal and maximimal values for a pool. so they express the heterogeneity of the pool. which is a quality of habitat favorable to salmon. The only environmental retention factor exhibiting a relation to the number and duration of visits by salmon to a pool is the arrival of fresh water. However this retention factor has not been formulated mathematically. Its influence will be considered during the process of choosing which pools to improve.

ASSESSEMENT OF THE COMFORT CONDITIONS OPTIMAL FOR SALMON

The relation of the retention potential as a function of the maximum agitation and the upstream pool depth (figures 1 and 2) may be represented by a single nomogram (figure 4). A retention zone (and therefore a comfort potential) can thus be identified as circumscribing the line of optimal retention. Each pool at each flow rate may be represented by a point whose ordinate gives the maximal agitation level and whose abscissa represents the upstream pool depth (figure 4). The pools were studied at both high and low water flow rates. and the two corresponding points are shown for each pool. The straight line segments joining the two points in each case characterize their respective behavior for varying flow rates.

The general position of this straight line segment in relation to the comfort zone indicates whether the pool has a tendency to be better at times of high water or low. or whether a retention pool. If the line intercepts the slope of optimal retention. then the retention potential of the pool is fully realized. at least temporarily. For example. the pools Gallagher (#52). First East (#80) and Kerby (#75) realize their potential of retention to the maximum. whether this is strong as in the cases of Gallagher and First East. or average as with Kerby. The pool #84 is a good example of a pool at low water. It ceases to be comfortable when the upstream depth rises due to an increase in flow. since the line passes outside the comfort zone as shown in figure 4.

The orientation of this line allows an immediate reading of whether or not the pool is sensitive to variations in flow rate. Projecting the segment in a direction perpendicular to the slope of optimal retention provides the indicator. The length of the projected segment is inversely proportional to the permanence of the pool. A pool's characteristic segment generally has a positive or zero slope. For example. among the pools studied. #96 Kransell is the least influenced by the flow rate.

DESIRABLE INTERVENTIONS FOR THE PURPOSE OF IMPROVING THE RETENTION QUALITIES OF POOLS

A detailed analysis of the various retention factors show that Second East pool. our example above. suffers particularly from a high upstream depth and from velocities which are too high at its entrance and at its exit.

The modifications would consist first in installing a sill (A) at its downstream end to raise the water level slightly within the pool. The second would aim at slowing down the flow in the pool and forcing it to the left. This would consist of placing one deflector in the upstream zone and two associated structures in the current within the pool. Tests of these modifications on a physical scale model have shown positive effects. Calculations of the various retention factors have established the IPR values for the pool modified. These values are 11. 11 and 15 at high summer flood discharge. mean and low water flow. respectively.

CONCLUSION

The application of this IPR method identifies those pools most amenable to improvement. The analysis of weak factors governing retention and IPRM values will give priority to methods of intervention likely to improve pools which already show promise from an angler's point of view.

REFERENCES

BOVEE. K. 1982. A guide to stream habitat analysis using the Instream Flow Incremental Methodology. Instream Flow information Paper 12. USDI. Fish and Wildlife. Services FWSI OBS-82/26. 221 p.

CHABOT. A. 1994. Projet expérimental d'aménagement de mini-structures de rétention du saumon sur la rivière Saint-Jean (Côte- Nord). Salmo Salar: vol. 17 no.3. p.23 et 24.

FRENETTE. M.. C. Rae et B. Tétrault. 1970. La création des fosses artificielles pour le saumon. p.17-24.

GRAS. R.. C. Sabaton et P. Gosse. 1994. Le comportement des poissons mis en équation; Aménagements hydrauliques et débits reservés. Epure 43. 19 p.

GROUPE SALAR. 1992. Plan de mise en valeur du potentiel salmonicole de la rivière Bonaventure. Rapport présenté à l'Association des pêcheurs sportifs de la Bonaventure inc. 98 p.

HYDROTECH Inc. 1987. Restoration of the salmon pools around Horse Island; Design and construction of the remedial works. Document présenté à la Société de gestion du saumon de la rivière Cascapédia inc. 36 p.

KLINE. S. J. 1965. Similitude and approximation theory. McGraw and Hill. New York.

LANGHAAR. H. L. 1951. Dimensional analysis and theory of models. Wiley. New York.

MINISTÈRE DE L'ENVIRONNEMENT ET DE LA FAUNE. 1995. Enquête auprès des pêcheurs québécois de saumon. Direction des communications et du marketing et la direction des territoires fauniques. 58p.

TREMBLAY. G.-A. 1994. La restauration de fosses de la rivière Saint-Jean. Côte-Nord. Salmo Salar: Vol. 17 no 1. pp. 37-38.

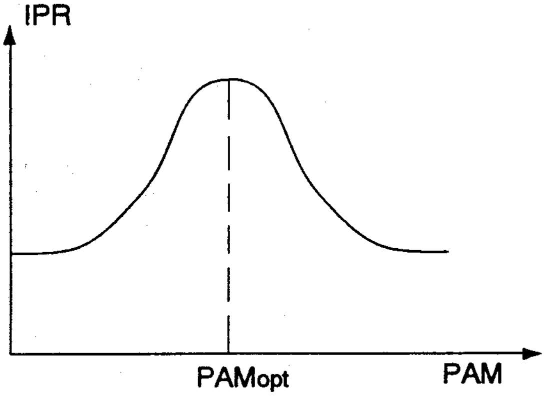


Figure 1: Optimum upstream depth (PAMopt) maximising the retention potential index (IPR) for salmon pools at any discharge.

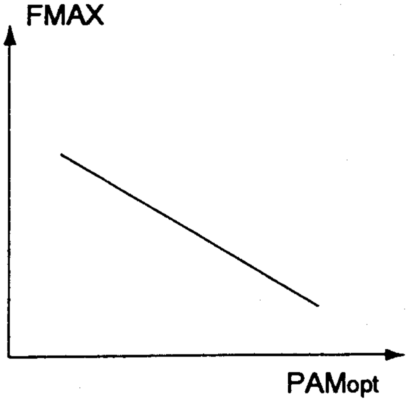


Figure 2: Relation between the optimum upstream depth (PAMopt) and turbulent agitation (FMAX) in a salmon pool.

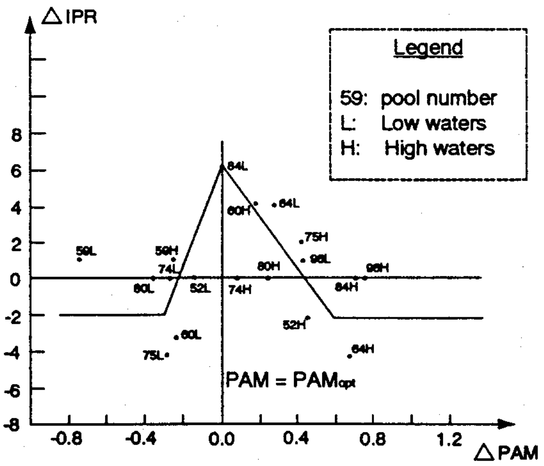


Figure 3: Variation in the retention potential index (IPR) as a function of upstream depth (PAM) determined by the discharge.

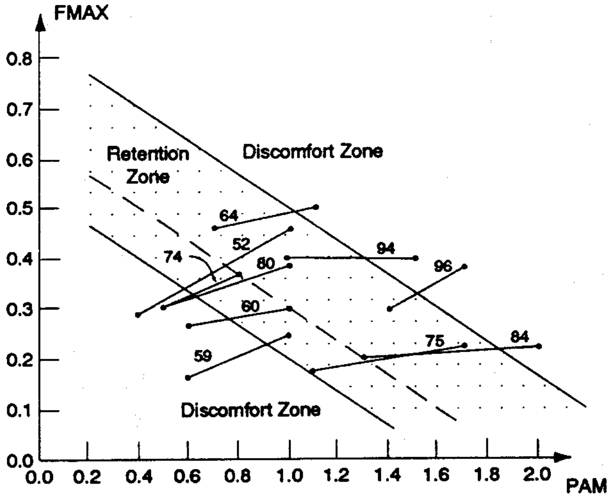


Figure 4: Retention potential as a function of discharge for the salmon pools studied.

MONITORING OF FISHERIES ON THE RIVER BLACKWATER

Karen Fisher, HR Wallingford
Robert Myers, John Lyness, John O' Sullivan, University of Ulster

From 1986 to 1992 river improvement works on the River Blackwater, Northern Ireland, and surrounding tributaries were undertaken with the primary objectives of improving drainage. During this time it was decided that the fish habitats on the River Blackwater and its tributaries should be enhanced to reinstate the previously good habitat conditions. The work has been undertaken by the Department of Agriculture for Northern Ireland (DANI).

The main part of the work involved the provision of spawning and nursery areas for salmon and trout. The changes to the river have involved the introduction of groynes or low stone weirs to aerate the flow, the introduction of large stones and boulders to provide nursery habitats for fish, the provision of holding pools where the fish can shelter and the provision of some fish passes. Over three hundred sites have been created throughout the River Blackwater catchment.

There is no data available on the hydraulic or ecological performance of the channels and the fish habitats. Through a research project, funded by the Ministry of Agriculture, Fisheries and Food (MAFF) in England and supported by DANI, a monitoring network was established to collect hydraulic, morphological, environmental and fisheries data from the River Blackwater and its tributaries. Although the hydraulic data collected in the project is being linked with the fisheries and morphological data, this paper concentrates on the hydraulic performance of the fisheries measures which can be used as an input to the ecological assessment of the measures.

For the research project, ten sites on the River Blackwater and its tributaries were monitored with one site being an undrained, natural site and the remaining nine at improved fisheries sites. The hydraulic information collected has shown the roughness characteristics of the different habitats, pools, spawning and nursery areas at a range of flows. Certain trends appear in the data. The paper shows that the hydraulic roughness over the low stone weirs is very high (>Manning's n of *0.1*) and the roughness values over the different habitats is generally greater than that given in the literature, especially at low flows. The roughness values calculated at the natural site are even greater than those calculated at the man-made sites with Manning's n values recorded as being in the range *0.03* to *0.3*. The paper indicates some surprising results which were found such as the roughness values measured over the nursery areas (up to Manning's n values of *0.07*), with large boulders, were generally smaller than those measured in the spawning areas (up to Manning's n of *0.2*), with smaller gravels.

Recommendations regarding further work as a result of this monitoring are given. The assessment of the hydraulic performance of the environmental and fisheries enhancements, which have been incorporated into the rivers will provide valuable information for the design of fisheries measures in the future.

KEY-WORDS: Hydraulic Performance/fisheries measures/low stone weirs/groynes/pools/spawning and nursery areas/roughness.

INTRODUCTION

The River Blackwater originates in Co. Monaghan and flows through Northern Ireland into the southern edge of Lough Neagh (Figure 1). During the 1980's the river was heavily dredged to improve drainage from the surrounding catchment which was predominantly used as grassland pastures. It was hoped that by lowering the water table, bearing capacities of the soils could be increased and desirable biological changes induced. Although these measures were hydraulically effective and land drainage did indeed improve, essential gravels and vegetation making up fish habitats were removed and consequently, fish stocks dropped. To counteract this depletion of fish stocks the Department of Agriculture in Northern Ireland (DANI) initiated a program of works to re-instate the fishery potential of the river to pre-drainage scheme values. During 1994 and the early part of 1995, approximately 350 artificial fish habitats consisting of spawning grounds characterised by a gravelly river bed in shallow water, and nursery areas comprised mainly of boulders were constructed along the river (Fisher et al, 1995). Groynes and specially built low stone weirs in a variety of configurations backed up by gravel were included in each habitat.

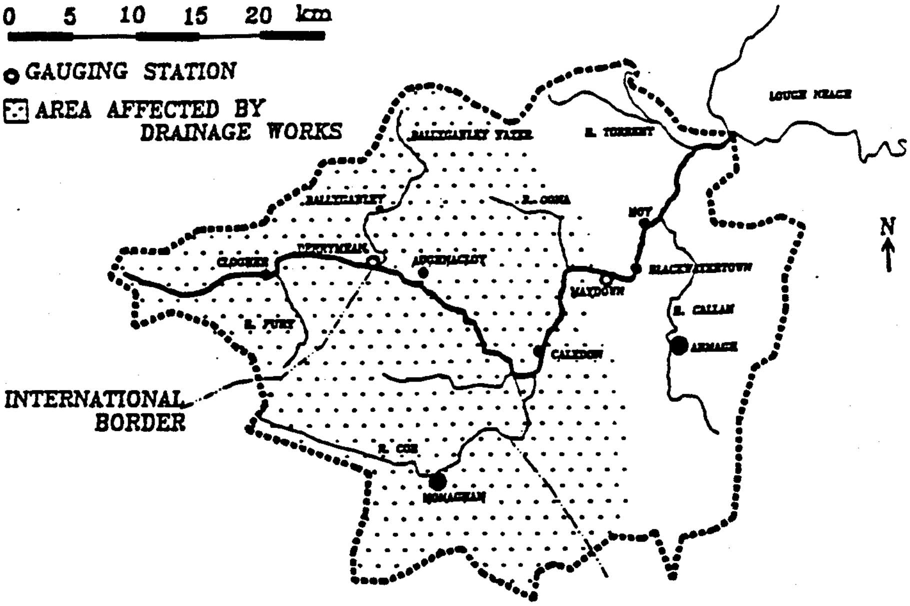


Figure 1: River Blackwater catchment

At present, the use of environmentally acceptable features such as groynes and low stone wiers in watercourses to improve their appeal for wildlife is considered desirable (Reeves and Bettess, 1990; Hydraulics Research, 1988). However, their addidion to channels is a recent development in river engineering and subsequently, little research, particularly that concerned with field data acquisition has been carried out in the UK (New Civil Engineer, 1996). Consequently, a program of research has been undertaken by the University of Ulster in collaboration with HR Wallingford Ltd. and funded by the UK Ministry of Agriculture, Fisheries and Food (MAFF). This research was carried out from October 1994 to March 1996. This duration allowed a comprehensive range of flow conditions to be analysed.

The objectives of this research were twofold:

- to understand the hydraulic characteristics of environmental features incorporated into artificial fish habitats,

and secondly,

- to establish hydraulic characteristics of a productive fish habitat.

FIELD DATA ACQUISITION

In order to carry out a detailed hydraulic study of the river it was necessary to concentrate on a selection of specific sites which best represented the overall characteristics of the river. Subsequently, ten sites were chosen for this investigation. Each site was chosen to incorporate the required elements of a fish habitat and hence consisted of a pool at the upstream end of the reach, a spawning area and a nursery area. Gravel supporting groynes and low stone weirs provided riffles for increased oxygenation of spawning grounds. A control site (site 10), on a tributary of the River Blackwater where no drainage works were carried out had a naturally occurring pool and riffle sequence with a nursery area. The sites selected offered a wide range of channel sizes, flow conditions, a range of types and numbers of structures, and were geographically spread over the Blackwater's catchment. This range was intended to give a good indication of the impact of the different features at different sites. Details of these sites are displayed in Table 1. A detailed topographical survey incorporating six river cross-sections and longitudinal sections was carried out at all sites.

Table 1: Details of sites studied in investigation

Site Name	Site No.	Bed Width (m)	Gradient	Features
Bawn's Burn	1	3	1/112	Small Stream; Straight Reach
Abel's Bridge	2	12	1/930	Large Stream; Bend at Upstream End of Reach; Angled Groyne
Lisdoart Mill	3	8	1/800	Medium Stream; Straight Reach; 3 Groynes; Good Fishing Site
Lisdoart Bridge	4	8	1/650	Large Stream; Straight Reach; Good Site For Fishing
Forthill Bridge	5	10	1/1320	Large Stream; Straight Reach; 1 Groyne; Velocities Too High For Good Fishing
Caledon Bridge	6	20	1/2700	Large Stream
Omagh Road	7	8	1/200	Medium Stream; Meandering Reach; 2 Groynes
Burn's Bridge	8	15	1/4100	Large Channel
River Oona	9	4	1/630	Medium Channel
River Fury	10	4	1/370	Medium Stream

The hydraulic measurements in this investigation consisted of gauge board readings and river gauging.

Gauge Board Readings

Six gauge boards were installed at each site as follows:

(1) upstream of the fish habitat;

(2) in the centre of the pool area;
(3) in the middle of the spawning area;
(4) at the downstream end of the spawning ground;
(5) in the middle of the nursery area;
(6) downstream of the nursery area.

These locations are shown in the site sketches represented in Figure 2. All gauge boards were levelled to a benchmark at each site and could be read to an accuracy of $\pm 0.01m$. At higher flows when turbulence gave rise to an unsteady water surface profile this degree of accuracy was reduced to $\pm 0.03m$. In addition to the boards, maximum level recorders were installed at each site at the upstream and downstream ends of the reach (i.e. attached to the boards at cross-sections 1 and 6). This ensured that in the event of a floods peak water levels were obtained at times when it was impossible to visit sites. Peak levels were matched to peak discharges obtained from nearby gauging stations.

Boards at all sites were read on a minimum of fifteen occasions during the project with the exception of Caledon Bridge (site no. 6) where boards were not in place until October '95. Boards were read at Caledon Bridge on seven occasions. Fifteen readings taken in the course of a year allowed for a full range of flow conditions to be assessed.

River Gauging

Current metering at all sites was carried out to comply, as far as possible, with the requirements outlined in *BS 3680: Part 3A: 1980*. Measuring locations (shown in Figure 2) were selected at a point in a straight river reach which was clear and unobstructed by trees and other obstacles and hence, provided the potential for a typical velocity distribution. All sites were gauged on a minimum of five occasions.

DATA ANALYSIS

Four sites have been selected for data presentation, representing a variety of habitat conditions. The habitat at Forthill Bridge (site no. 5) is in a relatively large stream and incorporates a variety of features. A concave shaped low stone weir is included along the reach (between cross-sections 2A and 3A). Lisdoart Bridge (site no. 4) is a medium sized stream and Bawn's Burn (site no. 1) having an average bed width of 3m is a narrow stream of low flow. Also included is the hydraulic analysis of the naturally occurring fish habitat on the River Fury (site no. 10).

Data analysis at all sites consisted of producing water surface profiles for a range of flow conditions, establishing stage-discharge relationships and calculation of Manning's roughness coefficients and Darcy Weisbach friction factors. Steady flow was assumed.

Water Surface Profiles

By combining the gauge board readings with the topographical surveys water surface levels were calculated at site cross-sections for all flows. A range of the resulting surface profiles are shown in Figure 3.

A low stone weir placed on the bed of a channel results in a local increase in the velocity of flow and a corresponding reduction in the elevation of the water surface. The effect of these weirs is less noticeable at high discharges when

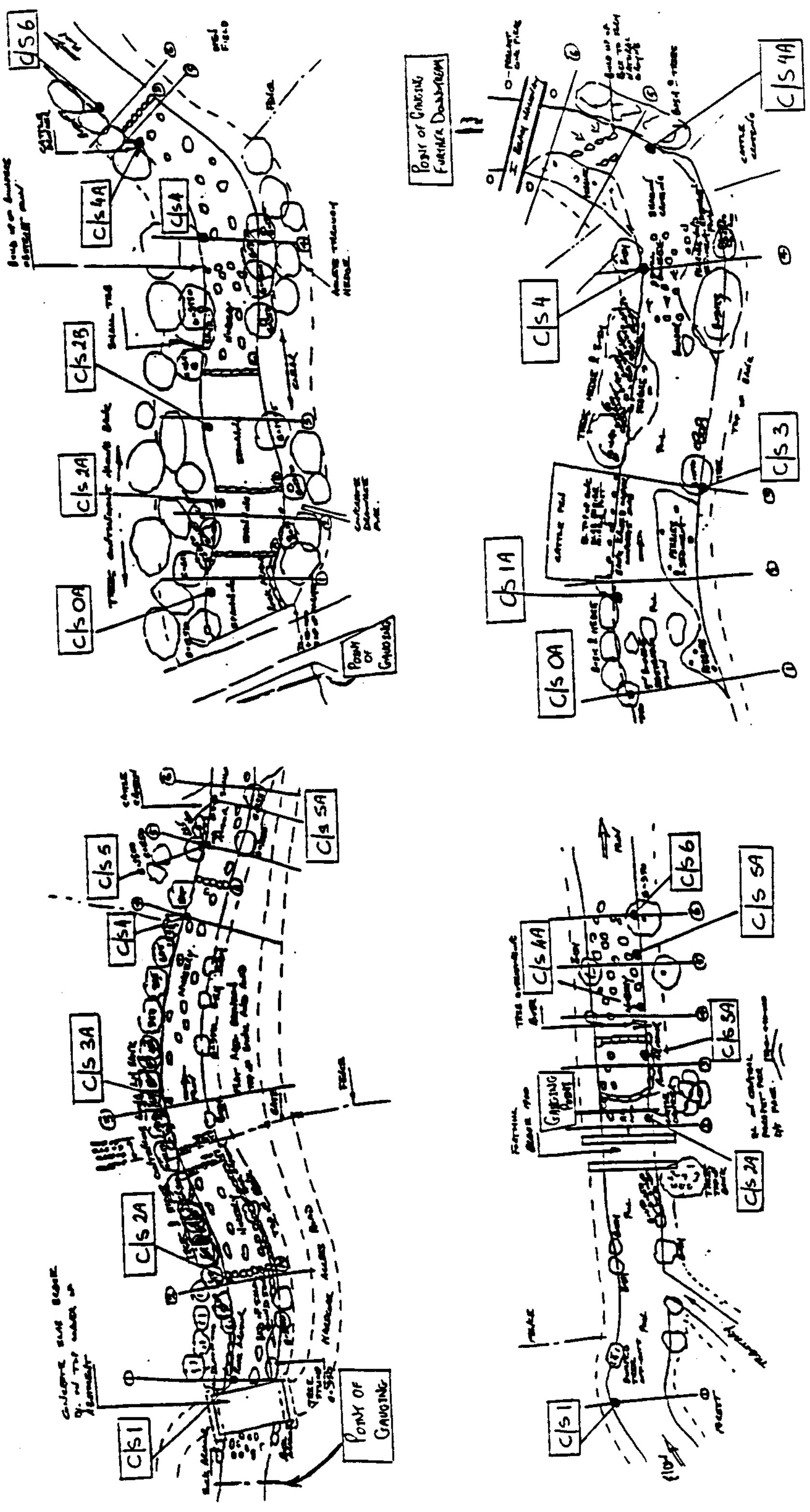


Figure 2: Clockwise from top left; site sketches of Bawn's Burn, Lisdoart Bridge, Forthill Bridge, and the River Fury

they are fully submerged. The size and type of weir, be it protruding the river bed or level with the upstream end of the reach, also appears to influence the water surface profile. Profiles from Forthill Bridge where the weirs are essentially deep steps result in a steeper depression of water surface elevation than the shallow protruding weirs at Lisdoart Bridge.

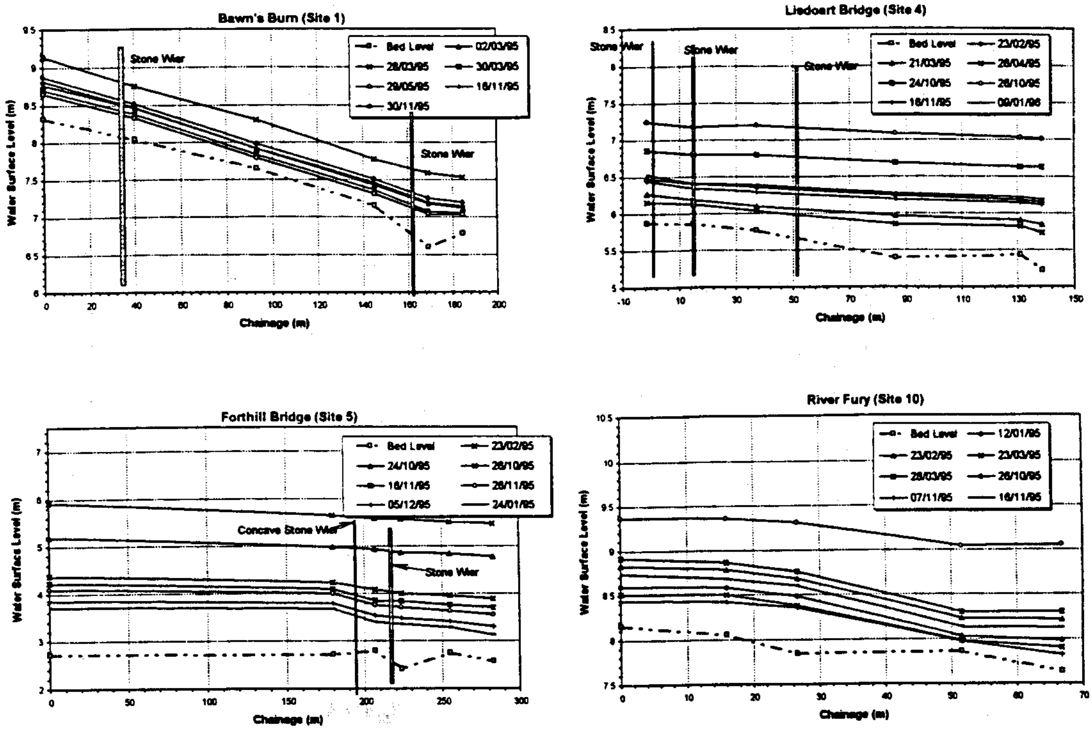


Figure 3: Water surface profiles for sites shown

Depth-Discharge Relationships

By using the water depths obtained from the gauge boards located at cross-sections no. 1, depth-discharge relationships were developed for all sites. These relationships were plotted on logarithmic scales for the four sites under discussion and power law equations established (Figure 4). The high values of the correlation coefficients associated with each relationship indicate the accuracy that was obtained.

Discharges for greater depths of water were obtained by extrapolating these relationships. Extrapolations were restricted to approximately *30%* of the maximum discharge. Discharges outside this range were obtained from local gauging stations and supplied by the Department of Agriculture for Northern Ireland (DANI).

Flow Resistance Coefficients

Manning's roughness coefficients and Darcy Weisbach friction factors were calculated at all sites for a range of water depths and discharges using the following equations:

(1) $$n = \frac{AR^{2/3}S^{1/2}}{Q}$$

where:
Q is the discharge (m^3/s);
A is the cross-sectional area (m^2) for the relevant depth of water;
R is the hydraulic radius (m);
S is the gradient of the water surface profile.

Friction factors f were calculated using the *Darcy Weisbach* equation:

(2) $$h_f = \frac{4flv^2}{2gD}$$

but $D = 4R$, $\frac{h_f}{l} = S$ and $Q = Av$

(3) $$\Rightarrow f = \frac{2SRA^2g}{Q^2}$$

where:
g is the acceleration due to gravity, taken to be 9.81m/s^2.

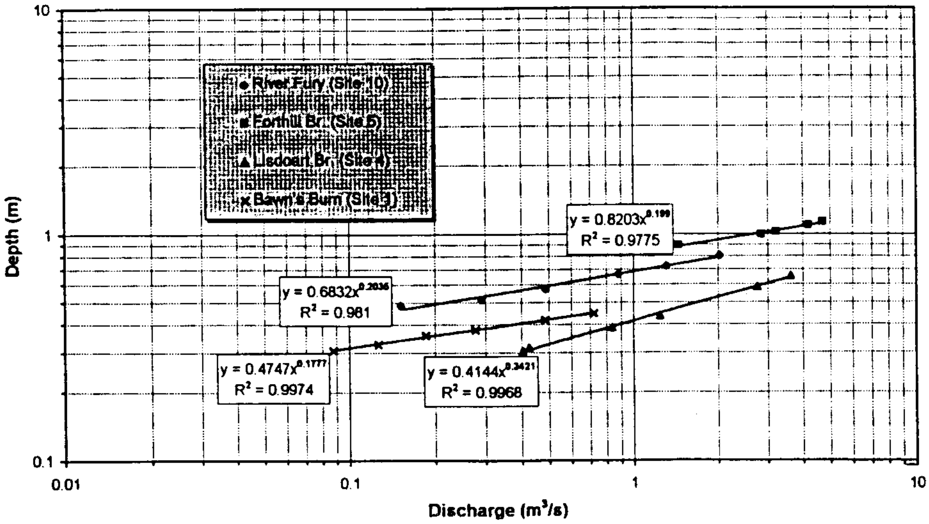


Figure 4: Depth-discharge relationships for sites shown

Graphs of discharge and depth versus Manning's *n* Coefficients are shown in Figure 5. Coefficients in these curves were calculated using the mean gradient of water surface profiles between cross-sections 1 and 6 along a reach. Cross-sectional areas, wetted perimeters and hydraulic radii were calculated for each section, summed and the resultant average applied to equation (2).

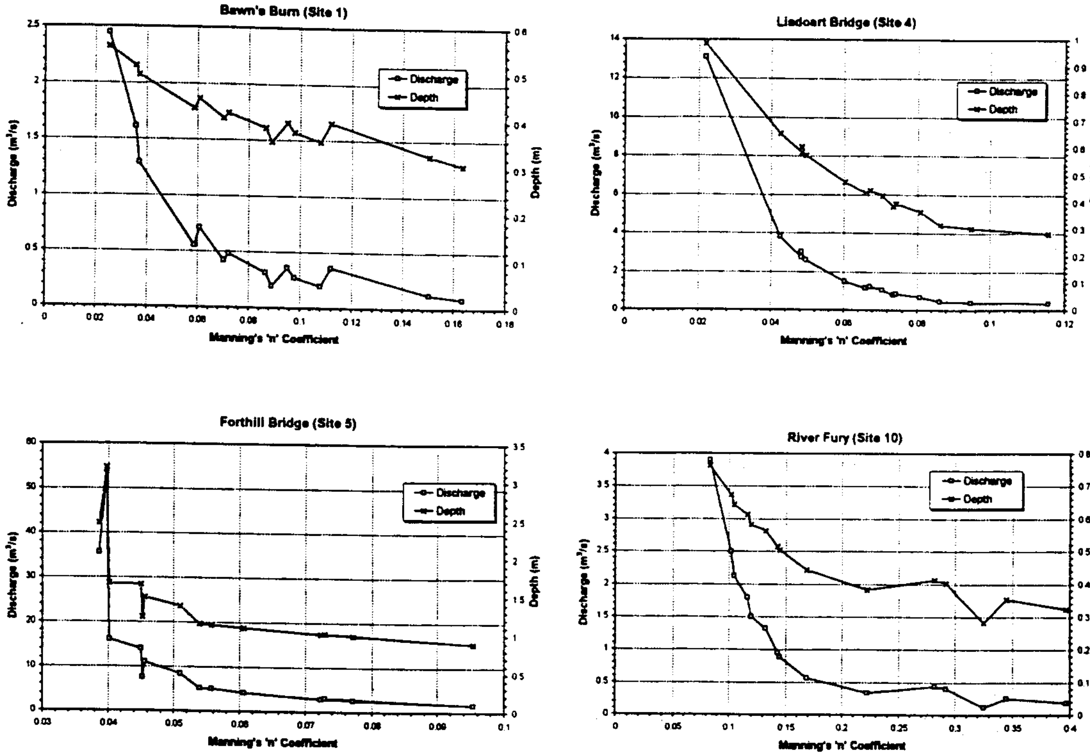


Figure 5: Manning's roughness coefficient v discharge and depth for overall reach at sites shown

As expected, Manning's *n* values decrease with increased depth and discharge. However, these values varied considerably and are, particularly at low discharges unexpectedly high, present literature specifying values ranging from *0.025-0.08* $(s/m^{1/3})$ for rivers of similar bed material and alignment (Chow, 1959). Figure 5 shows that the Manning's roughness coefficient was calculated as being *0.1* and above at low flows. The natural habitat and control site on the River Fury exhibits a significantly higher and larger range of *n* values than other sites. During periods of low flow, the rough river bed at this site results in a torturous flow pattern where water must flow around river bed material rather than over it. This possibly accounts for the excessive bed roughness at this site.

The construction of fishery measures and more specifically the addition of gravel to spawning grounds, boulders to nursery areas and the introduction of stone weirs and groynes all contribute to an increase in river bed roughness. However, these features, and stone weirs in particular cause localised increases in *n* values which are masked by an overall river reach analysis. Only by examining the variation in *n* values along a river reach can the impact of such features be assessed. Figure 6 shows variations of *n* with discharge at mid-points between cross-sections at the four sites under discussion. It is apparent from this figure that flows over isolated areas of high roughness, particularly weirs, result in increased values of Manning's roughness coefficients.

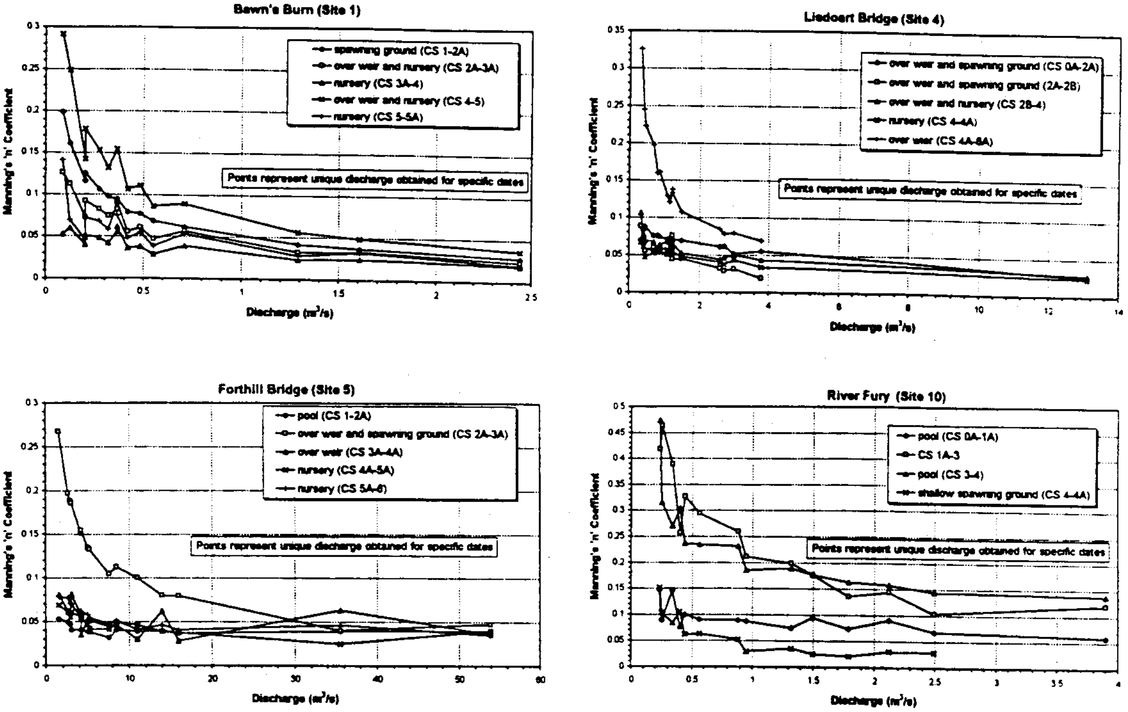


Figure 6: Variations in Manning's *n* at intermediate points between cross-sections for sites shown

The range of *n* values obtained for the different features over a range of discharges which were incorporated into the habitats at each site are shown in Table 2. At low discharges, the effect of the weirs is most noticeable and *n* values are significantly larger than elsewhere in the reach. Higher discharges however, result in drowning of the weirs and at these depths *n* values across the weir approach those obtained elsewhere. These isolated high *n* values contribute to the high *n* values obtained for the overall reach. At high flows Manning's roughness coefficients tend to one value along the reach, independent of the river features.

Table 2: Manning's roughness coefficients for features described

Site Name	Site No.	Pool	Spawning Ground	Nursery	Over Weir	Over Weir and Spawning Ground	Over Weir and Nursery
Bawn's Burn	1	-	0.028 - 0.199	0.019 - 0.141	0.037 - 0.292	-	0.023 - 0.126
Lisdoart Bridge	4	-	-	0.023 - 0.067	0.069 - 0.325	0.025 - 0.088	0.028 - 0.106
Forthill Bridge	5	0.037 - 0.053	-	0.025 - 0.069	0.028 - 0.080	0.04 - 0.267	-
River Fury	10	0.059 - 0.304	0.030 - 0.105	-	-	-	-

By comparing the effects of the different features, certain trends are apparent. The range of *n* values for nursery grounds (sites 1, 4 and 5) and spawning areas (Sites 1 and 10) are quite similar, although the upper end of the range at Bawn's Burn is greater than elsewhere. The nature, gradient and alignment of the channel at this site differs significantly from the other sites under discussion. It's steep gradient results in high flow velocities, which, combined with sharp curvature results in a more turbulent flow than was noticed at other sites for the range of discharges studied. The channel at both Lisdoart and Forthill Bridge is of smoother curvature with larger radius which results in lower values of *n* (Chow, 1959). In addition, the stage at Bawn's burn was consistently less than that recorded at Lisdoart and Forthill Bridge and consequently, the depth of cover over both the low stone weirs, and boulders included in the nursery area at this site, is less than that at the other sites. This results in a higher relative roughness at Bawn's Burn than at the additional sites under discussion.

It would be expected that the nurseries, being comprised of heavier bed material would have a higher roughness than other areas of the habitat. This was not the case and *n* values of spawning grounds at Bawn's Burn and the natural habitat at the River Fury, although comparable at low discharges, were higher than those calculated in the nurseries of other sites.

More generally, heavy foliage on the River Blackwater also assists in increasing roughness. Mid-channel vegetation is limited to mosses and encrusted algae. These will not significantly influence channel flow (Hydraulics Research, 1988). On the other hand, vegetation and foliage at channel edges contribute to increased roughness. This increase varies seasonally and is greatest during the summer months. Although fluctuation in *n* values can be attributed to variations in flow depth, existing literature (Powell, 1978) indicates that channel vegetation is the main cause for varying *n* values. During this research, dense foliage whether it be water iris, reeds, overhanging trees or a combination of these was noticed at all sites. At moderate and low flows the presence of this vegetation will significantly retard the flow and result in higher *n* values.

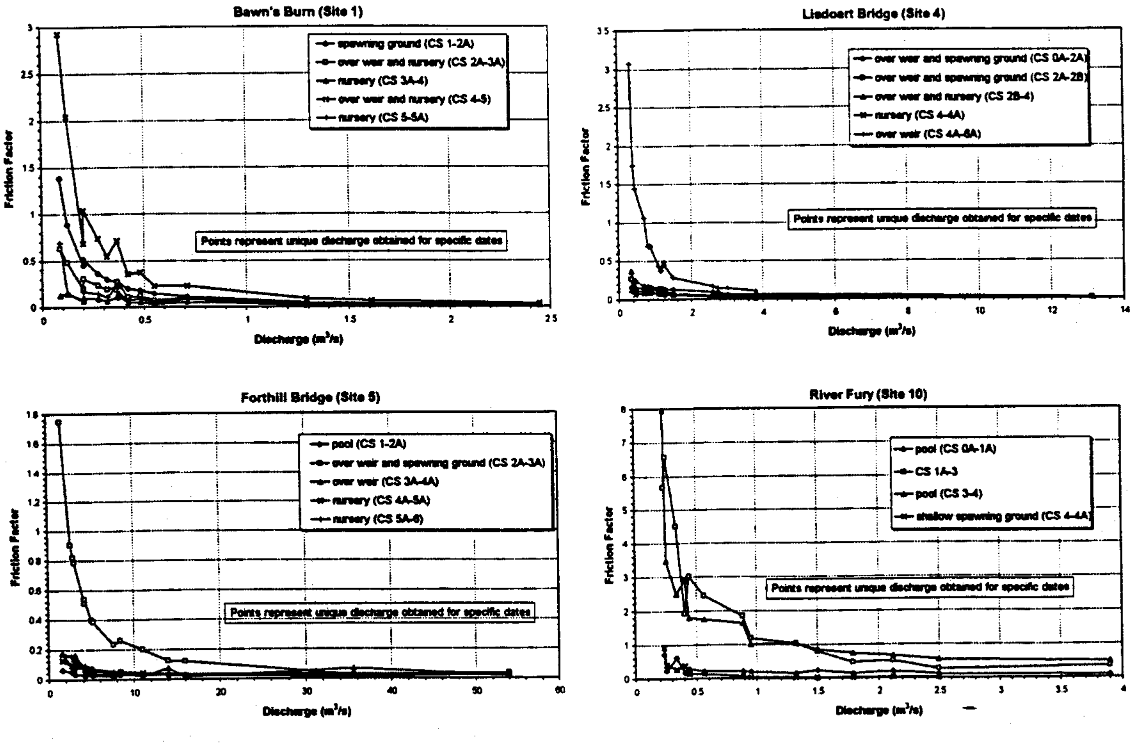


Figure 7: Variations in friction factors at intermediate points between cross-sections at sites shown

Friction factors were calculated using equation (3). These were also plotted against discharges at intermediate points along the reach (Figure 7). These plots serve to reaffirm the trends of Manning's roughness coefficients along the river reaches; friction factors being highest across low stone weirs.

CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE RESEARCH

The post-scheme monitoring of fish habitats on the River Blackwater appear to indicate that the introduction of fishery measures increase river roughness. Manning's roughness coefficients for the overall reaches at sites 1, 4 and 5 are higher, and in certain cases double those predicted in existing literature, but even at extremely low discharges they do not reach the n values of the natural habitat on the River Fury. The high n values at these sites are contributed to by isolated areas of extreme roughness associated with the added environmental features. Manning's roughness coefficients in excess of *0.3* across low stone weirs is very high and, incomparable with n values attributed to any channel characteristics to be found in existing literature. Data analysis for the environmental features and river characteristics described may be more appropriately undertaken using different methods of analysis as n values in excess of *0.1* suggest a flow approaching laminar in nature. However, the full impact of these features cannot be properly assessed until comprehensive field studies involving both pre- and post-scheme investigations on a number of river reaches incorporating such measures are undertaken.

Only by a better understanding of the hydraulic effects of these environmental features can the wider issues such as wildlife protection and amenity interests be balanced against operational hydraulic requirements. To assist in this understanding recommendations for future research, in addition to the further collection of field data, are proposed:

- a detailed series of laboratory based investigations to determine stage/discharge relationships, 3-dimensional velocity profiles, sediment transport characteristics of fisheries features. This will allow the monitoring of changes in hydraulic parameters induced by the addition of environmental features to a river reach;

- computational modelling using available computer packages to predict discharges, velocities and trends in sediment transfer. In addition the data acquired from the River Blackwater should be applied to a Riverine Habitat Simulation Model to test the packages suitability in the design of fish habitats in UK rivers;

- a series of both pre and post-scheme fishery surveys carried out over a sufficient period of time to allow a correlation of fisheries and hydraulic data to be made. Such a correlation should contribute to the establishment of hydraulic parameters for a productive fish habitat.

ACKNOWLEDGEMENTS

The completion of the research covered in this paper would not have been possible without input from other institutions. For this, the University of Ulster and HR Wallingford expresses its gratitude to the UK Ministry of Agriculture, Fisheries and Food for both its financial support and permission to write this paper. Additional support from the Department of Agriculture in Northern Ireland for their assistance in the field was also gratefully received.

REFERENCES

BS 3680: Part 3A: 1980. "Liquid Flow in Open Channels"

Channelling Effort. New Civil Engineer, pages 34--35, January 25, 1996.

Chow, V T, (1959). Open channel hydraulics. McGraw-Hill.

Fisher K *et al.* (1995). Hydraulic and environmental assessment of features ncorporated into works on the River Blackwater in Northern Ireland. In *proceedings of the MAFF Conference on river and coastal engineering, Keele, England*, pages 3.4.1--3.4.8.

Hydraulics Research Report EX1799 (1988). Assessing the hydraulic performance of environmentally acceptable channels, HR Wallingford, England.

Powell K E C, (1978). Weed growth - a factor of channel roughness, in hydrometry. Herschy R W (ed), John Wiley.

Reeve C E and Bettess R., (1990). Hydraulic performance of environmentally acceptable channels. In *Proceedings of The International Conference on River Flood Hydraulics, Wallingford, England*, Paper H1, 17--20.

APPLICATION OF THE PHYSICAL HABITAT SIMULATION (PHABSIM) MODEL AS AN ASSESSMENT TOOL FOR RIVERINE HABITAT RESTORATION TECHNIQUES.

C.R.N. Elliott
Institute of Hydrology
Crowmarsh Gifford, Wallingford, Oxfordshire, OX10 8BB,
U.K.

D.J. Willis
Environment Agency, Thames Region, Department of Water Management,
Kings Meadow House, Kings Meadow Road, Reading, Berkshire, RG1 8DQ
U.K.

M.C. Acreman
Institute of Hydrology
Crowmarsh Gifford, Wallingford, Oxfordshire, OX10 8BB,
U.K.

ABSTRACT

Historical river management in the UK has resulted in widespread degradation of physical and biological habitat quality. The legacy of past river engineering works has resulted in considerable potential for the restoration of riverine habitats as a sustainable method of improving the fisheries and wildlife resource. A key requirement for effective habitat restoration is the integration of hydrological and ecological techniques to assess the success of the work in producing the desired changes in habitat availability and target species.

The paper reviews the first application of the Physical HABitat SIMulation (PHABSIM) model to assess a restoration project in the UK. The project was undertaken on a small lowland river in southern England, where a variety of restoration techniques were implemented to improve the physical habitat diversity in a section of previously channelized river and increase the carrying capacity of target fish species.

A PHABSIM study reach was selected which included the area targeted for habitat restoration as well as areas which did not require significant improvement. PHABSIM calibration data were collected both before and after the scheme, and simulations were carried out to examine the changes in physical habitat (as defined by flow depth, velocity and dominant substrate) and the habitat availability for selected target species. Quantitative fish population surveys were undertaken pre- and post-scheme.

This initial study demonstrates the potential use of PHABSIM to quantitatively assess the resulting physical habitat changes and to examine habitat availability for target species given suitable habitat suitability indices. It has also indicated that there is potential for the model to be used to assist in the design of such restoration schemes.

KEY-WORDS: PHABSIM / Habitat restoration / Physical habitat / Evaluation / Habitat Suitability Indices

INTRODUCTION

Historical river management in the UK has resulted in widespread degradation of physical and biological habitat quality. The historical requirement to protect people and land from flooding has resulted in a legacy of insensitive land drainage and flood defence works (Holmes, 1993). Channelisation of rivers in England and Wales has been considerable with an estimated 24% of main river having been affected (Brookes *et al.* 1983).

The legacy of past activities, along with the recent progress in approaches to river management, provide considerable potential for the restoration of riverine and floodplain habitats as a sustainable method of improving the fisheries and wildlife resource. A recent survey of the UK Environment Agency fisheries managers responsible for England and Wales indicated that an estimated 21% of riverine fisheries had potential for physical habitat restoration (Mann & Winfield 1992).

There have been many factors behind the recent progress in the approach to UK river management, including environmental legislation, agricultural policy, improved information, education etc. (Ward *et al.* 1994). Riverine habitat and restoration activities are now routine with considerable public and private resources being expended.

A review of riverine fisheries habitat restoration in the UK has identified common areas of weakness in project management, including the subjective identification of limiting habitat factors and poor project appraisal (Mann & Winfield 1992). The review concluded that although significant resources were being expended on this activity, relatively little was being learnt and used to promote future effectiveness.

Given the relatively limited knowledge of the response of fish populations to physical habitat restoration techniques, there is a clear need to improve our project management and to develop tools to allow the assessment of such schemes. One key area is the need to evaluate the effectiveness of restoration techniques in producing the desired improvement in habitat availability and hence target populations. Only through improving our understanding of the relationship between restoration technique, habitat change and population response will we advance the effectiveness of this important management activity.

The Physical HABitat SIMulation (PHABSIM) model was first used in the UK to assess the instream flow requirements of two British rivers (Bullock *et al.* 1991). The model has now been applied to over 50 sites in the UK in both operational and research and development studies, primarily towards the assessment of ecologically acceptable flows. The first operational use of the model in the UK in the assessment of water resource issues took place in 1992-93 (Johnson *et al.* 1995) and to date PHABSIM has been applied to at least 15 sites under such studies.

The use of the method to assess physical habitat in relation to alteration of channel morphology has been limited. Only one previous study has been undertaken in the UK to assess the impact of a flood defence scheme (Johnson *et al*, 1993a). Examples in the wider literature are also limited with the study by Shuler *et al.* (1994) claiming to be the first to assess post project effectiveness of stream habitat improvement. This paper reviews the first UK application of the model, to aid evaluation of a habitat restoration scheme.

STUDY SITE AND DATA COLLECTION

River Wey Study Site

The River Wey is a major tributary of the River Thames and its catchment is located to the south of the North Downs in Surrey, South East England (Figure 1). This project was undertaken on the northern tributary of the main River Wey, referred to as the River Wey (north).

The River Wey (north) rises from the chalk aquifer near Alton and flows east through Farnham to the confluence with the main River Wey at Tilford, a distance of 33 km. The river then flows north east to join the River Thames at Weybridge. The geology of the Wey (north) catchment is dominated by chalk with some gault clay being found in the lower reaches.

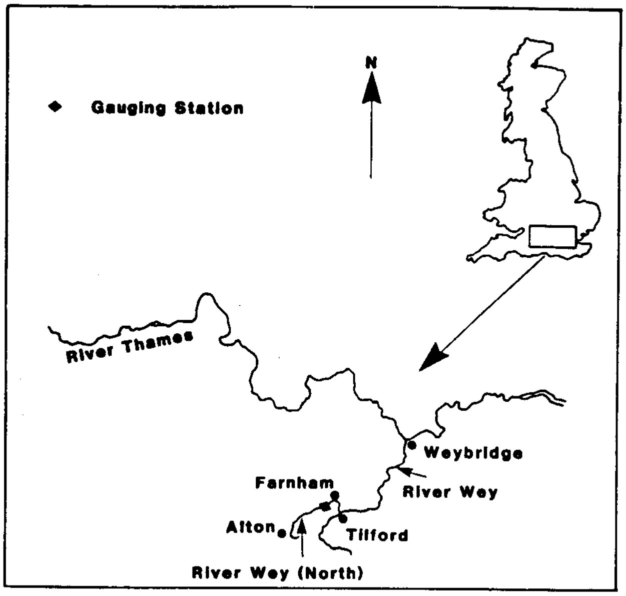


Figure 1: Location of the River Wey (north)

The study site was located to the south of Farnham (Ordnance Survey Grid Ref. SU 850 471) 2 km downstream of a flow gauging station. The catchment area at the gauging station is 19.1 km^2. The mean river flow at Farnham is 0.73 m^3s^{-1} and the 95 % exceedance flow is 0.18 m^3s^{-1}. The study site included a 260 m section of river with a mean channel width of 8.1 m. The upper 60 m of the site was morphologically diverse including riffle, glide and pool habitats. In contrast, the lower 200 m of the reach possessed poor physical habitat as a result of extensive channelization during previous river engineering works. Prior to the restoration scheme, this section of channel was of uniform width and depth, with little diversity of flow velocity or instream cover.

River Wey (North) Habitat Restoration Scheme

The primary objective of the habitat restoration scheme was to increase the carrying capacity of target fish species by improving the physical habitat within the channelized section of the river. In particular the scheme was designed to improve habitat for species including chub (*Leuciscus cephalus*) and brown trout (*Salmo trutta*).

The habitat factors that were considered to be limiting the carrying capacity of target species included:

1: The dominance of negligible flow velocities under summer discharges
2: The lack of diversity of flow velocities at a given discharge
3: The limited amount of cover provided by deeper water or instream and riparian features
4: The limited amount of gravel substrate with finer substrates forming the dominant substrate type

The restoration scheme was designed to ameliorate the limiting factors. The aim was to reduce the capacity of the existing channel at low flows and to increase the diversity of the channel morphology through the use of a variety of techniques as shown in Figure 2. These included:

1: Narrowing the existing channel to reduce capacity and increase channel velocities under summer flow conditions. This work included the construction of a two stage channel to maintain channel capacity at higher flows for flood defence purposes.
2: The channel narrowing was designed to create a more sinuous profile within the original channel to promote flow diversity.
3: The creation of pools to provide areas of increased depth and cover.
4: Placement of boulder groynes to create flow diversity and to provide cover.
5: Introduction of gravel substrate within the narrowed and shallower sections.

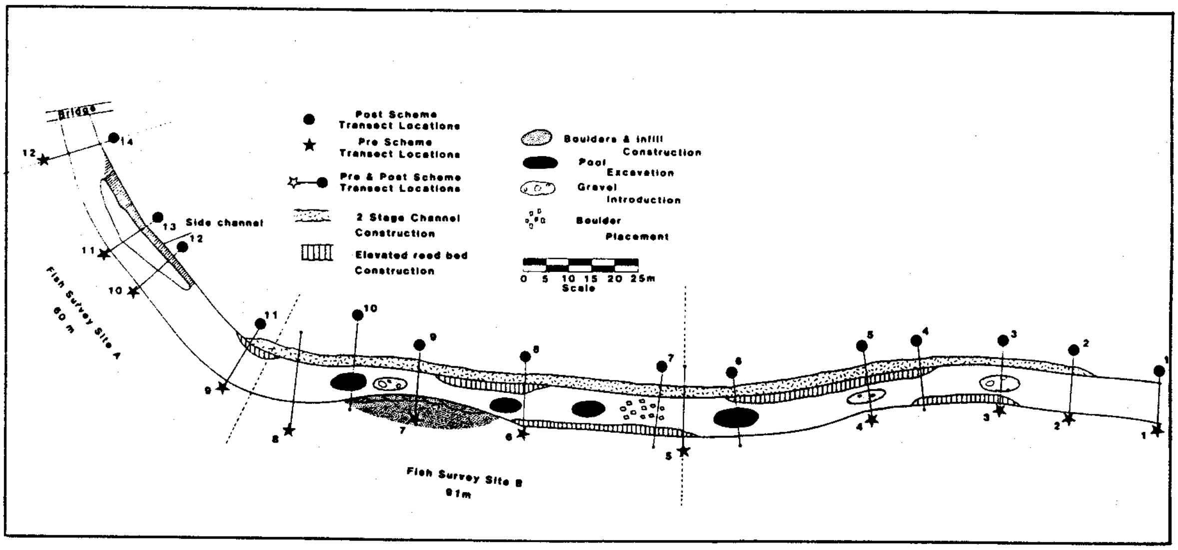


Figure 2: The River Wey (north) study site

PHABSIM Calibration Data Collection

In order to assess the availability of physical habitat within the study site prior to the habitat restoration works, data were collected to allow the calibration of the model to the river in its pre-scheme state. Twelve study transects were selected for hydraulic modelling purposes and to sample all of the habitat types available within the reach. Following the restoration works the location of the study transects was re-assessed and due to the increase in habitat diversity 14 transects were selected, some of which were common to both the pre- and post-scheme surveys. All transect locations are given in Figure 2. In both surveys PHABSIM calibration data were collected as detailed in Johnson *et al.* (1993b). The dates of each survey, along with the relevant river flow rate, are given below in Tables 1 and 2.

Table 1: **Pre-scheme hydraulic calibration data collection dates and discharge levels observed**

Calibration Data Set	Date	Discharge (m^3s^{-1})
1	28.10.93	0.560
2	2.2.94	1.605
3	26.7.94	0.417

Table 2: **Post-scheme hydraulic calibration data collection dates and discharge levels observed**

Calibration Data Set	Date	Discharge (m^3s^{-1})
1	20.3.95	1.618
2	6.6.95	0.565
3	4.7.95	0.418

Fish Habitat Suitability Data

Three fish target species and life stages were selected for assessment in this study, these were:

Adult/juvenile chub (*Leuciscus cephalus*)	(>10 cm length)
Adult brown trout (*Salmo trutta*)	(>20 cm length)
Adult roach (*Rutilus rutilus*)	(>15 cm length)

The Habitat Suitability Index (HSI) data for adult trout were developed by staff of the Environment Agency, South Western Region (Johnson *et al.* 1993c and 1995). The data upon which the curves are based were obtained through a field sampling program which included snorkelling within chalk streams in the south of England. The velocity and depth HSIs are Category 2 with the substrate data being Category 3 as defined by Bovee (1986).

HSI data for the life stages of roach and chub were originally developed under a commission from the UK Department of the Environment (Bullock *et al.* 1991). The criteria for adult roach were further developed under the National Rivers Authority R&D commission "Ecologically Acceptable Flows" (Johnson *et al.* 1993c). Development of these criteria was based on information from the literature and as a result these are Category I Bovee (1986).

The HSI data utilised in the study represent the best available information. However, the transferability of the brown trout data has not been assessed and the chub and roach data are base upon limited information (Johnson *et al.* 1993c). The limitations of the available HSI data are recognised and the results of the habitat modelling should be considered with the appropriate caution. The potential use and limitations of the HSI approach in relation to the ecology of UK riverine fish species is an area of ongoing debate.

Fish Population Survey Methods

Fish population surveys were undertaken pre- and post-scheme implementation. The survey objective was to quantify fish species composition and abundance within the study reach which could provide an initial comparison with pre- and post-scheme physical habitat availability.

The study reach consisted of two discrete sections, selected on the basis of professional subjective assessment of fish habitat quality, with the lower 200 m possessing poor habitat and the upper 60 m possessing good habitat.

The upper section was selected as fish survey site (A) (as shown in Figure 2) representing good habitat and with minimal alterations being implemented within the scheme. A 91 m section of the lower section was selected as fish survey site (B) being representative of poor habitat with significant alterations being implemented within the scheme.

Sites A and B were surveyed on 2nd November 1993 (pre-scheme) and 17th November 1995 (post-scheme). Conditions for sampling were good on both occasions with river flow at the nearby gauging station being 0.53 and 0.35 m^3s^{-1} respectively.

The survey sites were isolated by means of stop nets and electric fished with pulsed DC equipment. The four members of the survey team fished by wading upstream with two hand held anodes and two netsman. All fish captured were identified, enumerated, measured (fork length, mm) and weighed (g).

Population estimates were derived for each species using the catch removal method with 2 samples (Seber & Le Cren, 1967). Probability of captures (p) were generally high (>0.7). Results were presented as fish density (n/m2) and biomass (g/m^2).

PHABSIM MODEL SIMULATIONS

The procedures used to carry out the hydraulic simulations of flow within the River Wey (north) site were the same in both the pre- and post-scheme scenarios. In both cases the hydraulic model WSP, contained within the suite of PHABSIM programs, was calibrated using the three sets of observed water surface level (WSL) data and stage readings obtained from an automatic stage recorder installed on the most downstream study transect. WSP was then used to simulate water surface levels, at each of the study transects, at a range of selected discharges between 0.11-2.55 m^3sec^{-1}. Velocity simulations were then carried out at each of the selected flows using the IFG4 hydraulic model, calibrated using observed velocity data and utilising the observed and simulated water surface levels produced above. The distribution of the available velocities, depths and dominant substrate were examined to assess the changes in physical habitat resulting from the works.

The data produced from the above hydraulic simulations were then input into the PHABSIM habitat model HABTAT, along with the habitat suitability data for the target species life-stages detailed above, and simulations of the available physical habitat (Weighted Usable Area) were carried out.

RESULTS

Simulated Physical Habitat

The simulated physical habitat results demonstrate that the implementation of the scheme has produced a variety of changes in the physical habitat factors which were considered to be limiting to the target species. Figure 3 gives the distribution of flow depths along each of the study transects at an example discharge of 0.21 m^3s^{-1} (approximately equal to the annual 90 percentile exceedance flow). These data indicate that the scheme has resulted in a more diverse range of available depths at low flows as planned and in particular, increased amounts of pool habitat.

The distribution of flow velocities within each of the study transects is given in Figure 4. Again, the results are presented for the simulation flow of 0.21 m^3s^{-1}. The output indicates that, at the given flow, the scheme has produced a more diverse range of velocities within the study reach and that there are areas where velocities at low discharges have been increased, associated with areas where the channel cross section had been reduced. In general, however, the overall increase in velocities within the study reach was significantly less than anticipated. This may be due to the reduction in lower level channel capacity being less than required, or to backwater effects from the riffle at the bottom of the study reach.

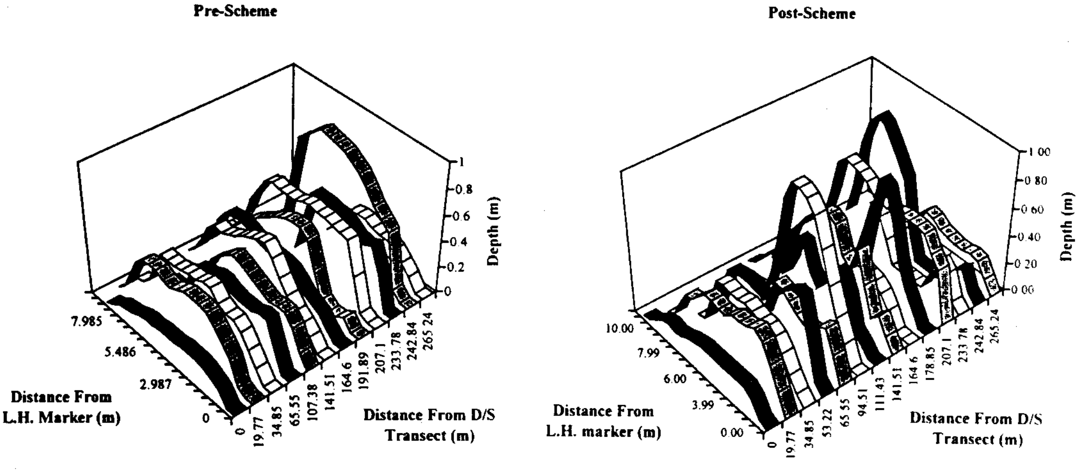


Figure 3: Pre- and post-scheme distribution of depths along PHABSIM study transects at 0.21 m^3s^{-1}.

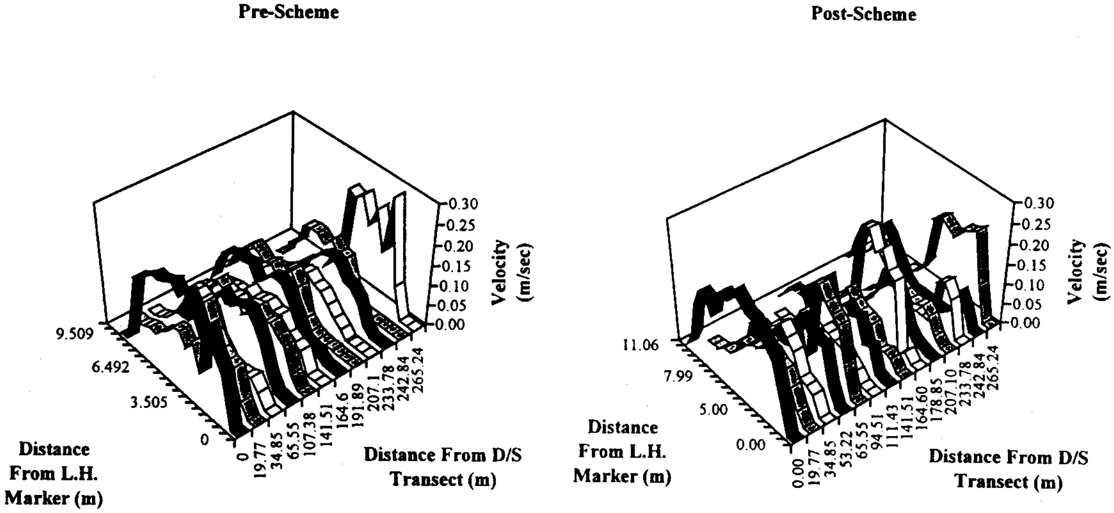


Figure 4: Pre- and post-scheme distribution of velocities along PHABSIM study transects at 0.21 m^3s^{-1}.

The relative distribution of substrate within the study reach, in the form of the percentage coverage of each of the dominant substrate classes, before and after the restoration works at a simulation flow of 0.21 m^3s^{-1} is presented in Figure 5. The results show that at low flows the diversity of substrate has increased but that the area of river covered by silt and sand has not changed. This suggests that the anticipated increase in gravel due to mobilisation of existing finer particles and introduction of additional substrate did not occur other than on a very localised basis. Overall, the proportion of gravel decreased due to additional fine sediment deposition on top of the introduced gravel. This was likely to be due to the general failure to increase flow velocities and increased sediment input resulting from the construction of the required second stage channel for flood defence purposes. This highlights the importance of assessing sediment dynamics within restoration schemes of this type.

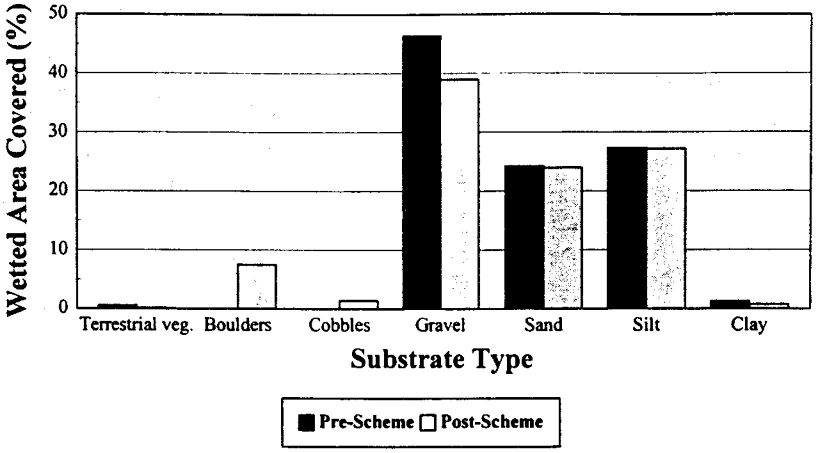


Figure 5: Pre- and post-scheme percentage of dominant substrate cover

Available Physical Habitat For Selected Target Species And Fish Population Surveys

Brown Trout (Adult)

Habitat modelling indicates that available habitat, expressed as Weighted Usable Area (WUA) vs Flow, has decreased following implementation of the scheme with the marginal reductions in habitat evident under low flows (Figure 6) increasing in the 0.5-1.0 m^3s^{-1} range. The output indicates that habitat remains comparatively limited at low flows. The fish population data indicate that although the adult brown trout have increased at both sites (Figure 7) the population at site B remains at a very low level and below that of site A.

Chub (Adult/Juvenile)

Habitat modelling indicates that there has been little change in the available physical habitat under low flows and reductions in habitat at medium and higher flows (figure 6). The results indicate that habitat remains comparatively limited under low flows. The fish population data indicates that chub have decreased at both sites (Figure 7). The population at site B has decreased significantly with an increase in the mean size of fish maintaining the biomass level. The population at site A has decreased but remains comparatively high with a biomass of over 100 g/m^2.

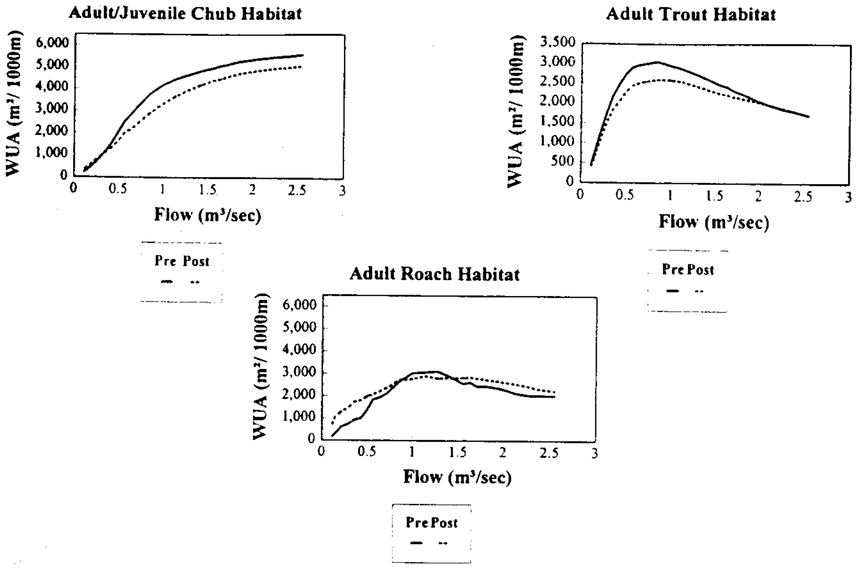


Figure 6: River Wey pre- and post-scheme available habitat area (WUA) for adult roach, adult /juvenile chub and adult brown trout.

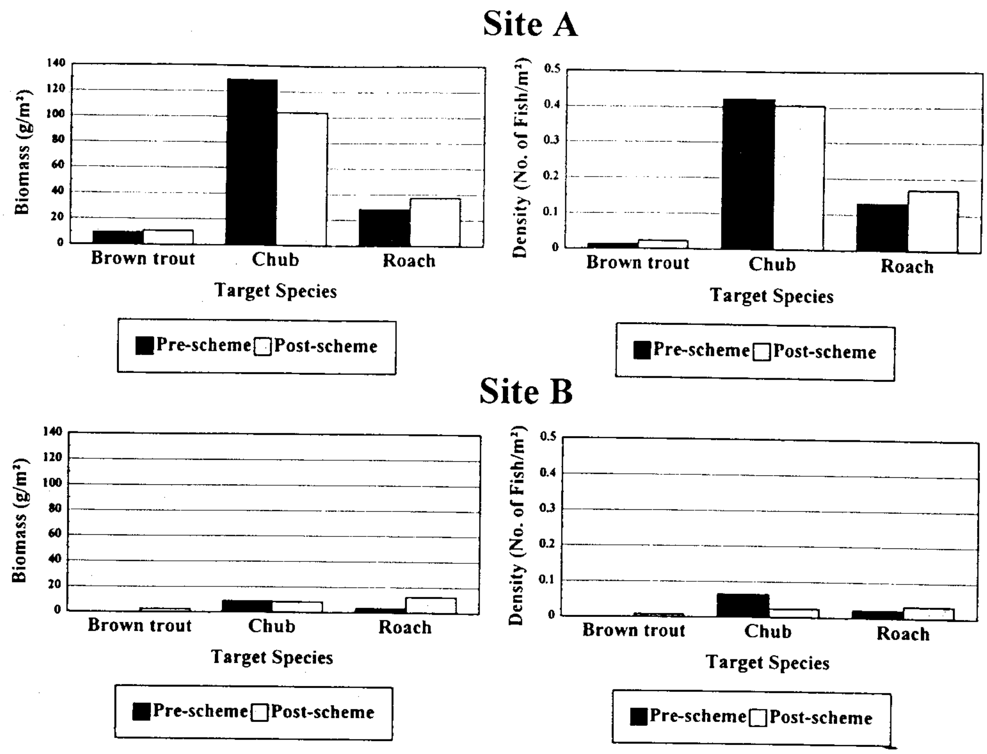


Figure 7: Fish population survey results

Roach (Adult)

Habitat modelling indicates that available habitat has increased under low and moderate flow (Figure 6), within the 0.1-0.7 m^3s^{-1} range. Although the level of habitat has increased dramatically (to approximately double the pre-scheme level at 0.21 m^3s^{-1}), the output indicates that habitat remains comparatively limited under low flow. The fish population data indicate that roach have increased at both sites (Figure 7). The population at site A has improved but remains lower than site B.

DISCUSSION

Physical Habitat Change

Overall, the extent of the changes in the physical habitat parameters considered to be limiting the target species were considered to be poor. Some aspects of improvement were evident but the combined result of the scheme was disappointing and did not achieve identified objectives. Although the model outputs indicate that the scheme has improved the diversity of depths within the channel, especially increasing the availability of pool habitat, the outputs also show that the scheme has been less successful in improving the diversity of flow velocities. In particular, the model output indicates that the general availability of areas of relatively fast-flowing habitat have not increased and that the change from silt and sand substrates to gravel has not occurred as anticipated. These results also demonstrate how PHABSIM may be used to allow an examination of the available physical habitat under simulated flow conditions, providing a useful quantitative assessment of the changes in instream habitat. These outputs are of clear value at this level, without linking the physical habitat to HSIs to assess weighted usable area for specific target species/life stages.

Changes In Habitat Availability And Fish Population

Overall, target species did not respond as planned in the design of the restoration scheme and the objective of significantly improving brown trout and chub carrying capacity was not achieved. The general status of these target species remained the same at both sites with the habitat restoration implemented at site B not improving the status of target species in relation to site A.

The failure to achieve the desired changes in the fish population is considered to be attributable to the restoration scheme not producing the planned changes to limiting habitat factors. Important aspects include the limited success of the techniques utilised to increase flow velocity and diversity and to decrease the amount of finer substrate types. The main reason for this is considered to be the limited extent to which the capacity of the low level channel was reduced.

The results of this initial study were assessed within a year of scheme implementation and it is considered that both habitat and associated fish population response will continue to develop over the coming years. In particular, the degree to which the scheme has restored habitat for re-colonization by desired macrophytes remains an important aspect to be monitored in the future.

The results of the habitat modelling indicate differing levels of agreement with respect to the relationship between WUA and fish population response. The decrease in chub habitat at moderate and high flows was supported by the reduction in the population and the significant increase in roach habitat at low and moderate flows was also supported by the increase in the population. However, the decrease in brown trout habitat across the flow range was not supported by the increase in the population. The consistency of changes in the fish population at sites A and B suggests that factors other than the restoration scheme may also be influencing the fish population.

Any firm conclusions relating to the relationship between habitat availability and target species response cannot be drawn from the results of this initial application due to the limited data. This initial application was undertaken knowing that the habitat suitability criteria were limited and validation of the habitat model with target species response was not an objective. A more robust study would be required to assess this issue and should be an objective of further work developing this approach.

PHABSIM As An Evaluation Tool

Fish habitat restoration projects are commonly evaluated by assessing fish population response in relation to the techniques utilised. This provides important information on the ability of restoration methods to improve target populations. However, it does not provide a direct assessment of the habitat changes produced and as a result does not give information that is critical for the assessment of possible shortcomings in the design or execution of such works. In addition, information on changes to identified limiting factors is essential in order to evaluate the response of target species.

The decision to implement a stream habitat restoration scheme is based on the finding, or accepted assumption, that identified habitat factors are limiting the target population. The nature of the limiting factor(s) will dictate both the restoration and appraisal methodology. This study has demonstrated that in cases where the limiting factors are related to physical habitat (i.e. depth, velocity, substrate, cover), the PHABSIM model can allow the quantitative assessment of habitat change. The method has the benefit of quantifying important aspects of physical habitat change over a range of discharges. The limited availability of suitable physical habitat is commonly an issue in cases where habitat destruction has been caused by channelization.

The use of HSI data to assess spatial and temporal habitat availability offers significant potential in evaluating relationships between habitat change and target species response. The use of time series analysis to assess seasonal habitat availability and the effect of stream restoration techniques in mitigating temporal habitat bottlenecks are important factors that should be considered. However the effective assessment of relationships between habitat availability and target species response are dependant upon robust HSI data. The development of such data remains a key requirement for further progress in the application of PHABSIM in the UK.

Published cases of the use of PHABSIM to evaluate fish habitat restoration are uncommon. Schuler and Nehring (1994) reported the use of the methodology to demonstrate increased habitat availability for brown trout following a stream habitat enhancement project. The study found a correlation between habitat availability and fish density.

The potential use of PHABSIM in the context of evaluating stream habitat restoration is subject to the wider debate surrounding the methodology. The range of criticism relating to the ecological (Scott and Shirvell 1987, Mather *et al.* 1985) and hydraulic (Osborne *et al.* 1988, Hearne *et al.* 1995) aspects of the methodology require careful consideration when assessing each potential application. As the use of hydro-ecological models such as PHABSIM develop there is a clear need to assess the relationship between habitat availability and target species.

A further potential development of the PHABSIM methodology would be to allow predictive modelling of aquatic habitats to assess the changes to physical habitat and WUA at the design stage. Work is currently in progress in the UK to develop the hydraulic modelling procedures within the model in order to facilitate this approach.

ACKNOWLEDGEMENTS

The views expressed in this paper are those of the authors and not necessarily those of their respective agencies. The authors would like to thank the Flood and Coastal Defence Division of the UK Ministry of Agriculture, Fisheries and Foods who provided support for this study under commission FD0505 "Assessment/Design of Habitat Improvement/Restoration Procedures" (Elliott *et al.* 1996). We would also like to thank our colleagues for their

assistance during the course of the work detailed here.

REFERENCES

Brookes, A., K.J. Gregory and F.H. Dawson (1983). An assessment of river channelization in England and Wales. *Science of the total environment.* 27, pp. 97-112.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the Instream Flow Incremental Methodology. *Instream Flow Information Paper No. 21.* U.S. Fish & Wildlife Service Biological Report. 86(7) pp. 235.

Bullock, A., A. Gustard and E.S. Grainger (1991) Instream flow requirements of aquatic ecology in two British rivers. *Institute of Hydrology Report No. 115.* pp. 138.

Elliott, C.R.N., I.M. Gowing, M.J. Dunbar and M.C. Acreman (1996). Ministry of Agriculture, Fisheries and Food Project FD 0505. *Assessment/design of habitat restoration./improvement procedures for flood defence schemes.* Institute of Hydrology report to MAFF. pp 78.

Hearne, J., I.W. Johnson and P.D. Armitage (1994). Determination of ecologically acceptable flows in rivers with seasonal changes in the density of macrophyte. *Regulated Rivers: Research and Management.* 9 pp. 177-184.

Holmes, N.T.H. (1993). River restoration as an integral part of river management in England and Wales. *Contributions to the European Workshop, Ecological Rehabilitation of river floodplains, Arnhem, The Netherlands, 22-24 September 1992.* Report No. 11-6 under the auspices of the CHR/KHR, pp 165-172.

Johnson, I.W., C.R.N. Elliott and A. Gustard (1995). Modelling the effect of groundwater abstraction on salmonid habitat availability in the River Allen, Dorset, England. *Regulated Rivers: Research and Management.* 10 pp. 229-238.

Johnson, I.W., C.R.N. Elliott, A. Gustard, S. Yaqoob, A. Fairhurst, J.H. Blackburn, P.D., Armitage and M. Ladle (1993a). Ministry of Agriculture, Fisheries and Foods Project FD 0503. *Quantitative environmental assessment of river flood defence schemes.* Institute of Hydrology Report to MAFF. pp 70.

Johnson, I.W., C.R.N. Elliott and A. Gustard (1993b). *Data requirements and collection procedures for application of the Instream Flow Incremental Methodology in the UK.* Institute of Hydrology Report to the National Rivers Authority. pp 32.

Johnson, I.W., C.R.N. Elliott, A. Gustard, P.D. Armitage, M. Ladle, F. H. Dawson and W.R.C. Beaumont (1993c). Ecologically Acceptable Flows: Assessment of the Instream Flow Incremental Methodology. Research and Development Project Record 282/1/Wx. National Rivers Authority UK publication pp 182.

Mann, R.H.K. and I.J. Winfield (1992). Restoration of riverine fisheries habitats. *Research and development note 105.* National Rivers Authority UK Publication. pp. 79.

Mathur, D., W.H. Bason, E.J. Purdey and C.A. Silver (1984). A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences.* 42 pp. 825-831.

Osborne, L.L., M.J. Wiley and R.W. Larimore (1988). Assessment of water surface profile model: accuracy of predicted instream fish habitat conditions in low gradient, warm water streams. *Regulated Rivers: Research and Management* 2 pp. 619-631.

Scott, D., and C.S. Shirvell (1987). A critique of the Instream Flow Incremental Methodology and observations of flow determination in New Zealand. In Craig, J.F. and J.B. Kemper (Eds) *Regulated streams advances in ecology.* Plenum Press, New York.

Seber, G.A.F. and E.D. Le Cren (1967). Estimating population parameters from catches large relative to the population. *Journal of Animal Ecology.* 36 pp. 631-643.

Schuler, S.W., and Nehring, R.B. (1994). Using the Physical Habitat Simulation Model to evaluate a stream enhancement project. *Rivers* 4 pp. 175-193.

Ward, D., N.T.H. Holmes and P. Jose (Eds) (1994). *The New Rivers and Wildlife Handbook.* Royal Society for the Protection of Birds. UK. pp. 426.

EFFECTS OF INSTREAM ENHANCEMENT STRUCTURES ON BROWN TROUT HABITAT AVAILABILITY IN A CHANNELIZED BOREAL RIVER: A PHABSIM-APPROACH

Ari Huusko
Finnish Game and Fisheries Research Institute, Kainuu Fisheries Research and Aquaculture
Manamansalontie 90, FIN-88 300 Paltamo
Finland
E-mail ari.huusko@rktl.fi

Timo Yrjänä
North Ostro-Botnia Regional Environment Center
P.O. Box 124, FIN-90101 Oulu
Finland
E-mail timo.yrjana@vyh.fi

ABSTRACT

Stream restoration managers employ instream structures to modify local hydraulic conditions to present preferred microhabitats to fish species, macroinvertebrates, plants and mosses, as well as increasing the retention of organic detritus. However, few studies report flow-related habitat conditions, their spatially distribution and fish habitat availability after habitat structures were installed. More often the post-restoration assessments have focused on fish population responses, usually affected by stocking. We measured stream channel morfology and hydraulic conditions before and after channel modification and boulder structure placements at three representative sites along a channelized boreal river to determine whether more favorable rearing habitat for brown trout was created by these rehabilitation activities. The assessment was performed by Physical Habitat Simulation (PHABSIM) procedure using summer and winter habitat preferences of brown trout for depth, velocity and substrate origin from a nearby trout stream. The results shows that the availability of potential trout habitat can be increased in this river at simulated low and moderate flow conditions by reconsructuring the river bed and placing instream boulder structures. The resulting diversity of depth and velocity conditions created a spatially more complex microhabitat structure. Improved habitat conditions are obviously able to sustain larger trout population compared to pre-restored one. Hydraulic habitat models, like PHABSIM framework, seems to be a suitable procedure to evaluate the benefits of physical habitat enhancement. Among other things the informative graphics output by PHABSIM-computer versions gives a useful tool for e.g. restoration managers informing the public of the success of certain remediation activities.

KEY-WORDS: Channelized river/habitat enhancement/boulder structures/physical habitat model/habitat evaluation/habitat availability/brown trout.

INTRODUCTION

A widely accepted assumption is that hydraulic conditions are one primary templates which govern the distribution and dynamics of stream organisms. Based upon this assumption, stream restoration managers employ instream structures to modify local hydraulic conditions to present preferred microhabitats to fish species, macroinvertebrates, plants and mosses, as well as increasing the retention of organic detritus. Often diverse habitat characteristics can be produced by mimicking the pristine stream conditions as a point of reference. Succesful habitat enhancement must usually re-create hydraulic conditions in all levels of habitat definitions (Newbury and Gaboury, 1993).

Log drives were a significant part of forestry in Finland until 1960's. For this purposes most of the small and medium sized rivers were channelized during 1940's and 1950's to allow the logs to flow rapidly from headstreams to lower reaches. The measures resulted in a situation where riverine habitats were impaired and monotonized, and the diversity of the water velocity, depth, and the substrate particle size were radically reduced, the extensive habitat destruction causing changes in the fish communities affected by the channelization (Jutila, 1985). During the last decade, several extensive projects have been initiated to restore the original habitat structure in the channelized rivers. The main goal of the restoration projects has been to increase the diversity of the riverine environment by reconstructing the substrate in the river channel and embankments (Yrjänä, 1995).

In general, few studies concentrate on reporting flow-related habitat conditions, their spatially distribution and fish habitat availability after habitat structures have been installed (e.g. Shuler, 1993; Harby and Arnekleiv, 1995). More often the post-restoration assessments have focused on fish population responses, usually affected by stocking. However, as knowledge is gained about how different manipulations modify the river channel, and thereby affect fish, more efficient and economical restoration strategies should follow (Rabeni and Jacobsen, 1993). The change in the physical habitat components (e.g. depth, velocity or substrate) varies according to the type of enhancement measures chosen. Thus, success in habitat enhancement presupposes the recognition of the habitat potential reached by different remedial actions so that more reliable guidelines for future habitat modifications and even management of the fisheries may be obtained. The need for objective information conserning the effectiveness of various enhancement measures and also the cost/benefit analysis of the restoration projects was also pointed out by the questionnaire directed to Finnish restoration project managers (Yrjänä and Huusko, 1992). Nonetheless, the physical habitat of river fish has proven difficult to quantify with precision, and, though many methods for habitat characterization abound, the consensus about effective methods for the collection and analysis of relevant information for habitat assessments and comparisons has not been reached. For the purposes of this paper, we chose to use physical habitat simulation (PHABSIM) -procedure (Bovee, 1982; Ginot and Trocherie, 1995) to obtain an instream view to habitat enhancement, and to develop the evaluation of habitat remediation from a 'snap-shot' study based on point measurements of the physical habitat components (Huusko and Yrjänä, 1995) to a more dynamic approach.

STUDY AREA

The river Kutinjoki (E 23°20'; N 65°40') is a third order river in northern Finland running through peatland and coniferous forests, with deciduous trees and bushes growing along the shorelines here and there, as the canopy of the river is mostly open. The River Kutinjoki is a branch of the River Kostonjoki, which is one of the main tributary of the River Iijoki, a river running to the Baltic Sea. The River Kutinjoki is 26 km long with a drainage area of 120 km^2 and the total gradient of it is 0.3 %, the annual mean flow (MQ) being 1.4 m^{3}s^{-1}, whereas the mean summer low flow (MNQ$_{summer}$) is 0.55 m^{3}s^{-1} and the mean high flow (MHQ) is 17.4 m^{3}s^{-1}. The river is ice covered from November to May.

The River Kutinjoki has been dredged for timber floating in 1950's. A majority of the rapids were destroyed and the quality of salmonid reproduction and nursery habitats, covering the total length of 6.6 km, decreased fundamentally. In the beginning of 1990's, before the restoration, the fish species with the highest significance for anglers were pike (*Esox lucius*), perch (*Perca fluviatilis*), grayling (*Thymallus thymallus*) and brown trout (*Salmo trutta*). According to the electrofishing experiments carried out in 1993 (Yrjänä and Seppänen, unpublished data) the most abundant fish species were minnow (*Phoxinus phoxinus*), bullhead (*Cottus gobio*), stone loach (*Noemacheilus barbatulus*), burbot (*Lota lota*), grayling and brown trout. The total fish density varied from 25 to 75 individuals/100 m^2.

Oulu Water and Environment district has been in charge of the restoration of the River Kutinjoki in 1993-1994 mostly for fisheries purposes. Habitat enhancement covered an area of 7.5 hectares of rapids. The habitats in the dredged rapids were improved by utilizing boulders, the cobble and rubble material originally removed from the channel being employed for consruction of boulder dams, deflectors, riffles and other enhancement structures. Reproduction areas for salmonids were increased by rehabilitating natural gravel beds and by consructing artificial spawning grounds from sifted gravel (details in Yrjänä, 1995). A supplemental stocking program for the first five post-restoration years consist of annual stocking of 8 000 young of the year grayling and 1 200 young of the year brown trout, together with additional stockings of older trout during the first post-restoration year.

MATERIAL AND METHODS

Three representative study sites was selected from the dredged river section. Hydrophysical habitat characteristics, including water depth, velocity and substrate were measured at 10-30 points along each of 10-12 transects at every study sites both before and after the restoration during the low discharge in August and September 1993 following guidelines of Bovee (1982) and Malavoi and Souchon (1989). Transects established within each site were selected to represent longitudinal segments of the river having similiar hydrological and morphological conditions. Because of increased diversity of the habitats in the post-restoration state the length of the river section covered by 10-12 transects was shorter in the post-restoration situation than that in the pre-restoration state. The data acquired was used as an input to the EVHA (see Ginot and Trocherie, 1995 and Ginot and Souchon, 1995), a french version of the original PHABSIM-procedure presented by Bovee (1982).

The habitat preference data of three brown trout size classes (4-9 cm; 10-15 cm; over 15 cm long) and two seasons (summer (the water temperatures exceeding +10°C) and winter (the water temperatures below +10°C)) (Mäki-Petäys *et al.*, 1996) were inputted to the EVHA as habitat preference for depth, velocity and substrate for the various size-classes of brown trout. The habitat preference data used here originates from the nearby River Kuusinkijoki, a river having a dense population of brown trout (Huusko, unpublished).

The Kolmigorov-Smirnov two-sample test (K-Sm) and the Wilcoxon-Mann-Whitney test (W-M-W) were used to determine the significancy of differencies occured in the frequency distributions of depth, velocity and the Froude number before and after restoration. The K-Sm-test is sensitive to any kind of difference in the frequency distributions from which two samples were drawn (differencies in location, in dispersion, in skewness etc.) while the W-M-W-test is testing whether two samples represent populations that differ in location (central tendency). If the W-M-W-test didn't react but the K-Sm-test showed a significant difference, then we postulate that the pre- and post-restoration samples differ mostly by the shape or by the dispersion of the distribution. The values for the above comparisons were simulated by the hydraulic model of the EVHA (Ginot and Trocherie, 1995) at the discharges of 0.8 m^3s^{-1}, 2.4 m^3s^{-1} and 4.8 m^3s^{-1}. The habitat values (Weighted Usable Area, WUA, m^2/100 m river reach) at the three study sites for the three brown trout size classes and the two seasons and for the three habitat components (depth, velocity and substrate) and their combination (multiplicative) were processed by the standard procedures available in the EVHA (Ginot and Trocherie, 1995).

RESULTS

The restoration increased the diversity and patchiness of available depths and velocities making the rapids wider and spatially more complex in contrast to the more or less homogenous and simplified pre-restoration flow pattern in a narrow channel (Figure 1). At the study rapids the pre-restorational simple longitudinal profile prevailing before the restoration, with no obstructing boulders or bottom elevations in the channel, the shape of which was a transversal 'U', was replaced by enhancement with complex cross-sectional and longitudinal profiles. The successive pool-boulder dam -structure was, consequently, clearly visible after the restoration.

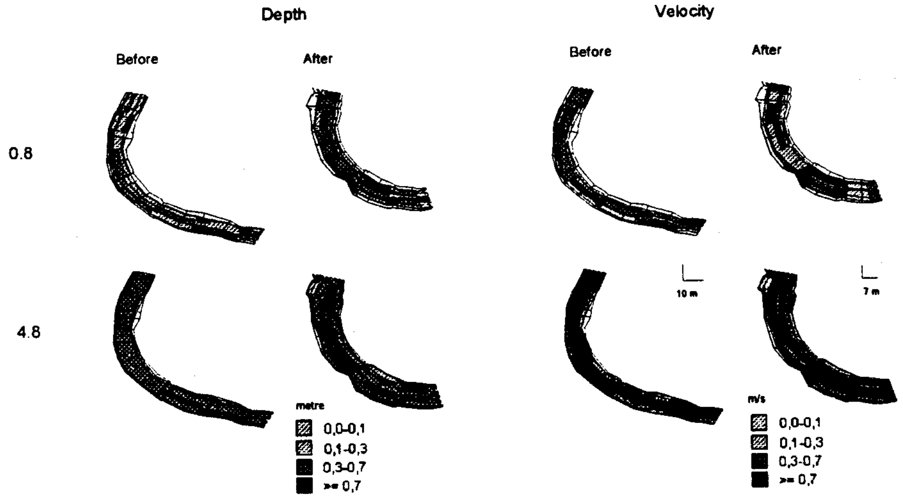


Figure 1. Spatial distribution of depth (metres) and velocity (m/s) cells at study site 1 at two discharges (m^3/s, numbers on the left) before and after habitat enhancement.

At all the study sites and at all the simulated discharges the frequency distributions of water depth, current velocity and the Froude number changed significantly due to the restoration with the exception of site 3 for velocity and the Froude number at the low discharges. The mean pre- and the mean post-enhancement values of the habitat components were quite close to one another, indicating the same location of the distributions (Table 1 and 2). The water depth and velocity median values decreased due to the enhancement at all the study sites and also at all the simulated discharges, but the highest depths and velocities were, nevertheless, almost always found among the post-restoration samples (Table 1). The zone of the velocity and the Froude number distribution values inside the quartile limits (50 % of measured values) was changed into a more narrow one around the mean in the post-restoration situation than had been the case in the pre-restoration state of the river, which indicated significant changes in the shape of the frequency distributions (Table 2). In contrast to this, the shape of the depth distribution and the range of all the distributions were, in most cases, wider in the post-enhancement situation (Table 1 and 2). The only post-restoration range to become narrower was that of the Froude number at high discharges at all the study sites and also at the moderate discharges at the site 2 (Table 1 and 2). However, a majority of the habitat cells had a Froude number indicating sub-critical flow conditions.

The physical habitat simulation modelling indicated that restoration changed particularly the velocity conditions into more favourable ones for all the three size-classes of brown trout both in winter and summer at almost every simulated discharges (Table 3 and 4; Figure 2 and 3). In its pre-restoration state of the river, the area with potentially suitable current velocities was evidently the most limiting habitat component for the trout size-classes I and II during both seasons. The larger trout would have preferred deeper water and coarser substrate than what was available. The post-restoration current velocities were limiting only for the trout size-classes I and II at the highest simulated discharges, whereas the limitation by the availability of suitable depth and substrate became more dominant for all the trout size-classes and the order of importance of these habitat components varied depending on the discharge and the fish size (Table 3 and 4; Figure 2 and 3). In general, the seasonal differences in the potential physical habitat area of trout of different sizes reflected the differences in the seasonal habitat preference curves used in the simulations. A definite lack of the EVHA is that it is not able simulate the effect of the ice cover on the habitat components.

Table 1. **Mean and range of hydraulic habitat characteristics at the study sites before and after enhancement at three discharges simulated by the EVHA.**

DISCHARGE 0.8 m^3s^{-1}	BEFORE Site 1	AFTER Site 1	BEFORE Site 2	AFTER Site 2	BEFORE Site 3	AFTER Site3
Mean depth cm	22	23	23	26	23	23
Range	0 - 45	1 - 63	0 - 81	0 - 86	0 - 58	1 - 74
Mean velocity cms^{-1}	33	27	30	25	27	23
Range	0- 107	0 - 135	0 - 90	0 - 152	0 - 92	0 - 116
Mean Froude number	0.22	0.20	0.20	0.17	0.16	0.16
Range	0.0 - 0.73	0.0 - 1.05	0.0 - 0.74	0.0 - 0.93	0.0 - 0.79	0.0 - 1.12
DISCHARGE 2.4 m^3s^{-1}						
Mean depth cm	35	35	35	38	35	34
Range	0 - 63	3 -78	0 - 99	1 -103	1 - 75	1 - 89
Mean velocity cms^{-1}	61	50	56	42	50	42
Range	0 - 171	0 - 165	0 - 138	0 - 192	0 - 164	0 - 151
Mean Froude number	0.32	0.28	0.29	0.22	0.25	0.23
Range	0.0 - 0.89	0.0 - 0.96	0.0 - 1.03	0.0 - 0.93	0.0 - 0.93	0.0 - 0.96
DISCHARGE 4.8 m^3s^{-1}						
Mean depth cm	48	46	48	51	49	47
Range	1 - 81	1 - 92	1 - 116	2 - 92	2 - 92	1 - 105
Mean velocity cms^{-1}	88	71	82	60	77	62
Range	0 - 215	0 - 193	0 - 206	0 - 240	0 - 220	0 - 220
Mean Froude number	0.40	0.33	0.37	0.27	0.34	0.29
Range	0.0 - 0.99	0.0 - 0.94	0.0 - 1.19	0.0 - 1.02	0.0 - 1.02	0.0 - 0.89

Table 2. **Statistical significance of the difference between the pre- and post-enhancement depth, velocity and Froude number distributions at three simulated discharges. Significance levels for the Kolmigorov-Smirnov test (K-Sm) and Wilcoxon-Mann-Whitney test (W-M-W) are the following: NS $p>0.05$, * $p<0.05$, ** $p<0.01$ and *** $p<0.001$**

DISCHARGE 0.8 m^3s^{-1}	K-Sm Site 1	W-M-W Site 1	K-Sm Site 2	W-M-W Site 2	K-Sm Site 3	W-M-W Site3
Depth	***		**	NS	*	NS
Velocity	***	**	***	*	NS	NS
Froude number	***	**	**	NS	NS	NS
DISCHARGE 2.4 m^3s^{-1}						
Depth	**	NS	***	NS	**	NS
Velocity	***	***	***	***	***	**
Froude number	***	***	***	***	***	**
DISCHARGE 4.8 m^3s^{-1}						
Depth	***	NS	***	NS	NS	NS
Velocity	***	***	***	***	***	***
Froude number	***	***	***	***	***	***

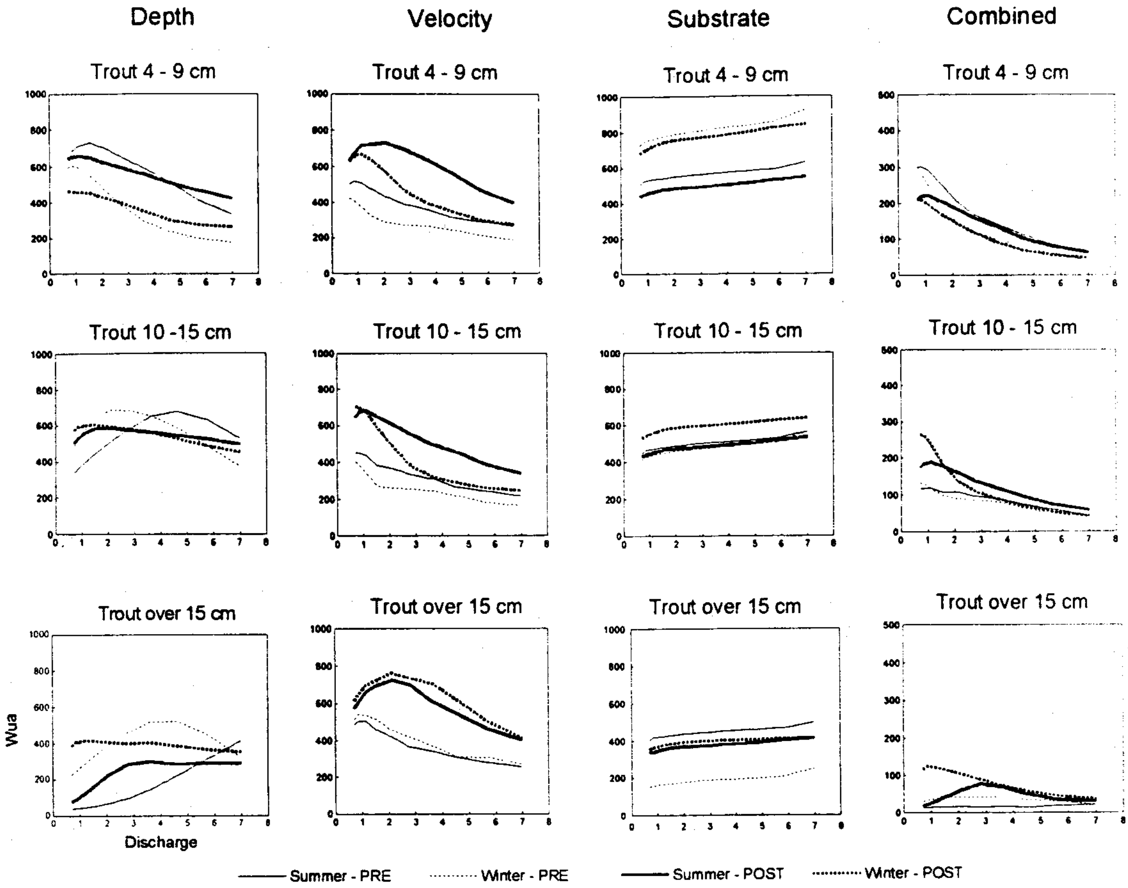


Figure 2. Seasonal weighted usable area, WUA (m^2/100 m river reach) vs. discharge (m^3/s) for depth, velocity, substrate and their combination by brown trout size-classes before (pre) and after (post) habitat enhancement at study site 1. Seasonal view is based on summer and winter habitat preference curves for brown trout size-classes (Mäki-Petäys *et al.*, 1996).

The combined potential physical habitat area (WUA) of the trout size-classes II (10-15 cm) and III (> 15 cm) increased in the enhancement measures at nearly every simulated discharge (Table 3 and 4; Figure 2 and 3). For example, at site 2 the combined potential physical summer habitat area of the size-class II decreased 12 % at the simulated discharge 0.8 m^3s^{-1}, remained unchanged at 2.4 m^3s^{-1} simulated discharge, whereas it increased 13 % at relatively high simulated discharge (4.8 m^3s^{-1}). For the largest trout size-class (> 15 cm) at the study site 2 the increase of the potential physical summer habitat area due to the enhancement was 31 %, 105 % and 100 % at the discharges of 0.8, 2.4 and 4.8 m^3s^{-1}, respectively.

Reciprocally to the changes found in the larger trout size-classes, the potential physical habitat area of the shortest trout size-class (4-9 cm) was smaller in the post-restoration state almost at every simulated discharges between 0.7 - 7.0 m^3s^{-1} and during both seasons (Table 3 and 4; Figure 2). The difference between pre- and post-restoration states was greatest at the low discharges. For example at the site 2 in summer the decrease in the combined potential physical summer habitat area of the trout in the size-class I was 55% at the simulated discharge 0.8 m^3s^{-1}, 45 % at 2.4 m^3s^{-1} and 10 % at 4.8 m^3s^{-1} (Table 3).

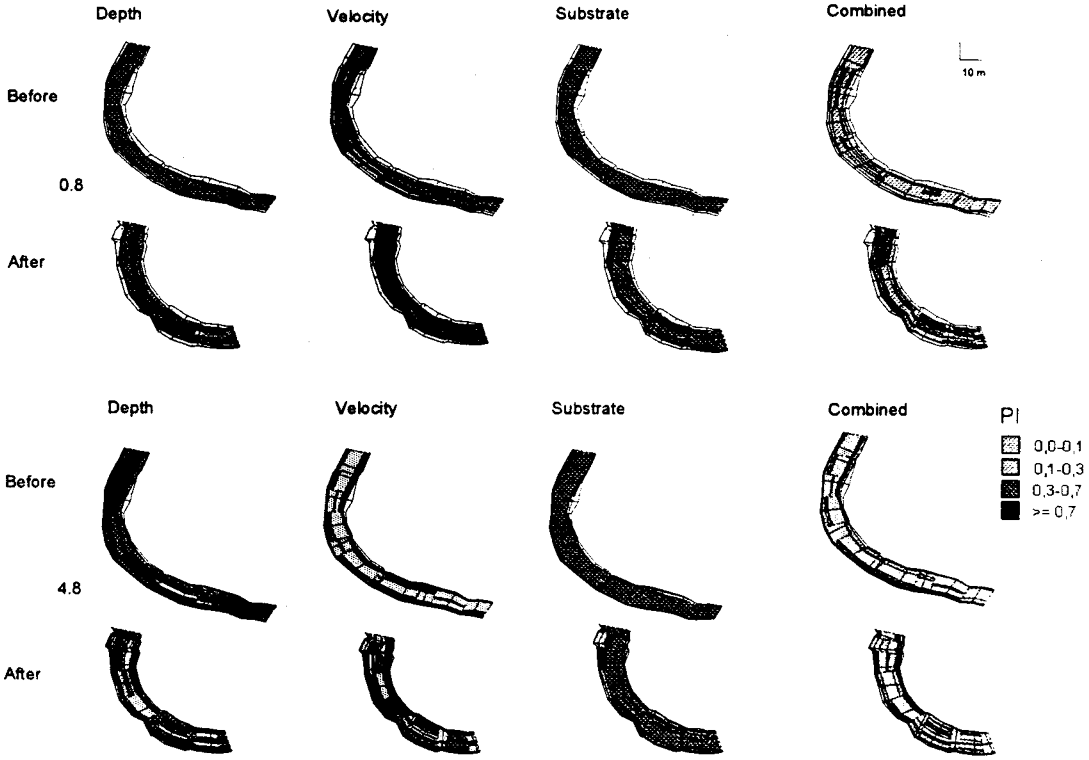


Figure 3. Spatial distribution of preference (PI) for depth, velocity, substrate and their combination by brown trout size-class II (10-15 cm in lenght) in summer and in two discharges (m^3/s, numbers on the left) before and after habitat enhancement at study site 1 as an example.

The physical habitat model simulations showed that before the enhancement the modelled rapids of the River Kutinjoki possessed a great number of suitable habitats for the small brown trout (the size-class I), whereas only a few of the cells were suitable for larger trout size-classes (Table 3 and 4; Figure 2 and 3). For example, at the site 2 at the discharge of 0.8 m^3s^{-1} in summer before the restoration the relation between the relative combined potential physical habitat area for the trout size classes I, II and III was 1: 0.4 : 0.005, respectively, whereas the figures in the post-restored situation were 1: 0.8 : 0.2, respectively. This trend was consistent at almost every study site and simulated discharges indicating that the physical habitat enhancement measures were most effective in increasing the potential physical habitat availability for the trout size-classes II and III.

DISCUSSION

The reconstruction of the river bed and the placement of boulder structures increased the availability of the potential physical trout habitat in the River Kutinjoki at simulated low and moderate discharges. Generally speaking, the remedial measures decreased current velocities, which was one of the main goals of the enhancement plan. On the other hand, the boulder dams, deflectors and other enhancement structures created habitats with high local gradient and fast currents, but these represent only a small area and can obviously be useful at very low flows. The enhancement structures employed certainly offer plenty of interstitial space suitable as hiding place for fishes, although this is not apparent in the modelling results. The resulting diversity of depth, velocity and substrate conditions created a spatially more complex physical microhabitat structure with a wider range of microhabitat quality than the pre-enhancement situation. The enhancement structures obviously provide a greater number of energetically more favourable locations not only for the brown trout but also for the other fish species and macroinvertebrates, thereby improving the chances for succesful enhancement of fish populations. The complex river bed structure, and, on a macrohabitat scale, the diversified channel form, increase the number of instream refugia. This, according to Lancaster and Hildrew (1993), can be very important for the structure and function of benthic invertebrates, especially during spates. However, at high discharges the water in the study rapids begins to crest the boulder-dam structures instead of being diverted by the boulders in the structures. It seems that the velocity refuge provided by the boulder-dam structures diminishes as the discharge increases, resulting to a lower availability of potential suitable habitats.

A homogenous biotope can add the amount of intercohort competion in a brown trout population and thus cause fluctuations in the density in the youngest age groups (Heggenes, 1994). Habitat diversifying can decrease not only the intraspesific competition between the trout year classes but also the interspesific interference. Bugert et al. (1991) report that habitat use by subyearlings of three salmonid species (in sympatry) differed primarily in

Table 3. Weighted usable area (Wua, $m^2/100$ m river reach) for depth, velocity, substrate and their combination by brown trout size-classes (I 4-9 cm, II 10-15 cm and III over 15 cm in lenght, respectively) at study site 2 and at three discharges before and after habitat enhancement. Seasonal view is based on summer and winter habitat preference curves for brown trout (Mäki-Petäys *et al.*, 1996).

SUMMER SITE 2	**BEFORE**				**AFTER**			
Discharge m^3s^{-1}	**Wua comb. $m^2/100$ m**	**Wua depth $m^2/100$ m**	**Wua veloc. $m^2/100$ m**	**Wua subst. $m^2/100$ m**	**Wua comb. $m^2/100$ m**	**Wua depth $m^2/100$ m**	**Wua veloc. $m^2/100$ m**	**Wua subst. $m^2/100$ m**
TROUT I								
0.8	295	700	536	527	129	468	446	313
2.4	220	726	432	568	119	514	543	383
4.8	92	494	239	596	82	468	493	425
TROUT II								
0.8	114	367	466	462	100	329	452	313
2.4	102	522	346	500	100	433	490	385
4.8	65	678	199	526	74	473	424	426
TROUT III								
0.8	16	45	533	423	21	80	415	237
2.4	17	79	412	456	35	139	521	290
4.8	16	215	222	478	32	221	486	322

WINTER SITE 2	**BEFORE**				**AFTER**			
Discharge m^3s^{-1}	**Wua comb. $m^2/100$ m**	**Wua depth $m^2/100$ m**	**Wua veloc. $m^2/100$ m**	**Wua subst. $m^2/100$ m**	**Wua comb. $m^2/100$ m**	**Wua depth $m^2/100$ m**	**Wua veloc. $m^2/100$ m**	**Wua subst. $m^2/100$ m**
TROUT I								
0.8	291	604	423	796	139	361	419	520
2.4	156	489	270	856	112	373	455	646
4.8	53	215	157	901	82	302	356	717
TROUT II								
0.8	138	529	388	493	151	395	464	445
2.4	100	683	222	531	121	486	431	558
4.8	56	596	147	566	99	460	328	617
TROUT III								
0.8	49	253	560	226	74	236	455	336
2.4	55	395	464	245	94	320	553	425
4.8	42	534	261	268	65	239	531	468

Table 4. Weighted usable area (Wua, m^2/100 m river reach) for depth, velocity, substrate and their combination by brown trout size-classes (I 4-9 cm, II 10-15 cm and III over 15 cm in lenght, respectively) at study site 3 and at three discharges before and after habitat enhancement. Seasonal view is based on summer and winter habitat preference curves for brown trout (Mäki-Petäys *et al.*, 1996).

SUMMER SITE 3	BEFORE				AFTER			
Discharge m^3s^{-1}	Wua comb. m^2/100 m	Wua depth m^2/100 m	Wua veloc. m^2/100 m	Wua subst. m^2/100 m	Wua comb. m^2/100 m	Wua depth m^2/100 m	Wua veloc. m^2/100 m	Wua subst. m^2/100 m
TROUT I								
0.8	335	799	606	533	254	753	692	498
2.4	242	714	561	555	281	932	783	548
4.8	99	391	360	574	151	769	682	566
TROUT II								
0.8	152	436	569	473	138	441	697	485
2.4	138	617	435	494	168	648	682	535
4.8	102	666	314	513	126	785	552	554
TROUT III								
0.8	17	50	565	427	15	67	649	373
2.4	28	101	569	444	28	126	757	410
4.8	38	356	356	459	48	264	661	422

WINTER SITE 3	BEFORE				AFTER			
Discharge m^3s^{-1}	Wua comb. m^2/100 m	Wua depth m^2/100 m	Wua veloc. m^2/100 m	Wua subst. m^2/100 m	Wua comb. m^2/100 m	Wua depth m^2/100 m	Wua veloc. m^2/100 m	Wua subst. m^2/100 m
TROUT I								
0.8	374	655	543	837	292	658	671	726
2.4	160	409	381	874	217	700	613	806
4.8	65	145	234	909	96	414	482	833
TROUT II								
0.8	237	643	547	544	185	573	717	526
2.4	141	742	297	575	157	810	561	588
4.8	91	476	316	604	86	823	411	608
TROUT III								
0.8	66	309	603	278	68	302	692	318
2.4	95	492	602	300	88	474	796	359
4.8	61	461	371	322	70	598	721	371

some hydraulic factors (depth and snout position). Enlarged channel depth variation may expand intraspesific and interspesific segregation, resulting to an increase in the cohabitation potential of the species with only small differencies in their physical habitat niche.

The habitat of the shortest size-class of brown trout seems to be tolerant of the dredging of rapids: in the River Kutinjoki 4 - 9 cm trout size-class had plenty of habitats also before the restoration. Chapman and Knutsen (1980) also found out that subyearlings of two trout and salmon species suffered no loss of habitat quality in contrast with the larger trout in the altered sections of small streams in Western Washington, U.S. The ideal habitat for trout should include more areas for larger trout size-classes because of larger territories needed by them and the stronger dominance effect prevailing in their intraspesific hierarchy in comparison with the smaller trout size-classes (Bovee, 1982; Heggenes, 1994; Mäki-Petäys *et al.*, 1996). According to the EVHA simulations carried out in the River Kutinjoki, the habitat areas of the shortest trout size-classes, are, however, essentially larger than those of the large trout. Therefore, a slight reduction in the simulated potential habitat area of the shortest trout size-class may not be harmful, since the potential habitat area of the larger trout size-classes increased at all the simulated discharges and also in proportion to the area for the shortest size-class.

In the cold season the trout adopt a 'shelter-and-move' strategy enabling them to avoid the risks of ice damage by being active at night and sheltering among the substrate during the day (Heggenes *et al.*, 1993). Both the riffle habitat created by the boulder, rubble and cobble structures and the pool habitat between boulder dams are necessary for the brown trout, for in cold water conditions the interstitial spaces within coarse substrate are a favoured form of shelter for the small trout, whereas deep pool habitats are preferred by large trout (Heggenes, 1994). It is therefore essential that stream enhancement programmes in areas with severe winter conditions ensure that appropriately sized (cobble-to-boulder) substrates are abundantly available in trout wintering areas. Apparently this can be reached by the placement of boulder-dam structures. Jutila *et al.* (1994) found good survival and highest densities of stocked brown trout in the areas restored by boulder dams which had several layers of boulders and cobbles, resembling those employed in the River Kutinjoki (see also Shuler, 1993).

During the night time activity period in winter the trout prefer slow-water and coarse substrate habitats (Heggenes, 1994). The winter preference data of Mäki-Petäys *et al.*, (1996) that we used in the present habitat simulations were sampled at daytime and thus characterizes the trout habitat preference during the inactive diurnal resting period. The changes in the hydraulic conditions caused by the ice cover, such as the formation of nonfloating ice in rapids, in particular, create temporally complex depth and velocity distributions (Calkins, 1990), making it really difficult for the hydraulic model to simulate the realized habitat conditions. Because of these uncertainty factors the simulated potential habitat values are not as reliable for the winter as for the summer period. However, in our case the winter habitat simulations can, nevertheless, produce useful information for the designing of habitat enhancement structures. For the enhancement of the streams in the high northern areas an efficient hydraulic modelling of local ice-processes is greatly needed.

The most critical and often most controversial elements of the PHABSIM analysis are the habitat preference criteria used to translate the physical characteristics of rivers into the WUA, an index of habitat availability and quality. Fish distribution and abundance on a local scale are known to be affected by the conditions of depth, velocity, substrate particle size and cover characteristics. Obviously fish do not respond to these variables independently, but rather to the combinations of variables in a hierarchical manner of a certain kind (Heggenes, 1994). The preference curves we chose to use in this study were developed in a study carried out in a nearby stream with a dense population of trout and diverse environment (Mäki-Petäys *et al.*, 1996). The validation of the preference curves is underway (Mäki-Petäys *et al.*, unpublished). From statistical and theoretical perspective, it would certainly have been more accurate to use site spesific preference curves, for the curves of this kind are able to reflect the ecological influences of the local microhabitat choices (including the responses for food availability, predation, competition etc; e.g. Orth, 1995). Since, however, our goal was only to see how the remedial actions will modify the physical environment from the trout perspective, the choice may be justified. A totally different question is whether this increased potential physical habitat availability will ever be achieved as the realized spatial habitat for the trout. The benefits of an increased suitable living space in the restored areas, especially in such a seasonally variable environment as a boreal river, are linked with the development of the biological complexity (aquatic vegetation, macroinvertebrates) and its temporal and spatial dynamics (e.g. Heggenes, 1994; Orth, 1995). Yet an assessment by a physical habitat simulation alone cannot capture these potential effects, and a combined analysis at multiple trophic levels is necessary (e.g. Orth, 1995; Wasson *et al.*, 1995). Mäki-Petäys *et al.* (1996) suggest that the minimum requirement for increasing the biological realism of instream habitat models is that the size structure of the fish population under study and the seasonality of resource availability and use be incorporated into the models. In streams of the far north the harsh winter conditions are far more likely to be critical for salmonid survival and production, thus emphasizing the importance of developing separate preference curves for winter conditions in these environments. Nevertheless, technical and theoretical concerns about the accuracy of habitat preference criteria (e.g. Mathur *et al.*, 1985; Gan and McMahon, 1990) reinforce the need to validate and verify models before using them for estimating the realized trout responses to different habitat enhancement measures. We have launched a study program to examine trout population responses and position choice in the restored areas.

Fishery managers rely heavily on the information gained from post-treatment evaluations for guidance in other enhancement efforts, but the majority of these evaluations have focused on population responses (e.g. Jutila *et al.*, 1994). Although studies at this level provide valuable information on how different species and lifestages respond to habitat enhancement, they tell only a little of the real reasons for success or setback in those fish responses. Quantitative estimates of the ecological impacts of river restoration require a description of the hydraulic local habitat. A quantitative tool, such as a PHABSIM framework (the EVHA and others (e.g. Payne, 1994)) combining habitat preferences with the appropriate physical habitat variables can be used to assess functional responses of the fish to the different habitat modifications (e.g. Shuler, 1993). Such ecological-engineering capabilities allow comparisons of structure types under different channel and hydraulic conditions and discharge regimes. An understanding of how enhancement structures modify the physical environment are essential for the design of enhancement treatments. On the other hand, the informative graphics of the output by the present PHABSIM computer versions give a useful tool for e.g. restoration managers informing the public of the success of certain remediation activities. PHABSIM framework can also be used to compare the potential habitat gains of various structure types during the design phase by changing the cross-sectional information to imitate hydraulic and structural changes (Harby and Arnekleiv, 1994), but the process seems laborous due to the difficulties in modelling local hydraulic parameters by the present hydrological modelling components. For planning purposes improvement could be gained from the development of 3-dimensional hydrological modelling component (Olsen and Alfredsen, 1994). We, however, share the view held by Shuler (1993) that, as with any model, the predictive accuracy and realism of the physical habitat model output is limited by the user's understanding of the underlying assumptions, by his or her willingness to recognize limitations inherent in the model and the degree to which the user incorporates general biological knowledge and expertise into the modelling and decision making process.

ACKNOWLEDGEMENTS

Ms Päivi Timonen, B. Sc., is greatly acknowledged for valuable assistance she offered in the analysis of the data. Without the help of Ms Eeva-Kaisu Kemppainen, an Official Translator in the English language, the language of this paper would have possessed a definite Finnish accent.

REFERENCES

Bovee, K. (1982). A guide to stream habitat analysis using the Instream Incremental Flow Methodology. *Instream Flow Information Paper.* 12, U. S. Wildlife Service. FWS/OBS-82/26.

Bugert, R. M., Bjornn, T. C. and Meehan, W. R. (1991). Summer habitat use by young salmonids and their responses to cover and predators in a small Southeast Alaska stream. *Transactions of the American Fisheries Society.* 120 pp. 474-485.

Calkins, D. J. (1990). Winter habitats of atlantic salmon and brook trout in a small ice-covered streams. *Proceedings of the 10th Symposium of Ice.* Vol. 3 pp. 113-126. IAHR.

Chapman, D. W. and Knudsen, E. (1980). Channelization and liverstock impacts on salmonid habitat and biomass in Western Washington. *Transactions of the American Fisheries Society.* 109 pp. 357-363.

Gan, K. and McMahon, T. (1990). Variability of results from the use of PHABSIM estimating habitat area. *Regulated Rivers: Research and Management.* 5 pp. 233-239.

Ginot, V. and Souchon, Y. (1995). *EVHA, version 1.0. Un logiciel d'evaluation de l'habitat physique du poisson en riviere. Guide de methodologique.* CEMAGREF-LYON, 3 bis quai Chauveau, 69336 Lyon Cedex 09, France. In French.

Ginot, V. and Trocherie, F. (1995). *EVHA, version 1.0. Un logiciel d'evaluation de l'habitat physique du poisson en riviere. Guide de l'utilisateur.* CEMAGREF-LYON, 3 bis quai Chauveau, 69336 Lyon Cedex 09, France. In French.

Harby, A. and Arnekleiv, J. V. (1994). Biotope improvement analysis in the river Dalåa with the River System Simulator. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* The Norwegian Institute of Technology, Trondheim. pp. 513-520.

Heggenes, J. (1994). Physical habitat selection by brown trout (*Salmo trutta*) and young Atlantic salmon (*Salmo salar*) in spatially and temporally heterogenous streams: implications on habitat hydraulics. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* The Norwegian Institute of Technology, Trondheim. pp. 12-30.

Heggenes, J., Krog, O. M. W., Lindås, O. R., Dokk, J. G. and Bremnes, T. (1993). Homeostatic behavioural responses in a changing environment: brown trout (*Salmo trutta*) become nocturnal during winter. *Journal of Animal Ecology.* 62 pp. 295-308.

Huusko and Yrjänä, (1995). Evaluating habitat restoration of rivers channelized for log transport: a case study of the river Kutinjoki, northern Finland. *Bulletin Francais de la Peche et de la Pisciculture.* 337/338/339 pp. 407-413.

Jutila, E. (1985). Dredging of rapids for timber floating in Finland and its effects on river spawning fish stocks. In: *Habitat modification and freshwater fisheries*, edited by Alabaster, J. S.. London, Butterworths. pp. 102-108.

Jutila, E., Karttunen, V. and Niemitalo, V. (1994). Better one stone in the rapid than ten on the bank - influence of various restoring methods on the parr densities of brown trout in the rapids of the tributaries flowing into the Iijoki River. *Kalatutkimuksia - Fiskeriundersökningar* 87. 29 pp. In Finnish with English summary.

Lancaster, J. and Hildrew, A. G. (1993). Characterizing instream flow refugia. *Canadian Journal of Fisheries and Aquatic Sciences. 50 pp. 1663-1675.*

Malavoi, J. R. and Souchon, Y. (1989). Methodologie de description et quantification des variables morphodynamiques d'un cours d'eau a fond caillouteux: example d'une station sur la Filliere (Haute-savoie). *Revue de Geographie de Lyon.* 64 pp. 252-259. In French.

Mathur, D., Bason, E. J., Purdy, Jr. and Silver, C. A. (1985). A critique of the Instream Flow Incremental Methodology. *Canadian Journal of Fisheries and Aquatic Sciences.* 42 pp. 825-831.

Mäki-Petäys, A., Muotka, T., Huusko, A., Tikkanen, P. and Kreivi, P. (1996). Seasonal changes in microhabitat use and preference by juvenile brown trout *Salmo trutta* in a northern boreal river. (submitted *manuscript*)

Newbury, R. and Gaboury, M. (1993). Exploration and rehabilitation of hydraulic habitats in streams using principles of fluvial behaviour. *Freshwater Biology.* 29 pp. 195-210.

Olsen, N. R. B. and Alfredsen, K. T. (1994). A three-dimensional numerical model for calculation of hydraulic conditions for fish habitat. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* The Norwegian Institute of Technology, Trondheim. pp. 113-123.

Orth, D. J. (1995). Food web influences on fish population responses to instream flow. *Bulletin Francais de la Peche et de la Pisciculture.* 337/338/339 pp. 317-328.

Payne, T. R. (1994). RHABSIM: user-friendly computer model to calculate river hydraulics and aquatic habitat. *Proceedings of the 1st International Symposium on Habitat Hydraulics.* The Norwegian Institute of Technology, Trondheim. pp. 254-260.

Rabeni, C. F. and Jacobsen, R. B. (1993). The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Frehswater Biology.* 29 pp. 211-230.

Shuler, S. W. (1993). Trout responses to boulder habitat structures in the Rio Grande River, Colorado, U. S. *Thesis for Master of Science.* Colorado State University. Fort Collins. Colorado. U.S. 95 pp.

Wasson, J. G., Bonnard, R. and Maridet, L. (1995). Global influence of physical habitat parameters on benthic invertebrates in trout streams: towards an integration in fish/habitat models. *Bulletin Francais de la Peche et de la Pisciculture.* 337/338/339 pp. 291-302. In French.

Yrjänä, T. (1995). Restoration of running waters within the Iijoki river area. In: *Rehabilitation of rivers*, edited by DeWaal, l., Large, A.R.G. and Wade, M.. Wiley Publication. London. (accepted manuscript).

Yrjänä, T. and Huusko, A. (1992). Enhancement of fish habitats in Finnish rivers. Suomen Kalastuslehti. 99(6) pp. 26-27. In Finnish.

Ecohydraulics of Spur Dikes in Ushizu River, Kyushu Region, Japan

Nobuyuki TAMAI, Yoshihisa KAWAHARA
Department of Civil Engineering, the University of Tokyo
Hongo, Bunkyo-ku, Tokyo 113, JAPAN
kawahara@hydra.t.u-tokyo.ac.jp

Masakazu AOKI, Aoki MATUMOTO
Asia Consulting Co., Ltd.
Tamura-cho, Atsugi-shi, Kanagawa Prefecture 243, JAPAN

Yutaka ISHIKAWA
Takeo Construction Office, Kyushu Regional Construction Bureau, Ministry of Construction
Takeo-cho, Takeo-shi, Saga Prefecture 843, JAPAN

Minoru YASUDA
Technology Research Center for River Front Development
Izumikan Sanbancho Bldg. Sanbancho 3-8, Chiyoda-ku, JAPAN

ABSTRACT

The necessity for river improvement works to provide a variety of suitable habitats have been well recognized in recent years. Spur dikes, which have been designed to protect bank from erosion and to control bed topography, have received much attention from the standpoint of fish habitats. To trace the changes in habitats after the construction of spur dikes, we carried out field measurement of bed level evolution, distributions of velocity and water depth, fish species in Ushizu river, Kushu region of Japan. The field observation continued from 1994 to 1996 covering the pre- and post-construction periods of the spur dikes. We found that the introduction of spur dikes made of stone riprap have altered the river bed from flat one to undulated one with local sours and riffles and that several fish species have been identified there, which implies the improvement of fish habitats. The characteristics of physical habitat at low and high water have been studied through field measurement and numerical simulation using two- and three- dimensional flow models. It was confirmed that the models can reproduce the velocity distributions at low water and at flood. Finally the role of the spur dikes in restoration of fish habitat in Ushizu river was summarized.

KEY-WORDS : Spur dike/Fish habitat/Velocity distribution/Water depth/Sediment transport/
Pool/Riffle/Field measurement/Numerical simulation.

1. INTRODUCTION

The restoration of rivers to an ecological system of the environment has become one of the most important tasks in many countries. The conventional river improvement works have often lost the habitat requirements for migration, reproduction, juvenile rearing, shelter and so on. Restoration efforts for riverine fish need the information and knowledge of physical and biological processes in river system.

Spur dikes, which have been constructed to protect bank from erosion and to stabilize bed topography, have recently received much attention from the standpoint of fish habitats. Velocity distribution and bed scour pattern associated with various arrangements of spur dikes have been studied over several decades (for example, Sanada, 1932; Yamamoto, 1995 in Japan). The design methods for suitable spur dikes have also been developed by many researchers (Kawamura, 1963; Klingeman, 1984). It has been reported that complicated bed profiles are the effective natural means to provide a wide range of flow conditions and suitable habitats for fish (Tamai,1993; Shields, 1995). Thus the construction of spur dikes in alluvial rivers, which leads to considerable change in bed topography, has the potential to realize its original objectives and the suitable habitats for fish at the same time. However, we need more reliable information on the impact of the spur dikes on fish habitats, which is the main topic we are pursuing in this paper.

The goal of this research is to develop a design method of spur dikes fully accounting the sediment transport for local scours, pools and riffles for fish habitats. In the present study, we aimed at the better understanding of the processes how the stream characteristics and fish species change. For such purpose, we carried out 1) field observation of sediment scour and deposition in the river reach with spur dikes, 2) field measurement of the flow pattern at low water and at flood, 3) fish capturing, 4) numerical simulation to understand the characteristics of flow and river bed and 5) identification of the role of spur dikes in improvement of fish habitats.

2. RESEARCH SITE

2.1 Location and Characteristics of Ushizu River

Figure 1 shows the location of Ushizu river, in Saga Prefecture, Japan. It is a typical meandering river on alluvium and the circle in Figure 1 represents the location of the observation site. Table 1 explains the physical characteristics of the Ushizu river. Figure 2 shows the view of the observation reach where the river width is about 50m together with the spur dikes constructed along the concave bank. The observation site locates in the tidally affected reach.

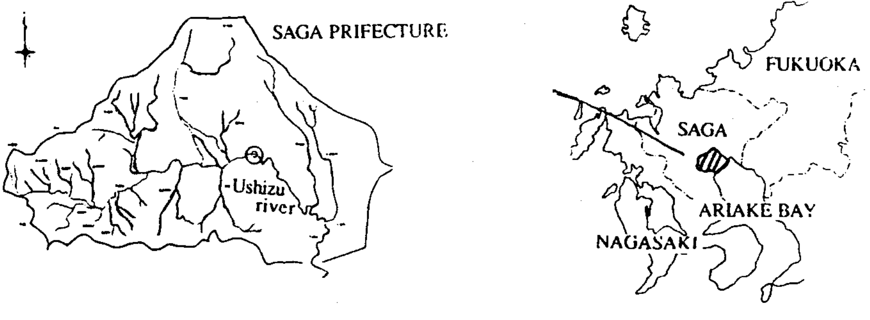


Figure 1 Location of Ushizu River.

Table 1 Characteristics of Ushizu River.

parameter		value
drainage area		165.5km^2
channel length		15.3km
mean radius of curvature		176m
bed material	d_{10}, d_{50}, d_{60}	0.002~0.51, 0.011~0.65, 0.018~0.77mm
bed slope		1/700
discharge	flood discharge	637.27m^3/s
	average discharge	0.4m^3/s
channel depth	flood	max=8m
	low flow	max=1m

Figure 2 View of Ushizu River and Spur Dikes.

2.2. Spur Dikes of Ushizu River

The spur dikes in Ushizu river belong to an impermeable riprapped rockfill type and were constructed at right angles to stream. Figure 3 shows the typical cross-sectional shape and its dimensions in cm unit.

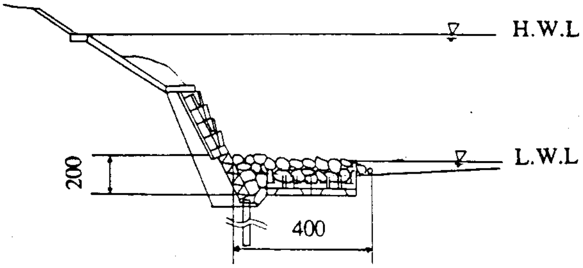


Figure 3 Spur Dikes in Ushizu River.

3. FIELD OBSERVATION

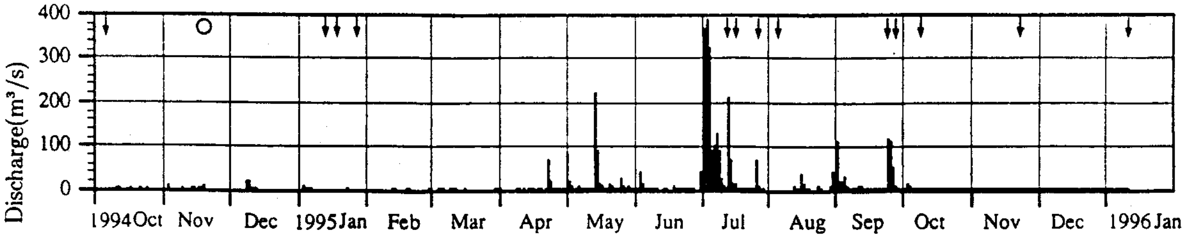
3.1 Data Acquisition Methods and Observation Dates

Discharge has been monitored continuously by the Takeo Construction Office, Ministry of Construction. Velocity distribution was measured in the reach with spur dikes using two ways, i.e., video image analysis of surface flow at low water and at flood and electro-magnetic velocimetry at low water. Water stage was measured using the standard surveying technique. Detailed geometry data of nine cross-sections at about 20m intervals were combined with the bed level data obtained at randomly distributed points to draw the precise contour lines of the bed topography. Fish capturing, diving and marking were carried out for fish habitats. Table 2 summarizes the measured quantities and the frequencies of each measurement. Figure 4 shows the daily discharge records during the course of our field measurement.

Table 2 Observation Dates and Collected Data.

year	1994	1995											1996
date	10/17	1/14	1/17	1/25	7/11	7/12	7/25	8/4	9/24	9/25	10/8	11/23	1/11
flow condition	L.W.	L.W.	L.W.	L.W.	H.W.	H.W.	L.W.	L.W.	L.W.	H.W.	L.W.	L.W.	L.W.
Velocity			◎		◎	◎			◎	◎			◎
Water stage			◎		◎	◎				◎			◎
Bed materials								◎					
Geometry	◎	◎					◎				◎		◎
Fish				◎				◎				◎	
Discharge	every hour using stage-discharge relation												

(L.W.:low water level at moderate flow, H.W.:high water level at flood flow)



(○: spur dikes completed, ↓: date of the observation)

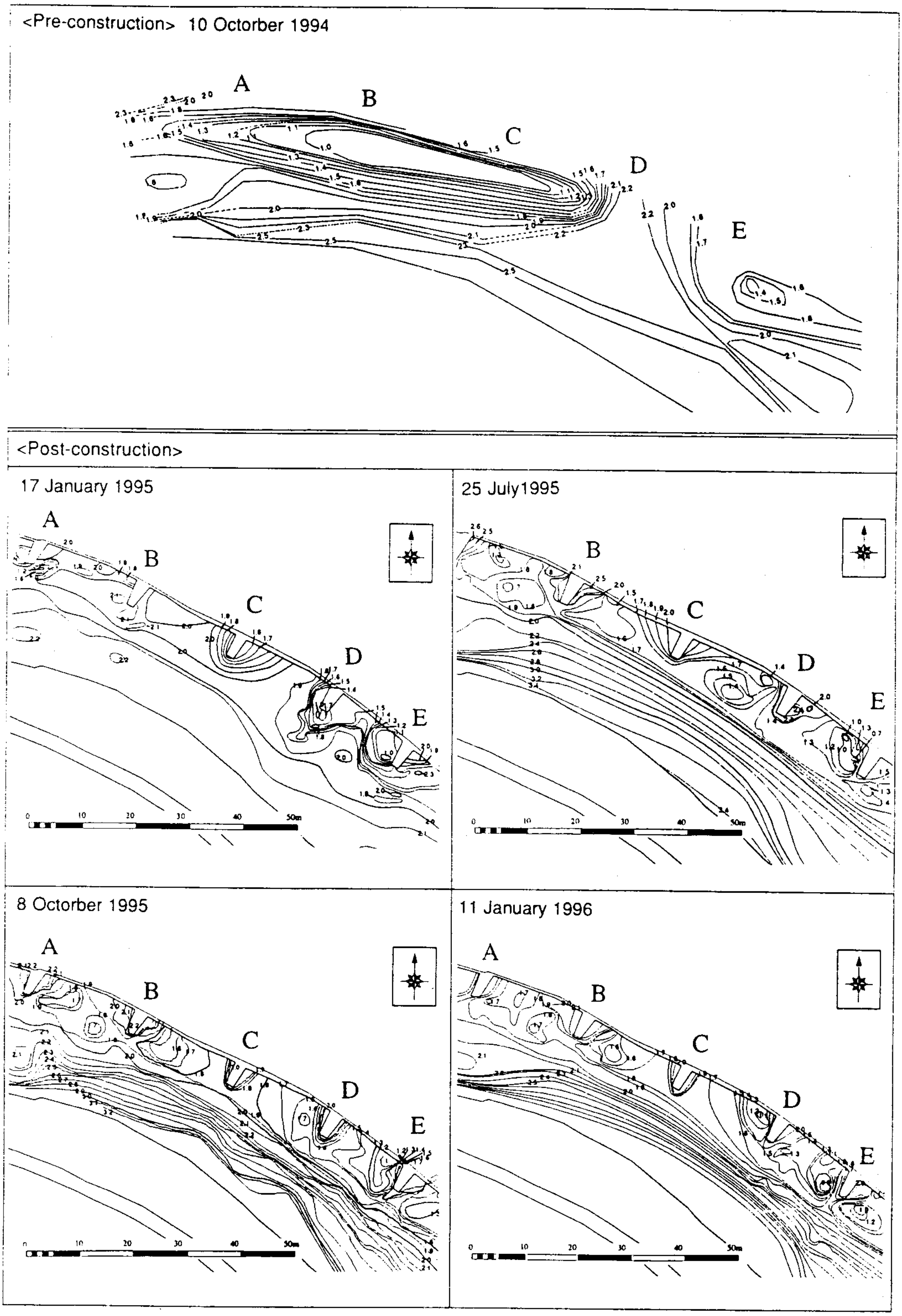
Figure 4 Variation in Daily Averaged Discharge during Observation Periods.

3.2 Result of Field Study

3.2.1 Bed Profiles

Main objective of our field investigation was to understand the effects of spur dikes on the stream flow and the bed topography and to get insight into their effects on fish habitats. Another objective was to obtain the detailed data which can be used to validate the numerical model we develop. Figure 5 compares the contour maps of the river bed observed at five different times. There existed a deep pool along the outer bank before the construction of the spur dikes. Due to the construction work and fine sediment transport, the data obtained on January 17th shows the complex bed shape near the spur dikes. The contour map of July 25th, which was measured after floods, shows the local sours around the spur dikes. The layout of the scoured area showed little variation after the flood of medium scale in September, 1995. The basic structure of the bed topography seems to be relatively stable.

Bed material is an important indicator of aquatic habitat quality. Clay and silt were scoured away and only sands and gravels were observed in the deep pools around the dikes. Clay and silt were found in the limited



<Pre-construction> 10 Octorber 1994

<Post-construction>

17 January 1995

25 July1995

8 Octorber 1995

11 January 1996

Figure 5 Change in Bed Configuration.

region near the heel of spur dikes. The rapid riffles did not include clay and silt. The properties of the bed materials at several locations are listed in Table 3.

3.2.2 Velocity Distribution and Local Scour near Spur Dikes

Figure 6 demonstrates the measured velocity distributions at low water around the dikes A, B, D and E in Figure 5. The figure (a) shows the results obtained by the video image analysis of surface flow, the figure (b) the time averaged velocity vectors measured by electro-magnetic velocimetry at the depth of 0.6 times water depth from water surface. From the figure, it is apparent that the spur dikes directly influence flow pattern. The tracking of the floats on water surface indicates such types of instantaneous stream patterns as (1) deviation away from the bank towards the center of the channel, (2) formation of large scale of vortices between spur dikes and (3) flowing toward the downstream spur dike and then deflected. These flow patterns are correlated with the bed profiles.

3.2.3 Fish Capturing

The creation of deep water zones by local scours and slack water areas by sedimentation between dikes is favorite for fish habitats. Identified fishes were three classes nine species in the winter season, three classes thirteen species in the summer season and two classes seven species in the autumn season, which is classified in Table 4. In terms of riverine habitats, they are classified into two groups. The first group includes Pseudogobio esocinus and Cobitis matubarai which like clean water and riffle areas. Their juveniles need the safety spaces. Riprap revetments of spur dikes, gravel and cobble provide holes for their escape. The other group includes Carassius cuvieri, Carassius gibelio largsdorfi, Pseudorasbora parva which were observed on the bed with clay and silt. They like waste water and stagnation pool. Diversity of physical condition created by spur dikes is considered to be essential to support wide variety in fish species.

4. NUMERICAL ANALYSIS

4.1 Two-Dimensional Flow Model

A two-dimensional model based on shallow water equations were applied to the stream at low water. The basic equations are written as;

- Continuity equation

(1)
$$\frac{\partial hU}{\partial x} + \frac{\partial hV}{\partial y} = 0$$

- Momentum equations

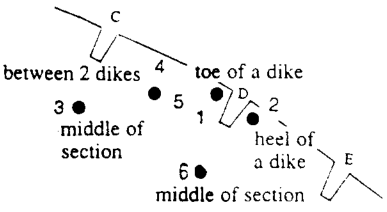
(2)
$$\frac{\partial hU^2}{\partial x} + \frac{\partial hUV}{\partial y} = -gh\frac{\partial z_b}{\partial x} - \frac{\tau_{bx}}{\rho} + h\nu_e\left(\frac{\partial^2 U}{\partial x^2} + \frac{\partial^2 U}{\partial y^2}\right)$$

(3)
$$\frac{\partial hUV}{\partial x} + \frac{\partial hV^2}{\partial y} = -gh\frac{\partial z_b}{\partial y} - \frac{\tau_{by}}{\rho} + h\nu_e\left(\frac{\partial^2 V}{\partial x^2} + \frac{\partial^2 V}{\partial y^2}\right)$$

where the depth averaged velocity components are written as U, V, the water depth h, the bed shear stresses τ_{bx}, τ_{by}, the bed level z_b and the effective diffusion coefficient ν_e. The basic equations are discretized by the finite element method. Figure 7 shows the velocity distributions in a part of the flow domain. Comparison between the observed data and the calculated results indicates that the present model is reliable in reproducing the flow feature around the spur dikes satisfactory under low water conditions.

Table 3 Bed Materials.

sampling point No.		1. toe of a dike	2. heel of a dike	3. middle of a section	4. between 2 dikes	5. between 2 dikes	6. middle of section
grain size	d_{70} (cm)	0.72	0.21	0.91	2.9	0.02	1.7
	d_{50} (cm)	0.49	0.11	0.65	1.8	0.01	0.93
	d_{30} (cm)	0.38	0.06	0.46	1.0	0.006	0.64
	gravel(%)	10	6	12	45	12	21
	sand (%)	89	59	86	55	13	78
	silt (%)	1	30	1		56	1
	clay (%)		5	1		20	



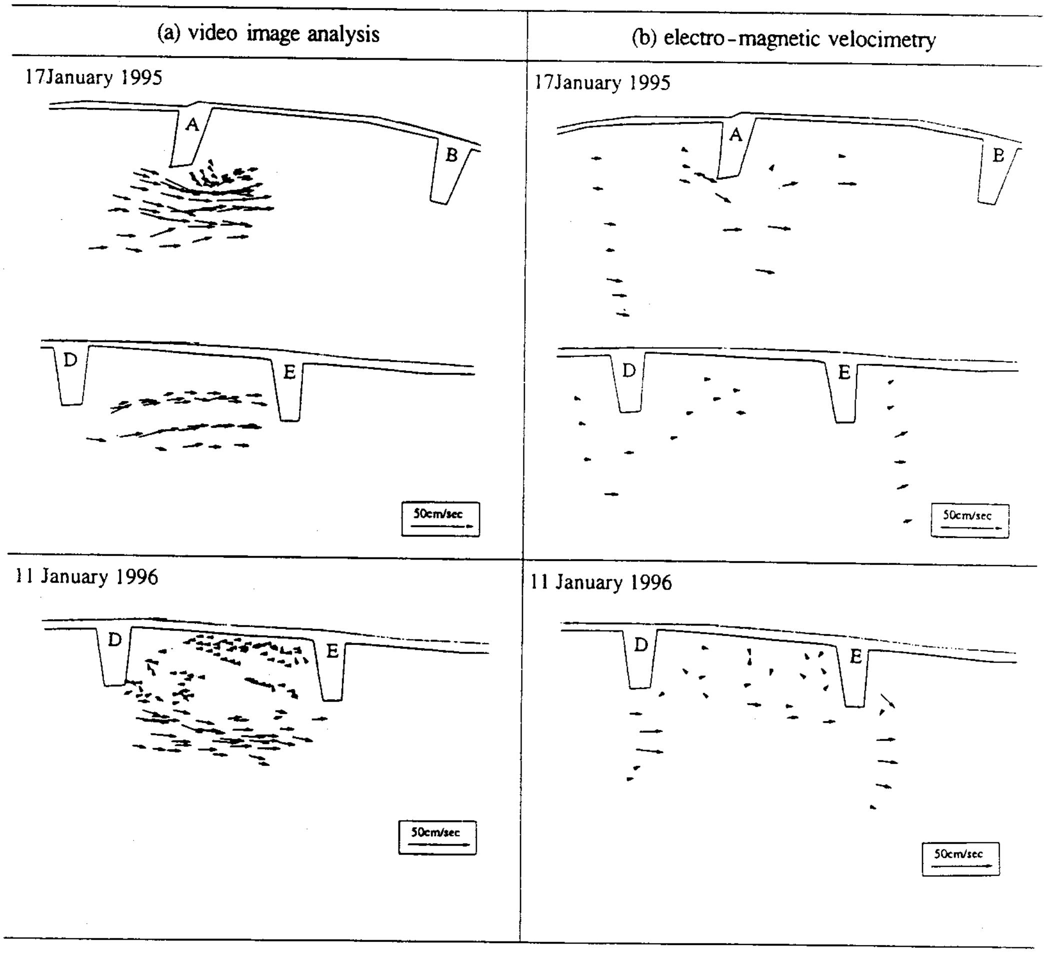
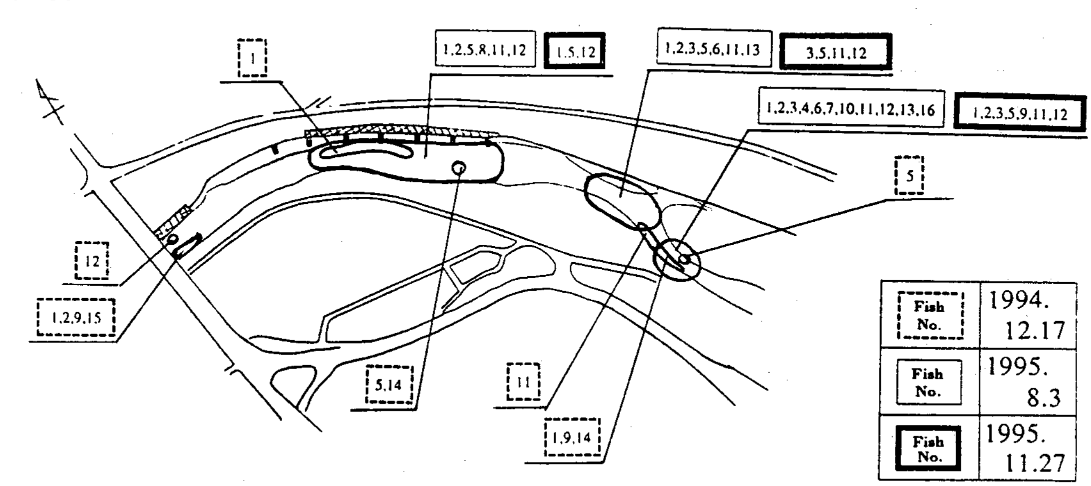


Figure 6 Velocity Distributions around Spur Dikes.

Table 4 Location of Fish Capturing and Identified Fish Species.

sampling points



		fish no.	winter season, 25 Jan 1995				summer season, 4 Aug 1995				Autumn season 23 Nov 1995			
			no.	size			no.	size			no.	size		
				min	mean	max		min	mean	max		min	mean	max
Cyprinidae	Carassius cuvieri	1	11	119	139	161	16	111	139	167	4	101	118	159
	Carassius gibelio langsdorfi	2	5	111	124	154	10	103	121	138	1		127	
	Tanakia lanceolata	3					27	55	72	89	9	36	46	53
	Acheilognathus rhombeus	4					1		81					
	Zacco platypus	5	7	34	40	62	19	35	80	125	21	32	72	124
	Zacco temmincki	6					2	112	136	160				
	Opsariichthys uncirostris	7					1		100					
	Pseudorasbora parva	8												
	Pseudogobio esocinus	9	2	51	63	75					2	32	55	78
Cobitididae	Cobitis matsubarai	10					1		89					
Gobiidae	Tridentiger trigonocephalus	11	2	40	45	50	2	24	27	30	3	33	38	41
	Tridentiger kuroiwae brevispinis	12	1		59		2	63	66	69	14	29	39	54
	Acanthogobius flavimanus	13					10	85	93	101				
	Glossogobius olivaceus	14	3	19	29	34								
	Acanthogobius lactipes	15	1		37									
Cottidae	trachidermus fasciatus	16	1		77									

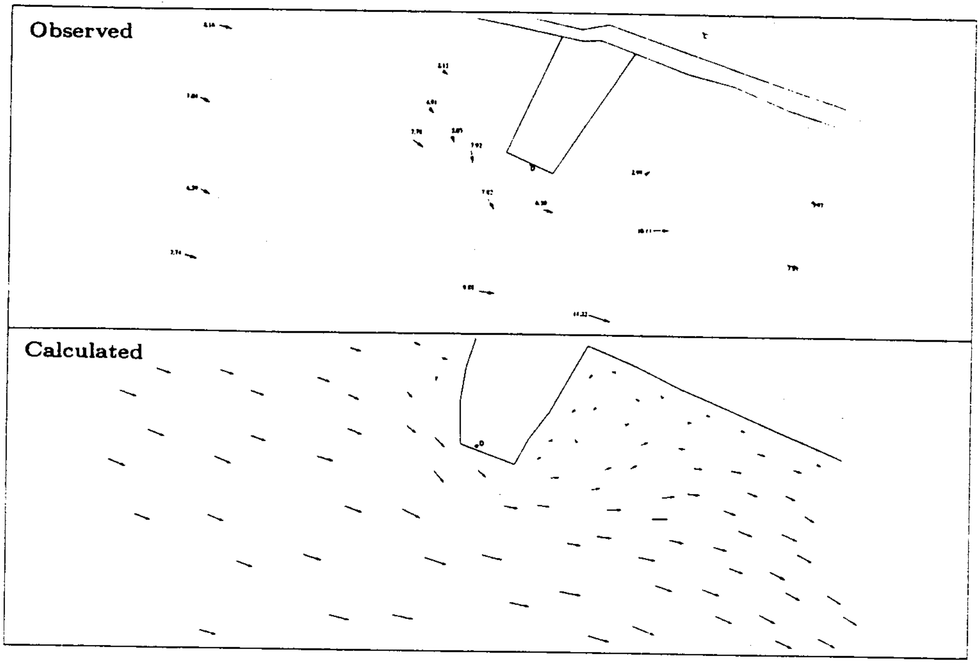


Figure 7 Velocity Distribution by a 2-D Model against Measurement.

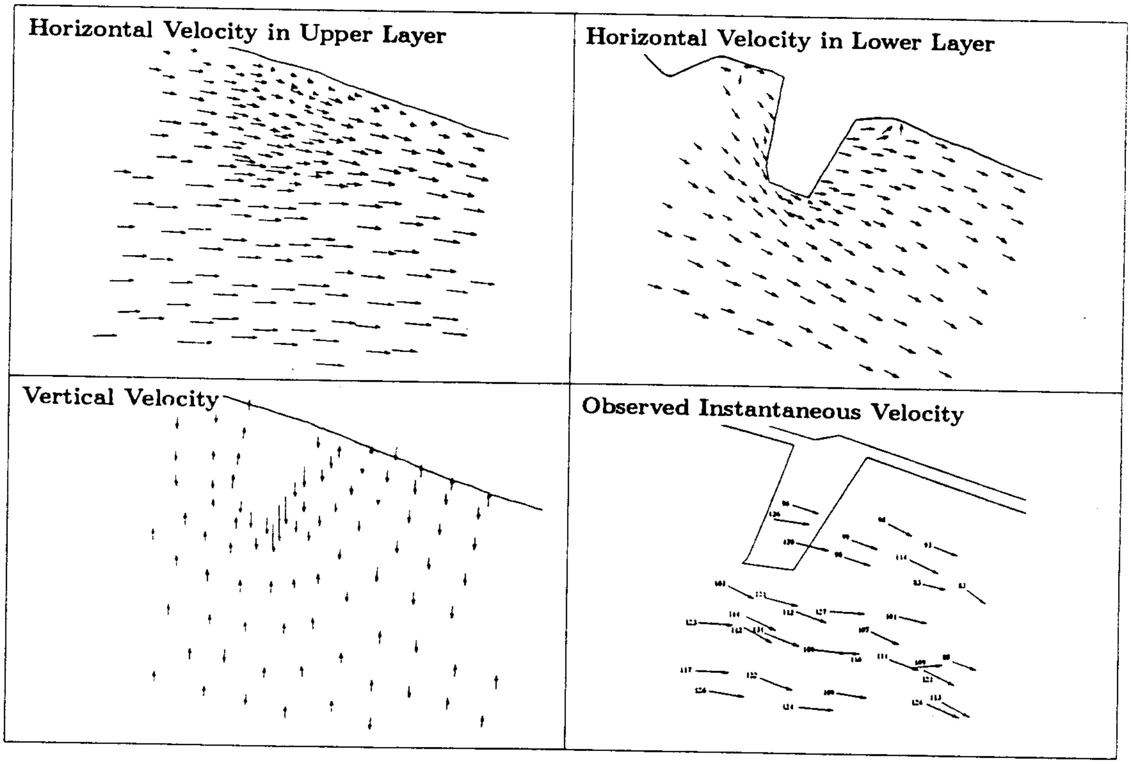


Figure 8 Velocity Distribution by a Two-Layer Model against Measurement.

4.2 Two-Layer Flow Model

A two-layer model was developed as the extension of the two-dimensional model explained above to account for the vertical motion around spur dikes during floods. The flood flow was divided into two layers, i.e., upper layer and lower layer. The upper layer was defined as the flow layer above the height of the spur dikes. The basic equations are the continuity equations and two mometum equations keeping the vertical velocity for both layers. Here we just show in Figure 8 the calculated results by this model in comparison with the observed data. The figure shows the flow overriding the spur dike in the upper layer, the flow in the lower layer and the vertical velocity around the dike with the measured one. Due to the vertical motion at the back side of the dike, recirculation region behind the dike does not develop, which is quite different from the flow at low water. Although this model yields the reasonable results about flow field at the expense of less computer burden, it would be necessary to elaborate this model for complex flow near spur dike for the discussion of fish habitat at flood.

4.3 Three-Dimensional Flow Model

A three-dimensional model was developed to discuss the complex flow phenomenon around the spur dikes. The model employs the standard $k-\varepsilon$ turbulence model to express the effects of turbulence. The continuity equation, three momentum equations and the transport equations for the turbulence energy k and its dissipation rate ε were solved simultaneously. The finite volume method is used to discretize the equations and the SIMPLE algorithm (Patankar, 1980) to obtain the converged solution. Figure 9 shows plan views of the calculated velocities at two different level above bed and the bed shear stress between the two dikes. A recirculation zone is formed near the bed and it disappears away from the bed. The bed shear stress takes a large value near the tip of the dike and behind the dike.

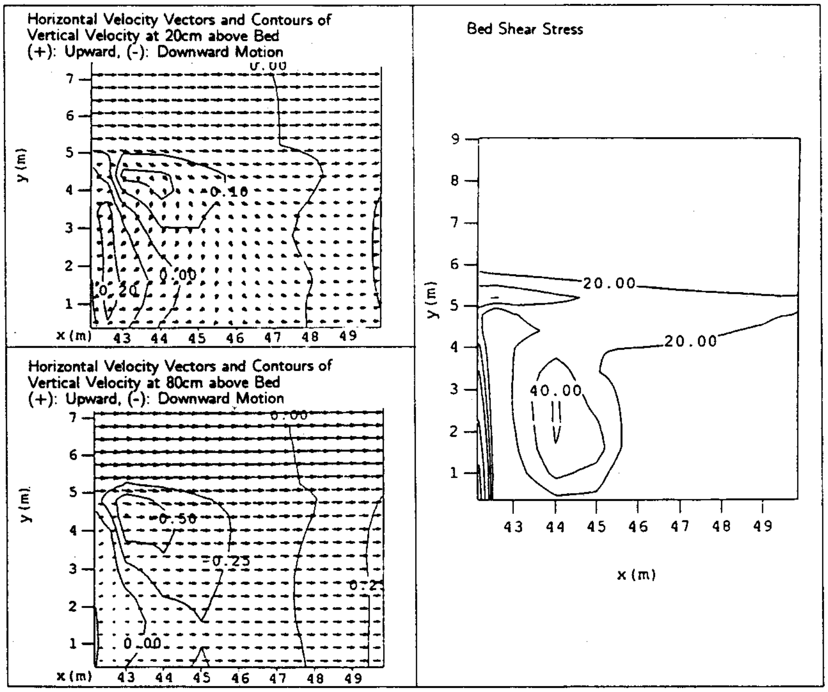


Figure 9 Calculated Velocity and Bed Shear Stress by a 3-D Model.

5. DISCUSSIONS AND CONCLUDING REMARKS

5.1 Physical Conditions and Fish Habitat

The spur dikes in Ushizu river were constructed along concave bank over a movable bed. The spur dikes directly affect the flow pattern and bed shear stress distribution. Then they exert significant effects on sediment transport, local scour and deposition near the structures. Numerical simulation showed the possibility to reproduce such complicated flow characteristics and bed topography around spur dikes, that is, the two–dimensional model is able to calculate the flow with vortices around spur dikes, while the two–layer model can take into account of the effect of vertical motion. The three-dimensional model offers more detailed information on complex flow field during floods. Physical conditions is so important to fish habitats that these numerical models help us understand and predict the flow and bed topography in the natural rivers. Major results obtained in the present study can be summarized in Table 5 and Figure 10. These attempt to rearrange the relation between physical conditions and fish habitats observed in Ushizu river in the presence of spur dikes. Evaluation methods have been proposed such as HQI method (Binns.1982) and IFIM (Stalnaker et al.,1994). Even though the study is devoted to the qualitative evaluation of for fish habitats based on filed observation, we would like to integrate this approach towards the development of quantitative evaluation method.

5.2 Influences on Fish Habitats

According to the field observation, the spur dikes provided the fish habitats with such various flow conditions as pools and riffles, variety of bed materials and the safety space within the riprap work of spur dikes.

ACKNOWLEDGEMENTS

The authors express their thanks to the members of Takeo Construction Office, Kyushu Regional Construction Bureau, Ministry of Construction for providing the data about Ushizu river, Messrs. Yoshimura and Yamauchi of Asia Consulting Co., Ltd. for their help in the field measurement and data analysis.

REFERENCES

Sanada,H.(1932): Dikes in Japan, Nikkankogyoshinbun Press.

Yamamoto, K.(1995): Groins in Japan, Sankaido Press.

Kawamura,M. and H. Miyauchi (1963): River Construction Works, Sankaido Press.

Klingeman,P.C., S.M. Kehe, and Y.A. Owusu (1984): Streambank erosion and channel scour manipulation using rockfill dikes and gabions, Water Resources Research Institute, Oregon State University, Corvallis, Oregon 97331.

Tamai,N., N. Mizuno, S. Nakamura (1993): Environmental River Engineering, University of Tokyo Press.

Shields Jr., F.D., C.M. Cooper and S.S. Knight (1995): Experiment in stream restoration, J. Hydraulic Engineering, ASCE.

Patankar,S.V. (1980): Numerical Heat Transfer and Fluid Flow, Hemisphere Publishing Corporation.

Binns,N.A.(1982): Habitat Quality Index Procedures Manual, Wyoming Game and Fish Department.

Stalnaker,C,.B.L.Lamb J. Henriken, K.Bovee, and J. Bartholow (1994): The Instream Incremental Methodology - A Primer for IFIM, National Ecology Research Center, National Biological Survey.

Table 5 Physical Conditions and Fish Habitats.

	Physical condition				Fish habitat	
	Depth	Velosity	Bed material	Other physical conditions	Identifid fish	Environment
A	L= 10 H= 50	L=4 Hu=190 Hl=	boulder (materials of dike)	cavity between rip-rap rivetment	*Carassius cuvieri* (juvenile fish) Carassius gibelio langsdorfi (juvenile fish) Pseudorasbora parva(juvenile fish) Tridentiger kuroiwae brevispinis (adult fish)	hideing place for juvenile fish
B	L= 40 H=180	L=6 Hu=180 Hl=120	gravel sand silt clay	tip of spur-dike primary vortex at low water level	Schismatogobius sp (juvenile fish) Tridentiger kuroiwae bervispinis (juvenile fish)	shoal of juvenile fish easy place to get food coming from upstream hideing place for fish
C	L= 20 H=160	L=2 Hu=80 Hl=30	silt clay	monotonous area	Zaccco platypus (juvenile fish) Carassius cuvieri (adult fish)	waste water a few number of fish
D	L= 80 H=220	L=5 Hu=180 Hl=80	gravel sand	pool current deflecter by downstream dike	Carassius cuvieri (adult fish) Carassius gibelio langsdorfi (adult fish)	shoal of fish by monotonous flow
E	L= 10 H=150	L=13 Hu=185 Hl=130	gravel sand silt	riffle	Zaccco platypus	fish like high velocity

L : in case of low water at 0.4m³/s(depth:cm,velocity:cm/s),

H : in case of high water at 38m³/s (depth:cm,velocity:cm/s), Hu : upper layer, Hl : low layer

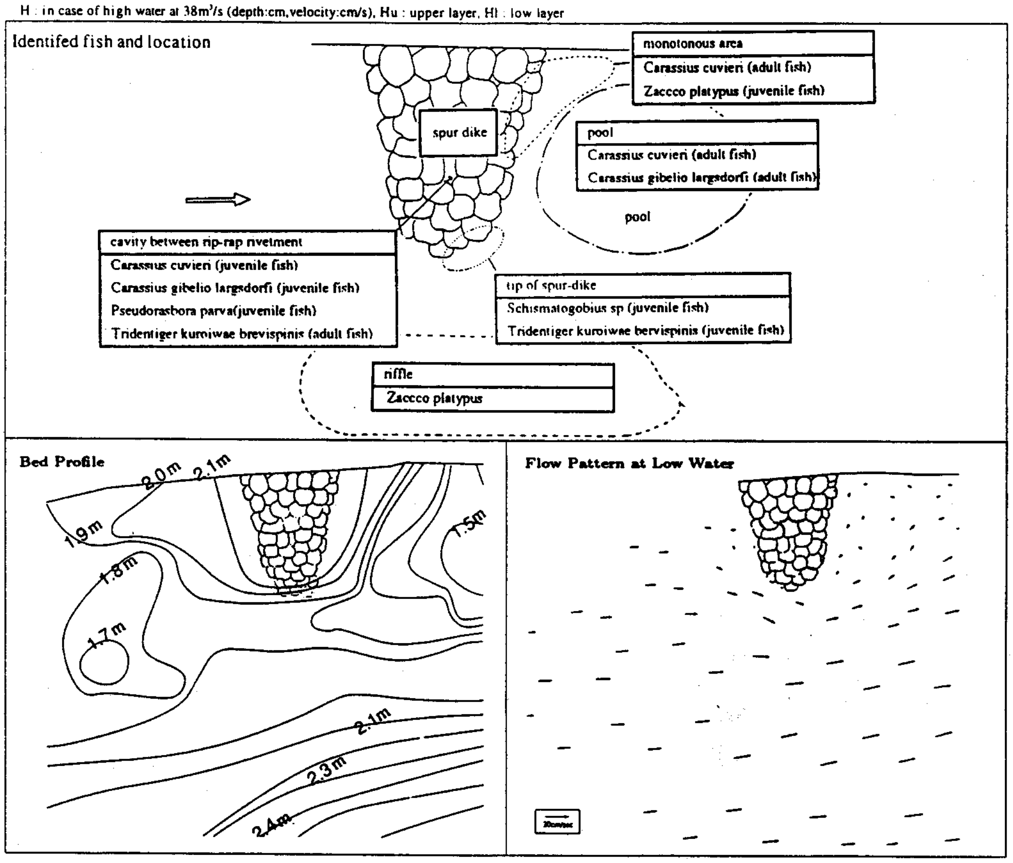


Figure 12 Physical Habitat for Fish.

An Analysis of Artificial Riffles and Enhancement of Benthic Community Diversity by Physical Habitat Simulation(PHABSIM) and Direct Observation

James A. Gore[1]

Dina J. Crawford[2]

David S. Addison[1]

[1]The Conservancy
1450 Merrihue Drive
Naples, FL 33942 USA

[2]Aquatic Resource Analysts
1638 Wellington Green
Franklin, TN 37064 USA

ABSTRACT

Critical to river rehabilitation decisions is the prediction of benefits of certain procedures. In low order systems, planning should focus on flow requirements of lotic organisms, especially benthic species. As a component of a whole-watershed restoration project to control water quality and enhance instream habitat, we examined the value of placement of artificial riffles in Holly Fork, a low-order tributary of the West Sandy River (west Tennessee). Our objective was to determine if the Instream Flow Incremental Methodology (IFIM) and its component computer model, PHABSIM, could adequately predict habitat value of the riffles for benthic macroinvertebrates and if, after colonization, this "value" was reflected by increased diversity of benthic macroinvertebrates. Holly Fork is a severely head-cut channel with two-meter high vertical embankments and a substrate dominated by sand and fines with occasional gravel riffles. Rather than target specific macroinvertebrate taxa, we chose to conduct analysis of each stream reach using macroinvertebrate diversity as the "indicator". Habitat suitability criteria were developed using diversity data from adjacent watersheds. These HSI curves predicted a range of suitable physical conditions which supported highest benthic community diversity. A traditional IFIM analysis of a reach of the Holly Fork indicated that, below 0.2 m^3/s (a flow exceeded 10% of the time), less than 5% of the wetted-area contained adequate habitat for benthic macroinvertebrates. At optimum flows (0.4 m^3/s and higher), only 15% of the wetted-area, primarily across small gravel bars, was adequate to support high community diversity. Two artificial riffles, composed of large cobble and boulder keystones, with leading and trailing aprons of medium cobble and gravel, were placed at 35 meter intervals in the test section. After hydrological stabilization and time for colonization by macroinvertebrates (six weeks), the artificial riffles were re-analyzed with PHABSIM. The simulation predicted that this reach contained significantly higher amounts of available benthic habitat at low flows (more than tripled) and over 40% of the total wetted-area should support high benthic community diversity at optimal flows. HABTAT options which predicted cell-by-cell composite suitability indicated that the presence of the artificial riffles contributed the majority of this habitat enhancement. During subsequent months, we collected benthic samples at random points on each riffle and at different discharges. A plot of cell-by-cell composite habitat suitability and sample diversity from these cells revealed a significant correlation between PHABSIM predictions and actual community diversity. This technique can be an aid in demonstrating the value of certain restoration structures during the rehabilitation planning process. Our macroinvertebrate data suggest that benthic community diversity can be a less labor-intensive HSI-target for evaluation of instream flow values that sustain ecosystem integrity.

KEY-WORDS: Restoration, PHABSIM, artificial riffle, hydraulic stream ecology, macroinvertebrates, habitat quality

INTRODUCTION

The restoration of lotic ecosystems involves an interdisciplinary approach which is focused upon recreating, at one level or another, an ecosystem which will attract plant and animal colonists and support a stable and sustainable ecosystem. Although stream restoration is most often a *response* to unplanned disturbance events, it is almost *as* often that some *a priori* planning can be made to minimize initial disturbance, thereby reducing the financial and personnel commitments to post-construction/operation/disturbance restoration activity. If direct control of point sources of pollution is also provided, the focus of the restoration scientist will be the placement and construction of instream habitat structures to enhance capture of organic detritus and *aufwuchs*, as well as, colonization by macroinvertebrate and fish species.

Traditionally, restoration work had been focused upon habitat structures designed and placed to enhance fish production. The most important components for fish production are acceptable water quality, food production areas, spawning/egg incubation areas, and cover (Wesche, 1985). Physical habitat structure is especially important for developing juvenile fish where the ability to avoid the shearing stresses of high velocity areas becomes a critical function. Thus, juveniles tend to prefer areas of brushy cover (Shirvell 1990) or deep holes (especially in small streams) (Heggenes 1990, 1991) rather than specific velocity requirements. In all cases, structural design is based upon the assumption that these habitat requirements can be controlled through design of structures which produce preferred physical and chemical conditions as related to flow conditions. In the same manner, restoration scientists are assuming that hydraulic conditions are one of a few primary templates which govern the distribution of lotic organisms. Statzner et al. (1988; for invertebrates) and Heede and Rinne (1990; for fish) have reviewed the body of data which support these assumptions. Both reviews have shown that complex flow conditions such as Reynolds velocities and shear stress, which incorporate combined factors of depth, velocity, bed roughness and viscosity of the water, are predictors of benthic densities and, perhaps, fish population response (Bartholow *et al.*, 1993).

Aquatic invertebrates are associated with a wide variety of substrates ranging from rooted vegetation and dead wood, to periphyton to all sizes of inorganic particulates (silt, sand, gravel, and cobble). Substrate composition is the most easily manipulated habitat characteristic in restoration projects. Although the opportunity for such precise study is frequently limited, the most effective rehabilitation of macroinvertebrate communities considers such factors as degree of embeddedness of particles, size of particles, contour of the substrate, and heterogeneity of substrate types in the source and recipient areas (Gore, 1985). Merritt and Cummins (1984) have listed major habitat types for families and genera of aquatic insects in North America. This can be a useful guide if source areas of colonizers are unknown or unrestored. In most cases, highest diversities and production have been reported from channels with medium cobble (256 mm diameter) and gravel substrates.

The most common structures for fish habitat enhancement have been current deflectors, overpour structures (dams and weirs), bank cover, and boulder placements. These instream structures also modify local hydraulic conditions to present preferred habitat to benthic invertebrates. Log-drop structures and other dams improve fish habitat by trapping sediment with minor impact on invertebrate densities while deflectors increase water velocity over riffles to remove accumulating sediments from cobble substrates (Luedtke *et al.*, 1976). For macroinvertebrates, then, primary considerations include formation of sediment-free riffle areas and structures to control sediment deposition. Such structures as deflectors (gabion, single, or double wing) check dams, and placement of large boulders have all been demonstrated to produce scoured areas of riffle habitat and pools (Gore and Johnson, 1981; Thompson, 1985; Wesche, 1985). Wesche has described construction details and installation characteristics.

Efforts to create artificial riffles to enhance macroinvertebrate diversity and production during rehabilitation of rivers and streams have been limited in application. This is surprising since the majority of studies on the benefits of creation of artificial riffles has indicated that diversity and production of macroinvertebrates most often duplicates communities in unimpacted areas (Gore, 1985) and results in enhanced fisheries diversity and production (Edwards et al., 1984; Bingham and Miller, 1989). The objectives of this research were to examine the impact of placement of artificial riffles during a watershed restoration project. Physical habitat modelling of artificial riffles provided predictions of change in macroinvertebrate diversity with changes in flow and allowed us a point of comparison to field-collected macroinvertebrate samples and a chance to verify the predictive ability of the models and their application to stream rehabilitation.

The Physical Habitat Model (PHABSIM) was used to evaluate the habitat quality of the artificial riffles. Shuler and Nehring (1993) have demonstrated that PHABSIM, the software package used in the Instream Flow Incremental Methodology, can be used to evaluate stream enhancement activities. The Instream Flow Incremental Methodology (IFIM) is a series of concepts, techniques and computer programs to combine channel morphology, characteristics of flow and biological preferences of target organisms to predict gains or losses in physical habitat under new or modified flow regimes (Bovee, 1982) (Figure 1). Based on the assumption that lotic biota have their distribution and certain phases of their life cycles controlled by the hydraulic conditions within the water column, various elements of PHABSIM predict changes in velocity, depth and channel resistance in each habitat element surveyed. These conditions are compared to the habitat preferences of target species, guilds or communities. The amount of usable habitat area (or volume) is predicted for each discharge of concern. Bovee (1982, 1986) has provided details of the IFIM procedures. Reviews of this technique and appropriate application have been provided by Gore and Nestler (1988) and Nestler *et al.* (1989).

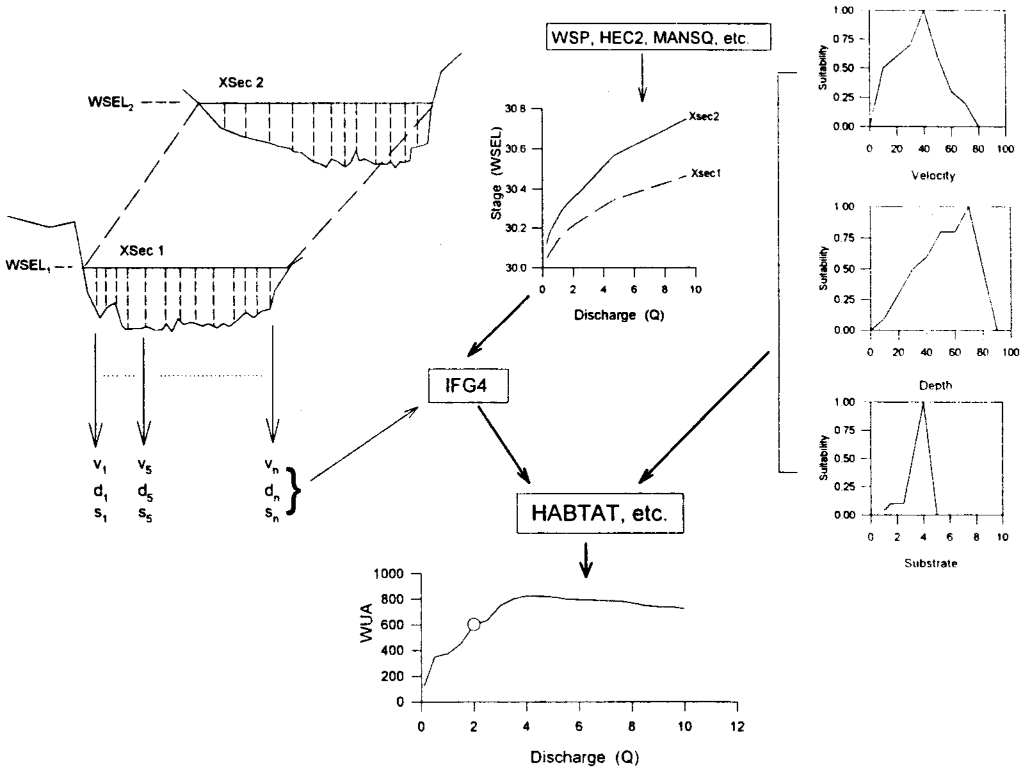


Figure 1. Schematic of the IFIM and PHABSIM process. Cross-sections are surveyed and combined with stage-discharge relationships in IFG4 to predict hydraulic change with discharge. HABTAT combines suitability criteria with hydraulic information to produce habitat quality estimates (WUA) as a function of discharge.

METHODS

Site Description

Holly Fork Creek (Henry County, Tennessee) is a second-order stream, draining agricultural areas in the Big Sandy/West Sandy drainage which empties into Kentucky Lake, an impounded portion of the Tennessee River. Holly Fork is a severely head-cut stream with almost vertical embankments of up to three meters height. Substrates are primarily sand and small gravel with isolated patches of coarser gravel and small cobble. Discharges range between 0.01 m^3/s and greater than 1 m^3/s. However, the majority of daily flows do not exceed 0.4 m^3/s.

According to the recommendations of Newbury and Gaboury (1993), two artificial riffles were constructed at 100 meter intervals along a typical reach of Holly Fork. These riffles were constructed of a mix of gravel with medium and large cobble. The riffles were hexagonal in shape with an upstream face with a 4:1 slope and a crest of 45 cm and a downstream face with a 20:1 slope. Each riffle was built in June of 1995 and allowed to stabilize, hydrologically and biologically, for approximately eight weeks.

During August, 1995, this section was surveyed according to recommended techniques and evaluated for available habitat for benthic macroinvertebrates at a series of typical discharges. Habitat suitability curves were not for a single species but, using the techniques proposed by Gore (1989), were curves which represented conditions which promote the highest community diversity. The raw data used to calibrate these curves came from a composite of over 75 0.1 m^2 samples, with accompanying depth, velocity and substrate characterizations, taken from similar low order, relatively undisturbed, low-gradient streams in the region. Sample diversity data, rather than density of individual taxa, were fit to exponential polynomial regressions against velocity, depth, and substrate (Gore and Judy 1981). Highest sample diversities occurred at velocities of 65 cm/s, depths of 27 cm, and substrates of gravel and small cobble.

PHABSIM analysis of the artificial riffles revealed the predicted gain and loss of available habitat to support high macroinvertebrate diversity over a range of discharges from 0.05 m^3/s to 1.15 m^3/s. Using the LSTCEL routine, the composite suitability (the product of depth, velocity, and substrate suitability values) were produced for all surveyed cells at all discharges. In theory, as composite suitability increases, macroinvertebrate diversity of those cells should also increase. We tested this prediction, by sampling cells, at random, at different discharges. In August, 1995, eight quantitative circular-bottom-samples were collected from both riffles at a discharge of 0.25 m^3/s and eight more samples were collected in October, 1995, at a discharge of 0.71 m^3/s. For each sample, Shannon-Weiner diversity (Pielou 1975), using log-base-2, was calculated and compared to predicted habitat suitability for the collection site at the discharge when the sample was collected.

RESULTS

Prior to construction of the artificial riffles, approximately 17% of the wetted-habitat area contained suitable habitat for benthic macroinvertebrates (Figure 2). This value could have been considerably lower in other sections of the stream since the majority of the habitat value was located at a single patch of gravel approximately half-way between the riffle areas. The results of the physical habitat simulated predicted a three-fold increase in available habitat at moderate discharges and an over-all increase in macroinvertebrate habitat over the entire range of discharges. Indeed, over the normal range of discharges during an annual hydrograph, PHABSIM analysis predicted that the effect of the artificial riffles was to maintain high quality habitat during periods of lowest flow by impounding water across the upstream face and discharging it at higher velocities across the gravel and cobble downstream aprons.

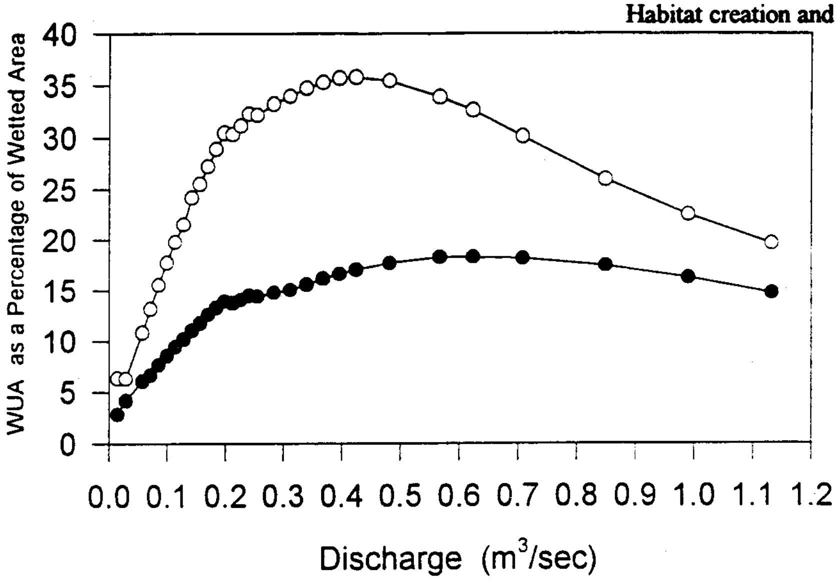


Figure 2. Prediction of habitat quality as a function of discharge before (solid circles) and after (open circles) installation of artificial riffles in a section of Holly Fork.

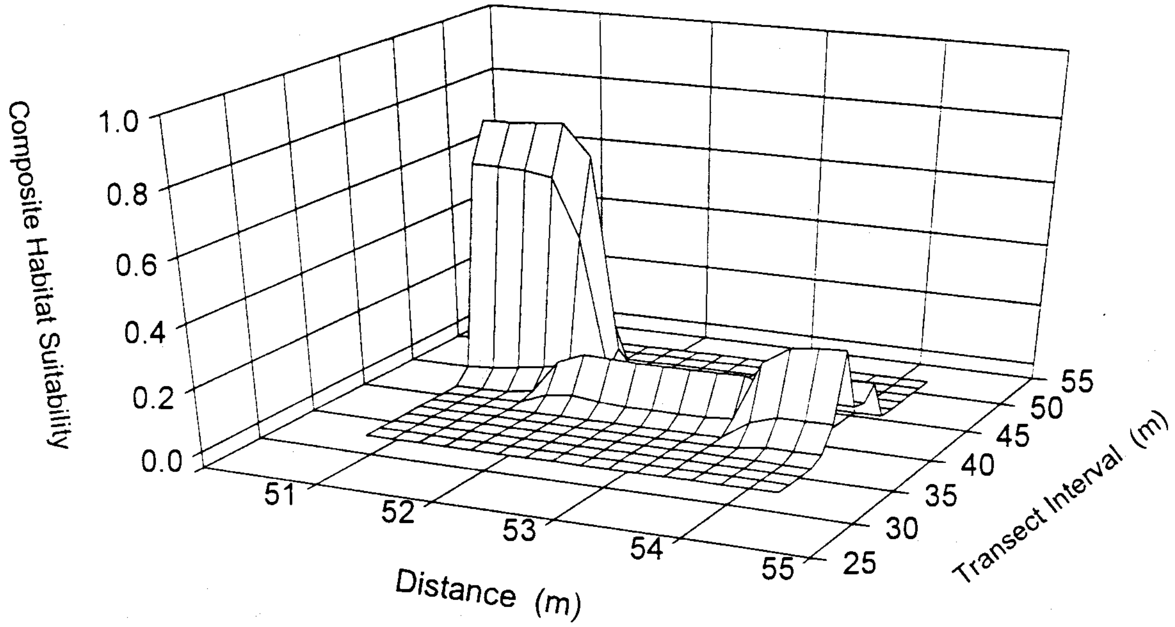


Figure 3. Composite habitat suitability at upstream artificial riffle from upstream (50 m) to downstream (55 m) at 0.06 m^3/s.

At low discharges, the greatest habitat value was predicted to occur in cells near the *thalweg* of the stream and near the crest of the artificial riffle, where velocities might be highest (Figure 3). At higher discharges, optimal combinations of depth and velocity occurred at the most downstream areas of the riffle (Figure 4). At the highest discharges simulated, areas of high quality habitat were predicted to be located in the mid-reaches of the riffle area and at areas more lateral to the *thalweg* of the stream (Figure 5). This "wave" of change in location of optimal habitat was predicted for both artificial riffles.

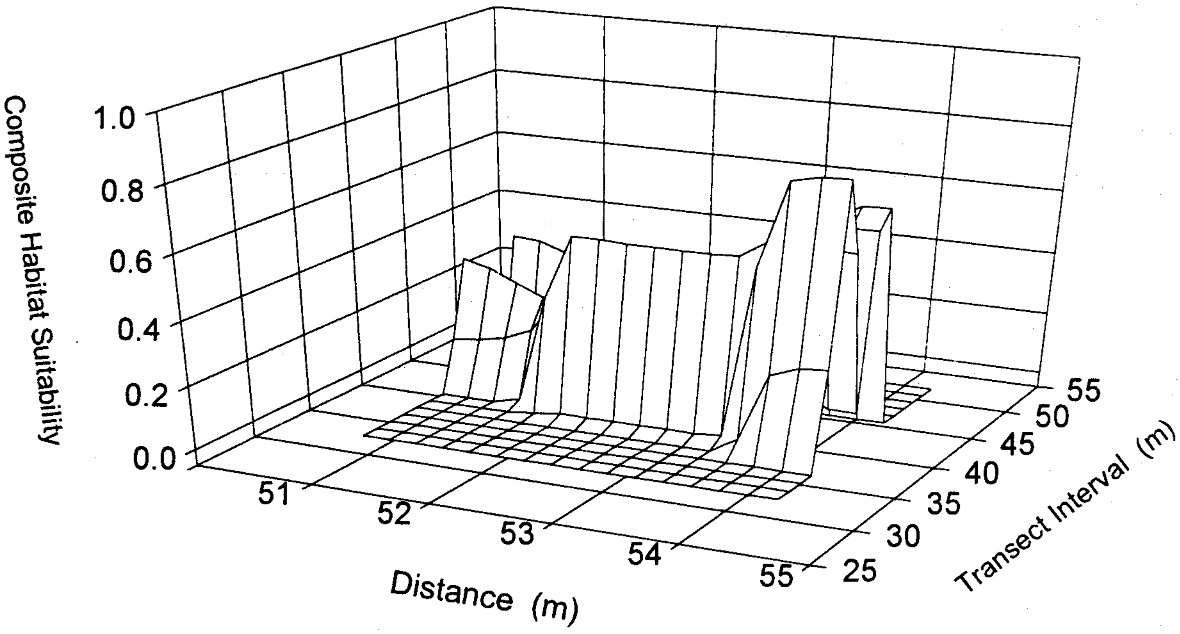


Figure 4. Composite habitat suitability at upstream artificial riffle from upstream (50 m) to downstream (55 m) at 0.7 m^3/s.

A comparison of sample diversity and predicted composite suitability for the cells from which the samples were taken is shown in Figure 6. There was a significant correlation between predicted habitat suitability and actual macroinvertebrate diversity for the cells examined in August and October of 1995. Although the data fit both first- and second-order regressions, the second-order regression (to fit the log-normal nature of diversity measures) was deemed to be the most appropriate predictor.

DISCUSSION

In low order rivers and streams under regulated flow conditions, say low flows from abstractions for irrigation, physical habitat modeling techniques may be useful in evaluating habitat structures which enhance preferred hydraulic conditions for target biota. These techniques can be used during the reconnaissance-phase of a restoration or rehabilitation project for design and placement of structures and, where necessary, for permitting of these structures by regulatory agencies.

The predictions of these simulations also indicate that habitat quality of any patch does not remain constant and that the riffle does not uniformly increase and decrease in habitat quality as discharge varies. Each cell ("habitat patch" displays a unique habitat condition with change in discharge (Figures 7 and 8). Those cells closest to the *thalweg*

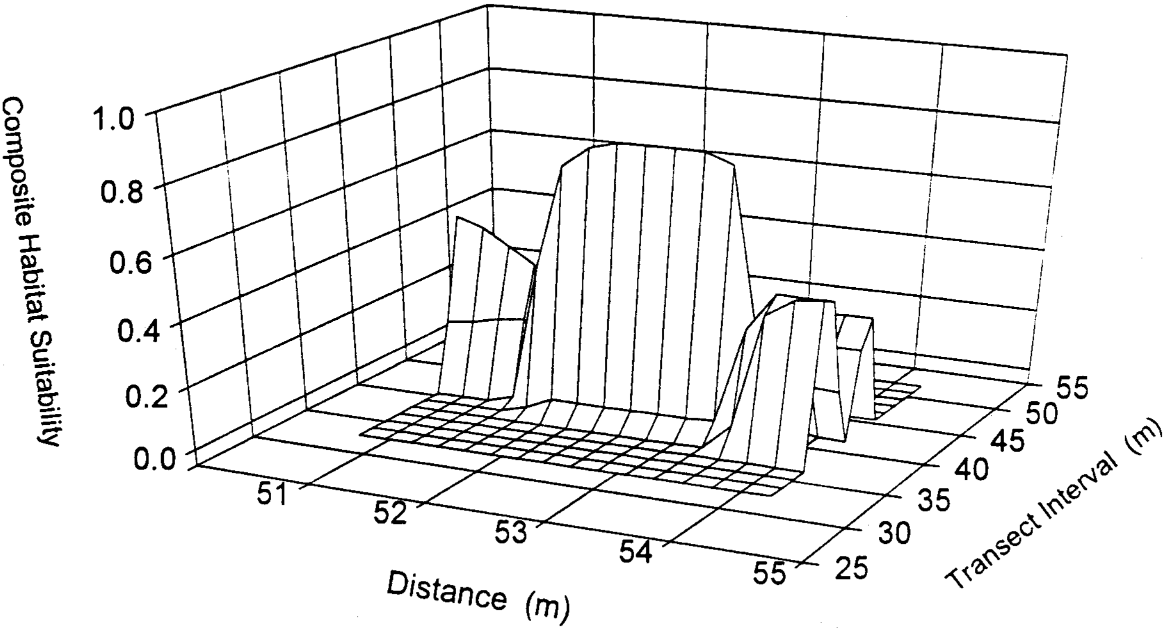


Figure 5. Composite habitat suitability at upstream artificial riffle from upstream (50 m) to downstream (55 m) at 1.12 m^3/s.

have high quality habitat at low discharges near the head of the riffle and those cells which are laterally removed from the *thalweg* have higher habitat quality at high discharges and at locations towards the tail of the riffle. This suggests that, like fish, macroinvertebrates must be able to "track" the changes in habitat quality and their location within the riffle. The simulation suggests that these changes occur relatively slowly and predictably over the change in hydrograph and the results of our field sampling suggest that this "tracking" does occur. The mechanism of this tracking is not identified; that is, we are unable to speculate if the change in position of high diversity samples is the result of migration of individuals from a patch of declining quality to a patch of increasing quality or if the change in position is a result of new colonization of newly occurring high quality cells and the abandonment (through drift?) of low quality cells as discharge changes.

Davis and Barmuta (1989) suggest that samples taken from an area as large as one-tenth of a square meter contain representatives from several hydraulic microhabitats, Statzner *et al.* (1988) have demonstrated that densities can be predicted, based upon composite sample-by-sample suitabilities. The data presented here indicate that, on a larger scale, a surveyed cell of 10 m^2, composite suitability predictions are adequate to determine macroinvertebrate community diversity. Although the scale used in the PHABSIM simulation is not an adequate representation of the hydraulic and ecological phenomena identified by Statzner *et al.* (1988), Davis and Barmuta (1989), and Barmuta (1990), this simulation does appear to be adequate to demonstrate the value of the placement of artificial riffles in certain areas of a stream during restoration activity. Similar merit has been found in the application of these modelling techniques to predict the value of other restoration and rehabilitation structures (weirs and deflectors) in similar application (Gore and Hamilton 1996).

The greatest drawback to application of IFIM techniques, especially PHABSIM, to stream restoration is the lack of a clearly-demonstrated relationship between usable habitat area and population numbers or biomass support. Although Bovee (1982) has indicated that the relationship between usable habitat area and carrying capacity of the

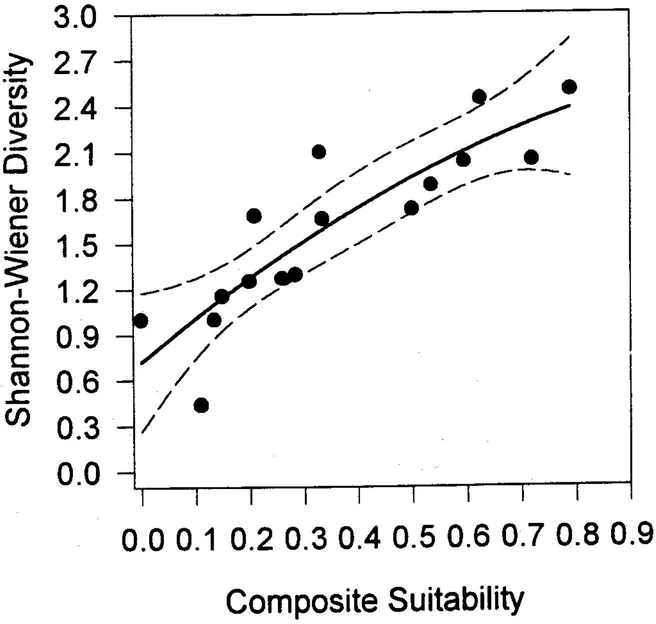


Figure 6. Regression of diversity and composite habitat suitability with 95% confidence intervals. Regression $r^2 = 0.78$.

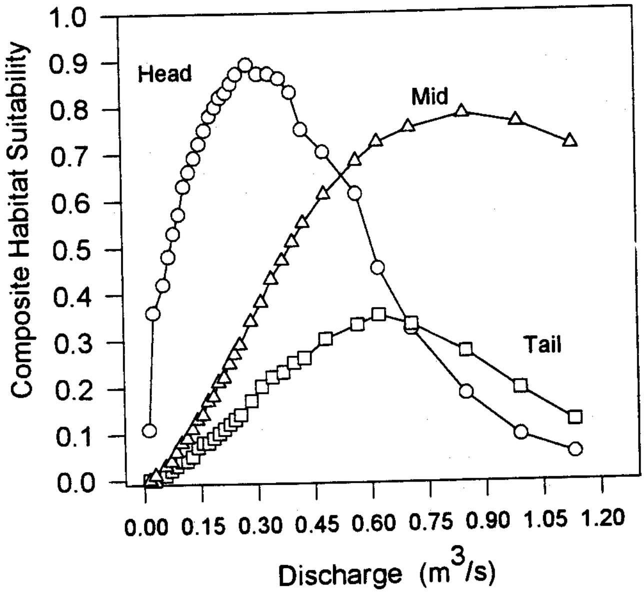


Figure 7. Habitat suitability of cell 11, within the *thalweg*, at upstream artificial riffle along the length of the cell.

stream reach is implicit in the development of PHABSIM, this has been difficult to demonstrate in practice. There have been successes (Cheslak and Jacobson, 1990) and failures (Scott and Shirvell, 1987) in predicting changes in fish populations using PHABSIM. However, Statzner *et al.* (1988) suggested that, for more non-mobile organisms, such as benthic macroinvertebrates, the relationship between physical habitat and density *is* predictable. The results

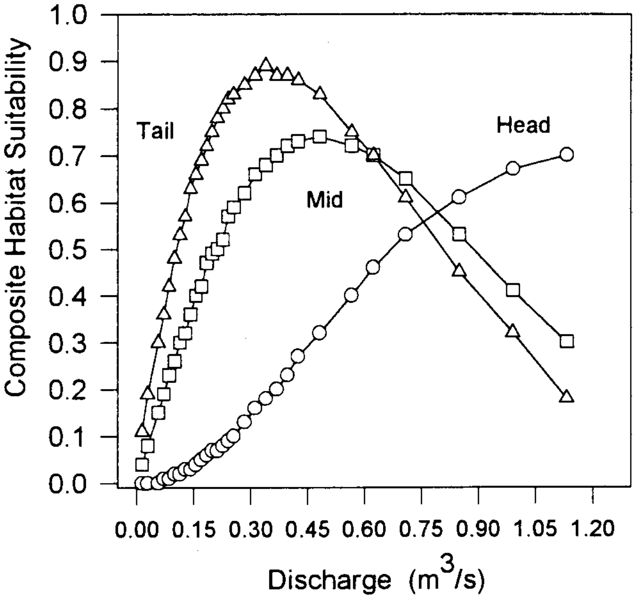


Figure 8. Habitat suitability of cell 8, two meters lateral from *thalweg*, at downstream artificial riffle along the length of the cell.

of this project do not suggest that biomass or density can be predicted using PHABSIM techniques, but do indicate that there is a significant correlation between macroinvertebrate diversity and predicted habitat quality.

Although certainly not the only templates which determine the successful occupancy of certain section of stream reach, hydraulic variables appear to be a major influence on distribution of most lotic species. Hydraulic habitat models, then, can be a useful tool to evaluate the benefit of certain restoration activities. IFIM is a flexible, general method for evaluation of habitat conditions under a wider variety of hydrological conditions. Only recently, has this technique been explored for application to restoration or rehabilitation. The initial findings, as reported here, suggest that this is a beneficial avenue for further research with a goal of improving our ability to restore or rehabilitate lotic ecosystems and sustain their communities for long periods of time.

ACKNOWLEDGEMENTS

Research on Holly Fork was funded under contract to Austin Peay State University, Clarksville, Tennessee, and the Tennessee Department of Environmental Quality as a component of the Big Sandy/West Sandy watershed restoration project. We thank Dr. Steven W. Hamilton, Department of Biology, APSU, and his most able graduate and undergraduate student assistants for their logistical support.

REFERENCES

Barmuta, L.A. (1990). Interaction between the effects of substratum, velocity, and location on stream benthos: an experiment. *Australian Journal of Marine and Freshwater Research* 41, pp. 557-573.

Bartholow, J.M., J.L. Laake, C.B. Stalnaker, and S.C. Williamson. (1993). A salmonid population model with emphasis on habitat limitations. *Rivers* 4, pp. 265-279.

Bingham, C.R. and A.C. Miller. (1989). Colonization of a man-made gravel bar by Oligochaeta. *Hydrobiologia* 180, pp. 229-234.

Bovee, K.D. (1982). A guide to stream habitat analysis using the instream flow incremental methodology. *Instream Flow Info. Paper No. 12*, U.S. Fish. Wildl. Serv., FWS/OBS-82/26.

Bovee, K.D. (1986). Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. *Instream Flow Info. Paper No. 21*, U.S. Fish. Wildl. Serv., Bio. Rpt. 86(7).

Cheslak, E.F., and A.S. Jacobson. (1990). Integrating the instream flow incremental methodology with a population response model. *Rivers* 1, pp. 264-288.

Davis, J.A., and L.A. Barmuta. (1989). An ecologically useful classification of mean and near-bed flows in streams and rivers. *Freshwater Biology* 21, pp. 271-282.

Edwards, C.J., B.L. Griswold, R.A. Tubb, E.C. Weber, and L.C. Woods. (1984). Mitigating effects of artificial riffles and pools on the fauna of a channelized warmwater stream. *North American Journal of Fisheries Management* 4, pp. 194-203.

Gore, J. A. (1985). Mechanisms of colonization and habitat enhancement for benthic macroinvertebrates in restored river channels. pp. 81-101 *in*: J.A. Gore (ed.) *The Restoration of Rivers and Streams*. Butterworth, Boston.

Gore, J.A. (1989). Models for predicting benthic macroinvertebrate habitat suitability under regulated flows' pp. 253-265 in: J.A. Gore and G.E. Petts (eds.) *Alternatives in Regulated River Management*, CRC Press, Boca Raton, FL.

Gore, J.A., and S.W. Hamilton. (1996). A comparison of flow-related habitat evaluations downstream of low-head weirs on small and large fluvial ecosystems. *Regulated Rivers* (In Press)

Gore, J. A., and L.S. Johnson. (1981). Strip-mined river restoration. *Water Spectrum* 13, pp. 31-38.

Gore, J.A., and R.D. Judy, Jr. (1981). Predictive models of benthic macroinvertebrate density for use in instream flow studies and regulated flow management. *Canadian Journal of Fisheries and Aquatic Science* 38, pp. 1363-1370.

Gore, J.A., and J.M. Nestler. (1988). Instream flow studies in perspective. *Regulated Rivers* 2, pp. 93-101.

Heggenes, J. (1990). Habitat utilization and preferences in juvenile Atlantic salmon (*Salmo salar*) in streams. *Regulated Rivers* 5, pp. 341-354.

Heggenes, J. (1991). Comparisons of habitat availability and habitat use by an allopatric cohort of juvenile Atlantic salmon Salmo salar under conditions of low competition in a Norwegian stream. *Holarctic Ecology* 14, pp. 51-62.

Heede, B.H., and J.N. Rinne. (1990). Hydrodynamic and fluvial geomorphological processes: implications for fisheries management and research. *North American Journal of Fisheries Management* 10, pp. 249-268.

Luedtke, R. J., M.A. Brusven, and F.J. Watts. (1976. Benthic insect community changes in relation to instream alterations of a sediment-polluted stream. *Melanderia* 23, pp. 21-39.

Merritt, R. W. and K.W. Cummins (eds.). (1984). *An Introduction to the Aquatic Insects of North America*, 2nd. Ed., Kendall/Hunt, Dubuque, Iowa.

Nestler, J.M., R.T. Milhous, and J.B. Layzer. (1989). Instream habitat modeling techniques. pp. 295-315 *in*: J.A. Gore and G.E. Petts (eds.) *Alternatives in Regulated River Management*, CRC Press, Inc., Boca Raton, FL.

Newbury, R.W., and M.N. Gaboury. (1993) *Stream Analysis and Fish Habitat Design*. Newbury Hydraulics Ltd. (co-publisher: Manitoba Natural Resources), Gibsons, British Columbia.

Pielou, E.C. 1975. *Ecological Diversity*. Wiley-Interscience, New York.

Scott, D.S., and C.S. Shirvell. (1987). A critique of the instream flow incremental methodology and observations on flow determinations in New Zealand. pp. 27-43 *in*: J.F. Craig and J.B. Kemper (eds.) *Regulated Streams*. Plenum Press, NY.

Shirvell, C.S. (1990). Role of instream rootwads as juvenile coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*O. mykiss*) cover habitat under varying streamflows. *Canadian Journal of Fisheries and Aquatic Sciences* 47, pp. 852-861.

Shuler, S.W., and R.B. Nehring. (1993). Using the physical habitat simulation model to evaluate a stream habitat enhancement project. *Rivers* 4, pp. 175-193.

Statzner, B., J.A. Gore, and V.H. Resh. (1988). Hydraulic stream ecology: observed patterns and potential application. *Journal of the North American Benthological Society* 7, pp. 307-360.

Thompson, C. S. (1985). Stream relocation on surface mined land. pp. 39-49 *in*: L.B. Starnes (ed.) *Fish and Wildlife Relationships to Mining*. Water Quality Section, American Fisheries Society, Milwaukee, Wis.

Wesche, T. A. (1985). Stream channel modifications and reclamation structures to enhance fish habitat. pp. 103-163 *in*: J.A. Gore (ed.) *The Restoration of Rivers and Streams.* Butterworth, Boston.

CONSTRUCTION OF GENERATING STATION AT THE MERCIER DAM: FEASIBILITY STUDY OF A MAN-MADE WALLEYE STREAM SPAWNING HABITAT USED AS A MITIGATIVE MEASURE

Benoit Gagnon[1], Marc Gendron[2] and Isabelle Thériault[3]

1. Hydro-Québec, Service Études environnementales, 800, de Maisonneuve est, 19th floor,
Montréal, Québec, Canada H2L 4M8, bgagnon@ge.hydro.qc.ca

2. Environnement Illimité inc., 1453, Saint-Thimothée, Montréal, Québec, Canada H2L 3N7, envill@cam.org

3. Tecsult inc., 85, Sainte-Catherine ouest, Montréal, Québec, Canada H2X 3P4

ABSTRACT

Hydro-Québec has been studying the possibility of building an 90-MW generating station at the Mercier dam on the Gatineau River. Construction of the Mercier dam in 1927 led to the creation of the Baskatong reservoir, which regulates the flow of the Gatineau River. The generating station would be built along the right bank of the existing dam. The preferred variant consists of excavating a 1.2-km canal between the dam and Chute du Lion falls so that a portion of the denivellation can be recovered and generating capacity increased by 6 MW. The excavation of a canal in the riverbed will decrease the water level over the entire section between the Mercier dam and Chute du Lion falls as well as concentrate most of the flow in the excavated canal. Depending on the flow, a decrease in the water level would dewater 4.8 to 9.8 ha of the riverbed, which represents 31 to 81% of the current surface area of the watercourse. The main impact of the project is the dewatering of a walleye spawning ground located downstream from the dam. It measures about 10,000 m^2 and represents over half of the spawning habitat for this section of the river. A large proportion of the fish population would be affected, with a resulting decline in the abundance of certain species or their disappearance. The corrective measures that have been retained are aimed at preserving the fish spawning habitats in this section of the river. These measures include relocating the spawning ground closer to the current spawning area, dredging three shoals so that a river flow is maintained, and creating a shallow berm in order to decrease flow velocity and allow spawners to migrate upstream. A method of managing the proposed generating station during the spring was put forth: it consists in allowing fish to access the developed spawning ground while taking into account the various reservoir management constraints as well as users' concerns regarding the reservoir periphery. The residual impact on the fish populations and their spawning habitat will be slight despite the major changes made to the river section. Previous experiences have shown that this type of development has a high rate of effectiveness with walleye. Environmental studies of spawning ground developments will be used to optimize the project and yield economic gains of about $0.3 million (1992) over 50 years.

KEY-WORDS: Hydro-Québec / Gatineau River / Hydraulic modelling / Environnmental Impact Assessment / Mitigative measure / Dam / Walleye / Spawning ground development

INTRODUCTION

The Mercier dam is located on the Gatineau River 186 km north of Ottawa (see Figure 1). Built in 1927, it led to the creation of the Baskatong reservoir (298 km^2; 3049 hm^3), which regulates the flow of the Gatineau River and supplies the Paugan (1929), Chelsea (1927) and Rapides-Farmers (1927) generating stations with a total nominal capacity of 495 MW and an annual average energy output of of 2,215 GWh.

Hydro-Québec is studying the possibility of building a 90-MW hydroelectric generating station to the right of the Mercier dam to develop the energy potential of the Baskatong reservoir. One proposed variant consists in excavating a long tailrace canal in the Gatineau riverbed. Excavation would decrease the flow in the residual riverbed and would dewater fish spawning and feeding habitats found in the river section downstream of the dam. Environmental impact studies, as per section 31.1 of the *Environment Quality Act*, were carried out over a two-year period (Hydro-Québec, 1995a, b). One of the environmental issues involves the preservation of the spawning and rearing habitat of walleye (*Stizostedion vitreum*) located in the reservoir tributaries and downstream of the dam.

This paper presents a study of the technical and biological feasibility of developing a walleye spawning habitat as a mitigative measure for decreasing the impacts of the new generating station. This mitigative measure is being proposed in order to preserve aquatic habitats within a context of sustainable development and to comply with the provincial no net loss policy.

AIM OF THE STUDY

The aim of the hydraulic and ichtyological studies consists in assessing the feasibility of developing the section of the Gatineau River in order to preserve the fish spawning and rearing habitats, especially that of walleye. More specifically, the study involves designing a spawning habitat, maintaining a constant flow in the residual riverbed, and facilitating the upstream migration of the fish to the spawning grounds.

PROJECT OVERVIEW

The Mercier generating station would be built on the right bank of the river to the right of the Mercier dam. Designed for a maximum usable flow of 470 m^3/s, the proposed development comprises a headrace canal, two water intakes integrated to a power house, two Kaplan generating units, and a tailrace canal that emerges at the foot of the dam. Two variants have been studied, i.e. the *Mercier variant* and the *Mercier variant with excavation*. For the Mercier variant, the net head of water is 20.2 m with an installed capacity of 85 MW and an annual energy output of 346 GWh. The development cost is estimated at $168 million in 1994 dollars (Hydro-Québec, 1995c).

The Mercier variant with excavation consists in excavating a tailrace canal that is 1.2 km in length by 20 m in width at level 194.5 m between the dam and Chute du Lion falls (Figure 1) in view of recovering part of the denivellation of the falls, i.e. 1.6 m, and increasing generating capacity by 6 MW, for an annual energy output of 381 GWh. The development costs have been estimated at $180 million in 1994 dollars (Hydro-Québec, 1995c).

A spawning habitat design has been established for each of the above variants. In this paper we will be presenting the design that was retained for the Mercier variant with excavation.

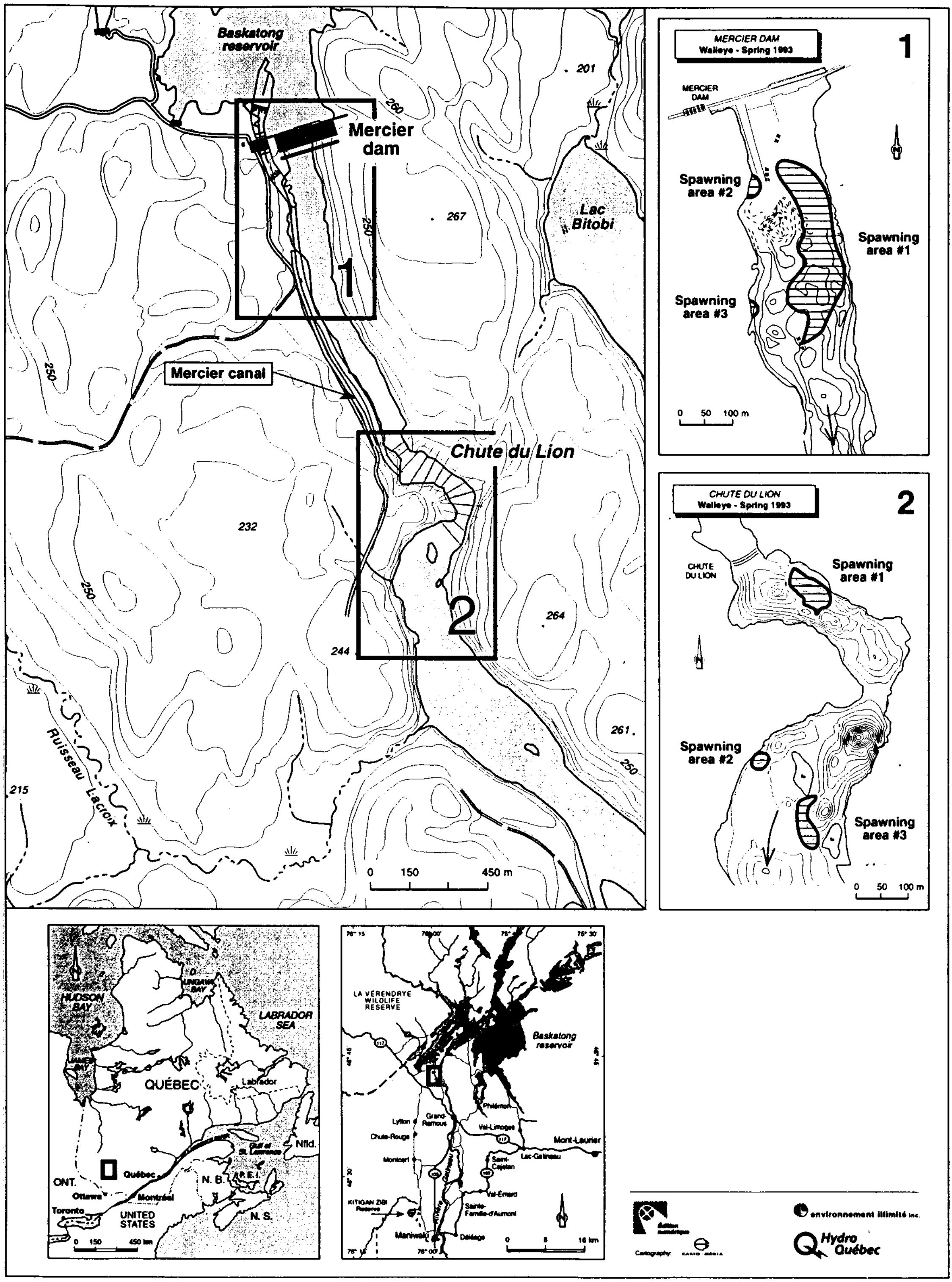


Figure 1: Location of the study area and walleye spawning habitats identified in the spring of 1993 downstream of the Mercier dam and Chute du Lion in the Gatineau River

DESCRIPTION OF THE AREA

Hydrological and Hydraulic Conditions

The Baskatong reservoir is filled in spring and progressively emptied in winter depending on energy requirements. Its maximum and minimum operating levels are 223.14 and 207.6 m, respectively. The river's mean flow at the Mercier dam is 261 m^3/s. Several operation specifications are used to control the flow and water levels in order to reduce floods downstream, especially in the town of Maniwaki, while maintaining a minimum flow for Maniwaki's water supply and providing a compensation flow during the summer.

The management of reservoir water levels also takes into account users' concerns with the immediate surroundings, such as dock and water access when the fishing season opens in May, navigability, access to beaches in the summer, shoreline erosion, and the maintenance of water levels to ensure the walleye's access to spawning grounds located in the reservoir's tributaries.

Hydraulic conditions found in the river section between the Mercier dam and Chute du Lion falls are described in Table 1. Flow velocities, which vary according to the discharge, are quite high. Backwater curves and water depths above the thalweg are shown in Figure 2. A 3-m denivellation is caused by the waterfall.

Table 1: Hydraulic conditions in the river section between the Mercier dam and Chute du Lion falls

	100 m^3/s		**235 m^3/s**		**470 m^3/s***	
Section	**Water level (m)**	**Velocity (m/s)**	**Water level (m)**	**Velocity (m/s)**	**Water level (m)**	**Velocity (m/s)**
Mercier dam – Chute du Lion falls	199.8 à 200.3	0.7 à 1.4	200.7 à 201.3	0.9 à 1.6	201.9 à 202.4	1.1 à 2.0
Above Chute du Lion falls	196.8 à 199.8	2.8	197.7 à 200.7	3.6	198.9 à 201.9	4.5

* Maximum usable flow of the future generating station

Aquatic Fauna

The Gatineau River and Bitobi lake have a varied fish community. The field survey yielded 16 species including longnose sucker *(Catostomus catostomus)* - 20.5%; yellow perch *(Perca flavescens)* - 19.5%; lake herring *(Coregonus artedii)* - 16.8%; shorthead redhorse *(Moxostoma macrolepidotum)* - 12.8%; walleye - 12.5%; lake whitefish (*Coregonus clupeaformis)*; and lake sturgeon (*Acipenser fulvescens*).

Walleye spawns between the first and third week of May. Three spawning grounds were located downstream from the Mercier dam, the main one being located on the west bank on a large shoal made of boulders and cobble (spawning ground no. 1; Figure 1). This site, which is mostly used by walleye, yields a maximum harvest of 500 eggs per net per night using a drift gill net. The highest number of walleye eggs were collected at locations with a depth of 0.5 to 1.5 m and a flow velocity of 0.3 to 1.5 m/s.

Additionally, three spawning grounds were located downstream from Chute du Lion falls. The main one was on the left bank, approximately 150 m downstream from the weir (spawning ground no. 1; Figure 1). This site is especially used by walleye and yields a maximum harvest of 317 eggs per net per night.

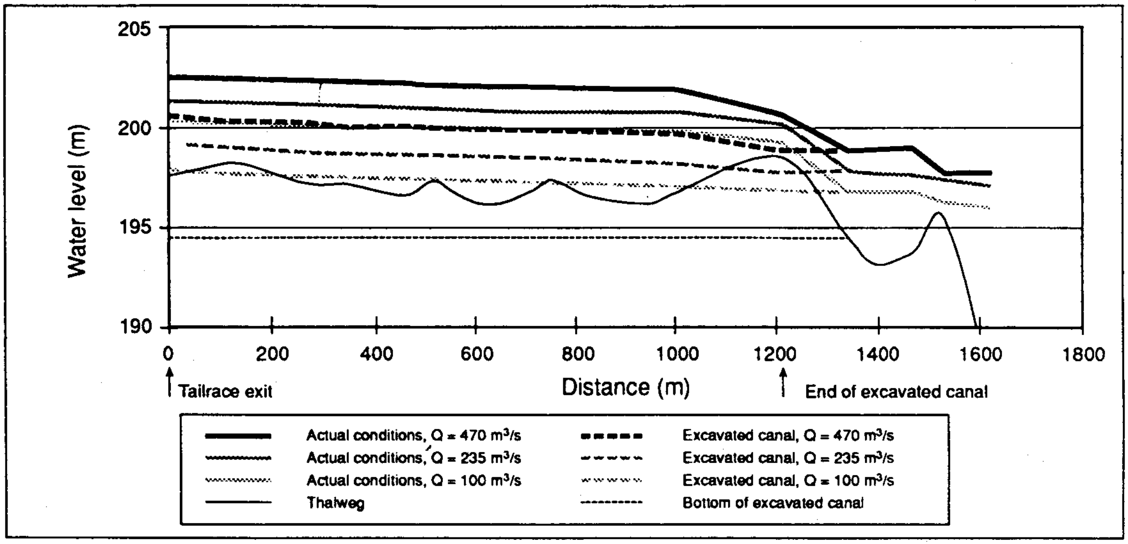


Figure 2: Water level profile between the end of the tailrace canal and Chute du Lion
(After Tecsult, 1995)

Hydraulic conditions at the spawning grounds are closely linked to the discharge from the dam. During spring spawning, the discharge varies considerably, leading to vast changes in flow velocity and water depth. The main spawning ground at the foot of the Mercier dam has a total potential area of approximately 10,000 m^2 and a denivellation of almost 3 m, thus providing adequate spawning habitat conditions for walleye, even for various flow scenarios. Field work and backwater curve simulations suggest that this spawning ground may offer appropriate spawning conditions for flows ranging from 50 to 500 m^3/s, with optimal flows found between 100 and 250 m^3/s. With these flow ranges, the spawning ground can cover an area of 5,000 to 7,000 m^2.

Downstream from Chute du Lion falls, the main spawning ground covers approximately 3,500 m^2 (spawning ground no.1, Figure 1). Favorable spawning conditions exist when the flow ranges from 50 to 250 m^3/s. When the flow exceeds 250 m^3/s, the walleye's spawning potential decreases as the flow increases due to the deeper water, faster currents, and steep shoreline slopes. Site no. 3, however, has considerable potential on the island shoreline when the water level is high (discharge > 250 m^3/s).

The current discharge at the Mercier dam does not provide optimal conditions for yearly reproduction of the walleye. During some years the dam is shut down during spawning season for the filling up of the reservoir. Also, abrupt changes in flow after spawning can threaten the survival of the eggs either because the water level drops, which exposes the eggs, or the current gains speed and possibly dislodges the eggs from the spawning ground.

IMPACTS OF THE PROJECT

Hydraulic and Hydrological Changes

By calculating the backwater curves (HEC2 software, 1988) we assessed changes in hydraulic conditions (flow, speed, level, depth) found in the river section between the Mercier dam and Chute du Lion falls.

A 1.2-km-long canal excavated along the river's right bank would lower the water level throughout the section between the Mercier dam and Chute du Lion falls and concentrate most of the flow in the excavated canal. A decrease in the water level in relation to natural conditions varies according to the flow and the location in the river (Figure 2). The decrease in water level is greater when the flow is low and peaks immediately upstream from Chute du Lion falls. Water levels decrease 2.3 m on average for the maximum usable flow, causing dewatering of the riverbed over an area of between 4.8 and 9.8 ha., which represents 31 to 81% of the area presently covered by that section of the river (Table 2; see also Figure 3).

Table 2: Watercourse surface area in the section between the Mercier dam and Chute du Lion falls for the variant with excavation under different flow conditions

Surface area	Flow		
	100 m³/s	235 m³/s	470 m³/s
Current surface area of the riverbed (ha)	12.1	14.8	15.7
Surface area after excavation (ha)	5.9	9.7	15.8
Dewatered surface area of the riverbed (ha)	9.8	8.8	4.8
Portion of riverbed that has been dewatered (%)	81	59	31

With turbine flow under 235 m³/s, the flow would be concentrated entirely in the excavated canal. Some depressed areas would still remain in the deepest parts of the river, but with no flow velocity. These areas would not connect to the excavated canal. Water would start flowing into the river when turbine flow reaches approximately 235 m³/s. Some of the water would flow into a river section parallel to the canal, over a distance of 890 m. However, the downstream area of the river section parallel to the canal would experience no flow, and Chute du Lion falls would be dewatered. Maximum flow velocity would occur at the downstream end of the canal, varying between 1.4 and 2.4 m/s for all of the flow scenarios that precede the spawning period. With the maximum usable flow from the generating station, the canal would overflow and some water would flow into the current riverbed and possibly cause a small amount, i.e. 15 m³/s, to spill into Chute du Lion falls.

Aquatic Fauna

The spawning grounds found in this section of the Gatineau allow many species to reproduce and are therefore vital to the stability of the fish populations of the river and of Bitobi lake. It is likely that these spawning habitats also contribute to populating the areas downstream of the river.

The canal would dewater spawning ground no. 1 located downstream from the dam, causing a loss of nearly 10,000 m² of spawning habitat, i.e., over one half of all the spawning area of this section of the river. Another result would be a much smaller aquatic habitat area. Flow velocity in the canal would be too high (1.4 to 2.4 m/s) to allow the fish to migrate upstream to residual spawning grounds. Upstream migration would be impossible due to the dewatering of some sections of the natural riverbed.

The loss of this spawning habitat would alter much of the fish population and may lead to a decline of some species or to their disappearance. The project is therefore judged to be of high impact. To mitigate this impact and promote the stability of habitat quality for aquatic fauna in this river section, the spawning grounds will be

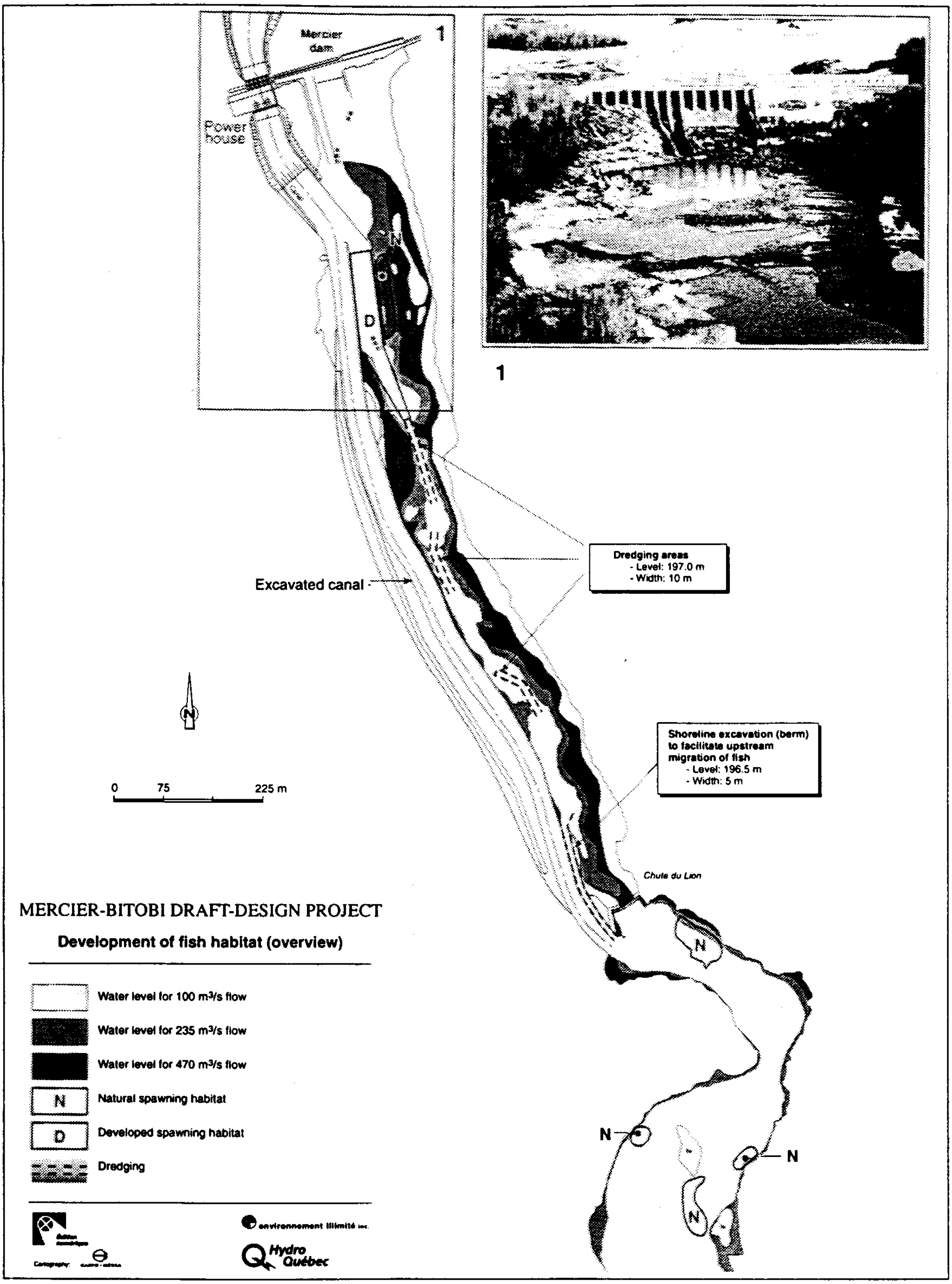


Figure 3: Overview of developed fish habitat in the Mercier–Chute du Lion section of the Gatineau River

relocated near the dam, the river's residual bed dredged to promote flow velocity, and a shallow berm built in the canal's downstream section to allow the fish to travel upstream.

DEVELOPMENT DESIGN

Various design criteria have been established in order to define different development designs that would preserve the aquatic habitats. The preferred solutions were proven development designs that were simple from an engineering as well as from a biological standpoint.

The development designs were optimized to take the following into account:

- Walleye reproduction, an abundant species prized by sport fishermen. Other species seeking stream habitats for spawning, such as suckers, coregonids and lake sturgeon, will also find appropriate spawning areas. Spawning habitat enhancement should replicate optimal depth conditions (0.5 to 1.5 m) and velocities (0.5 to 1.0 m/s) for walleye spawning. These conditions were determined on the basis of features noted in current spawning grounds and the main spawning grounds studied in the Baskatong reservoir (Gendron, 1995a,b). The conditions are also found among the parameters generally recognized for this species (Hazel and Fortin, 1986);
- The recreation of a quality spawning habitat that has the same physical characteristics of the current site and is located nearby;
- All reproduction-related physical and biological characteristics, i.e., upstream migration by spawners, spawning, egg development and the survival of fry;
- Constraints stemming from reservoir management and turbine flow and discharge;
- Site morphology and materials available during the construction phase.

Design and Optimization of the Proposed Development

The energy output simulations for the Gatineau River drainage basin and the operation of the proposed Mercier generating station were done using the SIMHYDE model (Technik Eaucan inc., 1995; Kang and Guerrero, 1995). These simulations allowed forecasts to be established for the weekly variations in Baskatong reservoir water levels as well as the turbine flow and discharge into the Gatineau River. The model was programmed to take into account the current operating and environmental constraints. These consist of maintaining the reservoir at a level under 222.14 m during the summer and fall and guaranteeing a minimum compensation flow from April 23 to May 28 each year in order to provide a cover of water for spawning grounds and adequate hydraulic conditions during the spawning period.

The spawning habitat was designed for the range of flows simulated by the SIMHYDE model during the spawning period. Weekly operating conditions were repeatedly simulated using different downstream flow scenarios in order to find an adequate solution. In one case, the compensation flow was 50 m^3/s while in another it was 150 m^3/s.

The HEC2 backwater curve was used to evaluate flow control and distribution between the canal and the natural riverbed, as well as flow velocity, water level and depth in order to optimize the development design according to the anticipated flow scenarios. The river and canal were modelled separately since the flow only meets up at the upstream and downstream ends of the canal. The flow conditions were simulated for the typical flow scenarios during the spawning period as well as for flood conditions in order to provide stability in the rockfill (Tecsult, 1995).

The downstream part of the tailrace canal was initially intended to be 20 m wide. However, flow velocity at this width was too great and hence did not favor the upstream migration of fish. A wider design of the canal (30 m) in its downstream portion (150 m) resulted in a slight increase (0.13 m) in the head. The widening of the canal not only yielded an increase in energy output but also led to a decrease in flow velocity, thus facilitating the upstream migration of the fish.

The development was finally designed to take into account a minimum turbine flow of 100 m^3/s. This allows the reservoir management constraints to be met while ensuring that there is sufficient water for the spawning grounds.

Spawning grounds

The siting of the spawning area was determined based on the proximity of the existing spawning ground as well as the possibility of using the residual riverbed as a lateral canal in which the flow may be partly channelled. Locating the development in a corridor that is separate to a certain extent from the main channel limits the effects of major variations in flow. As part of the enhancements made to the spawning grounds, additional excavation work was done in the riverbed immediately to the left of the canal. The excavations will be 30 m wide and 190 m long, with a depth level ranging from 197.25 to 198.0 m. The total surface area of the spawning habitat is 6,500 m^2 (Figure 4). The variable level of the spawning area will ensure that the velocity and depth criteria are met for a whole range of flow likely to occur during the spawning period. The spawning area riverbed will be covered by 0.5-m-deep cobble 0.1-0.3 m in diameter. Boulders 1 m in diameter and submerged islands will be placed around the spawning area to create diversity in the habitat and the hydraulic conditions. These materials will become stable at a flow velocity of 2.2 m/s, generated by a millenial flow assessed at 2,720 m^3/s. The hydraulic characteristics of this section of the river are found in Table 3.

Table 3: Estimated hydraulic characteristics for the developed spawning ground and main canal

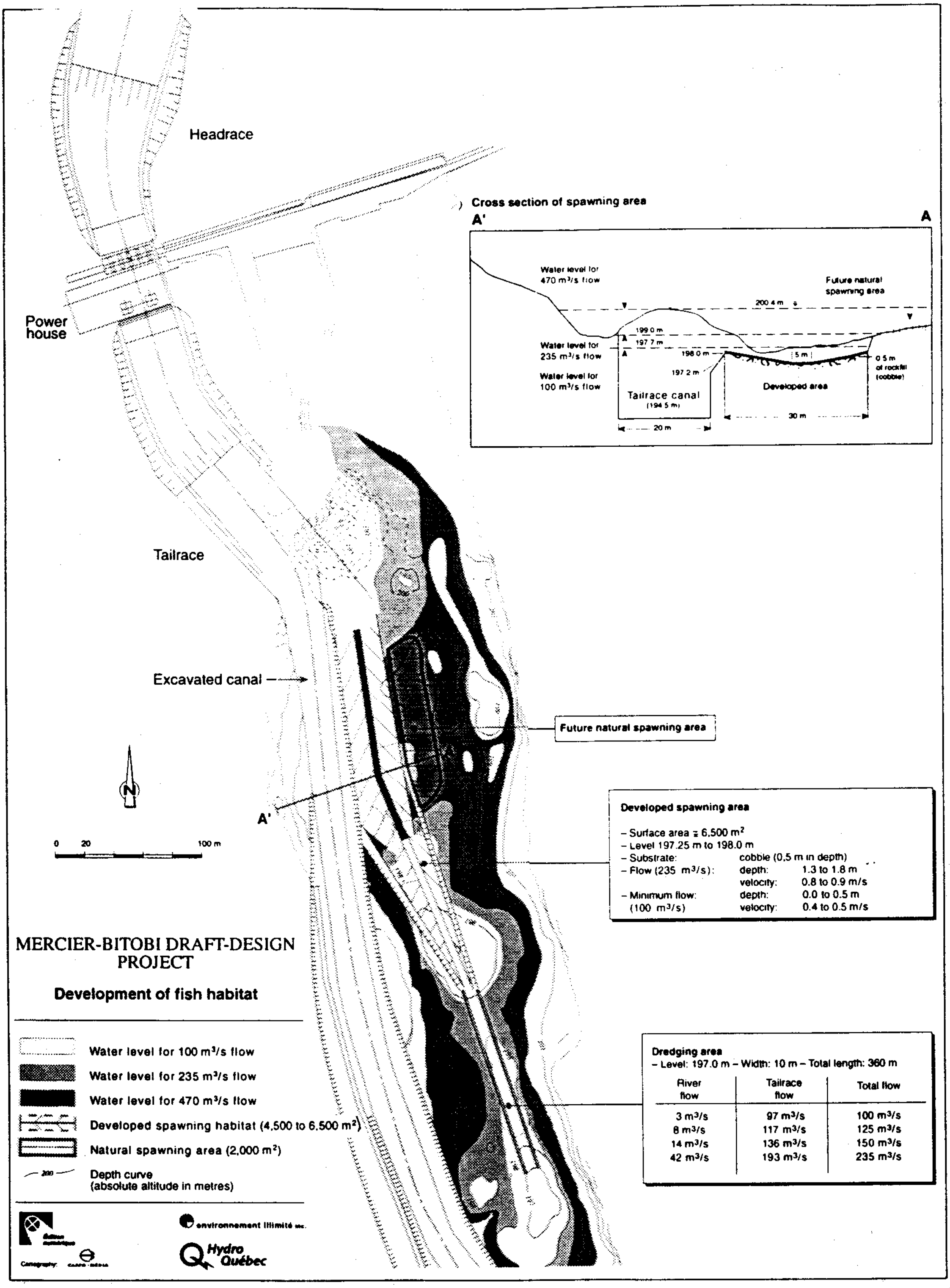
Total flow (m3/s)	Level (m)	Spawning ground			Canal		
		Flow (m^3/s)	Velocity (m/s)	Depth (m)	Flow (m^3/s)	Velocity (m/s)	Depth (m)
100	197.7 – 197.8	3	0.4 – 0.5	0.0 – 0.5	97	1.4 – 1.5	3.3 – 3.4
125	198.0 – 198.1	8	0.5 – 0.6	0.3 – 0.9	117	1.5 – 1.6	3.5 – 3.6
150	198.3 – 198.4	14	0.6 – 0.7	0.5 – 1.1	136	1.6 – 1.7	3.8 – 3.9
235	199.0 – 199.1	42	0.8 – 0.9	1.3 – 1.8	193	1.8 – 1.9	4.5 – 4.6
470	200.3 – 200.4	125	1.0 – 1.1	2.2 – 2.7	345	2.2 – 2.4	5.5 – 5.6

Dredging of the residual riverbed

Certain shoals will have to be dredged in order to provide maximum flow to the spawning ground and in the residual riverbed for all of the possible flow during the spawning period. Dredging will be carried out in the middle part of the river in the overburden. The dredging level has been set at 197.0 m. The canals will be about 10 m wide and will each be from 40 to 70 m long for a total length of about 360 m (Figures 3 and 4).

Berm

The tailrace canal flow conditions are such that there is little cover from the current to allow the fish to cross this corridor. The studies carried out on the swimming performance of walleye reveal that the speed that can



Developed spawning area

- Surface area ≅ 6,500 m²
- Level 197.25 m to 198.0 m
- Substrate: cobble (0,5 m in depth)
- Flow (235 m³/s): depth: 1.3 to 1.8 m; velocity: 0.8 to 0.9 m/s
- Minimum flow: (100 m³/s) depth: 0.0 to 0.5 m; velocity: 0.4 to 0.5 m/s

Dredging area

- Level: 197.0 m – Width: 10 m – Total length: 360 m

River flow	Tailrace flow	Total flow
3 m³/s	97 m³/s	100 m³/s
8 m³/s	117 m³/s	125 m³/s
14 m³/s	136 m³/s	150 m³/s
42 m³/s	193 m³/s	235 m³/s

Figure 4: Detailed view of a developed walleye spawning habitat downstream of Mercier dam on the Gatineau River

be sustained by a specimen measuring about 400 mm is about 0.7 to 0.8 m/s over about 100 m for about ten minutes (Jones et al., 1974). The flow velocity in the canal can reach between 1.4 and 3.5 m/s for discharge ranging from 100 and 470 m^3/s and is relatively constant over the entire water column, which substantially exceeds the swimming capacity of the fish. In order to be able to swim upstream to the spawning grounds, the spawners must use the residual riverbed, which is characterized by covered areas and allows fish to swim through. To access the riverbed, the fish must travel over a distance of about 150 m in the downstream part of the canal. In this section of the river, the development of a 5-m-wide shallowberm (at level 196.5 m) is proposed for the left bank in order to facilitate the upstream migration of the fish (Figure 3). Additional excavation will be required while leaving the fill in place to ensure adequate cover from the current and allow the fish to cross this corridor. Additional boulders may be placed on top of the berm to maximize the number of rest spots. The estimated flow velocity in the canal and over the berm for various flow scenarios is shown in Table 4. For flows ranging from 100 to 235 m^3/s, the average flow velocity over the berm would be 0.3 to 0.9 m/s; these velocity rates will decrease substantially at greater depths between the covered areas. The downstream part of the berm will be widened to 20 m to facilitate access by the fish to the downstream portion of Chute du Lion falls. The spawners travelling upstream will probably swim along the left bank where the flow velocity is not as great and will thus have easy access to the residual section of the river.

Table 4: Estimated hydraulic characteristics for the developed berm and main canal

Total flow	**Berm**				**Canal**		
(m3/s)	**Flow (m^3/s)**	**Velocity (m/s)**	**Depth (m)**	**Level (m)**	**Flow (m^3/s)**	**Velocity (m/s)**	**Depth (m)**
100	0.5	0.3	0.4	196.9	99.5	1.4	2.36
125	1.2	0.4	0.6	197.1	124.0	1.6	2.56
150	2.0	0.6	0.7	197.1	148.0	1.8	2.74
235	5.5	0.9	1.3	197.8	229.0	2.4	3.26
470	19.0	1.1	2.3	198.8	451.0	3.5	4.33

Management protocol

The protocol consists of reservoir management guidelines aimed at optimizing the upstream migration and spawning of walleye. As of April 21 – or when the temperature of the water downstream of the dam reaches 4.0°C – it is recommended that a discharge of 10 m^3/s be maintained to favor the upstream migration of spawners. As of May 1 – or when the temperature of the water reaches 6.0°C – a minimum flow of 100 m^3/s should ideally be maintained. During the years characterized by favorable runoff conditions, a flow ranging from 100 m^3/s and 250 m^3/s should be maintained while limiting the daily flow variations under 50 m^3/s.

EFFICIENCY AND COST-EFFECTIVENESS OF THE DESIGN

The development of the spawning grounds downstream of the dam combined with the proposed flow management scenarios will favor a stream spawning habitat similar to the one that is currently found. The quality of the spawning habitat and its surface area will be ideal when the flow is between 150 and 235 m^3/s. During high-flow years (>470 m^3/s), the conditions favorable to spawning will be ideal in the natural residual spawning habitats.

We believe that the residual impact on fish populations and their spawning and feeding grounds will be slight despite the major transformations that will be made to the river section. Although there are few examples of spawning areas that have been developed for walleye under stream conditions, the success rate for this type of development is about 60% (Beak Consultants Limited, 1993). The knowledge acquired during the course of a follow-up study on a spawning habitat downstream of Rivière-des-Prairies generating station in Montréal was put to use in the habitat design (Gendron, 1988). This study was conducted over a three-year period and dealt with the use of the developed spawning habitat in the springtime by lake sturgeon, longnose sucker, white sucker, walleye and sauger (*Stizostedion canadense*).

The cost-benefit analysis reveals that the construction and operation of the spawning ground development will not adversely affect the project from a financial perspective. The cost of building the spawning habitat, canals and berm are negligible compared to the total cost of the project. The equipment and personnel will already be available at the site to carry out the work. The berm and canals will be excavated at the same time as the canal, while fill material will be used to create the spawning ground.

The widening of the downstream part of the tailrace canal from 20 to 30 m resulted in an average head gain of 0.13 m. This increase in head provides an economic gain of $0.81 million (1992) over 50 years. Maintaining a compensation flow of 150 m^3/s during the springtime spawning period would result in a shortfall of $0.52 million (1992) over 50 years. Since the design takes into account a flow of 100 m^3/s, the shortfall will be less than this value. Overall, the environmental impact assessment of the spawning ground development has allowed the project to be optimized while resulting in a financial gain of about $0.3 million (1992) over 50 years.

REFERENCES

Beak Consultants Limited. (1993). Evaluation of the Effectiveness of Man-Made Spawning and Rearing Habitat in Reservoirs and Streams. *Submitted to the Canadian Electrical Association.* 116 p.

Gendron, M. (1988). Rivière-des-Prairies. Suivi de l'aménagement du haut-fond, synthèse 1982-1988. *Produced by the Groupe de Recherche SEEEQ ltée for the service de Recherches en environnement et santé publique. vice-présidence Environnement. Hydro-Québec.* 95 p.

Gendron, M. (1995a). Étude de l'accessibilité printanière des frayères à Doré jaune dans les principaux tributaires du réservoir Baskatong. *Report produced by Environnement illimité inc. for Hydro-Québec. Service Études environnementales.* 49 p.

Gendron, M. (1995b). Évaluation des impacts sur les frayères en aval du barrage Mercier et élaboration de mesures d'atténuation. Avant-projet Mercier Bitobi – Étape 2. *Report produced by Environnement illimité inc. for Hydro-Québec. Service Études environnementales.* 30 p.

Hazel, P.-P. and R. Fortin. (1986). Le Doré jaune (*Stizostedion vitreum* Mitchill) au Québec – Biologie et gestion. *Report presented by Université du Québec à Montréal to the ministère du Loisir, de la Chasse et de la Pêche. Direction de la faune aquatique. Service des espèces d'eau fraîche.* 417 p.

HEC2 (1988). U. S. Army Corps of Engineers. *Hydrologic Center. California.*

Hydro-Québec (1995a). Mercier-Bitobi. Rapport synthèse des études environnementales de la phase 2 de l'avant-projet. Volume 1 – Comparaison des variantes et évaluation des impacts. *Vice-présidence Ingénierie et Services.* 167 p.

Hydro-Québec (1995b). Mercier-Bitobi. Rapport synthèse des études environnementales de la phase 2 de l'avant-projet. Volume 2 – Description du milieu. *Vice-présidence Ingénierie et Services.* 239 p.

Hydro-Québec. (1995c). Rapport synthèse des études techniques de la phase 2 de l'avant-projet. Volume 1. *Vice-présidence Ingénierie et Services. Direction Conception des installations.* 155 p.

Jones, D. R., J. W. Kiceniuk and O.S. Bamford. (1974). Evaluation of the Swimming Performances of Several Fish Species from the Mackenzie River. *J. Fish. Res. Bd. Canada.* 31 (10).

Kang, N. and A. Guerrero (1995). Avant-projet phase 2. Mercier-Bitobi. Système hydrique Gatineau. Rapport synthèse des études hydrologiques. Volume 1. *Hydro-Québec. Direction Ouvrages civils. Service Hydraulique.* 14320-RA-95-11. 41 p.

Technik Eaucan inc. (1995). Manuel de référence. Modèle SIMHYDE 2.67. Simulation Hydroénergétique. *Produced for Hydro-Québec.* 385 p.

Tecsult inc. (1995). Avant-projet Mercier – Bitobi. Évaluation des modifications des conditions hydrologiques et hydrodynamiques dans le réservoir Baskatong et la rivière Gatineau. *Hydro-Québec. Vice-présidence Ingénierie et Services. Service Études environnementales.* 50 p.

PROBLEMATICS OF SHORELINE HABITAT EROSION ALONG THE ST. LAWRENCE RIVER: MAGNITUDE AND POTENTIAL SOLUTIONS

Denis Lehoux
Environment Canada, Canadian Wildlife Service
1141, route de l'Église, Sainte-Foy
Québec, Canada
G1V 4H5

ABSTRACT

Research programs conducted in recent years by Environment Canada and various associates in the course of the St.Lawrence Action Plan have brought new light on the problematics of shore erosion along the St.Lawrence River, particularly in the river section between Cornwall and Quebec City. These studies indicated that approximately 400 km, or 25% of the 1 500 km of shoreline investigated, showed traces of erosion. More detailed analyses reveal that 50% of the 224 islands found in the Montreal-lac Saint-Pierre section, representing the equivalent of 250 km of shoreline, present shoreline receding rates ranging from low (<1m/year) to high (≈3m/year). It is estimated that almost 1500 ha of island shore habitats have been lost during the last 35 years, between Montreal and lac Saint-Pierre alone.

It is difficult to quantify the total habitat losses due to shore erosion between Cornwall and Quebec City, other than island shorelines, since no systematic studies have been conducted. However, these losses are thought to be in the range of 500 ha, for a total habitat loss of 2000 ha in the study area. Finally, erosion data for the sectors downstream from Quebec City, in the middle and lower estuaries, are not sufficient to provide for accurate estimates of habitat losses along that portion of the St.Lawrence. Several factors are responsible for shore erosion, among which are ship-induced waves, water level variations, wind-induced waves, ice, and river currents.

Considering the great biological and economical value of shore habitats and the major impacts caused by erosion, including the loss of important bird and wildlife habitats, it is becoming urgent to stabilize and restore the most highly eroded shores, using relatively soft techniques. For these reasons, Environment Canada, along with various partners, has conducted in recent years several pilot projects for shore protection, stabilization and enhancement. So far, several techniques have been tried, including gabions, branch mats and bundles, weirs, wooden cribs, revegetation of barren shores, and sedimentation parks. Costs of implementation vary greatly between techniques and may occasionally reach several hundred dollars per linear meter.

KEY-WORDS: St.Lawrence / Erosion / Impacts / Stabilization techniques / Wetland value/ Vegetation / Wooden cribs / Fencing / Sedimentation parks

INTRODUCTION

Anyone who has worked along the St.Lawrence River at any given time must have noticed the disastrous impacts of erosion along the shores of the St.Lawrence. However, nobody had yet attempted to document the magnitude of these impacts. Several research projects conducted at specific locations along the St.Lawrence river and its estuary had reported local impacts. For example, shore receding rates ranging from 1.5 to 4.6 meters/year during the period of 1967 to 1995 were reported for some sections of the upper Rivière-du-Loup marsh (Argus, 1995; Dionne, 1986). Receding rates ranging from 0.1 to 1.5 meter/year were measured at the Cap Tourmente marsh (middle estuary) over a 26 year period between 1964 and 1990 (CWS, unpublished data). D'Agnolo (1978) reported an erosion rate up to a few meters/year at certain locations between Montreal and Sorel. Finally, Pilon *et al.* (1980) have mapped in greater details the shore erosion occuring between Boucherville and Berthier and have classified shorelines according to the following categories: severe, moderate and weak erosion.

Studies conducted by Environment Canada (in collaboration with the Quebec Ministry of Transport, the James Bay Energy Corporation, Ducks Unlimited and Argus Consultants) in the course of the St.Lawrence Action Plan have for the first time, documented this problematics on a larger scale, covering the whole length of the St.Lawrence River and upper estuary, from Cornwall to Quebec City (Cap-Tourmente) (Figure 1). Data were recorded on the actual length of shoreline affected by erosion, the magnitude of erosion and the potential impacts on wildlife habitats. In addition, a few pilot projects were initiated to determine natural restoration techniques adequate for the St.Lawrence system. New techniques were tried in order to replace conventionnal shore stabilization techniques such as riprap, gabions and embankment works.

DESCRIPTION OF THE STUDY AREA

The St.Lawrence is among the largest rivers in the world, flowing over 3 060 km and draining a total watershed of 1 344 000 km^2; (excluding the Gulf of St.Lawrence) (Centre Saint-Laurent, 1991) (Figure 1). The river links the Great Lakes to the Atlantic Ocean through a river section, an estuary (upper, middle and lower) and a gulf. The river section extends from the outlet of the Great Lakes to lac Saint-Pierre and includes rapids, lakes and archipelagos; it is characterized by an alternance of pronounced narrow zones (rapids, rivers, deep channels) and wide shallow areas (lakes). The upper estuary covers approximately 160 km between lac Saint-Pierre and Quebec City; it is characterized by freshwater and tidal influence. The highest tides in the St.Lawrence (up to 6.2 m in mid-May) occur in this section. The middle estuary is 195 km long, from Île d'Orléans to the confluence with the Saguenay River. It presents a marked increase in width from a few kilometers to more than 20 km. The water is brackish and turbidity is high.

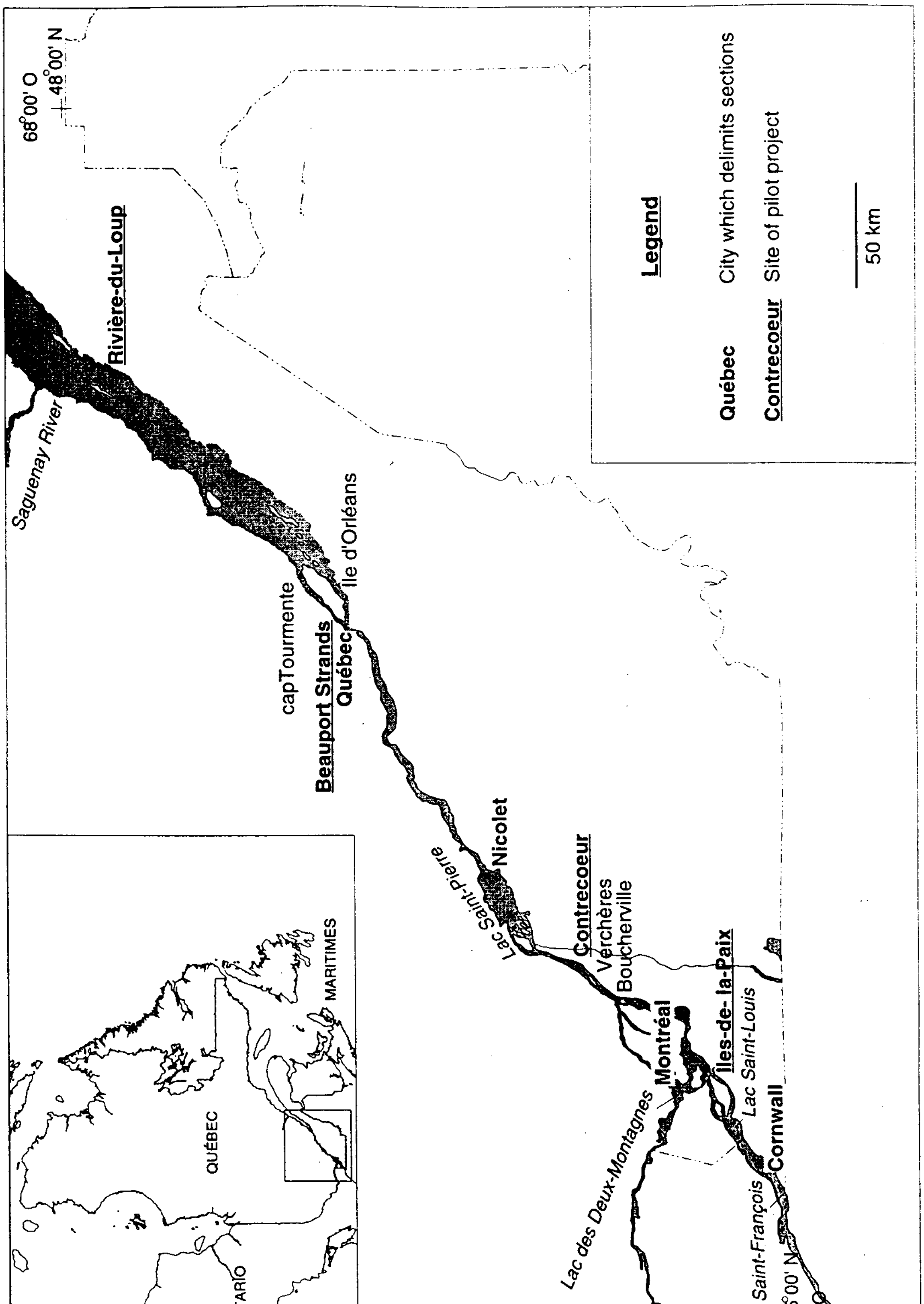


Figure 1. Location of the studied area.

METHODS

Erosion studies

Two series of studies were conducted in order to assess the total amount of shoreline affected by erosion both in the river section and in the upper estuary, and the magnitude of the phenomenon (annual shore receding rates). In the first project, the whole study area was surveyed by boats, covering approximately 1500 km of island and mainland shorelines. About fifty different items were potentially recorded at each sie, the most important being: location on a cadastral map; length of homogenous segments; shore status (natural or man-made); nature of soil deposit; heigth, slope and stability of the riverbank.

The study on shore receding rates was conducted only in the river section between Montreal and lac Saint-Pierre. It compared two sets of black and white aerial photographs taken in 1964 and 1983 respectively. The 1964 set were taken during the low-water period. However the 1983 set were shot between the end of May and mid-July. The water level was still very high in May; therefore the determination of shore limits and the assessment of erosion rates were somehow more difficult.

Pilot projects for shore stabilization

Several pilot projects for shore stabilization were conducted, four of which are of greater interest. In the Îles de la Paix National Wildlife Preserve, in the Montreal area, a mixed approach using gabions, branch mats and bundles was tested in order to find an economical and ecological solution to a major problem of erosion threatening, at mid-term, the actual existence of the archipelago. In the Contrecoeur National Wildlife Preserve, about 25 km east of Montreal, wooden cribs were tested on small islands protecting several hectares of marshes, prime habitats for fish and waterfowl. In the Beauport area, immediately downstream from Quebec City, revegetation techniques were tried in the hope of bringing back productivity to a degraded environment. Finally, in Rivière-du-Loup where an important spartina marsh was totally eroded away, sedimentation parks, adapted from Dutch and German techniques were tested.

RESULTS

A surprising assessment

Shore erosion along the St.Lawrence is impressive. Twenty-five percent, nearly 400 km, of the total 1500 km of shoreline in the study area, are under active erosion; 450 km (30%) have retained natural conditions and are stable; and almost 700 km (45%) are man-made shores relatively stable (Table 1). The magnitude of erosion is far from uniform. The effects of erosion are more profound in the river section between Montreal and lac Saint-Pierre, where 50% (≈ 270 km) of the shoreline is affected. Twenty percent (≈ 90 km) of the river section between lac Saint-Pierre and Quebec City, and 10% (≈ 40 km) of the section above Montreal are also affected.

Table 1. Status of shorelines along the St.Lawrence river between Cornwall and Quebec.

Section	Location	Length (km)	Natural				Man-made			
			Stable		Unstable		Stable		Unstable	
			Length	%	Length	%	Length	%	Length	%
Cornwall-Montréal (Boucherville)	Shores	318	35	11	10	3	265	83	8	3
	Islands	157	23	15	29	19	99	62	6	4
	Total	475	58	12	39	8	364	77	14	3
Montréal-Nicolet	Shores	214	86	40	36	17	92	43	<1	<1
	Islands	360	103	29	232	64	17	5	8	2
	Total	574	189	33	268	47	10	19	8	1
Nicolet-Québec (cap Tourmente)	Shores	390	171	44	51	13	165	42	3	<1
	Islands	93	34	38	38	40	21	22	0	0
	Total	483	205	42	89	18	186	39	3	<1
Synthèse	Shores	922	292	32	97	11	522	56	11	1
	Islands	610	160	26	299	49	137	23	14	2
	Total	1532	452	30	396	26	659	43	25	2

The importance of the erosion recorded in the mid-section of the study area is explained in large part by the fact that this section is narrow, often less than three kilometers wide, and is characterized by the presence of several archipelagoes. The St.Lawrence is considered a major shipping lane; it is estimated that more than 15 000 ship passages occur each year on the river. Several of the ships recorded travel to ports near Montreal and up to the Great-Lakes, with tonnage occasionally reaching 80 000 tons. Because of its narrowness, the mid-section is more sensitive to ship traffic. All the shores located within 610 meters of the shipping lane (20% of all shoreline in the mid-section) are likely to be affected by frequent ship-induced waves (D'Agnolo, 1978). Also there are two to four times more islands in the mid-section than in the upstream or the downstream sections (360 km versus 157 and 93 km, respectively). Erosion of island shores is very high and since most of these islands are uninhabited, riverside residents are less concerned by this phenomenon. Thus they are less prone to initiate stabilization works, which are often expensive, on lands mostly devoted to wildlife or farming.

Multiple causes of erosion

Among, the several other factors potentially inducing shore erosion, the most important are: water level variations, wind-induced waves, ice and river currents. During the period between 1965 and 1974, water levels rose to spectacular heights, inducing a major destabilization of riverbanks. In fact, these level variations induced serious changes in shore vegetation terracing; as a consequence, the actual nature of the terrain was also modified drastically. This phenomenon, along with man-made clearings along the shores, have made the shoreline more vulnerable to erosion. This factor plays an important role, particularly along lac Saint-Louis.

Wind-induced waves are a natural phenomenon. The height of these waves depends mostly on wind speed, fetch and water depth near the riverbank. The impacts of waves will be greater when the winds are strong, the fetch is long, the river is deep and the shore materials are crumbly. Wind-induced waves are the main cause of shore erosion along the wide river lakes in the study area: lac Saint-François, lac Saint-Louis and lac Saint-Pierre. Ice usually forms in mid-December and remains until early April on the St.Lawrence river. Shores are subjected to the ice grinding actions in winter and, mostly, during the spring break-up. Usually the impact of ice is limited or influenced by exposure of the riverbank to long fetch and to zones of strong currents.

Finally, the river current can erode crumbly materials such as clay, silt, sand and gravel. Under certain exceptional circumstances, this erosion process can also undermine the riverbanks to the point where large amounts of materials cave in and are carried away by currents. The erosion power of river currents is directly dependant upon the hydrodynamics of the river, the morphology of the riverbanks, the depth profile of the near-shore slope, the nature of the substrate, and the presence of emerging or submerged aquatic vegetation near the shores. It is mostly in the narrow sections of the river that the currents can erode and carry large amounts of material, over varying distances.

On Several Wildlife Species

Several islands within the St.Lawrence River are considered prime waterfowl nesting habitats, particularly for dabbling ducks. Bélanger (1989) estimated the mean nest density per hectare at 1.3 on the islands between lac des Deux-Montagnes and Trois-Rivières. Thus, it can be assumed that a loss of 1 500 ha of island shore habitats or 40 ha/year over the past 35 years prevented the building of almost 40 000 nests. Each nest built usually produces approximately 1.2 fledgling (taking into account several mortality factors such as predation at nest sites, natural mortality of chicks, etc); therefore, in theory, erosion would have prevented the production of 48 000 ducks in the Montreal-Sorel river stretch during the last 35 years.

Several islands provide protection to river marsh habitats from the negative effects of waves and currents. For example, the îles de la Paix provide a protecting barrier to some 240 ha of emerging and submerged river marsh (De Repentigny, 1994). This is also the case in the Contrecoeur archipelago where islets provide shelter and protection to about 135 ha of river marsh (Dryade, 1980). These marshes are important brood rearing sites for waterfowl and spawning sites for several species of fish (Giroux and Picard, 1993; Shooner, 1989). The marshes in the îles de la Paix archipelago are spawning sites for yellow perch; in the Contrecoeur archipelago, they provide spawning sites to northern pike, brown bullhead, yellow perch, pumpkinseed and black crappie. Erosion may induce the mid-term disappearance of these spawning sites.

On some islands, rare or endangered species are found. Thus, ten species designated threatened or vulnerable have been collected in the îles de la Paix archipelago (De Repentigny, 1994). These species may also disappear from this area in the mid-term. Additionnal impacts are also expected on all the other species associated with shore habitats, such as muskrats, frogs, turtles, songbirds, etc.

Promising Restoration Techniques

Considering the great biological and economical value of shore habitats, and the major impacts caused by shore erosion, it is becoming urgent to stabilize and restore the shore habitats affected by erosion using soft techniques which preserve global productivity. This was the basis for a research program initiated in 1990 by Environment Canada and various partners in order to test and eventually recommand restoration and protection techniques, appropriate to the St.Lawrence system. Some of the pilot projects conducted are (Figure 2):

Îles De La Paix

The Iles de la Paix National Wildlife Preserve is located in lac Saint-Louis, in the Montreal area. It consists of a dozen islands totaling nearly 50 ha over a distance of about 10 km. A rise of water levels in this sector since the ealry 70s, plus the effects of dominant winds and of smallcraft-induced wave action, have resulted in a marked erosion and a mean shore receding rate of 2.2 ha/year; the natural shores of the îles de la Paix have yet to reach a new stabilization and equilibrium slope. The total length of eroding shoreline is about 8.5 km (Argus, 1995). Assuming that the erosion rate will maintain the same rythm over the coming years, the îles de la Paix could well disappear totally within about 20 years; this would, in turn, induced the disappearence of all aquatic vegetation currently under the leeway of the islands.

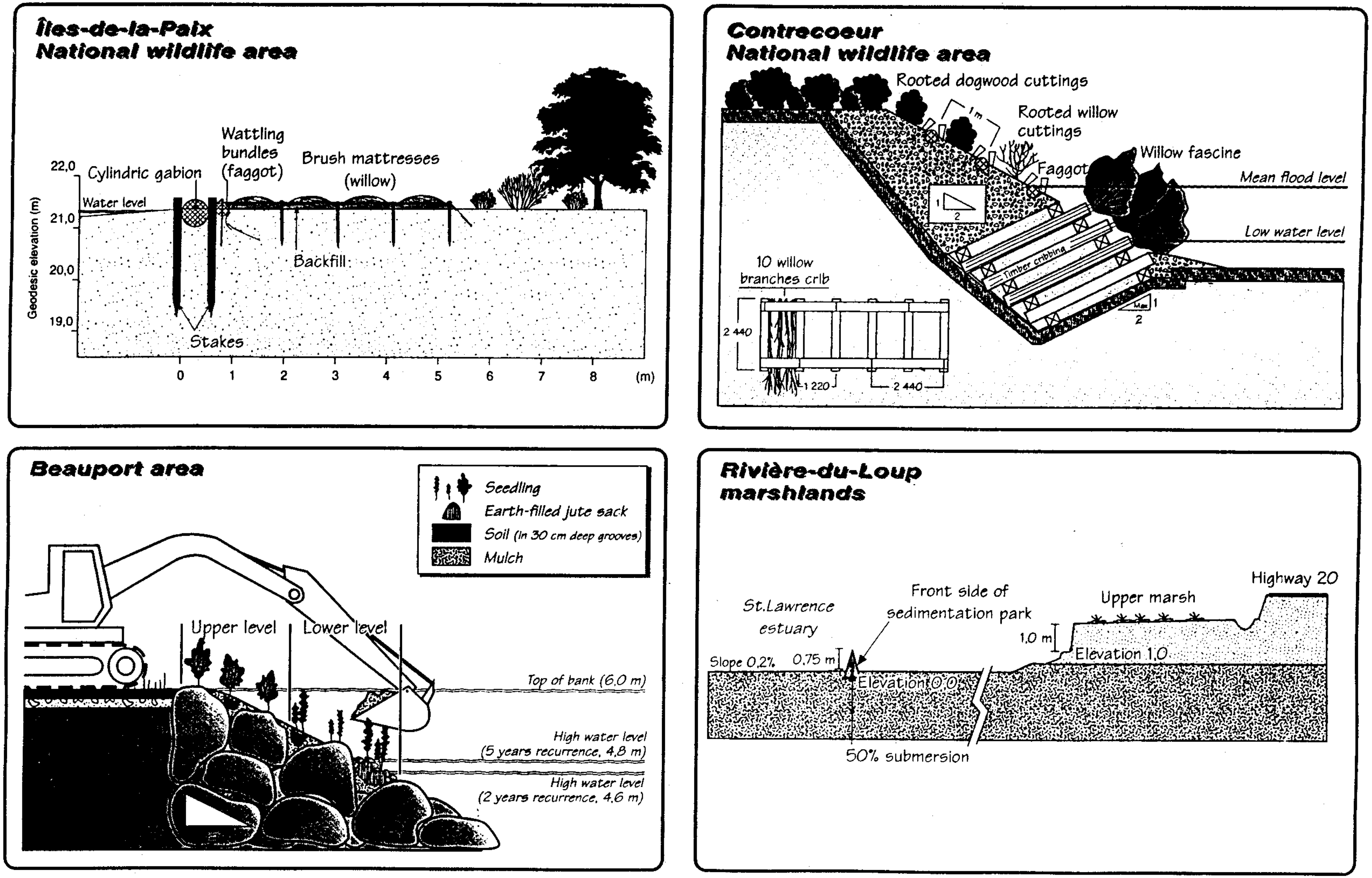


Figure 2 Examples of shore restoration techniques

In order to stabilize the eroding embankment and to protect valuable wildlife habitats, it could be possible to build protection structures made essentially of rocky material (retaining wall and pierhead). However since such structures imply the use of inert materials which do not create a productive and diversified ecosystem, we believe that a more ecological approach, better adapted to the environmental orientation of the archipelago, should be favoured. In this perspective, we have conducted a pilot project on a shore section in erosion (approximately 50 meters) in the fall of 1995. The eroded embankment was stabilized by a gabion, and branch mats and bundles. First, dead trees and strongly affected trees were felled to facilitate the stabilization operations. Then, the shore was reprofiled to have a slope of 1:10. A trench was dug at the base of the embankment and a gabion, filled with rocks varying from 100 to 150 mm in diameter, was set in the trench. Two rows of stakes, one on each side of the gabion, keep it in place. Two sets of bundles of dogwood and/or willow branches solidely tied together to create a uniform coil were used. The first set of bundles was placed at the very base of the slope to be stabilized, immediately above the gabion, inside a trench which depth was approximately 2/3 the thickness of a bundle. The second set was positionned in the highest portion of the embankment. Bundles are fixed in the trench by stakes and covered with earth. Finally, a branch mat was set in the mid-section of the embankment in order to stabilize the residual substrate. Branch mats are made of a branch arrangement deposited on the slope and maintained in place by wire-mesh. Willows (*Salix exigua* and *Salix eriocephala)* were used in our mats.

Two years of monitoring will be needed to verify the stability of the structure. If this technique produces the desired results, it could be possible to use it integrally or partially, depending upon the extent of erosion, on approximately six kilometers of shoreline at an estimated cost of about 2.5 million dollars.

Contrecoeur

The Contrecoeur Islands National Wildlife Preserve covers an area of 207 ha. It is located about 25 km east of Montreal. It is made of 27 islands distributed in two strings parallel to the river shores (Lehoux et Grenier, 1995). The first string of islands, located near the shipping lane and surrounded by vegetation beds, constitutes a genuine protection barrier against wave action. Blocked away, waves cannot reach the second string of smaller islands located behind. Most of the barrier islands are man-made, being the deposit sites of dredging material from the construction of the St.Lawrence Seaway. Not being protected, the dredged materials erode progessively under the action of wind-induced and ship-induced waves, along with the effect of water level variations. In fact, several islands have been staved in by erosion and the original dredged materials have completely disappeared. All of the remaining barrier islands are also affected by erosion and their current annual receding rates suggest that they may well disappear at mid-term. Their disappearance would automatically result in the decline of aquatic vegetation and the progressive erosion of the second string of islands (Argus, 1993).

Faced with the magnitude of the erosion problems observed in this National Wildlife Preserve, we have tested, through a pilot project, a stabilization technique which could eventually be used on a larger scale, not only in the Contrecoeur archipelago, but also in other areas along the St.Lawrence River. This technique, tested in 1991 on a 49 m segment of island shoreline, consists essentially in interweaving a few rows of wooden cribs along the eroding embankment and filling the cribs with several branch bundles and live shrubs. More precisely, the technique requires excavations on the banks to set the cribs built of cedar logs. During the operation, the embankments were reprofiled to produce a slope of 1:2. The soil between cribs was stabilized by inserting a geotextile membrane, some bundles of willow branches collected on the site, and fine or coarse material.

A monitoring program conducted the following year revealed severe erosion occuring at the front and the base of the structure. This erosion results from the fact that the structure was built too high on the slope. By simply adding a row at the bottom, the magnitude of erosion would have been much less. Follow-ups also revealed that plant survival ranged from 23% to 61% depending on their location in the cribs; the plants in the lower levels showed a higher growth rate and were healthier. Mortality rates can be attributed to a large number of rodents on the site. With this technique, the cost of shore stabilization is in the order of $400/m^2.

The Beauport Strands

Lining the shores with stones remains one of the most commonly used stabilization techniques. Between Cornwall and Quebec City, more than 60% of the 650 km shoreline protected by man-made structures were lined with stones (excluding concrete walls). This technique provides an adequate physical stabilization of the riverbanks and a good protection against the various erosion factors. However, the substrate, often made of large blocks with or without slits in between, does not favour vegetation growth. The resulting shoreline has only little ecological and aesthetic value.

The project conducted in June 1992 on the Beauport Strands, immediately downstream from Quebec City, aimed at introducing vegetation on a 100 m stretch of stone-lined embankment in order to increase its ecological and aesthetic value, and to develop a technique which can be applied to most of the river shores lined with stones along the St.Lawrence (Argus, 1995). The experimental site is located near a highway. It is made of large grade granit rocks with a mean diameter of more than 70 cm. Interstices between rocks create large holes (mean size = 0,125 m^3), which contain no fine particles; these are totally washed away by leaching and by wave action. In the pilot projects, three types of mulch were tested to fill in the holes: a mulch of shredded evergreen bark, a mulch of pruned-branch shavings from deciduous trees and a mulch of shredded leaves. Eleven plant species, including willows, dogwoods, alders, wild roses, elders and wild grapes, were selected. Two types of seedling were planted: four to five month old rooted cuttings and two year old seedlings.

Alternating grid plantation was favoured over other methods. Distances between plants ranged from 75 to 120 cm. A compost rich in nutrients was added to accelerate growth rate.

Follow-ups conducted in subsequent years indicated that the bark mulch showed less subsidence than the other two mulches. Mean plant survival rates were 46% and 47% for the upper and lower levels, respectively. The three species with the best survival were the Canadian Elder (*Sambucus canadensis*) (no mortality), the Siberian Pea (*Caragana arborescens*) (85% survival) and the Sweet Gale (*Myrica gale*) (62% survival). Species with the highest mortality rates were the green alder (*Alnus crispa*), the rough rose (*Rosa rugosa*) and the sumach (*Rhus typhina*) (survival rate ranging from 29% à 31%). Survival rates for all species in the three mulches ranged from 33% in the shredded leaves, to 46% in the evergreen bark, to 52% in the branch shavings. Plant mortality was caused by frost, salt and flood tides. Costs of this restoration technique are estimated at 12$/plant.

Rivière-Du-Loup

The Rivière-du-Loup marsh, located approximately 125 km east of Quebec City, has been severely affected by erosion for the past fifteen years. Spartina marshes, considered as rare habitats in the province of Quebec, are threatened.

The solution proposed to reduce the erosion rate is inspired from a technique well-proven in Europe, but rarely used in Quebec, the establishment of sedimentation parks. Three parks of a total length of about 90 meters were experimented with. Results of this study are detailed in another paper presented to the symposium.

CONCLUSION

The projects conducted in the course of this study resulted in a more accurate assessment of the extent of shore erosion in the St.Lawrence river and estuary, and of the impacts created by erosion. It has become obvious that the magnitude of the problem and the major impacts created require that shore stabilization works must be undertaken in the coming years. If not, valuable wildlife habitats may vanish at mid-term. We estimate at 150 million dollars the amounts required to stabilize eroding embankments and to introduce vegetation on the stone linings already built within the study area. Considering the length of shoreline affected and the relatively high costs associated with the restoration and stabilization techniques, priorities must be established. The pilot projects on stabilization techniques conducted in recent years have identified, tested and described about a dozen techniques more environmentaly acceptable. It is hoped that these techniques will be favoured in future interventions on the St.Lawrence River.

REFERENCES

Argus (1995). Perspectives éco-technologiques de restauration des rives de la Paix. *Rapport présenté au Service canadien de la faune.* 43 p. + annexe.

Argus (1995). Restauration du marais intertidal de la baie de Rivière-du-Loup: suivi du projet pilote de parc de sédimentation. *Rapport présenté au Service canadien de la faune, au Ministère des transports du Québec et à Canards Illimités.* 19 p.

Argus (1995). Végétalisation de berges enrochées: projet pilote. Rapport de suivi 1995. *Rapport soumis au Service canadien de la faune, au Centre Saint-Laurent à la Société d'énergie de la baie James, au Ministère des transports du Québec et à Canards Illimités.* 25 p.

Argus (1993). Projet pilote d'ouvrage de protection contre l'érosion et de restauration des dépôts de matériaux dragués dans le secteur des îles de Contrecoeur (rapport synthèse). *Rapport présenté au Service canadien de la faune et à la Garde côtière canadienne.* 49 p. + annexes.

Argus (1991). L'érosion des îles du Saint-Laurent, tronçon Montréal-lac Saint Pierre. *Rapport soumis au Service canadien de la faune et à la Garde côtière canadienne.* 55 p. + annexes.

Bélanger, L. (1989). Potentiel des îles du Saint-Laurent dulcicole pour la sauvagine et plan de protection. *Environnement Canada, Service canadien de la faune.* 71 p.

Centre Saint-Laurent (1991). Un fleuve, des estuaires, un golfe: les grandes divisions hydrographiques du Saint-Laurent. *Atlas environnemental du Saint-Laurent.*

D'Agnolo, F. (1978). Étude d'érosion. Partie du fleuve Saint-Laurent entre Montréal et Sorel, 1974-1976. *Ministère des transports, Garde côtière canadienne, région des Laurentides.* 23 p.

De Repentigny, 1994. Potentiel biologique actuel de la RNF des îles de la Paix. *Service canadien de la faune, région du Québec.* 17 p.

Dionne, J. C. (1986). Érosion récente des marais intertidaux de l'estuaire du Saint-Laurent, Québec. *Géogr. Phys. Quatern.* 40(3) pp. 307-323.

Dryade (1980). Les habitats propices aux oiseaux migrateurs. *Rapport présenté au Service canadien de la faune, région du Québec.* 66 p. + annexes.

Giroux, J. F. and M. Picard (1993). Utilisation de l'habitat par les couvées à la Réserve nationale de faune des îles de Contrecoeur. *Rapport présenté au Service canadien de la faune.* 13 p. + annexes.

Lehoux, D. and C. Grenier (1995). Sommaire des informations concernant les îles de juridiction fédérale et propositions d'aménagement. *Environnement Canada, Service canadien de la faune.* 82 p.

Pilon, C., J. M. Boisvert, D. Carrière, J. Champagne, P. Chevalier, D. Le Quéré, V. Simard and G. Sylvain (1980). Les îles du Saint-Laurent, de Boucherville à Contrecoeur: environnement biophysique. *Centre de recherches écologiques de Montréal.* 292 p. + annexes.

Shooner, G. (1989). Localisation des sites de reproduction des principales espèces de poissons du fleuve Saint-Laurent (Cornwall-Montmagny). *Atlas cartographique présenté à Pêches et Océans Canada.* 16 cartes.

Bioengineering for shore protection

Techniques végétales de protection des berges

BIOENGINEERING TECHNIQUES TO PREVENT SHORE EROSION

Bernard Lachat
BIOTEC Biologie appliquée SA
Route de Courroux 17
CH-2824 Vicques (Switzerland)
Tel: ++41 6635 66 66*
Fax: ++41 6635 56 46*
E-mail: 100770.3301@ compuserve.com

ABSTRACT

Bioengineering techniques employ plant species to solve erosion problems, particularly along water streams. This paper reviews the natural structures of stream beds, banks and slopes; it then presents different aspects of bioengineering: concepts, methods, materials and expected results. The basis of bioengineering are explained and the parameters to consider for a successful technical and biological intervention are summarized. Man-made regulation and protection works on a stream are not always desirable; a zero intervention alternative must be considered before any action is taken, following a natural logic where vegetation has priority on mineral intervention. Thus, this paper is more of a general overview of current knowledge and techniques in bioengineering, rather than a specific step-by-step method which is readily available in various publications. However, it is clearly demonstrated that plants are used as construction materials and not only as cosmetic elements to enhance the aesthetics of civil works! The concepts herein are based on a European context and may not apply integrally to other continents.

KEY-WORDS: Bioengineering / Erosion control / Bank protection / River works / Morphology / Stabilization / Slope stabilization / Root / Willow

FOREWORD

The objective of this paper is to provide information on the notion of bioengineering and to present possible developments in this domain to solve shore erosion problems or to restore streams with strongly affected morphology. The goal is not to produce specialists on these techniques; the fields of knowledge are too numerous and diversified, and several publications address these topics: Florinath (1982), Lachat (1994), Morgan and Rickson (1995), Regione Emilia-Romagna (1993), Schiechtl (1973) are all examples from various countries, in different languages.

Therefore, I will not discuss the pertinence of bioengineering techniques in a conceptual context. Several recent and on-going interventions, initiated all around the world in the past decades, more particularly in Europe, show the originality of the method and, looking back to appreciate the results, reveal ecological applications for more superior than conventional rock protection methods.

The examples and concepts in this paper are relevant to a European reality and may not actually apply to other continents. However, hydraulics and vegetation are universal realities; bioengineering techniques must therefore be transposable worldwide.

BACKGROUND INFORMATION

The first living cells appeared in the aquatic environment. Ever since, life has existed and evolved in this medium. An infinite variety of animal and plant life forms have remained in water over millions of years. Dependent upon particular climatic and morphological conditions, and various flow characteristics, various forms of life have evolved and developed together in close communities, the biocenosis, closely linked to the biotope or physical habitat.

The relationships between the various life forms and with their habitat form a functional unit, the ecosystem. When a simple disturbance of one component occurs, the whole system is affected. The "stream" ecosystem, comprised of water, stream bed, banks and slopes, provides a variety of habitats to a wide range of living organisms.

Stream morphology is modulated by weathering, according to the consistency of the terrains through which they flow; thus different streams have different configurations which give them each a different "personality". But streams are also shaped by man-made impacts. Often neglected, and often restored, streams in most industrialized countries are subjected to the assaults of modern technology, and pristine reaches are gradually disappearing.

Our actions on the environment are guided by immediate economic concerns. Our evolving lifestyle and philosophy, and our social behaviours have greatly affected a variety of natural habitats over the last fifty years; the consequence is an ever-increasing uniformity and lack of originality. Unfortunately, streams are part of the process.

THE BIOCENOSIS UNDER THE INFLUENCE OF STREAM MORPHOLOGY

Freshwater wildlife is widely diversified. The variety of organisms (mammals, birds, reptiles, batracians, fishes and almost every category of invertebrates) is largely superior than in other productive environments, such as lean grasslands and hedgerows. This should be a sufficient reason to consider a stream as more than just a channel through which liquid flows.

Unfortunately, man-made structures are continuously affecting this wildlife, gradually decreasing to the point of total disappearance in certain regions. The major sources of impacts are presented in Figure 1; three important groups of factors are identified (Lachat, 1991a):

- water quality (physical-chemical: chemical, organic, mineral, pollutions, etc.);
- water quantity (flow regime: artificial fluctuations, residual flows, etc.);
- stream bed and bank morphology (inadequate hydraulic works, etc.)

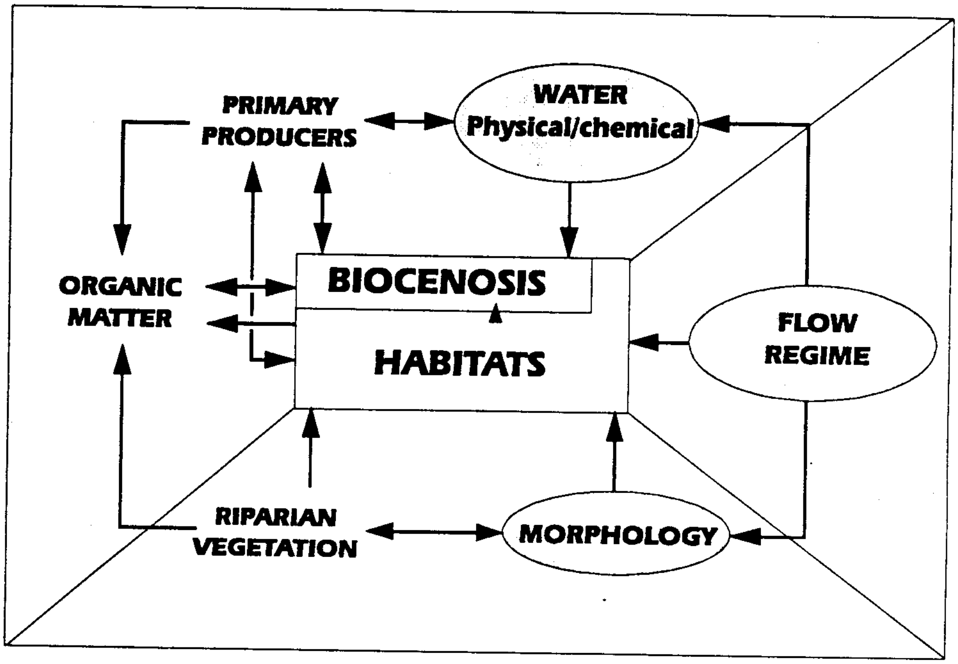


Figure 1: Factors having an influence on biocenosis of watercourse.

Regarding morphology, there remains a lot to be done to harmonize hydraulic and ecological constraints. Protection systems must be developed in the perspective of combining public safety and life survival.

BEWARE OF THE OPTICAL ILLUSION

Watercourse regulation is gradually evolving. In most cases, old rigid hydraulic principles are gently changed, often too lightly, in more rustic ensembles, which nevertheless are still pretty far from a natural watercourse: artificial turf or log protection walls replace semi-aquatic vegetation beds; boulders, sometimes cemented or concreted, sometimes percolated with earth, replace ancient walls or smooth ripraps. Unfortunately, rock protections are still masking soils which could be stable if proper natural set-ups were established. Although these works are sometimes a small improvement over previous situations, some of these "heavy" interventions are wrongly considered as ecological models, because they create one or two trout pools, or because vegetation manages to grow upon the structures. But a simple plant survey reveals, without any doubts, that they are biologically poor (Lachat, 1991b). Stream bed morphology and associated habitats are often simplified outrageously. This leads to the destruction of ecological balances and of functional processes; as a result, fish biomass may be significantly reduced, by as much as 80% (Wasson *et al*, 1995).

The essential characteristic of a natural watercourse resides in its complexity and diversity, at the physical, chemical and biological levels. But the biocenosis, including fish populations, cannot exist if the ecological requirements are not satisfied.

Therefore, a watercourse must present a mosaic of different physical elements to support complex biocenoses. Heterogeneity and diversity are morphologically essential to maintain and develop an ecosystem, even if it complexifies hydraulic calculations.

WATERCOURSE VEGETATION

Animals are not the only life forms in a watercourse. Plants have also invaded this environment in a special manner. With the exception of apical mountainous and alpine conditions, or at rock bed or crumble slope crossings, watercourses do not naturally have boulder structures. Normally, a site-specific vegetation will develop on particular substrates, from the center of the minor stream-bed to the edge of the major bed. Conditioned by stream-specific hydraulics and water quality, natural riparian vegetation follows a more or less well defined serial distribution (Figure 2), also influenced by subtle climatic parameters and by various pedological factors.

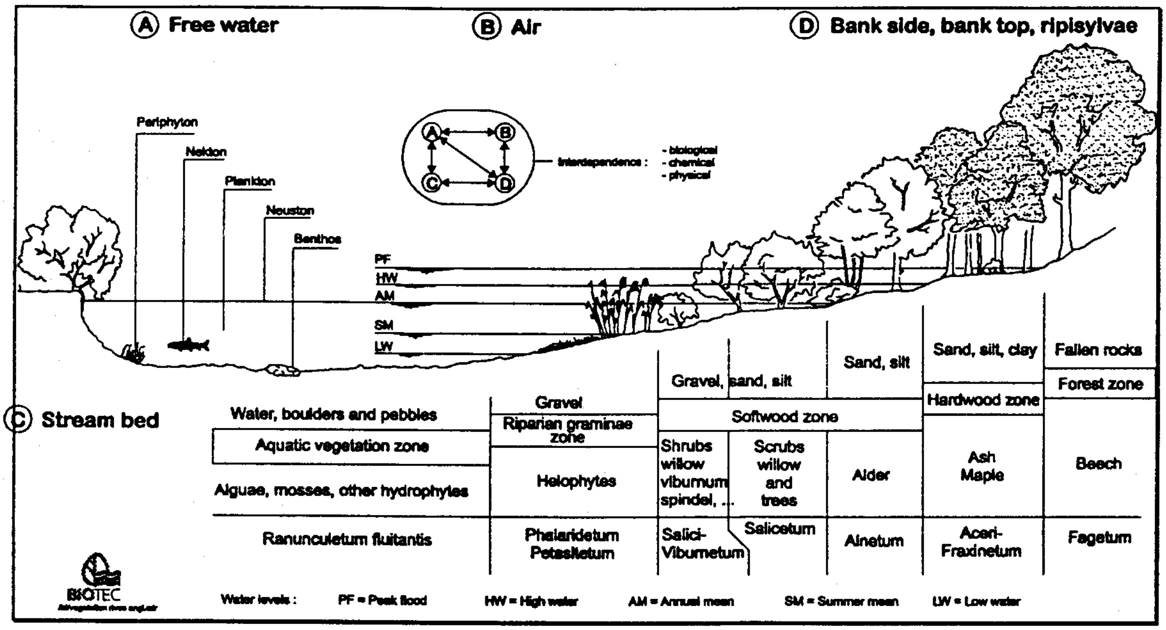


Figure 2: Serial distribution of riparian vegetation on a stream .
The River Doubs between France and Switzerland (after Lachat 1991a).

Because of a rather hostile physical environment, aquatic and riparian plants have developed (at least most species have) highly performing root systems, providing for great substrate stability. Also, several species have roots with hollows permitting gas circulation while in submerged anoxic conditions. Better yet, faced with serious environmental difficulties, most species have solved their problem of sexual reproduction by developing vegetative reproduction (cutting capacity). These are important attributes that will be used in bioengineering techniques.

For several decades, stream regulation (corrections, dyking, hydroelectrical dams, etc.) and the general state of eutrophication towards which a vast majority of streams have evolved have considerably modified the global vegetation picture.

For example, in Alsace, the riparian vegetation of the Rhine river has drastically changed while that of the nearby Ill river, in a more natural state, has remained largely unchanged. In Switzerland, alluvial zones are thoroughly investigated and concerns are raised about the disappearance of several plant species in the vegetation, when compared with surveys conducted less than twenty years ago, even if the number of species may have remained the same. Species quality is changing, reflecting a real genetic erosion. Again we observe a trend towards riparian vegetation uniformity under man-made influences.

VEGETAL ENGINEERING

Definition

Public work domains using inert materials are known as civil engineering.

Construction domains using live materials are part of various "engineering" categories.

Vegetal engineering consists in using plants or parts of plants to solve mechanical engineering problems related to erosion control, and soil stabilization and restoration. Unavoidably, biological aspects and functions must be considered. The remodeling of streams towards natural conditions or the restoration of original conditions exceeds the notion of vegetal engineering and is encompassed in the notion of biological engineering and ecological engineering (Figure 3).

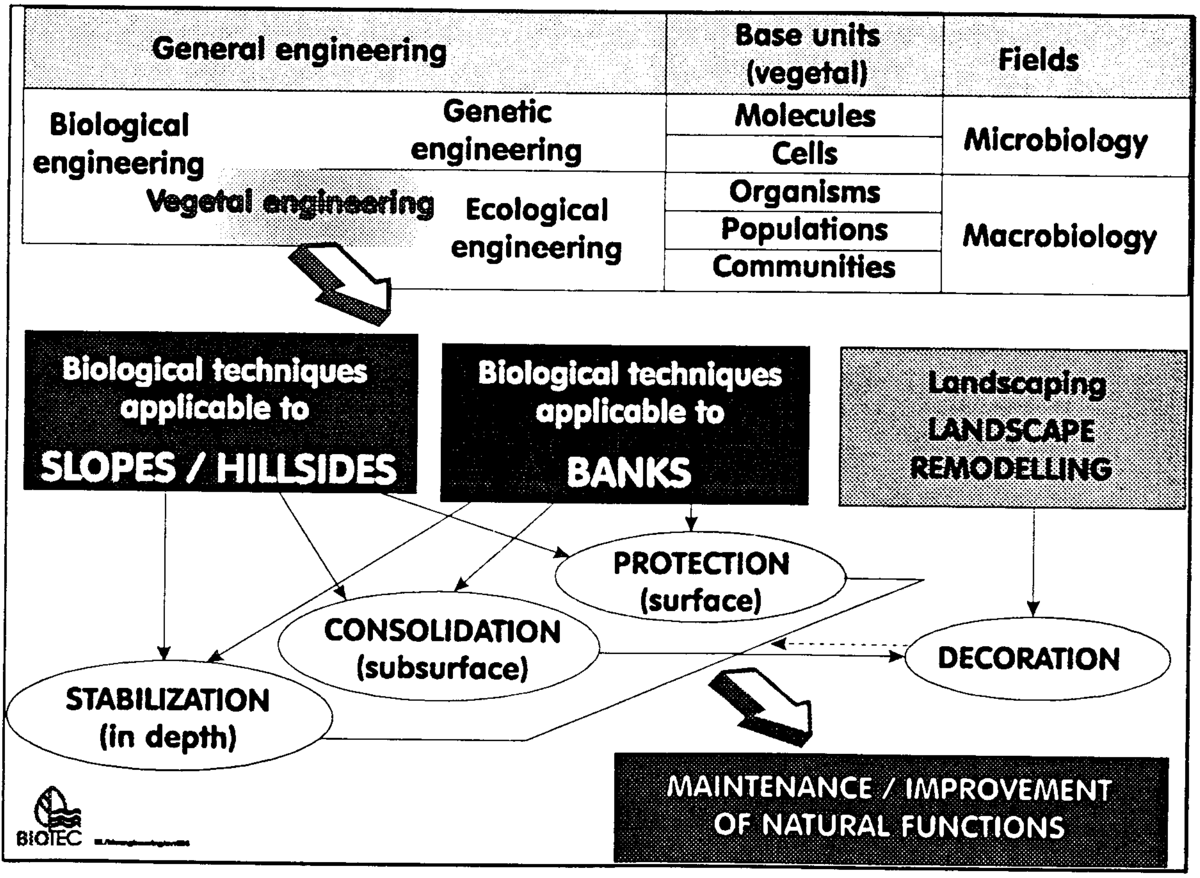


Figure 3: Fields and purposes of vegetal engineering.

According to the ecological requirements of plants, the various turfs of interventions are slopes (roads, trails, railroads, ravine, hillsides, etc.), banks (streams, ponds, lakes, etc.) and various zones to be naturally remodeled, as shown in Figure 3. Live plants will be used in different forms and will be implanted in specific order and sequences according to specific processes. Various methods have been described by different authors (Begemann and Schiechtl, 1986; Lachat, 1991a; Lachat, 1991b; Lachat, 1994; Lewis and Williams, 1984; Morgan and Rickson, 1995; Regione Emilia-Romagna, 1993; Schiechtl, 1973).

In fact, vegetal engineering is a hybrid science issued from different horizons. To be successful, it requires a wide range of multiple and varied insights. Botany, geobotany, phytosociology and plant ecology are the major inputs in terms of live components. But physics, soil mechanics and dynamics, auxiliary materials, geology, pedology and hydraulics, when we consider watercourses, are also essential tools for a successful application of bioengineering techniques.

Thus, based upon known and observed natural growth mechanisms, vegetal engineering develop processes which solve soil erosion control problems, sometimes at a large scale. It is precisely in the development of techniques and their applications that is the true essence of bioengineering. In other words, not only does vegetal engineering exploits the natural capacities of plants as a model, but it uses vegetation as the basic material for shore protection works.

Vegetal engineering techniques are very appealing because of their capabilities to offer very efficient, aesthetic and personalized solutions to a variety of specific problems. Generally, they have no impacts on a stream ecosystem, especially when compared with conventional civil works usually proposed. In certain cases, according to the project design and to the state of shore and stream bed degradation, they can improve the biological quality of the alluvial environment and the general morphology of the stream.

Vegetation being alive, it is, by defenition, subjected to fast ranoom fluctuations, capable of evolution and growth, and strongly influenced by environmental components. As a consequence, it is impossible to quantify vegetation in the same manner as inert materials. Nevertheless, in terms of shore protection with plants, the empirical approach based upon biological and hydraulic data is reliable, and likely one of the few valuable approaches.

Fashion or science?

Vegetal engineering is not new, although the term is very recent (Lachat, 1994). In fact, Leonardo da Vinci discussed the usefulness of willows along certain channels. During the 17th, 18th and 19th centuries, plants were commonly used to control landslides or shore erosion (Demontzery, 1878; Thiéry, 1891). Unfortunately, insights on plant physiological functions were insufficient to fully understand certain responses of plants to environmental events. Mechanical means to make the techniques more efficient were lacking. And as a result of two world wars, mechanization evolved rapidly and vegetal engineering was forgotten. It could have remained asleep for a long time, had it not been for biologists and forest engineers of various countries who tried to reassert the value of and improve these techniques (Lachat, 1991a; Lachat, 1994; Schiechtl, 1973).

Today, with new scientific insights in physiology and ecology, and with new materials and improved machinery such as geotextile and articulated hydraulic diggers, vegetal engineering is in full growth. Now, it is part of university curricula in several domains; there is even a chair of vegetal engineering at the University of Vienna, in Austria.

Why should we talk of construction and not only of vegetation?

The capacity to integrate a protection work in the natural landscape is a serious asset of vegetal engineering; but live material is not used only for decoration purposes.

Plants are mostly used for their biological holes, as highlighted by Maridet (1994) and for their biotechnical functions:

- absorption of mechanical constraints;
- soil stabilization via the root system;
- soil drainage by evapotranspiration and creation of hollows;
- protection against weather constraints (strong winds, excessive sunshine, snow slides, etc.).
- soil improvement in terms of humus content.

Vegetal engineering is often viewed (because of poor understanding or contempt) as strictly replanting vegetation. In fact, it is much more than that and much more than using plants as cosmetic or camouflage for traditional civil works. Thus, covering gabions and ripraps with willow cuttings or any other vegetation should not be considered actual bioengineering.

In reality, vegetal engineering techniques result in true structures meeting strict technical and mechanical criteria, in which live materials (stakes, branches, cuttings, twigs, turfs of heliophytes, etc.) are the foundations of the construction. This term is realistics since it implies combining and fixing to the banks a variety of materials, from different origins and of different quality, to create a live protection structure which is compact, cohesive and efficient against erosion immediately upon implementation.

Plants are not the only component of these structures; for improved efficiency, it is sometimes necessary to include inert (or non-living) materials. Generally, logs and certain natural fiber geotextiles (such as jute, coconut fiber, etc.) are favoured (Lachat, 1984; Lachat, 1986; Lachat, 1994; Schiechtl, 1973). Their main function is to increase resistance in the early stages of the structure, before the root systems have become performant.

Over the long term, this auxiliary materials will, for the most part, decay gradually and be replaced by growing vegetation.

If inert materials, such as boulders, are used, their function is to influence water flow and currents by building transverse structures such as thresholds and groins. Thus, it is hoped that they will last forever. But, in a vegetal engineering project, mineral materials are only used in complementary structures to bioengineering techniques and rarely constitute a full component of the main stabilization structure.

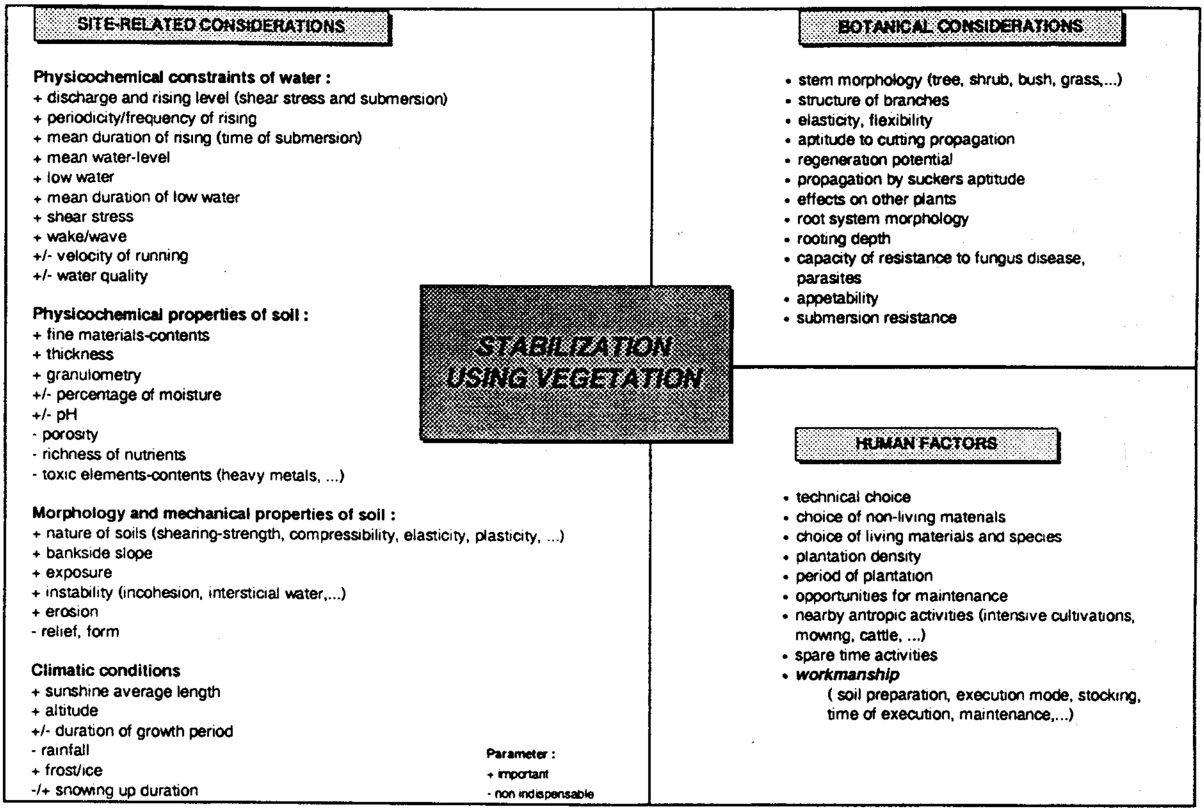
However, there are exceptions, particularly when a stream has been overly modified or when the stream banks are subjected to regular intense floodings, caused, for example, by tides. The normal tidal range, silting and salinity may be limiting factors to bioengineering techniques; mixed (or combined) structures are than required.

Site-specific adaptations

Revegetation, including plants used as foundations, must be preceded by an analysis of plant growth conditions at the site (Figures 4 and 5). This evaluation is generally conducted entirely in the field; a global approach is defined and essential data are collected. Further important information is acquired by laboratory analysis or by plain calculations.

Field experience and site overview are fundamental. It is useless to plant willows if nothing is known on the various species, or their edaphic requirements, or their growth succession related to water levels, etc.

In general, an important step is the production of a summary map to assess the current state and orient future works. However it is not essential, but a survey of local and neighbouring vegetation is unavoidable to detect and select appropriate species for active shore protection.



SITE-RELATED CONSIDERATIONS

Physicochemical constraints of water :
+ discharge and rising level (shear stress and submersion)
+ periodicity/frequency of rising
+ mean duration of rising (time of submersion)
+ mean water-level
+ low water
+ mean duration of low water
+ shear stress
+ wake/wave
+/- velocity of running
+/- water quality

Physicochemical properties of soil :
+ fine materials-contents
+ thickness
+ granulometry
+/- percentage of moisture
+/- pH
- porosity
- richness of nutrients
- toxic elements-contents (heavy metals, ...)

Morphology and mechanical properties of soil :
+ nature of soils (shearing-strength, compressibility, elasticity, plasticity, ...)
+ bankside slope
+ exposure
+ instability (incohesion, intersticial water,...)
+ erosion
- relief, form

Climatic conditions
+ sunshine average length
+ altitude
+/- duration of growth period
- rainfall
+ frost/ice
-/+ snowing up duration

Parameter :
+ important
- non indispensable

STABILIZATION USING VEGETATION

BOTANICAL CONSIDERATIONS

- stem morphology (tree, shrub, bush, grass,...)
- structure of branches
- elasticity, flexibility
- aptitude to cutting propagation
- regeneration potential
- propagation by suckers aptitude
- effects on other plants
- root system morphology
- rooting depth
- capacity of resistance to fungus disease, parasites
- appetability
- submersion resistance

HUMAN FACTORS

- technical choice
- choice of non-living materials
- choice of living materials and species
- plantation density
- period of plantation
- opportunities for maintenance
- nearby antropic activities (intensive cultivations, mowing, cattle, ...)
- spare time activities
- ***workmanship***
  (soil preparation, execution mode, stocking, time of execution, maintenance,...)

Figure 4: Considerations affecting the choice of a biological method and the choice of the plants.
(d'après Lachat, 1994)

Also, visual representations of objects affecting regular flow and causing erosion can be a precious tool to understand and develop revegetation (Figure 5).

Plant selection is determinant for the success of any intervention. As a rule, we should use native species, adapted to local growth conditions and to site-related conditions. Lists of plants are readily found in the literature. For streams with natural riparian vegetation, the best solution is to collect on-site specimens and to avoid exotic species. Then, the species selected offer the best protection tools available. If a category of woody species, for example willows, is missing because of the absence or the modification of riparian vegetation, nurseries may become necessary to provide sufficient material. Willow cuttings take readily and the production of stems is usually rapid and functional, if the right soils are used.

It is always recommended to vary species varieties for plants with vegetative root systems, or genus varieties when bare root plants are involved, not only for biological diversity, but also to increase revegetation potential and improve the efficiency of the structure. The respect of site-specific factors also means that species will be planted according to natural plant sequences related to water quality and hydraulic constraints. Basic knowledge in plant ecology is essential.

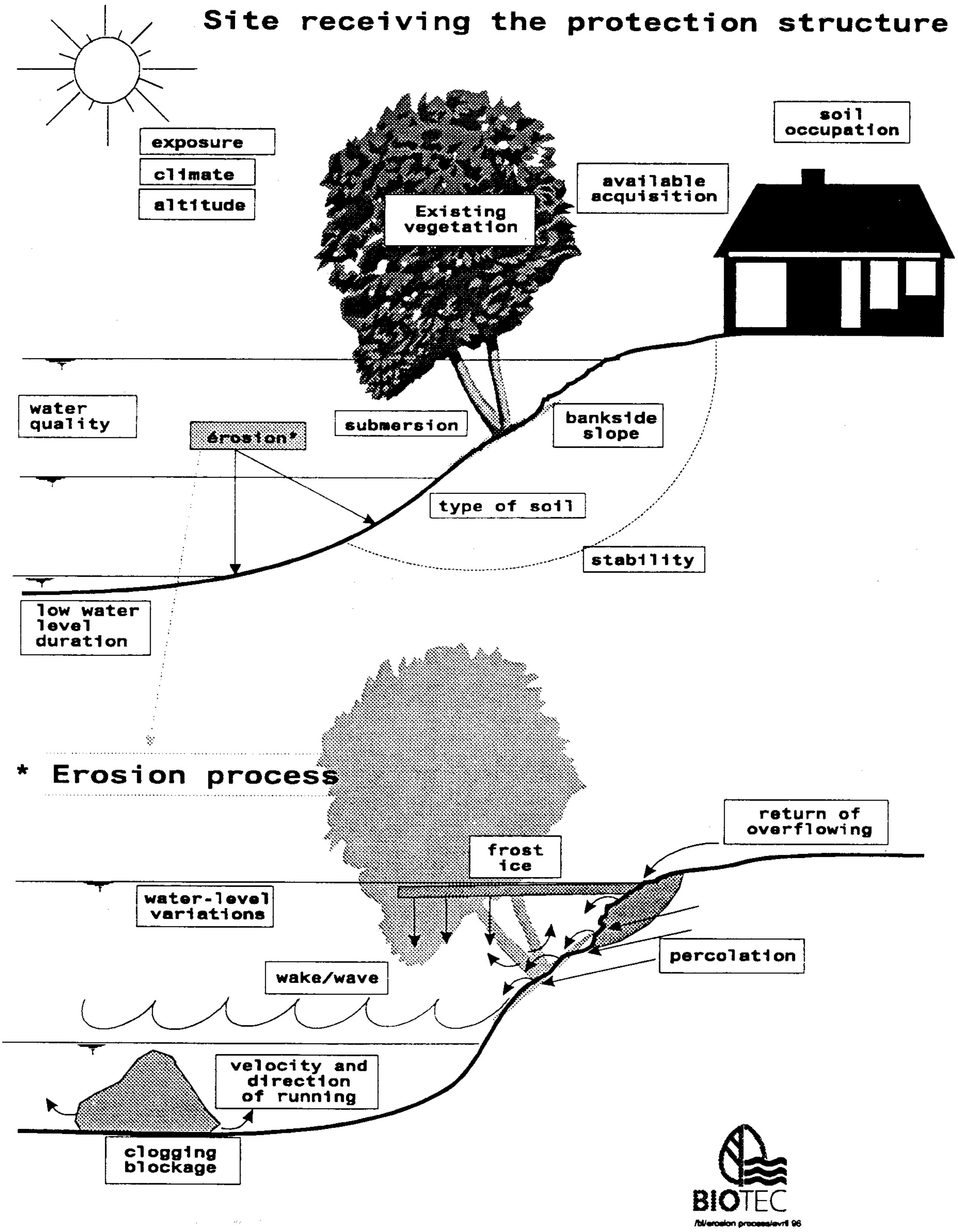


Figure 5: Erosion process and considerations affecting the site receiving the protection structure.

Selecting the right species must also consider the capabilities to be used as construction components, whether it be throwing out new shoots, suckering, plant flexibility, growth characteristics or future development.

A considerable improvement in efficiency from early practices allows bioengineering on large expanses of shoreline (Lachat, 1994). Vegetal engineering can be extended to various watercourse types, related to either hydraulic regime, flow facies, shore types, substrate roughness, or the origin of disturbance. In addition the re-opening or the restoration of a watercourse should involve appropriate vegetal engineering techniques.

New methods have also been developed and it is now possible to solve several specific cases.

The basic techniques of bioengineered structures are included in various categories according to several authors (see Lachat, Schiechtl op. cit., and Zeh *et al.*, 1990). The better known techniques are:

- cuttings;
- planting;
- weiring (single or double row of stakes made of willow branches or heliophytes);
- layers of shoot throwing branches;
- plants and sapling beds;
- combing;
- planted single or double-walled log cribs;
- log latticing (mostly for landslides).

Of course, each specific erosion case requires a personalized concept and control structure. In every project, a selection of banks or shores to be left untouched or to be improved upon, totally or in part, must be made. Depending on the local context, the local constraints and the true values to preserve, some eroding kanks may not need interventions.

ADVANTAGES, POTENTIALS AND LIMITS

What are the advantages?

Advantages provided by vegetal engineering can be summarized roughly as follows:

- growing stabilization efficiency;
- structure flexibility and resistance (anchoring rather than resting on the banks; pulling and sliding tensions counteracted by traction);
- hydromechanical actions of plants in soil: water captation, absorption and transpiration, increase in soil cohesion, bending effects, slowing of runoff speeds;
- relationships watercourse/undisturbed groundwater;
- minimised impacts of field work;
- improved water purification by roots;
- increased biodiversity and enhanced habitats (support, shelter, food);
- genetic and patrimonial reservoir;
- increase in water oxygen content and coolness (shade, oxygen release);
- decrease in turbidity and clogging;
- decrease in invading vegetation (algae...);
- preservation and enhancement of landscape, perfect integration in the neighbouring environment;
- educational asset;
- regular works: financial gains of 40 to 50% over conventional techniques;

- difficult works (highly modified stream and/or strong erosion): similar or slightly higher costs than conventional techniques, but:
  - much better ecological results;
  - living, therefore everlasting, structures;
- materials available on-site, simplified transportation (energy gains), no construction roads needed;
- solutions limiting legal procedures and claims (important financial advantages);
- possible product exploitation (creates a revenue).

What are the limits?

Despite all the above listed advantages of vegetal engineering techniques, there are some disadvantages and limiting factors (Lachat 1994; Zeh, 1990) directly related to plant growth:

- depending on the technique used, stabilization is not fully efficient upon structure achievement. However, the lower limits can be increased by a judicious use of geotextiles and by more sensible fixation methods;

- there are some limiting factors (altitude, regular strong pollution, lighting, soil hardness, salinity, tides, rocky substrate, steep slope and high energy level). In some extreme cases of erosion, bioengineering cannot be applied without financial and technical means similar to conventional civil works;

- in some cases, man-power requirements are important.

- a competent man-power and a serious scientific and technical supervision are essential. Specialized firms and qualified man-power are still rare: only ten firms exist in France;

- depending on stream size, regular maintenance at intervals may be needed;

- results are not always immediately apparent and may occur only after one period of vegètation.

For example, mountain streams present several limiting factors making bioengineering hard to apply: rocky substrate, altitudinal limit of plant species, torrential regime, steep stream slope and high energy.

FINAL REMARKS

Vegetal engineering is part of a wide range of techniques available to engineers to solve erosion problems. The use of live plants leads to extremely functional and durable structures. Plants are not cosmetic elements but true construction materials.

Regulating and landscaping a watercourse is not only a matter of hydraulic calculations. A knowledge of biological functions is needed to develop appropriate interventions. A fine analysis of the whole stream according to hydroecological criteria has become essential. For a successful intervention, the following guidelines should be followed:

- Stream bed, banks, slopes, pools and watershed form a whole which must be considered globally.

- The watercourse is a natural dynamic element from a chemical, physical and biological point of view; it must remain free of pollution sources and of disturbing artificial impassable obstacles.

- The watercourse requires space to maintain a natural behavior.

- The natural watercourse tends to show a maximum diversity of habitats and functional facies. We must preserve, restore or increase this diversity.
- Each watercourse has its own specific personality; interventions must be site-specific.
- On a watercourse, construction and maintenance works must respect natural wildlife and vegetation cycles.
- If technical measures must be taken to control damages caused by streams, hydroecological and biotechnological must be favoured. The philosophy governing this principle is based upon the following notions (Table 1):

Table1. Philosophy governing interventions to control shore erosion.

1) Assess the need for interventions according to impacts and issues (option zero)

2) Assess if vegetal engineering can solve the problems.

3) If not, assess if mixed interventions are efficient.

4) Only at this stage, should a conventional civil work technique be applied if it proves reasonable and coherent.

If the initial intention was to technically solve an erosion problem, then the "engineer" role is fulfilled. But for the biologist, the goal is not reached. As mentioned previously by several experts (Leuzinger and Lachat, 1995), the complete and successful achievement of a bioengineering intervention cannot be only viewed technically. Satisfaction will be complete only if the environment keeps or acquires superior ecological values than in the initial state, and superior values than gained form conventional techniques.

Bioengineering must solve technical problems with grace; but mostly, it must be a living and enhancing component of the natural environment.

The numerous financial, aesthetic, ecological, etc. advantages must now lead to an extended use of vegetal engineering techniques to control bank erosion. Successes obtained in Europe prove that these techniques are widely applicable on various stream types and in different regions. Solid insights in plant ecology and physiology, and in fluid mechanics are essential for successful interventions. But that is not sufficient; it is necessary to gain expertise in implementing the techniques.

Infatuation derived from spectacular results will certainly cause failures from people not completely mastering the necessary expertise. We want to believe that bioengineering will not suffer too much from these failures; indeed we could have abandoned, a long time ago, other more mineral techniques producing more costly failures.

Figures 6: LA TRAME A TAVANNES (CH)

Figure 6.a:
The bank prior to intervention. Erosion and landslides were threatening a road on the bank crest.
October 29, 1992.
(Photo : B. Lachat)

Figure 6.b:
Installation of the crib frame for earth, cuttings and plants.
November 26, 1992
(Photo : P.A. Frossard)

Figure 6.c:
When completed, the structure is compact and solid. Vegetation will grow and achieve the objectives.
June 17, 1993
(Photo : B. Lachat)

Figure 6.d:
Nature at work. This technique can also be used on large rivers.
August 19, 1994
(Photo : B. Lachat)

Figures 7: THE BIRSE RIVER AT SOYHIERES (CH)

Figure 7.a:
Digging the new river bed.
June 1, 1984
(Photo : B. Lachat)

Figure 7.b:
Bioengineering technique for shore protection, using three different geotextiles to counteract weather constraints and work delays. Protection is provided by fast action grasses. The river flows in its new bed with banks protected by grasses.
November 27, 1984
(Photo : B. Lachat)

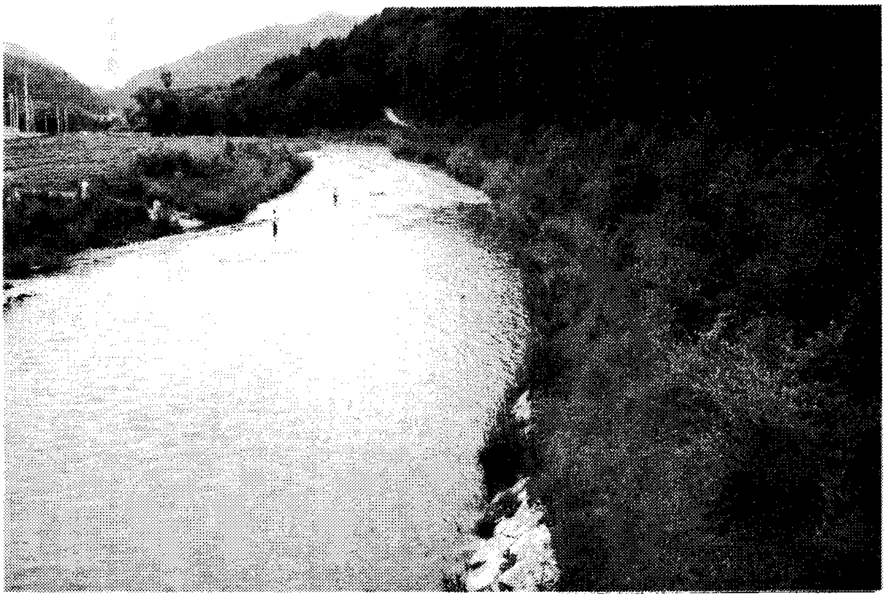


Figure 7.c:
Later shrubs were planted to provide general stabilization.
June 19, 1988
(Photo : B. Lachat)

Figure 7.d:
With a varied path, different bank types and vegetation cover, the river shows a natural aspect and offers true biological functions.
One of the goals of vegetal engineering is to camouflage completely man-made interventions.
May 31, 1994
(Photo : B. Lachat)

FIGURE 8: THE RHÔNE RIVER AT LYON (Fr)

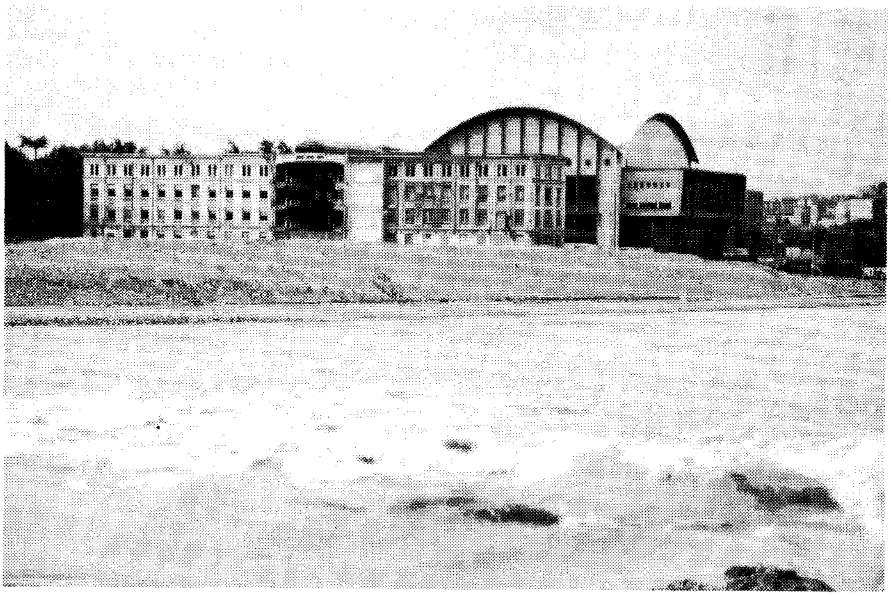


Figure 8.a:
The embankment prior to bioengineering protection.
September 16, 1993
(Photo : B. Lachat)

Figure 8.b:
Layering of branches of 5 species of willow, covered by a putrescible geotextile and fixed by willow stakes.
January 24, 1994
(Photo : B. Lachat)

Figure 8.c:
Work completed: vegetation serial distribution.
March 3, 1994
(Photo : B. Lachat)

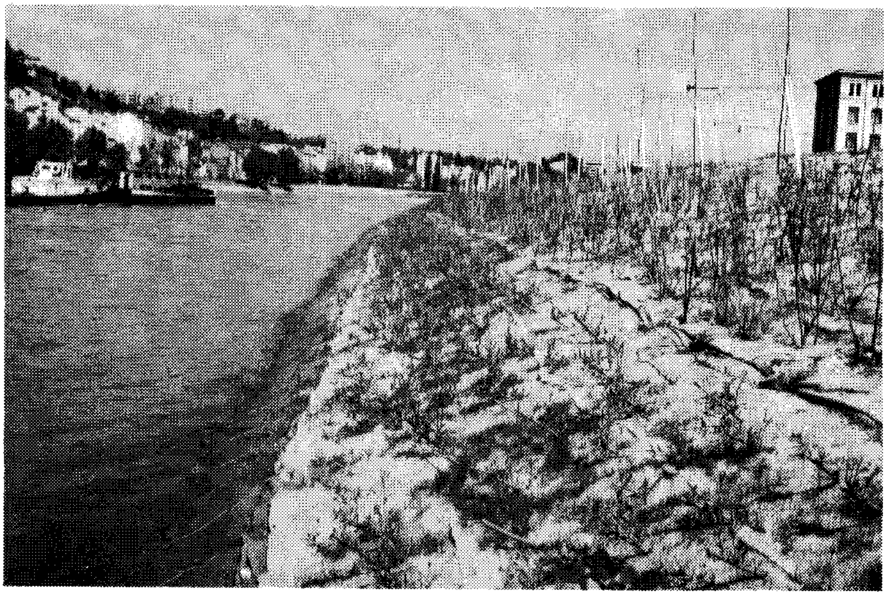


Figure 8.d:
One month after completion, plants show their growth potential. The technique used was conceived to resist ship-induced waves.
April 11, 1994
(Photo : B. Lachat)

Figure 8.e:
Vegetal engineering in full force.
July 4, 1994
(Photo : B. Lachat)

Figure 8.f:
A few months later, vegetation has grown to form water-resistant bushes. One species of willow (saule des vanniers) is dominant. Even on large streams, bioengineering techniques can be applied.
September 8, 1994
(Photo : B. Lachat)

ARTIFICIAL SILTING OF SHORE MARSHES : THE EXAMPLE OF THE BAY OF RIVIÈRE-DU-LOUP, QUÉBEC, CANADA

Pierre Bertrand and Bernard Massicotte
Les consultants en environnement Argus inc.
3075, chemin des Quatre-Bourgeois, Sainte-Foy (Québec) G1W 4Y4
Canada
pbertran@roche.ca

Jean-Pierre Beaumont
Service de l'environnement, ministère des Transports du Québec
35, rue de Port-Royal Est, 3e étage, Montréal (Québec) H3L 3T1
Canada
mcassetti@mtq.gouv.qc.ca

Denis Lehoux
Canadian Wildlife Service, Environment Canada
1141, route de l'Église, Sainte-Foy (Québec) G1V 4H5
Canada
derepentigny@cpque.am.doe.ca

ABSTRACT

The St. Lawrence encompasses more than 6 000 km of shoreline between the Great Lakes and the Gulf of St. Lawrence. The hydraulic dynamics of the system is characterized by three specific components: a river, an estuary and a marine environment, the Gulf of St. Lawrence. These components are greatly affected by erosive processes; in order to find adequate restoration techniques, we have elaborated and tested stabilization techniques respecting the ecological integrity of these ecotones. This paper presents the results of an experimental project in which sedimentation structures were set up in an intertidal zone within the St. Lawrence estuary. We have used a reinforced version of sedimentation fields, a technique conceived in the Netherlands and widely used around the North Sea. More than 160 concrete modules tied together and shaped into three contiguous crescents were installed in the Bay of Rivière-du-Loup, in the lower estuary of the St. Lawrence. The more than 70 m long frontage of the sedimentation field is located near the geodesic zero level, corresponding in the area to a 50 % submersion level, the lower limit of implantation of alternate-flowered Spartina (*Spartina alterniflora*). The sedimentation field was designed to let the mud penetrate in the park and be trapped in the back at flood-tide. Preliminary results are promising. The structures have resisted well to ice dragging; only four blocks were moved by ice over a distance of more than 100 m. A monitoring of sedimentation conducted from the spring to the fall of 1995 revealed that there were net sediment deposits on the lower marsh during that period, both inside and around the park. However, a thorough analysis of elevation data indicates that accumulation inside the field is significantly greater than outside the field. Although the monitoring does not cover a complete year cycle yet, the sedimentation field appears efficient at catching and retaining sediments, at least under summer and fall conditions. This successful approach should contribute to restore part of the tidal marsh lost to erosion during the past twenty years in the Bay of Rivière-du-Loup by way of the natural sedimentation processes occurring in the area. When located near ports and small craft harbours, these structures could also be used as controlled sites for dredging material deposits. The artificial silting of intertidal zones may prove to be a very interesting approach for the restoration of shore habitats, in particular tidal marshes.

KEY-WORDS: Erosion / Sediment balance / Shore processes / Sedimentation fields / Tidal marsh / St. Lawrence estuary / Ecological restoration.

INTRODUCTION

The St. Lawrence encompasses more than 6 000 km of shoreline between the Great Lakes and the Gulf of St. Lawrence. The hydraulic dynamics of the system is characterized by three specific components: a river, an estuary and a marine environment, the Gulf of St. Lawrence. The St. Lawrence shore dynamics reflects these various components, which are greatly affected by erosive processes. In order to find adequate restoration techniques, we have elaborated and tested stabilization techniques respecting the ecological integrity of these ecotones.

In the specific case of the St. Lawrence estuary, the approach selected uses large amounts of sediments transiting naturally over the intertidal zones of the estuary, as is the case in the Bay of Rivière-du-Loup.

The Rivière-du-Loup tidal marsh has been severely affected by erosion for more than fifteen years. Erosion would have begun in the 1980s following an unfavourable combination of high water levels and particularly strong winds (Drapeau, 1992). Between 1967 and 1985, receding rates in the upper marsh ranging from 1,6 to 2,2 m/year were recorded (Dionne, 1986; SOGEAM, 1984).

In the course of a pilot project cosponsored by Environment Canada, the Québec Ministry of Transport, the James Bay Energy Corporation and Ducks Unlimited, a solution for the restoration of the Rivière-du-Loup marsh was proposed by Les consultants en environnement Argus. This solution, inspired by a well proven technique currently used in Europe but hardly known in Quebec, is based upon the use of sedimentation fields. These structures established in the intertidal zone catch the suspended matters, abundant in the Bay of Rivière-du-Loup, stimulating regeneration of the substrate and helping the natural progression of the marsh (Figures 1 and 2).

A small-scale pilot project is underway in the Bay of Rivière-du-Loup in order to assess the efficiency of this technique. In the fall of 1994, a sedimentation field was built on a segment of the shoreline under active erosion (Bélanger and Verreault, 1995); structural and sedimentology monitoring were conducted.

The first results of the monitoring of this pilot project are presented in this paper. The objectives of the monitoring program are:

- to assess the state of the structure and its resistance to ice grinding;
- to evaluate the impact of the sedimentation field on the receding rate of the upper marsh;
- to make a preliminary assessment of the efficiency of the sedimentation field.

METHODS

Structure stability

In June 1995, distance and elevation readings were taken with an electronic theodolite (total station) to establish precisely the location of the sedimentation field. These measurements assess the stability of the structure and provide reference coordinates for the park location.

Reference lines

On the upper marsh, a reference line parallel to the edge of the embankment is used to measure the receding rate. Forty-eight meter long reference lines were positioned from a geodesic marker. Permanent stakes are placed at the center and at both ends of each line. The perpendicular distance between the edge of the embankment and the reference line was

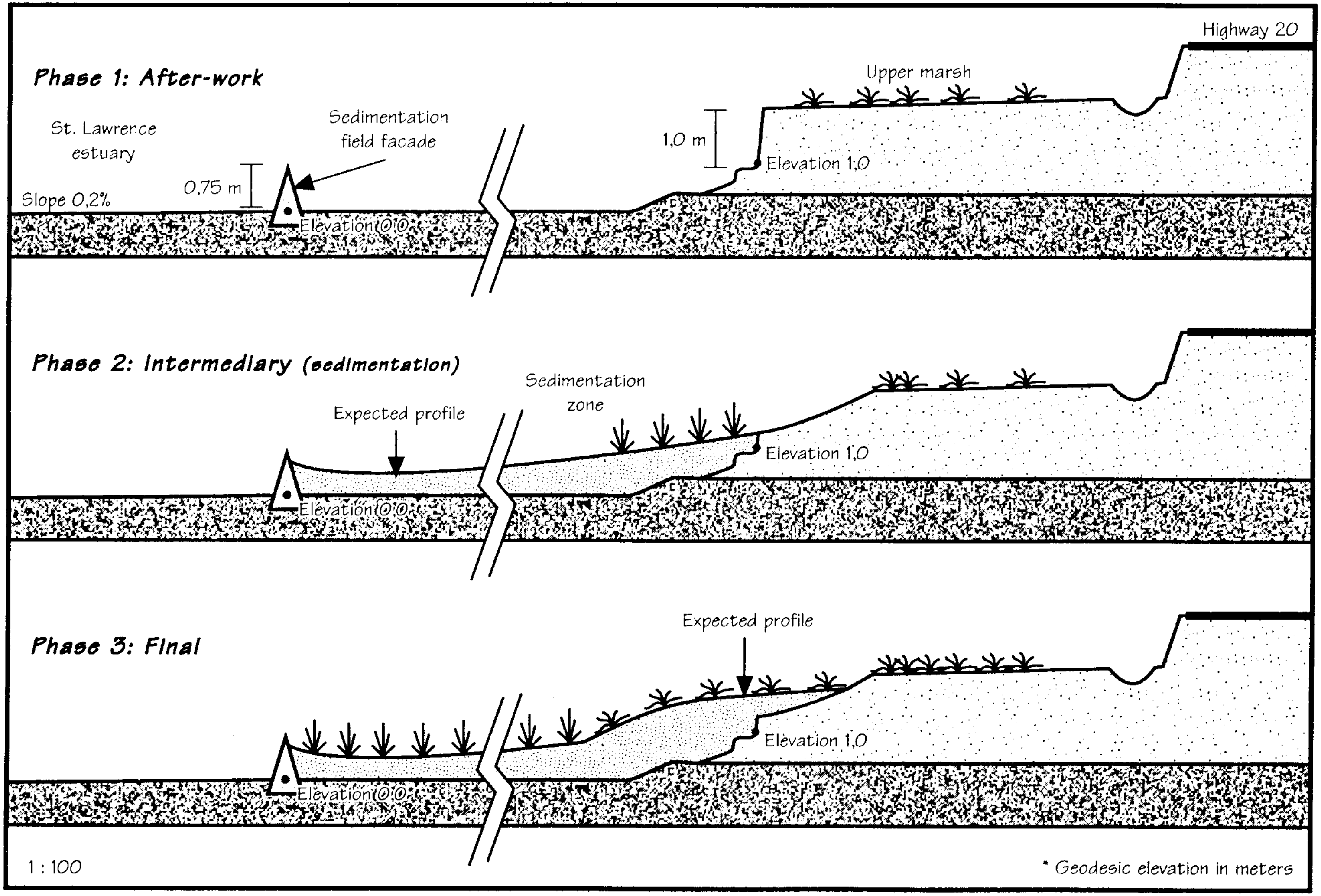


Figure 1. Cross-section view of the profile evolution of the Rivière-du-Loup marsh.

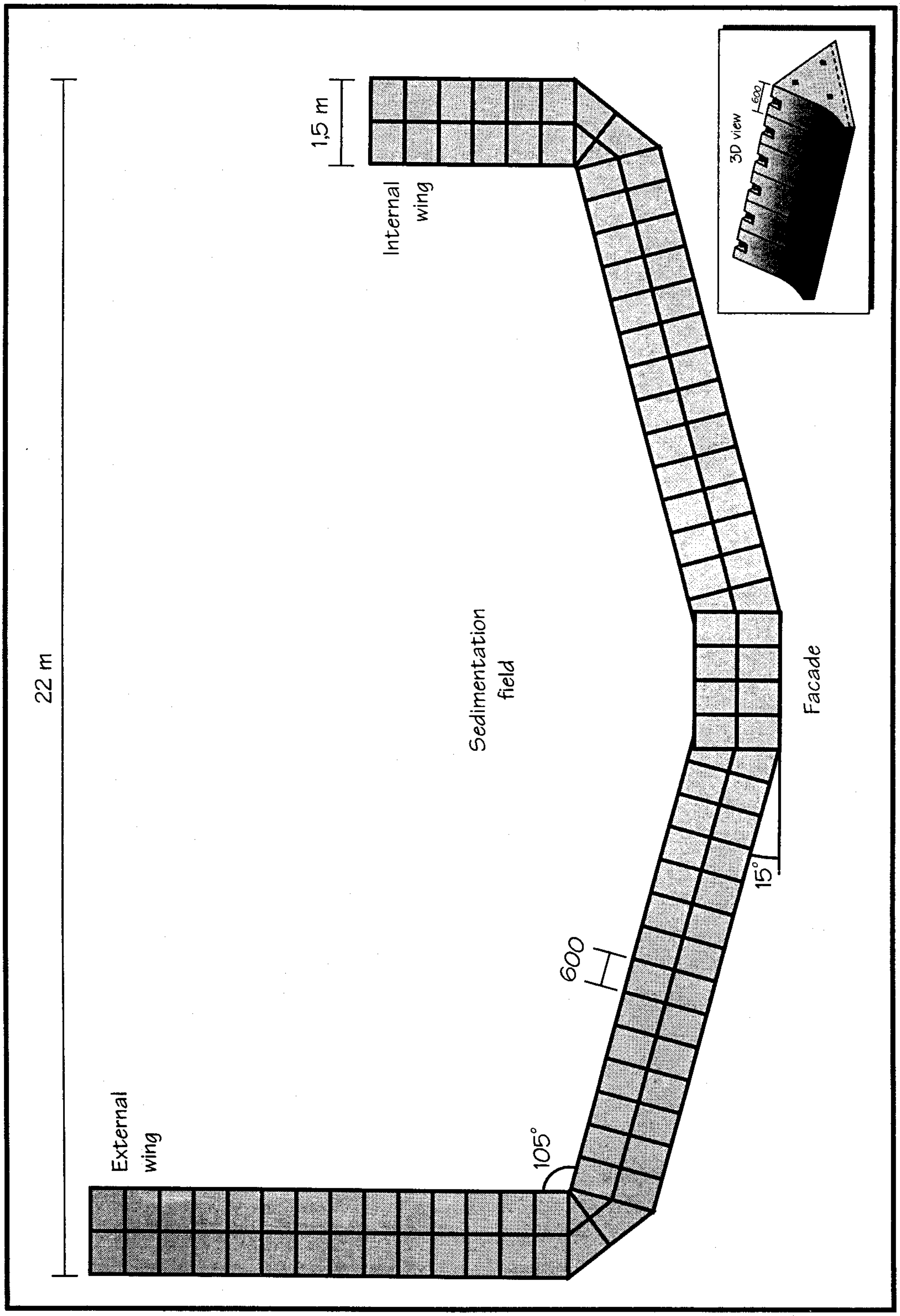


Figure 2. Detailed long-section view of a sedimentation field module.

measured every two meters with a measuring tape. In June, measures were taken at the top of the embankment along reference lines A and B. An additional reference line (C) was established in September, north of lines A and B (Figure 3). In November, data were collected on the position of the bottom and the top of the embankment at the three reference lines.

Profile of the foreshore

The second monitoring technique used consists in measuring with an electronic theodolite the foreshore elevation along three transects perpendicular to the shoreline (Figure 3). One of the transects passes through the center of the sedimentation field; the other two transects are located about 100 m away on each side of the park.

The foreshore profiles originate in the upper marsh, sloping down to the lower marsh over a distance of about 210 m. Measurements were all taken with an electronic theodolite and were positioned in relation to the International Great Lakes Datum (IGLD) in order to establish a relation between the geodesic levels and the tidal levels.

Sedimentation stations

In addition to the profile transects, 16 permanent sedimentation stations were established inside and outside of the sedimentation field (Figure 3). Elevation data are collected more intensively at these stations to compare sectors either under the influence or not of the park and to assess the efficiency at trapping sediments. These stations were also positioned and monitored with an electronic theodolite.

RESULTS

In the spring of 1995, we noted that the sedimentation field had been partly disturbed by ice movements. Four concrete blocks in the central module had been carried toward the lower foreshore over a distance of about 107 m. These blocks were picked up and repositioned in their original settings. We believe that the partial and temporary opening of the park following the displacement of these blocks had no major impacts on net sedimentation.

Upper marsh receding rate

An important receding rate was recorded at the top of the embankment during the monitoring period. Between June 8 and November 23, 1995, mean receding rates along line A and B were 4,6 and 3,1 m respectively. At line C established on September 14, the embankment had receded on average 3,4 m in two months. This rate is comparable to those observed at the other transects over a five month period. It may be the result of a more severe erosion in the line C sector, or of a particularly intensive erosion during the final two months of monitoring in all sectors, or of a combination of both causes. Future monitoring will clarify this point.

The primary objective of the sedimentation field is to favour sedimentation and to trap and retain new deposits. The park also acts as a breakwater during intermediate stages of the tidal cycle. However the structure is not sufficiently high to protect the upper section of the embankment during Mean High Water (Figure 4). Surveying data show that the top of the concrete blocks were at a mean elevation of 1,1 m in June while the level of Mean High Water is 2,0 m and the level of Spring Tide is 3,0 m (geodesic levels).

To become an efficient breakwater, a structure higher than 2 m (rather than 1 m) would have been required. It is quite possible that the park may protect the lower portion of the embankment during intermediate stages of the tidal cycle.

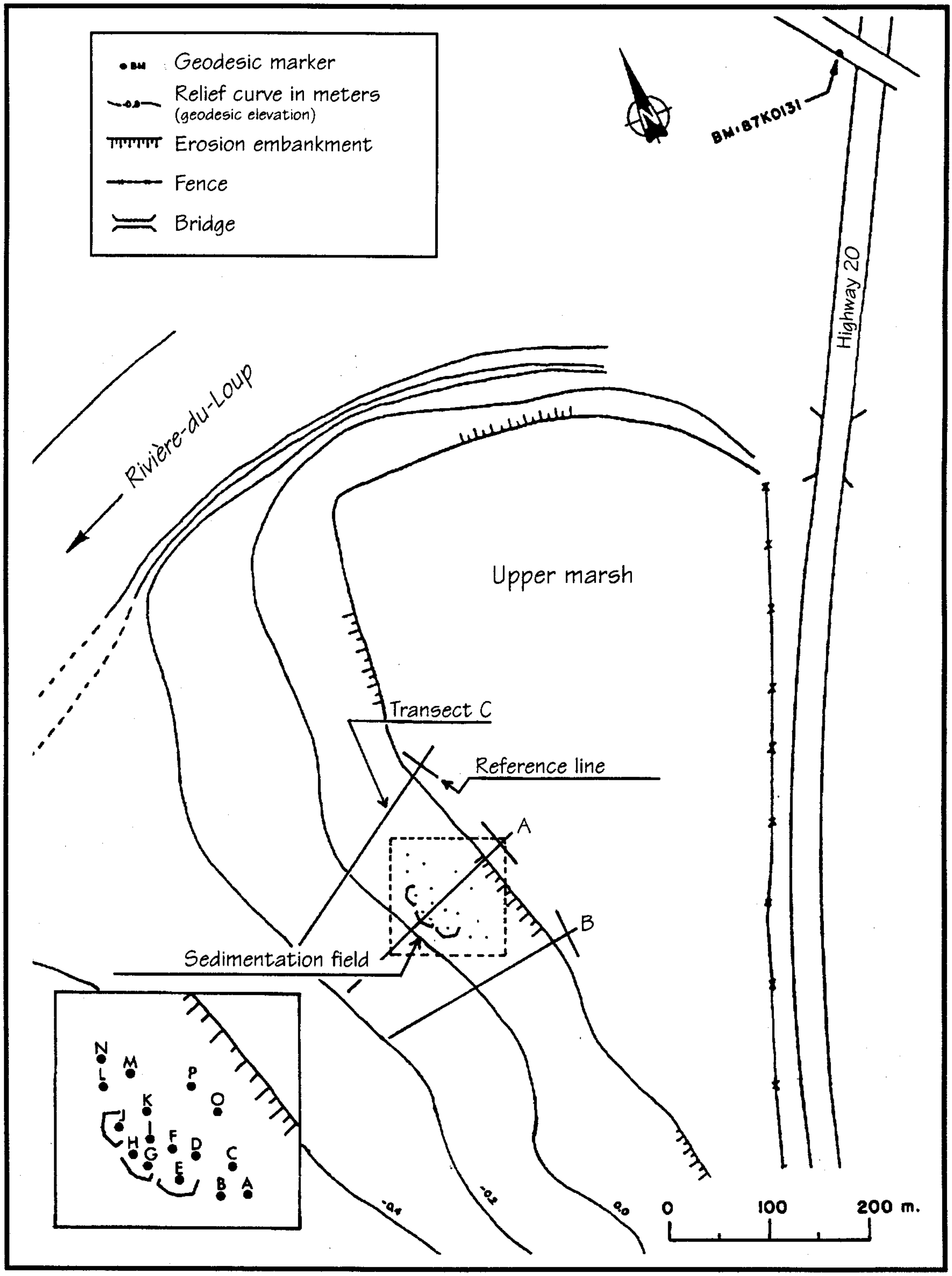
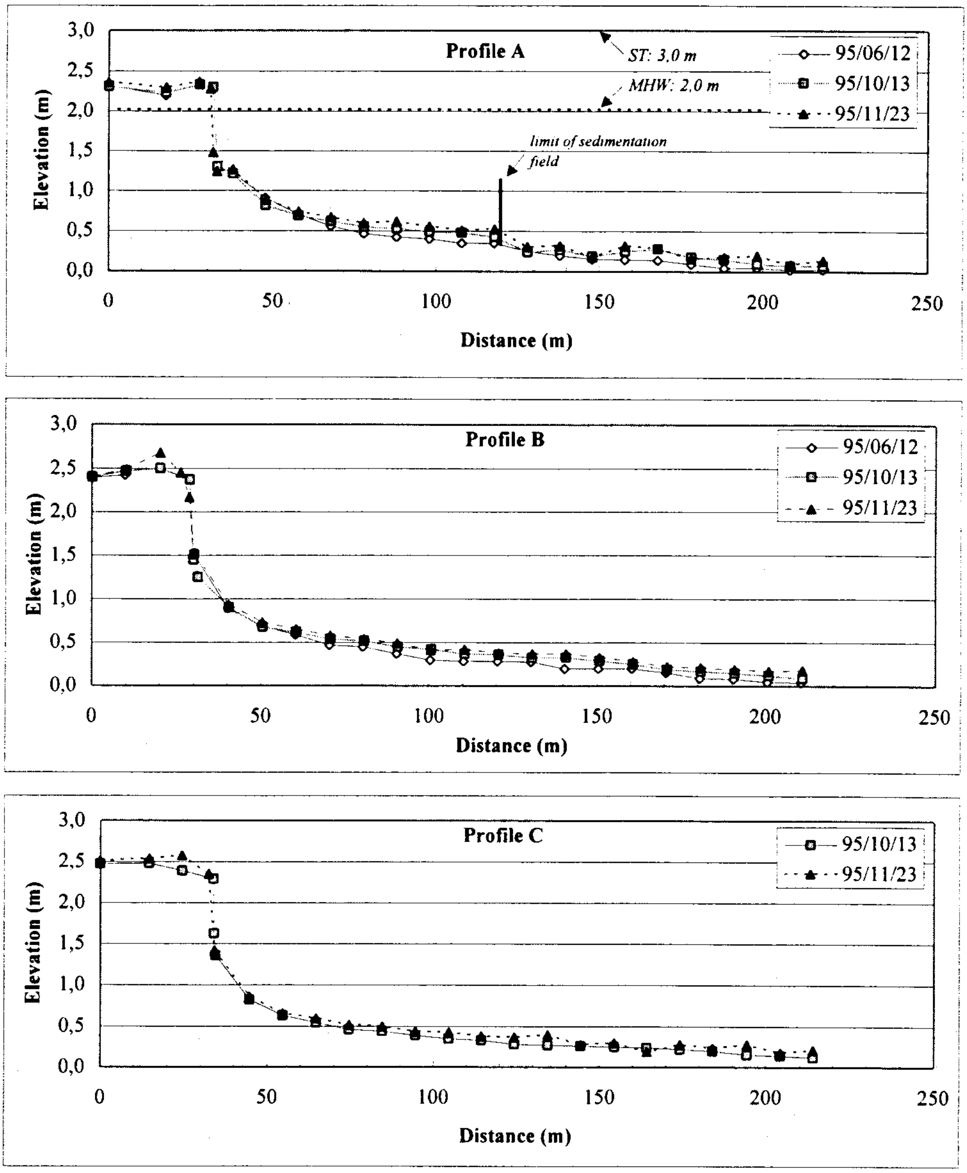


Figure 3. Location of the sedimentation field and of monitoring stations.



Notes: MHW : Mean High Water
ST: Spring Tide
Elevations according to geodesic marks
Vertical exageration : 24,7 x

Figure 4.
Foreshore profiles in the Bay of Rivière-du-Loup in the area of the sedimentation field (profile A) and south (profile B) and north of the field (profile C).

Sedimentation on the lower marsh

Sedimentation data on the lower marsh are from surveying conducted in June, October and November 1995. Figure 5 presents the sediment deposits on a 40 m strip parallel to the shore, in and around the sedimentation field. Between June and November, there was a net sediment deposit at all the surveying stations, with the exception of one station located approximately 50 m south of the sedimentation field. Net sediment deposit zones are observed within the park and along the three profile transects, on the whole width of the foreshore. Between June and October, sediment deposits ranged from 6 to 22 cm, most often above 10 cm. Between October and November, weaker deposits, almost always below 10 cm, were recorded. The net deposit during the whole monitoring period ranged from 15 to 27 cm. The highest values were recorded inside the sedimentation field.

A statistical comparison of sediment deposits between two groups of stations was conducted in order to assess the impact of the sedimentation field:

Group 1: Stations under the influence of the sedimentation field. This group includes stations D to K inclusively (Figure 6) and four points on transect A located in the park.

Group 2: Stations not protected by the sedimentation field, but located at the same level on the foreshore as Group 1 stations (40 m strip). This group includes stations A, B, C, L, M and N (Figure 6) located on either side of the park and stations along transect B.

The statistical analysis covered sediment deposits over the whole monitoring period, between June 12 and November 23, 1995.

In general, net deposits were recorded for both groups of stations. However, accumulation was greater at stations in Group 1, under the influence of the park. The difference is highly significant (p=0,0068, Wilcoxon-Mann-Whitney test). Figure 6 summarizes the means and standard deviations of the two groups.

As this paper is prepared, the monitoring data on the sedimentation field is only partial, extending over a short period of a few months. Important sedimentation events such as winter sediment deposits and ice break-up are not yet integrated in the overall sedimentation assessment. Only after these data are considered, will the actual efficiency of the sedimentation field be assessed with precision. Depending upon weather conditions, it is possible that sediment deposits outside of the park may be eroded away in coming months. We can expect milder erosion within the park, resulting in a higher positive balance of sedimentation than at reference stations. At this time, we can conclude that there is a preferential sediment accumulation inside the park. The capacity of the park to trap and retain the deposited sediments will have to be assessed in future monitoring.

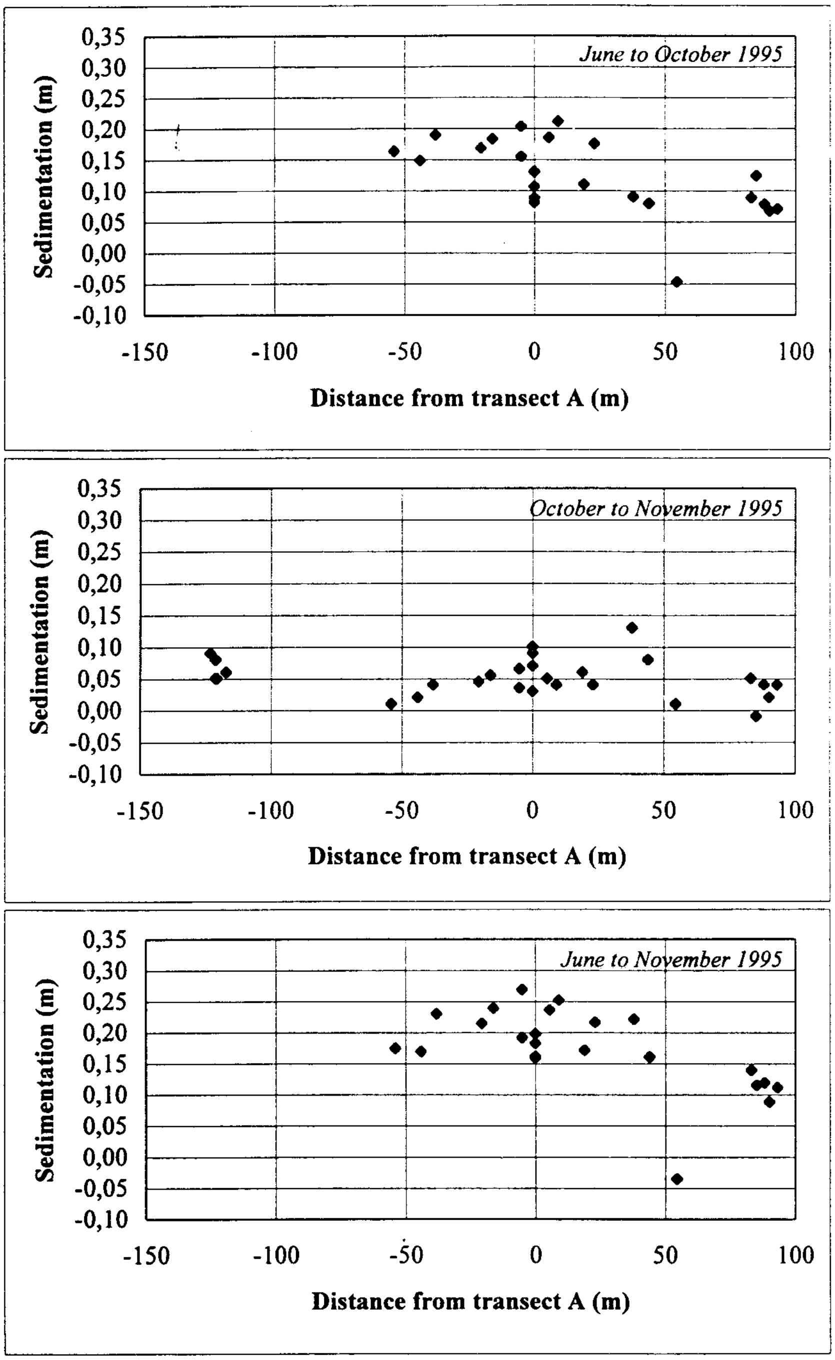


Figure 5.
Net sedimentation along the foreshore in relation with the distance from transect A.

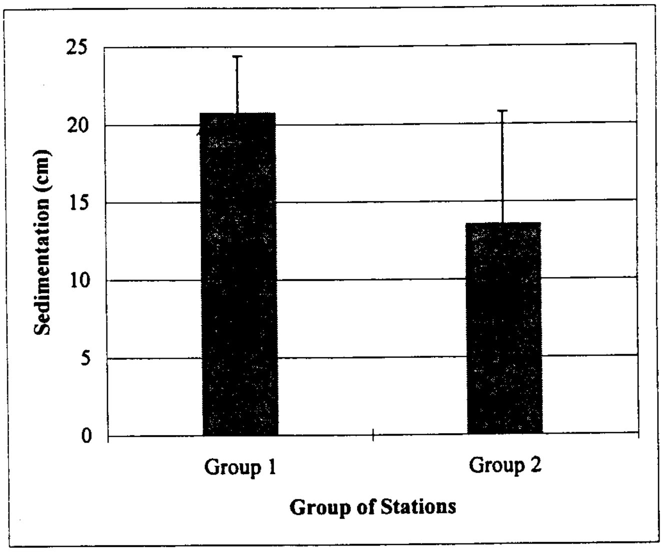


Figure 6
Net sedimentation between June and November at stations located inside (Group 1) and outside (Group 2) of the sedimentation field (mean and standard deviation). The difference between groups is significant at p=0,0068
(Wilcoxon- Mann-Whitney test).

CONCLUSION AND RECOMMENDATIONS

Preliminary results of the monitoring indicate that there is a preferential sediment accumulation inside the sedimentation field over a relatively short period of 5 months. It appears that the park as designed is capable of trapping and retaining sediments under summer and fall conditions.

In order to determine whether the structure will satisfy the main objectives of rebuilding the substrate and inducing a natural revegetation of the marsh, it is necessary to continue the monitoring. Additional data are needed to integrate important events of the sedimentation cycle such as winter sedimentation under the ice-cover and the subsequent ice break-up. We recommend that:

- a survey, similar to surveys already conducted, be conducted immediately after the ice break-up in April;
- a survey be conducted around mid-June 1996, 12 months after the first sampling, in order to cover a full year cycle.

During the monitoring, an important receding rate was recorded in the upper marsh, with a net loss of prime wildlife habitat. Under the present conditions, the sedimentation field probably limits the receding of the bottom of the embankment but does not prevent receding at the top of the embankment.

REFERENCES

Bélanger, S. and G., Verreault. 1995. Mise en valeur du marais de Rivière-du-Loup pour la faune et protection d'un tronçon de l'autoroute 20. Report submitted by Les consultants en environnement ARGUS inc. to Environment Canada, Canadian Wildlife Service, to the Québec Ministry of Transport and to Ducks Unlimited. 50 p.

Dionne, J.-C. 1986. Érosion récente des marais intertidaux de l'estuaire du Saint-Laurent, Québec. Géographie Physique et Quaternaire. 40(3): 307-323.

Drapeau, G. 1992. Dynamique sédimentaire des littoraux de l'estuaire du Saint-Laurent. Géographie Physique et Quaternaire. 46(2): 233-242.

SOGEAM, Les consultants. 1984. Étude de faisabilité de la restauration d'une zone de marais intertidal dans la baie de Rivière-du-Loup (Phase IV). Report submitted to Ducks Unlimited. 26 p. + appendix.

Troude, J.-P. 1991. Restauration et création de marais à scirpe dans l'estuaire du Saint-Laurent en aval de Québec. Report submitted by Robert Hamelin et Associés inc. to Environment Canada, Canadian Wildlife Service, Conservation and Protection (Québec Region). 83 p.

THE STABILIZATION OF THE OTTAWA RIVER BANKS: A COMPARATIVE STUDY BETWEEN BIOENGINEERING METHODS AND A METHOD BASED ON NATURAL GRAVEL EMBANKMENTS INSTALLED IN WINTER

Michel Bérubé [1], Guylaine Gagnon [2], Léon Hardy [3] and Line Bariteau [3]

1. Hydro-Québec, Vice-présidence ingénierie et services, Service études environnementales,
800 de Maisonneuve est, 19 ième étage, Montréal, Québec, H2L 4M8.

2. Hydro-Québec, Vice-présidence projets, Service lignes de répartition est, division environnement,
500 Sherbrooke ouest, Montréal, Québec, H3A 3C6

3. Poly-Géo inc. , 624 avenue Notre-Dame, St-Lambert, Québec, Montréal, Québec J4P 2L1

CANADA

mberube@ge.hydro.qc.ca

ABSTRACT

A major shore protection program began during the 1995-1996 winter on the Québec side of the Ottawa river. Over 70 km of shore will be stabilized along the reservoir of the Carillon dam, between the municipalities of Carillon and Gatineau. Because of the size of the project, an impact study was required in order to get the legal authorizations. The different shore protection methods were reviewed and a method was developed to fit the context of the Ottawa river. The bioengineering methods were closely examined because they offer an environmentally interesting alternative to traditional techniques. It appears that the multiple operations needed to install bioengineering works, such as clearing, slope reduction, soil amendment, plant rearing, planting and fastening with various materials, make these methods complicated and expensive for such a large scale project. These operations could also disturb the abundant wildlife in the area and reduce biodiversity. Furthermore, the shore access by landowners becomes more difficult because of the tight vegetation needed to stop the erosion. It was instead decided to embank the foot of the slopes with natural gravel of 20 cm to 40 cm maximum diameter containing about 15 % of sand. The embankments and their grain-size distribution are the smallest needed to meet the local erosion conditions. Different tests were carried out in the field to confirm the technical and the environmental performances of the works. The impacts related to the access on site for the machinery are avoided by using the winter season for construction. The technical operations are simple and the slope morphology as well as the shoreline itself are not modified. The edge of the work facing the river is remolded by the waves to adjust to the local energy conditions, and it merges completely with a natural gravel shoreline after a few years. The use of natural pebbles and cobbles mixed with a sufficient amount of sand size particles makes the works easy to walk on and facilitates the vegetation retake. Field tests have demonstrated that in most cases the natural retake will cover more than 80 % of the works in five years or less.

KEY-WORDS: stabilization/ riverbanks/ erosion/ Ottawa river/ Rivière des Outaouais/ bioengineering/ embankments.

INTRODUCTION

The stabilization of lakes and river banks is an important subject of discussion among environmental science specialists for many reasons. First of all, because there is an increased awareness of the multiple consequences of soil erosion on the aquatic ecosystem. Indeed, not only is the loss of land an incentive to stop erosion, but also new environmental considerations such as water pollution control (especially in agricultural areas) and conservation of wildlife habitats. Another reason is that many types of shore protection techniques exist, with different environmental "performances". If erosion can create environmental impacts, it is also the case for the erosion control measures. Therefore, even if erosion control is considered a mitigation measure in general, large shore protection projects must be carefully examined through an environmental impact assessment. According to Québec regulations, an impact study is required for projects that require more than 300 m of shore protection.

The increasing interest for environmentally improved stabilization methods have brought many authors to suggest the use of live material (herbaceous plants and hardwood cuttings) combined with fastening devices (stakes, steelwires) or other inert material (crib walls, stones). These methods are known under the name of "bioengineering" or sometimes "ecoengineering". Most of them have been developed as early as the thirties and forties in Europe and the United States and are described in Schiechtl (1980) and Gray and Leiser (1982). In Québec, among other trials, some experiments were conducted along the St-Lawrence river to develop alternatives to large riprap embankments (Argus 1991). This approach is also becoming the one encouraged by the authorities at various levels (St-Laurent Vision 2000 1996; COBBIR 1992; Gratton 1989).

The following presentation is based on the impact study that was submitted by Hydro-Québec (Hydro-Québec 1994), in order to get the approval of the *Ministère de l'environment et de la faune du Québec*, to stabilize more than 70 km of shore on the Ottawa river. It describes why methods using live material were not considered the best "environmental" alternative for this large scale project, and how mineral material constructions were rather designed to meet the environmental and technical requirements.

THE STUDY AREA

The study area is located on the lower reach of the Ottawa river between the city of Gatineau (West) and the Carillon dam (East). This stretch of the water plane is approximately 110 km long and its level is determined by the Carillon dam, except during major spring floods. The study is restricted to the Québec side of the river, that constitutes a borderline between the two provinces (figure 1). The riverside landowners are composed of 1113 private residents, 50 companies, the federal and provincial governments and Hydro-Québec. As for land use, 21% of the shore is considered urban; 47% is used for agriculture; 16% for recreational parks and 12% for conservation parks. Over two thirds of the shore are bordered by wooded bands. Eight municipalities are by the river. Of particular interest is the Plaisance wildlife reserve (near Papineauville) and a series of wetlands soon to be protected by the creation of provincial parks, from Plaisance upstream to Gatineau. These wetlands were created in part by the flooding following the Carillon dam construction, completed in 1963. Canada geese, among other fauna, use these areas to rest and feed in great numbers mostly in the spring.

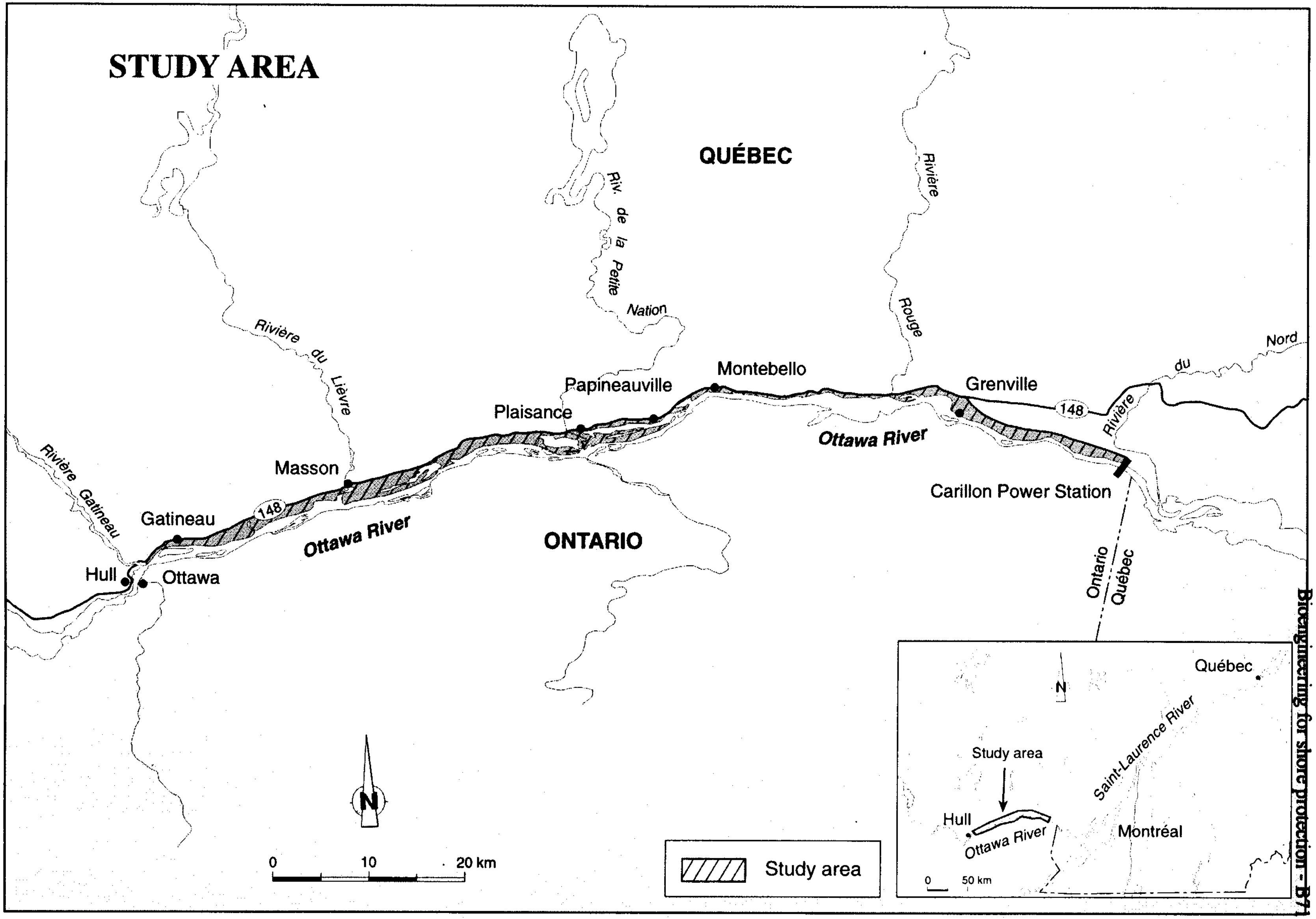


Figure 1: Study Area

BANKS MORPHOLOGY AND EROSION PROCESSES

The lower reach of the Ottawa river crosses the Saint-Lawrence lowlands which are characterized by a low and flat topography, developed on generally horizontal sedimentary rocks. The bedrock is usually overlained by unconsolidated material, mostly marine silt and clay or a mixture of fine sand, silt and clay of alluvial origin. These fine materials are susceptible to erosion. Banks heights in the study area vary from less than 1 m to about 10 m. Before the construction of the Carillon dam, the water levels used to vary with the river discharge. The main agents of erosion were then currents, waves and ice. The completion of the Carillon dam, in 1963, raised the water level to a 41,15 m elevation at the dam site. This corresponds to a maximum rise of 18 m on a short section between Carillon and Grenville, and only 1,4 m on the longer section upstream of Grenville, where most of the erosion occurs. The actual summer level in this last section is close to what used to be the average spring flood level in natural conditions.

The formation of a higher water plane upstream of Carillon had the effect of reducing the erosive forces of currents and ice. However, the fetch has been slighly increased. It varies generally between .5 km to 2 km, with a maximum of 5 km on short sections of the river.

On the whole, the waves acting on the shore are relatively weak when compared to larger reservoirs or to the ocean. But the fact that the silt and clay banks are now attacked by the waves at an almost constant level during the ice-free season increases significantly the erosion rate. The waves take away the fine particles, which are too light to deposit and form beaches at the foot of the banks. No erosion occurs everywhere the shore is made out of gravel. The substrate is therefore considered to be the most important erosion factor.

A survey of the evolution of banks erosion has been carried out with different series of aerial photographs, taken before and after the construction of the Carillon dam. The study reveals that about 42 km of shores was being eroded before 1963 and that close to 57 km of eroded banks were added after 1963. The total length of shore including all the bays is over 300 km. The study reveals also that the narrow lower terrace which bordered many sections of the river before 1963, is completely eroded in many cases, allowing waves to erode the higher terrace and initiate larger land movements. This is particularly the case in the Plaisance region. Some cottages and roads needed urgent stabilization measures in 1992, 1993 and 1995. Because the actual erosion process continues, it was decided to stabilize everywhere needed before bigger damages occur, needing more complicated and expensive interventions.

Even though landslips and landslides are more spectacular because they occur on higher cliffs (photos 1 and 2), the erosion rate in this study refers strictly to the receding rate of the shore, and this rate can be fast even along lower terraces. On the Ottawa river, simple sapping by the waves is very efficient to remove pieces of grassland (photo 3). Even shrubs and trees cannot maintain the fine substrate in place since the waves bare the roots easily (photo 4).

ENVIRONMENTAL KEY ISSUES

Before choosing a shore protection method, we examine here what are the major environmental issues.

As for all shore protection projects, the major environmental issue is to maintain, if not recreate, a contact zone between the aquatic and terrestrial ecosystems, that is suitable for both the local flora and fauna.

Photo 1: Landslide

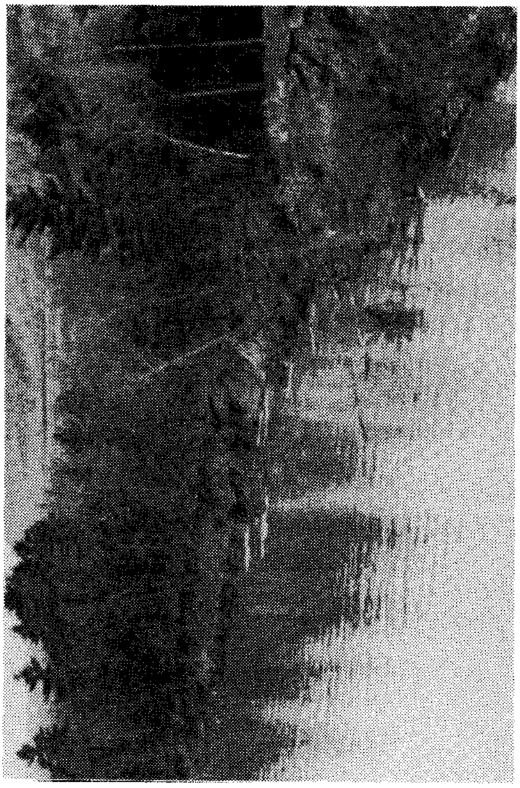


Photo 2: Landslip

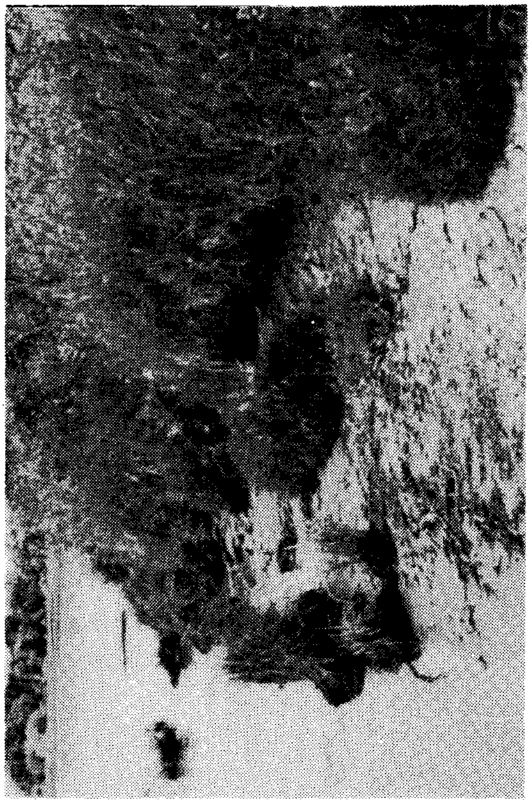


Photo 3: Sapping

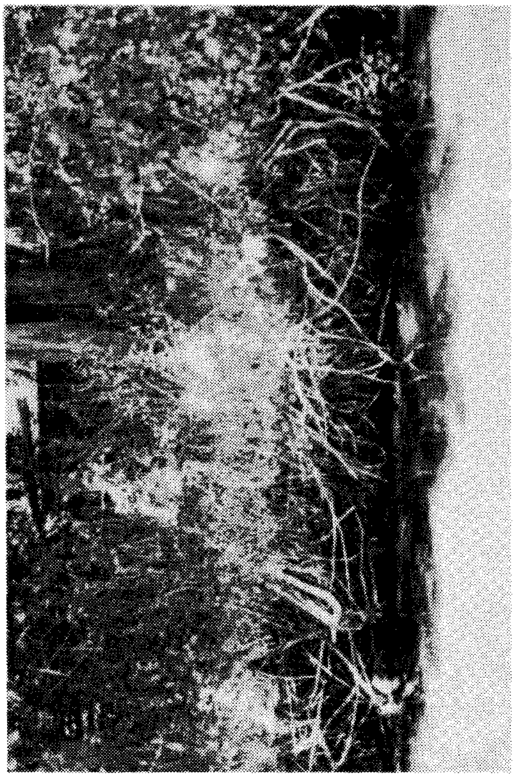


Photo 4: Undermining of vegetation

The maintenance of the existing local biodiversity is also a main issue. As mentioned earlier, the upper half of the river contains a rich flora and many fragile wildlife habitats. The flora along the shore was inventoried during the impact study. Over 209 species were found. Many rare and protected species colonize the area (Dignard 1994). As a matter of fact, the whole region is one of the richest in rare plant species in Québec. Unfortunately, the narrow band of forest left, especially in agricultural areas, is now rapidly falling in the river as the erosion continues. The stabilization method must preserve as much as possible the existing vegetation.

The timing of the construction period is important. Because of the abundant fauna in the natural zones, an attention must be given to the critical periods for wildlife. Most of the fish species in the Ottawa river are spring and summer spawners. Some areas near erosion zones, especially in the upper half of the river, are important spawning grounds for pike in early spring (Fournier et al. 1990). Ducks and shorebirds are present in the area in great numbers during spring and fall migration periods, and for nesting from April to July. As far as Canada goose is concerned, the Plaisance wildlife reserve and its surroundings are densily used during migration for resting and feeding from the beginning of April to the middle of May. On the people side, the project must disturb as less as possible the normal recreational and leisure activities going on during the summer time, which is the peak season of occupation for the many cottages along the shore. Waterfowl hunting in fall is also an important activity that construction on a large scale could disturb.

Another major issue is the access for heavy machinery on the construction sites. An access strategy must be planned in order to avoid potential impacts to private properties, especially in zones of high occupation density. The construction of temporary roads must also be limited to a minimum not only for economical considerations, but also to avoid land clearing. The access to forested areas is also particularly difficult.

After stabilization, the shore must remain easily accessible to the residents. This is considered a very important issue by most landowners.

Finally, the stabilization works must integrate into the landscape and leave as few evidence as possible of human intervention.

TECHNICAL AND ECONOMICAL CRITERIA

Technically, the obvious and most important criteria is the efficiency to stop the erosion on a long term basis.

The cost, needless to say, must be as low as possible. The two next criteria derive from this one.

The method must be adaptable with a minimum of modifications from site to site (there are 70 km of shore to protect).

We are looking also for a method that is simple. It must be done in a few operations, quickly, and with little or no need for further maintenance.

METHOD SELECTION

Many stabilization methods have been examined for the Ottawa river project. It is obvious that most classical methods did not perform very well when it comes to the environmental considerations, neither did they on the economical criteria. We would like to examine here more closely the bioengineering approach that would apply, and a method based on simple natural gravel embankments. Each method is first briefly described, and then evaluated according to the issues and criterias explained above.

Bioengineering methods

In the case of the Ottawa river, we have established that the banks made out of silt and clay were very susceptible to erosion, and that sapping at the foot of the banks was responsible for most of the damage. The use of woodplants (willows in that case) could break the energy of the waves before they get to the substrate. Since this substrate is so easily transported by the waves, a possible minimum design could be to put a riprap key with a geotextile sheet under the lower summer water level; stones and fastened wood cuttings between the lower and higher levels; and various appropriate plantations above. Other variants are possible but their listing is unnecessary here. What is important is that for most stabilization methods using woodplants, there is a need to control the erosion during the time that the rooting system is developing, which implies a combination with another method using inert material (Gratton 1989).

As far as technical criteria are concerned, the following observations are made: The technical operations are complicated and need many steps such as: the choice of plant species and local sources; the rearing facilities when big quantities are needed; the clearing and levelling down of the slope; the operations regarding soil amendment and preparation; the storing, planting and fastening of live material; and the maintenance during the first and often further post construction years. Additional operations are needed when combined with riprap keys or other inert material protections at the waves action level.

The adaptability of the method is low since a single basic design cannot be used for different types of banks and erosion conditions. An important effort must be made to elaborate adapted designs for each site.

Environmentally, bioengineering certainly adresses the main issue, which is to create a good transitional zone between the terrestrial and the aquatic ecosystems. But the quality of the wildlife habitat that is created needs to be characterized. It has not been shown yet that such a shore is easily used by either waterfowl, shorebirds or small mammals. Indeed, in some cases, the willows or other shrubs need to be planted so tightly that it forms almost a barrier. Studies have to be done to determine the actual use of these different types of works for riparian wildlife. We didn't find such information in the literature.

The biodiversity would be affected by the clearing of the natural vegetation in favour of a few species, even if chosen locally, selected for their ability to resist to erosion. It may be possible to limit the extent of this impact by a careful selection of multiple local species, but at an unaffordable cost on a large scale project.

The time period required for the work would be in late spring or early summer if possible (after the flood). The mechanical work needed to prepare the slope for plantation and to install inert material at the base, would create turbidity, disturbing fish spawning and egg incubation. Bird nesting grounds would be perturbed as well as the resting areas for geese in migration. The recreational and leisure summer activities would also be strongly disturbed by major construction activities.

The access to the shore is problematic since the same heavy machinery used for traditional stabilization measures is required, with the usual terrain modifications this implies.

For local residents, the dense mixture of willows and other shrubs would make the access to the shore difficult. Indeed, unless willow poles are planted so that root nets form a continuous mat, they do not contribute to protect against erosion. On the contrary, "isolated root nets are likely to enhance erosion more than hinder it" (Oplatka *et al* 1995). So the owners will have to condemn a significant part of their shoreline and build specific accesses to the water.

Cost wise, the demonstration has not been made that bioengineering is cheaper. For the moment, most reports on bioengineering do not give enough information on that point. It is said to be cheaper than "hard" methods, but rarely are the real construction costs revealed. In the present project, we have estimated that a reasonable price for bioengineering techniques would vary from 100 to 500 dollars per meter for a minimum intervention. The embankments proposed in the following section cost between 50 to 150 dollars per meter.

Gravel embankments installed in winter

What was finally proposed was to stabilize the foot of the banks with natural gravel of sizes corresponding to local erosion conditions. The solidity and reliability of the works are assured by a precise knowledge of the local erosion forces involved, and by different tests in the field before starting the project on a large scale.

In order to determine the optimal gravel size needed for stabilization, a study on local wave actions was done. The height of the waves and their impact on the shore have been monitored in a variety of conditions. From these informations, the minimum diameter required and the optimum grain-size distribution were calculated.

Field tests have also shown that the works are stable as long as a good percentage of the right size stones were present (here 20 to 40 cm diameter), and that the works contain a variety of intermediate sizes down to sand size. The size of the embankment itself is defined by the height of the bank. The higher banks (more than 5 m) are stabilized with an embankment of 3 m width at the top, finished with a 3 in 1 slope. After stabilization at the shore level, the unsteady slopes will gradually reach a more stable profile (25° to 30°) and will partially cover the work (figure 2). The smaller banks need embankments as small as a 0.5 m wide at the top.

The height of the work is the difference between the lower summer water level and the average spring flood level, plus an additional 0.5 m to break most waves. The maximum height usually reaches 1 to 1,4 m. The work needs neither a stone key at its base nor a geotextile sheet against the bank. The contact with the clay material of the bank is cut by a sufficient amount of sand and gravel within the work. The generally flat littoral zone of the Ottawa river on which the embankment is installed permits the work to progressively get to an about 10° slope, as the material is rearranged by the local waves action. This 10° slope is the same as the one observed on natural gravel shores found on the river.

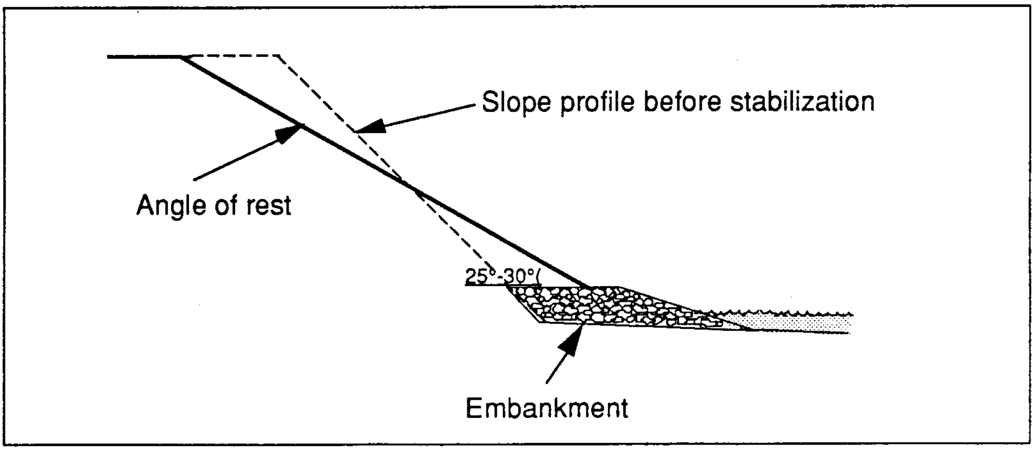


Figure 2: Evolution of the slope after the embankement installation

One of the major advantages of this method is definitely that it can be done during the winter time by circulating on the ice. Indeed, when the slope of the littoral zone is gentle, the ice is often thick enough to rest on the bottom, and needs little work to permit machinery circulation. The access problems are solved, and wildlife or residents activities are not disturbed. Also, the bank doesn't have to be levelled down or cleared of its vegetation. Indeed, it is left as it is. Once stabilized at its base, the bank takes slowly an angle of rest and allows natural vegetation recovery.

The wildlife habitat issue is also met by a rapid retake of natural vegetation from both ends of the work, the shore and the bank. The only impact on wildlife habitat is the fact that the protected shore becomes inappropriate for muskrat and other small mammal burrows.

Field tests were done in 1992 and 1993, showing that the vegetation retake on the embankment is very good if the amount of sand size particles is sufficient (around 15%). The results show that vegetation covering will be over 80% five years after the construction for most works. Within 5 to 10 years, the plant community will be sufficiently structured to integrate totally the landscape. In some field tests the proximity of mature trees and a dense plant community allowed an extremely rapid retake (photos 5, 6, 7). No soil amendment is generally needed since spring floods bring sufficient amounts of fine material and organic matters.

Concerning the aquatic habitat, a study has demonstrated that in most cases, the works are colonized by a more diverse benthic fauna than the original clay material. The works are also suitable for the spawning of many fish species. The only potentially harmful operation is the clearing of fallen trees before the construction. This is done from a barge in late fall though, when fish spawning and egg incubation periods are over. The encroachment of the embankments on the fish habitat as such is limited by the construction of the smallest works needed.

Finally the use of natural gravel also permits an access to the river for the residents or the public because the stones are round and easy to walk on. A summary of this comparative analysis is given in table 1.

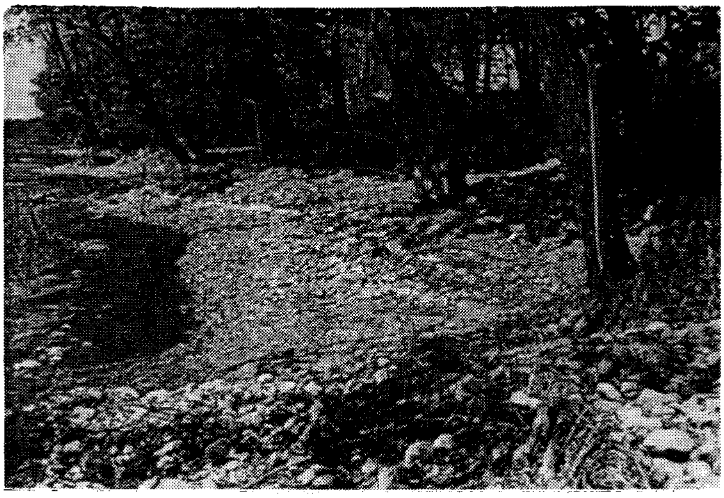


Photo 4: May 1992, the first spring after the bank stabilization

Photo 5: June 1993

Photo 6: September 1993
Vegetation retake at Baie Noire near Plaisance

Table 1: Comparison of stabilization methods according to key issues and technoeconomical criteria in the context of the Ottawa river.

KEY ISSUES	BIOENGINEERING	GRAVEL EMBANKMENTS
Maintenance of biodiversity	Difficult but possible.	Yes
Future use of shore by wildlife	Yes, but quality of habitat to confirm.	Yes, improving with vegetation retake
Easy access of machinery to the shore	No	Yes if done in winter.
Low impact construction activities	No	Yes, if done in winter.
Future use of shore by residents	Can be restrictive.	Easy with proper material.
Efficiency	Yes, when properly designed.	Yes, when properly designed.
Simplicity of design and construction	No	Yes
Flexibility, adaptability	No, many designs needed.	Yes, one basic design.
Low cost	Variable, to confirm.	Yes

DISCUSSION AND CONCLUSION

The method developed here was inspired in part by the Alcan stabilization project of the Saint-Jean lake banks (Alcan 1983). This project was the first large scale stabilization program in Québec to be carried out in winter, using the ice as an access to the banks. It was also the first one submitted to an environmental impact assessment. As for the use of natural gravel at the foot of riverbanks, it was never used before in a major project in Québec to our knowledge. Argus (1991) has found that the simple deposition of stones in front of banks in the St-Lawrence river, with no intervention on the bank itself, has been tried only by some private owners, with various levels of success. One similar embankment was installed on the La Grande river near Chisasibi (Vézina et Fortier 1985) with similar material to what is proposed here, and is still perfectly functional.

Our design is based both on the observation of nature, and experimentation. On the Ottawa river, the natural gravel banks are stable. In most cases there is no need for big riprap constructions. We have demonstrated this theoretically and in the field. These conditions permit the use of mineral materials that are more "user friendly" for people, fauna and flora.

If stabilization methods using plants are generally presented as being more environmental and less expensive than methods using inert material (Oplatka *et al* 1995, Houde 1994, Schiechtl 1980), we think that mineral (inert) material can also be used in an environmentally improved way and be competitive on an economical basis, especially for large scale projects. Many eroded banks do not need to be replanted when their basis is stabilized and protected from the waves action above the mean flood level. They gradually acquire a stable slope and are colonized, along with the embankment at their base, with the local vegetation. When the grain-size distribution of the material is proportional to the local erosion forces, and when the size of the construction is meant to avoid overdesign, the environment is protected at its best and at low cost. Furthermore, when natural rounded stones are used, the shore stays easily accessible to both wildlife and humans and the work integrates easily to the landscape. After all, these stones were originally carved by water, and belong naturally to river beds and lake shores.

The Ottawa river stabilization program is a large scale project, with very specific needs. The solution found here does certainly not applie to all cases. In our opinion, there is in fact no *environmental* erosion control measure *perse.* There is instead an *environmental approach* to erosion control, that we think implies necessarily a complete environmental impact assessment.

ACKNOWLEDGEMENTS

Many consultants have participated, in collaboration with the authors, to the impact study from which most of the material contained in this paper was taken. These are Genivar inc. (formerly named Groupe environnement Shooner), Anne-Marie Boulva (independant consultant), Nicole Lavoie (independant consultant), Marc Tanguay and Nicolas Roy (from the École Polytechnique de Montréal), and Laboratoires SAB inc..

REFERENCES

Aluminium du Canada Ltée (Alcan) (1983). Programme de stabilization des berges du lac Saint-Jean; Résumé de l'étude d'impact sur l'environnement et le milieu social. Dans *Le Lingot,* septembre 1983. 23 p.

Argus Groupe-Conseil Inc. (1991). L'érosion des îles du Saint-Laurent tronçon Montréal-Lac Saint-Pierre. Environnement Canada, 63 p.

COBBIR, Conférence municipale sur les bois, les berges, les îles et les rivières (1992). Guide d'aménagement et de renaturalisation des berges, Ville de Laval, 8 p.

Dignard, N. (1994). Les plantes susceptibles d'être désignées menacées ou vulnérables du secteur de Plaisance, rivière des Outaouais, Québec. Rapport complémentaire (Baie Lochaber-Baie Noire). Rapport préparé pour la Direction du pein air et des parcs. Ministère de l'Environnement et de la faune, 20p..

Fournier, H.; Houde, P. and G. Thellen (1990). Rivière des Outaouais. Évaluation de la qualité de l'habitat de fraye du grand brochet dans le bief Hull-Carillon. Ministère du loisir de la chasse et de la pêche, service de l'aménagement et de l'exploitation de la faune. Hull, 18 p.

Gratton, L. (1989). L'utilisation des plantes ligneuses dans la stabilization des berges en milieu agricole. Ministère du loisir de la chasse et de la pêche, 61 p.

Gray, D. H.; Leiser A. T. (1982). Biotechnical slope protection and erosion control. Van Nostrand Reinhold publ., 271 p.

Houde, B. (1994). Projet d'aménagement de rive au Québec. L'arbre en ville et à la campagne, colloque sur les pratiques de végétalisation, pp 243-249.

Hydro-Québec (1994). Programme de stabilization des berges québécoises de la rivière des Outaouais. Rapport d'avant-projet. 3 vol.

Oplatka, M. and A. Sutherland (1995). Tests on willow poles used for river bank protection. *Journal of Hydrology (NZ)* 33(1), pp. 35-58.

Saint Laurent Vision 2000 (1996). Les plans d'aménagement et les projets de démonstration de restauration d'habitats fauniques. *Le Fleuve, bulletin d'information* 6(1) pp. 3-4

Schiechtl, H. (1980). Bioengineering for land reclamation and conservation. Department of the environment. Government of the province of Alberta. U. of Alberta press. Edmonton, 404 p.

Vézina, Fortier et associés (1985). Travaux de protection de la berge de Chisasibi. SEBJ, Note technique IXC-808-85-002, 54 p.

Bank Protecting Functions of Common Reed and Ditch Reed

Shoji Fukuoka
Professor
Department of Civil and Environmental Engineerring,
Hiroshima University
1-4-1, Kagamiyama, Higashi Hiroshima, 739,
JAPAN
sfuku@ue.ipc.hiroshima-u.ac.jp

Akihide Watanabe
Associate Professor
Ditto

ABSTRACT

Through field observation and hydraulic experiments, we have revealed the mechanism by which reeds growing along river banks reduce bank erosion by mitigating the erosive force acting on natural banks of, for instance, flooding and waves produced by boats. We also theoretically assessed the critical velocity at which the bank-protecting functions of a vegetation revetment can be brought out; the engineering validity of this critical velocity was proved in field research.

KEY WORDS: natural vegetation / bank protection / vegetation revetment /flood current / boat waves /

INTRODUCTION

Channel vegetation not only represents a component of flow resistance during flooding and affects the water level of a river, but it also plays an important role in terms of the scenic beauty, ecosystem and other environmental aspects of the river.

Because of the necessity of determining the role played by channel vegetation in resistance during flooding and river environment, in recent years much hydraulic and environmental research on channel vegetation has begun to be performed(Fujuoka and Fujita, 1990). Some research is being carried out from the perspective of the ecology of channel vegetation(Okuda, 1976) and we have come to know the types of vegetation that grow in a channel and the types of locations in the channel where they grow.

In this research, which focuses on the soil-maintaining effects of vegetation -- particularly common reed and ditch reed -- growing near river banks, investigations have been carried out with the objective of determining the viability of using common reed and ditch reed in bank protection works. Up until now there has been almost no research from the standpoint of actively using, in bank protection, living vegetation that can be obtained in actual rivers(Fukuoka,1993). However, recent close-to-nature river improvement work involves using natural vegetation in river improvement, and while this trend is a positive one, technical evidence of its effectiveness is still necessary. The objective in this research is the collecting of just such evidence. Then, we investigated over time the relationship between common reed growing along river banks and the amount of bank erosion in order to confirm the effectiveness of common reed in reducing bank erosion. First, we after conductied field investigation on the growth environment of common reed and on bank protection conditions. In addition, field experiments on tensile strength of bank covered with reed and analysis based on these results was conducted to determine the velocity up to which banks with common reed can withstand erosion.

THE EFFECTIVENESS OF COMMON REED IN REDUCING BANK EROSION

In order to gain an understanding of the overall effectiveness of common reed in reducing bank erosion, we investigated the relationship between the amount of bank erosion (as determined with cross-sectional surveying) and

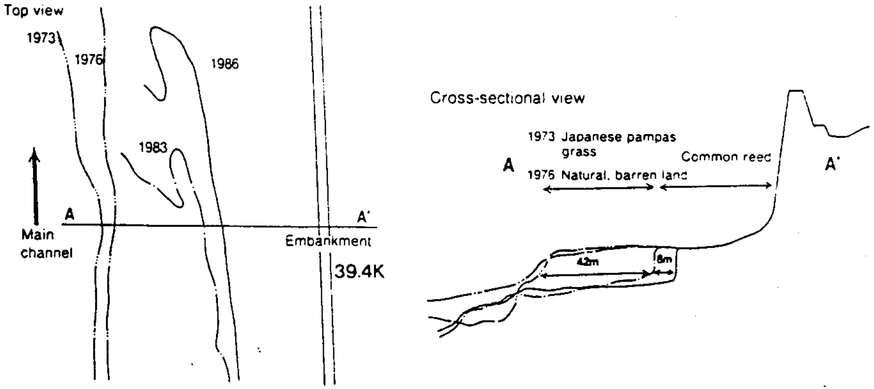
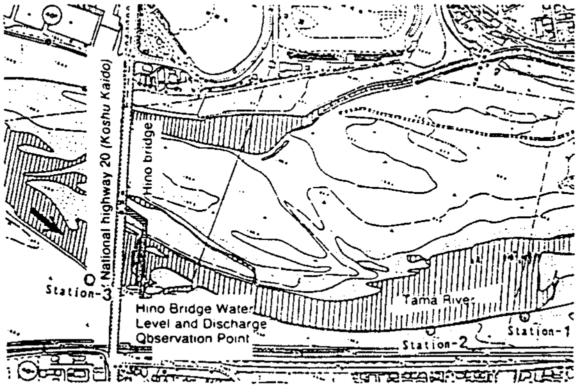


Figure 1: Change over time in bank vegetation and flood channel vegetation



Figiure 2: Field survey area

changes over time in flood channel vegetation in the middle reaches of the Tama river (right bank at the 39-40 km point; see Figure 1).

Floods in which water entered the flood channel occurred in 1974, 1979 and 1982 (August and September). Although the vegetation in the flood channel was colonies of Japanese pampas grass (Miscanthus sinensis) in 1973, the effects of large-scale flooding in 1974 had transformed the flood channel into natural barren land by 1976. This is believed to be because suspended sediment accumulated in the colonies of Japanese pampas grass, eventually burying and killing it. Four floods occurred thereafter in the seven-year period between 1976 and 1983 causing the banks to erode and recede, but we found that bank erosion had nearly stopped around the boundary of the common reed colonies in the flood channel. Examination of the bank in 1986 showed that erosion has progressed insignificantly since 1983, a fact that demonstrates the effectiveness of common reed in reducing bank erosion in the flood channel.

THE STATE OF BANK VEGETATION IN ACTUAL RIVERS

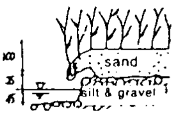
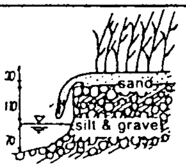
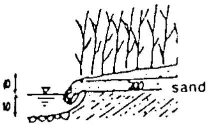
Common Reed And Their Growth Environment

We performed a field survey on how common reed protects river banks in stations 1, 2 and 3 in the research area (see Figure 2). Located here is the Hino Bridge Water Level and Discharge Observation Point. Stations 1 and 2 are where the current strikes the bank during flooding; at station 3, although the current is concentrated at normal water level. during flooding the main current shifts to the center of the channel. Compared to other locations, the erosive force of flood currents is lower here.

Common reed is a herbaceous plant often found in flood channels and shoal sandbar in the middle and downstream reaches of rivers with thick deposits of sediment. Common reed, in fact, is known for sprouting anew after being covered with sediment during flooding, and can grow to heights of three to four meters. It also grows in quite dense colonies with a distance between stems of roughly 10 centimeters, and although above ground a cluster may appear to be comprised of individual reeds, they are interconnected through subterranean stems. Around the research area the bed slope is 1/280, distance between embankments 300 meters, and flood channel width roughly 100 meters. Since 1960 the bed height has dropped roughly three meters, and so the banks were formerly river bed, and the banks of the main channel are mostly gravel containing silt, atop which fine sand transported during flooding has accumulated to a maximum thickness of 70 centimeters. Throughout this layer of fine sand common reed spreads its roots and grows in colonies.

Table 1 shows the characteristics of the three locations in the research area and the common reed in these locations. Research here shows that the distribution of vegetation colonies in the channel is greatly affected by the frequency of inundation there; that the common reed grows in places where inundation never exceeds medium-scale flooding; and that since the subterranean stems grow only in the sand layer, reed growth is greatly affected by the thickness of this sand layer, more so than by such factors as the diameter of sand particles or the particle-void ratio.

Tabel 1: Characteristics of common reed and their location

	Station 1	Station 2	Staion 3
Height (m)	3.0	3.0	2.0-3.0
Density (reeds/m^2)	40-60	120-140	170-250
Stem Diameter (cm)	1.0	0.5-0.8	0.7-1.0
Subterranean stem depth (cm)	15-50	20	10-30
subterranean stem diamter (cm)	1.5-2.0	0.7	0.7-1.0
Bank conditions			
Partial diameter (mm)	Fine Sand (50cm deep) d_{10} 0.05 d_{30} 0.11 d_{50} 0.18	Fine sand (10cm deep) (30cm deep) d_{10} 0.03 0.04 d_{30} 0.09 0.12 d_{50} 0.19 0.30	Fine sand (50cm deep) d_{10} 0.03 d_{30} 0.08 d_{50} 0.15
Volume ratio (%)	Soil particles 35-45 Water 20-30 Air 30-40	Soil particles 40-50 Water 30-40 Air 20-30	Soil particles 40-50 Water 15-35 Air 25-40
Void ratio	1.2-2.0	1.0-1.7	1.0-1.2
Coeffient of permeability (cm/s)	5.5x10^{-3}	(10cm depp) (30cm deep) 1.1x10^{-3} 1.3x10^{-3}	0.84x10^{-3}

The Mechanism Of Erosion In Banks Where Common Reed, Etc., Grow

The state of reeds during flooding is shown in Figure 3(a) as can be seen in Photo 1 and Figure 3(b), after flooding the bank layer containing the reeds has an overhanging shape. This shape results from the soil-maintaining properties of the subterranean stems, which gives the bank's upper layers a relatively high erosive resistance compared to the lower layers. This shape is actually an important one as a technique for countering flooding. And although there are

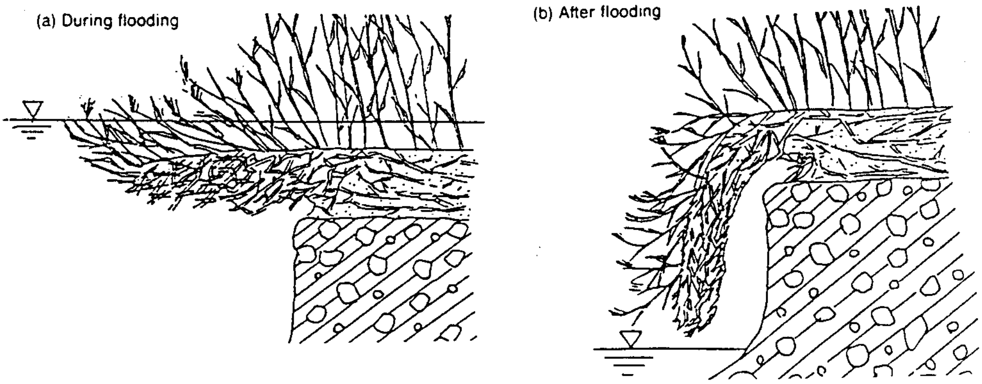


Figure 3: Shape of a bank with common reed

Phot 1: A bank with common reed after flooding

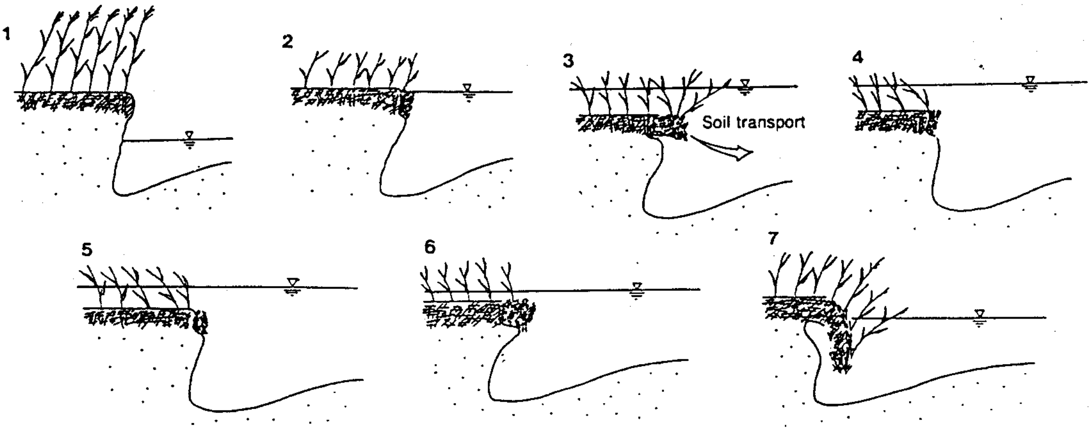


Figure 4: The process of erosion in banks with common reed

differences in lengths of overhang, the upper layers of banks with vegetation other than common reed generally take on this overhanging shape through erosion.

We will now describe the process by which a bank with reeds takes on this overhanging shape.

Figure 4 shows the process of bank erosion during flooding as changes over time. As the water level rises, the river bed undergoes scouring and the lower layers of the bank are eroded. The upper layers are firmly maintained by the subterranean stems and therefore relatively resistant to erosion. As the sand around the subterranean stems (which comprise a continuous mat-shaped mass in the upper layer) is washed away, the erosive force reaches the subterranean stem cluster. At this point, it is believed, the lower layers of the bank have been considerably eroded. These subterranean stems grow out from the nodes, and as force is concentrated on these nodes, some of them snap. As erosion in the bank's lower layers progresses even further, stability in the upper layer, which is comprised of soil and the reed cluster, is lost. As a result, the subterranean stems are overcome by the force and undergo tensile damage, and bank erosion continues. Repetition of this process results in bank erosion progressing further. Buoyancy acting on reeds causes them to exist in a floating state when the water level is high, but they and the soil hang down when the water level during flooding drops below the height of the flood channel. While the velocity at this water level is quite fast, the floating upper layer that covers the bank not only results in increased roughness around the bank and hence reduces velocity, but also serves to reduce the amount of water that directly strikes the bank.

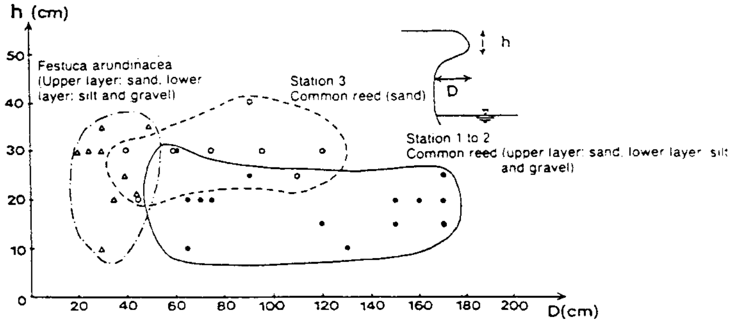


Figure 5 : Diagram of erosive shapes

A velocity of roughly 1.5 m/s was measured, when the water level dropped to the height of the flood channel.

After investigating the bank's overhanging shape, an erosive shape diagram (see Figure 5) was created with overhang length D as the horizontal axis and overhang thickness h as the vertical axis. The overhang length is the length that the sagging overhang would have were it straightened out.

This illustration hence shows that common reed can maintain the upper layers of the bank even if the lower layers are considerably eroded (although this depends on the conditions of the location and the thickness of the upper layer in which the roots have spread). The reason that longer overhangs were maintained at stations 1 and 2 than at station 3 is because of the greater depth of the main channel here, which allowed the upper layer to hang down with a longer overhang shape.

THE STRENGTH OF COMMON REED AS A BANK PROTECTION WORK

The longer the overhang is, the greater is the effectiveness in preventing bank erosion. But in order to plant common reed as a bank protection work, it is necessary to determine the velocity up to which the overhanging shape can be maintained; learning this would make it possible to determine the locations where common reed could be planted as a bank protection work. We therefore decided to measure the tensile strength of underground stems of common reed in an actual river, then use these measurements to determine the relationship between velocity and critical overhang length.

The Tensile Strength Of The Subterranean Stems Of Common Reed

An overhanging soil mass is held in place by the subterranean stems of common reed. In field research after flooding, overhanging soil masses that had collapsed from their own weight were observed in the water. We therefore roughly estimated the force that had to be assigned per subterranean stem to support these collapsed soil masses. To do this, we first calculated the weight of the collapsed soil mass from its volume and the specific gravity of the soil mass containing subterranean reed stems, then divided by the number of subterranean stems left in the soil mass in order to determine the tensile strength per subterranean stem. The value obtained for strength per subterranean reed stem was an average of 23.0 Kgf. Then, in order to accurately measure this tensile strength, we built the in-situ tester shown in Photo 2.

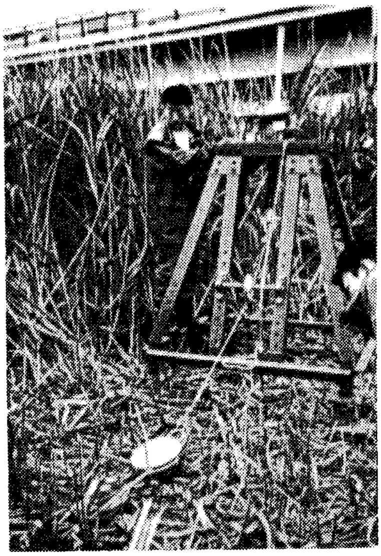


Photo 2: In situ tester

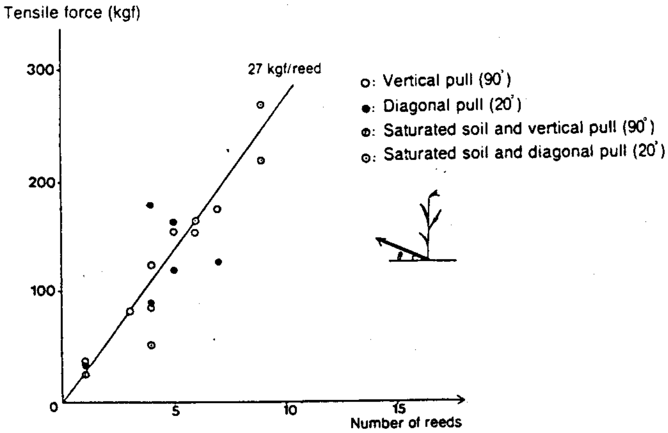


Figure 6: Results of measurement of common reed subterranean stem tensile strength

This tester was comprised of a scale attached to an arm suspended from the truss-structure base. A reed stem protruding from the surface was clamped at its base between two testing plates, then pulled by turning the handle to apply tension. The angle of pull can be changed by raising or lowering the position of the pulley on the base. This experiment was performed four times in different locations: with a vertical angle of pull; a diagonal (roughly 20 degrees from horizontal) angle of pull; with unsaturated soil; and with saturated soil. Reeds were always pull out by the node of the subterranean stem. The measured results for tensile strength are shown in Figure 6.

Figure 6 shows that the tensile strength of common reed is approximately 27 kgf per reed, a value that remains roughly constant regardless of the conditions. The reason for this is that the flexibility of the subterranean stem enables the reed to change its orientation in accordance with the direction of external force.

Erosive Critical Velocity For Banks With Common Reed

Now let us estimate the erosive critical velocity (the velocity at which an overhanging bank breaks) from the balance between the external force produced by the current and the tensile strength of the subterranean reed stems. Figure 7 shows our model for overhanging banks.

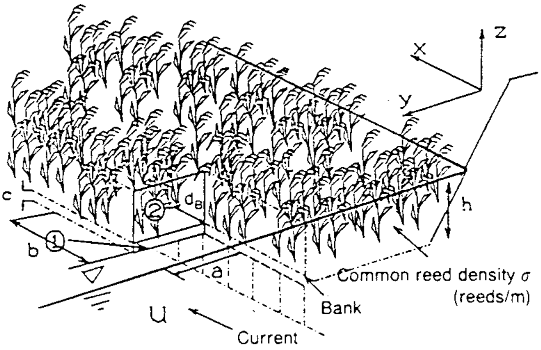


Figure 7: Model of overhanging bank

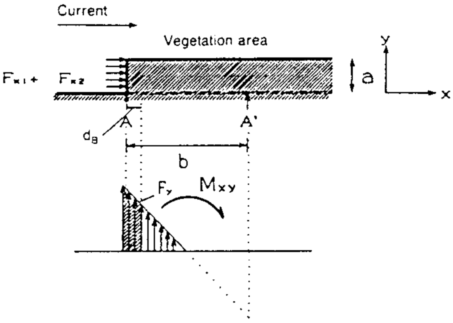


Figure 8: Distribution of bending stress in x-y section

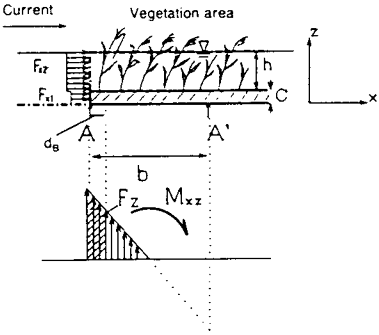


Figure 9: Bending stress in x-z section

Equations (1) and (2) below are used to represent the external forces produced by the current, i.e., 1) F_{x1}, the fluid force acting on the front surface of the overhang, and 2) F_{x2}, the fluid force acting on the reeds in the overhang.

(1) $$F_{x1} = C_D \cdot \rho \cdot a \cdot c \cdot u^2 / 2$$

(2) $$F_{x2}(x) = C_D \cdot \rho \cdot \phi \cdot \sigma \cdot a \cdot h \cdot \int_0^x u'^2 / 2 dx'$$

where C_D is the drag coefficient (1); ρ, water density; ϕ, reed stem diameter; σ, reed density; a, overhang length; u, approach velocity of the main current; and u', velocity inside a reed colony (u·exp [-2.15x]). Reed colony velocity u' attenuates exponentially in the downstream direction, and the attenuation constant (2.15) was determined through hydraulic model experiments. b is the distance it takes for the velocity to sufficiently attenuate. (Here, this experiment was performed with a distance b of 0.8, 1.0 and 1.2 meters, in which the velocity attenuated 82%, 90% and 93%, respectively.)

The fluid force in the downstream direction that acts on the most upstream break position of the overhang (length in downstream direction: d_B) is represented with the following equation.

(3) $$F_x = F_{x1} + F_{x2}(d_B)$$

Fluid force F_x causes the overhanging bank to undergo bending moments in the xy surface and the xz surface (M_{xy}, M_{xz}; see Figures 8 and 9). These moments are represented by the following equations.

(4) $$M_{xy} = \{F_{x1} + F_{x2}(b)\} \cdot a / 2$$

(5) $$M_{xz} = F_{x2}(b) \cdot (c + h) / 2$$

Bending stresses F_y and F_z (produced by these bending moments and occurring on A-A' plan) are assumed to have triangular distribution.

Also, this distribution of the bending stress allows us to determine the stresses in the y and z directions acting on the portion d_B as shown below.

(6) $$F_y = \int^{d_B} 6M_{xy}(1 - 2x / b) / b^2 dx$$

(7) $$F_z = \int^{d_B} 6M_{xz}(1 - 2x / b) / b^2 dx + \{(\rho_m - \rho) a \cdot c + (\rho_v - \rho) a \cdot h \cdot \sigma' \pi \cdot \phi^2 / 4\} g d_B$$

The above enables us to determine the resultant force F vector (F=(Fx,Fy,Fz)) of the external forces acting on d_B, the area where breakage occurs, and from the balance between this external force F and the tensile strength of the subterranean reed stems at the breaking surface, we can determine u_{cr}, the critical velocity for the overhang.

(8) $$R = r \cdot \sigma' \cdot d_B$$

(9) $$u_{cr} = [2g \cdot r \cdot \sigma' \cdot d_B / \{\rho \cdot C_D a \cdot (K_1^2 + K_2^2 + K_3^2)^{1/2}\}]^{1/2}$$

$$K_1 = c + \phi \cdot h \cdot \sigma \int_0^{d_B} f(x')^2 dx'$$

$$K_2 = 3a / b \cdot d_B / b \cdot (1 - d_B / b) \cdot \{c + \phi \cdot h \cdot \sigma \int_0^b f(x')^2 dx'\}$$

$$K_3 = 3d_B / b \cdot (1 - d_B / b) \cdot \{\phi \cdot h \cdot \sigma (c + h) \int_0^b f(x')^2 dx'\} / b$$

where u_{cr} is the critical velocity; r, tensile strength of the subterranean reed stem (27 kgf/reed); σ', number of subterranean reed stems per unit of downstream distance; and f(x')=exp (-2.15x').

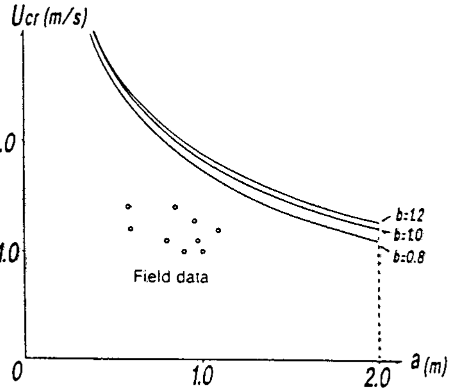


Figure 10: Length of over hanging bank (a) and critical velocity

Figure 10 is a comparison of field data (for the 40-km point on the Tama river) and the results for calculations to determine critical velocity u_{cr} when overhang length a is the variable and the submerged depth (h) is one meter. The field data has been plotted with the value for a determined through bank surveying after flooding, and a value for u determined by measuring the velocity during flooding.

The Figure shows that the calculated values correspond to the results of field measurement. The reason why the overhang length (a) ends at 2.0 meters is that when the difference in height between the flood channel and the water level in the main channel is greater than two meters, bank stability becomes extremely low and, as a result, the bank-protective properties of the reeds becomes ineffective. According to this graph, when the velocity is 2 m/s or less, an overhang length of 0.8 m or more can be maintained, and this overhang can be counted on to mitigate bank erosion.

THE EFFECTIVENESS OF DITCH REED IN ATTENUATING THE ENERGY OF BOAT WAVES (Fukuoka et al.,1992)

The increase in pleasure boats in rivers has been accompanied by illegal docking and other new problems, one of which is the waves produced by commercial and pleasure boats. When a boat passes, the waves it produces negatively affect areas used by people enjoying the river and also cause considerable bank erosion.

In North America and Europe, where boat usage is advanced, research and damage reduction countermeasures concerning bank erosion caused by boat waves are already being carried out.(Hemphill and Bramley, 1989). In many places in Southeast Asia, however, boats are also used as a means of transporting people, and this river transportation causes extensive bank erosion that has resulted in serious land- and housing-related problems.

Ditch reed grows along the banks in the downstream reaches of some rivers. If these ditch reed colonies can keep the water's surface calm and prevent erosion by canceling out the energy of boat waves, then this reed's flood-control and environmental functions should be able to contribute to both conservation and restoration. In order to determine the extent to which ditch reed colonies reflect and attenuate the energy of boat waves, a field experiment was performed in

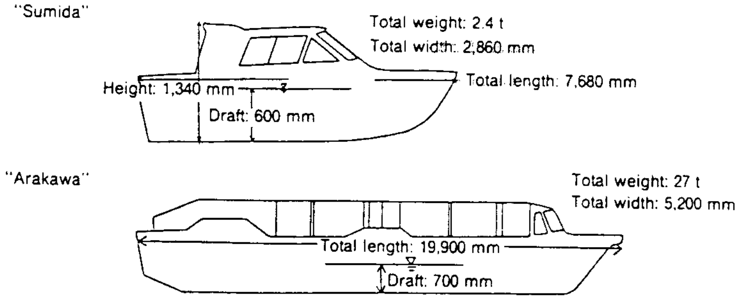


Figure 11 :River patrol boats

Table 2: Characteristics of waves created by boats

	Patrol boat Sumida	Patrol boat Arakawa
Wave height (cm)	13-23	6-52
wave celerity (m/s)	1.85-2.67	2.11-3.56
Wave period (sec)	1.7-2.4	2.2-4.6
Wave length (m)	5.5-6.6	5.1-15.9
Number of waves	10-12	10-14

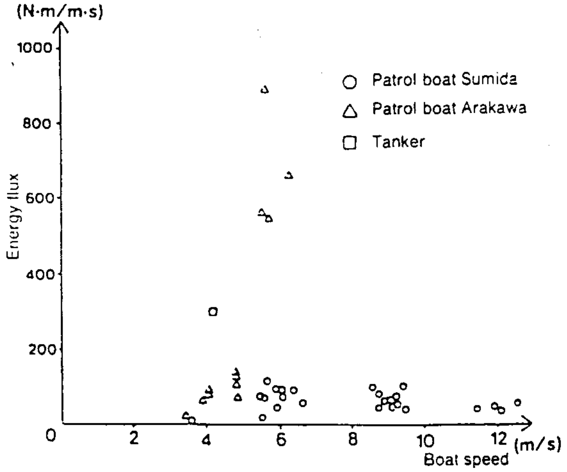


Figure 12: Relationship between boat speed and wave energy flux

the lower reaches of the Ara river, and an experimental wave tank was also used. The site of this research was the right bank at the 14.5-km point. Two boats were used: the "Sumida" and "Arakawa," both of which are patrol boats belonging to the Ara River Lower Reach Construction Office. The dimensions of these boats are shown in Figure 11.

As the test area is in a tidal region, the current velocity is nearly negligible. Depth is approximately six meters where boats pass and 1.0 to 1.5 meters at the front edge of the reed colony.

The dimensions of the waves measured are shown in Table 2, while Figure 12 shows the relationship between boat speed and the average transmitted wave energy per unit crest width and unit time ($W= 1/8\ \rho g H^2 C_g$). The Sumida, traveling at a speed (Vb) of 6 to 9 m/s, produces waves with a large amount of energy that surged towards shore, when traveling at 6 m/s. The waves produced by the Arakawa had a far larger energy flux than those of the Sumida, and maximum energy flux.

Where ditch reed colonies grow, wave energy is comprised of reflected wave energy and transmitted wave energy, with the rest being energy lost inside the ditch reed colonies. The energy conservation law is exprèssed with the following equation.

$$W_I = W_T + W_R + W_L \tag{10}$$

where W_I, W_T and W_R is the average transmitted wave energies per unit crest width and unit time by, respectively, incident waves, transmitted waves and reflected waves; W_L is the average energy loss per unit crest width and unit time. Equation (10) can be changed to equation (11) when the reflection coefficient K_R is H_R/H_I, the transmission coefficient K_T is H_T/H_I, and K_L is the coefficient of energy loss.

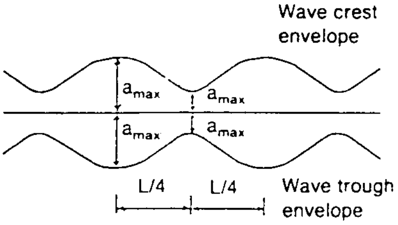


Figure 13: Envelope of partial standing waves

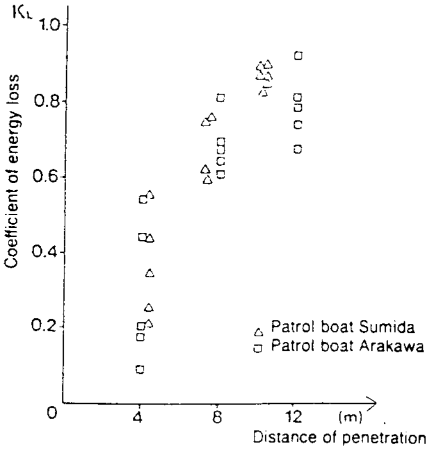


Figure 14: Coefficient of energy loss of boat waves in ditch reed cluster

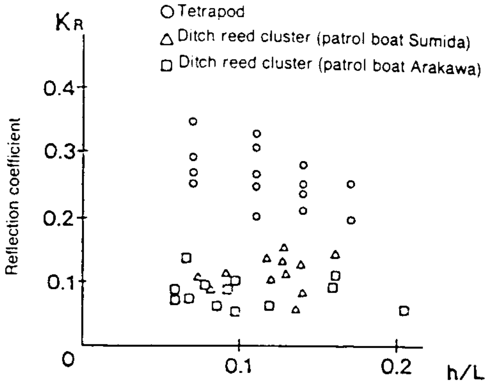


Figure 15: Wave reflection coeffient in ditch reed cluster

$$1 = K_R^2 + K_T^2 + K_L \tag{11}$$

In the ditch reed colony waves reflect incompletely, resulting in crests and troughs like those shown in Figure 13. Measuring the envelope of the crest and troughs of such waves, the reflection coefficient K_R was determined with Healy's method.

$$K_R = \frac{a_{max} - a_{min}}{a_{max} + a_{min}} \tag{12}$$

The coefficient of energy loss in ditch reeds (K_L) is determined with equations (11) and (12). Figure 14 indicates the relationship between the distance that a wave travel through the ditch reeds and the resulting energy loss coefficient. This coefficient of energy loss approaches 60 to 80% when the wave penetrates eight meters into a ditch reed colony, thus demonstrating that the effectiveness of ditch reeds in attenuating wave energy is considerable even with large waves like those created by the Arakawa, and that the erosive force of waves reaching the bank behind the ditch reed colonies is nearly zero. Also, Figure 15 shows that the coefficient of reflection of waves in a ditch reed colonyis merely 0.05 to 0.15, lower even than that produced by tetrapods (0.2 to 0.35), which shows that ditch reed colonies are capable of sufficiently reducing reflected waves. This means that ditch reed colonies can quickly return the water's surface to a calm state after the passage of a boat, and that they are also effective in the area of water-surface utilization.

The above establishes that the energy-attenuation and reflective characteristics of ditch reeds are sufficiently high in comparison with other artificial wave-dissipation structures. The effectiveness of ditch reed in mitigating bank erosion and returning the water's surface to a state of calm is definitely worth use even from a flood-prevention standpoint, while at the same time, ditch reed's natural environmental functions, in combination with this effectiveness, can be used to create a river environment that is both safe and rich in natural beauty. The administration, conservation and restoration of ditch reed in rivers is therefore called for.

CONCLUSION

The objective of this research was to investigate the use of common reed and ditch reed as bank protection works. The principal conclusions reached through the field research, field testing and experimentation performed for this research are as follows.

1. It was demonstrated that common reed growing along a river bank can maintain the bank's overhanging shape during flooding; also demonstrated was the mechanism by which bank erosion is mitigated during flooding and in the flood attenuation stage by the overhang covering the banks.
2. The tensile strength of common reed was determined, as were the critical velocities for using these reeds as bank protection works.
3. It was confirmed that the growth environment of common reed is affected principally by frequency of flood channel inundation and other hydraulic characteristics, and by the characteristics of the soil in which their roots grow.
4. The effectiveness of ditch reed in attenuating the transmitted energy of waves produced by boats was also demonstrated.

REFERENCES

Fukuoka,S. and K.Fujita(1990): Hydraulic effects of luxuriant vegetations on flood flow, *Report of Public Works Research Institute*, pp. 129-192, (in Japanese).

Okuda, S.(1976): Vegetation maps in the Tamagawa River Basin, *Report of Natural Environment in the Tamagawa River Basin*, (in Japanese).

Ministry of Construction (1973): Vegetation of the Tamagawa, (in Japanese).

Fukuoka, S.(1993): Flood control measures that utilize natural functions of rivers, *Proc. XXV Congress of IAHR*, Vol.VII, pp.71-78.

Fukuoa, S., K.Komura and A.Watanabe.(1992): Effect of marsh reeds in the river on the energy dissipation of boat-generated waves, *Proc. of Hydraulic Engineering, JSCE*, Vol. 36, pp. 713-716, (in Japanese).

Hemphill, R.W. and M.E.Bramley(1989): Protection of river and canal bank, *CIRIA Butterworth*, pp. 18-22.

DESIGN AND MANAGEMENT OF BANKS: AN INTEGRATED APPROACH.

J.O. Reinhold & J.T. Klein Breteler
Ministry of Transport, Public Works and Water Management
Directorate-General for Public Works and Water Management
Road and Hydraulic Engineering Division
P.O. Box 5044
2600 GA Delft
The Netherlands

ABSTRACT

There are many many river and waterway bank areas in the Netherlands. Most banks require protection against erosion by currents, wave attack caused by ships and wind or trampling by cattle. Until recently, many Dutch bank protections were artificial and steep, with few possibilities for vegetation and fauna to settle, reproduce, shelter, feed and migrate.

Fortunately, changes in thinking have resulted in a government supported policy of applying an integral approach, in which a water system as a whole is taken care of, including the banks and their adjacent surroundings. This approach takes into account the various functions of a water system, and its physical, chemical and biological components.

In 1989, the Road and Hydraulic Engineering Division has started a research, consultancy and knowledge proliferation program on banks of fresh water systems, in which experts in ecology, hydraulics and materials technology are working together. In the program of the Road and Hydraulic Engineering Division, rivers, shipping canals and fresh water lakes are dealt with. The different research projects link up with the several steps in the planning process of banks.

In the first step the initiative is taken and inventories are made of the current situation. This includes the management situation like dedicated functions and ownership, the position of the regarded location within the water system and the state it is in, concerning amongst others the geomorphology, present vegetation and fauna and connections with nearby nature reserves.

This step ends with the formulation of the target: the desired state of the bank. To support the execution of this step, we described ecological reference states for natural and ecologically-sound banks.

The second step starts with formulating the design principles. This is achieved by combining the knowledge of various disciplines. In our program many research projects focus on providing the necessary information.

This second step results in design alternatives and a final choice.

The third step deals with the necessary management schemes. In one of our research projects we study the effects of different ways of managing vegetation.

After a bank project has been realised a plan for monitoring and evaluation has to be made and put into practice. In support of this, we compiled guidelines with directives for monitoring the biotic and abiotic aspects of banks and we developed a method for measuring vegetation in narrow zones along inland waterways.

KEY-WORDS: bank / water system / design / integrated management / monitoring / evaluation / vegetation / fauna

INTRODUCTION.

Seventy procent of the land area of the Netherlands lies below sea level, and many projects have been set up to protect the land from flooding. But artificially changing the natural state of water systems can cause many problems.

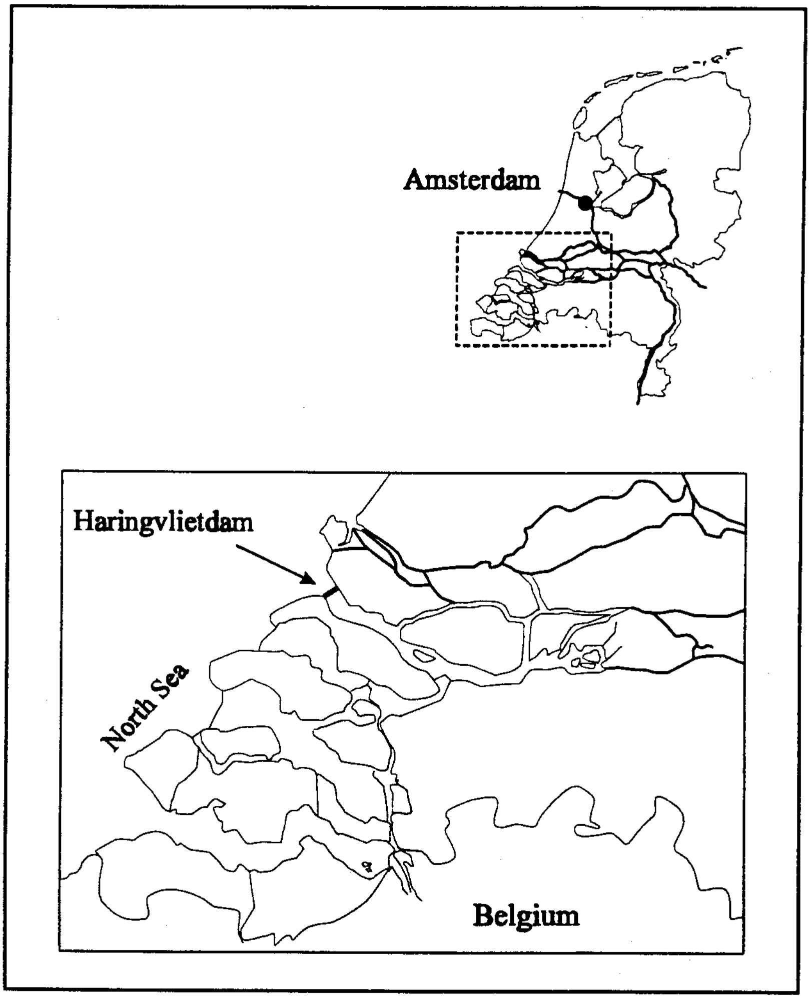


Figure 1. The Netherlands, a delta region of several rivers.

An example:

In 1970 the construction of the Haringvlietdam was completed. It was build as one of the protective works that have to ensure the safety of the Netherlands against inundations, such as in 1953. As a result, the Haringvliet, a part of the Rhine-Meuse estuary became isolated from the North Sea. However, water system managers had not sufficiently realised that:

- this dam is a barrier for anadrome fishes such as the Salmon, the Allis shad and the Twaite shad. Numbers of these fish species have strongly decreased since 1970 (de Nie, 1996).
- the tides disappeared, therefore wind wave attacks have more grip on the bank and cause erosion.
- polluted silt deposits in the area. Effects of this pollution are measurable in the breeding success of Cormorant (Dirksen et al., 1995).

If the managers of water systems had taken into account the problems caused by their solution, they probably would have chosen another solution to protect our country. However, it was not in the spirit of that time to think about an integrated solution.
Water systems include banks, floodplains and dikes are now regarded more multi-functional (Ministerie van Verkeer en Waterstaat, 1989). In addition to functions as water supply, shipping and fishing industries, aspects such as preservation of our nature and the recreational value of such areas are also growing in importance.

Because of the changed mentality more often an integrated approarch is used in bank construction works in which economic, ecological and social consequences are incorporated.
The consequence of this more integrated approach is that a growing number of relevant aspects has to be taken into consideration. To make the right decisions a clear and structural approach is necessary.

The Road and Hydraulic Engineering Division (DWW) is a governmental organisation that provides advice on how to construct and manage banks of water systems. To make sure that the best possible bank is constructed in a water system the Road and Hydraulic Engineering Division designed a step by step method with different phases to come to an optimal bank construction (Simons et al., 1994).
During the development of this method gaps of knowledge were found. After completing the planning process we now try to fill the gaps and develop instruments to make the decision steps during the process easier.

This article is an introduction to the planning process. In special text boxes we describe the research and instruments we have developed to support each planning step. Especially aspects related to flora and fauna will be treated.

PLANNING PROCESS FOR INTEGRATED BANK MANAGEMENT

The planning process deals with six subsequent phases, each divided in one or more steps (figure 1).
The six phases are:
1. developing an objective management vision for a bank
2. describing which aspects of the bank have to be rearranged
3. describing a plan to execute these rearrangements
4. describing a plan to maintain the developed bank
5. describing a plan to monitor the development of the bank
6. describing a plan to evaluate the whole process.

Developing an objective management vision

A vision on the targets of a water system and its bank provides the possibility to compare the existing situation to a desired situatioHowever, a vision has to be realistic and there has to be acceptance for this vision by different groups in society. Before making a vision, research should take place to the existing and potential values of a bank. Together with the different groups of society a (for all groups) acceptable vision should be made based on the research.
The complexity of this stage makes it necessary to divide it into three steps (figure 2):

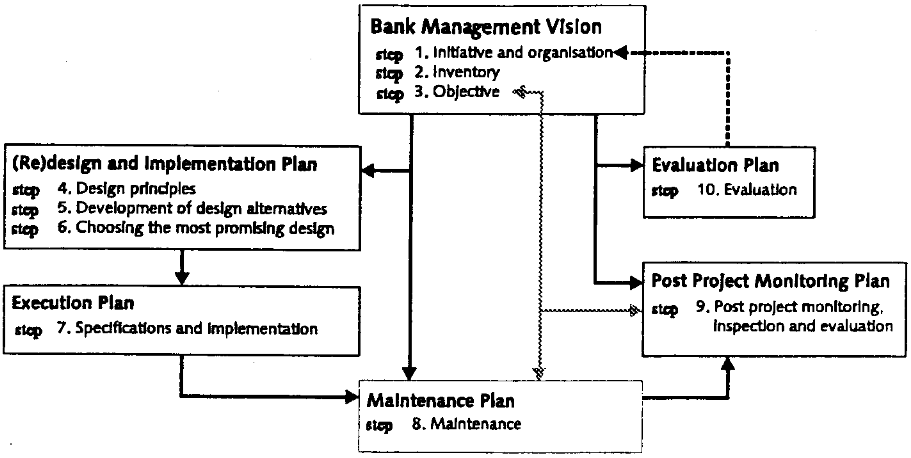


Figure 2. The planning process for integrated management of banks

Step 1: Initiative and organisation.
At the beginning of the planning process a person or organisation has to take the initiative. This initiator starts the process and sorts out:
1. the persons and organisations who have to take part in the planning process.
2. the financial possibilities.

Together with persons and organisations involved, the initiator produces a plan describing which decisions have to be taken and at what moment these decisions take place. The opportunities for comment have to be well planned and described in this step.

The initiator also has to investigate the financial possibilities for formulating the whole planning process and the possibilities for restoration and maintenance in the future.

Clearness and frankness to all participants and others characterise this stage and may well prevent delays in further steps.

Step 2: Inventory
An inventory of ownership of the bank and management duties, visualised on a map gives insight into the responsibility of each party involved.

For the vision on the management of banks information about the functions of the bank and direct surroundings is necessary. Does the bank have an important economic function because of shipping or farming in the floodplains? Or is the safety against flooding the main function. In a country with a land

area of 70 % below sea level this function has to be well taken into account.
In our small country with 15 million inhabitants the importance of nature is growing. Banks have important functions; as living habitat or maybe as a corridor between nature reserves.

Changes in the importance of functions must be considered, since they can result in large changes in the opportunities for the design and management of the banks.

Having determined all relevant functions, an inventory of abiotic and biotic components is required. What is known about aspects such as: land-use; hydrology; geomorphology; quality of water, (deposited) silt and soil; value of the landscape; cultural and historical values; ecological values; processes like desalination, dehydratation, acidification.
This information is an important basis for the next steps. Without the recognition of the (autonomous) processes it is hardly possible to set up the objectives.

> A useful instrument for this step, which we have developed at DWW is the DI(mensioning) PRO(tections) program. An important function of the water systems in the Netherlands is to offer the possibility of transportation of goods by shipping. Consequently, shipping puts strong demands on the lay-out and the maintenance of water systems. DIPRO can calculate the hydraulic loads on banks, structures and beds, caused by shipping. It runs on a PC. DIPRO is the result of two decades of research, in laboratory as well as in practice.

Step 3: Objective
In this step a general decision on the targets for the bank has to be described, taking into consideration the functions of the bank. With the results of steps 1 and 2 a well weighed up description of the objectives can be made.

Homogeneous parts of the bank have to be confronted with the list of relevant functions to establish which functions determine the future lay out of these parts.
To describe the function nature, reference situations are used; situations which occur when man would not interfere. Especially the situation of a water system at the beginning of 1900 is popular because then human influence was scarce and the situation is quite well described. Maps are available with the geomorphological and hydrodynamical situation, aspects which conduct the potentials of flora and fauna.

To achieve the target, changes in management and/or design have to be made; so for each homogeneous part these changes have to be listed. Because every change will have its own influence on the target, a priority of these changes has to be made.
To make sure the best solution is described all possible alternatives in changes of management and design have to be considered.

At this stage the financial possibilities for changing the design or management should be clear. Also the potential financial situation after the reconstruction has to be considered. If financing of the maintenance will not continue in the future no bank design with high maintenance costs should be chosen.
Decisions also have to be made about the ownership of the bank and the responsibility for the maintenance in the future.

Information about the current distribution of animal and plant species can be provided by a large number of flora and fauna societies.

Figure 3. Fauna in banks.

In the Netherlands almost every taxon of species has its own society. There is for example a society of birds, bats, butterflies, moths, snails, mushrooms, mosses and lichens. If someone needs information about the species in an area he or she pays these societies for their information. Regarding information on fauna, DWW is a frequent 'client' of these societies.

> Within EKOS, a PC program for analyzing banks used by animal species, rough information on the Dutch distribution pattern of characteristic bank species is described. For species still colonizing a potential distribution is presented based on the favourite habitat. In combination with information about the landscape surrounding the bank a decision can be made which species could be target species.

plan to rearrange the bank

Step 4: Formulating design principles

In a (re)design for the bank the preconditions and objectives laid down in the vision are translated into design principles. This is achieved by applying knowledge of various disciplines.

The various demands, for example by shipping, flora, fauna, recreation, should be given different priorities, which should be indicated. The level of priority depends on the location of the bank and its assigned functions and will have to provide the criteria for testing the design at a later stage.

This is one of the most difficult steps, in particular when it comes to formulating design principles for flora and fauna. At DWW, we have come up with two useful tools.
> EKOS provides the user with information about the main habitat variables and describes the relation between the values of the variables and suitability for the species using HSI-models. By weighing the suitability of each variable against another, it is also possible to predict an average suitability.
So if a bank manager has selected his target species he gets a list of the most important variables. When he has measured the values of these variables in the field and puts them into EKOS, it calculates the average suitability and also calculates which variables are responsible for this average suitability. If he wants to improve the bank for these species he should work on the limiting variables.
> For a period of five years we have studied the relationship between bank design and the occurrence of bank vegetation. We did this mainly along inland waterways. Indeed we have been able to find a relation between bank design and occurring vegetation types. Abiotical components such as width, slope, forces of wave attacks and flow determine the possibilities for bank vegetation.
As a result of our research on this relation, we are able to advise on bank designs which will result in a desired vegetation.

Step 5: Alternatives in design and/or management.
Knowing the functions, the targets of a bank and the design principles, alternatives of bank design and management can be prepared.

Figure 4. Bank protection consisting of stone protection and shallow pool.

During this phase various alternative designs can be drawn up, e.g. by using different widths for the bank, different types of bank protection and different materials.

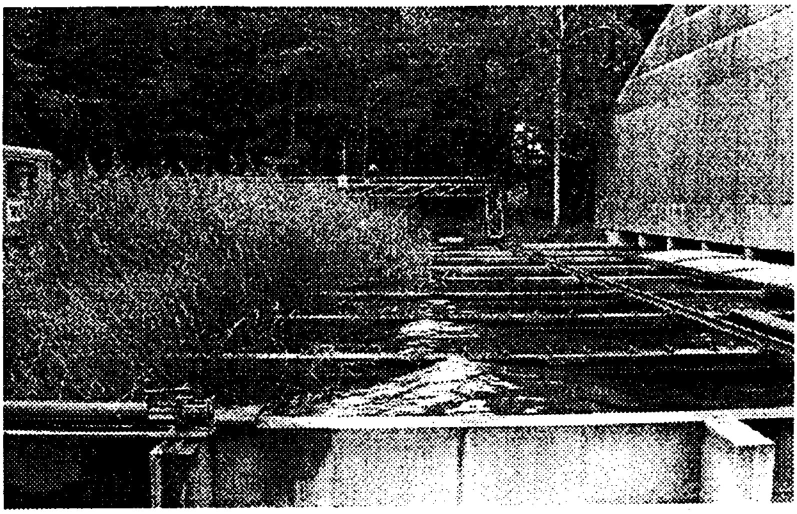


Figure 5.Test of the protective properties of reed and bulrush at the Delft HydraulicsLaboratory

Figure 6.Test of the penetrability of geotextiles

Instruments developed at DWW deal with the lay out of protective structures and the use of materials.

> With the forementioned PC program DIPRO, DWW offers designers the possibility to dimension the structural elements of many kinds of bank protections, based on the calculated hydraulic loads caused by shipping.

> A nowadays much applied bank protection along inland waterways consists of a fixed protection, bordering a shallow pool that changes into the bank itself. The shallow pool is meant to provide a habitat for all kinds of plants and animals, and offers the bank protection a much more agreeable outlook, more befitting the surrounding environment. An important aspect of this type of construction is the exchange of water between waterway and pool, to keep the quality of the water in the pool on a sufficient level. Therefore, gaps are made in the fixed construction, through which water exchange is possible. Furthermore, these gaps can be used by fish or other animals to get in and out of the pool.

The distance between these apertures as well as the effective area largely governs the rate of water exchange, and consequently fluctuations of the water level in the pool.

In order to make a proper design, regarding dimensions of the gaps and the distance between these, we have developed numerical models, PLONS 1-dimensional for relatively narrow pools and PLONS 2-dimensional (Vermeulen, 1996). The models are governed by the following processes:

- Water exchange is possible by the water level depression caused by a passing ship
- During the passage time of a ship, the water level in the waterway is lowered and due to the head difference between pool and waterway, water flows out of the pool.
- After the ship's passage, the level in the waterway is restored, resulting in an inverse flow, since the water level in the pool has decreased during the stage of outflow.

The processes are modelled using one- and two-dimensional partial differential equations, that describe non-stationary flow in open channels, combined with suitable discharge-head relations for the applied gaps.

> Reducing the amount of materials used in construction works is one of the targets of our government. Re-used materials are therefore growing in importance. Also materials with low environmental impact are preferred. DWW researches how to reduce the flow of materials and how to use more ecologically sound materials, like for instance reed zone plants.

From practice and several earlier studies it has become clear that reed zone plants can effectively reduce wave attack by wind or shipping and subsequent erosion of the subsoil with their stems and root system. However, knowledge proved to be insufficient for defining the circumstances under which reed zone plants can effectively and lastingly protect the slope they grow on. Therefore we commissioned an unique research project to the De Voorst branch of Delft Hydraulics Laboratory. During four years, reed and bulrush have been tested, undergoing very frequent wave attack in a specially adapted, outdoor wave tank facility. The whole year round, waves attacked the plants growing on a four meter wide, unprotected berm. During the growing season and in winter time, the vitality of the plants was studied, and the amount in which the plants reduced the incoming wave heights and the erosion of the sandy subsoil. The research resulted in design rules for the circumstances under which reed zone plants provide an effective protection against wave attack. With these it is possible to determine whether the plants can protect the banks of an inland waterway or a lake by theirselves or whether an extra defensive structure is needed and if so, which dimensions this should have.

> Thus, in banks vegetation can be used as protective material. However, in a newly constructed bank the vegetation needs a few years to develop. In the meantime protection of the bank is needed against erosion. Geotextiles can provide this protection by covering and armouring the subsoil. Once the geotextiles are penetrated by riparian plants, the root system of the plants can take over the geotextile's function. So in many cases, a temporary geotextile may suffice. The possibilities for riparian plants to penetrate the geotextile have become an important characteristic. However there was no test method available to determine the penetration resistance of geotextiles. Research by DWW on this resistance led to two computational methods. For woven geotextiles the pore size is as the most determining factor, for non-woven geotextiles the weight of the geotextile. With these two known variables, penetration by for instance reed shoots can be predicted.

Step 6: Choosing the most suitable alternative.
During this phase all parties involved will have to assess the designs and make a choice between the various alternatives. In making the best choice the priorities and criteria of step 4 and the costs and benefits need to be incorporated. The design which best answers to the target formulated in the management vision and is still affordable, is the obvious choice.

Plan of execution

step 7: Elaborating the most suitable alternative
When the choice for a design has been made and (if necessary) the choice has been approved, the specifications will need to be drawn up and the work put out for tender, after which the work can start. Both in the specifications and during the implementation, details on the role of ecology in the design require special attention. It may be a bit pretentious to speak of an implementation plan, but these details should indeed be thought of and laid down.

Plan for maintaining the desired situation

step 8: Maintenance plan
A maintenance plan mentions all the activities that are relevant to the entire management plan that should be carried out during the period planned. They include both the periodical maintenance that is carried out regularly and the occasional and/or the more fundamental maintenance.
The nature function is not optimal directly after the intervention. The colonisation of a bank with plant and animal species may take a few years. In the meanwhile maintenance is necessary to reach this optimum.

> DWW is carrying out research on ecological aspects of maintenance.
> \> The most limiting factors for a desired vegetation are taken into account in step 4. Besides a proper design also maintenance of the vegetation is necessary to reach this desired vegetation. However, the influence of different measures is almost unknown. Research takes place on the impact of changes in water level and mowing on riparian vegetation. Results are at this moment not available.
> \> Any maintenance that has large impact on animals using banks, can be found in the PC program EKOS. In this program maintenance is one of the variables used for the described Habitat Suitability Indices.

Plan for monitoring

step 9: monitoring
After completion of the bank construction or after adjusting the maintenance, the development of the bank should be monitored. Monitoring makes use of a number of measurable parameters and consists of at least two successive measurements carried out in one location and each time using the same method. Monitoring should be well documented in order to test the development of the bank against the objectives set (evaluation). To ensure proper monitoring and evaluation it is necessary that these objectives are quantifiable and described in as much detail as possible.

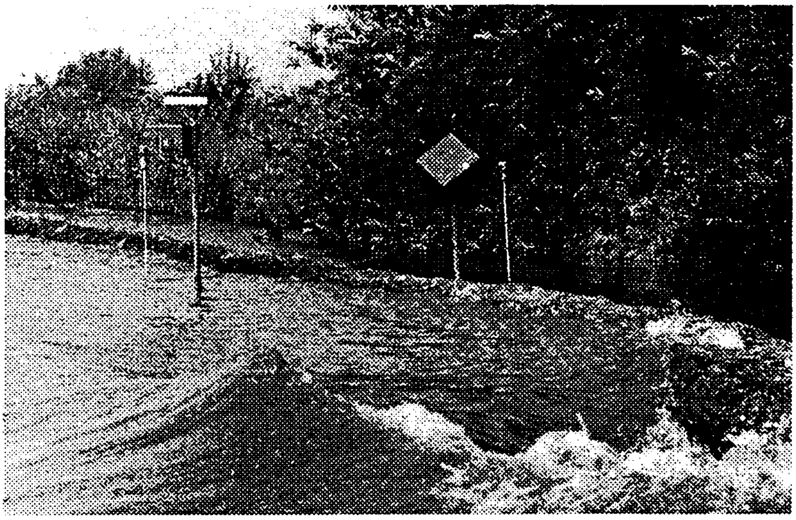


Figure 7. Monitoring the reduction of wave attack on the bank vegetation

Managers of banks have not always realised that biological monitoring and evaluation differs from technical monitoring and evaluation. The quality of the structural parts of a bank only deteriorate after the building of the construction. Biological aspects fluctuate in quality within the year and during the years and are at least in the first years not optimal.

DWW has developed a method for planning the monitoring and the evaluation to prevent the collecting of useless data. Important aspects of this method are the determination of the objectives of monitoring, the choice of relevant variables and monitoring techniques in relation to availability of money and time and the storage and analysis of data.

We have come up with a new method to monitor the development of typical bank vegetations. With this simple method data are collected and analysed to assess the extent of natural quality of the vegetation.

Monitoring of fauna is still hardly done. Until now the function nature in bank design had mainly vegetation aspects. Fauna aspects are rather new in this process.

In the future monitoring of fauna will be more common. Monitoring of fauna is more difficult than monitoring flora. For each group and sometimes even every species different methods have to be used. The manager of a bank will need assistance to make a good (fauna) monitoring plan. We can offer this assistance.

Plan of evaluation.

Step 10: Evaluation

Monitoring and evaluation are not the end of the process. Also the overall management should occasionally, for instance at the end of each (mostly ten-year) plan term, be assessed with regard to the objectives mentioned in the management plan. The objectives themselves should be assessed and adjusted if necessitated by social developments. Such assessments may have consequences for the design and maintenance of the bank(s) involved. The precise nature of these consequences can be determined by using the action plan. Even feedback of this kind, therefore, benefits from a systematic and planned approach.

CONCLUSIONS

When watersystems (including banks) are regarded more multifunctional an integrated approach of designing and planning seems logic. The method leads to well troughed descisions during the planning process and the questions for advise to our division are more specific. People know more precisly which questions have to be solved and they also know why they want to solve that question.
The use of this step by step planning process gives our division the opportunity to develop on forehand new knowledge and instruments. We know which steps have to be taken and it is possible to predict where problems arise. Research on this problems and developing instruments can take place before we are asked for assistance. For both an efficient way of working.

REFERENCES

Cappendijk, A.M. (1994) A manual of PLONS (In Dutch)-DWW-94-294, Rijkswaterstaat DWW, Delft.

Coops, H., R. Boeters and H. Smit (1991) Direct effects of wave attacks on helopfyten. Aquatic Botany 41: 333-352

Dirksen, S., T.J. Boudewijn, L.K. Slager, R.G. Mes, M.J.M. van Schaik and P. de Voogt (1995) Reduced breeding success of Cormorants *(Phalacrocorax carbo sinensis)* in relation to persistent organochlorine pollution of aquatic habitats in the Netherlands. *Environ. Poll.*88 pp.119-132

Ivens, E. (1993) Penetration through geotectiles by shoots, roots and rhizomes of the common reed (Phragmite australis): a pilot study. In Limnologie Aktuell vol 5 "Seeuferstörung und Seeuferrenaturierung in Mitteleuropa" Gustav-Fisher Verlag, Stuttgard.

Ministerie van Verkeer en Waterstaat (1989) Water in the Netherlands: a Time for Action. Summary of the Third National Policy Document on Water Management. -Staatsuitgeverij, Den Haag.

Nie, H.W. de (1996) Distribution maps of fresh water fishes in the Netherlands (in Dutch). -Media Publishing, Doetinchem.

Simons, H.E.J., J.L. Koolen and G.J. Verkade (red.). Natuurvriendelijke oevers. CUR Handboek 168. Gouda

Vermeulen, I. (1996) Ecologically sound banks; a 1 and 2-dimensional method to calculate the refreshment (In Dutch). Report Technical Universtity Delft 68pp.

DYNAMICS AND RESTORATION OF LAKESIDE REEDBELTS IN A PREALPINE LAKE (LAKE CONSTANCE, GERMANY)

Wolfgang Ostendorp(1)
Botanisches Institut, Universität Greifswald

Herbert Stark(2)
Naturschutzbund Deutschland, Ortgruppe Konstanz

Priska Krumscheid-Plankert(3)

Annelie Pier(2)
Naturschutzbund Deutschland, Konstanz

(1) Botanisches Institut, Universität Greifswald, D-17487 Greifswald, Germany, (2) Naturschutzbund Deutschland, Ortsgruppe Konstanz, Kindlebildstr. 87, D-78479 Reichenau, Germany, (3) Alte Poststraße 29, D-88690 Uhldingen-Mühlhofen, Germany

ABSTRACT

Between 1954 and the beginnings of the 1980s a serious 'die-back' of lakeside *Phragmites* reeds was observed at Lake Constance. The main features and causal factors of the decline are described. A monitoring program which was started in 1984 shows a slow lakeward progression of most of the test areas. The dynamics of the reed front was found to be partially controlled by mechanical threat due to wave and breaker action, and by flooding events. Two models are presented to quantify the potential impact of flooding and mechanical damage. Lakeshore restoration works and *Phragmites* plantations were performed from 1986/87 till 1988/89 at Lake Constance-Obersee. Most of them were successful, since they took these factors into account.

KEY-WORDS : *Phragmites australis*, lakeshore, bank erosion, reed decline, flooding, mechanical damage, power of mechanical resistance, lakeshore restoration, model, monitoring program

INTRODUCTION

The lakeshore zones are among the most intensively occupied elements of the central European landscape. Bank erosion, deterioration of submerged vegetation, and the decline of fringing reeds are consequences of many interconnected factors that are in change due to human impact. A serious decline of lakeside reedbelts (*Phragmites australis* (Cav.) Trin. ex. Steud., Poaceae) has been documented in about 50 lakes in central Europe (for a review see Ostendorp, 1989). One of the sites that are most seriously affected is Lake Constance, the greatest lake in the northern Alpine forelands. In this paper, a concise summary is given on the main features of the retreat, its temporal pattern, and on the underlying causes. The results from a monitoring program that ran from 1984 to 1995 are presented, and some methods and techniques to restore threatened shore sections, and to re-establish *Phragmites* reeds are discussed.

SITE DESCRIPTION

Lake Constance is the largest and deepest lake in the northern Alpine foreland (9°18' E Gr., 47°39' n. Lat., 395 m a.s.l, 571 km^2 water surface, 254 m max. depth; Braun and Schärpf, 1994). It is devided into three basins, Obersee, Überlinger See and Untersee, which are of different morphometric and trophic nature (Figure 1, Table 1). Lake Constance-Obersee is the deep 'upper lake'. It is a mesotrophic lake with low nutrient concentrations, since 64 % of the annual discharge comes from the alpine region (Luft *et al.*, 1990).

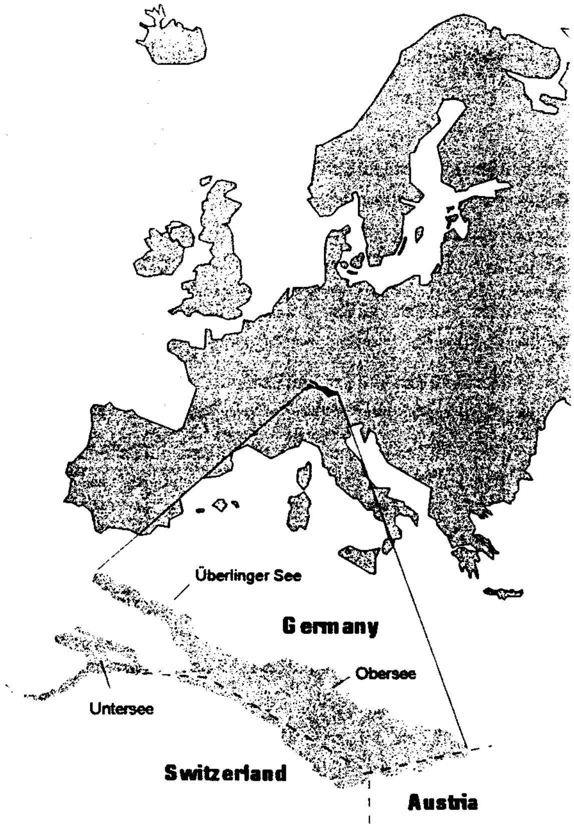


Figure 1 :

Lake Constance with its lake basins, Obersee, Überlinger See, and Untersee

Table 1 : Morphometric, hydrologic and trophic data of Lake Constance

	Obersee	Überlinger See	Untersee
Surface area [km^2]	432.0	68.0	71.5
Shoreline [km]	186		87
Maximum depth [m]	254	148	46
littoral area (0-5m below m.w.l.) [km^2]	12,5 (2.9 %)	7.0 (10.3 %)	24.7 (34.5 %)
mean water level (mean of the period 1887-1987) [m a.s.l.]	395.33		395.11
high water level (June/July, mean of the period 1887-1987) [m a.s.l.]	396.56		396.33
low water level (Jan/Febr, mean of the period 1887-1987) [m a.s.l.]	394.54		394.30
trophic status	mesotrophic	mesotrophic	eutrophic

Überlinger See is a canyon-like basin sheltered from strong winds by ridges of hills at the NE and SW edge of the lake. The littoral platform is extremely narrow except in delta areas. The hydrological regime and the trophic status is the same as for the Obersee. Lake Constance-Untersee, the 'lower lake' is a shallow eutrophic lake with broad shelves and extended reedbelts which cover about 31 % (c. 295 ha) of the eulittoral area, or 54 % of the total shore length (northern shore only). Of the eulittoral reed area, 99.7 % is comprised of monospecific stands of *Ph. australis*, a perennial grass with thick culms, about 6 - 13 mm in diameter at their base, and about 2.5 - 4.5 m in height. The sublittoral zone is covered with submerged macrophytes, mainly *Chara* spp., *Potamogeton* spp., and filamentous algae (e.g. *Cladophora* sp.) which are uprooted and washed ashore during storms, mainly in late summer and autumn.

The lake levels of the three basins are unregulated and undergo the same annual hydrological regime: the water level is high in summer as a result of ice and snow melting in the inneralpine region, and low in winter. In summer all lakeside reeds are submerged, and in winter the greater part of them become dry. Westerly winds prevail in the Lake Constance region, changing from WNW in spring and summer to WSW in winter. From November to June strong winds and storms with strengths between 6 and 10 °Beaufort occur frequently (5 - 6 days per month, average of the period 1959-1984).

REED DECLINE AT LAKE CONSTANCE

The temporal patterns of retreat are best documented for Lake Constance-Untersee, based upon series of aerial photographs from 1954, 1962, 1967, 1972, and 1978 (Ostendorp, 1991). In the first half of the century the reedbelts expanded towards the mean low water bathymetric line, and in the beginning of the 1950s a maximum extension of c. 380 hectares was reached (northern shore only). A few years later the decline began. Initially, the losses were restricted to the landward section, and were due to direct destruction (land reclamation for recreational activities, construction of buildings, etc.). A dramatic die-back ocurred between 1965 and about 1978 when 37 % of the reed beds below the mean water level were lost (Figure 2). The die-back was suspected to have been initiated by the eutrophication of the lake since the 1960s. Recently, however, it became clear from the evaluation of water mark records and meteorological data, that serious flooding events in 1965, 1966 and 1967, together with storms, hail storms and macrophytic washes were responsible for the lakeside decline in the 1960s and 1970s (Ostendorp, 1990).

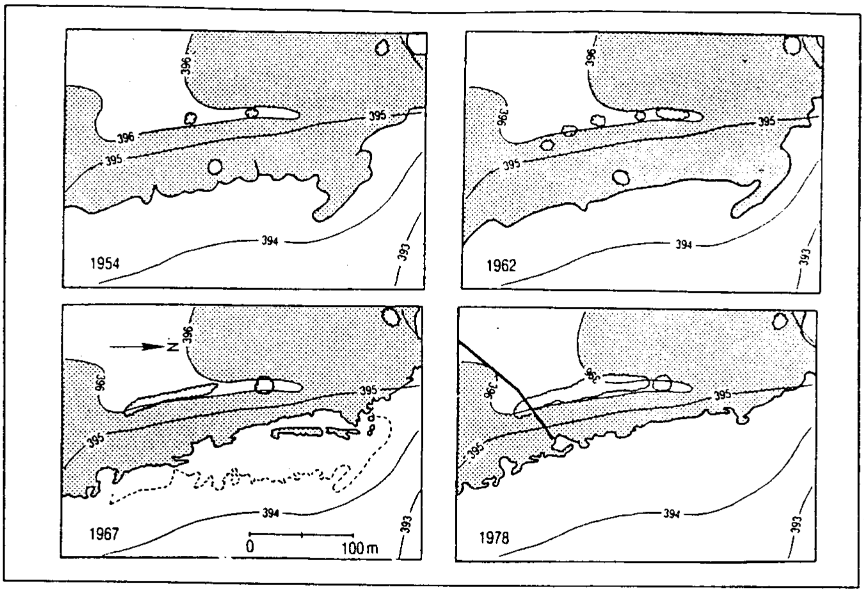


Figure 2 : 'Die-back' of lakeside reeds in Lake Constance-Untersee
(drawn to scale from aerial photographs from 1954, 1962, 1967, and 1978);
dotted area - monospecific *Phragmites* reeds; 394 etc. - bathymetric line [m a.s.l.])

Similar to Lake Constance-Untersee *Phragmites* belts at the Überlinger See and at the northern shore of the Obersee extended far lakewards in the 1930s and 1940s. Due to a lack of adequate aerial photographs the maximum extension is unknown for most places. An analysis of reed regression by interpretation of vertical aerial photographs from 1967, 1978 and 1984 showed a total loss of at least 15 hectares of submerged reed beds. (i.e. 50 % of the area of 1967, considering only the shoreline in the State of Baden-Württemberg) (Krumscheid *et al.*, 1989). At Lake Constance-Obersee lakeside reeds were greatly affected by both mechanical stress by waves loaded with cultural debris, algal and macrophytic mats and driftwood, and by bank erosion.

Hence, it was hypothesized that the longterm dynamics of the lakeside reed front is controlled to a great extent by hydrological factors (i.e. height, duration and frequency of flooding) and by mechanical factors (i.e. action of waves and breakers loaded with drifting matter). This was the main reason for setting up a monitoring program of reed development.

MONITORING PROGRAM

A monitoring program was started in 1984/85 to detect small lateral shifts of the outermost reed front, which are below the detection limits of aerial photographs, and to determine the causes underlying the year-to-year regression or progression of the reed belt. In representative test areas of Lake Constance-Untersee and Überlinger See testing lines were marked with wooden poles along the lakeside reed front. One testing line consisted of 5 sections with a length of 5 m, each (Figure 3). In wintertime, when the water level was low, all culms from the previous season were counted, which stood lakewards of the line between two poles. Based on these data, the annual shift of the reed front can be calculated with the following equation:

$$PR = 100\,(N_{t2} - N_{t1})\, L^{-1}\, D^{-1}\, \Delta t^{-1} \qquad (1)$$

PR - progression/regression rate [cm a^{-1}] , N - number of *Phragmites* culms at time t1, and t2, respectively [1] , L - length of a section [m], i.e. generally 5 m, D - mean culm density in lakeside reeds, i.e. 30 m^{-2}, Δt - time difference, i.e. generally one year . Positive values indicate a lakeward progression of the reed front, negative rates a regression.

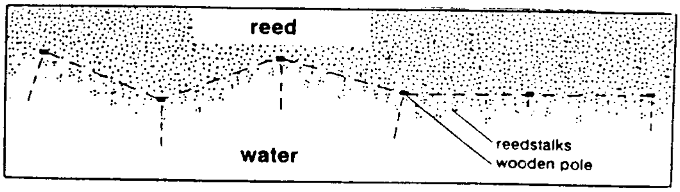


Figure 3 : Schematic top view on a test line with 5 sections

A total of 72 test lines were established in 13 different types of habitats :

1 - reed beds on onkolithic carbonate sediments ('Schnegglisande'), poor in nutrients (n = 5)
2 - reed beds on glacial boulder clay, poor in nutrients (n = 15)
3 - stands on substrates with medium nutrient level and subjected to mechanical stress (n = 8)
4 - stands strongly affected by organic debris, very rich in nutrients (n = 17)
5 - stands near the mouth of a polluted creek, rich in nutrients (n = 3)
6 - reed beds subjected to beach ridge movement due to breaker action, poor in nutrients (n = 7)
7 - storm exposed reed stands (n = 7)
8 - edges of an isolated reed island (n = 1)
9 - reed beds infested by larvae of the fly *Lipara spec.* (Dipt., Chloropidae) (n = 3)
10 - reed stands shaded by trees and bushes (n = 3)
11 - reed beds affected by washes of algal mats (mainly *Chara* sp. and *Cladophora* sp.) (n = 1)
12 - reed beds influenced by bank erosion (n = 1)
13 - reed stands protected from erosion by geotextiles (n = 1)

The data set presented here consists of 51 test lines from Untersee (monitored from 1984/85 till 1994/95) and 21 test lines from Überlinger See (1986/87 - 1994/995). Data were evaluated using SPSS for Windows Vers. 6.0 procedures. The results are given in Table 2 (Untersee) and Table 3 (Überlinger See).

Mean PR Rates At Untersee And At Überlinger See

Tables 2 and 3, right columns, show the grand means of annual rates of reed front progression/regression at Untersee and at Überlinger See, respectively (all test lines pooled). The high average PR rates at the Untersee in 1984-1987 were followed by a regression in 1988 (-4,8 cm). Compared to the Untersee the Überlinger See PR rates were much lower in 1987/88, 1988/89 and in 1992/93 ($p<0,01$ for all three years, Anova). The rate in 1988/89 was lowest in both lake areas; in the Untersee it was negative ($p<0,01$, Anova). The progression rates from 1989/90 to 1991/92 were comparable at the Untersee and Überlinger See and reached high values which were never recorded before. The PR rates in 1992/93 were significantly different for Überlinger See and Untersee ($p<0,001$, Anova).

PR Rates Of Different Habitat Types At Untersee

In only a few cases the mean PR differences between habitats were significant. In 1984 reed beds with average nutrient level (type 3) and stands on shores with shifting beach ridges (type 6) differed significantly from all other types (Student-Newman-Keuls [SNK]-test , $p<0,05$), in 1985 this was the case for habitat type 2. In 1986 the rates of type 1 and types 6 and 3, respectively, were significantly different. Habitat types 8 and 9 exhibited higher rates than all others in the period 1993/95 ($p < 0.05$, SNK-test).

Table 2: Mean annual progression/regression rates [cm a^{-1}] for test areas at the Untersee from 1984/85 till 1992/93 for different types of habitats
(# - number of test areas with progression or regression, resp., all periods pooled)

Period	**Habitat Type**										
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**mean**
1984/85	6.0	18.0	20.7	10.8	8.7	28.4	3.1	2.5			15.5
1985/86	7.0	24.3	- 2.4	4.6	2.5	7.6	2.7	5.4			5.5
1986/87	16.9	4.0	- 3.4	4.6	14.8	- 5,7	- 2.5	10.7	8.5		4.3
1987/88	17.3	10.6	15.8	13.2	20.0	10,2	9.1	16.9	- 2.7	2.3	12.9
1988/89	- 4.6	0.8	- 3.6	-10.5	- 8.5	0.7	- 3.5	- 7.9	-3.7	- 8.3	- 4.8
1989/90	66.4	15.8	40.2	16.7	39.9	31.4	26.2	62.9	16.5	11.3	31.1
1990/91	43.6	4.5	70.2	48.2							49.2
1991/92	10.7	10.0	13.0	13.6	22.4	24.0	0,0	35.4	41.0	- 3.8	16.1
1992/93	22.5	- 1.4	29.0	22.0	65.8	25.1	8.4	32.9	25.5	23.2	23.6
1993/95	12.5	9.8	11.5	7.0	33.8	10.4	9.7	26.5	19.4	- 0.4	11.4
# progr.	9	9	7	9	8	8	6	8	5	3	72
# regr.	1	1	3	1	1	1	3	1	2	3	17

PR Rates Of Different Habitat Types At Überlinger See

The PR rate of the storm exposed habitats (type 7) was significantly higher than the rates of all other habitat types for the period 1989/90 ($p<0.05$, SNK-test). In 1990/91this was the case for the habitat types 13 (protected by geotextile: 240 cm a^{-1}) and 5 (polluted creek: 65.6 cm a^{-1}) ($p<0.05$, SNK-test). They did not differ significantly from each other. Both, in 1992/93 and 1993/95 the progression rate of type 5 was the highest among all habitat types ($p < 0.05$, SNK-test) No significant differences between habitat types could be detected for the periods 1986/87, 1987/88, 1988/89 and 1991/92. In 1992/93 and 1993/95 the PR rate of habitat type 5 was significantly higher ($p<0.05$, SNK-test) compared to the rates of all other types.

Table 3: Mean annual progression/regression rates [cm a^{-1}] for test areas at the Überlinger See from 1986/87 till 1992/93 for different types of habitats
(# - number of test areas with progression or regression, resp., all periods pooled)

Period	**Habitat Type**										
	2	**3**	**4**	**5**	**7**	**9**	**10**	**11**	**12**	**13**	**mean**
1986/87	7.1										7.5
1987/88	17.3	0.9	- 2.5	8.2	- 0,1	7.0	8.5	- 0.4	3.2	32.0	4.5
1988/89	- 3.3	- 2.0	5.7	- 4.0	5.0	- 5.3	- 3,8	- 3.6	4.4	- 6.0	0.7
1989/90	29.8		13.9	31.3	108.7	21.7	5.2	0.2	- 6.8	66.7	3.5
1990/91	52.3		34.8	65.6	60.2	78.5	27.7	0.9	6.3	240.0	44.9
1991/92	16.9		9.3	28.4	33.9	26.8		- 0.2	10.3	83.3	19.9
1992/93	- 0.7		1.6	69.1	17.4	13.8		0.0	-15.9		5.0
1993/95	8.4		9.3	133.0	16.8	10.2	- 2.8		5.0	30.4	13.7
# progr.	6	1	6	6	6	6	3	3	5	5	47
# regr.	2	1	1	1	1	1	2	3	2	1	15

PR Rates In Relation To The Water Level

The mean annual course of the water level of Lake Constance is characterized by minimum values from January to March, then raising like a sigmoid curve to reach a maximum in the beginning of July. The raise is somewhat parallel to the growth of the *Phragmites* shoots. There are, however, great deviations from this average course. It is hypothesized that water levels below the average will promote the lakeward spreading of the reed belts, and water levels much higher than the average will lead to a regress. This hypothesis was tested for 25 test lines for which the altitude h_0 was known. The PR rate was correlated with the effective water depth above ground level $h_{eff} = h_{wl} - h_0$ (h_{wl} - maximum water level in a given month) for April, May, June and July. A highly significant negative correlation was yielded in all four months (all test lines pooled). Seperated into different types of habitat the relationships are diverse. Figure 4 gives an example for type 2 (boulder clay) in May. Table 4 shows a summary of the statistical results. For all habitat types a strong correlation exists between the maximum water depth and the PR rate either in May or in June. In July no significant correlation could be found. In this month the reed culms are tall enough, so that high water levels cannot exert any influence on them, anymore.

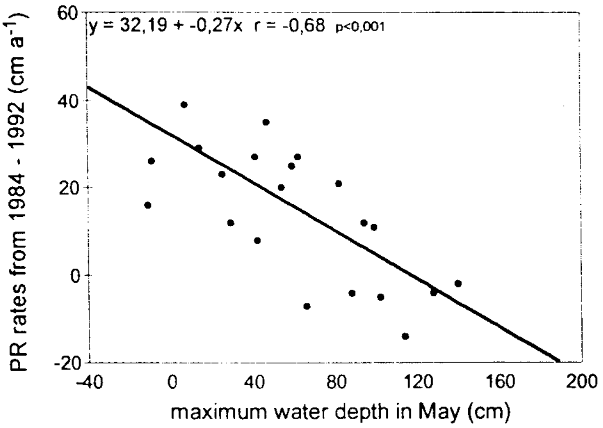


Figure 4 : Correlation between maximum water depth in May and PR rates of habitat type 2 (period 1984 /92)

Table 4 : Correlation between PR rates and maximum water depth for five habitat types at Lake Constance-Untersee (r - Pearson correlation coefficient, p - significance, n.s. - not significant, n - number of 5m-sections)

Month		**Habitat Type**					
		1	2	3	4	7	average
	n	18	36	45	81	18	180
April	r	-0.22	-0.65	-0.31	-0.42	-0.67	-0.39
	p	n.s.	< 0.001	n.s.	< 0.001	< 0.05	< 0.001
May	r	-0.33	-0.68	-0.45	-0.43	-0.67	-0.44
	p	n.s.	< 0.001	< 0.01	< 0.001	< 0.05	< 0.001
June	r	-0.57	-0.20	-0.52	-0.32	-0.32	-0.42
	p	< 0.05	n.s.	<0.001	< 0.01	n.s.	< 0.001
July	r	-0.23	-0.05	-0.18	-0.08	-0.07	-0.18
	p	n.s.	n.s.	n.s.	n.s.	n.s.	< 0.05

FACTORS THAT CONTROL THE DYNAMICS OF THE LAKESIDE REED FRONT: FLOODING AND MECHANICAL STRESS

Phragmites australis is a flooding tolerant plant which normally achieves the most vigorous growth when the water table lies a few decimetres above ground. The flooding tolerance is, however, limited: to photosynthesize and to accumulate assimilation products the leaves must not be submerged, as the CO_2 diffusion rate in water is much lower than in air (Rodewald-Rudescu, 1974, p.81). Similarly, oxygen transport from the culm and the leaves into the rhizomes is possible only if the plant is in contact with the atmosphere (Armstrong and Armstrong, 1991). Since *Phragmites* cannot stand anaerobic metabolism for long (Brändle 1983, 1985, Studer and Brändle 1984), the submerged leaves turn yellow and the rhizomes die down after a few days of flooding. It is assumed that at least three leaves must be emersed to secure a culm's survival (any mechanical threat being absent). Abnormal high flooding events in spring-time and early summer have been shown to be crucial for the survival of *Phragmites* culms at the lakeside reed front. This was the case in 1965, when c. 74 % of the outermost fringe died back (Ostendorp, 1990, 1991).

The results of the monitoring program demonstrate that the reed belts of Lake Constance Untersee and Überlinger See are in progress after a long period of decline. The same yields for the test areas of the Obersee (Krumscheid-Plankert, unpubl. data). There is a continuous lakeward spreading of the reed beds, interrupted by the period 1988/89 which followed a year with an extraordinarily high flooding in July and August. In the periods 1987/88, 1988/89, and 1992/93 the progression rates of the Überlinger See reeds was found to be significantly lower than in the Untersee. Here, the reeds form a continuous belt along the shore over kilometres, whereas in the Überlinger See the greater part of the reed belts form isolated isles of a few hundred square metres each. Presumably, these reeds, due to their greater edge factor, are more susceptible to all kinds of mechanical threat like drifting matter, agitated by breakers, grazing waterfowl, and bathing, boating and other human activities.Our results show a reduced progression, or a regress, in the season following a year with high water events, even if the stalks have not been completely submerged. This indicates some kind of sublethal damage to the rhizomes which becomes evident in the following season, as the rhizomes produce fewer shoots in spring.

In case of mechanical damage, the *Phragmites* culms are bent down, and the leaves may be ripped up or stripped off. If the culm is pressed below the water surface, the leaves are not able to photosynthesize. Hence, the transport of assimilates to the rhizomes is severely reduced. In many cases the culms are broken at their base. Then the central cavity of the internode and the aerenchyma are filled with water. As the diffusion rate of oxygen is much lower in water than in air, oxygen supply to the underground organs is inhibited. The resulting oxygen deprivation in the rhizome tissue sets up an energy-consuming anoxic metabolism, which may lead to a die-off of parts of the rhizome complex within a few days. The degree of damage to lakeside reed stands can be regarded as the result of external forces and the capability of the reed to withstand these forces („power of mechanical resistance", Ostendorp, 1995). If the acting forces are great enough, the mechanical tolerance of the culm will be exceeded, and the culm will fail, resulting in some kind of damage, as described above. Binz-Reist (1989) attempted to put the interaction of wind and wave forces with the resistance of the stem into a mathematical form. Whether a culm fails or not depends on numerous factors which must be quantified to take advantage of this model (Figure5). Many of the input variables cannot be measured with sufficient precision. Hence its practical value for quick and simple estimation of mechanical impacts to lakeside reeds, and their consequence for the plants' survival is doubtful.

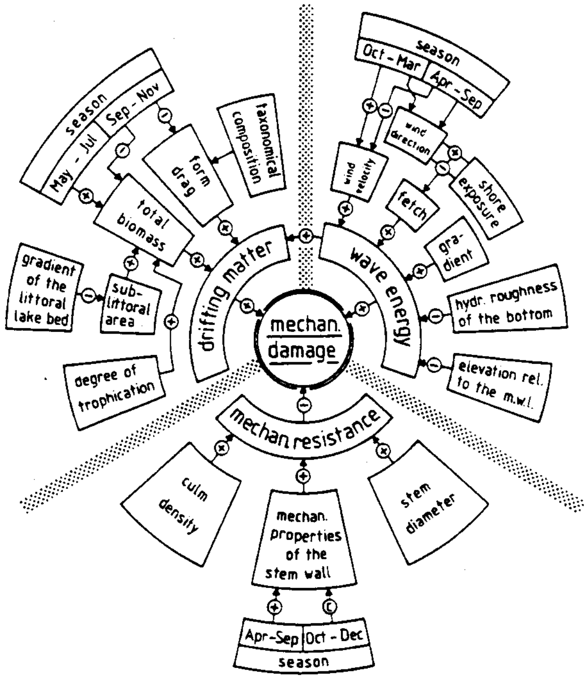


Figure 5 : Factors affecting the damage to lakeside reeds in Lake Constance: (+, accentuating influence; - , weakening influence; c, constant, time-independent influence; m.w.l., mean water level)

Ostendorp (1995a) proposed a simple stochastic model that allows assessment of the mechanical susceptibility of a reed stand, based on the mechanical properties of the stem wall tissue, the dimensions of the stem's cross section and the density of the stand. The model implies that, under constant mechanical impacts, the 'degree of mechanical damage' (DMD), i.e. the proportion of culms that fail, will be inversely proportional to the 'power of mechanical resistance' (PMR), i.e. the sum of bending stiffness values of all individual culms per square metre :

$$\mathrm{DMD} \sim \mathrm{PMR}^{-1} = [\,A^{-1} \textstyle\sum M_i\,]^{-1} \tag{2}$$

with, A - area, 1 m^2, M_i - bending stiffness of stem i within area A;

The bending stiffness M can be regarded as the ratio of the bending momentum to the curvature of the stem induced by it. According to Binz-Reist (1989) this ratio depends on mechanical properties of the stem wall material (modulus of elasticity, E) and geometric dimensions of the stem's cross section (momentum of inertia, J)

$$M = E\,J \tag{3}$$

By this model it is possible to compare the resistance of two *Phragmites* stands against an unknown, but constant, impact of drifting matter powered by waves. It is not practicable to prove the equality of wave energy input as the damaging event takes place, since the measurement of wave energy input requires platforms in front of these two reed stands, and data recording equipment installed to await an adequate storm on the lake. A simple way to test the model is to find a uniform shore section affected by waves, without alongshore variation of wave energy input and drifting matter load. Then the variation of percentage of damaged culms should only be due to differences in PMR of the *Phragmites* belt. One such stand could be examined at Lake Constance-Untersee: the model explained 56 % of the variability of the proportion of damaged culms, and 67 % of the variability of damaged biomass. These relatively low figures are attributed to the fact that longshore variability of wave+drifting matter energy cannot be neglected.

RESTORATION WORKS

The results of our investigations demonstrate that flooding and mechanical impact are among the main factors that control the dynamics of lakeside *Phragmites* reed at Lake Constance. Several countermeasures have been taken against lakeshore deterioration and reed decline (Ostendorp *et al.*, 1995). At Lake Constance-Untersee the idea was to improve the power of mechanical resistance of reeds by biomanipulation: winter harvesting was performed to reduce the nutrient and organic matter load of the reed beds. It turned out, however, that winter cut *Phragmites* stands had thinner stems with unfavourable mechanical properties, so that their PMR was lessened. Field observation confirmed that this kind of management was responsible for a high degree of damage to treated reeds (Ostendorp, 1995b)

Based upon these findings, the objective of restoration works at Lake Constance-Obersee was to protect damaged reeds from mechanical threat by (i) filling up erosion scarps at the reed front, (ii) creating a new shore platform or enhancing natural sedimentation, (iii) locating the breaker zone away from the reed front, and so reducing wave energy input into the reeds, and if new reeds were planted (iv) keeping away drifting matter by network fences (Figure 6). A five-year project was initiated 1987 in order to find an optimal technique. Three different restoration methods were tested: at three sites a shallow refilling of substrate was heaped up (by aerial cableway or by building machines) to compensate for the erosion. A stone dam was constructed as a lakeward boundary of the restoration area at a distance of about 30 m from the shore and in the altitude of the mean water level (Figure 7). After refilling with substrate the detrimental effects of waves on the remaining reed were reduced by breaking high energy waves near the dam, and by friction on the (new) littoral bottom in front of the reed stands. Two other methods were designed to promote accumulation of sediment by longshore currents. In one case sedimentation lozenges were constructed with coir fabric in front of the reed stand (Figure 8). The caskets were 15 to 30 cm high and were built into the sediment by hand. They were found to be effective for 2 to 3 years until the material decays. The method was built on a small scale compared with the others, and forms a quite gentle type of restoration. In the other case stone groynes were constructed parallel to the shore line to promote sedimentation in the calm areas.

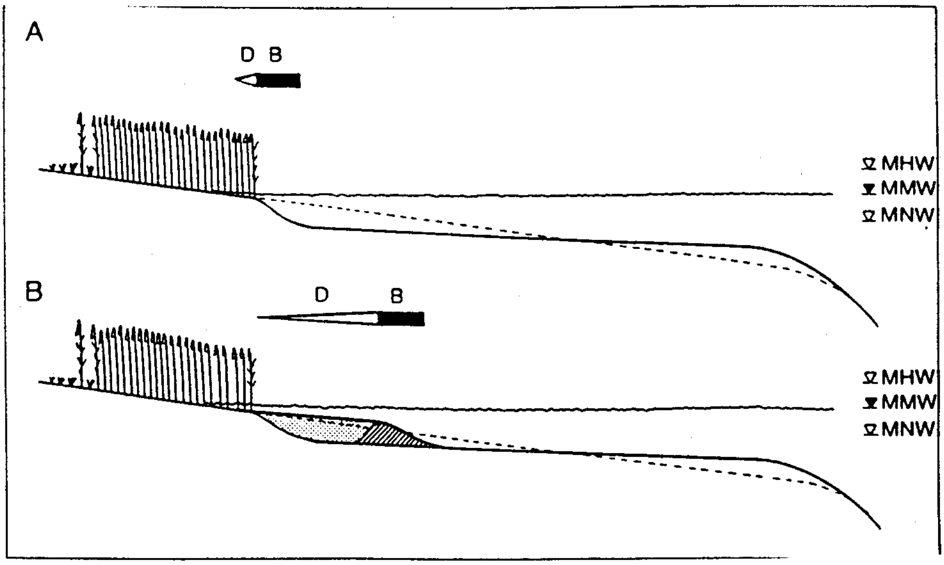


Figure 6 : Schematic cross section of the littoral zone before (A) and after (B) restoration, showing the erosion scarp near to the reed front, the former littoral slope (dashed line), the stone dam (dashed area), and the refilled fine matter (dotted area); B - breaker zone, D - dissipation zone, MMW (MNW, MHW) - mean water level (mean low water, mean high water)

In addition, attempts were made to re-plant reed beds where they once grew, the intention being to close gaps and create continuous stands. However, at Lake Constance, the annually fluctuating water level and the unpredictable flooding and storm events are crucial for the establishment of new plantations. In May and June, *Phragmites* shoots grow c. 4 cm per day. Usually this is sufficient to keep up with the rising water level. Planted reed, however, grows only at a rate of about 2 cm per day (Krumscheid-Plankert,unpubl. data), and may be flooded during the normal annual course of the water level. In order to improve the vitality of reed, and to reduce the shock effect of transplantation, the rhizome clusters were precultivated in basins with a constant water level, and then transferred to the lake shore a year or two later. The precultivated culms then showed higher growth rates (Krumscheid-Plankert, 1993). The 'new' reed stands grow on a higher level and their ecological importance, e.g. for fish breeding or waterfowl nesting, is thought to be less than that of permanently flooded reeds. They are, however, less susceptible to flooding disasters.

After five years of observation the following results can be presented: The main objective of the restoration measures was to stabilize the shoreline and the lakeside reed front. This was reached by all methods (refilling behind a stone dam, sedimentation caskets, stone groynes). However, the lake's dynamic work, i.e. different grades of 're-erosion' can be seen at all sites.The method of sediment refilling behind a dam shows good stability, although it requires a lot of material and is relatively expensive (1000 - 1500 DM per running metre of shoreline). Additionally this is a large-scale method which cannot be altered if negative effects should become evident. Furthermore the method is not advisable for river mouths: negative side-effects were re-erosion and canalizing of the river flow. Regarding the restoration with caskets and stone groynes, problems of stability arose, and together with the settling of solid matter an undesired accumulation of filamentous algae occurred in some years.

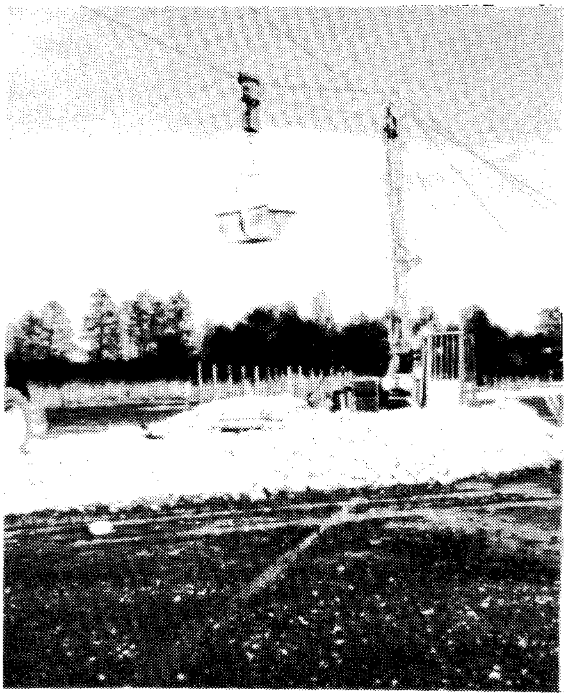


Figure 7 : Constructing a stone dam using a cableway (the photo was taken in late winter at low water level)

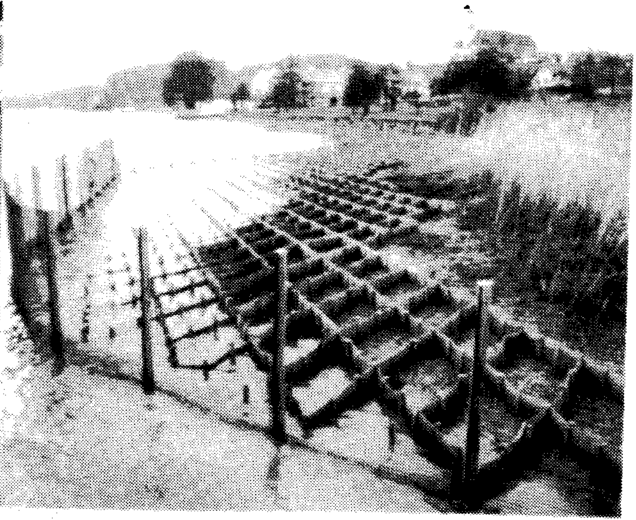


Figure 8 : Sedimentation lozenges in front of a damaged reed stand
(the photo was taken in late winter at low water level)

At all restoration sites a rapid lakeward progression of the remaining reed stands took place, and the plantations spread quickly, too. Interestingly, the method "refilling behind a dam" offers a refuge for endangered plant species on the replaced bare sediments. If this type of construction is made, the sediments for refilling should be chosen carefully, not only regarding maximum stability. Since the shoreline and the habitat conditions are altered considerably, anyway, the colonization of the bare sediments by certain species may be desired and promoted. Comparing the costs per m^2, the restoration measures ranged from ca. 20 DM for the stone dam methods to 30 DM for the hand-work methods of sedimentation caskets or stone groynes.

CONCLUSIONS

Extreme flooding events and mechanical factors have led to a serious decline of fringing reeds in many Central European lakes. One of these lakes is Lake Constance, where predictive models have been developed to reconstruct damaging events, and to elucidate the boundary conditions to which a lakeside *Phragmites* stand can proceed. It is hypothesized that for a long-term view the outermost reed front is in equilibrium with external forces, i.e. the yearly biomass increment is equal to the yearly amount of biomass destroyed by these forces. The results of the monitoring program suggests that flooding and mechanical damage are among the most important factors that control the dynamics of the lakeside reed front in Lake Constance. Hence, lake shore restoration works should take into account strength, duration and frequency of flooding and storms events. The restoration concept at Lake Constance-Obersee was to shelter the damaged reeds from these two impacts, giving them the chance to spread anew. After five years of observation it became clear that the efforts are successful, and that the methods tested might be quite suitable for a number of lakes with mainly mechanically-caused reed damage.

REFERENCES

Armstrong, J. and W. Armstrong (1991). A convective gas through-flow in *Phragmites australis. Aquatic Botany.* 39 pp. 75-88.

Binz-Reist, H.-R. (1989). Mechanische Belastbarkeit natürlicher Schilfbestände durch Wellen, Wind und Treibzeug. *Veröffentlichungen des Geobotanischen Institut der ETH Zürich.* 101 pp. 1-536.

Brändle, R. (1983). Evolution der Gärungskapazität in flut- und anoxiatoleranten Rhizomen von *Phalaris arundinacea, Phragmites communis, Schoenoplectus lacustris* und *Typha latifolia. Botanica Helvetica.* 93 pp. 39-45.

Brändle, R. (1985). Kohlehydratgehalte und Vitalität isolierter Rhizome von *Phragmites australis, Schoenoplectus lacustris* und *Typha latifolia* nach mehrwöchigem O_2-Mangelstreß. *Flora* .177 pp. 317-321.

Braun, E. and K. Schärpf (1994): Internationale Bodensee-Tiefenvermessung 1990. - Stuttgart, 98 pp. + app.

Krumscheid-Plankert, P. (1993). 'Restoration of reed stands at Lake Constance-Obersee' - a preliminary report on results of the BMU-research project. *Limnolgie aktuell.* 5 pp. 179 - 187. Stuttgart, G.Fischer.

Krumscheid, P., H. Stark and M. Peintinger (1989). Decline of reed at Lake Constance (Obersee) since 1967 based on interpretations of aerial photographs. *Aquatic Botany.* 35 pp. 57-62.

Luft, G., G. van den Eertwegh and H.J. Vieser (1990). Veränderung der Bodensee-Wasserstände von 1887 bis 1987. *Handbuch Hydrologie Baden-Württemberg.* 6.2 pp. 1-77 + app.

Ostendorp W. (1989). 'Die-back' of reeds in Europe - a critical review of literature. *Aquatic Botany.* 35 pp. 5-26.

Ostendorp, W. (1990). Die Ursachen des Röhrichtrückgangs am Bodensee-Untersee. *Carolinea* . 48 pp. 85-102.

Ostendorp, W. (1991). Damage by episodic flooding to *Phragmites* reeds in a prealpine lake: proposal of a model. *Oecologia.* 86 pp. 119-124.

Ostendorp, W. (1995a). Estimation of mechanical resistance of lakeside *Phragmites* stands. *Aquatic Botany.* 51 pp. 87-101.

Ostendorp, W. (1995b). Effect of management on the mechanical resistance of lakeside reeds in Lake Constance. *Acta Oecologica.* 16 pp. 277-294.

Ostendorp, W., Chr. Iseli, M. Krauss, P. Krumscheid-Plankert, J.-L. Moret, M. Rollier & F. Schanz (1995). Lake shore deterioration, reed management and bank restoration in some Central European Lakes. *Ecological Engineering.* 5 pp. 51-75.

Rodewald-Rudescu, L. (1974). Das Schilfrohr. *Die Binnengewässer.* 27 pp. 1-302 . Stuttgart

Studer, C. and R. Brändle (1984). Sauerstoffkonsum und -versorgung der Rhizome von *Acorus calamus*, *Glyceria maxima*, *Menyanthes trifoliata*, *Phalaris arundinacea*, *Phragmites communis* und *Typha latifolia*. *Botanica Helvetica.* 94. pp.23-31.

BIOENGINEERING TECHNICS APPLIED ON AN HUMID AREA BIOLOGICAL RECOVERY AND CLEARING

T. FERREIRA MACHADO* and A. TELHADO**

WATER INSTITUTE. AVª ALMIRANTE GAGO COUTINHO Nº 30
1000 LISBON, PORTUGAL
FAX: (01) 8480933

ABSTRACT

Clearing is one of the fundamental actions on flood control politics, but depending on the way this action is implemented, the consequences could eventually become prejudicial to the riparian ecosystems.

For this reason, an experience was done by clearing and restoring a part of a river, situated on an humid area (Natural Reserve of Paul do Boquilobo), classified and included on the World Program of the Biosphere Reserves, by using bioengineering technics instead of using other conventional river engineering technics. This way it was our intention to protect and mitigate environmental impacts on the river channel and banks.

Furthermore and for the rehabilitation of the natural ecosystem, new solutions were proposed for bank stabilization with the utilization of living vegetation and promoting river biodiversity, for the areas considered to have more ecological value, while for the other areas were human activities prevail, it was analyzed the possibility of promoting the landscape value of the river associated with its recreational use.

KEY-WORDS: Bioengineering, river, restoration.

---

* Bioengineer
** Environmental Engineer

1 Introduction

Flood defense politics has clearing as one of the fundamental actions to diminish flood hazards, but depending on the way this action is done, the results could eventually become prejudicial to the riparian ecosystem.

An experience was initiated on a Portuguese marshland, at the river Almonda, named Boquilobo Natural Reserve, where bioengineering techniques were used to clear and restore the river, instead of using other conventional river engineering techniques. This action was implemented taking into account that this Reserve has a particular conservationist value as it was classified and included on the World Program of the Biosphere Reserves.

The principals' objectives of this work were to integrate conservationist ideas together with the natural resources planning, promoting and valorizing together, the hydraulic conditions of the river without affecting the whole ecosystem.

For rehabilitating the natural ecosystem, new solutions were proposed for bank stabilization and protection using living plants and promoting river biodiversity, on the areas with more ecological value, while for the areas were human activities prevail, it was promoted the landscape value of the river associated with its recreational use.

2. The Almonda River Basin

The Almonda river, is situated at the center of Portugal, in the Portuguese part of the Tejo river Basin (Fig. 1). The Tejo river Basin, with a drainage area of 80629 km2 (69% located in Spain and 31% in Portugal), is a very important river for the Water Management and Hydrological Plans of the two countries.

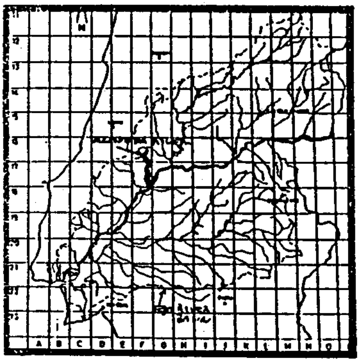


Fig. 2- The Tejo and Almonda river Basin

Fig. 1 The Tejo river Basin

In this specific case, we analyze the characteristics of the Almonda river Basin (Fig. 2), which has a drainage area of 228,6 km2, divided between two geological regions, a karst mountainous region and an alluvial valley, with altitudes that varies from 25 to 675m.

The north part of this river basin is characterized by the lack of surface drainage and the groundwaters are the river springs because in this area all the river system is situated beneath the karst mountain of "Serra de Aire", while the south part of this river is associated with the Tejo river floodplain. Between these two parts we could find a transition area, without specific characteristics. For these reasons the channel morphology of the Almonda river has on his first part meander characteristics and on the last part, when it is situated on the alluvial valley, it has a straight pattern.

Almost all the watercourses of this river basin, except the Almonda river, have a temporary and torrential regime, as a consequence of the physical and climatic characteristics of this region, with mean temperatures of 16 °C, and a mean annual precipitation with values among 600 and 800 mm.

Pasturing cattle and forresting are the main activities at the high altitudes, and only near the main watercourse, on the lower areas, agriculture is practiced intensively. In terms of population, the karst region is uninhabited, and the main villages (Ribeira Branca, Lapas and Torres Novas) are located in the lowlands.

The Almonda river is considered a polluted river, because of the industrial discharges, being nowadays in practice a Program that pretends to improve the water quality of this river.

As natural characteristics, and in the alluvial valley, it is situated a Marshland named Boquilobo (Fig. 3). This area was classified by the Portuguese Government - (Decreto-Lei n° 198/80, 1980 June the 24th) as a Protected Area -Natural Reserve-, as a consequence of the existence of fauna species with great interest, that were urgent to preserve.

Later this area was also integrated on the European BIOTOPOS CORINE Program, (Biotopos n° 12) and in 15/12/81 the International Council of the MAB, classified it as a Biosphere Reserve.

Fig. 3- Boquilobo Natural Reserve

The Natural Reserve of Boquilobo Marshland with 529 hectares, has two different areas, an agriculture land and a natural area which is partially occupied by common reeds and by a swamp, being this last area totally dependent on the Hydrological regime of the Tejo and Almonda rivers. Being frequent in this region the occurrence of floods, the water level variations during the two main seasons (winter and summer) is a significant phenomenon.

One of the principal characteristics of this classification resulted from the importance of this area on ornithology species, being this Reserve a nest building area for several species of the family Ardeidae, commonly named as herons and bitterns [which includes with importance at the national level *Buculcus ibis* and *Egretta garzetta* (little egret) and with international importance *Ardea- purpurea* (purple heron), *Ardeola ralloides* (squacco heron), *Ixobrychus minutus* (little bittern) and *Nycticorax nycticorax* (night heron)].

At the national level it is also a wintering area for species of the Anatidae Family and a migration route for trans-Saharan small birds like the ones that belong to the Turdidae Family.

At the botanical level the species diversity is limited being dominant the following species: *Polygonum amphibium* (amphibious bistort), Xanthium strumarium subs. italicum (rough cocklebur), *Alisma lanceolatum* and *Paspalum paspalodes*, depending their number on the drainage conditions. The number of vascular plants is diminished in places where during summer the water table exists, dominating *Sparganium erectum* (branched bur-reed) and seaweed's. Considered as an infesting plant that affects other endemic plants and fishes, is the *Eichornia crassipes* (water hyacinth) that proliferates in the ditches.

3. Intervention Areas

Having the intention to guarantee the correct hydraulic function and at the same time the preservation of the ecological component, it was proposed and initiated for two different areas, the clearing and restoring of the Almonda River.

a) Torres Novas, urban area - Fig.4

In Torres Novas the river has meanders but the river channel has been modified presenting rectangular sections, being the margins totally transformed with retaining walls and buildings, so the flow conditions are variable. In most cases the channel has artificial characteristics without any vegetation recovery, having only on the transition between the urban and rural environment, arboreal vegetation and scrubs that have invaded the river bed or are located on their margins.

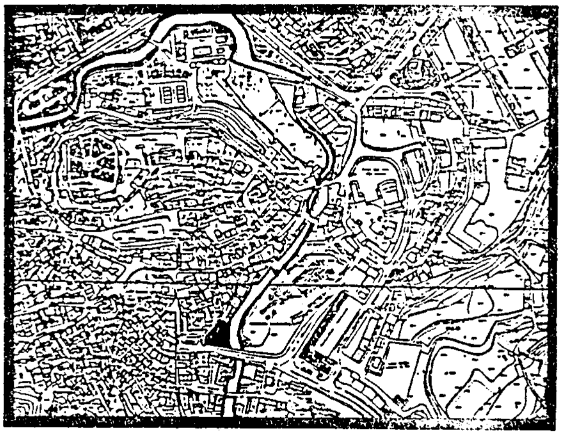


Fig. 4- Torres Novas

The principal problems on this part of the river were related with the insufficient hydraulic capacity, as a result of the location of bridges and buildings, the accumulation of litter on the river bed and the existence of weir or small dams.

b) The Almonda river and Boquilobo Natural Reserve

The Natural Reserve is limited by a part of the Almonda river, where the channel is linear and their sides are limited by levees that were in some places ruined. As a result of this degradation the stream flow was diverted to the agricultural areas.

Generally the riparian vegetation hasn't diversity, being essentially composed of willows and some specimens of ash, dog rose, blackberry vine and bulrushes (Reedmace Family, Typhaceae). The obstruction to the stream flow was frequent as the river bed was strongly silted and invaded by infesting species and arboreal vegetation, tree trunks and other materials.

4 Intervention Proposals

The works of dredging that were done on the Almonda river that is situated at the Natural Reserve, were basically for clearing like the following:

-selection of the trees that have to be preserved as a consequence of their ecological value;
-removal from the river bed, with certain criterion's, the reeds taking into account that some plants have to be protected;
-removal of loose sediments, predominantly silt and vegetation (desilting), in order to maintain and adequate capacity or level (Fig. 5);
-removal of the materials deposited on the river bed - like trunks and branches, that obstruct water circulation;
-trimming of overhanging branches (Fig.6);
-cutting the trees, scrubs and bushes that were installed on the river bed;
-enlargement of the strangulation points maintaining a medium section of 9m;
-consolidation of the margins taking into account the necessity to maintain a medium section (re-sectioning);
-utilization of material removed for modeling (re-grading).

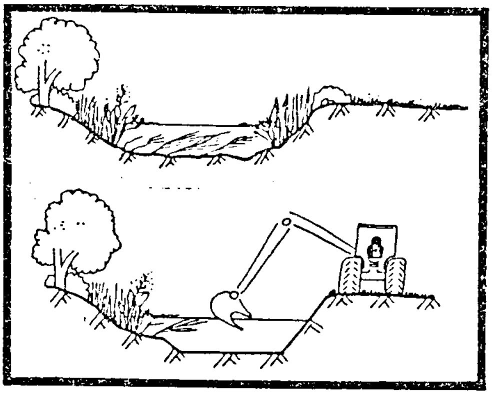


Fig. 5- Partial Dredging(before-after)

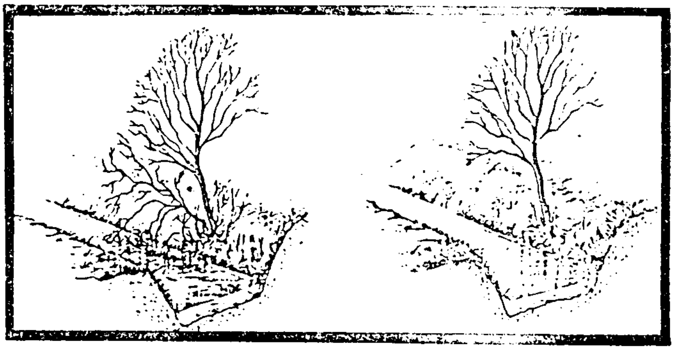


Fig. 6- Trimming branches

During these actions some precautions were taken like the seasonality of the works, selection of adequate locals for the deposition of sediments and types of machinery. These precautions were practiced to avoid biological damage as a result, for example of the machinery circulation.

The proposal also included the intention to create punctual meanders on the watercourse (in an extension of 300m), to modify the linear aspect of the river and consequently water velocity, but this intention was not implemented.

On a second phase, the future works will include:

-the installation of endemic species of riparian vegetation to increase biological diversity on the margins;
-the introduction of small gates that could control water levels and sometimes could permit the marshland flood;
-the conclusion of the re-grading and re-sectioning works with the utilization of living plants and other techniques (Fig. 7 - Gabions and reed rolls of *Phragmites communis* and reed clump planting).

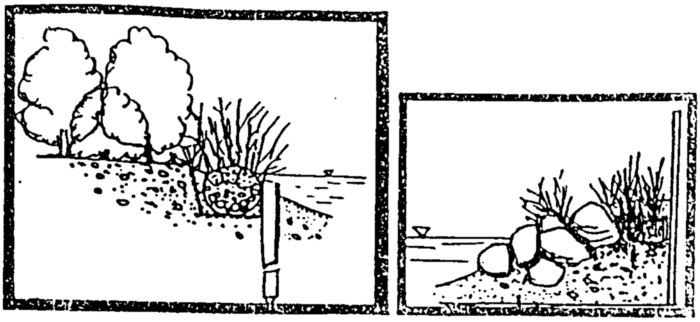


Fig. 7 - Gabions and reed roll and reed clump planting

The works done at the Urban Area of Torres Novas had the objective to protect this village from floods and to promote and maintain the amenity and recreational use at the adjacent land of the river. On the river bed the actions were the following:

- tree and scrub selective removal and pruning in order to permit stream flow optimization (Fig 8);
- removal of garbage from the river bed and margins;
- removal of silts and seaweed's from the river bed (de-silting) and their transportation to a specific place because of their toxicity;
- dig up the trees located on the top of the slope.

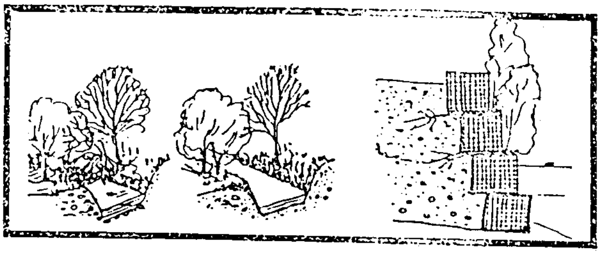


Fig 8 - Clearing a river (before-after)

The actions proposed were not limited to the river bed, they also included other works that will the realized in the next future, like:

- modeling the unstable margins and plantation of vegetation to reconstitute the riparian vegetation, to promote the stabilization of the margins; The specimens that were proposed to install were: Salix atrocinera, Tamarix africana, Populus nigra (black poplar), Rosa canina (dog rose) and Nerium oleander (oleander);
- installation of vegetated gabions and reed rolls;
- stabilization of banks with the utilization of stones (Fig. 9);
- stone walls without using mortar;
- dam recuperation for the use of the populations;
- recuperation of a mill for didactic purposes;
- distribution to the population of pamphlets and other material of disclosure that motivate their interest in using the river for recreational purposes.

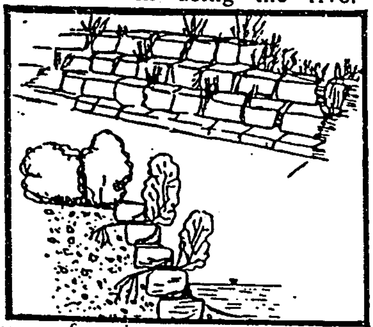


Figure 9 - Type of action proposed- Stone walls vegetated

5. Conclusions and Recommendations

In the last years it is intense the opposition made by the conservationists to the clearing river process, because most of the times this process is associated not with restoring techniques but to channelization. One other reason for this opposition resulted from the fact that the methods usually used cause the redution of the riparian vegetation and the river margins habitats

In this specific case and as a result of the ecological value of the Natural Reserve "soft techniques" where used instead of other conventional river engineering solutions.

To mitigate the adverse effects of channel realignment, the creation of meanders was proposed, as well as the selective clearing of the river bed and the indication of the trees and scrubs that had to be cut down or the removal of obstructions that were identified, being all this works, simple actions that could be effective to mitigate as floods.

Channel restoration in this case involved the use of living plants for bank protection in conjunction with inanimate materials such as gabions. We think that this experience could become an example that could be followed, too adequate other techniques or validate the ones that were implemented so that these actions could be used and adapted in other rivers.

Bibliography

-Brooks A., 1988. Channelized Rivers Perspectives for Environmental Management. John Wiley & Sons. Great Britain

-Chinery M. and, 1988. The Natural History of Britain and Europe. Guild Publishing. 4th edition, London

-Davies P., Gibbons B. 1993. Field Guide to Wild Flowers of Southern Europe. The Crowood Press(ed.) Great Britain

- Ferreira,T. 1993. Contribuição para a valorização biofísica do rio Almonda, troço do Paúl do Boquilobo e Torres Novas. Trabalho Fim de Curso. Èvora (non publis.)

- Gordon, N.D., Mcmahon T. A., Finlayson B. L. 1992. Stream Hydrology an Introduction for Ecologists. John Wiley & sons. Great Britain

-Nature Conservancy Council. 1989. Nature Conservation and the Management of Drainage Channels. Association of Drainage Authorities, England

-National Rivers Authority Thames Region. 1984. Flood Defence Guidance for Conservation in Water Courses Maintenance Works. NRA. Great Britain

-Schiechtl H. 1980. Bioengineering for land reclamation and Conservation. The University of Alberta Press. Alberta

IMPLEMENTATION OF NEW TECHNOLOGIES TO CONTROL EROSION ALONG THE BANKS OF THE ST. LAWRENCE RIVER

Marie-Claude Massicotte, Architecte paysagiste
Ville de Montréal
2155, rue Pie IX
Montréal (Québec)
Canada, H1V 2E4

ABSTRACT

The objectives of this project were to develop shore stabilization methods and to restore riverbanks representative of public properties belonging to the city of Montreal (in four municipal parks), and typical of erosion conditions along the St.Lawrence River, using new technologies combining bioengineering and more conventional techniques.

The goals are to restore, protect and preserve Montreal shorelines, to recreate adequate wildlife habitat conditions, to restore landfill sites banked up with inappropriate materials (concrete, waste, etc.), to reduce erosion caused by water level fluctuations, by surface runoffs and by ship induced waves, and finally to consider potentials and constraints related to various activities in the parks.

The banks of these parks are restored using one or more of the following techniques: riprap at the bottom of the slope and erosion control blanket; planting; branches and wooden stakes; gabion mats; combined rock laying and revegetation.

These shore restoration techniques are less costly than conventional techniques; they offer adequate protection to existing wildlife habitats and they allow the creation of new habitats when the potential is present.

The proposed techniques were developed by Montreal's park development section. Some were inspired by previos expertise (Panasuk,1987; Dumas, 1981; Schiechtz, 1980; Argus, 1991) but had never been tried on the Montreal island. They present innovations in terms of plants and materials used (erosion control blanket, gabions,etc.) and in implementation methodology. For example, the riprap-erosion blanket controls both soil surface erosion and seed washout, and is fully biodegradable. The blanket breaks down gradually over five years, progressively replaced by revegetation. If successful, these interventions can be repeated on eroding banks all along the St.Lawrence River.

KEY-WORDS: Erosion control blanket / Shore Stabilization / Bioengineering / Gabions / Riprap / St.Lawrence River / Revegetation

INTRODUCTION

As part of the St. Lawrence Vision 2000 action plan (technology demonstration component) a $300,000 project is currently under way, jointly and equally funded by the Ville de Montréal and Environment Canada. Its goal is to explore various ways of stabilizing eroded riverbanks, and to use a combination of bioengineering techniques and standard methods to restore sections of shoreline along four of Montreal's riverside parks: Bellerive, Bout-de-l'île, Raimbault and Stanley. These parks were selected because they represent the range of riverbank contours found on Montréal's shores as well as the erosion conditions present all along the banks of the St. Lawrence and its islands.

Of Montréal's 75 kilometers of riverbank, over 50% are plagued by erosion and stabilization problems. The riverside parks, which make up about 20% of the 75 kilometers, provide important recreational opportunities for the city's citizens, as well as a window on the St. Lawrence River and the Rivière des Prairies. They contribute to the city's quality of life by offering a unique context for a wide range of sports, as well as recreation and leisure activities.

Because these parks face the water, they are of special interest to people who enjoy aquatic sports, quiet contemplation of the environment and the beauty and attraction of the riverscape. Riverbank erosion problems must be solved because they compromise the "green" value of the parks. Any land reclamation work must be carried out with a view to maintaining existing activities and creating new land use possibilities, while at the time enhancing the aesthetic appeal of the shoreline. Riverbank erosion is a problem all along the St. Lawrence, affecting its islands and tributaries as well.

MAIN OBJECTIVES

The main objectives of the project are to:

- restore, protect and conserve riverbanks located in various Montréal parks;
- recreate environments conducive to the propagation of wildlife habitat;
- rehabilitate sites previously filled with inappropriate material such as waste, concrete, etc.;
- reduce the erosion resulting from water level fluctuations, surface runoff, and the undermining caused by the wave action of passing boats;
- stabilize the riverbanks, paying special attention to the possibilities and limitations created by the kinds of activities that occur in the parks.

The solutions explored by the project have very high exportability potential, since any successful interventions can be repeated on other eroded riverbanks in the city, as well as on sites all along the St. Lawrence River where habitat has been degraded or inappropriate fill used.

The project will lead to increased expertise in the field of erosion control techniques consistent with the natural character of Montréal's riverside parks.

As part of the project, we have tried to pinpoint the sections of riverbank most seriously affected by erosion, to identify the biophysical and human components of erosion and finally, to identify the techniques most likely to slow down or even stop the erosion affecting each section of riverbank.

The technologies used in the context of the project draw on bioengineering expertise as well as rock fill materials and methods. They include:

A) rock fill at the toe of the bank and erosion control blanket;
B) planting;
C) branches and wooden stakes;
D) gabion mats;
E) combined rock placement and planting.

These riverbank reclamation techniques are less costly than conventional techniques (concrete walls, etc.), and they promote the growth of important wildlife habitats, thus recreating a natural milieu that blends smoothly into the green spaces. It should be noted that several of the techniques involve working from the waterway itself because the riverbanks are sometimes inaccessible to heavy machinery.

The proposed techniques have been developed by Montréal's parks development section. Some were inspired by the work of experts in the field, including Panasuk (1987), Dumas (1980), Schiechtl (1980) and Argus (1991). None, however, have previously been tried on the island of Montréal. They are innovative in terms of either the type of planting and support material used (erosion control blanket, gabions, etc.), or the way they are implemented. For example, the blanket used in technique A (rock fill at the foot of the bank combined with an erosion control blanket) controls both surface soil erosion and seed washout, and is fully biodegradable. The blanket breaks down slowly over five years, and the stabilizing vegetation planted on the bank takes its place.

Description of the four riverside parks: wildlife potential and erosion control work

Promenade Bellerive Park

Promenade Bellerive Park stretches for 2.2 kilometers along the St. Lawrence. It is the first park downriver from Montréal's Old Port and includes the first stretch of unwalled riverbank outside that area. It is an important regional park, and considered a relaxation area.

Its banks are made up of fill from the construction of the Louis-Hippolyte Fontaine tunnel and the excavation of Montréal's subway system.

A large portion of its banks are denuded and have slope gradients ranging from 75% to 100%. Erosion has gained ground steadily over the years, compromising the park's green potential.

The port activity in the area and the proximity of the St. Lawrence seaway have sped up the degradation of the riverbanks, which have been altered by the effects of the undermining caused by the wave action of passing boats.

The park's wildlife habitat potential is as follows:

Strong points:
1. fast-running water - potential habitat for certain fish species (fishing in waders);
2. open water during winter, with potential for attracting diving ducks such as common golden-eyes and mergansers.

Weak points:
1. river corridor constantly disrupted by landfill and erosion;
2. sparse or nonexistent shrub and tree vegetation;
3. busy waterway.

At present, the riverbanks in the park have low potential for attracting bird life because of poor habitat (constant disruption, lack of tall grasses, shrubs and trees) and disturbances caused by visitors (bike path, grass playing field near the bank, domestic animals).

Recommendations:

1. restore degraded areas (clean and stabilize);
2. increase shrub cover on banks;
3. diversify the vegetation to include grasses, shrubs and trees;
4. plant vegetation that produces dense cover and fruit to attract wildlife (dogwood, elder, viburnum, etc.);
5. plant clumps of conifers to created sheltered areas for wildlife during winter;
6. conserve as much existing vegetation as possible;
7. conserve and develop observation points and access areas.

Work was carried out along a 150-meter stretch of riverbank where the greatest levels of inappropriate fill were present (concrete, cinders), material which compromises the visual quality of the site. Work included:

- placing rock fill at the toe of the bank to slow basal erosion;
- recontouring the slope;
- introducing humus and a biodegradable erosion control blanket;
- planting species of trees and shrubs such as willow, sumac, ash, walnut, thuja and dogwood, to recreate a riverside habitat designed to sustain wildlife.

Bout-de-l'île Park

This park is located at the eastern tip of the island of Montréal and borders on both the Rivière des Prairies and the St. Lawrence River. It is within a diversified zone that includes both strong currents and water plant communities in shallow, slow-moving water. The site is valuable for its habitat and potential as a wildfowl nesting area. Its present erosion problems involve washout and breakup of the riverbank.

The park's wildlife habitat potential is as follows:

Strong points:
1. presence of tall grasses (*phalaris arundinacea*) conducive to duck nesting;
2. proximity of island groups attractive to waterfowl (feeding and nesting) such as herons, paddling and diving ducks, terns, etc.;
3. some sections colonized by water plant communities (*scirpes*);
4. sport fishing areas;
5. observation point for water birds.

Weak points:
1. sparse shrub and tree cover on riverbank;
2. riverbank disrupted regularly (erosion);
3. busy waterway;
4. lack of water plant communities.

Recommendations:

1. increase riverside shrub and tree cover;
2. restore eroded banks;
3. develop observation points;
4. prohibit or limit access to certain areas (water plant communities) in order to conserve natural wild areas (territorial zoning).

Two sites has been developed:

A 77-meter stretch of grassy shore has been divided into parcels 10m x 10m in size. Rooted cuttings of trees and shrubs (willow, dogwood, spirea, silver maple, red maple) were planted, with a view to reforesting the riverbank, establishing a diversified root system to stabilize it and studying the competition between herbaceous and woody plants. This demonstration will enable us to evaluate the performance of particular shrub species under specific conditions.

On Marion Island at the tip of the park, work is to be carried out along a 50-meter stretch of shoreline, aimed at halting the crumbling and caving in of the banks caused by basal erosion (wave action), high spring water levels and ice. We have excavated a ditch at the toe of the slope and installed a barricade of cedar stakes, sunk to a depth of 1.2m and interwoven with sandbar willow branches, thus creating a reinforced plant base resistant to ice pressure and fluctuating water levels. The slope will be contoured and then stabilized by using layers of branches and fascines made of sandbar willow, as well as by planting willow cuttings on the bank.

Stanley Park

This north shore park stretches for 500 meters along the banks of the Rivière des Prairies and offers an exceptional view of l'Île Jésus. The park was constructed using fill from various excavations (subway, etc.). Washouts and high spring waters have eroded its banks year after year, threatening the survival of mature trees at the top of the slope and herbaceous vegetation along the riverside.

The park's wildlife habitat potential is as follows:

Strong point: 1. fast-running water - potential habitat for certain fish species.

Weak points:
1. very narrow shore;
2. very sparse shrub and tree cover;
3. very busy riverside area (bike path);
4. regular disruptions.

At present, the shores of this park offer minimal potential for attracting bird life because of poor habitat (constant disruption, lack of tall grasses, shrubs and trees) and disturbances caused by visitors (bike path, grass playing field near banks, domestic animals).

Recommendations:

1. restore sections of riverbank damaged by erosion;
2. increase shrub cover on the banks (willow, dogwood, etc.);
3. diversify vegetation groups (include grasses, shrubs and trees);

The work includes contouring the bank and placing rock fill at the toe of the slope, as well as planting rooted cuttings of dogwood and bog myrtle along a 50-meter stretch of shoreline.

Raimbault Park

This park was built in the 1950s and contains important mature tree growth. It also offers a variety of facilities, including playing fields, a children's playground and a wading pool. The shoreline includes a low plain whose trees are threatened by soil washout, as well as steep embankments with shrubs and trees at the bottom. There are also denuded sections of concave riverbank which have been undermined by wave action and high spring waters. The park includes an area of water plant communities in slow-moving water, which provide good wildlife habitat and spawning possibilities.

The park's wildlife habitat potential is as follows:

Strong points:
1. fish spawning area;
2. belt of shrub and tree cover along most of the banks with waterfowl nesting potential (paddling ducks);
3. water plant communities (*zizania palustris, saggitaria, nymphea* and *valiisneria*) and calm waters offers potential for attracting paddling ducks (widgeons, mallards, etc.), reptiles (turtles and frogs) and batrachians.

Weak points:
1. narrow shoreline, limited potential for attracting riverside birds (sparrows and water birds);
2. homogenous riverside habitat (no tall grasses);
3. very busy riverside site, resulting in continual disturbances (domestic animals, etc.)
4. ongoing disruption caused by erosion and deforestation.

Recommendations:

1. restore degraded areas (riverbank stabilization);
2. diversify the tall grass and shrub vegetation;
3. widen the shoreline if possible;
4. prohibit or limit access by pleasure craft to the cove facing section D2-D3, especially during spawning season (spring);
5. avoid stirring up sediment.

Two types of land reclamation has been carried out:

- At the first 100-meter-long site, gabion mats (wire mesh casings filled with rocks) are used in combination with a biodegradable blanket. In addition, the riverbank is reforested by placing successive layers of sandbar willow branches in the recontoured slope. The gabion mat is used to cover the denuded riverbank and secure the soil until the vegetation is well established. The work is carried out from the bottom up. Ditches have been dug in the slope for the willow branches, and cedar stakes have been driven in to support the plant-based structure and secure the willow branches. This site will eventually serve as a source of vegetation to be used in other riverside reclamation work.

- Work at the second site involves stabilizing 35 meters of riverbank by placing rock fill on the slope and planting rooted cuttings and dogwood in earth-filled jute bags. This technique affords an opportunity to introduce plant varieties with high rooting capability into uncultivated soil and rocky banks.

Courses given to municipal horticulturalists

In autumn 1995, a six-hour theoretical course was given to horticulturalists employed by the city, followed by two full eight-hour days in the field. The sessions were aimed at providing information about the riverbank reclamation work required in various parks and the techniques used to carry it out.

The theoretical course focused on:

- the identification of plant species used in protecting riverbanks (such as various willow species);
- the role of riverbank vegetation, the importance of plant diversity, and its impact on wildlife;
- the description of different kinds of riverside vegetation (planting, cleaning, pruning, support structures, etc.);
- short, medium and long-term approaches to waterway reclamation;
- various techniques used to consolidate riverbanks using plant materials, including wattle fence, fascines, branch mats, etc.

Completed works

Riverside landscaping was carried out in the four parks during autumn, 1995 and spring, 1996.

The work carried out in autumn 1995 focused on landscaping work in Raimbault and Stanley parks. At Promenade Bellerive, rock fill and slope contouring was completed, while at Bout-de-l'île, the 10m x 10m experimental parcels were marked out.

By spring, it was evident that construction at the Raimbault Park site had not been affected by ice. A protective blanket installed in the autumn was removed at the beginning of April, 1996. At that time, the willows had to be straightened because the weight of the soil had bent them over during the winter. At the second site, vandalism had damaged the wire mesh surrounding the jute planting bags, and high water and snow had caused some caving in of the planting ditches.

CONCLUSION

Taken together, these interventions have provided an opportunity to establish and evaluate the various steps involved in riverbank stabilization, from design to implementation.

The use of plant materials in the reclamation work has made it possible to evaluate the huge stabilization potential of vegetation while at the same time creating a supply source for future erosion control projects.

On the other hand, the harvesting and storage time for plants is relatively short. Consequently, the erosion control work itself must be performed in a limited period, since it must occur during very late autumn or very early spring when climatic and aquatic conditions are not at their best.

The use ob bioengineering techniques in riverbank stabilization must be built on a process of follow-up and experimentation. Monitoring and adjustement should be an integral part of the work. If a project succeeds, it is important to recognize and understand that a series of appropriate adjustments have been made all along the way. If a project falls short, we have to be able to identify its shortcomings so that the strengths and weaknesses of various techniques can be understood rather than rejected altogether.

Finally, we should remember that erosion is a natural phenomenon, and that not every eroded riverbank needs to be stabilized. Expert studies evaluating biological, ecological, pedological and especially hydraulic factors can serve to identify the causes of the instability and evaluate the risks involved in correcting it.

REFERENCES

- Ville de Montréal, *Proposition, mise en oeuvre de nouvelles technologies de contrôle d'érosion de berges en bordure du St-Laurent*, August 1994, 25 pages.
- Bernard Lachat, ministère de l'Environnement, *Guide de protection des berges de cours d'eau en techniques végétales,* Diren, Rhône Alpes, 1994, 144 pages.
- St. Lawrence action plan, St. Lawrence Centre, Environment Canada, *L'Érosion des îles du St-Laurent tronçon Montréal, Lac St-Pierre*, 1991, 120 pages.
- Bernard Lachat, Comité directeur pour la protection et la gestion de l'environnement et du milieu naturel, *Le cours d'eau, conservation, entretien et aménagement*, Strasbourg 1991, 83 pages.
- Schiechtl, H., *Bioengineering for land reclamation and conservation*, University of Alberta Press, 404 pages.
- Ville de Montréal, *Contrôle de l'érosion des berges dans divers parcs*, submission 6585.
- Verein Für, Ingénieur biologie, *Dictionary of Soil Bioengineering*, Zurich, 1996, 250 pages.
- Verein Für, Ingénieur biologie, *Bibliographie ingénieur biologie*, 1992, 207 pages.
- Special Bioengineering Landscape Design, September 1991.
- Course notes, Claude Anctil 1995.
- Roche, *Contrôle de l'érosion des berges en bordures du St-Laurent, Évaluation des techniques proposées et recommandations*, September 1995, 26 pages.

ALCAN QUEBEC: LAKE SAINT-JEAN SHORELINE STABILIZATION PROGRAM AND PRESERVATION OF NATURAL HABITATS

Suzanne Dupuis
Société d'électrolyse et de chimie Alcan Limitée
100, rue Saint-Joseph sud, bureau 104
Alma (Québec)
Canada
G8B 7A6

ABSTRACT

In the Saguenay–Lake-Saint-Jean area, Alcan operates a hydroelectric system that encompasses six generating stations and three reservoirs. Since 1926, when Alcan completed the construction of the Isle-Maligne power station, Lake Saint-Jean has been one of the reservoirs. The mean water level has risen by two to three meters in the summer and the fall, flooding lakefront lands and increasing shore erosion.

In 1986, Alcan and the Quebec government concluded a ten-year agreement which outlined the stabilization work to be carried out on approximately 100 kilometers of shoreline, and proposed new water level management rules. The objective of the shore stabilization program in Lake Saint-Jean is to offset the effects of erosion, taking into account technical, economic, social and environmental issues.

By 1996, 29 km of beaches were replenished and various types of rock structures will now protect more than 80 km of shoreline in farming, forest, cottage and city areas. A monitoring program has been established in order to prevent any secondary negative impacts on the biophysical environment; among the concerns addressed by the monitoring program are the integration of all the sensitive elements in the area and the preservation of wildlife potentials.

During the ten years of the program, the stabilization interventions have proven efficient in controlling or curbing erosion in the sectors originally affected. The protection of 150 hectares of wetlands had a positive impact on the preservation of their wildlife potential.

Forage fish species found in beach areas were monitored; important variations in fish community structures were recorded between 1989, 1990, 1991 and 1995. A comparison between reference stations and rebuilt beach stations showed that fish communities have retained their integrity, in terms of density and diversity, in locations where granular materials were spread on the beaches.

KEY-WORDS: Erosion / stabilization / conservation / wetlands / forage fish / beach replenishing / groin / breakwater / marsh / Lake Saint-Jean.

INTRODUCTION

The Lake Saint-Jean shore stabilization program initiated by Alcan is one of the most important stabilization programs conducted on inland water bodies in North America. Its purpose is to offset the effects of erosion along approximately 100 kilometers of shoreline, taking into account technical, economic, social and environmental issues and concerns. By the end of this ten-year program in 1996, 52 million canadian dollars will have been invested in stabilization works, in public consultation and involvement, and in a control and monitoring program of the physical, socioeconomical and biological impacts.

Since 1986, 29 kilometers of beaches have been replenished with granular material, protected or not by hydraulic works, and more than 80 kilometers of shoreline are now protected by long-lasting works such as stone linings, gabions, rip raps. The impact of these works on the resources and the multiple uses of the shoreline of Lake Saint-Jean and its tributaries are issues adressed by the monitoring and control program. Wetland and forage fish communities of the beaches have been the object of special attention in order to preserve their integrity.

BACKGROUND INFORMATION

Lake Saint-Jean, the size of an inland sea, is one of the largest freshwater bodies in southern Quebec. It is one of three reservoirs of the Alcan hydroelectric system in the Saguenay–Lake-Saint-Jean area (Figure 1). Six generating stations, tapping these waters, have an annual power production of 1940 megawatts, mostly used for aluminum production in the Alcan electrolysis plants in Quebec.

Following the start-up in 1926 of the Isle-Maligne power station at the outlet of Lake Saint-Jean, the mean water levels in the summer and the fall have risen by 2 to 3 meters (Figure 2). This increase in the summer-fall water levels has caused the flooding of lakeside lands, thereby accelerating the natural process of erosion.

As early as the 1930s, Alcan undertook shoreline protection measures, mostly the construction of rip rap in urbanized sectors. During the 1960s and 1970s, beaches were rebuilt by replenishing with sand or gravel.

Since the 1950s, summer cottages have sprouted along the lakeshore and several recreational and touristic activities have developed, attracted by long sandy beaches and interesting fisheries resources dominated by walleye (*Stizostedion vitreum*) and land-locked salmon (*Salmo salar ouananiche*).

In the early 1980s in order to maintain its shore protection efforts and to comply with the provincial regulations in the Environmental Quality Act, Alcan undertook an environmental and social impact study of shore stabilization works and of different water level management scenarios. Following public hearings, Alcan concluded a ten-year agreement with the Quebec government to conduct shore stabilization works along Lake Saint-Jean and its main tributaries. The agreement also established new rules for the management of lake water levels.

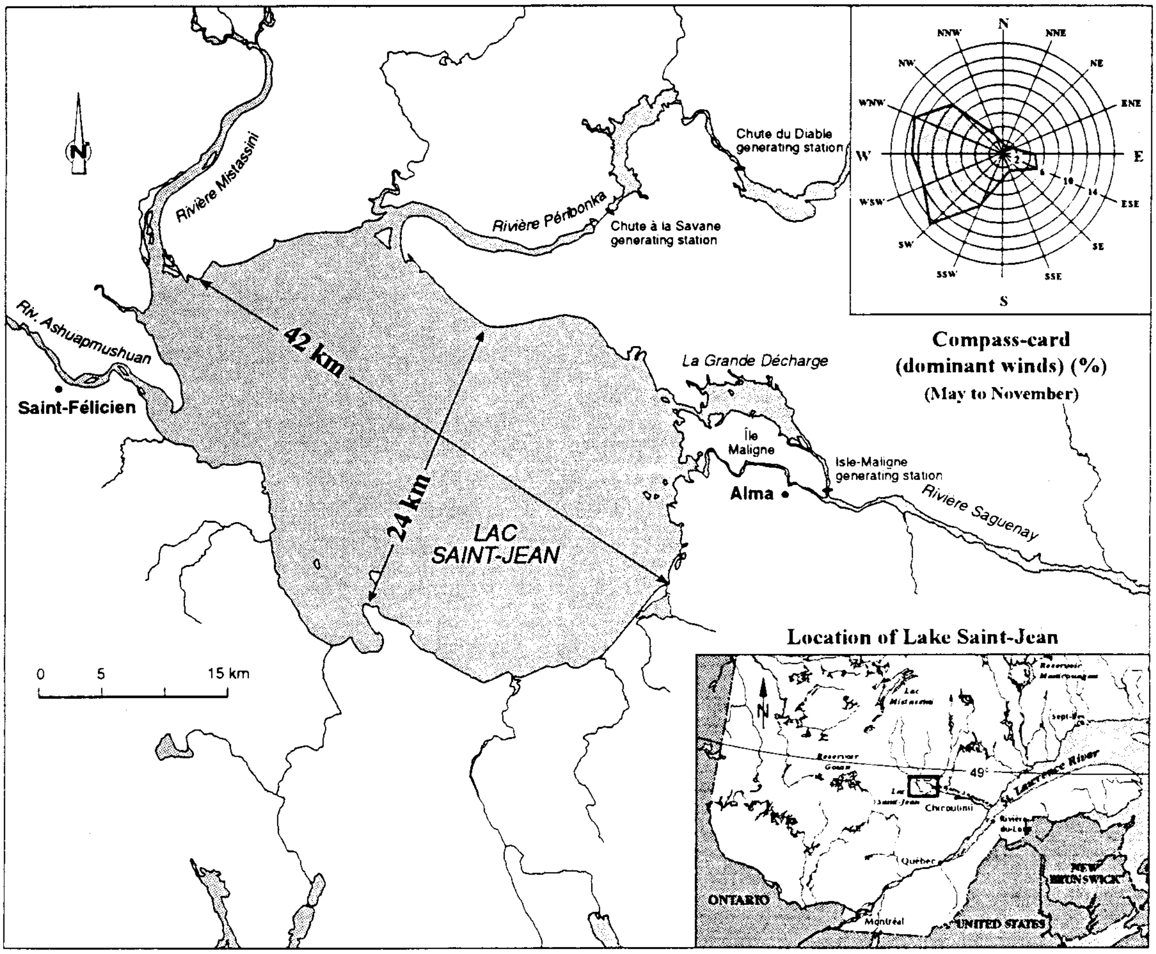


Figure 1: Site location of Lake Saint-Jean and Alcan hydroelectric system (partial)

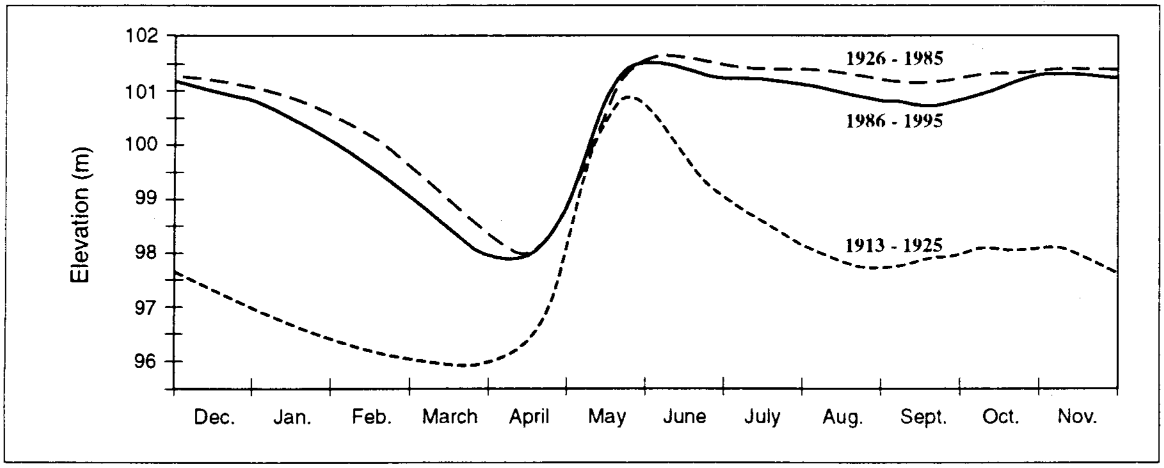


Figure 2: Lake Saint-Jean mean water level

THE PHYSICAL ENVIRONMENT AND EROSION

Lake Saint-Jean covers an area of more than 1000 km2; its main tributaries, the Mistassini, Mistassibi, Peribonka and Ashuapmushuan rivers provide for a rapid water replacement of 90 days on average (Leclerc, 1978).

The lake morphology is characterized by a reduced volume and a large surface area subjected to the action of southwest and northwest dominant winds. The total fetch in the northwest-southwest axis stretches for 42 km. The action of wind-induced waves, which may reach a height of 2.5 m with a high erosion energy, is felt on the lake shores during the open-water season, particularly during fall storms (Marsan, 1983).

Lake Saint-Jean has a total perimeter of 210 km. The shores of the lower reaches of the main tributaries and the shores of the lake outlet, la Grande Décharge, are included in the stabilization program, adding an extra 236 km of shoreline.

Of the total 436 km of shoreline considered, almost 47% are made of sand; clay represents the dominant shore deposit on 25% of the perimeter studied. Therefore, most of the shoreline (72%) are highly sensitive to erosion. Rocks and gravel cover 22% of the perimeter. Organic deposits occupy only 6% of the shoreline, but nevertheless they constitute wetlands of importance for wildlife (Marsan, 1983).

Prior to the stabilization program, between 1964 and 1981, almost 50 km of lakeshore had recided by 5 m to 30 m and more. An equivalent accumulation of sediments was observed at the mouth of the large rivers at the north end of the lake (Marsan, 1983). In the past ten years, active erosion was reduced on most of the affected sectors.

THE SOCIOECONOMIC ENVIRONMENT

In 1995, more than 37% of the studied shoreline was used for recreation purposes. From a socioeconomic point of view, holiday resorts and summer cottages are the most important users of lakefront, with more than 4000 houses and cottages. Swimming, boating (cruisers and sailboats), camping and fishing are at the heart of recreational activities and tourism development. Farmlands and forests cover 42% of the total shoreline, while the remaining 20% is divided between urban dwellings and marshes (Lusinchi, in prep.). The intensive human occupancy is a secondary cause of shore erosion.

THE LAKE SAINT-JEAN SHORE STABILIZATION PROGRAM

By the end of the Alcan program in July 1996, 108 km of shoreline will have been protected by various types of stabilization works.

Types of measures

The types of interventions applied were conditionned by shore composition, lakefront occupancy and the nature of erosion (Table 1). In beach areas, sifted natural material from nearby sand quarries are unloaded along the shore. In total, 28.7 km of beaches have been restored in the past ten years. In some sectors where the erosive energy is such that replenishing is short-lasted, hydraulic structures such as groins or breakwaters reduce the effects of erosion on replenishing materials. This added protection has proven efficient on 9.8 km of rebuilt beaches (Massé, in prep.).

Table 1: Types of measures and length of shoreline protected from 1986 to 1996

Shore utilization	Type of measures	Length (km)
Beach area	Sand replenishing (0-5 mm)	8.5
	Fine gravel replenishing (0-20 mm)	20.2
	Groin, breakwater	(9.8)
Cottage areas with no beach	Rip rap and concrete or flat rock accesses	16.0
Farmlands, forests, industrial areas, cities	Rip rap	37.6
	Gabion	2.7
	Stone lining (25-150 mm)	9.1
	Other	8.3
Wetlands	Dykes and rip rap	5.8

In farming and forest zones, in urban and cottage areas with no beaches and in wetlands, long lasting protection structures such as rip rap, stone lining (25-150 mm), gabions and water accesses built of concrete blocks or flat rocks protect 79.5 km of shoreline.

Revegetation by seeding or tree and shrub planting was used as a complement to rip rap in order to accelerate invasion by native embankment vegetation, and to ensure long term stability and integration to the neighbouring environment.

Control and monitoring program

A control and monitoring program was established from the beginning in 1986 by the Alcan multidisciplinary team in charge of managing the shore stabilization program. Its objectives were to ensure that the measure carried out are efficient, to assess and control impacts on the biophysical and socioeconomic environment, and to make proper adjustments in the course of the program, when judged appropriate.

The control and monitoring program comprises a follow-up of active erosion in order to better comprehend the process, a monitoring of the hydraulic structures to assess their resistance and efficiency, the surveillance of on-going work to ensure technical and environmental compliance to plans and blueprints, a socioeconomic follow-up addressing the evolution of shore occupancy and utilization, of pleasure boating, and of public satisfaction in which citizens can express their expectations.

The biophysical monitoring scrutinizes the potential secondary negative impacts on the biophysical components and proposes mitigation measures. It ensures that the wildlife potential of shore habitats is preserved, by assessing the evolution of wetlands and of forage fish communities.

RESULTS

Erosion control

Since 1986, Alcan has measured annually the width of the 45 km of beaches along lake Saint-Jean. Globally, all beaches including the 28.7 km of rebuilt beaches, have presented a mean increase in width of about 6 meters. In beach sections protected from the erosive energy of waves by hydraulic structures, groins or breakwaters, erosion has decreased significantly and, in certains cases, an important accumulation of material deposits was recorded. A better controlled shore dynamics will provide a greater stability to these dynamic shores and decrease the need for recurring replenishments.

The long-lasting protection structures (rip rap, stone lining, gabions, concrete blocks) have shown adequate structural behaviour. These structures now provide efficient protection to 16 km of cottage area, 53 km of farmland or forestland and 5 km of city area.

In total, erosion has been controlled or curbed on 108 km of shoreline along Lake Saint-Jean. Erosion, soil losses and consequent turbidity are considerably reduced from the original conditions observed prior to the program.

Preservation of wetlands

There are 3000 hectares of wetlands distributed around Lake Saint-Jean and the mouth of its tributaries. These wetlands provide important spawing sites for northern pike (*Esox lucius*) and yellow perch (*Perca flavescens*) and for some forage species, and nesting grounds and nursing areas for the 13 species of ducks nesting in the Lake Saint-Jean area.

At the beginning of the program in 1986, information was scarce on these habitats, some of which were affected by erosion. In the early stages of the program, various biophysical surveys were conducted on vegetation, physico-chemical conditions, and habitat bathymetry among others, over more than 1500 hectares of wetlands. Concurrently, sectors affected by erosion were identified in order to establish a protection program for lakeshore wetland habitats.

Priority was given to the 8 hectares Golf de Saint-Prime marshland. Recent habitat area losses were important and this wetland was threatened in the short term; in fact, a mean receding rate of 0.5 m/year had been measured between 1947 and 1985. In 1988, a 800 m dyke of fine granular material protected by stone lining was built, surrounding this habitat on the lake side. An opening, with a threshold, was included to allow for water and aquatic life exchange between the lake and the wetland (Figure 3).

These structures have provided efficient protection against erosion while maintaining adequate water levels in the marsh area to satisfy basic requirements for duckling rearing, even during low-water periods or during exceptionnally low hydraulicity events.

Prior to these interventions, the marsh could be totally emerged during the summer. With the improved habitat conditions since the measures were implemented, there are now between 8 and 14 duck clutches produced every year, belonging to six species; before, no more than three clutches had been recorded (Bouchard, 1996).

The positive results at this experimental site led to the protection of other wetlands affected by erosion. From 1988 to 1995, this type of measure was achieved on 5.8 km, corresponding to almost 5% of the total Alcan program for a total cost of 1.7 million dollars (Can.). More than 150 hectares of lakeshore wetlands loosing ground to erosion were thus preserved.

Also, the annual monitoring of shore habitats conducted during the past 10 years and to be maintained in coming years, ensures an adequate surveillance of the more sensitive sectors and of the evolution of wetlands. If necessary, measures can be taken to offset losses caused by erosion, a major contribution to the preservation and enhancement of wildlife potentials.

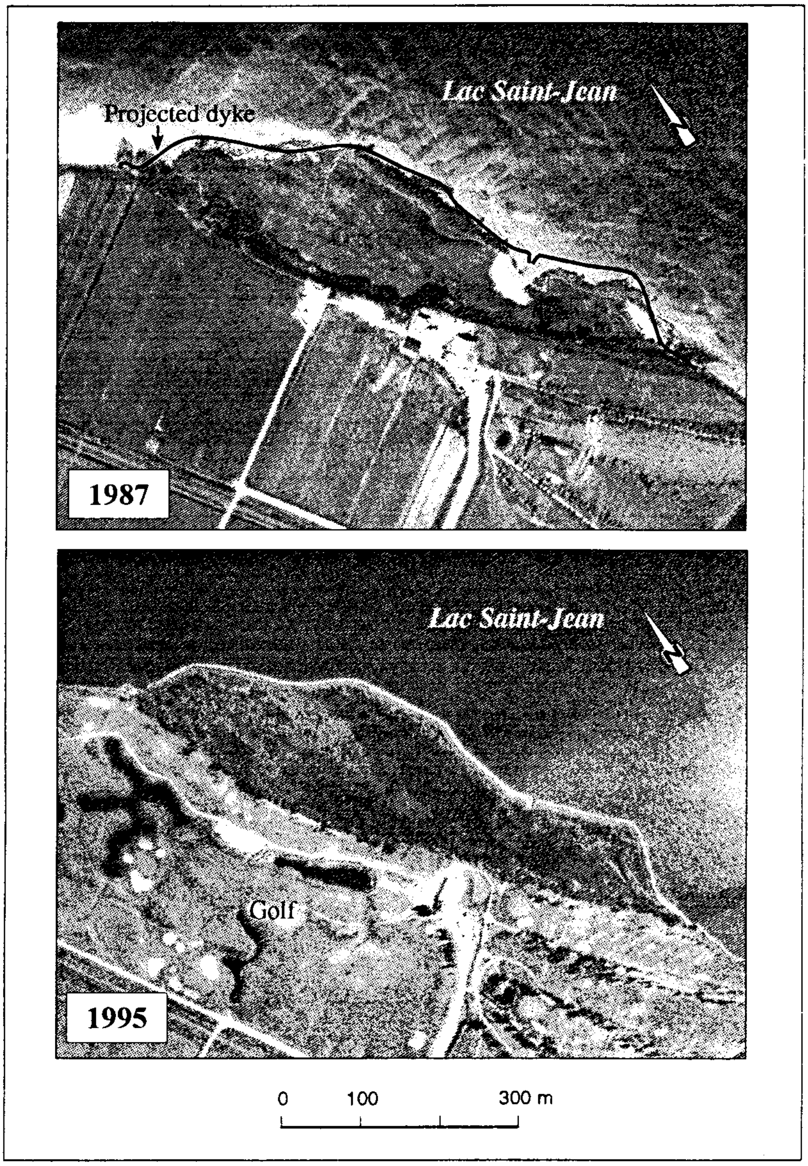


Figure 3: The Golf de Saint-Prime marshland in 1987 and in 1995

Evolution of forage fish communities

As part of the environmental monitoring of the stabilization measures, a survey of the forage fish communities present at beaches was conducted. Two objectives were pursued: a characterization of the communities, unknown at the beginning of the program, and an assessment of the potential effects of beach replenishing works. Replenishing material varied in size from 0-5 mm to 0-20 mm depending upon the sectors.

Following exploratory surveys in 1987 and 1988, the fish communities were sampled with a beach seine in 1989, 1990, 1991 and 1995, at 20 stations distributed in six sectors of the lake (Figure 4). Eleven stations at sites where interventions had been conducted and nine reference stations were sampled. Each station was divided in three sub-stations in order to limit sampling variability; sampling was conducted five times during the summer (June, July, August), in order to cover the fry emergence of all important species.

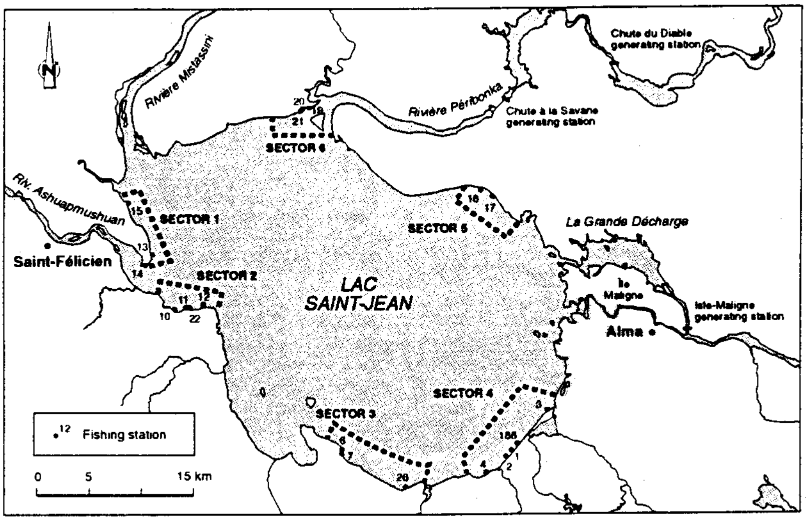


Figure 4: Location of fishing sectors and stations

Characterization of fish communities

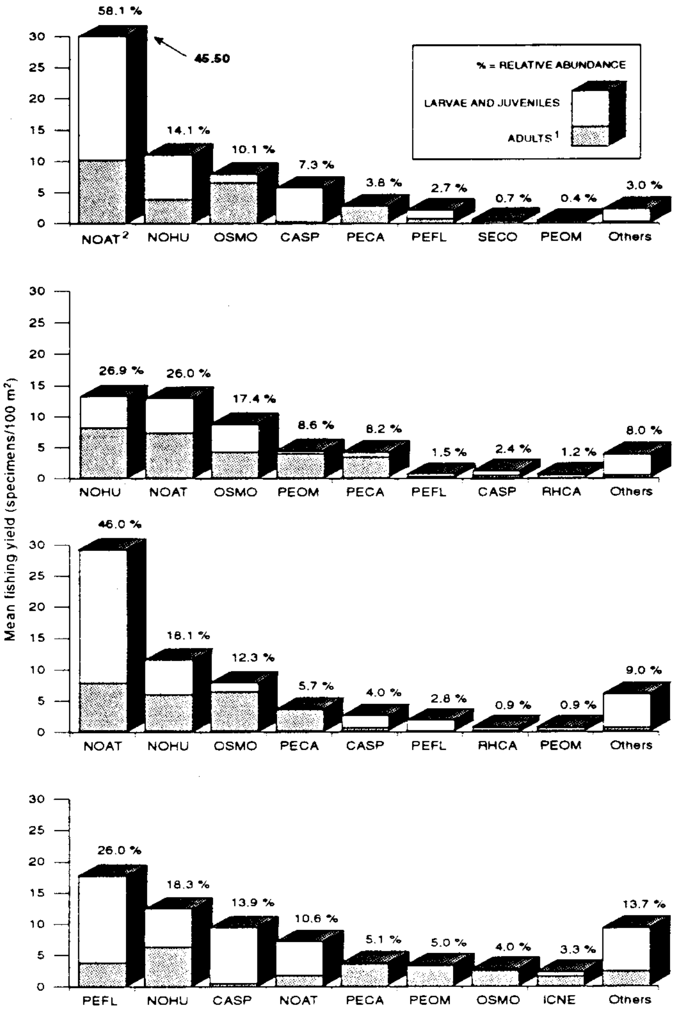
This monitoring program is one of the few large studies conducted in Quebec on fish communities present near beaches. The data collected is sufficient to draw a global picture of the fish populations using this type of habitat in Lake Saint-Jean.

Fishes of Lake Saint-Jean comprise 28 species of which 6 are predators and 22 are considered forage species. Surveys along the beaches indicated the presence of 21 species. The most abundant species, with an annual occurrence greater than 80%, are the emerald shiner (*Notropis atherinoides*), the spottail shiner (*Notropis hudsonius*), the rainbow smelt (*Osmerus mordax*), the yellow perch (*Perca flavescens*), the logperch (*Percina caprodes*), the trout-perch (*Percopsis omiscomaycus*) and catostomids. The other species collected are marginal (Figure 5).

The mean densities, all stations included, were respectively 78.3, 49.2, 62.8 and 67.7 specimens/100 m^2 in 1989, 1990, 1991 and 1995. Results also indicate that the species composition and the total yields vary greatly between years, according to sectors, ranging from 8.2 to 487.5 fish/100 m^2.

The emerald shiner showed the largest annual abundance fluctuations. Fishing yields of this species are higher in the eastern part of the lake while the spottail shiner dominates in the western and northern sectors of the lake. The highest densities of rainbow smelts were observed in the southern part of the lake.

In species where they were collected, larvae and yearlings frequently accounted for more than 50% of the density. At some stations, the proximity of tributaries or of marsh areas may have influenced the abundance recorded for species such as walleye, catostomids and yellow perch. However, fry occurrences higher than 70% and frequent high densities of emerald shiners and spottail shiners are indicative that these habitats are spawning sites for these species.



1989

1989 Total catch: 88 744	Mean fishing yield (spec./100 m²)	Shannon diversity index (H)
Sector 1	71.1	0.84
Sector 2	18.8	1.05
Sector 3	98.9	0.79
Sector 4	145.2	0.96
Sector 5	40.8	0.56
Sector 6	57.6	0.96
General mean (20 stations)	**78.3**	**0.86**

1990

1990 Total catch: 48 560	Mean fishing yield (spec./100 m²)	Shannon diversity index (H)
Sector 1	43.2	1.41
Sector 2	50.2	1.21
Sector 3	68.1	1.33
Sector 4	33.4	1.31
Sector 5	66.9	1.36
Sector 6	49.7	1.49
General mean (20 stations)	**49.2**	**1.35**

1991

1991 Total catch: 68 907	Mean fishing yield (spec./100 m²)	Shannon diversity index (H)
Sector 1	37.9	1.27
Sector 2	13.8	1.57
Sector 3	51.5	1.56
Sector 4	104.8	0.98
Sector 5	79.3	1.02
Sector 6	83.8	1.34
General mean (20 stations)	**62.8**	**1.29**

1995

1995 Total catch: 63 295	Mean fishing yield (spec./100 m²)	Shannon diversity index (H)
Sector 1	104.0	1.21
Sector 2	115.2	1.22
Sector 3	91.6	1.71
Sector 4	28.5	1.56
Sector 5	47.0	0.67
Sector 6	23.5	1.55
General mean (20 stations)	**67.7**	**1.32**

(1) Yields presented for adults include yearlings which have terminated the juvenile stage according to Auer (1982).

(2) NOAT : *Emerald shiner*
NOHU : *Spottail shiner*
OSMO : *Rainbow smelt*
PECA : *Logperch*
PEFL : *Yellow perch*
SECO : *Fallfish*
PEOM : *Trout-perch*
RHCA : *Longnose dace*
ICNE : *Brown bullhead*
CASP : *Suckers sp*

Figure 5: Mean density of the main species and mean yield per sector of forage fish communities captured by beach seining on the beaches of Lake Saint-Jean (1989-1995)

Effects of protection measures

Data analysis was conducted on a sector basis in order to account for important temporal and spatial variations of the fish communities along the beaches of Lake Saint-Jean and to isolate the effects related to replenishing. In this way, the incidence of environmental factors that could influence the structure of forage fish communities, such as the proximity of tributaries or spawning sites, the direction of dominant winds or other local factors, is limited.

Substrate granulometry was estimated visually at each fishing operation. The analysis of the evolution of this parameter between 1989 and 1995 showed that, with the exception of sector 4, it remained unchanged, both in the replenished sectors and the reference sectors, and that the granulometry of borrowed material is comparable to the natural substrate found at most stations (Table 2). In sector 4, a high wave energy and a major drift induce a segregation of material by size, resulting in the deposit of coarser material in this area.

In every sector, there were between two and five stations, including at least one reference station. The results at each of the stations where stabilization measures were applied were compared for temporal differences in diversity and yield with the reference stations, before and after replenishing operations (Table 2).

A comparison between the fish species diversity at stations with measures and at reference stations revealed higher or lower values in 55% and 45% of the cases, respectively. These propportions are 39 and 59% for local yields. The same type of analysis conducted on the densities of the main species revealed similar proportions, except for yellow perch and catostomids in which densities are more frequently higher at stations under stabilization measures. At the opposite, the densities of emerald shiner and spottail shiner are more frequently higher at reference stations. The variations are low when compared to annual fluctuations observed in these species and cannot be attributed to the effect, positive or negative, of replenishing operations. Temporal comparison between catches before and after interventions revealed no significant changes in the structure of forage fish communities (Environnement Illimité, in prep.).

As a rule, the results of this study indicate that beach replenishing with materials comparable in size to the original substrate had no major perceptible impacts on the structure of forage communities found in this type of habitat in Lake Saint-Jean.

CONCLUSION

The Alcan Quebec Lake Saint-Jean shore stabilization program will end in July 1996. During the ten-year program, erosion has been controlled or reduced on 108 km of shoreline, preserving more than 29 km of beaches, 16 km of cottage areas, 53 km of farmland or forestland, 5 km of city lakefront and nearly 6 km of wetlands.

The global approach adopted by the Alcan team, which integrates technical, economical, social and environmental aspects, resulted in efficient measures protecting the natural environment while satisfying public interests and concerns. The final situation assessment shows that the interventions have not only helped preserve the sectors affected by erosion, but, in several cases, have also enhanced the recreational and tourist or wildlife potentials.

In addition to acquiring knowledge on erosion processes and on the Lake Saint-Jean ecosystem, the monitoring program showed that interventions had no secondary negative impacts on the biophysical components. The building of dykes and regulation works around wetlands subject to wave action resulted in the preservation of more than 150 ha of prime wildlife habitat. Forage fish communities found along sandy shores retained their integrity, in terms of density and diversity, in the presence of borrowed granular material used to replenish beaches. The monitoring of forage fish along beaches is one of the rare extensive studies conducted in Quebec on these communities.

These actions are in the scope of the Environment Protection Policy adopted by Alcan in 1989, which stipulates that "the company must harmonize its operations with the environment, respecting all life forms that thrive in the system".

Table 2: Variation (%) of the Shannon diversity index and of the total yield at stations under stabilization measures and at reference stations

Sector	Stabilized station	Reference station	Work*	Survey year	Diversity (%)	Total yield (%)
1	13	14	G88, G90	1989	17	-26
	(2 2)**	(4 4)	E90, B90	1990	25	-68
				1991	-14	-54
				1995	1	184
2	10	12	G87	1989	36	30
	(2 2)	(4 4)		1990	-14	334
				1991	-23	166
				1995	-18	144
2	11	12	E91, S91	1989	23	-42
	(2 2)	(4 4)		1990	-26	203
				1991	-2	34
				1995	3	-21
2	22	12	S88, E89	1989	-3	-35
	(2 2)	(4 4)		1990	-8	-41
				1991	-26	71
				1995	30	-63
3	6	7	G87, E87	1989	57	-61
	(3 5)	(3 3)	G89	1990	-4	3
				1991	-4	111
				1995	1	411
3	26	7	S87, GA91	1989	-13	-19
	(2 2)	(3 3)	S94	1990	-29	-33
				1991	-6	-2
				1995	-17	40
4	1	188	G87, E88	1989	78	-63
	(2 3)	(4 5)	S92, E95	1990	-13	-13
				1991	38	5
				1995	-14	42
4	2	188	G86, E89	1989	57	-76
	(3 5)	(4 5)	E91, G91	1990	9	-73
				1991	34	-14
				1995	-26	63
5	17	18	S88	1989	57	79
	(2 2)	(2 2)		1990	32	4
				1991	93	-2
				1995	194	0
6	19	21	G86, S87	1989	3	43
	(2 2)	(2 2)		1990	50	-68
				1991	-9	-78
				1995	65	-22
6	20	21	S87	1989	2	-3
	(2 2)	(2 2)		1990	46	-29
				1991	-20	-12
				1995	58	-22
Years-surveys with higher values (%)					55	39

* G88: fine gravel replenishing, fall 1988; S89: sand replenishing, fall 1989; E91: groin, winter 1991; B92: breakwater, winter 1992; GA91: gabion, winter 1991.

** Granulometry (1989-1995)

	Grain size classes				
	1	2	3	4	5
Dominant	Clay	Sand	Sand	Gravel	Gravel
Subdominant	-	-	Gravel	Sand	-

Clay (<0.075 mm); sand (0.075 to 0.5 mm); gravel (>0.5 mm)

REFERENCES

Bouchard, L., M. Larose (1996). *Suivi de l'impact des ouvrages de stabilisation au marais du Golf de Saint-Prime*, Centre écologique du lac Saint-Jean, pour Société d'électrolyse et de chimie Alcan Ltée (Sécal), 35 pages, 5 annexes.

Environnement Illimité (en prép.). *Programme de stabilisation des berges du lac Saint-Jean - Évolution des milieux humides et des communautés de poissons fourrages*, pour Sécal.

Leclerc, M. (1978). Quelques caractéristiques limno-physiques du lac Saint-Jean. *Rapport sur la productivité biologique du lac Saint-Jean*. INRS-eau, 49 pages.

Lusinchi, P. *et al.* (en prép.). *Programme de stabilisation des berges du lac Saint-Jean - Bilan socio-économique 1986-1996*, Le Groupe Leblond Tremblay Bouchard, pour Sécal.

Marsan, A. *et al.* (1983). *Programme de stabilisation des berges du lac Saint-Jean, Étude d'impact sur l'environnement et le milieu social*, pour Aluminium du Canada Ltée (Alcan), 19 annexes, 3 rapports-synthèses.

Massé, G. *et al.* (en prép.). *Programme de stabilisation des berges du lac Saint-Jean - Envergure et efficacité des travaux 1986-1996*, Les Consultants RSA, pour Sécal.

Fishways, By-pass structures

Passes migratoires, systèmes d'évitement

SALMON STOCKS AND HYDROELECTRICITY: INTEGRATED MANAGEMENT ON THE JACQUES-CARTIER RIVER

Claude Beaulieu, hydraulician
Groupe-conseil Génivar inc.
5355 boulevard des Gradins
Quebec (Quebec)
Canada G2S 1C8

ABSTRACT

The development of the territory along the Jacques-Cartier River, near Quebec City, is representative of the environmental and industrial evolution of a watershed from the beginning of colonization to the modern era. For more than two centuries, forest and salmon exploitations have been the basis of the regional economy. At the beginning of the 20th century, the industrial revolution marked the disappearance of Atlantic salmon in the river in 1913. But the involvement of locals and a massive stocking program reintroduced salmon in the Jacques-Cartier in 1982. A fishway was erected at the Donnacona damsite in 1985 and the salmon population has been growing ever since. Several small hydroelectric stations, shutdown during the 1980s, are recommissionned between 1993 and 1996. And now, the salmon resource and the production of hydroelectricity, both closely linked to the river hydrology, must coexist. Meanwhile energy production is questioned in Quebec and the recurring financial involvement of government agencies in fish and wildlife protection is significantly reduced. The environmental issues on the Jacques-Cartier relate mostly to Atlantic salmon:

- the upstream migration over three power stations and one natural impassable obstacle;
- the downstream migration of thousands of salmon parrs between early May and mid-June;
- the determination of minimum instream flow needs;
- the preservation of a historical and cultural site;
- and the replacement of a fish migratory device commissioned about ten years ago only.

In conclusion, integrated resource management principles are presented, indicating that hydroelectricity cannot be produced to the prejudice of the environment and that studies conducted with competence and expertise are sufficient to avoid fatal errors. Our methods have to be revised to correct biased observations and recurring errors induced by lack of experience, and to account for growing public awareness and government requirements. In addition to expertise, these methods will include sufficient in-depth analyses to provide enlightened site-specific diagnosis and regulate promotors' performances.

Along with this management challenge, Quebec has several hydroelectricity and environmental projects requiring a sound selection to optimize territorial use.

KEY-WORDS: Jacques-Cartier River / Atlantic salmon / Fishway / Hydroelectricity / Integrated management / Fish stocking.

INTRODUCTION

The coexistence of hydroelectricity production and Atlantic salmon is not always easy. This paper presents an example of sustained development on the Jacques-Cartier River, an example which could be exported to other salmon rivers.

The objective of the paper is to demonstrate the possible coexistence of two very different interests in a given watershed. First, the recent history of the basin is reviewed. Then, the current socio-economic context is briefly addressed, pointing to the social concerns about hydroelectricity production in Quebec.

Finally, the salmon-hydroelectricity issue on the Jacques-Cartier river is summarized. The paper concludes with general principles for future interventions to harmonize hydroelectricity production and salmon preservation.

HISTORICAL NOTES

This chapter reviews briefly some historical elements compiled in CRJC (1995), Gingras (1994) and Germain (1984).

In the early days of the French colony around 1700, the resources in the Jacques-Cartier River Basin were shared by the French, the Hurons, the Algonquins, the Atikamekw and the Montagnais. The river itself was the major access way between Lake Saint-Jean and the Quebec City area. In 1763, the colony becomes British and significant changes occur in land uses. The mass arrival of Loyalists between 1792 and 1800, and of Irish and Scottish immigrants between 1817 and 1830, saw the development of the northern English-speaking villages of Saint-Gabriel de Valcartier, Stoneham and Tewkesbury. Some exclusive fishing rights are then granted to noblemen and to rich riverside landowners. Finally, the British Crown gives to soldiers and militians the right to establish the first fish and game clubs.

Early accounts and illustrations of salmon fishing on the Jacques Cartier are found in the paintings of Frédéric Tolfrey (1841 and 1845) at the Hospital Pool and Déry Bridge (Figure 1), Thomas Davies (Figure 2), Cornélius Krieghoff, James Cockburn and several others. The history of the Jacques-Cartier is a mirror image of the development of North America over several centuries. Over exploitation and poaching had reduced the salmon population by 30% between 1850 and 1856. Already, the authorities had to implement legal protection measures and the first salmon hatchery was born from necessity (Richard Nettle, 1857). Famous Canadian master-brewer, Joseph Knight Bosewell built the first fishway (54 meters long) on the Jacques-Cartier in 1867 to facilitate salmon migration above Lamothe's Hole (Figure 3).

In the Quebec area, industrial development begins with increasing timber commerce and shipbuilding; several timber merchants and shipyards occupy river mouths and coves between Sillery and Portneuf. In 1860, the industry slows down and, with the crisis of 1873-1879, a massive emigration to the USA and New-England is recorded. Around 1872, the construction of a railroad opens the river basin to new important companies (Jacques-Cartier Pulp and Paper, Bird and Son). Small business such as flow-milling, ironworks, smelters, bakeries, and spinning-mills flourished. Salmon exploitation continues and the northern part of the basin booms with recreational activities near lac Saint-Joseph and Tewkesbury.

In 1908, Henri Saint-Georges and John Foreman, owners of the Jacques-Cartier Fund, attract american businessmen to establish the Donnacona Paper Co. Ltd. founded in September 1912. A paper mill and a dam (6.5 meter high) are built. The dam supplies power and water to the mill processes. During the same period, two other dams with generating stations are built about 12 kilometers upstream (Bird and McDougall). Environmental issues were not considered and, by 1914, salmon has disappeared from the river (Figure 4).

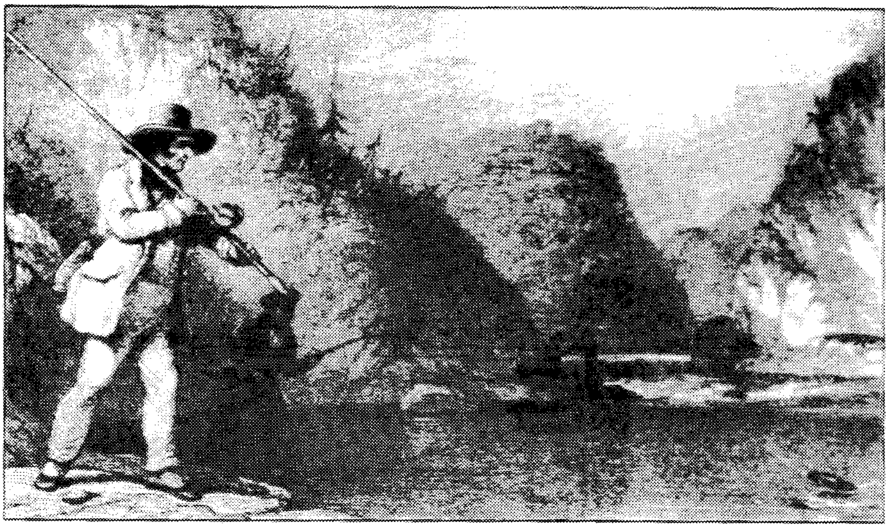


Figure 1. Engraving by Frédéric Tolfrey (1816) : salmon fishing on the Jacques-Cartier.
(In Gingras 1994)

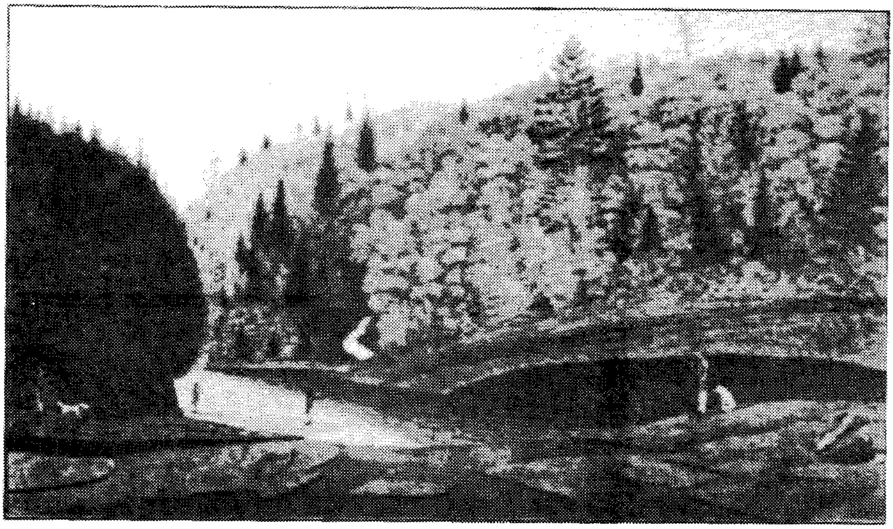


Figure 2. Water-color by Thomas Davies (1790) : salmon fishing in Hospital Pool.
(In Gingras 1994)

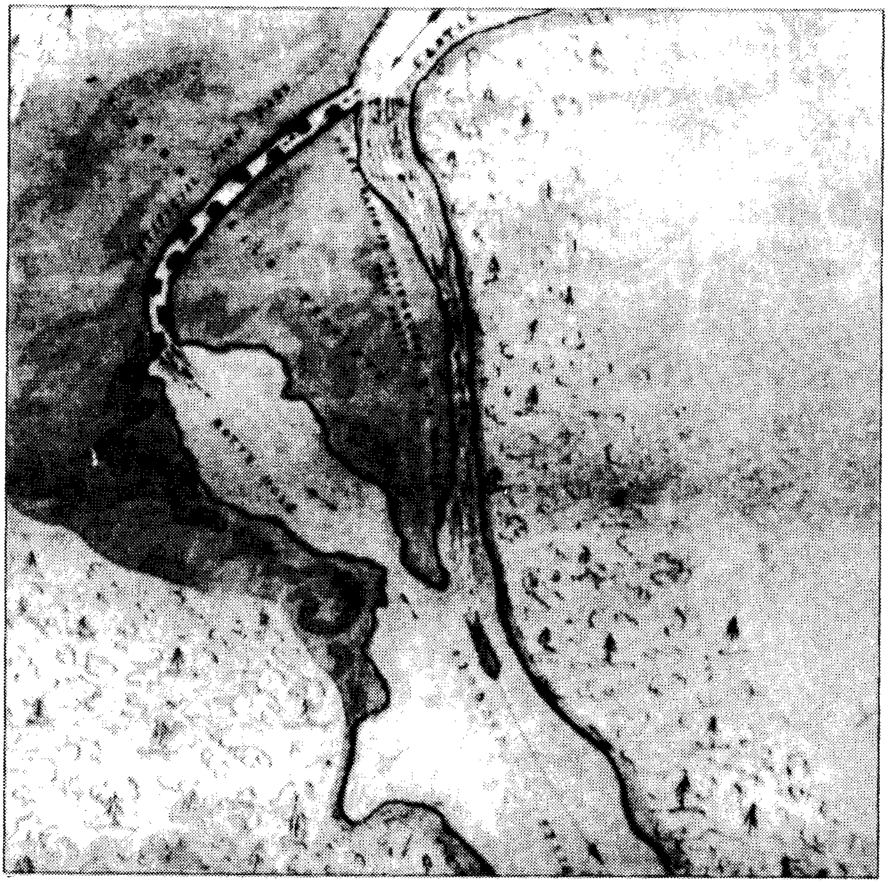


Figure 3. Drawing of the Joseph Knight Boswell fishway on the Jacques-Cartier River in 1867. *(In Gingras 1994)*

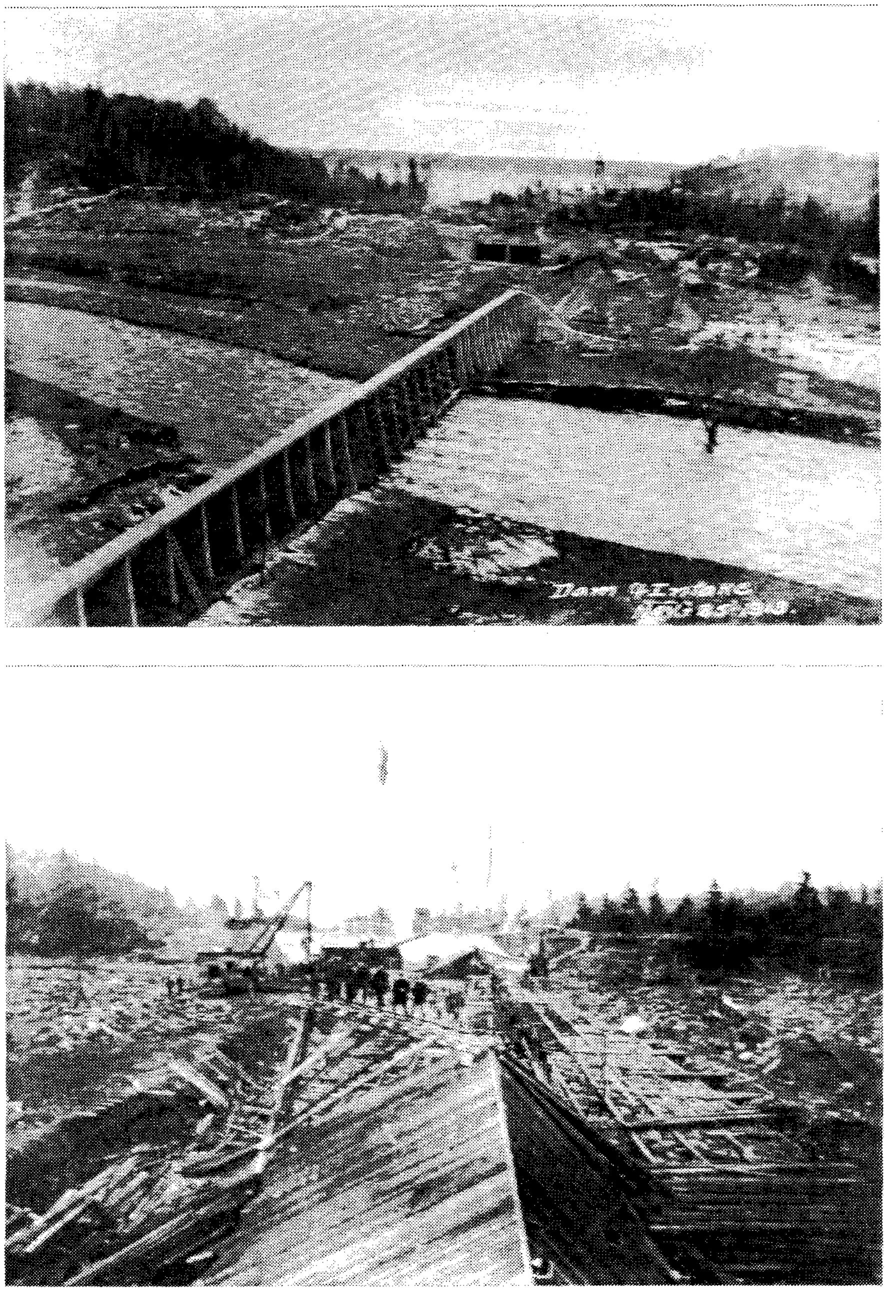


Figure 4. Photographs of the Donnacona dam in 1913. *(Domtar 1989)*

Only in 1979 will the "Comité de Restauration de la rivière Jacques-Cartier" (CRJC) formed by anglers and scientists, attempt to reintroduce salmon in the river. Stocking is conducted by the CRJC and the "Ministère du Loisir, de la Chasse et de la Pêche" (MLCP) and the first mature salmon return in July 1982. In 1985, a successive pool fishway is built on the left shore of the Donnacona dam. Since then, more than 5926 salmon have used it. A portion of spawners are collected and transported to upstream spawning grounds, in a conservation park (Figure 5). In over fifteen years, successes have followed mistakes and the population now totals about 1000 individuals. Current analysis estimate the potential at between 4000 and 5000 spawners per year on the Jacques-Cartier.

Between 1987 and 1990, several studies initiated by Hydro-Quebec indicated that it was financially sound to purchase a limited amount of energy (750 MW) from private sources. Among these, several small hydroelectricity stations were restored and recommissionned by private firms. On the Jacques-Cartier, three small generating stations are being restored. Socio-economic and environmental issues were addressed to avoid earlier mistakes and to implement true integrated management on the Jacques-Cartier.

THE SOCIO-ECONOMIC CONTEXT

The project to fully restore the salmon potential on the Jacques-Cartier while recommissionning small generating stations is happening in a social turmoil induced by the need to create employment, to cut-back government spending to protect our environment, to supply energy, and to satisfy people awareness and involvements. This chapter will summarize some of the major aspects of the reflection and discussion between government representatives, private promoters and regional interest groups.

THE ENERGY SITUATION

In our modern societies, agreements are note reached easily on topics such as economy, energy, environment, health, education, etc. After several years of expectations, Québec residents finally expressed their opinions and concerns in the course of a Public Commission on energy.

The Commission was rendered essential after the cancellation of hydropower sales to the USA, combined with energy conservation programs, native claims, pressure from ecologists, private electricity production, increased natural gas use, special industrial rates, the North American free-trade agreement, etc. The Commission should produce an Energy Policy in June 1996.

At first sight, it appears that Quebec citizens still favour hydroelectricity as a source of energy and of regional development. But consumers are not ready to pay more than for energy sources from western Canada, Ontario or the USA. Private production should be limited to hydropower and satisfy local and regional expectations. Ecologists want to see more emphasis put on energy conservation, while developing hydroelectricity production rather than CO_2 producing thermal sources or nuclear power stations. Several briefs propose a «green tax» on CO_2 producing energy to protect the environment and limit unfair competition.

Our operating power stations produce more than 35 000 MW of which less than 1 % is by private promoters. Hydro-Québec will not get involved in small-scale projects (less than 25 MW), because of high construction, maintenance and exploitation costs. But private promoters (with less overhead) have proven the viability of these projects in partnership with Hydro-Québec.

In June 1995, a report analyzing market trends in energy requirements and production for the next twenty years was released by the «Centre de développement technologique de l'École Polytechnique de Montréal». The study shows that the local market is saturated; if high energy-consuming industries stay away from the province, mega-projects will not be justified in coming years. The report also states that energy demands can be increased

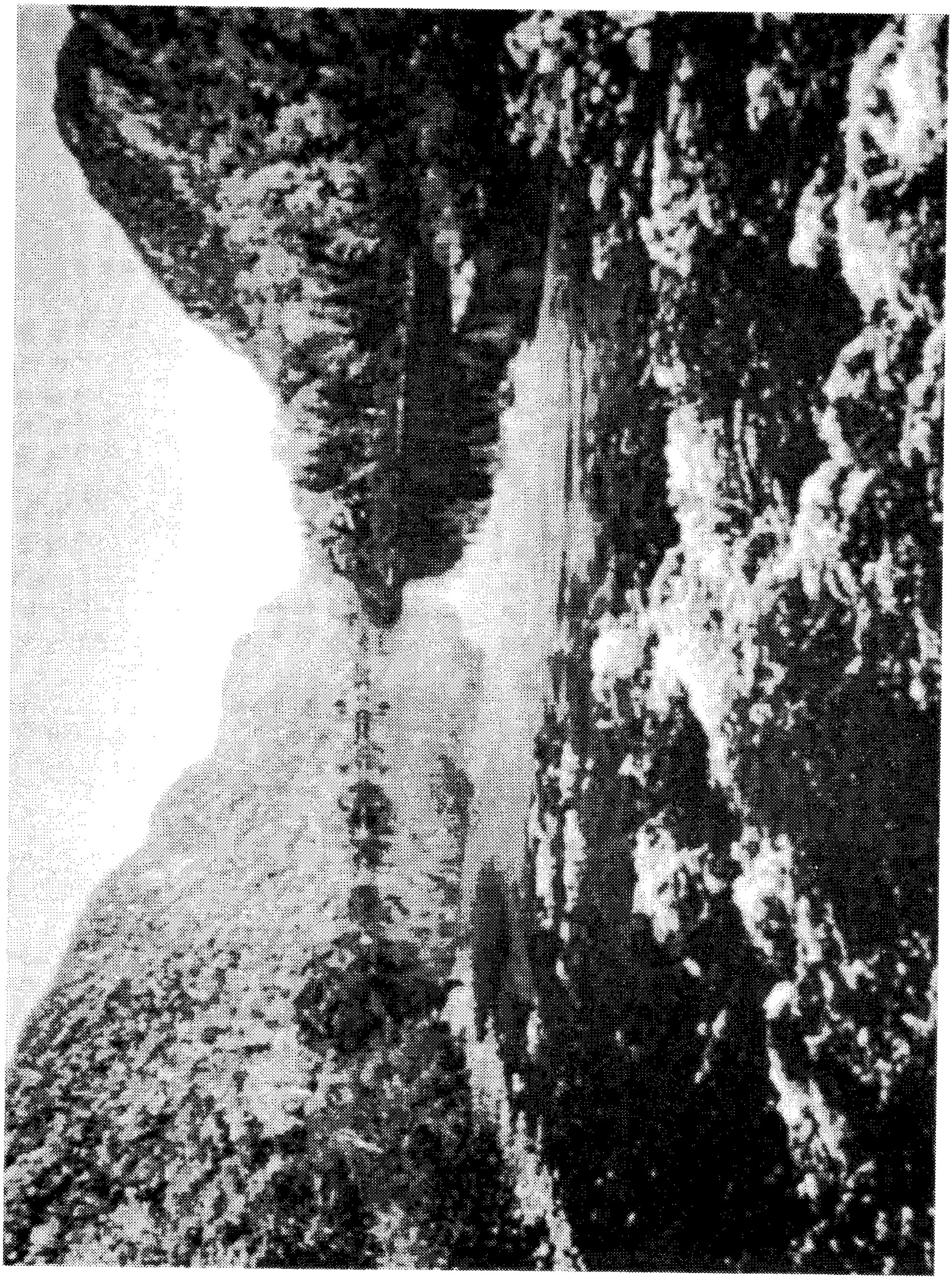


Figure 5. The Jacques-Cartier conservation park.

by invading the open USA market. The forefront private producers are waiting for government authorization to directly export to the USA. In fact, the 1994 North-American Free Trade Agreement favours electricity exportation.

Figure 6 shows that most US energy is produced by thermal generating stations (coal, natural gas, oil); nuclear stations and hydropower stations represent only 8 % of total production. The main goal of the free trade is to induce a 1 % rate decrease by the year 2005.

SALMON AND REGIONAL DEVELOPMENT

Regional development is at the heart of politicians' preoccupations and a varied and independent economic network in a given region is absolutely essential, creating public awareness and inducing government's interventions.

In our regions, economic development is mostly based on natural resources and primary productions, which are submitted to international market fluctuations. In several regions, monolithic economies have proven near-fatal in the face of stock collapses or vigorous international competition : ironore (Schefferville), copper (Murdochville), lumber (Gaspé and North-Shore), commercial fisheries (North Shore, Magdalen Islands, Gaspé).

With increasing unemployment in most regions, the governments have reacted by promoting job-creating recreo-touristic activities.

Journalist Rejean Lacombe of Le Soleil newspaper recently drew a portrait of the economic importance of salmon fishing in Québec; hecited a study by economist Luc Michaud (1996) revealing that salmon angling generates a net benefit of 97 million CAD annually, 92 % of which remains in Quebec, for a gross value of 1.2 billion CAD. Zins Beauchenes and Ass. (1995) reported that salmon sport fishing supports more than 616 direct and indirect jobs per year in Québec.

Despite government's investments of more than 25 millions CAD in about thirty development projects on salmon rivers, several regional managers are facing financial problems after subsidies cut-backs. Also, the local salmon fishing market is becoming saturated. The Zins Beauchesne survey indicates that the sale of salmon licenses decreased significantly in 1994 and the number of anglers has levelled off. However, the study shows that a minimal marketing effort and an opening of US and european markets would double the demand in three years only. The study also recommends to emphasis high and mid value products to cover exploitation costs, generate benefits and satisfy a market demand.

On the other hand, the full fish and game, and recreo-touristic potentials are far from being reached in any Quebec river basins. Several rivers offer great attributes to attract clientele and promote regional development. But these potentials have to be clearly identified and development projects have to be selected on the basis of gaining maximum value for governments, local interveners and private industries. Curiously, these high potentials are often found on rivers with high hydroelectricity potentials.

Integrating hydropower projects and salmon reintroduction on the Jacques-Cartier River has induced a sustainable development based on partnership, and shared efforts and benefits. The quick return on investments of hydropower projects has supported the reintroduction of salmon and the maintenance of several non-specialized jobs. This example of partnership represents a new approach in industrial and regional development, integrating the exploitation of two divergent resources on the same river.

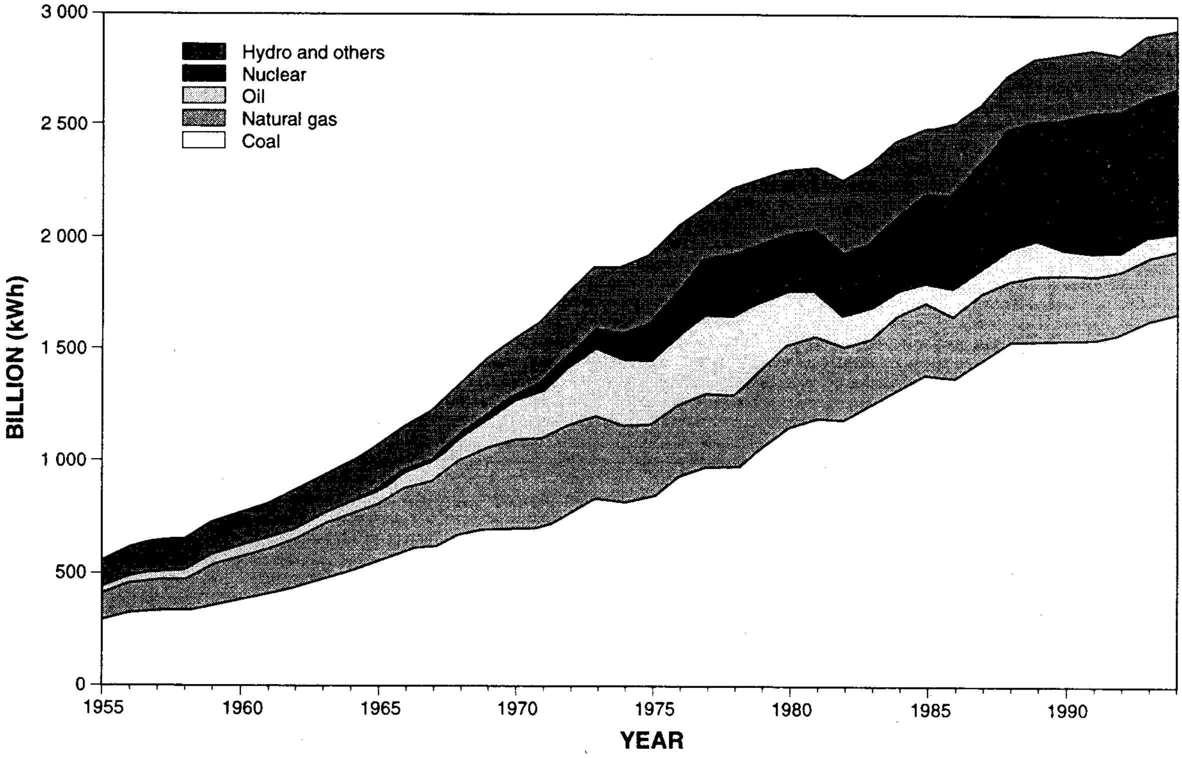


Figure 6. Electricity production in the USA between 1955 and 1994, by source.
(Centre de développement technologique de l'École Polytechnique de Montréal)

ENVIRONMENTAL CONTEXT ON THE JACQUES-CARTIER RIVER

The recommissionning of small generating stations on the Jacques-Cartier required a careful assessment of environmental parameters in order to harmonize the projects with environmental protection and to ensure profitability of both ventures (Figure 7). History shows that salmon life cycle and behaviour on the Jacques-Cartier are readily affected by the presence of impassable obstacles. Thus the main environmental issues concern the maintenance of salmon migration, both upstream and downstream, and the regulation of minimum flow requirements for the species. The Déry gorges above the Hospital Pool at Pont-Rouge are a historic site and required a special attention; the Donnacona dam fishway also needed specific controls. The analysis criteria used to integrate these concepts with the commissioning of small generating stations are described in the following paragraphs.

- Species involved : four environmental studies have reported the presence of 26 fish species in the basin. For resident species (more than 20), river dams have little impacts on their habitats and their occasional passage through turbines should not affect overall population survival (Paul Ruggles, pers. comm.). Eel and brown trout movements are controlled at the fishway and both species are kept below the obstacle (Boudreault et al., 1995). Thus, Atlantic salmon is the only migratory species potentially affected by the small power stations.

- Impassable obstacles : The Donaconna dam, built in 1913, had a major impact on the upstream salmon run. This obstacle was finally overcome in 1985, by the construction on the left side of a fishway with a catch and transport device to release spawning salmon above current obstacles (at Pont-Rouge), near the protected spawning grounds in the Jacques-Cartier Park.

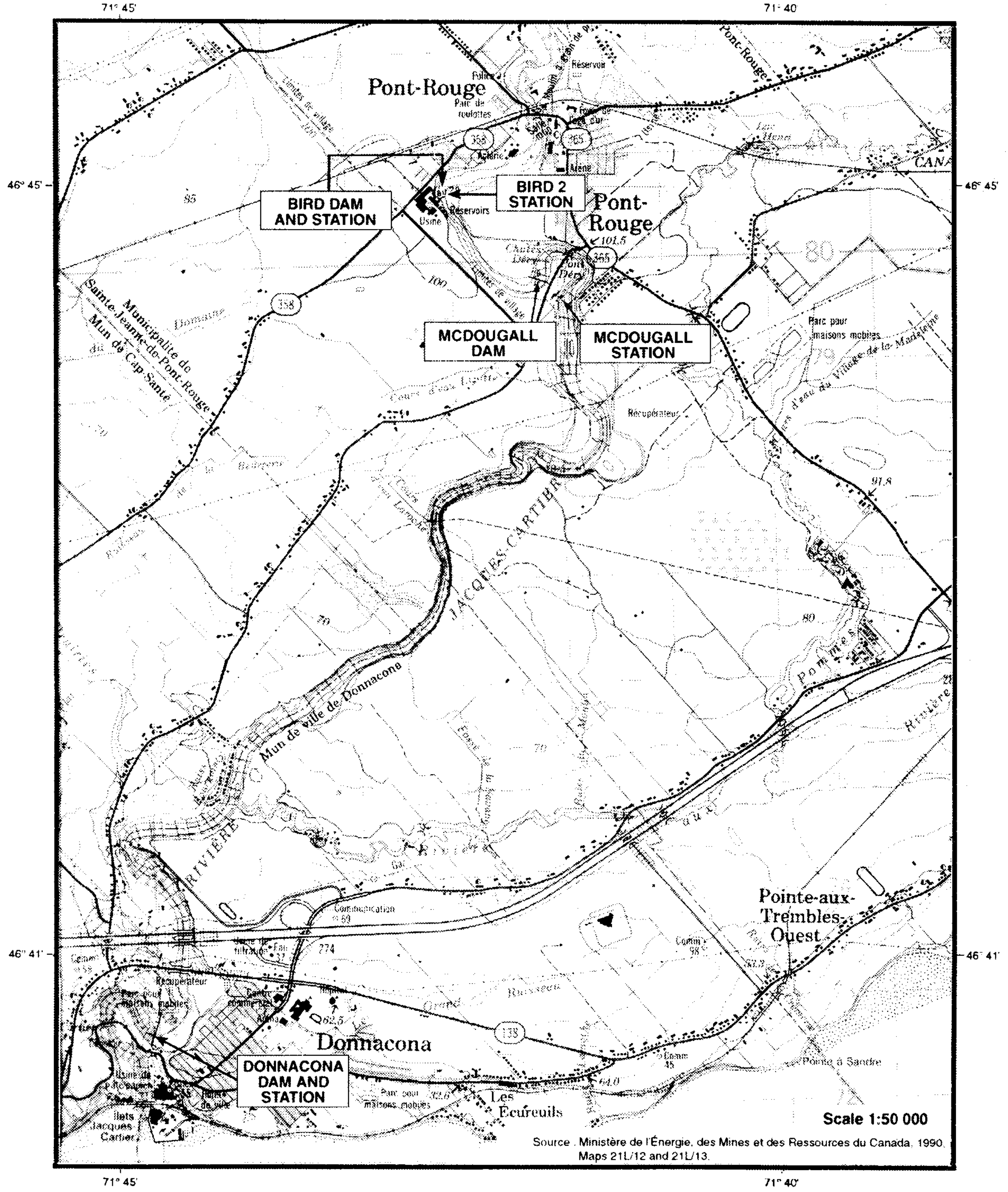
Donnacona fishway

The release of 25 000 salmon parrs in 1981 resulted in the return of four grilses at the Donnacona dam in 1982. The following year, a behaviour study on 68 returning salmon was conducted. A temporary migration device, made of removable wooden boards, was moved along the dam's apron (Figure 8). This technique was used until 1985 in order to validate the hydraulics hypothesis and to support negotiations with riverside neighbouring landowners for the construction of a fishway.

Hydrotech conducted biological studies and hydrological and hydrodynamics analyses to design an appropriate fishway. A review of existing technologies added useful elements in selecting a proper design. Hydraulic data were collected during two summer campaigns and water levels were modelized above and below the obstacle. Depth contours were measured, a statistical analysis of recurring hydrological events and a study of flow pattern behaviour were conducted.

At the end of each field season, engineers and biologists discussed the relevant parameters for the design of the fishway. In 1982, the site for the fishway was finally selected (on the left bank) on the basis of hydrodynamic and bathymetric data, general site morphology, angle of the dam with the river, and salmon behaviour at the foot of the dam. The mean flow rate measured during salmon migration is 55 m^3/s; the water level variation above and below the fishway is approximately 1.1 m over a 15 year recurrence period.

Four options were considered before a final design was retained. The 47 meter long fishway answers the requirements presented in Table 1 and Figure 9. A 0.75 m^3/s flow is added to the basic successive pool fishway discharge.



Source : Ministère de l'Énergie, des Mines et des Ressources du Canada, 1990. Maps 21L/12 and 21L/13.

Figure 7. Location of dams and power plants on the Jacques-Cartier River.

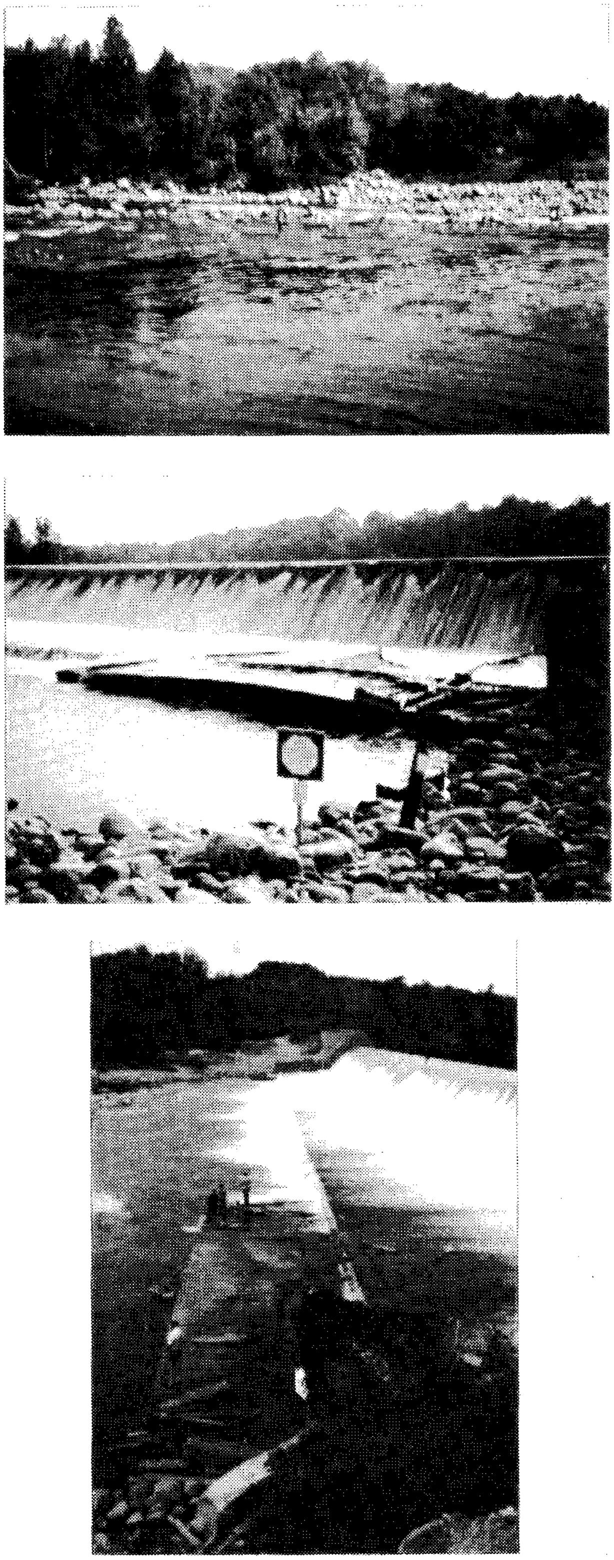
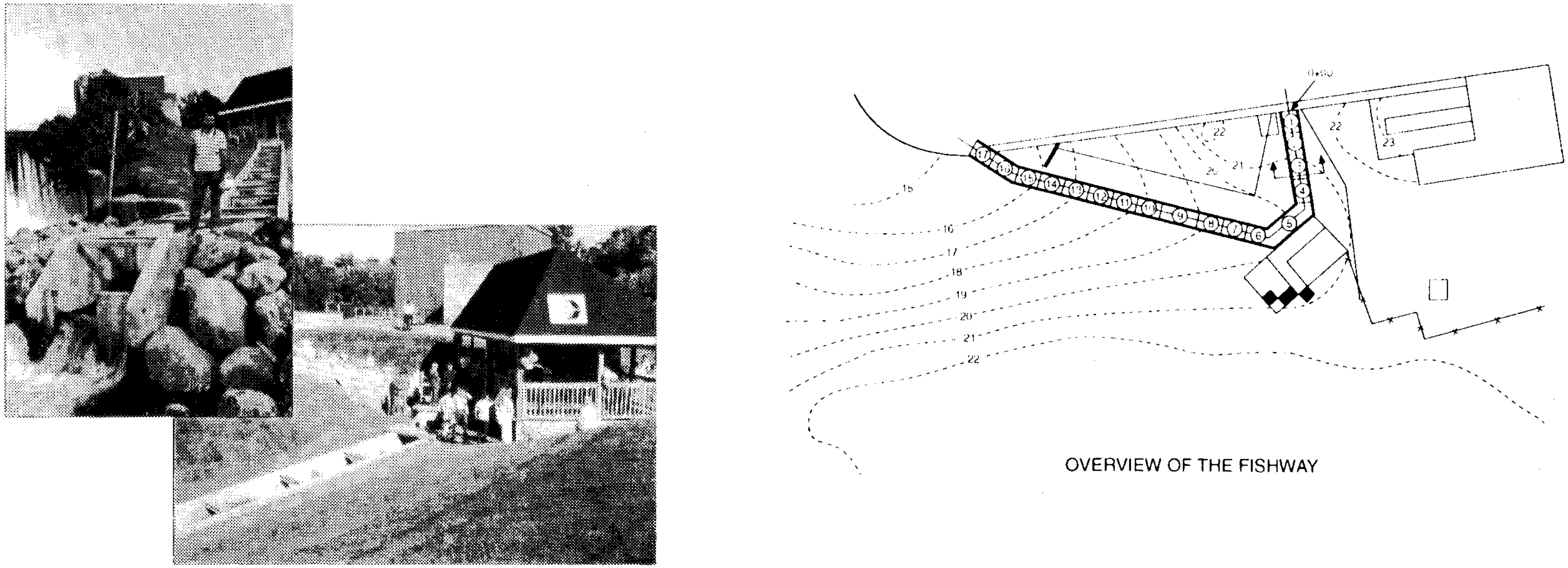
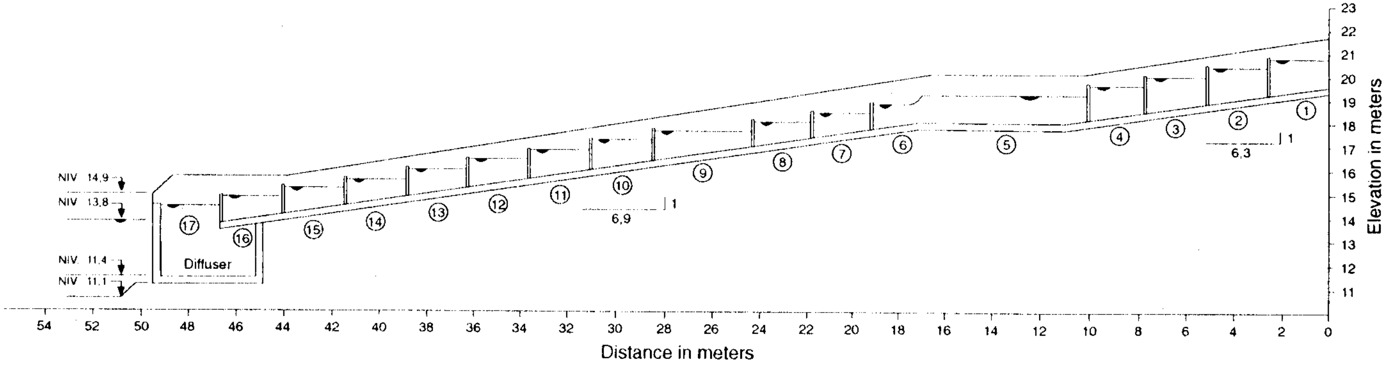


Figure 8. Biological sampling and hydrological measurements for the design of the Donnacona fishway in 1982.



OVERVIEW OF THE FISHWAY



DETAILED LONGITUDINAL VIEW OF THE FISHWAY

From Hydrotech, 1985

Figure 9. Diagram of the Donnacana fishway on the Jacques-Cartier River.

Overall, the fishway has satisfied the expectations, although performance was somehow mitigated when discharges were greater than 50 m^3/s; salmon then tend to aggregate on the opposite bank and may not find the fishway as easily. For example, the hydraulic conditions were extreme in 1994 and the majority of salmon were delayed by two to three weeks at the fishway. This is explained by a turbulent front masking the entrance of the structure under strong discharge conditions. The initial design was based on limited behaviour observations on very few salmon during normal hydraulic conditions. The 15 year recurrence period for hydrology analysis could be extended. Limited budgets in the mid-80s refrained the construction, exploitation and maintenance of a second fishway on the right bank to ensure upstream runs during strong hydraulicity periods. With current knowledge, the design of the fishway would allow a higher discharge to counterbalance spillings over the dam; a second entrance would be built 5 to 7 meters downstream and civil works would prevent undercuttings at the foot of the concrete structure.

The Donnacona fishway remains one of the most efficient in Quebec, along with the Matane and à Mars River fishways (Beaulieu, 1993), even if the design could be improved.

The Donnacona fishway experiment demonstrate the importance of :

- sound hydrological and hydraulics database;
- deep insights in salmon behaviour at the foot of the obstacle under various and extreme flow conditions.

Thus, an efficient fishway necessarily requires deep planning to avoid mediocre performances requiring adjustments at additional costs. From now on, fishway projects should be thoroughly examined by a committee of experts or professionals familiar with this technology.

Future fishway at the Donnacona dam

The small hydropower station at the Donnacona dam will be equipped with eight small turbines providing a total outflow of 86 m^3/s, concentrated on the bankside opposite to the current fishway. It is assumed that salmon behaviour will be modified in accordance and that a new fishway will be needed on the right bank. Observations on salmon behaviour under these new conditions will be collected in 1996 and 1997. Hydrometrics and civil work data will be updated and a preliminary design should be available during the summer of 1997. Construction of this new structure should take place in the fall of 1997. Meanwhile, salmon will be directed to the left bank fishway for another year. Performance guaranties are required by the ministère de l'Environnement et de la Faune (MEF) and by CRJC, based upon a statistical analysis of 10 year salmon run data. The efficiency of the new fishway will be continuously monitored; whence it has proven efficient, the old fishway will be decommissionned. The private hydropower producer will finance the use and maintenance of the new fishway.

SALMON MIGRATION IN THE DÉRY GORGES

The Déry gorges at Pont-Rouge are classified a Canadian patrimonial and historical site. The ministry of Cultural Affairs desires to maintain the site as pristine as possible. On the other hand, MEF and CRJC want to regulate this almost impassable obstacle to salmon migration. With the arrival of a private electricity producer, favourable solutions to maintain the aesthetics of the landscape while building flow regulation and fish passage works are now possible (Beaulieu, 1993; Gauthier, 1994).

Since salmon reintroduction has been initiated on the Jacques-Cartier River, CRJC never had to worry about salmon migrating through the Déry gorges, 12 kilometers upstream from Donnacona; few salmon actually return between 1982 and 1993 and a catch and transport device at the fishway provided spawners upstream from the gorges. In 1992, CRJC ordered a prefeasability study for potential regulation works at this highly selective site for migrating salmon.

TABLE 1. Characteristics of the Donnacona fishway

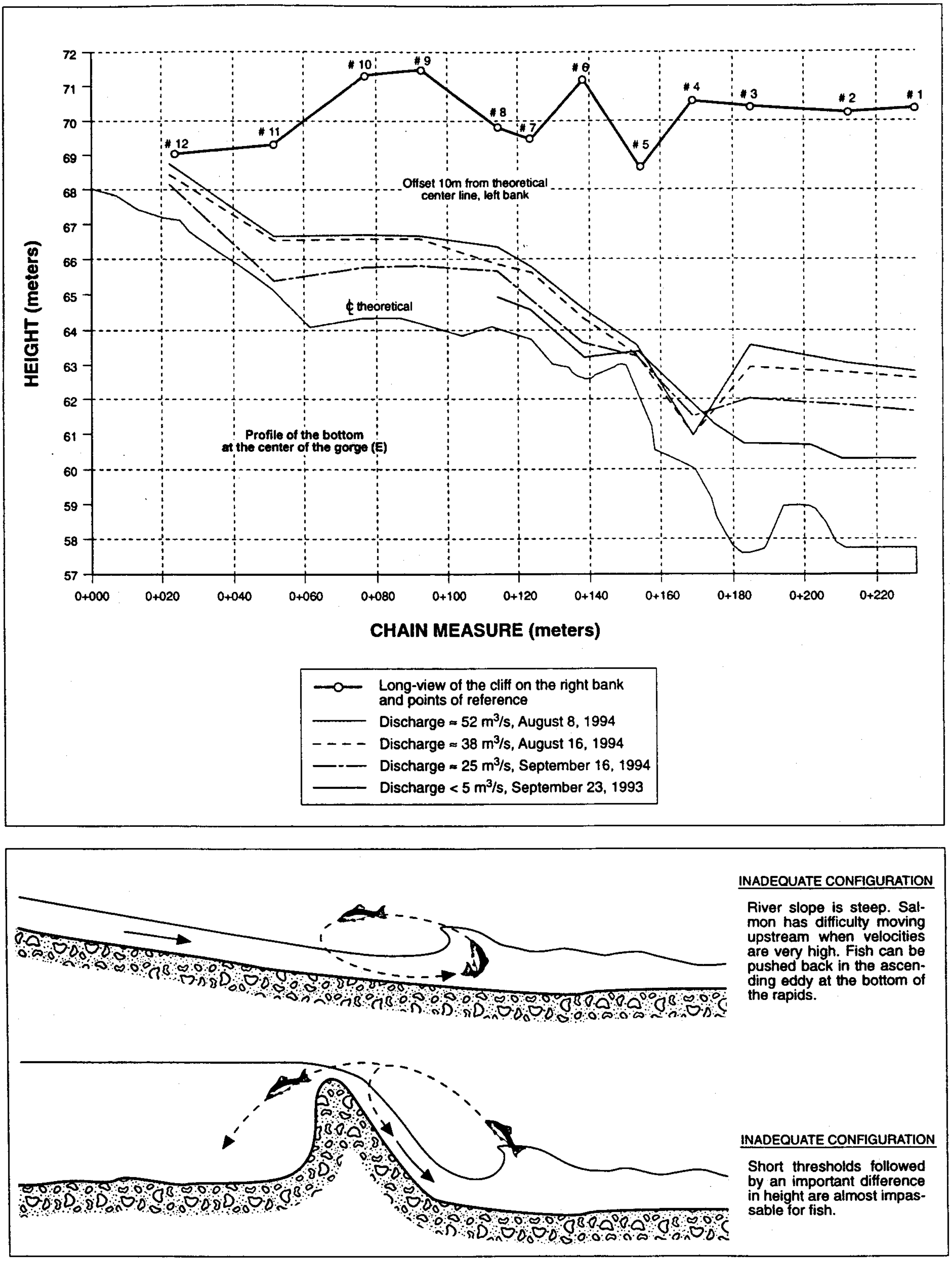
Type	Alternate notch weir
Total length	47 m
Width	2 m
Chamber length	2.5 m
Height	1.8 m
Notch	500 mm x 500 m
Average slope	15 %
Hydraulic depth	1.20 m
Flow	1 m^3/s
Average velocity in chamber	0.42 m/s
Difference in height between chambers	0.40 m
Headbay level	20.80 m
Afterbay level	13.80 m
Period of utilization	June 15 - September 15
Annual upstream run	
• mean (1985 to 1991)	496 salmon
• optimal	3500 salmon
Alliance water intake	capacity : 0.7 m^3/s

High flow velocities (5 to 6 m/s) and two hydraulic rises strongly hamper salmon migration success at the Déry gorges (Figure 10) (Beaulieu, 1993). Two scenarios are proposed and both imply major site transformations at high costs (Figure 11).

In 1993, a project to recommission the McDougall generating station rises and the environmental assessment reveals new avenues to regulate the flow in the gorges. A telemetry study (Beaurivage, 1983) had showed that salmon rarely passed the gorges on first attempt; a mathematical model also revealed potential migration difficulties caused by hydrodynamic changes in the Déry gorges and in Hospital Pool after contact with the McDougall plant tail race (Carter, 1992). Validation and additional measurements were required.

A second telemetry study followed the behaviour of 17 salmon between Hospital Pool and the McDougall dam, during most of the summer of 1994; it confirmed the difficulty to overcome the gorges (Figure 12). Even by diverting 22 m^3/s through the McDougall station, hydrodynamic conditions remained improper for salmon migration. Only when regulating flow rates at the Bird dam to simulate discharge conditions equal to 10 m^3/s in the gorges, was salmon activity recorded (Tremblay et al., 1994); however no fish overcame the obstacle.

During the summer of 1995, a third survey was conducted for eight weeks under severe drought condition. Only 10 salmon overcame the Déry gorges. It became evident that physical interventions were needed. The proposed regulating works are an application of the pre-dam technique in which all the residual discharge (4 to 30 m^3/s), partially regulated from the power plant, passes through the gorges. For a turbined flow of 22 m^3/s, the proposed concept will be efficient 45 % of the time during the upstream salmon run, while an additional turbine regulating up to 44 m^3/s would increase the efficiency to about 75-80 % of the migration period.



INADEQUATE CONFIGURATION

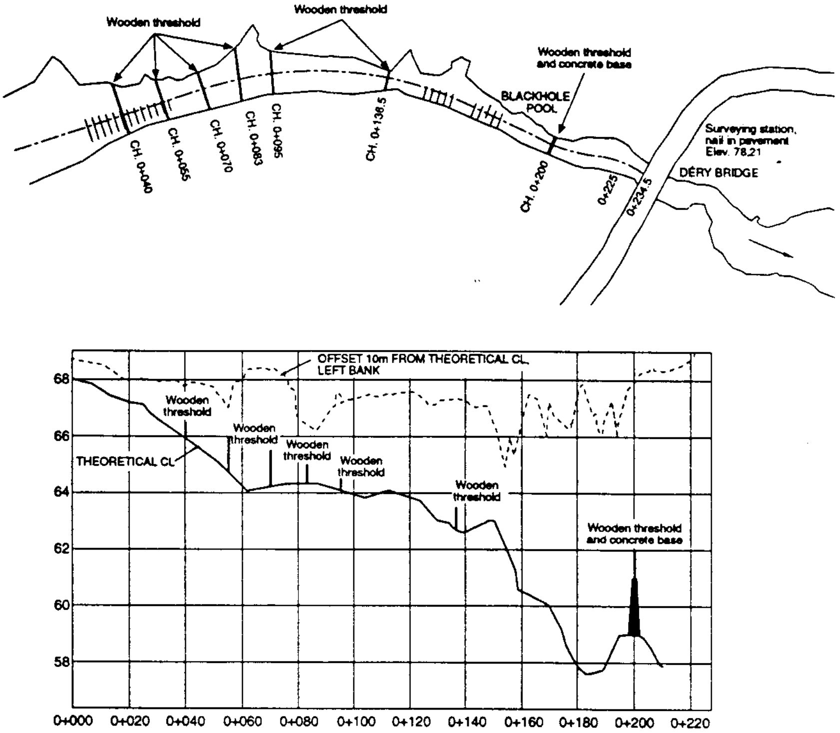
River slope is steep. Salmon has difficulty moving upstream when velocities are very high. Fish can be pushed back in the ascending eddy at the bottom of the rapids.

INADEQUATE CONFIGURATION

Short thresholds followed by an important difference in height are almost impassable for fish.

Figure 10. Diagram showing the hydrodynamic conditions in the Déry gorges, Jacques-Cartier River. *Écohydraulique 2000, juin 1996, Québec*

OPTION 1



OPTION 2

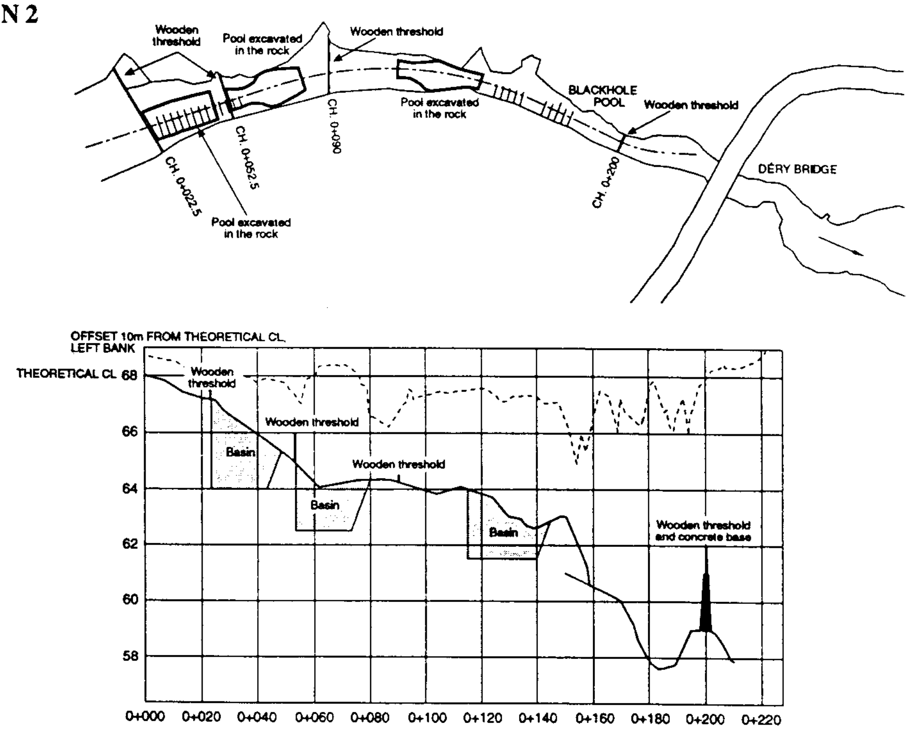


Figure 11. The two regulation options in the Dery gorges under natural flow conditions.

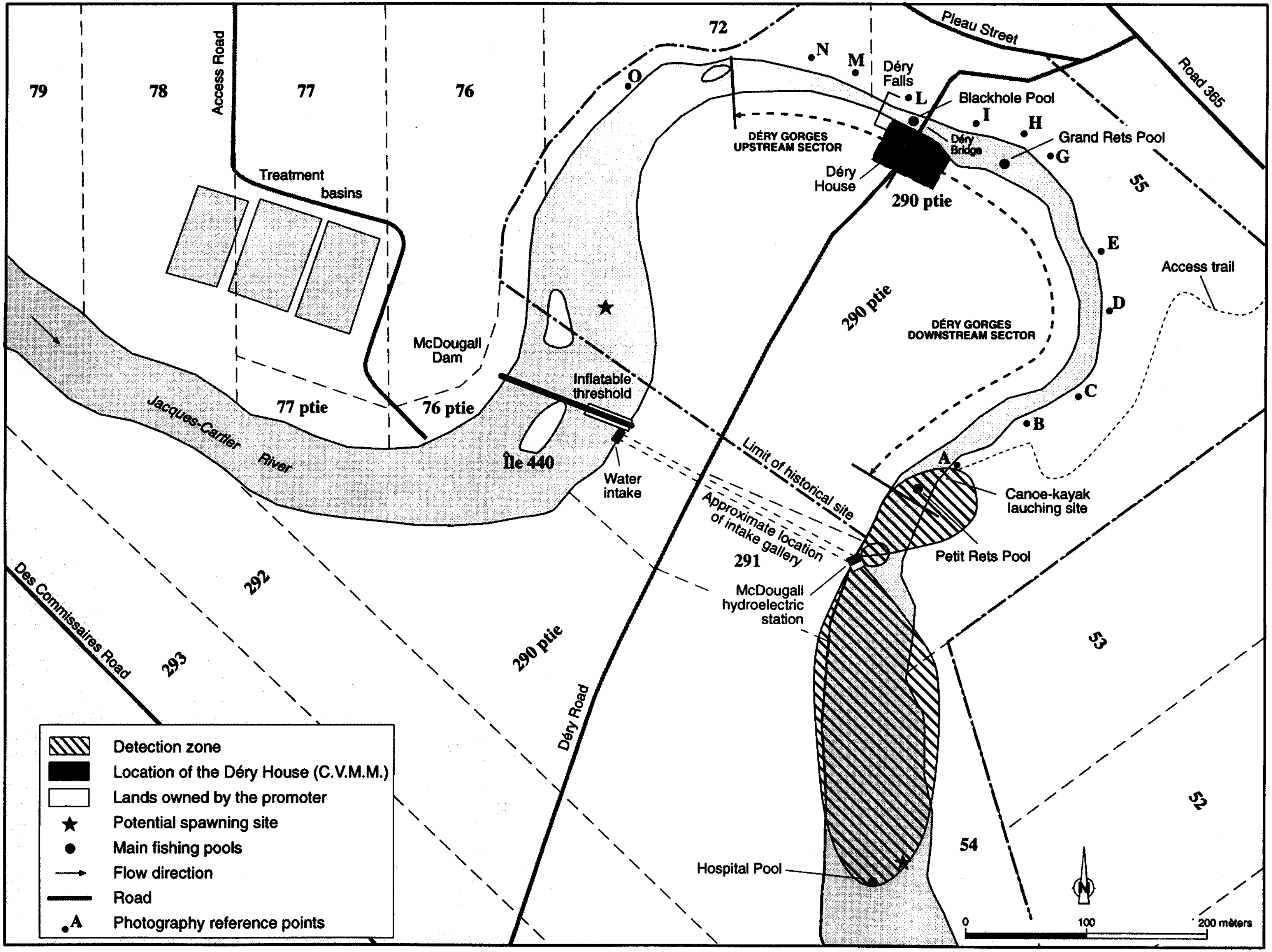


Figure 12. Location of the summer 1994 telemetry study area in the Déry gorges.

Hydrodynamics analysis indicate that seven adjustable impounding structures would be sufficient. These structures, shown in Figure 13, divide the height differential of 6-8 meters into 1 meter stages. At a low discharge rate (5 m^3/s), each stage is easily passable and the dissipated energy is less than 200 watts/m^3, acceptable salmon migration conditions. At moderate rate (15 m^3/s), passage conditions are still acceptable since the dissipated energy remains largely below 500 watts/m3, the recommended value in most reference works. Finally, when the flow will reach 30 m^3/s in the Déry gorges, salmon will still have acceptable migration conditions (around 500 watts/m^3) in all the pools, despite some local turbulences inducing a dissipated energy of 1000 watts/m^3.

The adjustable structures were designed to resist the action of ice and high waters; construction was completed during the 1995-1996 winter. Figures 14, 15 and 16 illustrate these structures and the effect on the water line. Pre-dams were selected over pool excavation in rock because the latter would have been irreversible; the ministry of Cultural Affairs preferred removable steel and wood structures rather than permanent modifications of the rock bed.

Fish migration parameters were defined by an advisory board composed of local interest groups , government representatives and the promoter. Migration performance must be monitored by competent personnel following approved protocols.

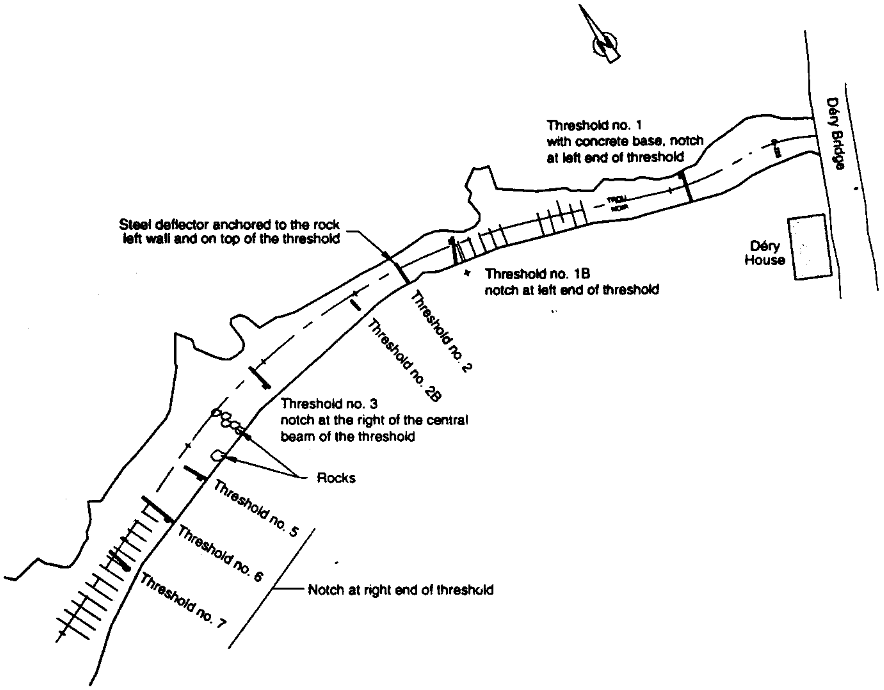
McDOUGALL AND BIRD DAMS

McDougall dam (rebuilt in 1994) and Bird dam, located about 12 kilometers from the river mouth, are the final obstacles to access to more than 95 km of prime salmon habitat. In each case, the private owner will build an acceptable fishway. Biological surveys were initiated in 1994 and will be more intensely conducted in 1996 and 1997. Monitoring measures have been planned to verify the efficiency of the fishways and propose required adjustments.

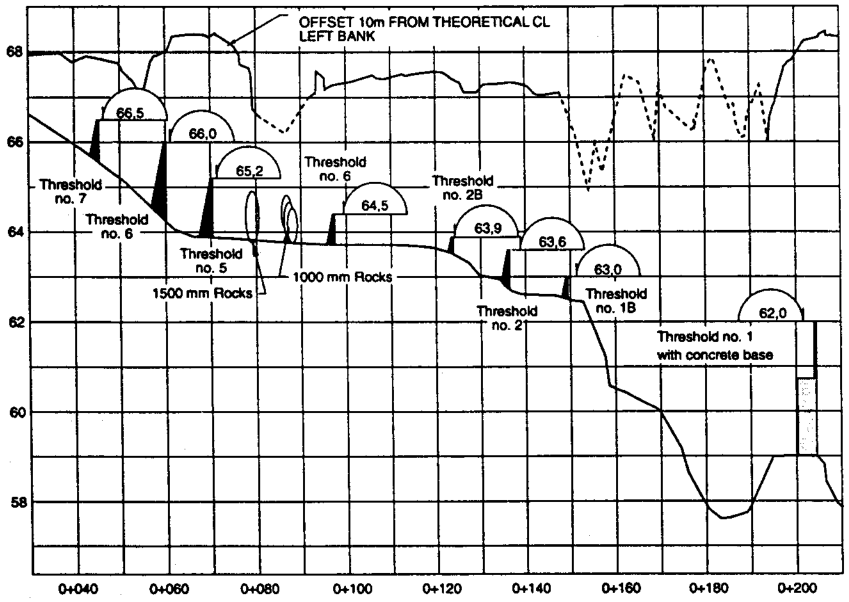
- Downstream migration : when studying the juvenile sea-run migration, two important parameters emerge: size and swimming capacity. In the Jacques-Cartier, parrs reach 12 to 18 centimeters on average, much longer than western America salmon parrs (2.5 to 3 cm only). Parrs can swim faster than 30 cm/s and thus avoid unfavourable hydrodynamic conditions , or death by pressing on diversion structures (Therrien and Beaulieu, 1996). **The cumulative impact** of parrs going through a series of generating stations and turbines may eventually affect the survival of a population. Accepted models (Larinier and Dartiguelongue, 1989) indicate that parr passage through the Bird, McDougall and Donnacona stations could result in a mortality rate of 35 to 55 % of total population, assuming that 100 % of juveniles pass through the water intakes. An assessment of the proportion of parrs actually going through the turbines should precise this mortality rate.

  The risk of mortality in the turbines depends upon the proportion of parrs drawn into the water intake. The literature indicates that this proportion is not necessarily proportional to intake inflow (Tremblay, 1995; Ruggles, 1992; Larinier, pers. com.). However, when 100 % of the flow goes through the station, there are no doubts left; the whole population is affected. At Donnacona, this will occur only 25 % of the time during parr migration. With the configuration of the site (Figure 17) and the hydrological conditions during this period, it can be assumed that a portion of the parr population will pass over the Donnacona dam, for 40 to 50 % of the time. At the Bird and McDougall stations in Pont-Rouge, flow in the intake will be about 45 m^3/s resulting in a considerable volume of water spilling over the dams, for more than 90 % (Figure 18).

  Since 1994, several studies have been conducted on the downstream migration of parrs in the Jacques-Cartier. Tremblay (1994) estimates a rate of 16 % of parrs going through the water intake and a mortality rate of 15 % at the old Bird station on the right bank. A literature review, combined with discussions with american, european and Canadian experts, including Paul Ruggles, summarized the advantages and



BIRD'EYE VIEW
1:1000



LONGITUDINAL SECTION
1:1000 Hor.
1:100 Ver.

Figure 13. Diagram of the pre-dam concept in the Déry gorges, Jacques-Cartier River.
Écohydraulique 2000, juin 1996, Québec

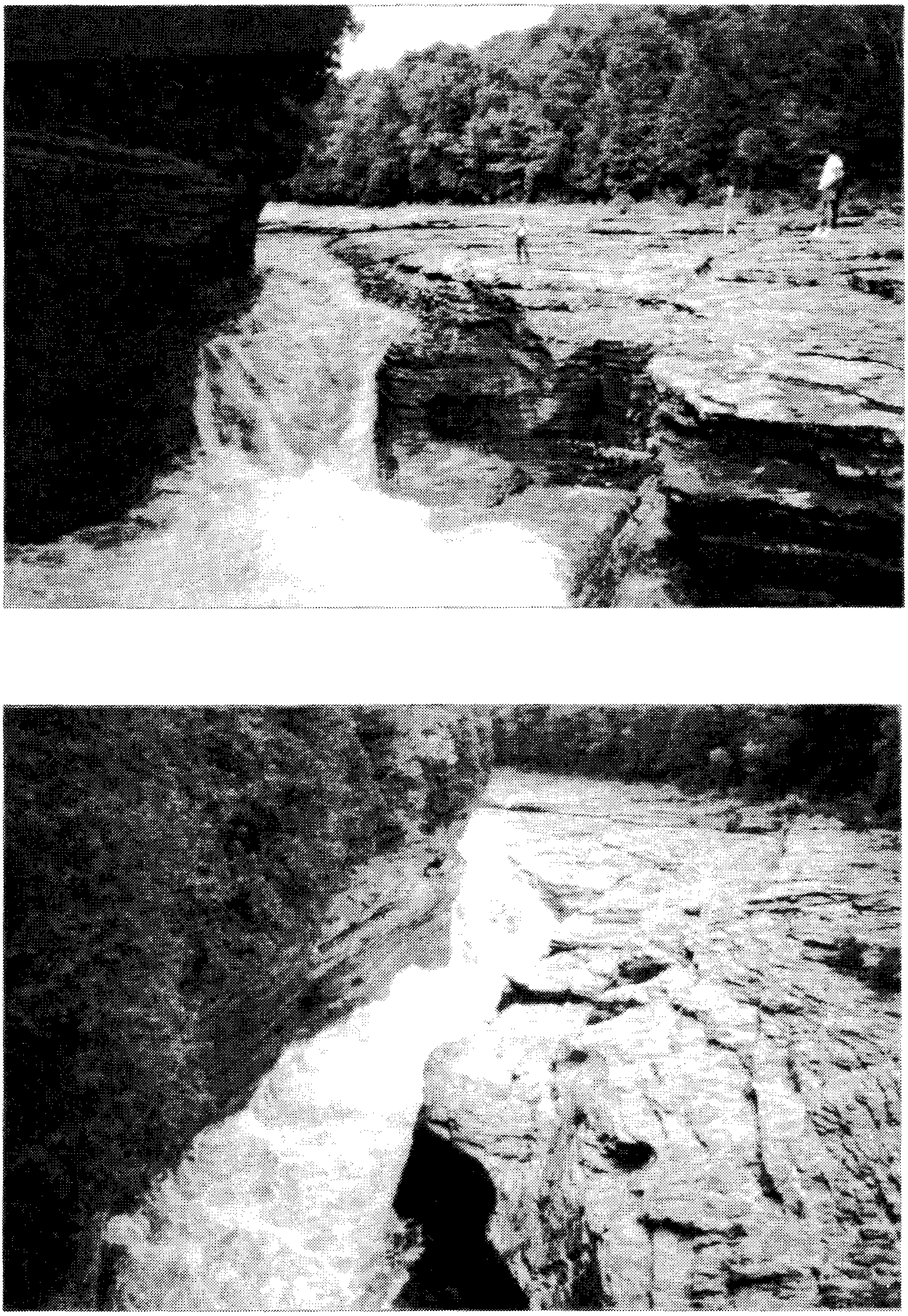


Figure 14. The Déry gorges on the Jacques-Cartier River, prior to regulation works.

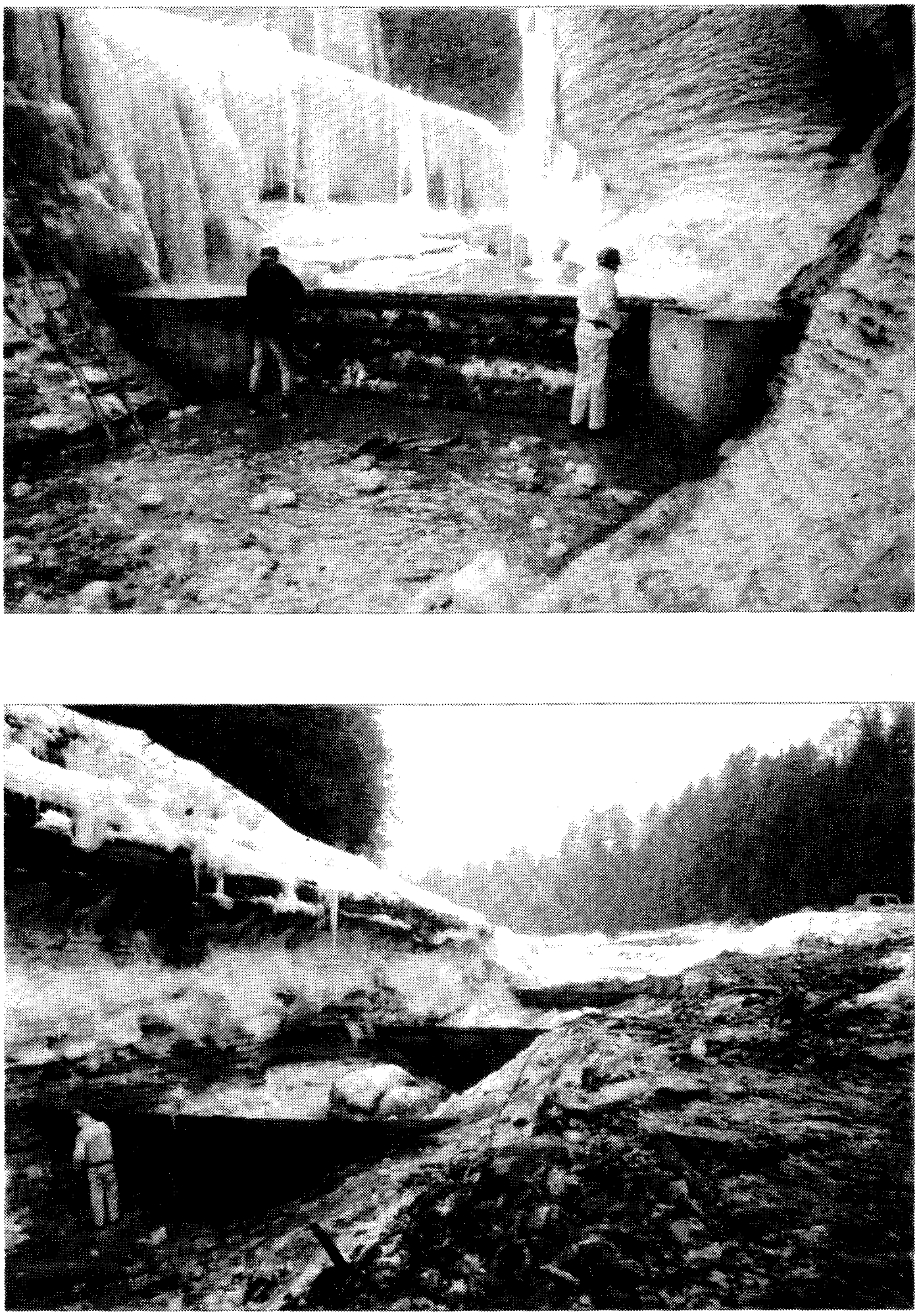
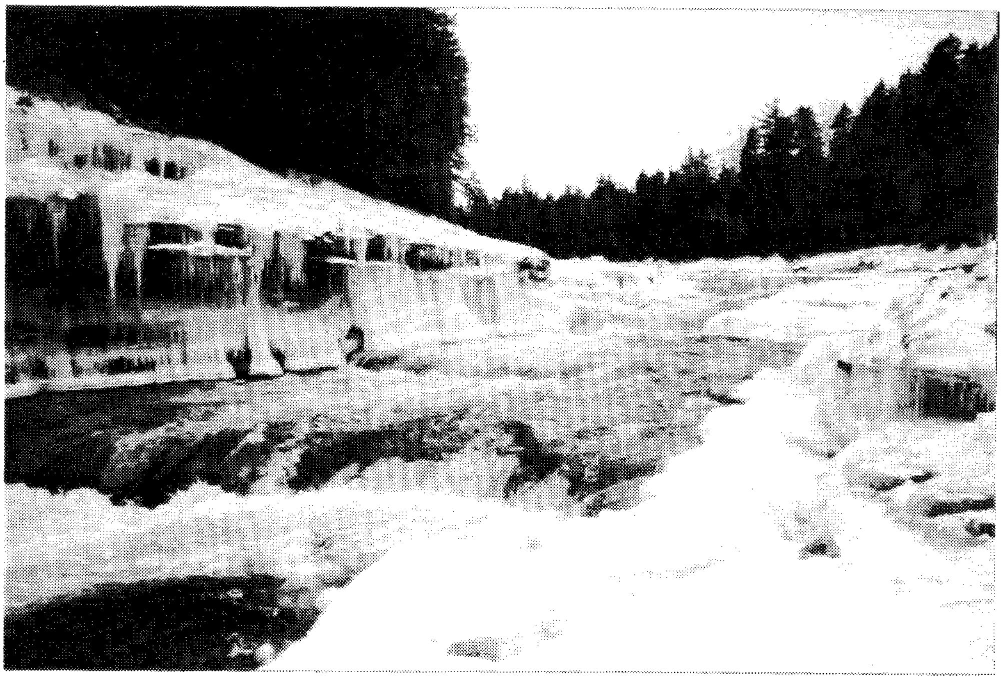


Figure 15. The Déry gorges during pre-dam construction.



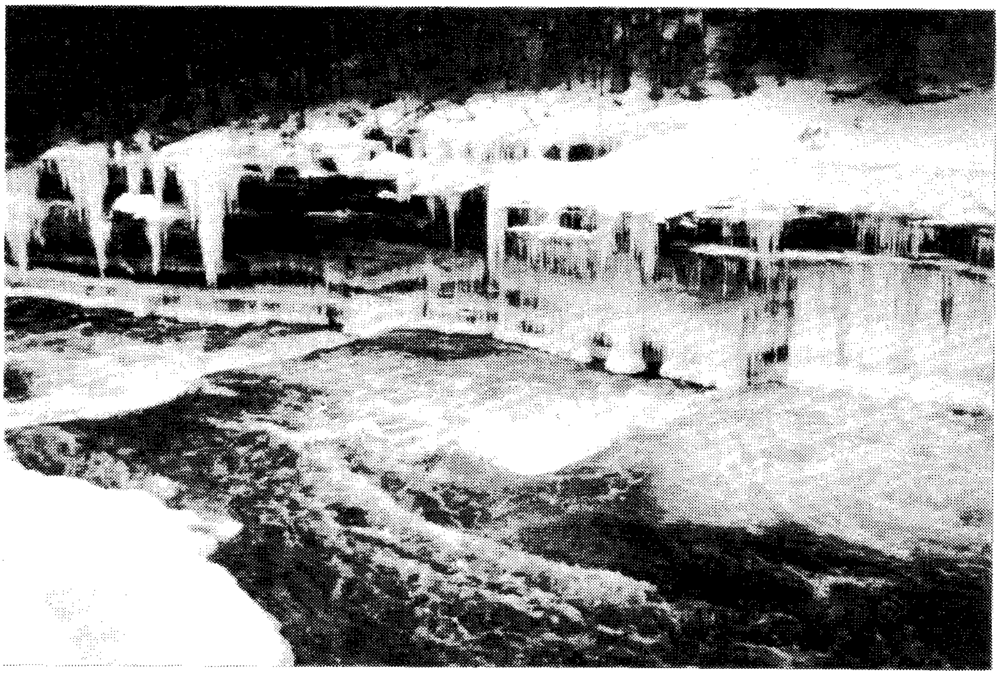


Figure 16. The Déry gorges after pre-dam construction.

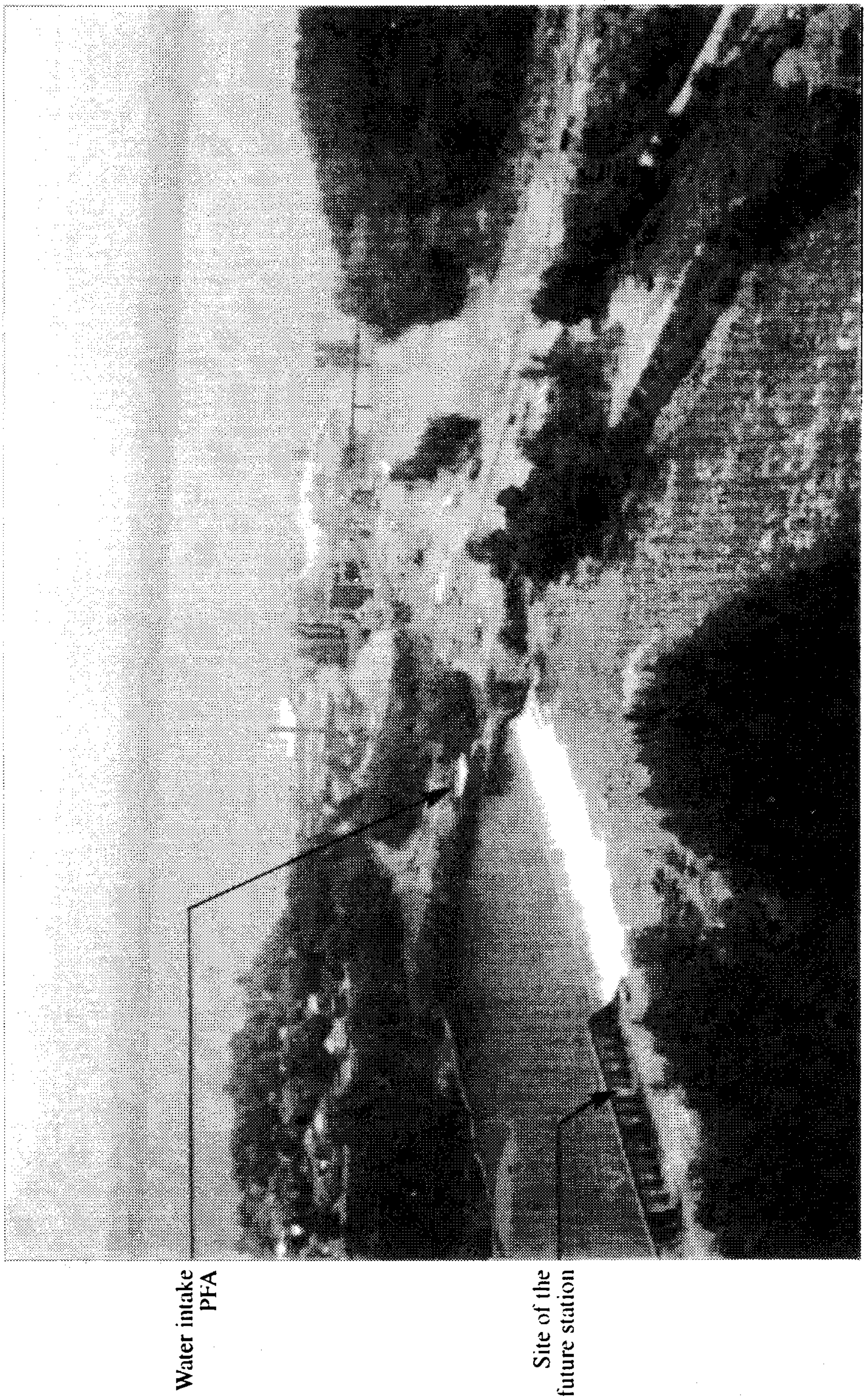


Figure 17. The Donnacona dam and site configuration.

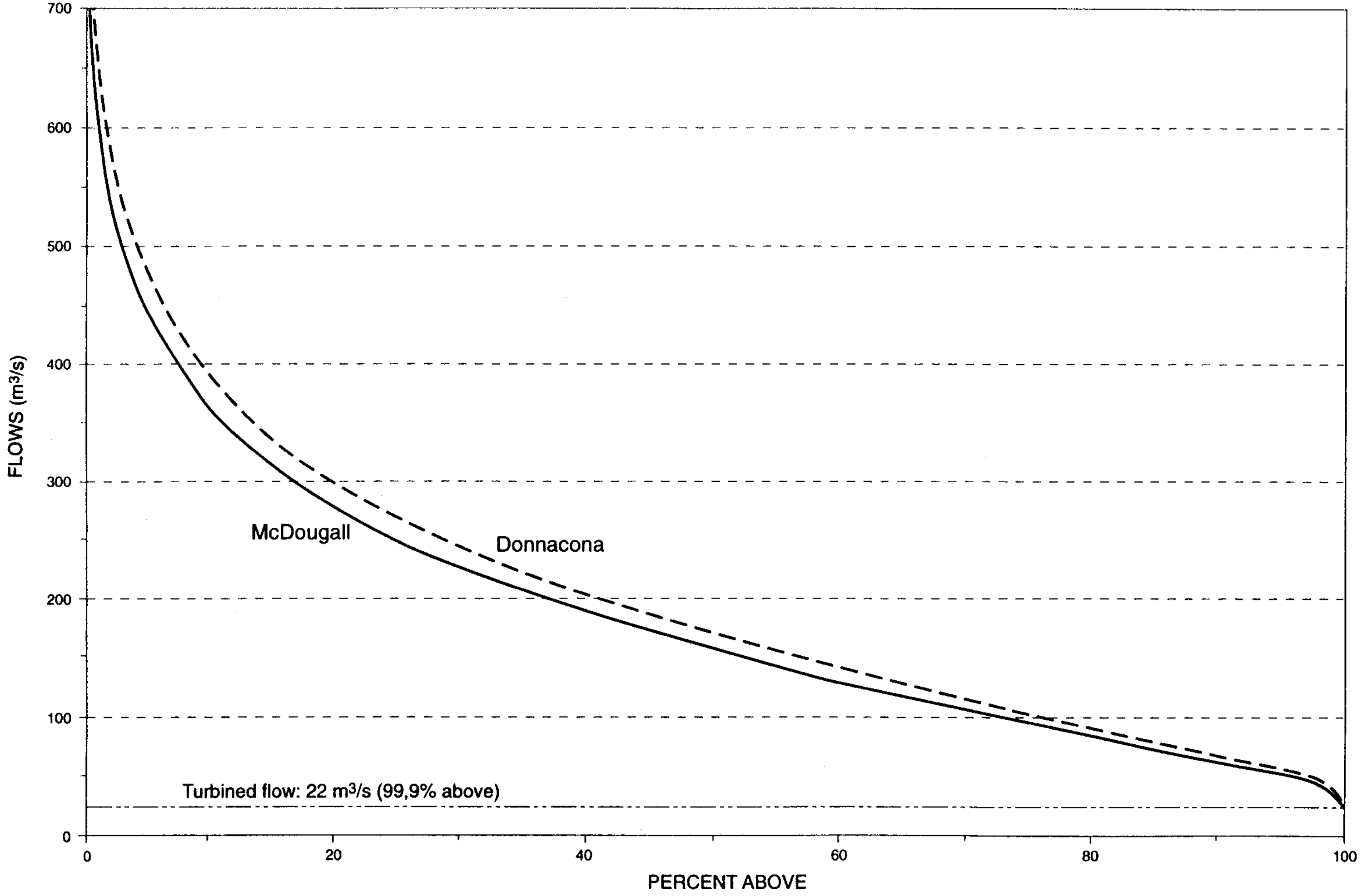


Figure 18. Hydrological conditions recorded during the downstream migration on the Jacques-Cartier, related to turbined flows at the Bird, McDougall and Donnacana stations (Appendice 3.2, Donnacana report; Appendice 8.1, McDougall report; Bird report).

disadvantages of about forty avoidance devices during downstream migration (Therrien and Beaulieu, 1996). The analysis concluded that there are no universal system adequate for every projects and that the more reliable techniques are the use of shutters, eicher grids, Modular Incline System (MIS), fixed grids, oı a combination of devices with cumulative effects. However, a poorly design protection device could be more fatal to fish than passage through the turbines.

Downstream migration protection seems easier to implement in Quebec than in the USA or Europe. In Québec, promoters have to protect only a few fish species, often only one. In addition, parr migration is limited to only a few weeks every year and the size of fish is relatively high. Repeated successes with Eicher grids (98 to 100 %) and successful laboratory experiments with MIS (100 % in 1993, 1994 and 1995) provide hopes for designing and implementing reliable devices for the Donnacona, McDougall and Bird stations. The first full-scale experiments will be completed at the Bird station in 1996 and at Donnacona in the Spring of 1997. Government authorities and the CRJC are expecting acceptable results.

- Minimum flow requirements : In Québec, most small hydropower station projects, such as Donnacona and Bird, do not require to maintain a minimum flow for fish habitat protection since the station is generally straddling the dam. However, a flow must be maintained for the proper functioning of fishways and avoidance devices. As for the McDougall site, the power station is located 600 m below the dam. A relevant minimum flow can be determined through a sound assessment of potential habitat losses.

Theoretical minimum flow values for habitat protection can be found in the literature. Normally, the minimum flow needs were based upon a dry-weather flow for 7 consecutive days and of two years recurrence. However, the protection of habitats' identified by MEF, the aesthetic constraints imposed by MAC, the conditions needed for quality sport fishing, and the migration through the Déry gorges made it a more complex problem. The recommended first minimum flows ranged from 15 to 25 m^3/s. However, a closer assessment revealed that salmon habitats were of poor quality at a high velocity flow over bedrock and that ice cover eliminates the aesthetics constraints during winter.

Because of these factors and of difficulties met by migrating salmon in this sector, government authorities agreed to reduce the minimum flow to about 5 to 10 m^3/s during the upstream run as long as salmon could more easily pass through the gorges. During winter, the minimum flow need is reduced to a symbolic value of 1 m^3/s.

These two examples are good indications of project specific flow requirements. A substantial portion (50 %) of the benefits generated by these projects will go to CRJC to cover wildlife development projects or part of the operation costs.

CONCLUSION

At a time when our society is questioning energy use and production, the commissioning of small power stations introduced a relatively uncomfortable situation for government authorities, salmon river managers, private promoters and environmental consultants. In fact, the expertise on fish counts versus small power stations is rather limited in Québec; caution was required to avoid costly errors (environmental, economic, and social). A literature review, combined with foreign expertise, readily highlighted troublesome situations :

- in northeastern USA, more than twenty devices have been proposed by government authorities to improve fish migration; none satisfied the expectations;

- fish mortality rates in turbines have been regularly overestimated (Therrien and Beaulieu, 1996) : results were biased by inappropriate experimental protocols;

- most fish protection devices used in the USA were never adequately monitored (Therrien and Beaulieu, 1996) : their true performances are not really known;
- in France, Michel Larinier (an expert in the field) reported several cases of bad functioning, in both fishways and intake avoidance devices, caused by poor designs or lack of experience.

On the Columbia River, intake avoidance devices have cost around three billion dollars over thirty years and results are still unsatisfactory (Paul Ruggles, pers. comm.).

In Québec, the situation is less dramatic, but more than half of the fishways constructed between 1975 and 1992 had deficiencies which required expensive corrections (Beaulieu, 1993). In most cases, some information needed for an optimal design were overviewed. One or two years are needed to gather all the pertinent information needed.

Currently, real efforts are made in Québec by government authorities (MEF, DFO), APPHA, certain private promoters and their environmental consultants to assess all the issues associated with small hydropower stations. For example, a synthesis of advantages and disadvantages of most intake avoidance devices was recently produced to support judicious choices. Another document addresses the issue of minimum flow requirements and establishes guidelines. Fish migration monitoring methods are improving; performances required from private promoters are better defined. Decision-making is easier than it was only one year ago, a clear witness to rapid progress, although more knowledge is still needed.

The Jacques-Cartier River venture is showing that fish population and small power stations can coexist in a context of sustained development. Despite the fact that construction is still underway and that results are still preliminary, the projects show great promises.

The Jacques-Cartier experiment has showed that :

- close coordination is essential between government authorities, resource managers, private promoters, engineers, and environmental consultants : decisions must be acceptable to all;
- one or more small power stations on a river may represent a unique occasion to enhance habitats and natural resources. Through mitigation, compensation and enhancement measures; it is possible to generate significant positive impacts of various nature :
    - e.g. environmental :
        - overcoming obstacles increases accessibility and production capacity;
    - e.g. socio-economic :
        - higher fish production translates into better fishing and greater benefits;
        - seasonal and permanent jobs;
        - possible financial partnership for local groups;
        - monetary compensation to support the development of a species;
- small power stations should not be implemented if their environmental and socio-economic profitability is not demonstrated at all levels;
- undeveloped hydroelectric potentials are numerous in Québec. Only projects well integrated in the environment should be authorized;
- satisfactory performances are expected from all devices implemented (fishways or intake avoidance systems) and should be imposed to all private promoters;

- structure designs must be based upon solid database. For example, river hydrology and fish behaviour must be thoroughly analysed over one or two years of feasibility studies; it is better to invest judiciously at this stage rather than spend large amounts on corrections and adjustments;

- each small power station project must become a source of information on fishways, fish monitoring, fish behaviour on resource enhancement, useful for future projects.

The analysis justifies or rejects certain structures and devices; it gathers all the pertinent information; it assesses the viability of the project; and finally proposes a final concept. Control and correction measures can be introduced, if needed. Fishway design and construction last on average one year, two years for more complex projects.

Fisheries resources and hydroelectricity production can coexist on any Quebec rivers. The example of the Jacques-Cartier constitutes the minimum tolerance threshold for sustainable development projects. Several projects have greater potentials than the Jacques-Cartier River. When will we witness the development of a salmon river totally financed by private electricity project ?

REFERENCES

Beaulieu, C. 1993. Les passes migratoires pour le saumon et la ouananiche au Québec. Dossier synthèse préparé pour le ministère du Loisir, de la Chasse et de la Pêche et la Fondation de la faune du Québec par le Groupe-conseil Solivar, 23 p.

Beaulieu, C. and F. Gauthier 1993. Analyse d'options visant à rendre la gorge Déry franchissable par le saumon. Rivière Jacques-Cartier – Préliminaire. Document présenté à la Corporation de la restauration de la Jacques-Cartier par le Groupe Environnement Shooner inc. 14 p. + annexes.

Beaulieu, C. and M. Frenette 1984. Principes d'aménagements de la passe migratoire de Donnacona sur la rivière Jacques Cartier. Les carnets de zoologie. Société zoologique de Québec, vol 44, no 1, p. 10-11.

Beaurivage, M. 1987. Étude du comportement migratoire du saumon atlantique par télémétrie aux Rapides Déry et au barrage Bird sur la rivière Jacques-Cartier. Corporation de la Restauration de la Jacques-Cartier, 84 p.

Boudreault, J., Therrien, J., Bourgeois, G., Beaulieu, C. and R. Bouchard. 1993. Évaluation environnementale, Projet d'aménagement hydroélectrique au barrage Donnaconna, rivière Jacques-Cartier. Rapport réalisé par Groupe Environnement Shooner inc. pour 9006-8776 Québec inc. 70 p. + annexes.

Carter, D. 1992. Conditions hydrauliques pouvant faciliter la montaison naturelle du saumon dans la gorge Déry. Ministère du Loisir, de la Chasse et de la Pêche. 12 p.

Corporation de restauration de la Jacques-Cartier. 1994. Zec de la rivière Jacques-Cartier : bilan de la pêche 1994. 15 p. et annexes.

Gauthier. F., Beaulieu, C. and R. Bouchard. 1993. Évaluation environnementale, Projet d'aménagement hydroélectrique McDougall, rivière Jacques-Cartier. Rapport réalisé par Groupe Environnement Shooner inc. et présenté à Cascades Énergie inc., 72 p. + annexes.

Germain, H. Historique et inventaire du potentiel culturel et patrimonial du bassin hydrographique de la Jacques Cartier. Comité de restauration de la rivière Jacques Cartier. Duchesnay, décembre 1984, 214 pages.

Gingras, S. La pêche au Québec, un siècle d'histoire...

Larinier, M. 1983. Guide pour la conception des dispositifs de franchissement des barrages pour les poissons migrateurs. Bulletin français de pisciculture. 29 p. + annexes

Larinier, M. and J. Dartiguelongue. 1989. La circulation des poissons migrateurs: le transit à travers les turbines des installations hydroélectriques. Bulletin Fr. Pêche Piscic. 312-313, 94 p.

Larinier, M. and S. Boyer-Bernard. 1991. "Dévalaison des smolts et efficacité d'un exutoire de dévalaison à l'usine hydroélectrique d'Halsou sur la Nive", *Bull. Fr. Pêche Piscic.* 321: 72-92.

Ministère de l'Environnement et de la Faune. 1994a. *Projet de politique sur la faune.* Version de mai 1994. 48 p. + 1 annexe.

Ministère de l'Environnement et de la Faune. 1994b. *Cadre d'analyse des projets de petites centrales hydroélectriques.* Document de régie interne.

Ruggles, C.P. 1992. *What's new in downtream fish passage?* Paper presented in the Fourth International Atlantic Salmon Symposium. St-Andrews, New Brunswick, Canada, June 14-17, 22 p.

Ruggles, C.P. 1993. "Effect of stress on Turbine Fish Passage Mortality Estimates", *Proceedings of the Workshop on fish passage at hydroelectric developments : March 26-28, 1991, St.John's, Newfoundland.* Canadian Technical Report of Fisheries and Aquatic Sciences. n° 1905. p. 39-57.

Ruggles, C.P. and N.M. Collins. 1980. *Mortalité des poissons en fonction des propriétés hydrauliques des turbines.* Association Canadienne de l'électricité. Section recherche et développement, Montréal, Québec; projet G144. Monenco, Halifax, N.-É., 108 p. + annexes.

Ruggles, C.P., Palmeter, T.H. and K.D. Stokesbury. 1990. *A critical examination of turbine passage mortality estimates.* Report prepared by Monenco Maritimes Limited for the Canadian Electrical

Taft, E.P. 1990. *Fish Protection Systems for Hydro Plants.* Rapport préparé par Stone & Webster Engineering Corporation.

Taft, E.P. 1993. "An Update of Methods for Preventing Turbine" *Proceedings of the Workshop on fish passage at hydroelectric developments : March 26-28, 1991, St.John's, Newfoundland.* Canadian Technical Report of Fisheries and Aquatic Sciences. n° 1905. p. 70-86.

Taft, E.P., Amaral, S.V., Winchell, F.C. and C.W. Sullivan. 1993. "Biological Evaluation of a New Modular Fish Diversion Screen", *Proceeding of a Symposium on Fish Passage Policy and Technology. Portland, Oregon, U.S.A. Bioeng. Sec. Am. Fish. Soc.* p. 177-188.

Tremblay, G., Gauthier, F. and C. Beaulieu. 1994. Étude par télémétrie du franchissement de la gorge Déry par le saumon atlantique. Rapport présenté à RSP Hydro par le Groupe Environnement Shooner, 20 p. et annexes.

Tremblay, G., Therrien, J., Asselin, S. and C. Beaulieu. 1994. *Problématique de l'entraînement des saumonneaux et des saumons noirs dans la centrale Mitis 1 et étude des systèmes de déviation applicables.* Rapport du Groupe Environnement Shooner au service Milieu naturel de la Vice-présidence Environnement d'Hydro-Québec. 58 p. et 4 annexes.

Therrien, J and C. Beaulieu. 1996. Guide d'évaluation de la problématique de la dévalaison des poissons en relation avec les petites centrales hydroélectriques. Rapport du Groupe-conseil Génivar présenté au ministère des Pêches et des Océans et à l'Association des producteurs privés d'hydroélectricité du Québec. 118 p.

UPSTREAM PASSAGE OF MIGRATORY FISH THROUGH NAVIGATION LOCKS

M. ZYLBERBLAT
Compagnie Nationale du Rhône
2 rue andré Bonin, 69004-Lyon
France

J.Y. MENELLA
Association Migrateurs Rhône Méditerranée
Espace La Beauvalle, rue Gandhi, 13084, Aix-en-Provence
France

ABSTRACT

An original technical solution optimizing existing structures was selected to facilitate the upstream run of fish at the Beaucaire hydroelectric station on the Rhône River; the use of the navigation locks has been recommanded to facilitate fish passage.

The Beaucaire lock has become a reference site to elaborate techniques which require adjustments to navigation locks to be used as fishways.

It is essential to induce upstream attraction and to verify its efficiently on shads from the lower reach into the locks and from the locks to the upper reach.

Experimental conditions vary considerably depending on the number and the variability of external factors; field work is difficult and complex, limiting the number of trials acceptable.

Sampling by nets inside the lock indicated the attraction at a flow-rate of $60 m^3/s$ was relatively efficient.

Shads exit the lock when the upstream gate is completely lowered, without actually selecting an attraction flow value; this technique was prefered to a partial lowering because it can be reproduced at the two locks located at some distance upstream.

Fish migration could eventually be controlled by accoustic monitoring (echosonding).

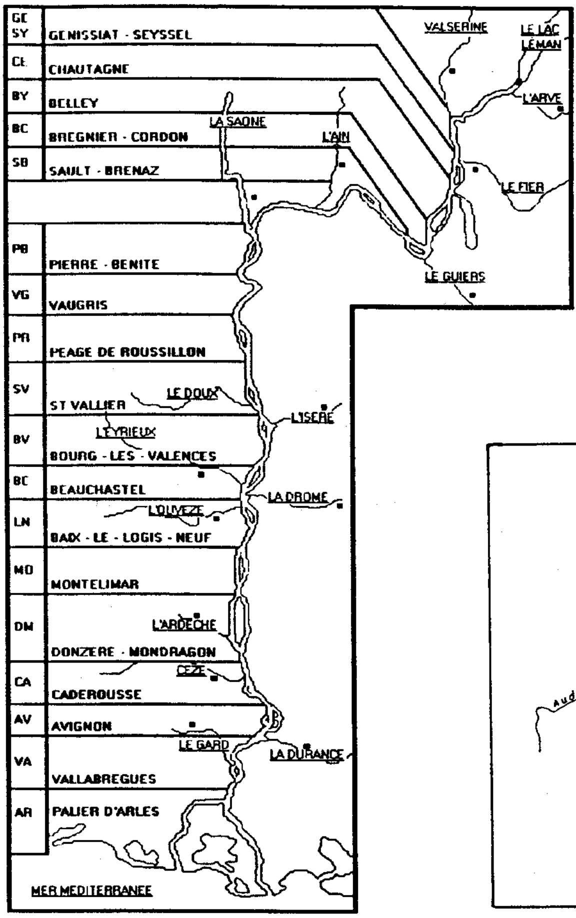
The unconventional use of navigation locks requires some adjustments to the equipment; the routine operation in a fishway mode also implies the automation of the device.

KEY-WORDS: Rhône / Migratory fish / Shad / Obstacle / Lock / Attraction flow / Echosounding / Management rules.

INTRODUCTION

The Compagnie Nationale du Rhône (CNR) proceeded with the regulation for multiple usage of the Rhône River between 1938 and 1986 (Figure 1). The regulation had three objectives: energy production, navigation and enhancement of farmlands. La CNR currently exploits and maintains the existing structures.

On most of the 18 falls, the regulation followed a similar model (Figure 2); construction of a dam with a spillway, construction of a generating station and a ship lock on the diversion canal.



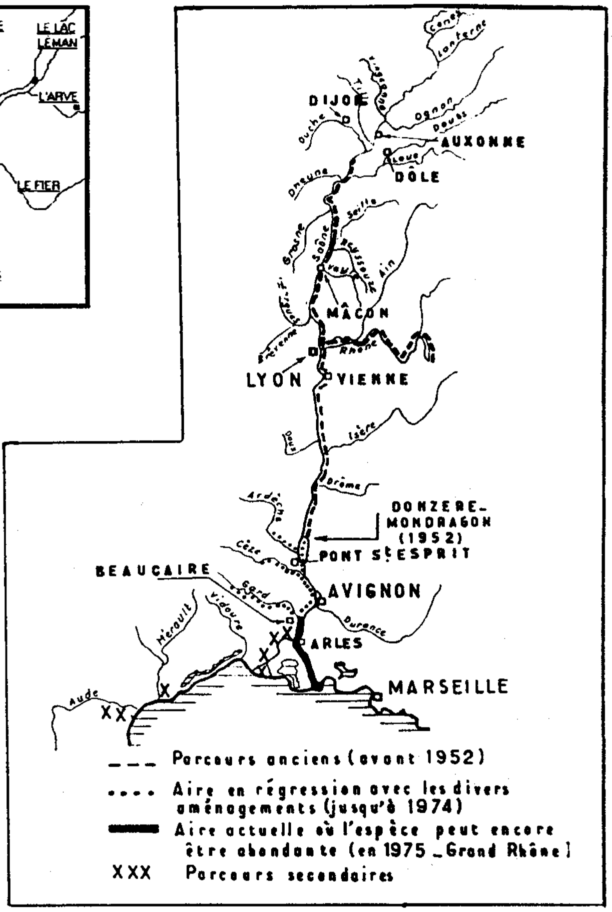
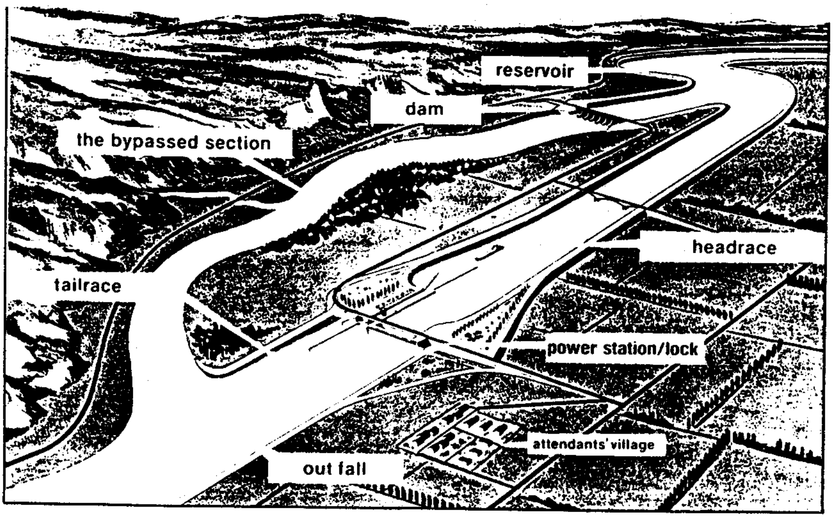


Figure 1:Rhône regulation works - Location of the waterfalls

Figure 3:Summary of shad regression in the Rhône River (Rameye et al., 1976)

Figure 2: Typical regulatio works on the Rhône River



Increased public environnemental awareness and concerns, in particular since 1976, has gradually justified the integration of this aspect in the regulation works of waterfalls: increased reserved flow, maintenance of alternative channels, construction of fishways ...

However, even with these conservation measures, regulation works, in particular those located on the lower third of the river, remain an obstacle to migratory fish movements. Despite existing specific devices, the dam and associated works at Vallabrègues (Beaucaire power station) are the first obstacle to fish migration on the Rhône River. Different migratory species have specific biological characteristics which affect their capacity to surmont an obstacle; for exempple, European eels (*Anguilla anguilla*) manage to get over (at least in part) the Beaucaire site while shads (*Alosa alosa, Alosa ficta rhodanensis*) are totally refrained by the obstacle.

Therefore, the distribution area of shads has gradually decreased with each new regulation works (Figure 3).Currently, the dominant species is *Alosa ficta rhodanensis* (Rameye *et al.*, 1976; Douchement, 1981). In the Beaucaire sector, fischeries statistics are not sufficient to estimate with any accuracy the catch per unit of effort (Délégation de bassin RMC, 1989); inquiries conducted between 1988 and 1993 suggest that catches of shad, all fisheries included, range between 5 and 10 tons per year - 7 tons in 1989 -(Comité de gestion des poissons migrateurs du bassin RMC, 1995).

Made aware of this situation, the ministry of the Environment implanted a shad rehabilitation plan, the first on the French mediterranean basin (Délégation de bassin RMC, 1991). A first phase will cover the period up to 1998.The initial goal is to reintroduce the species on known spawning grounds on the tributaries of the west shore: the Gard, Cèze and Ardèche rivers (Gallois, 1947). This corresponds to an homogeneous area with high historical potentials. The plan requires free migration on the main river and the clearing of three regulation structures on the Rhône: Beaucaire, Avignon and Caderousse. Several solutions were examinated and evaluated in search for the optimal tecnique to facilitate obstacle clearing by shad; finally the use of the navigation locks was selected (Délégation de bassin RMC, 1990), advantages outscoring drawbacks under the current conditions (Table 1).

Table 1: Advantages and drawbacks of clearing an obstacle by way of the navigation lock

Advantages	Drawbacks
relatively low cost	discontinuous use
probably little selective	requires a human presence
	specific usage outside normal ship locking

Several research projects have been conducted since 1989 at the Beaucaire navigation lock, which has become a reference site for the elaboration of this technique and of the new management rules required.

DESCRIPTION OF THE BEAUCAIRE SITE

The mean reference flow of the Rhône at Beaucaire is 1696 m^3/s, calculated over a period of 74 years (1920-1993). Low-water flow is normally reached at the end of summer and corresponds to a sustained flow (conventional low-water=570m^3/s).The hydrological regimen of the Rhône is the resultant of the following river influences: mountains rivers (high-Rhône, Isère), oceanic (Saône) and the Cévennes (Ardèche and Gard rivers).

The minimum turbine flow is approximatively 2400 m^3/s; at lower values, the generating turbines take water from the diverted flow, leaving a reserved flow of 10 or 20 m^3/s in the original channel (vieux- Rhône); in its downstream portion, this channel has a threshold which maintains a minimum depth but refrains the free circulation fish.During a flood, the flow portion exceeding 2400m^3/s passes through this channel when the dam spillway is open.

Up to this value, except when a rise in the water level of the Gard river floods the threshold in the vieux-Rhône, shads are attracted in the diversion canal where the navigation lock is set; A fish ladder with stoppers exists between the tail race and the original river bed, but it is not very efficient for shads (Baril, 1988).

The ship lock, 195 meters long and 12 meters wide, is located on the east side of the power station from which it is seperated by a dividing wall. The lock chamber has two distinct mechanisms: a filling-emptying mechanism operated by two sluices of 60 m^3/s nominal flow each; and a fish-gate (flow: 5 m^3/s) used for migrating shads.The upstream gate is of the lowering-type and can be operated while in charge; the downstream gates, hooked, have two lock paddles delevering a total flow of 5 m^3/s.

The total drop varies according to upstream and downstream levels and can reach a maximum of 16 m.

The generating station is operated by sluicing waters whenever the Rhône discharges fall below the minimum turbine flow; this may induce a level variation of more than one meter in a few hours in the downstream reach.The upstream level remains relatively stable, at more or less 10 cm.

MATERIALS AND METHODS

During the shad upstream run, approximately between March 15 and July 1, various tests are conducted; each test answers a different and complementary objective.

The particular configuration of the Beaucaire lock allows to experiment with, in manual operation, different attraction flows both at the entrance and the exit of the lock (Figure 4).

Three possibilities exist to attract shads inside the lock chamber (Délégation de bassin RMC, 1989):

. 0 m^3/s: the downstream gates are open, all the filling sluices are closed,
. 5 m^3/s: the downstream gates are open, the fish-gate is operating,
. 60 m^3/s: the downstream gates are open, one of the filling sluices is open; this is an exceptional operation not complying with the conventional use of the lock.

Three modes are possible to attract shads toward the exit (Larinier, 1991; Jolimaitre, 1992, Ruchon, 1993; Bechu *et al.*, 1994; Gendre, 1995):

. upstream gate partially lowered:

Normally, the discharge above the gate is slightly greater than the calibration of the downstream doors. After the lock chamber is filled, the level stabilizes and then decreases because the downstream lock paddles are in the open position; the upstream gate is them partially lowered so that a layer of 20 to 30 cm of water creates a surface stream flow, prompting shads to exit; the initial hydraulic conditions are gradually modified by the filling of the lock chamber. Some operations are repeated to verify the efficiency of shad passage.

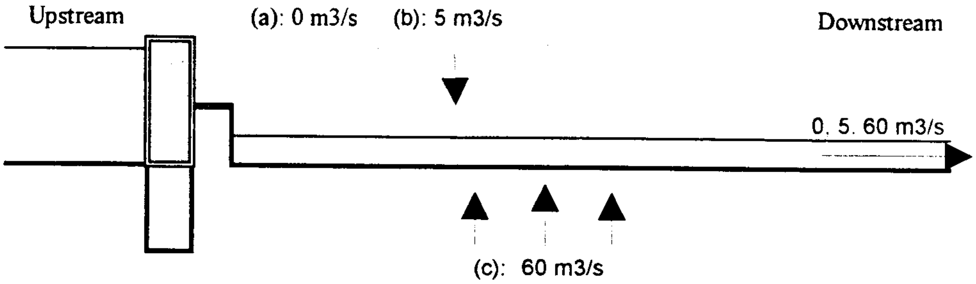
. upstream gate totally lowered:

The opening of an emptying sluice releases a flow of 60 m^3/s in case of an incident on the sluice; it is also possible to close the upstream gate which can be operated while in charge.It is also possible to test this position with a zero flow and a 5m^3/s flow, depending if the lock paddles are closed or open.

From 1991 to 1993, a tail race flow of about 5m^3/s located at the base of the upstream gate feeds in the lock chamber.

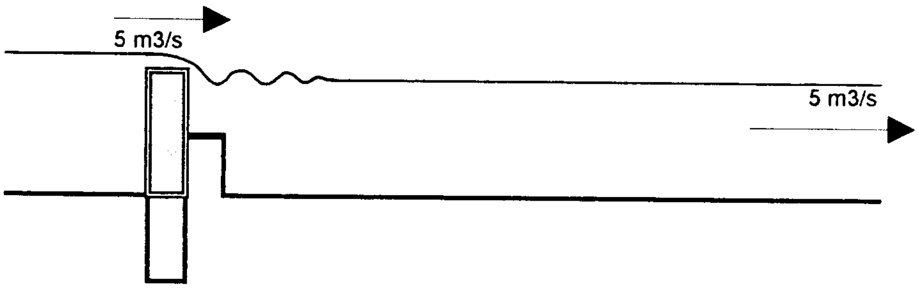
In total, six campaigns have been conducted between 1989 and 1995 (Table 2).The duration of each is a function of the objectives, the hydraulic conditions, and the means available. Experimental conditions are

Shad entrance: attraction phase to the lock chamber

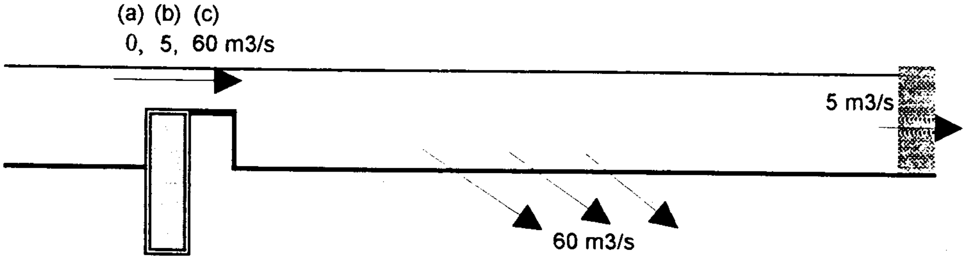


(a) No attraction: no sluice open
(b) Opening of doors on the downstream gate
(c) Opening of an emptying sluice (experimental operation)

Shad exit: attraction phase to the upstream reach



Partial lowering of the upstream gate (experimental operation)



(a) No attraction: no sluice open
(b) Opening of doors on the downstream gate
(c) Opening of an emptying sluice (experimental operation)

Total lowering of the upstream gate

Figure 4: Operation diagram of the Beaucaire lock (Rhône) in the fishway mode (from Larinier, 1991)

Table 2: Description of experimental conditions during the studies at the Beaucaire lock(Rhône)

Year	Observation period	Rhône discharge (m3/s)	Type of experiments	Number of sluicing waters	Entrance flow(m3/s)	Exit flow (m3/s)
1989	May 10 to 12, June 1	1500-2000	Net setting in the lock	2	0	_
				3	5	_
				2	60	_
1989	May 10 to 12, June 1	1500-2000	Total lowering, nets upstream of the lock	10	5	
				1	60	
1991	May 14 to 22	700-910	Partial lowering	21	5*	5*
				3	60*	5*
1992	April 22 to May 20	700-2100	Partial lowering	3	0*	5*
				23	5*	5*
				23	60*	5*
1993	May 3 to 27	1100-1900	Partial lowering	10	60	5
1994	May 5 to June 23	1100-3200	Single partial lowering	54	60	5
			Double partial lowering	7	60	5
			Total lowering, nets upstream of the lock	3	60	5
1995	April 7 to July 9	1400-5000	Single partial lowering	55	60	5
			Double partial lowering	2	60	5
			Total lowering	2	60	0
			Total lowering	2	60	5

* not including an escape flow of about 5m3/s

rarely repeatable in the course of a compaign because of variations in uncontrolled external parameters (flow, temperature, structure behaviour, etc.) and a strict protocol is hardly applicable because of priority to shopping; water sluicing for fish occurs when the lock is available for one hour, as 20 minutes are required for the entrance in the lock chamber, 15 minutes for the filling of the chamber, and 15 to 20 minutes for the exit.

Factors susceptible to influence measurements are systematically recorded: total discharge and levels in the Rhône, turbinated flow, daily water temperature, limpidity, duration of attracting during the entrance and exit phases.

Controls are conducted:

. in the lock chamber with several fixed nets with mesh sizes of 40, 45 and 60 mm,

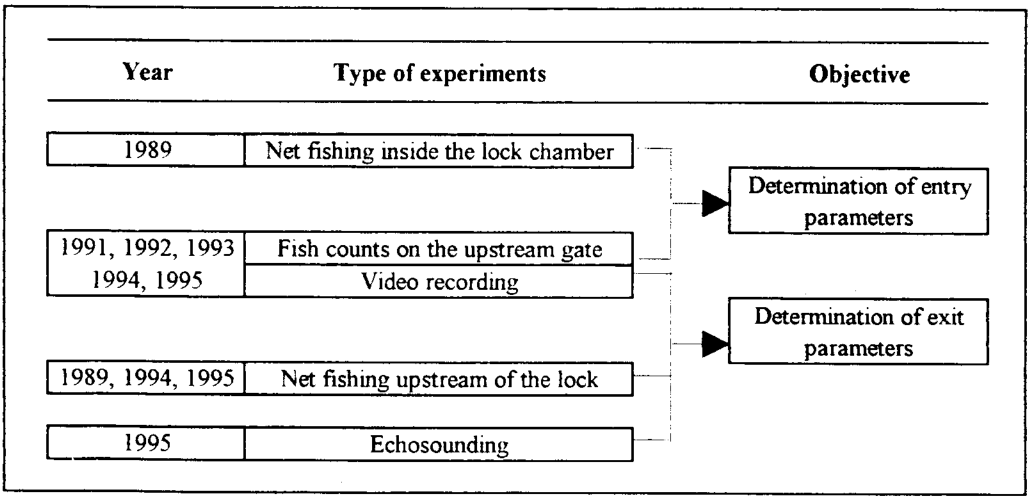
. at the exit:

-with fixed nets (mesh size 40 and 60 mm) positionned upstream,
-by visual counts on the upstream gate partially opened, by observers and by video recording control covering half the width of the lock,
-by accoustic detection using monofrequency echosounders EY 200 (Simrad) of 70 and 200 khz equipped with three transducers (7, 11 and 22 degrees) providing vertical and horizontal sampling of the section of the lock, immediately above the lowering gate.

RESULTS

Since a type of experimentation could correspond to several objectives and that an objective could be adressed by several tests, results have been examinated by common groups of homogeneous objectives (Figure 5).

Figure 5: Efficiency of shad passage in the Beaucaire lock (Rhône)
Relations between the objectives and the type of experiments



Results relating to the general conditions of shad migration and to shad distribution between the tail race and the old river channel (vieux-Rhône) are not addressed in this paper. However these results indicate that the total number of shads counted per water sluicing during partial lowerings may vary considerably (Table 3) and that daily migration peaks may represent as much as 34% of the total number of shads recorded during a field campaign (Jolimaitre, 1992).

Table 3: Summary of the number of shads recorded during partial lowerings of the upstream gate at the Beaucaire lock (Rhône).

Year	Rhône discharge (m3/s)	Number of shads	Number of sluicing	Mean of shads per sluicing
1991	700-900	3324	24	138,5
1992	700-2100	10544	49	215,1
1993	1100-1900	1912	10	191,2
1994	1100-3200	40990	40	102,2
1995	1400-5000	308	55	5,6

Analyses of variance have highlighted the influence of the time of day of the sluicing (Larinier, 1991) (Figure 6): passages between 1 AM and 5 PM are significantly greater than between 5 AM and 11 AM and between 5 PM and 10 PM. These results were not confirmed in 1992 (Jolimaitre, 1992), but the duration range in this case was two hours.

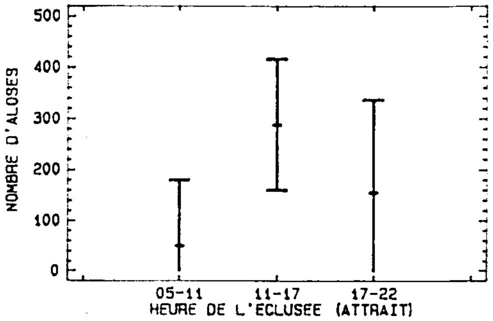


Figure 6: Influence of the time of day of the sluicing on the passage of shads at the Beaucaire lock (Rhône) (from Larinier, 1991).

Lock entry parameters

Catches made inside the lock chamber in 1989 show a 4 to 8 fold higher attraction for a 60 m^3/s flow than for a 5 m^3/s flow (Délégation de Bassin RMC, 1989); there does not seem to be any significant differences between 0 and 5 m^3/s (Table 4).

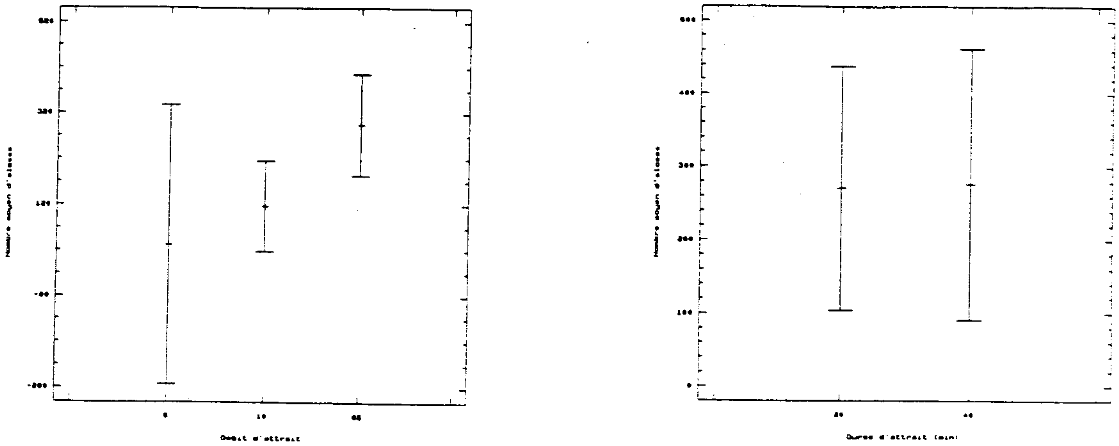
Table 4: Shad entries in the Beaucaire lock (Rhône). Gillnet catches inside the lock chamber.

Date	Attraction flow	Yield	
	m3/s	g/m2/minute of fishing	kg/net
May 9 1989	0	1,2 - 2,1	8,1 - 15,9
May 10 1989	5	1,4 - 2,7	12,1 - 21,6
May 11 1989	60	11,3 - 11,9	29,8 - 44,6

The partial lowering of the upstream gate also tests the entry attraction, assuming that the number of shad exiting is directly proportional to the number entering.

Analyses of variance conducted by Larinier in 1991 did not demonstrate any influence of the attraction flow values (0, 5, 60 m^3/s) or of the duration of the flow (20 or 40 minutes).However, analyses of variance of the

1992 data show that the influence of the entry flow is statistically significant, while the duration of the flow has no significant influence (Figure 7).



(1) (2)

Figure 7: Efficiency of the attraction flow on the entry of shads in the Beaucaire lock (Rhône).Comparison of the mean number of shad for:
(1) three attraction flows (0, 5, 60 m^3/s) with 95% confidence intervals (Jolimaitre, 1992).
(2) two flow durations (20, 40 minutes) with 95% confidence intervals (Jolimaitre, 1992).

Lock exit parameters

Partial lowering of the upstream gate

Analysis of the hydrodynamics conditions on the partially open upstream gates show a succession of 4 hydraulic phases (Gendre, 1995) characterized in Table 5.

Table 5: Determination of shad exit parameters at the Beaucaire lock (Rhône).Characteristics of the flow regime during a partial lowering of the gate (Gendre, 1995).

Phase timing	Hydraulic characteristics	Speed range (m/s)	Level drop (cm)
1	free flowing spurt	3,1 à 4	49 à 81
2	flowing spurt with ondulation vein	2,4 à 3,1	29 à 49
3	surface spurt	2 à 2,4	20 à 29
4	submerged flow	1,4 à 2	10 à 20

Visual observations, supported by statistical analyses, reveal that:

-during phase 1, fish do not surmont the obstacles,
-during phase 2, mullets get over the obstacle,
-during phase 3, both mullets and shads can cross the gate,
-during phase 4, shads are hesitant and swim back and forth in the lock.

Shad distribution over the gate is not homogeneous ; comparison tests on means indicate that shads favour the two lateral zones over the two central zones (Gendre, 1995).

The efficiency of partial lowering was tested by proceeding with two consecutive lowerings of the gate (Béchu *et al.*, 1994; Gendre, 1995). On average, 52% of the shads cross the gate on the first opening. If we exclude atypical sluicing without phase 4, efficiency increases to about 65%. However, a strong variability is noted on the first passage (Figure 8).

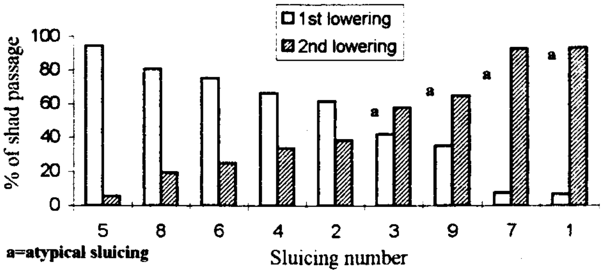


Figure 8: Efficiency of two concecutive lowerings with the same attrction flow (from Gendre, 1995).Beaucaire lock (Rhône).

Total lowering of the upstream gate

Net catches after the visual count on the partially lowered gate do not permit a calibration of the method: results vary greatly and the efficiency proportion of net catches varies from 5.1% to 31.2% under identical exit conditions, with a mean of 13.1% (Béchu *et al.*, 1994).Therefor, it is not possible to compare the efficiency of different exit flows (0, 5, 60 m^3/s). However, it should be noted that the average number of shad netted upstreap (31.6 shad for 5 sluicings) with a fully open gate and a flow of 5 m^3/s is around three times the average number of shads netted with a partially lowered gate (11.5 shad for 8 sluicings), experimental conditions at netting sites were similar.

Species could not be identified by echosoundings (Jouffre, 1995).Nevertheless, the vertical sounder revealed two preferred migration depths: between 0.7 and 1.7 meter, and between 2.7 and 4.7 meters.

There is a good linear correlation (r>0.8), for the vertical and the horizontal sounders (22° and 11°), between the number of recorded echos (Figure 9); this is not true for the 7° sounder (Jouffre, 1995).

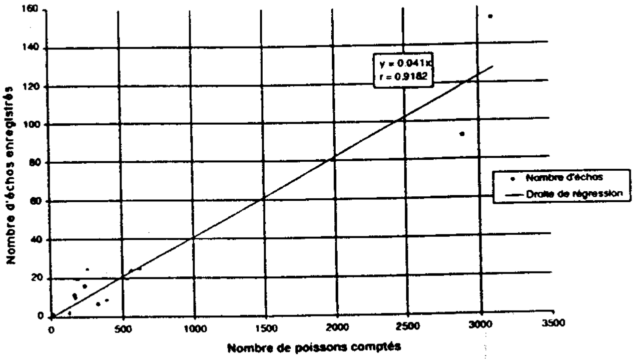


Figure 9: Correlation between the number of recorded echos and the number of fish counted at the Beaucaire lock (Rhône) - (Jouffre, 1995).

This relation gives an acurate estimate of the total number of fish migrating during a total lowering.

Finally, it appears that almost half of the echos are recorded during the first 5 minutes of the total lowering operation, lasting a total of 20 minutes. After 5 minutes, the number of echos varies, for equal time laps, between 5% and 11% with a slight increasing trend after 15 minutes (Figure 10) (Jouffre, 1995).

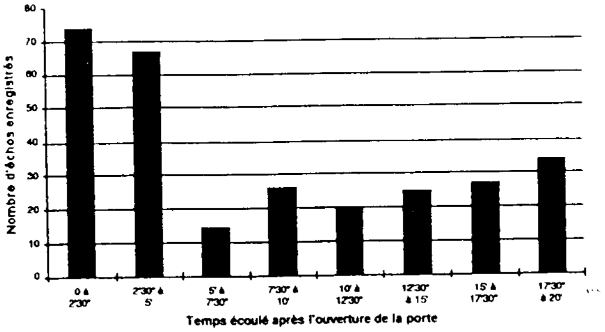


Figure 10: Temporal evolution of the number of echos during the total lowering of the upstream gate at the Beaucaire lock (Rhône) (Jouffre, 1995).

MANAGEMENT PROPOSALS

The Beaucaire navigation lock could become an efficient device to facilitate the migration of shads when the Rhône discharge is lower than the minimum turbine flow required at the generating station.

Management of the lock requires a particular and well adapted operation : indeed, the current equipment must be adjusted, particularly the filling sluice, and the entry and exit phases must be automated and time-separated.

Lock entry

A 60 m^3/s flow value is sufficient to attract shads from downstream of the lock into the lock chamber. The apparent lack of influence of the attraction flow duration suggests that the attraction flow is efficient on shads located near the lock.

Lock exit

The partial lowering of the upstream gate allows the passage and the visual monitoring of the migration and of the efficiency of the technique.However this technique is less efficient, for a 5 m^3 /s flow, than a total lowering; also it cannot be reproduced at upstream locks with horizontally sliding gates. Therefore, it appears preferable to use total lowering.

The problem associated with the exit flow value (0, 5, 60 m^3/s) has to be solved; echosounding may provide elements of a solution if it is demonstrated that the echos recorded at separate depth strata correspond to two different species, shad an mullet. While waiting for a quantitative tool, it has been demonstrated that passages occur at a 5 m^3/s flow. The exact exit duration will have to be eventually clarified.

Management rules

With the current state of knowledge and field experience, the number of daily sluicings is established at a minimum of two and a maximum of five in order to answer the fleeting aspect of the migration. Current rules recommand a routine attraction flow of 60 m^3/s during 20 minutes for lock entry and a complete lowering of the upstream gate with an attraction flow of 5 m^3/s for lock exit.

These rules will be improved upon as additional data become available. In particular, we should harmonize the number of sluicings with the importance of the migration.

ACKNOWLEDGEMENTS

The authors wish to thank all the financial and technical interveners involved various degrees in the "Plan migrateur Rhône", which represent approximately fifty people. These include the "Association Migrateurs Rhône Méditerranée" regrouping the departmental fishing federations, the professional fishermen and the F.R.A.P.N.A. 07, the Compagnie Nationale du Rhône, the Agence de l'eau R.M.C., the ministry of the Environnment and its decentralized agencies (D.I.R.E.N.), the Conseil Supérieur de la Pêche, the Rhône-Alpes, Provence-Alpes-Côte d'Azur and Languedoc-Roussillon regions, the Bouches-du-Rhône et du Gard departments, the Voies Navigables de France, Electricité de France, the C.R.N.S. (Arles) and the I.N.R.A. of Thonon-les-Bains.

REFERENCES

Baril, (1988).Etude des conditions de franchissement par l'alose (*Alosa ficta rhodanensis*) d'une passe à ralentisseurs de fond expérimentale (site expérimental de Beaucaire).CSP Montpellier.EDF Chatou.Rapport d'étude.36 p.+annexes.

Béchu, Blanchet, Tabardel, Menella, Gendre,(1995).Etude préalable du franchissement par les aloses des écluses de navigation sur le Bas-Rhône.Plan Migrateurs Rhône-Méditerranée.Association Migrateurs Rhône-Méditerranée.33p.

Comité de gestion des poissons migrateurs du bassin R.M.C, (1995).Plan de gestion des poissons migrateurs 1995-19999.Rapport de synthèse.33 p.+annexes.

Délégation de bassin RMC, (1989).La pêche professionnelle et amateurs aux engins sur le Rhône-Suivi des captures.Schéma de vocation piscicole du fleuve Rhône.Document n°7 .41 p.+annexes.

Délégation de bassin RMC, (1990).Suivi de la migration des aloses à l'écluse de navigation de Beaucaire sur le Rhône.Schéma de vocation piscicole du fleuve Rhône.Document n°4.7 p.+ annexes.

Délégation de bassin RMC, (1991).Schéma de vocation piscicole du fleuve Rhône.Rapport de synthèse.202 p.+annexes.

Douchement, 1981.Les aloses des fleuves français:Alosa fallax,Lacipède (1803) et Alosa alosa, Linnée (1798)-Biométrie,Ecobiologie:autonomie des populations.Thèse de 3ème cycle.Univ.Sc. Tech. Languedoc.377 p.

Gallois, (1947).Aménagement du Rhône vivarais par la Compagnie Nationale du Rhône.Bull. Fran. Pisc., 146, 25-34.

Gendre, (1995).Validation de la manoeuvre d'abaissement partiel de la porte amont.Campagne d'étude 1995.Plan Migrateurs Rhône-Méditerranée.Association Migrateurs Rhône-Méditerranée.31 p.

Jouffre, (1995).Etude de faisabilé d'une technique de comptage par échosondage des poissons transitant par l'écluse de Beaucaire-Vallabrègues.Campagne d'étude 1995.Plan Migrateurs Rhône-Méditerranée.Association Migrateurs Rhône-Méditerranée.27 p.

Jolimaitre, (1992).Franchissement par l'alose feinte de l'aménagement de la chute de Vallabrègues.Etude du franchissement de l'écluse de navigation.Avant-projet de passe à poissons sur le seuil de Beaucaire.ENGREF.CSP.42 p.+annexes.

Larinier, (1991).Utilisation de l'écluse de navigation pour le franchissement de l'usine de Beaucaire par les poissons migrateurs.Campagne 1991 et propositions.5 p.

Rameye, Kiener, Spillman, Biousse, (1976).Aspects de la biologie de l'alose du Rhône-Pêche et difficultés croissantes de ses migration.Bull. Fran.Pisc., 263, 76 p.

Ruchon, Menella, Tabardel, (1994).Reproductibilité du franchissement par les aloses des écluses de navigation sur le Bas-Rhône.Plan Migrateurs Rhône-Méditerranée.Association Migrateurs Rhône-Méditerranée.47 p.

FLOW STRUCTURE OF STREAM-TYPE FISH WAY

Tetsuro Tsujimoto
Dept. of Civil Engineering., Kanazawa University
2-40-20, Kodatsuno, Kanazawa, 920
Japan
khl@kenroku.ipc.kanazawa-u.ac.jp

Yoshihiko Shimizu
Dept. of Civil Engineering., Gunma University
1-5-1, Tenjin-cho, Kiryu. 350
Japan

ABSTRACT

The flow structure of stream-type fishways are discussed with numerical analysis. As typical examples, (i) standard Denil fishway, (ii) superactive-type bottom baffle fishway and (iii) Alaska steeppass are discussed in the same framework. The stream-type fischway has a rectangular flume with closely spaced fins. The flow is three-dimensional, and locally varied particularly downstream of fins. Hence, previously the design of fishways depended upon experience and model or prototype tests.

In this paper, the authors believe that the function of fishway is brought about by the longitudinally averaged flow structure: a special depth-discharge relation, a pair of cellular motion and the subsequent peculiar cross sectional velocity distribution. If the flow can be treated by longitudinally averaged version, comparatively easy analysis is possible. If the presence of fins is represented by adding the term of drag, which is longitudinally averaged, the complicated boundary of flow due to the existence of fins need not to be considered. Only the side walls and the bottom are solid boundaries of flow.

If the fins are vertical and perpendicular to the side walls, they play only as a cause of resistance. However, fins in stream-type fishways are obliquely arranged, not vertical, not perpendicular to the side wall. The interval, height and angles to represent oblique arrangement define a type of fishways. If the fins are obliquely arranged, the drag has other components beside the longitudinal one. They induces a secondary fluid motion. Particularly, previous fishways are designed to bring about a pair of cellular motion which causes the efficient velocity distribution for fish migration.

The present numerical analysis provides the primary velocity distribution, secondary flow vector distribution and the depth-discharge relation. Thus, the present method is a powerful means to design more efficient fishways easily. In calculation, some numerical parameters should be determined empirically, and thus, more systematic comparisons of the calculation and flume tests are necessary.

KEY-WORDS : Fishway hydraulics / Denil fishways / Alaska steeppass / Superactive-type bottom baffle / Numerical analysis / k-ε model / Cellular motion

INTRODUCTION

Fishways are constructed to enable fishes to overcome obstructions such as weirs in rivers to their migrations. Fishways are classified into three groups : (i) stream-type; (ii) vertical slot fishways; and (iii) pool and weir. The stream type is often called "Denil fishway," which has essentially a straight rectangular flume with closely spaced fins (baffles or vanes) on the bottom and/or side walls. These fins are arranged to keep proper depth and to bring about proper flow pattern for fishes. Among many types of this type of fishways, the following three (see Fig.1) are common: (a) standard Denil; (b) superactive-type bottom baffle fishway; and (c) "Alaska steeppass". In the following, Denil is to imply a standard Denil.

In stream-type fishways, the flow is controlled by fins or baffles on the bottom and/or side walls. These at first bring about higher hydraulic resistance to keep the depth. The fins are oriented in oblique manner to the longitudinal axis, and they changes the flow pattern. The flow without fins is driven by the longitudinal component of the gravity, and the vertical fins arranged perpendicular to the longitudinal axis simply act as resistance. However, if fins are arranged in oblique manner, they bring vertical and/or transverse forces on the fluid to drive the secondary motion.

For example, the velocity profile at the centerline of a standard Denil fishway is rather linear, where the range of velocity is wide and the zone with some velocity has equal ratio to the total depth. Such a flow characteristic is caused mainly by cellular motion driven by oblique fins on the side walls (which brings downward force to the fluid near the side walls), and the upward motion along the centerline brings a linear profile of the primary velocity. In case of superactive-type fishway, oblique fins on the bottom (V-type arrangement) produce a pair of cellular motion and the downward current along the centerline brings a peculiar profile of primary velocity, where the surface flow is retarded. Such a flow is suitable for passage of boats. Additionally, the surface flow concentrates to the center by the pair of cellular motion to avoid collision of a boat with the side walls. The flow with "Alaska steeppass" is more complicated.

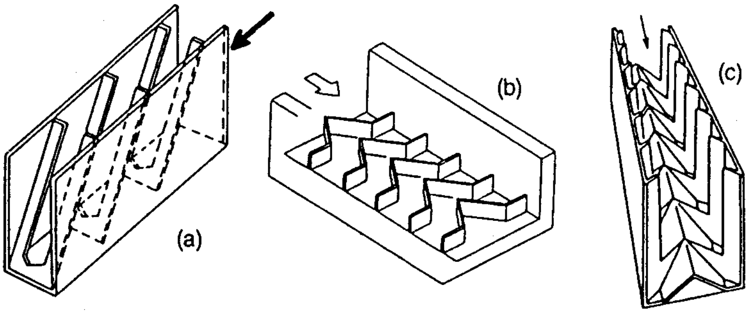


Fig.1: Stream-type fishways
(a) Denil, (b) Superactive-type bottom baffles and (c) Alaska steeppass

Most of efforts in hydraulics of fishways were conducted by laboratory experiments (Katopodis, 1982; Rajaratnum and Katopodis, 1984; Larnier, 1992), and one has to conduct hydraulic model test to design fishways though a simple design manual have been proposed. However, if one would employ a new geometry of fishway to improve the flow structure or the depth-discharge relation, such a manual does not work and one must repeat enormous hydraulic model tests. In this study, the essential mechanism for fishway with some devices to bring proper velocity profile is investigated, and numerical model to describe it is proposed. One can reasonably change the geometry of stardard design of fishways or look for new types of strream-type fishways by consulting numerical analyses based on the present study.

The conventional fishways, not only steam type but also vertical slots and pool and weir, are artificially prepared part of rivers which has a fuction to help migration of fishes. However, recent needs may be to look for more natural fishway, and then the flow in a stream sould be controled by more natural materails such as stones, vegetation and so on instead of artificial fins. The concept and method of this study would help to design such a flow-structure control of a stream to improve more general fish habitat (Tsujimoto, 1996).

SECONDARY CURRENT CONTROL BY SKEWED ARRANGEMENTS OF FINS

The hydrodynamic drag force acts on a fin, and it is vertical to the face of the fin if the friction is negligible. If the streamline near the fin is negligible, only the drag which acts on the fin vertically to its face is significant. Then, it reacts on the fluid, and it drives the secondary current. Fig.2 shows some basic arrangements of fins in oblique manner, where x, y, z=longitudinal, vertical and transverse direction. Fig.2(a) shows fins vertically arranged but with angle θ in the x-z plane, and the drag force has the transverse component D_y besides the longitudinal one D_x. Fig.2(b) shows fins vertically arranged on a side wall with angle φ in the x-y plane, and the drag force has the vertical component D_z besides D_x. Fins shown in Fig.2(c) has angles (θ, φ) both in the x-z and x-y planes, and the drag force has three components. These components of the drag are written as follows:

(1)
$$D_x = D\sin\theta\sin\varphi\ ;\ D_y = -D\sin\theta\cos\varphi\ ;\ D_z = -D\cos\theta\sin\varphi$$

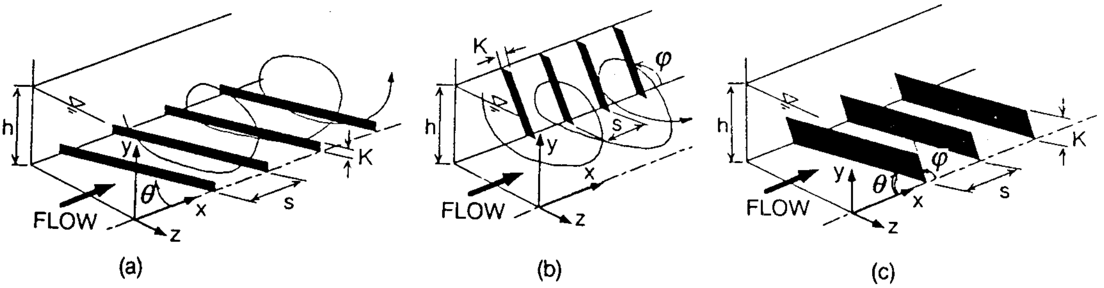
(2)
$$D = \frac{1}{2}C_D\rho(U\sin\theta\sin\varphi - V\sin\theta\cos\varphi - W\cos\theta\sin\varphi)\cdot\sqrt{U^2+V^2+W^2}\,K$$

in which D=amplitude of drag force; (U, V, W)=longitudinal, vertical and transverse components of time-average velocity; C_D=drag coefficient of a fin, K=height of fin; and ρ=mass density of water.

Fins are arranged in some spacing in fishways. In this study, the local flow behavior near fins are neglected, but we focus the longitudinally averaged flow field, where cellular motion is formed. The average force for the obstacles (fins) to act on the fluid per unit mass (F_x, F_y, F_z) are expressed as follows:

(3)
$$F_x = \frac{D_x}{\rho s K}\ ;\ F_y = \frac{D_y}{\rho s K}\ ;\ F_z = \frac{D_z}{\rho s K}$$

in which s=interval of fins. These forces should be added in the momentum equation of fluid.



(a) (b) (c)

Fig.2: Drag acting on fins obliquely arranged on bottom or/and side wall(s)

If the additional force is taken into account, the energy loss due to the workdone by it appears. Such a loss is the result of the process that the turbulent energy is produced and then it dissipates. Hence, the production of turbulence energy corresponding to the workdone by the drag is considered in k- and ε-equation (k=turbulent energy; ε=dissipation rate of turbulent energy). Then, the governing equations of the flow are written as follows after parabolic flow assumption (dimensionless version):

(4) $$\frac{\partial}{\partial x^*}(\phi U^*)+\frac{\partial}{\partial y^*}\left(\phi V^*-\Gamma\frac{\partial}{\partial y^*}\right)+\frac{\partial}{\partial z^*}\left(\phi W^*-\Gamma\frac{\partial}{\partial z^*}\right)=S_\phi$$

in which $x^*\equiv x/h$; $y^*\equiv y/h$; $z^*\equiv z/h$; h=depth of flow; $U^*\equiv U/U_\mathrm{m}$; $V^*\equiv V/U_\mathrm{m}$; $W^*\equiv W/U_\mathrm{m}$; U_m=cross-sectionally averaged velocity; ϕ, Γ, S_ϕ=quantity transported (non-dimension), turbulent diffusion coefficient of ϕ (non-dimension); production term of ϕ (non-dimension); and they are written as shown in Table 1.

Table 1: Quantities in governing equations of flow

ϕ	Γ	S_ϕ
U^*	$\nu_\mathrm{T}^*+\frac{1}{R_\mathrm{e}}$	$-\frac{\mathrm{d}P^*}{\mathrm{d}x^*}-\frac{2}{3}\frac{\partial k^*}{\partial x^*}-F_\mathrm{x}^*+\frac{\partial}{\partial y^*}\left(\nu_\mathrm{T}^*\frac{\partial V^*}{\partial x^*}\right)+\frac{\partial}{\partial z^*}\left(\nu_\mathrm{T}^*\frac{\partial W^*}{\partial z^*}\right)$
V^*	$\nu_\mathrm{T}^*+\frac{1}{R_\mathrm{e}}$	$-\frac{\partial P'^*}{\partial y^*}-F_\mathrm{y}^*+\frac{\partial}{\partial y^*}\left(\nu_\mathrm{T}^*\frac{\partial V^*}{\partial y^*}\right)+\frac{\partial}{\partial z^*}\left(\nu_\mathrm{T}^*\frac{\partial W^*}{\partial y^*}\right)$
W^*	$\nu_\mathrm{T}^*+\frac{1}{R_\mathrm{e}}$	$-\frac{\partial P'^*}{\partial z^*}-F_\mathrm{z}^*+\frac{\partial}{\partial y^*}\left(\nu_\mathrm{T}^*\frac{\partial V^*}{\partial z^*}\right)+\frac{\partial}{\partial z^*}\left(\nu_\mathrm{T}^*\frac{\partial W^*}{\partial z^*}\right)$
k^*	$\frac{\nu_\mathrm{T}^*}{\sigma_\mathrm{k}}+\frac{1}{R_\mathrm{e}}$	$P_\mathrm{k}^*-\varepsilon^*+C_\mathrm{fk}(U^*F_\mathrm{x}^*+V^*F\mathrm{y}^*+W^*F_\mathrm{z}^*)$
ε^*	$\frac{\nu_\mathrm{T}^*}{\sigma_\varepsilon}+\frac{1}{R_\mathrm{e}}$	$\frac{\varepsilon^*}{k^*}\left\{C_{\varepsilon 1}\left[P_\mathrm{k}^*+C_{\mathrm{f}\varepsilon}(U^*F_\mathrm{x}^*+V^*F\mathrm{y}^*+W^*F_\mathrm{z}^*)\right]-C_{\varepsilon 2}\varepsilon^*\right\}$

In Table 1,

(5) $$P_\mathrm{k}^*\equiv\nu_\mathrm{T}^*\left\{2\left[\left(\frac{\partial V^*}{\partial y^*}\right)^2+\left(\frac{\partial W^*}{\partial z^*}\right)^2\right]+\left(\frac{\partial U^*}{\partial y^*}\right)^2+\left(\frac{\partial U^*}{\partial z^*}\right)^2+\left(\frac{\partial V^*}{\partial z^*}+\frac{\partial W^*}{\partial y^*}\right)^2\right\}$$

P_k^*=dimensionless production rate of turbulent energy; $P^*\equiv P/(\rho U_\mathrm{m}{}^2)$; P=cross-sectionally averaged pressure; P'=pressure deviation; $F_\mathrm{x}^*\equiv F_\mathrm{x}h/U_\mathrm{m}{}^2$; $F_\mathrm{y}^*\equiv F_\mathrm{y}h/U_\mathrm{m}{}^2$; $F_\mathrm{z}^*\equiv F_\mathrm{z}h/U_\mathrm{m}{}^2$; $C_{\varepsilon 1}$, $C_{\varepsilon 2}$, σ_k, σ_ε, C_fk, $C_{\mathrm{f}\varepsilon}$=numerical parameters; $k^*\equiv k/U_\mathrm{m}{}^2$; $\varepsilon^*\equiv\varepsilon h/U_\mathrm{m}{}^3$; $\nu_\mathrm{T}^*\equiv\nu_\mathrm{T}/U_\mathrm{m}h$; ν_T=kinematic eddy viscosity; $R_e\equiv U_\mathrm{m}h/\nu$; ν=kinematic viscosity; and the kinematic eddy viscosity is related to k and ε as follows:

(6) $$\nu_\mathrm{T}^*\equiv C_\mu\frac{k^{*2}}{\varepsilon^*}$$

in which C_μ=empirical parameter.

The above explained model was proposed for calculation of flow with cellular motion controlled by oblique arrangement of strip roughness by Tsujimoto (1995).

The numerical parameters in the model except for C_{fk} and $C_{f\varepsilon}$ are determined according to the previously proposed standard values as follows: C_μ=0.09; $C_{\varepsilon 1}$=1.44; $C_{\varepsilon 2}$=1.92; σ_k=1.0; and σ_ε=1.3. The parameters C_{fk} and $C_{f\varepsilon}$ are determined by consulting the agreements of the numerical calculation with the experiments for flow with vegetation (Shimizu and Tsujimoto, 1994), as follows: C_{fk}=1.0; $C_{f\varepsilon}$=1.3.

The governing equations are discretized in staggered grid with SIMPLE algorithm (Patanker and Spalding, 1972). As the boundary condition, the log-law is applied at rigid boundaries (bottom and side wall(s)), and a condition of local balance of energy production and dissipation of turbulence is assumed there. The calculation provides the cross-sectional distributions of respective velocity components, Reynolds stress and dP/dx, which provides the value of energy gradient of the flow, I_e. For the time being, fishway is designed by considering mainly the time-averaged flow structure, but the fish behavior may be affected by turbulence and the local flow behavior. The numerical calculation based of the present method can provide the cross-sectional distribution of turbulence intensity (turbulent energy) though it is longitudinally averaged.

DENIL FISHWAY

For Denil fishway, a standard geometry is proposed as follows: b/B=0.64 ($K=(B-b)/2$); s=0.45 (s/K=2.5); and ψ=135°, K_v/B=0.24 (see Fig.3). This is recommended by the minsistry of agriculture, forestry and fisheries in Japan (1994), but is a little different from that after Larinier (1992). The governing equations are made dimensionless by the depth h and the discharge ($U_m Bh$). Thus, the calculations were conducted with changing the value of h/B under the condition of the constant Reynolds number (discharge).

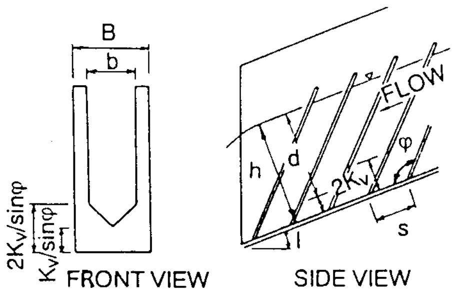


Fig.3: Dimensions of Denil fishway

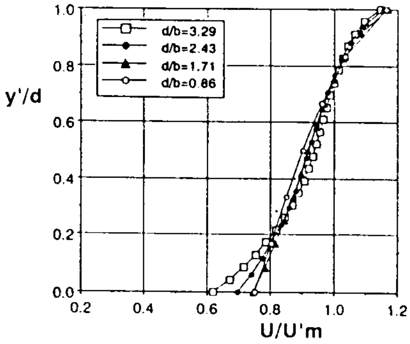


Fig.4: Vertical profile of primary velocity along centerline

The calculated results are shown as the vertical profile of the primary velocity at the centerline $U(y')$, the isovels of the primary velocity $U(y, z)$, and the vector expression of secondary flow components (V, W) in Figs.4, 5 and 6, respectively. In the figures, U_m'=velocity at the point 25% depth below the water surface along the centerline; y=distance measured from the bottom; y'=height measured from the top of the fin; and d=depth measured from the top of the fin. The drag coefficient of the fins, C_D, is not known particularly for rapid flow with closely spaced situation and it would be treated as an empirical parameter in the mean time. When fins are regarded independent each other, C_D is the order of 1.0, and the demonstrated results shown in this chapter were obtained by assuming this value. The structure of the flow field is well described by the present analysis. Particularly, the velocity profile in Fig.6 demonstrates a linear profile to provide wide range of the velocity and equal chance for respective favorite velocity for species of fish. On the contrary, the calculated energy gradient is too small compared with the experimental data (1982). The calculated velocity gradient in Fig.6 is also smaller than the Katopodis's data (1982). It might be caused by poor estimation of the drag coefficient. It might be much larger than the assumed value. In order to obtain more reasonable estimation of the drag coefficient, it is necessary to measure the force acting on the fin directly. Furthermore, the cross-sectional distribution of the turbulent energy ($k=u^2+v^2+w^2$; (u,v,w)=velocity fluctuation but longitudilally averaged) is shown in Fig.7.

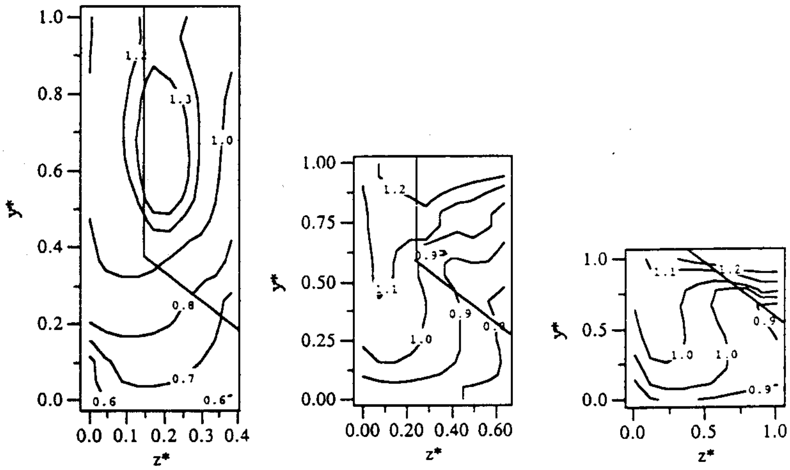


Fig.5: Isovels of primary velocity in cross section of Denil fishway

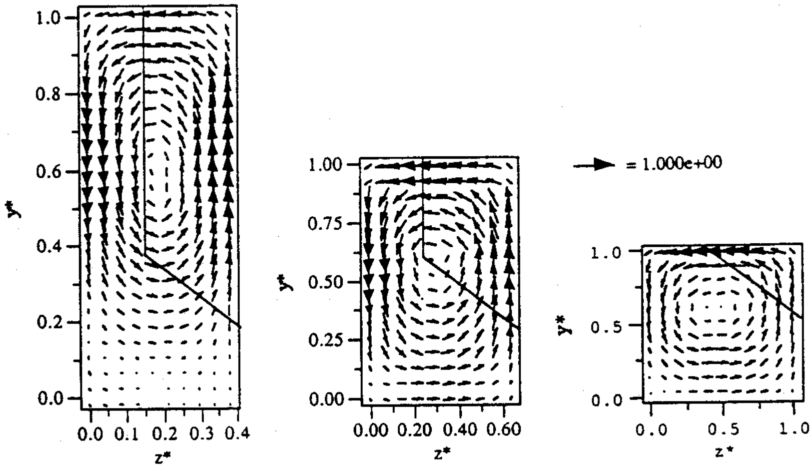


Fig.6: Vector expression of secondary current appearing in Denil fishway

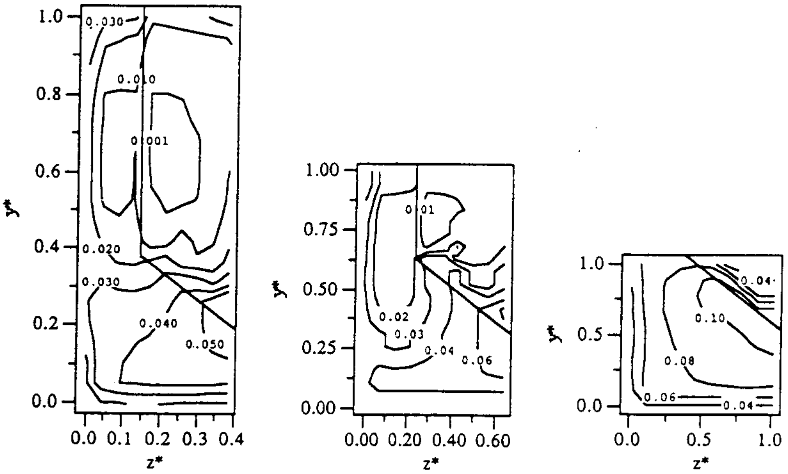


Fig.7: Cross-sectionally distribution of turbulent energy

SUPERACTIVE-TYPE BOTTOM BAFFLES FISHWAY

The standard geometry of superactive-type bottom baffles fishway (Larinier, 1992; Ministry of agricture, forestry and fisheries, 1994) is as follows (see Fig.8): s/K=2.60; B/K=6.0, θ=150°. The center zone of the width $B/2$ has V-type arrangement of baffles (fins), and the zones near the side walls ($B/4$ respectively), the fins are perpendicular to the walls ("wall zone").

The calculations were conducted by changing the ratio of the flow depth to the width. The calculated results are shown as the vertical profile of the primary velocity at the centerline $U(y')$, the isovels of the primary velocity $U(y, z)$, and the vector expression of secondary flow components (V, W) in Figs.9, 10 and 11, respectively. As shown in Fig.10, the surface flow along the centerline is retarded and it brings an advantage for passage of boats with the fact that the cellular motion concentrates the surface flow at the centerline as seen from Fig.11.

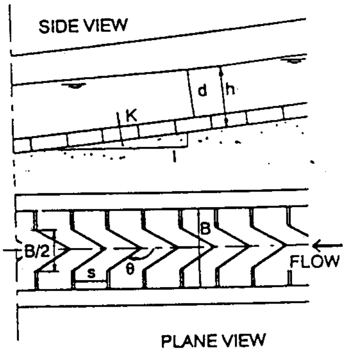


Fig.8: Dimensions of superactive-type bottom baffle fishway

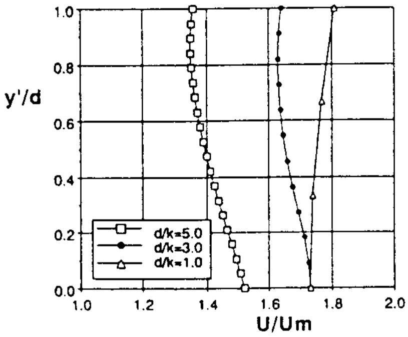


Fig.9: Velocity profile along centerline of superactive-type bottom baffle fishway

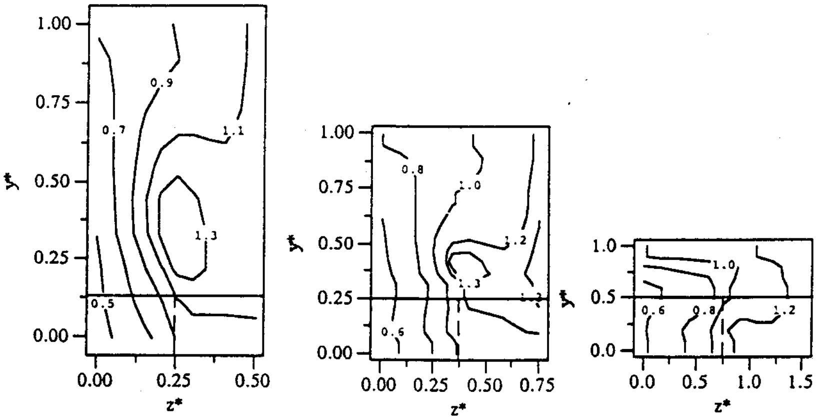


Fig.10: Isovels of primary velocity in cross section of flow over superactive-type bottom baffles

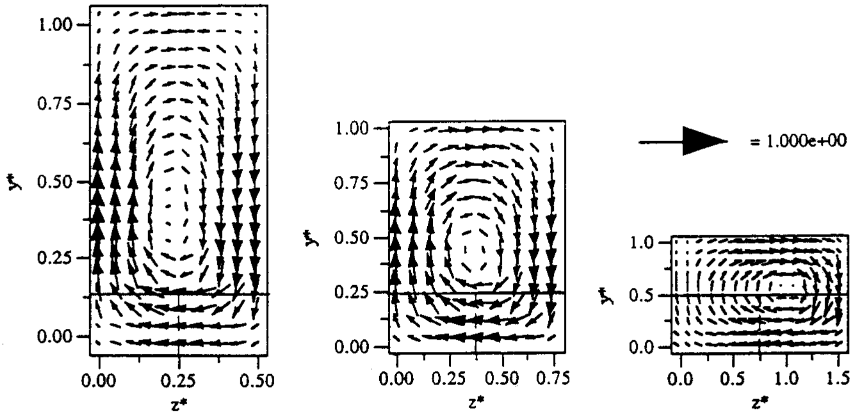


Fig.11: Vector expression of secondary current appearing in flow over superactive-type bottom baffles

There is a slower flow region near the side walls for a stream with "wall zones" as seen from Fig.9. Without perpendicular arrangement of fins near the side walls (without wall zone), such a fact never appear (see Fig.12). In the calculation, the drag coefficient has been assumed to be 10.0, and the calculated results have been able to explain the relation of the discharge and the depth (resistance relation), as shown in Fig.12 (comparison with the curve given by Larinier (1992)).

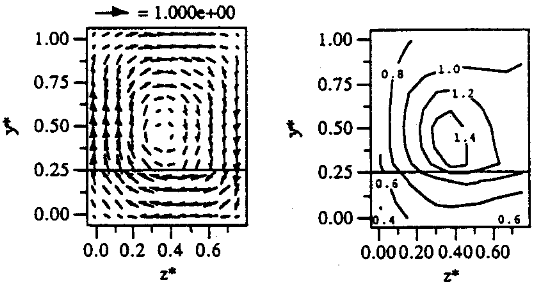


Fig.12: Flow behavior without wall zones

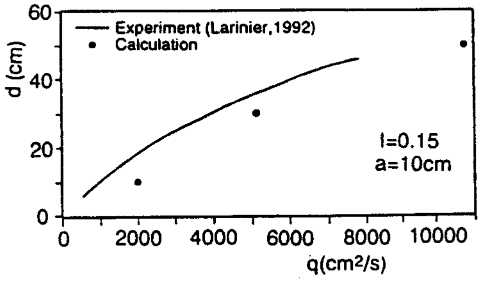


Fig.13: Relation between depth and discharge

ALASKA STEEPPASS FISHWAY

The standard Alaska steeppass fishway has the following standard geometry (see Fig.14; Ministry of agriculture, forestry and fisheries, 1994): B/b=1.58 ($K=(B-b)/2$; K/B= 0.18); s/b=0.715 (s/K=2.51); K_V/b=0.357.

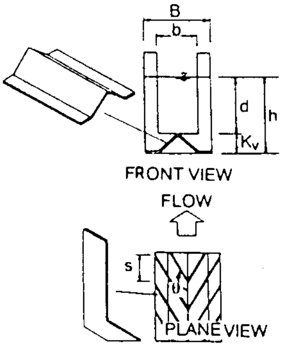


Fig.14: Dimensions of Alaska steeppass steeppass

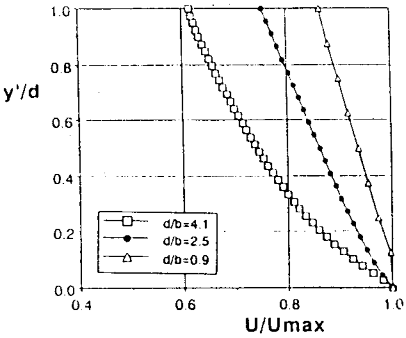


Fig.15: Velocity profile along centerline of Alaska

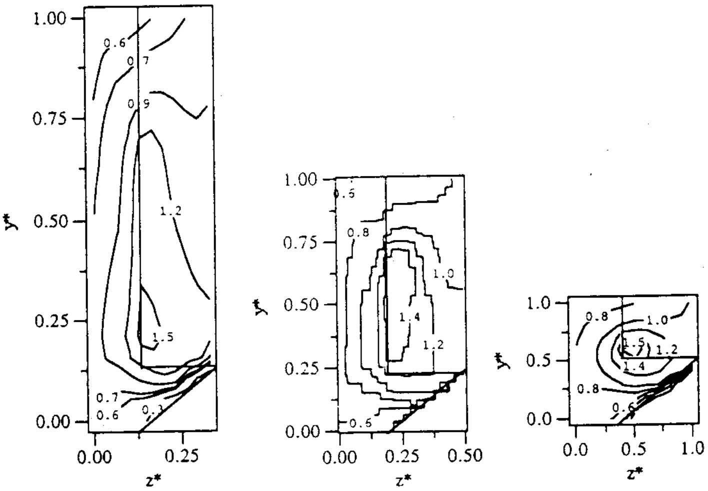


Fig.15: Isovels of primary velocity in cross section of flow over Alaska steeppass

The calculations were conducted by changing the ratio of the flow depth to the width. The calculated results are shown as the vertical profile of the primary velocity at the centerline $U(y')$, the isovels of the primary velocity $U(y, z)$, and the vector expression of secondary flow components (V, W) in Figs.15, 16 and 17, respectively. In case of Alaska steeppass fishways, a pair of cellular motion also appears, and the downward flow appears along the centerline, and the velocity near the bottom at the center line is comparatively high. The flow pattern is somehow similar to the flow over superactive-type bottom baffles. The faster flow appears near the bottom along the centerline, and it is intended to be used by bigger fishes. The vertical profile of the primary velocity is peculiar and quantitatively consistent to the measured one by Katopodis (1982) though the velocity gradient (negative) is slightly overestimated. It must depend on the evaluation of the drag coefficient.

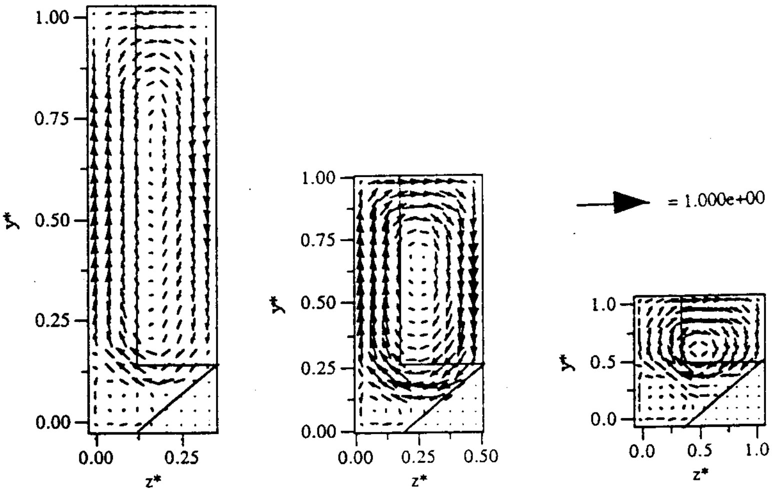


Fig.16: Vector expression of secondary current appearing in flow over superactive-type bottom baffles

In order to compare the experimental data and the calculated results and particularly to investigate the drag coefficient of the fin, hydraulic model of Alaska steeppass was made (Fig.17 in which scale is shown in cm). And, the velocity measurements were conducted by using an electromagnetic current meter. It has a cylinder type probe of which diameter is 4mm, and depending on the orientation arbitrary two components of velocity can be measured simultaneously. The calculated and experimental results on the isovels of the primary velocity and the velocity vector are compared in Figs.18 and 19, respectively. The shaded area in the figure to show the calculated results implies the region where the velocity measurements were carried out. The experimental and the calculated results show a good agreement qualitatively.

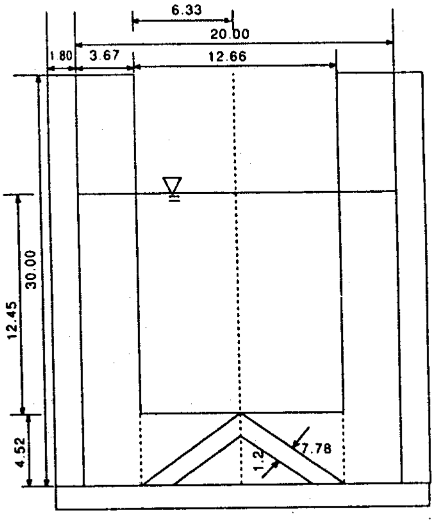


Fig.17: Cross-section of model steeppass fishway

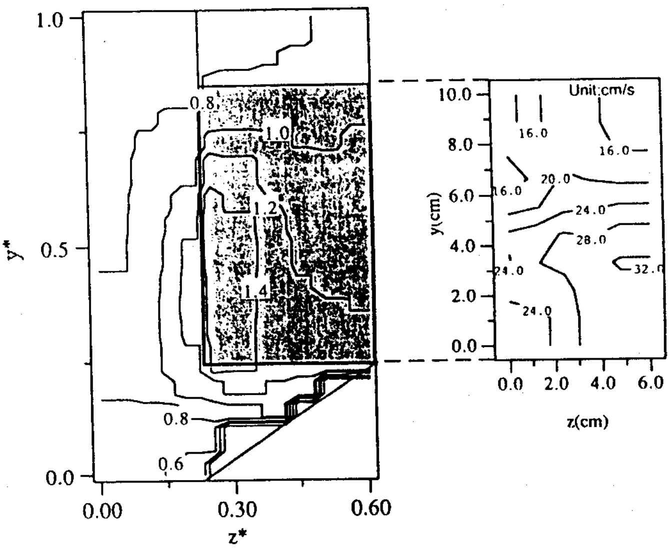


Fig.18: Comparison of isovels of steeppass fishway between measurements and calculation

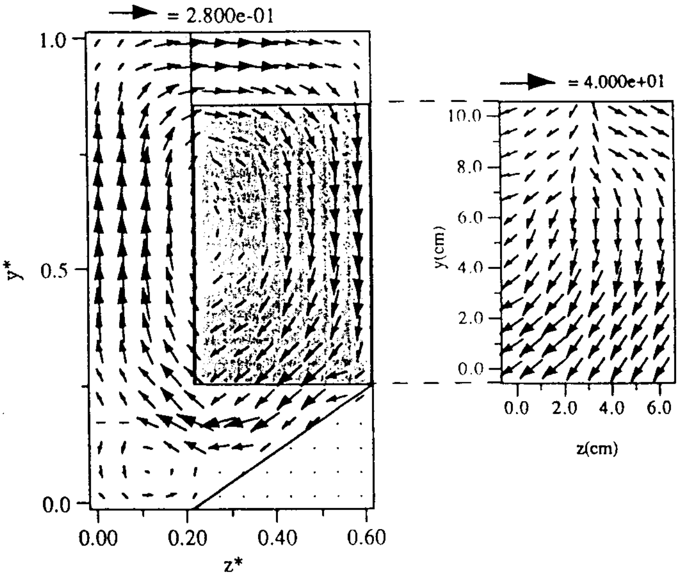


Fig.19: Comparison of velocity vector of steeppass fishway between measurements and calculation

CONCLUSIONS

In this study, numerical analysis based on k-ε turbulence model is applied to flows in stream-type fishways: Denil type, superactive-type bottom baffles and Alaska steeppass. In these fishways, the hydraulic resistance and cellular motion are controlled by obliquely arranged fins (or baffles), and the effect of fins are idealized by adding the term of the drag and the production of turbulent energy corresponding to the workdone by the drag to solve the longitudinally averaged turbulent flow characteristics. The present model can describe the outline of the flow behavior. However, the depth-discharge relation strongly depends on the estimation of the drag coefficient including the sheltering effects. The more local behavior of the flow near fins and the hydrodynamic forces acting upon the fins should be studied in order to obtain more reliable prediction of the flow in various types of fishways by numerical analysis on the present model.

Furthremore, the present method to analyze the flow structure can be applied to evaluate fish habitat from the view point of micro structure of flow such as cellular motion (Tsujimoto, 1996). On the other hand, approaches only from the hydraulics cannot clarify the preference of organisms. Cellular motion and fluctuation might play a role of trigger for the action of organisms. Such an aspect should be investigated by corporation of biologists and hydraulic reserachers.

REFERENCES

Katopodis, C. (1982). *A Study of the Hydraulics of Denil Fishways*, MS Thesis, The Univ. of Alberta, Canada, 158p.

Katopodis, C. (1990). Advancing the art of engineering fishways for upstream migrants, *Proc. Int. Sym. on Fishways '90*, Gifu, pp.19-27.

Larinier, M. (1992). Les passes a ralentisseurs, Chapt. 6, *Bull. Fr. Péche Piscic.*, N°326-327, pp.73-94.

Ministry of Agriculture, Forestry and Fisheries (1994). *Design Manual of Fishways at Head Works*, 194p., (in Japanese).

Patankar, S.V. and Spalding, D.B. (1972). A calculation method for heat, mass and momentum transfer in three-dimensional parabolic flow, *Int. J. Heat Mass Transfer*, Vol.15, pp.1787-1806.

Rajaratnam, N. and Katopodis, C. (1984). Hydraulics of Denil fishways, *Jour. Hydraul. Eng.*, ASCE, 110(9), pp.1219-1233.

Shimizu, Y. and Tsujimoto, T. (1994). Numerical analysis of turbulent open-channel flow over a vegetation layer using a k-ε turbulence model, *J: Hydroscience & Hydraul. Eng.*, JSCE, 11(2), pp.57-67.

Tsujimoto, T. (1995). Cellular-motion control by skewed arrangements of strip roughness in open channel, *Advances in Hydro-Science &-Engrg.*, Vol.II, pp.1839-1846.

Tsujimoto, T (1996). Fish habitat and micro structure of flow in gravel bed stream. *Proc. Ecohydraulics 2000,* Quebec, Canada (in printing).

NATURALLY SHAPED FISHWAY OF THE NAGARA RIVER ESTUARY BARRAGE

Saburo Komura
Emeritus professor of Gifu University, Director, Institute of River Environment
3-14, Mishima-cho, Gifu 502, Japan

Shuhei Kazusa
Chubu Regional Construction Bureau, Ministry of Construction
2-5-1, Sannomaru, Naka-ku, Nagoya 460, Japan

Toshiaki Hara
Chubu Branch Office, Water Resources Development Public Corporation
1-2-1, Sannomaru, Naka-ku, Nagoya 460, Japan

ABSTRACT

Over seventy percent of the surface area of Japan is mountainous and the vast majority of the people and properties are concentrated in the narrow plains. Rainfall in Japan quickly turns into discharge which may descend on and flood the plains to cause vast amounts of damage. The Nobi Plain, through which the Nagara River flows, has always been a particularly flood-prone area. Recently 3,500 houses were severely damaged when the dykes on the Nagara River breached in 1976. The implementation of higher level of flood control measures is urgent.

The major purposes of the Nagara River Estuary Barrage, which was completed in March 1995, are to protect the local residents of the flood-prone area along the river from repetitious flood disasters as well as to develop a new and stable water resources in this area which includes Nagoya, the third largest city in Japan.

Prior to the implementation of the barrage construction project, an intensive environmental impact study had been conducted with the participation of about 90 academics (KST: Kisosansen Survey Team) over a period of four years between 1963 and 1966, which was followed by many other surveys and studies. Based on the results obtained from these studies, two types of fishways, called the lock-type fishway and fishladder with guide flow were to be constructed to provide fish in the river with means of travelling up and down the river. Further environmental impact studies had been conducted since then, and the results confirmed the need of an additional naturally shaped fishway. This naturally shaped fishway is called "*Seseragi Gyodo*" in Japanese, in which "*Seseragi*" means a brooklet.

The naturally shaped fishway of approximately 320 m long and 15 m wide (water surface width 3 m) was constructed on the overflow-type levee on the right-bank side of the Barrage. The fishway produces a near-natural stream to help a variety of fish ascend the river. Many other considerations are given to aquatic biota in the river. The fishway is designed to have an easy gradient of 1/110 (average), riffles and pools are alternately arranged in the fishway filled with cobbles to produce different depths and velocities. It is also meandered to provide fish with resting spots and refuges. Furthermore, at the intake of the fishway, four rubber weirs are constructed to cope with the water level fluctuations.

This paper describes the design of the naturally shaped fishway, hydraulic model testing, hydraulics of the completed fishway, as well as the present state of various aquatic biota travelling up the fishway.

KEY WORDS: Estuary Barrage / Naturally-shaped Fishway / Guide-flow Fishway / Lock-type Fishway / Hydraulic Model Test / Migratory Fish / Demersal Fish / Diverse Velocity / Rubber Gate / Fish Migration Survey

1. THE FISHWAYS OF THE NAGARA RIVER ESTUARY BARRAGE

1.1 Target Fish Species

Important fish species in the Nagara River are migratory species of fish such as ayu (*Plecoglossus altivelis*) and satsukimasu salmon (*Oncorhynchus masou f.*). When constructing the Nagara River Estuary Barrage, fishways were installed in order to minimize as far as possible the effect of the barrage on such migratory fish. Guide-flow fishways and lock-type fishways were installed on both the right and left banks of the barrage. In addition, on the right bank a newly-designed naturally-shaped fishway was constructed.

Past environmental surveys have found 104 species of fish and 48 species of crustaceans in the lower reaches of the Nagara River (from the estuary up to the 30.2 km-point). According to their habitat, the fish can be classified into freshwater fish, migratory fish, fish dwelling in brackish waters, and coastal fish. The species inhabiting brackish or coastal waters can survive without travelling upstream, and it was judged that the construction of the estuary barrage would have little effect on these species. Also the freshwater fish, which spend their entire lives in fresh waters, are not thought to be affected much by the construction of the estuary barrage.

Migratory fish in the lower reaches of the Nagara River, meanwhile, have been confirmed to include anadromous fish, such as the satsukimasu salmon and salangid (*Salangichthys microdon*), that mature in the sea and climb up the river to spawn, catadromous fish, such as eels, that mature in the river and descend the river to spawn in the sea, and fish that have the properties of both, such as ayu and small-spawn common sculpin, which hatch in freshwater but immediately descend to the sea, and climb back up the river to mature and spawn. An example of a fluvial crustacean with a migrant life-cycle is the mokuzugani crab (*Eriocheir japonicus*).

At the Nagara River Estuary Barrage, fishway structures were thus primarily designed taking into account migrant aquatic biota such as ayu, satsukimasu salmon, as well as small-spawn common sculpin (*Cottus pollux*), four-spine sculpin (*Cottus kazika*), and mokuzugani crab.

1.2 Fishway Positions

Based on suggestions from the Kisosansen Survey Team (KST) mentioned above, a lock-type fishway and a guide-flow fishway were installed on the banks, and in addition a naturally-shaped fishway was installed on the overflow-type levee of the right bank. The guide flow channel of the guide-flow fishway on the left bank was fitted with a 1 m-wide denil fishway, and cobbles were used for the right-hand fish ladder on the right bank.

In total, the 30 m-wide lock-type fishway and the 9 m-wide guide-flow fishway on the left bank, plus the 15 m-wide lock-type fishway, 7 m-wide guide-flow fishway, and 15 m-wide naturally-shaped fishway on the right bank, amount to a width of 81 m, equivalent to approximately 12% of the river width of 661 m under normal conditions (Figure 1).

2. DESIGN POLICY FOR THE NATURALLY-SHAPED FISHWAY AND THE STRUCTURE THEREOF

A naturally-shaped fishway involves the positioning of cobbles and boulders on the channel bed, thereby diversifying the flow within the channel, and creating varying flow velocities. Naturally-shaped fishways thus mimic conditions generally observed in brooklets in nature. Such a naturally-shaped fishway was installed in the overflow-type levee on the right bank of the Nagara River Estuary Barrage. As the flow in this fishway is indeed similar to that of a mountain river flow in nature, this fishway can be utilized by a variety of aquatic species, such as small fish that are not powerful swimmers, such as demersal fish, shrimp, crabs, such as the mokuzugani crab, as well as relatively powerful swimmers, such as ayu. (Photograph 1)

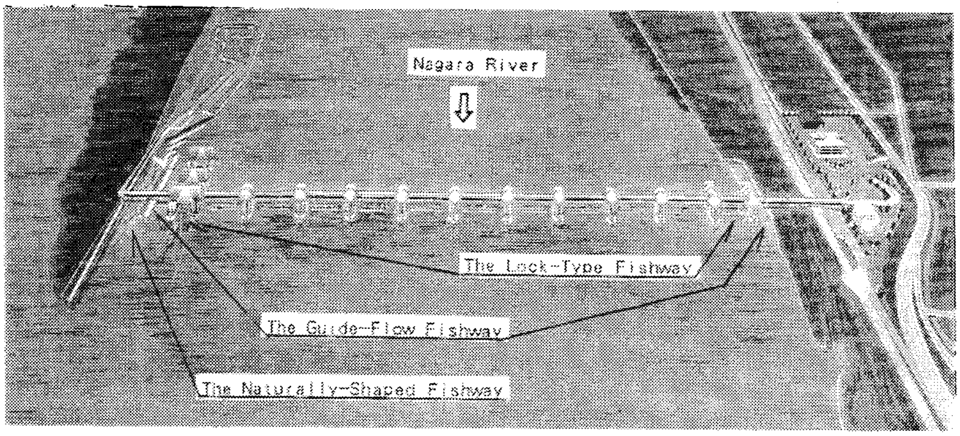
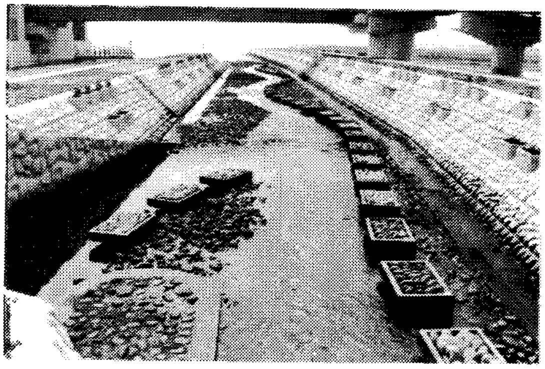
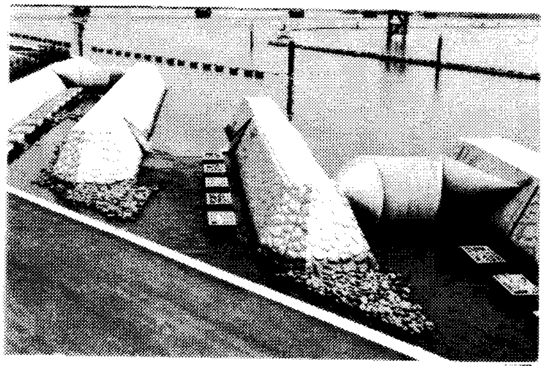


Figure 1: The fishways of the Nagara River Estuary Barrage and their locations



Photograph 1: Naturally-shaped fishway viewed from upstream



Photograph 2: Rubber gates (upstream end of fishway)

The following considerations were taken into account when designing the naturally-shaped fishway.

1) The width should be 3–9 m in total and 3 m for the watercourse, owing to limits on the amount of space available on the overflow-type levee.

2) The mean gradient of the naturally-shaped fishway was set at 1/110. Additionally, in order to uphold the impression of riffles flows, the duration of channel submergence was kept to a minimum by designing the vertical gradient so that the average tide level lies in the vicinity of the guide flow channel's outlet.

3) The velocity and depth should allow upstream travel by biota with less powerful swimming capabilities, such as demersal fish; the velocity should thus be around 0.3 m/s, and the depth around 0.3 m.

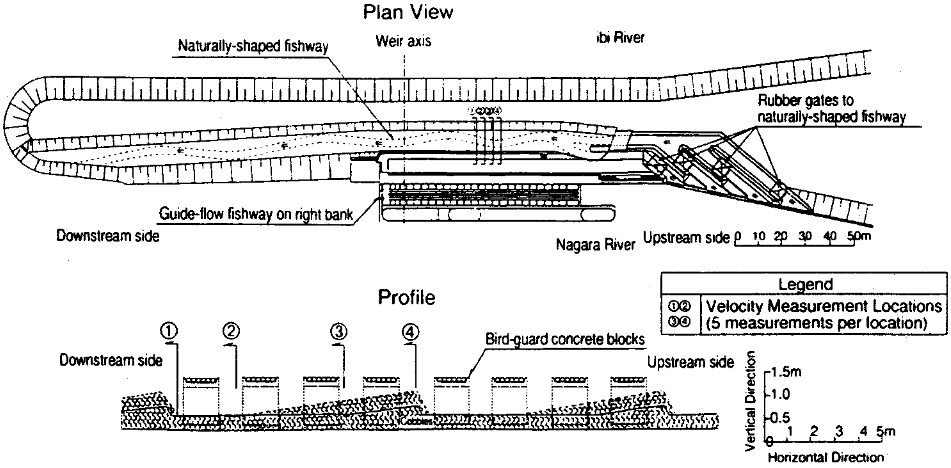


Figure 2: The naturally-shaped fishway

4) The transverse cross-section chosen for the watercourse was triangular, as this will enable continuous changes in water depth and gentle changes in velocities.

5) The longitudinal cross-section was such that riffles and pools — which serve as resting places for the fish — alternate.

6) To allow upstream migration of demersal fish, shrimp, crabs, and eel, cobbles were loosely spread on the floor of the watercourse, allowing diverse velocities to exist.

7) Boulders should be positioned as appropriate, to provide rest areas and refuges. Raising the roughness coefficient also helps to control flow discharge and water depth.

8) In order to cope with fluctuating water levels upstream (T.P. +1.3–0.3 m), exits (with rubber gates to allow full closure or full opening) were positioned in four levels, allowing the flow discharge to be adjusted to a velocity and water depth suited to upstream migration (Photograph 2).

9) The elevation of the entrance to the fishway shall be at lowest tide level, to allow upstream migration even at minimum tide T.P. -1.5 m.

10) Naturally-shaped fishway is shallower compared to other fishways, and hence predators such as birds are expected to prey on the fish in the fishway. Therefore bird-guard concrete blocks should be installed to provide places of refuge for the fish. Figure 2 shows the naturally-shaped fishway.

3. HYDRAULIC MODEL TESTING

Hydraulic model tests were carried out in order to gain advance understanding of the flow conditions in the naturally-shaped fishway. Figure 3 shows an example of velocity distribution, and flow conditions at a typical cross section can be summarized as follows:

- In Section A–A, at which the flow is fastest, the slowest velocity is 0.4–0.6 m/s, surpassing the goal of 0.3 m/s — however, sufficient rest can be obtained at portions of slow current, such as behind boulders or in-between loose cobbles. In section C–C, where the water depth increases, the flow is gentle, at only 0.1 m/s. It was thus determined that flow conditions would be suitable for upstream migration even by species that are less powerful swimmers, such as demersal fish.

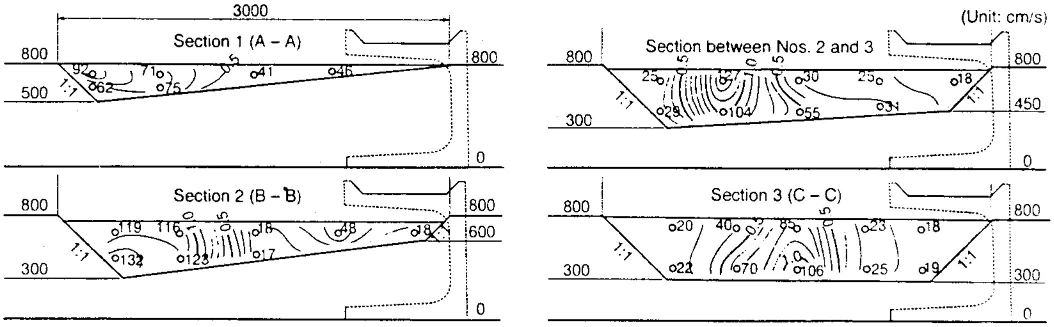
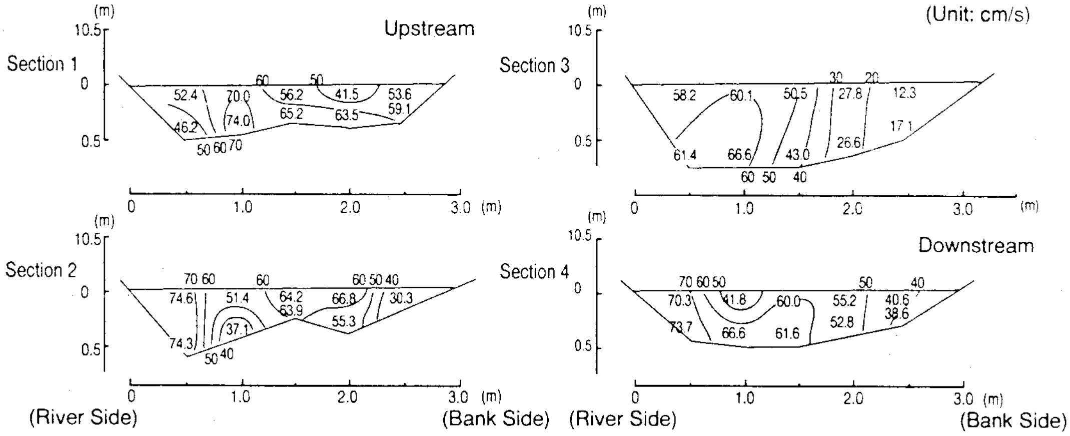


Figure 3: Velocity distribution (within naturally-shaped fishway) in hydraulic model testing (gradient 1/110, discharge: 0.5 m^3/s)



Measurement Conditions: Discharge of 0.83m^3/s

Figure 4: Velocity observations within completed fishway

4. FLOW CONDITIONS IN THE COMPLETED FISHWAY

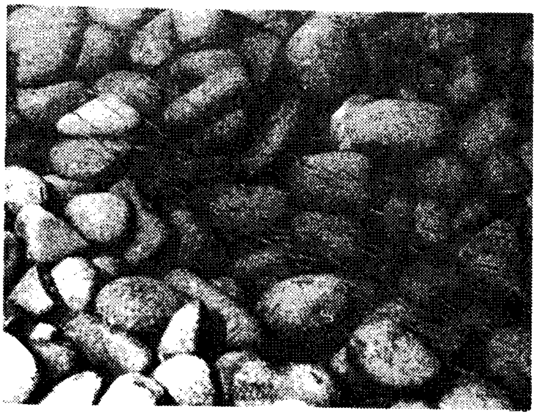
Velocities were measured in order to verify flow conditions in the completed naturally-shaped fishway. Measurements were carried out at the four sections shown in Figure 2. As given in Figure 4, the velocity at an average flow volume of 0.83 m^3/s was 12–75 cm/s, demonstrating both the presence of zones with slow velocity and the diversity in the flow conditions.

5. MIGRATION OF AQUATIC BIOTA

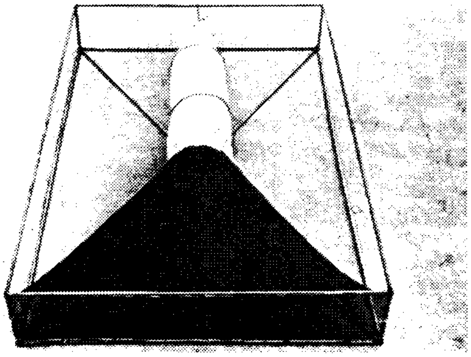
Herewith follows an outline of the results from a survey conducted at the fishway April through May in 1995, after the completion of the barrage, to study the state of fish migration.

5.1 Visual Observation

Visual observations involved counting the number of migrating ayu fry at the upper end of each fishway, by alternately counting for 10 minutes and resting for ten minutes, from sunup to sundown. It was found that over a period of 36 days in 1995 approximately 320,000 fish swam by; 74% of ayu fry used the naturally-shaped fishway.



Photograph 3: Migrating ayu fry



Photograph 4: Small trap

The body length of the ayu in this season was around 5–10 cm; ayu of this size swim at 40–80 cm/s, and can momentarily manage maximum speeds (bursting speeds) of 150 cm/s–200 cm/s. Photograph 3 shows ayu swimming in the naturally-shaped fishway. If one compares the actual flow conditions in the fishway and the capabilities of the fish, it is evident that they can easily swim upstream.

5.2 Sampling Test

Photograph 4 shows a small trap installed at the upstream portion of the fishways, in order to conduct a sampling survey. A total of 42 species (27 species of fish, 6 species of shrimp, and 9 species of crab), such as small-spawn common sculpin, four-spine sculpin, and mokuzugani crab, were observed, proving the naturally-shaped fishway and the other fishways positioned at the estuary barrage to be functioning effectively.

6. CONCLUSIONS

Provisions were taken at the Nagara River Estuary Barrage Project to allow unhindered upstream migration by ayu and satsukimasu salmon, as well as less powerful demersal fish, shrimp, and crabs. Enhanced fishways were installed with the purpose of preserving ecosystems. In the migration survey carried out in 1995, the fishways at the estuary barrage were found to be performing adequately. Observations of the aquatic species will continue, in order to help us understand the conditions surrounding the upstream and downstream migration of fish using the fishways; in addition, we hope to work on establishing gate control mechanisms which are more effective in terms of allowing aquatic biota to pass by the barrage.

7. ACKNOWLEDGMENTS

The authors thank those members of the Fishway Advisory Committee of the Water Resources Development Public Corporation who helped in the planning of the fishway.

HYDRAULICS OF RIFFLES AND THEIR APPLICATIONS TO FISHWAYS

Saburo Komura
Emeritus Professor of Gifu University
Director, Institute of River Environment
3-14 Mishima-cho, Gifu City, Gifu 502, Japan

ABSTRACT

Predicting velocity and depth of flow require knowlege the resistive properties of river channels. Most investigations of this subject have been concentrated on flows with small-scale roughness elements, in which the size of the bed sediment is small compared with the depth of flow. Accordingly, flows in riffles with protruded large-scale roughness elements. such as cobbles and boulders, have received little attention among researchers and engineers, and so far no generally acceptable resistance equation has been proposed for these flows. For developing the hydraulics of riffles and their applications to natural-like fishway designs, research on resistace to flows with cobbles/boulders protruded and over cobbles/boulders beds, such as those in mountainous areas is very important. In this paper, computational equations on the resistance to flow are newly developed from the Froude number distributions and from newly defined roughness density. In order to test the applicability and reliability of newly developed resistance equations, published experimental and field data on flows over and through large-scale roughness elements are used.

KEY-WORDS: Riffles / Large-scale roughness / Resistance to flow / Friction factor / Manning's roughness / Flow through protruded boulders / Standard deviation of bed material / Relative submergence / fishway / Froude number

INTRODUCTION

In mountain river, bed material consists of gravel, cobbles and boulders, and the relative submergence, or ratio of depth, d, to sediment size, D_{84}, ranges from the region of large-scale roughness, via intermediate-scale roughness, to the region of small-scale roughness. The regions are defined as

Large-scale roughness: $d/D_{84} < 1.2$
Intermediate-scale roughness: $1.2 < d/D_{84} < 4$
Small-scale roughness: $d/D_{84} > 4$

Here, d = mean depth of flow, D_{84} = the size of median axis of the bed material which is larger than 84% of the material.

FLOW RESISTANCE EQUATION FOR LARGE-SCALE ROUGHNESS

Previous studies have indicated that the flow resistance of large-scale roughness is related mainly to the form drag of the elements and their disposition in the channel (Bathurst, 1978, Thompson and Campbell, 1979, Bathurst et al. 1981). Since the associated resistance processes are different from those for small-scale roughness, it is not possible to use the semi-logarithmic resistance equations developed for small-scale roughness (Bathurst et al. 1981). In developing a theoretically based resistance equation for large-scale roughness, it is necessary to start from the analysis of the Froude number equation which determine the drag of individual elements, and the processes of roughness geometry and channel geometry.

Derivation of Fr Equation

From the results of Figures 1and 2, the Froude number equation can be expressed as

(1) $$Fr = a \exp[b(d/D_{84})]$$

in which, Fr = the Froude number defined $Fr = U/(gR)^{1/2}$ (here U = mean velocity o f flow, and R = hydraulic radius), a and b = constants varying with the bed material properties and channel geometry. From observed data published in the world, it is assure that the terms of a and b would be a function of roughness size distribution, and channel geometry, so those variations with the standard deviation of bed material distribution, σ, and water surface slope , I, were studied.
It is assumed that the constants of a and b have the following function forms:

(2) $$a = fn(I)$$

(3) $$b = fn(\sigma, I)$$

in which, $fn(\ )$ = a function. Quantification of Eq. (1) was carried out, term by term, by using observed data.

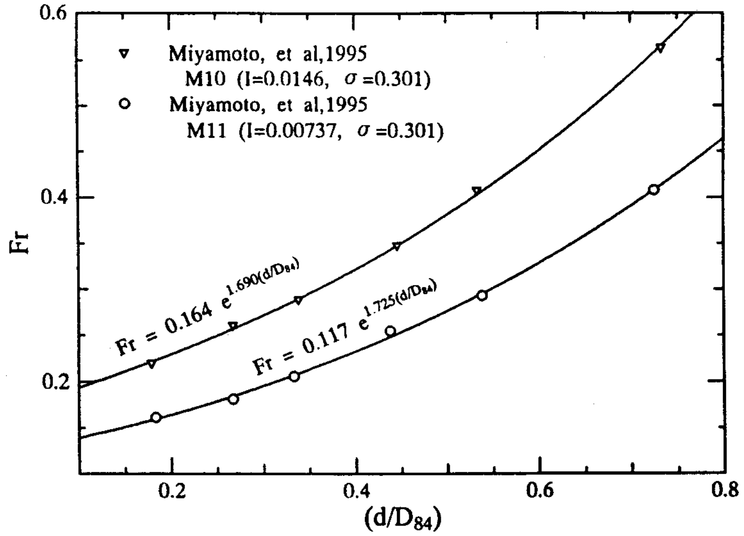


Figure 1: The relationship between Fr and d/D_{84} (M10 and M11)

The standard deviation, σ, is defined by Eq. (4).

(4) $$\sigma = \log\left(\frac{D_{84}}{D_{50}}\right)$$

Also, the following standard deviation, σ_{ϕ}, is widely used.

(5) $$\sigma_{\phi} = \sqrt{\frac{D_{84}}{D_{16}}} = \frac{D_{84}}{D_{50}}$$

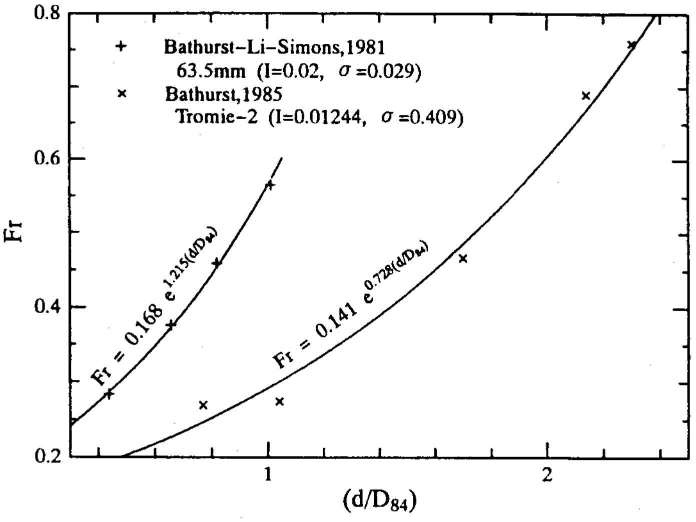


Figure 2: The relationship between *Fr* and d/D_{84} (Tromie-2 and 63.5mm)

The relationship between σ and σ_ϕ can be expressed as

(6) $$\sigma_\phi = e^{2.3\sigma}$$

The results support the above mentioned assumption, giving

(7) $$a = 1.18\ I^{1/2}$$

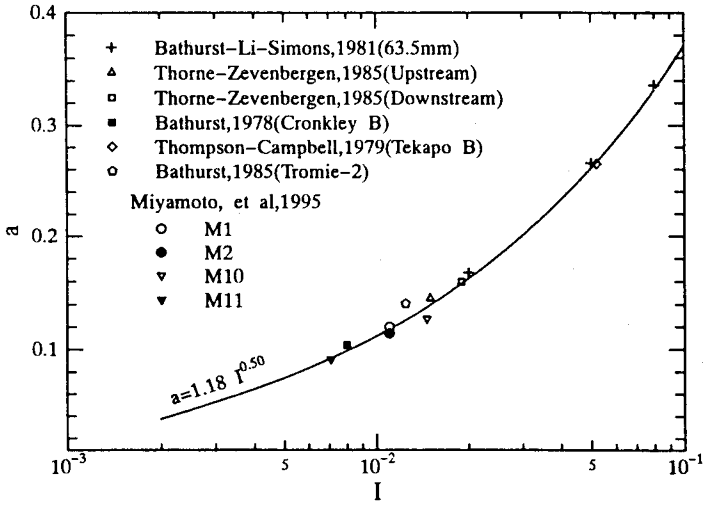


Figure 3: The relationship between *a* and *I*

Figure 3 supports this relationship. Since the function form of *a* was obtained, Eq. (1) can be rewritten as

(8) $$\ln\left(\frac{Fr}{a}\right) = b\left(\frac{d}{D_{84}}\right)$$

Generally, the relative submergence, (d/D_{84}), is given by observed data for analyzing the term b.

(9) $$Y1 = \frac{\ln(Fr/a)}{(d/D_{84})} = b$$

In order to determine the function form of b, at first, the effect of σ should be analyzed. Denoting $Y1$ as Eq. (9), the relationship $Y1$ and σ is analyzed by observed data in the region of $I = 0.007\sim0.02$. Figure 4 shows this relationship.

(10) $$Y1 = 1.18\ e^{-0.8\sigma}$$

Using Eq. (10), Eq. (9) can be rewritten as

(11) $$Y2 = \frac{\ln(Fr/a)}{(d/D_{84})\ \exp(-0.8\sigma)} = fn(I)$$

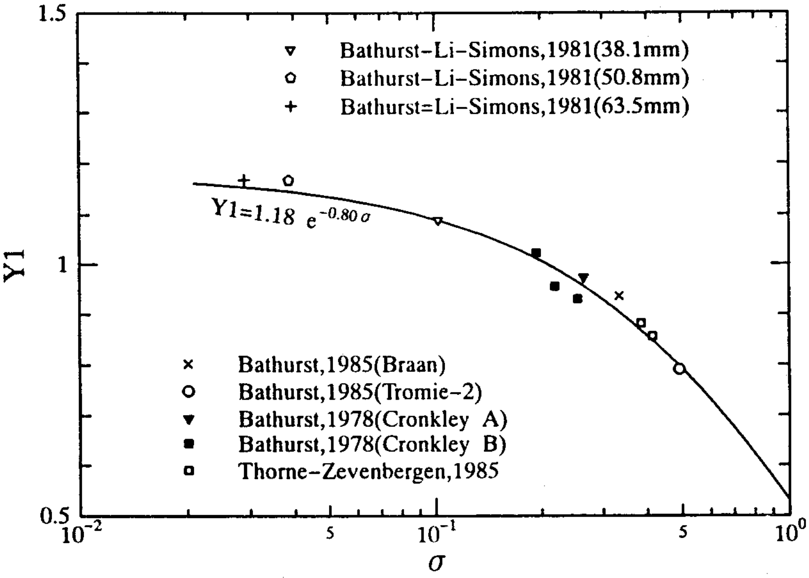


Figure 4: The relationship between $Y1$ and σ

Again, it is assumed that $Y2$ should be a function of water surface slope, I. Plotting the values of $Y2$ and I as Figure 5, the following equation was obtained.

(12) $$Y2 = 1.1\ e^{5.5I}$$

Using Eq. (12), Eq. (11) can be rewritten as

(13) $$Y3 = \frac{\ln(Fr/a)}{(d/D_{84})\ \exp(-0.8\sigma)\ \exp(5.5I)} = constant\ = c$$

in which, c = a proportional constant. The value of c = 1.1 was obtained from observed data.

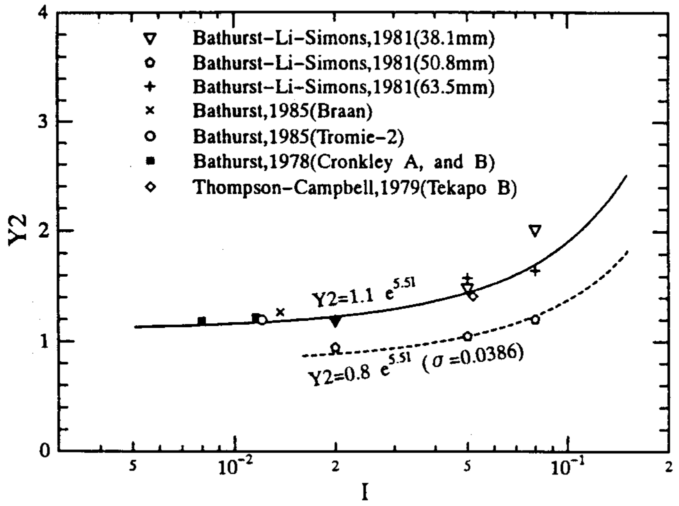


Figure 5: The relationship between *Y2* and *I*

Combination of Eqs. (3), (10) and (12) gives

(14)
$$b = c\, e^{-0.8\sigma}\, e^{5.5I}$$

In order to confirm the values of c in the wide range of σ/I, the relationship between c and σ/I was plotted in Figure 6. From the relationship between the natural logarithm and the exponential function, and from Eqs. (1), (7), and (14), the following equation was finally obtained.

(15)
$$Fr = 1.18\, I^{1/2} \exp\left[c\, e^{-0.8\sigma}\, e^{5.5I}\, (d/D_{84})\right]$$

A comparison of observed values of *Fr(obs)* and values of *Fr(cal)* calculated by Eq. (15) is shown in Figure 7. It was confirmed that the relationship obtained is very good, and that the calculated values do not exhibit trends away from the line of perfect agreement.

Derivation of Friction Factor Equation

The fundamental equation for flow resistance is

(16)
$$\sqrt{\frac{8}{f}} = \frac{U}{u_*} = \frac{Fr}{\sqrt{I}} = \frac{R^{1/6}}{n\sqrt{g}}$$

in which, n = Manning's roughnes coefficient, g = acceleration due to gravity, u_* = the shear velocity, defined by $u_* = (gRI)^{1/2}$. Substituting Eq. (15) in Eq. (16), the following equation is obtained.

(17)
$$\frac{U}{u_*} = \sqrt{\frac{8}{f}} = 1.18 \exp\left[c\, e^{-0.8\sigma}\, e^{5.5I}\, (d/D_{84})\right]$$

Then, the equation of friction factor, f, becomes

(18)
$$f = 5.75 \exp\left[-2\, c\, e^{-0.8\sigma}\, e^{5.5I}\, (d/D_{84})\right]$$

The standard deviations, σ_{18} and σ_{58} indicated in the note of Table 1 are defined as

$$\sigma_{18} = \frac{1}{2}\log\left(\frac{D_{84}}{D_{16}}\right) \tag{19}$$

$$\sigma_{58} = \log\left(\frac{D_{84}}{D_{50}}\right) \tag{20}$$

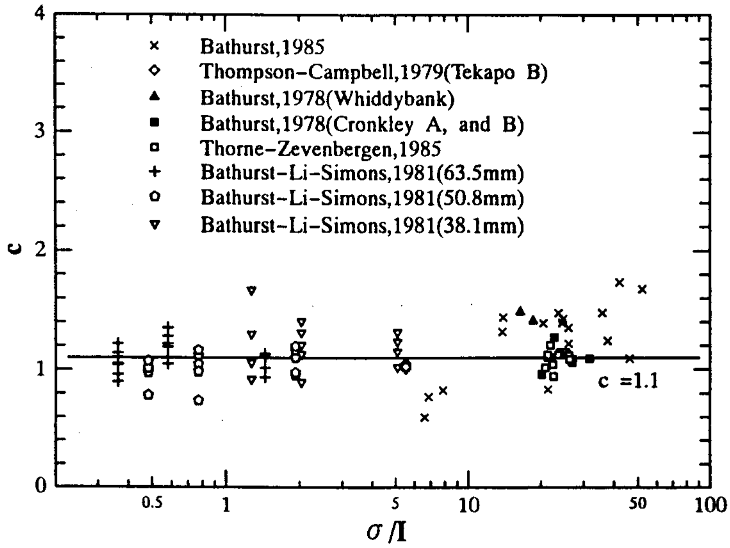


Figure 6: Distribution of proportional constant *c*

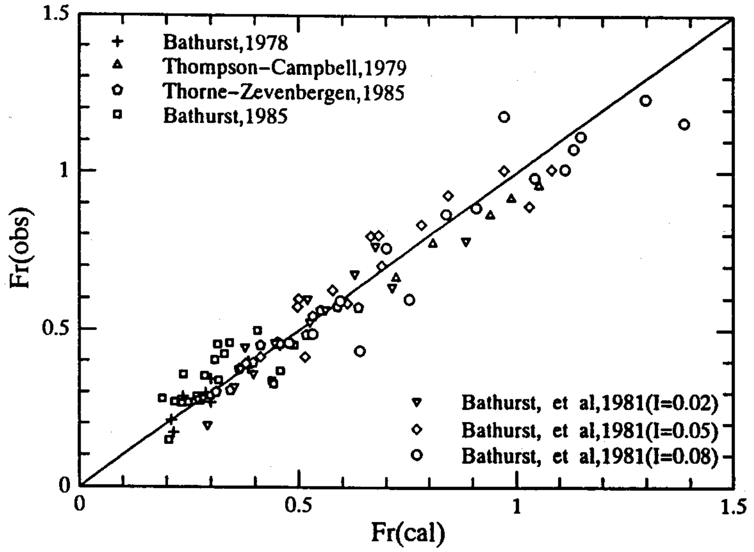


Figure 7: The relationship between observed values of *Fr(obs)* and predicted values of *Fr(cal)*

A comparison of observed values of $(8/f)^{1/2}$ and values predicted by Eq. (17), the equations of Bathurst, Li and Simons (1981), Thompson and Campbell (1979), Kellerhals (1970), and Hey (1979: small-scale roughness equation) is shown in Table 1, based on the river data of Bathurst (1978). In this limited test, Eq. (17) gives good predictions.

Table 1: Comparison of observed and predicted values of $(8/f)^{1/2}$ using data from Upper River Tees, UK (Bathurst, 1987)

Site (1)	Discharge in cubic meters per second (2)	Relative submergence R/D_{50} (3)	Relative submergence d/D_{84} (4)	Observed value of $(8/f)^{1/2}$ (5)	This equation Eq(17) (6)	Bathurst, Li and Simons (7)	Thompson and Campbell (8)	Kellerhals (9)	Hey (10)
Whiddy-bank*	0.90	0.73	0.452	2.18	1.80	1.83	0.61	1.54	0.77
	3.90	1.34	0.948	3.07	2.94	2.81	2.07	2.28	2.23
	7.20	1.77	1.169	3.50	3.55	3.34	2.83	2.68	2.92
Cronkley A	1.37	1.02	0.468	1.60	2.02	1.52	1.36	1.59	1.74
	4.00	1.36	0.888	2.50	2.80	2.16	2.10	2.13	2.47
	7.10	1.65	0.905	3.18	2.79	2.65	2.62	2.44	2.95
Cronkley B	1.10	1.09	0.739	2.35	2.36	1.89	1.53	2.08	2.19
	4.00	1.79	1.012	3.33	3.20	2.86	2.86	2.79	3.41
	7.10	2.17	1.331	4.22	4.27	3.38	3.39	3.23	3.88

* Note $\delta = \sigma_{58}/\sigma_{18} < 0.85$

Figure 8 shows the relationship between observed values of *U/u*(obs)* and prdicted values of *U/u*(cal)* by Eq. (17).

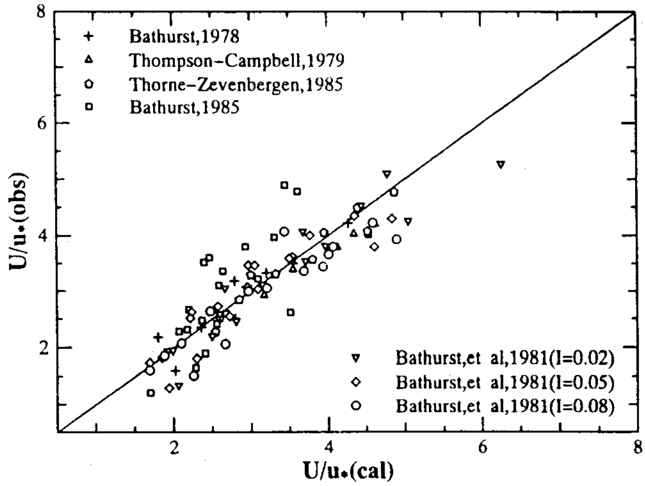


Figure 8: The relationship between observed value of *U/u*(obs)* and predicted values of *U/u*(cal)*

Derivation of Manning's Roughness Coefficient Equation

From Eqs. (16) and (17), the equation of Manning's roughness coefficient can be expressed as

$$n = 0.847 \frac{D_{84}}{\sqrt{g}} \left(\frac{R}{D_{84}} \right)^{1/6} \exp[-c\ e^{-0.8\sigma}\ e^{5.5I}\ (d/D_{84})] \tag{21}$$

A comparison of observed values of *n(obs)* of Manning's roughness coefficient and values of *n(cal)* predicted by Eq. (21) is shown in figure 9, based on river data of Thorne and Zevenbergen (1985).

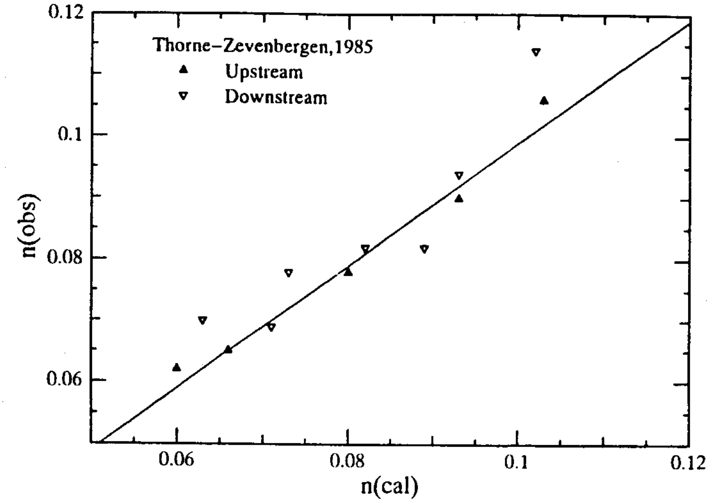


Figure 9: The relationship between observed values of *n(obs)* and predicted values of *n(cal)*

Derivation of Roughness Density Equation

Direct Observation of Roughness Density

Considering the case in which the river flow has boulders protruded on the water-surface for a given flow discharge, the total area, A_s, of boulders protruded on the water-surface, can be expressed by counting the number N of boulders protruded on the water-surface in the predetermined measuring area, A.

(22)
$$A_s = \sum_{i=1}^{n} \left(\frac{\pi D_i^2}{4} \right)$$

in which, i indicates ith boulder in the measuring area, A, and D_i = the diameter of portion protruded on the water-surface. The roughness density, R_d, defined as

(23)
$$R_d = \frac{A_s}{A}$$

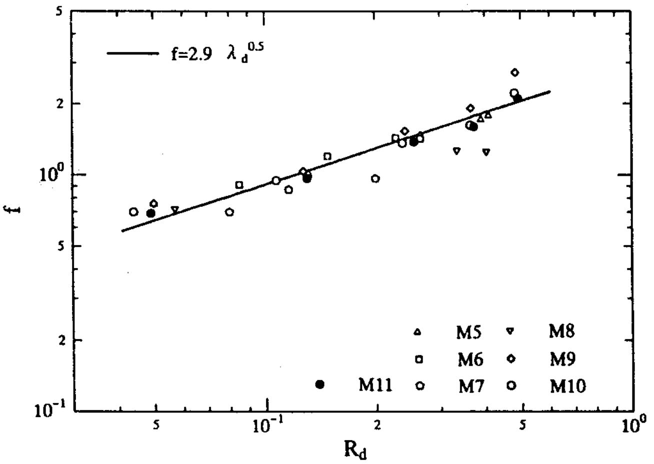


Figure 10: The relationship between the friction factor, *f*, and roughness density, R_d

Since the roughness density indicates an area ratio of the exposed roughnes elements to an area of bed A, R_d is a non-dimensional number ranging from zero to unity.

Derivation of Roughness Density Equation

From the experimental data (Miyamoto et al. 1995), the relationship between the friction factor, f, and roughness density. R_d, is obtained. Figure 10 shows this relationship.

(24)
$$f = 2.9\, R_d^{1/2}$$

Equating Eqs. (18) and (24), and rearranging, the equation for predicting the roughness density, R_d, is obtained.

(25)
$$R_d = 3.93 \exp[-\, 4\, c\, e^{-0.8\sigma}\, e^{5.5I}\, (d/D_{84})]$$

Derivation of Packing Coefficient

Roughness elements of observed data (M5~M11) on roughness density, R_d, by Miyamoto et al. (1995) are do not closely packed by three types of concrete sphere elements, so that a packing coefficient has to be introduced into Eq. (25). Table 2 shows bed-material data of M5~M11, in which λ_b = basal roughness concentration. The basal roughness concentration is defined as Eq. (26).

Table 2: Bed-material data

No	D_{50} (m)	D_{84} (m)	σ	λ_b
M5	0.13	0.20	0.187	0.409
M6	0.14	0.20	0.155	0.273
M7	0.14	0.20	0.155	0.204
M8	0.13	0.20	0.187	0.409
M9	0.08	0.16	0.301	0.498
M10	0.08	0.16	0.301	0.498
M11	0.08	0.16	0.301	0.498

Values of D_{50} and D_{84} were determined by total element numbers installed to the experimental channel-bed.

(26)
$$\lambda_b = \frac{\sum_{i=1}^{N} A_{D_i}}{A_{bed}}$$

in which, $\sum A_{Di}$ = total basal area of bed elements (spheres) installed on the bed, and A_{bed} = predetermined channel bed area (plan area). For natural rivers, the value of λ_b becomes unity. On the other hand, for naturally shaped channels, such as *Ramp Fishways* having boulders dipersedly installed, the value of λ_b takes a value in the range of $\lambda_b<1.0$. As shown in Table 3, the computaion of $Y3$ on observed data (M5~M11; Miyamoto et al.(1995)) were conducted for obtaining a packing coeficient, ϵ. Since $4c = 4.4$, values of $Y3$ were divided by 4.4. The relationship between the packing coeficient, ϵ, and the basal roughness concentration, λ_b, was derived from Table 3.

(27)
$$\varepsilon = 2.28\, e^{-0.82\lambda_b}$$

In Eq. (27), when the value of λ_b is unity, the packing coefficient becomes $\epsilon = 1.0$. Figure 11 shows this relationship.

Table 3: Relationship between basal roughness concentration, λ_b, and packing coefficient, ϵ

No	λ_b	Y3	ϵ=Y3/4.4
M5	0.409	7.266	1.651
M6	0.273	7.920	1.800
M7	0.204	8.471	1.925
M8	0.409	7.310	1.661
M9	0.498	6.518	1.481
M10	0.498	6.606	1.501
M11	0.498	6.720	1.527

Note: 4c=4.4

Introducing the packing coefficient, ϵ, into Eq. (25), the euation of R_d becomes

(28)

$$R_d = 3.93 \ \exp\left[- 4.4 \ \varepsilon \ e^{-0.8\sigma} \ e^{5.5I} \ (d/D_{84})\right]$$

In order to quantify the function of Eq. (28), the roughness density, R_d, was plotted against X. The variable X is denoted by Eq. (29).

(29)

$$X = \varepsilon \ e^{-0.8\sigma} \ e^{5.5I} \ (d/D_{84})$$

Plotted results are shown in Figure 12. Frm Eq. (29), Eq. (28) can be expressed as

(30)

$$R_d = 3.93 \ e^{-4.4X}$$

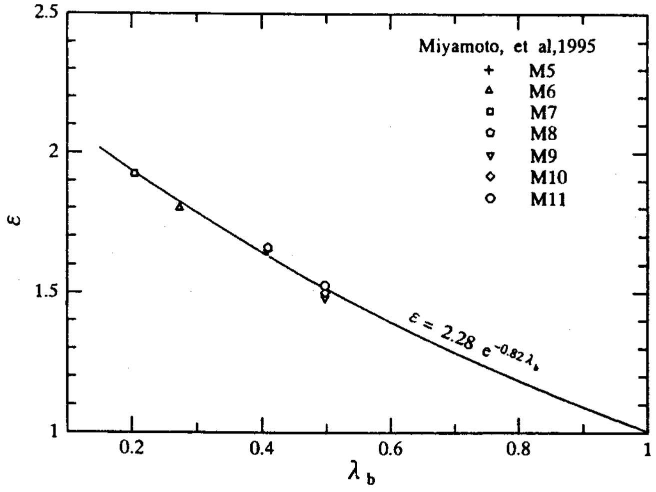


Figure 11: The relationship between the packing coefficient, ϵ, and the basal roughness concentration, λ_b

Also, the packing coefficient, ϵ, is introduced into Eqs. (17), (18), and (21). For example, Eq. (17) of U/u_* can be expressed as

(31)

$$\frac{U}{u_*} = \sqrt{\frac{8}{f}} = 1.18 \ \exp\left[1.1 \ \epsilon \ e^{-0.8\sigma} \ e^{5.5I} \ (d/D_{84})\right]$$

Generally speaking, it can be seen that the data points lie about one line for each run.

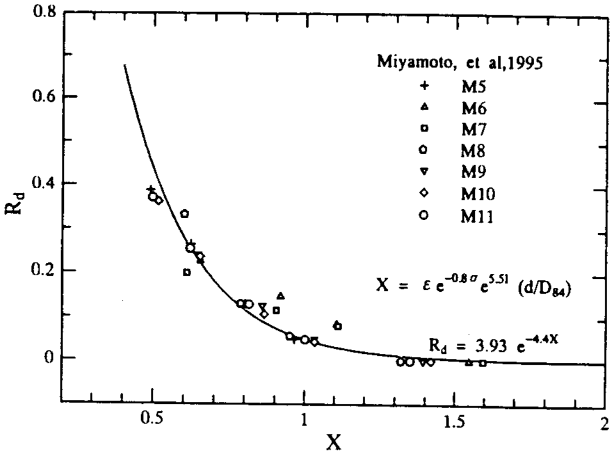


Figure 12: The relationship between the roughness density, R_d, and X

When pertinent data on R_d is supplied by field observation, the friction factor, f, would be predicted by the following equation. Equations for Fr and n also can be obtained from Eqs. (16) and (31).

(32) $$\frac{U}{u_*} = \sqrt{\frac{8}{f}} = 1.66\ R_d^{-1/4}$$

(33) $$Fr = 1.66\ I^{1/2} R_d^{-1/4}$$

(34) $$n = \frac{D_{84}^{1/6}}{1.66\sqrt{g}} \left(\frac{R}{D_{84}}\right)^{1/6} R_d^{1/4}$$

ADJUSTMENT METHOD OF D_{84}

When $\delta = \sigma_{58}/\sigma_{18} < 0.85$, it is needed to adjust the value of D_{84}. Tentatively obtained adjustment equation is .

(35) $$D_{84e} = D_{84}\ (1 + 1.4 \log\delta)$$

If the size distribution of sampling sediment data obeys theoretically to a log-normal distribution with a standard deviation of σ, the value of δ will be unity. So that, in this case Eq. (35) gives $D_{84e} = D_{84}$.

APPLICATIONS TO FISHWAYS

Using the research results of resistance to flow with boulder protruded bed, (1)design boulders protruded naturally-shaped fishway and (2) design ramp fishway.

(1) Design of Naturally-Shaped Fishway Design conditions are as follows: Fishway slope $I = 0.01$, design flow depth $d = 0.35m$, fishway width $B = 6m$, and the fishway bed should be packed by cobbles and boulders (standard deviation $\sigma = 0.15$, boulder size $D_{84} = 0.5m$, and basal roughness concentration $\lambda_b = 1.0$). Find the values of roughness density R_d, U/u_*, friction factor f, Manning's roughness Coefficient n, Froude number Fr, mean velosity U, and discharge Q.

Solutions:

From Eq. (27), $\epsilon = 2.28 \exp(-0.82\lambda_b) = 1.0$; Relative submergence $(d/D_{84}) = 0.7$, and hydraulic radius $R = 0.31m$.

From Eq. (28), $R_d = 3.93 \exp[-4.4 \times 1.0 \times e^{-0.8 \times 0.15} \times e^{5.5 \times 0.01} (0.7)] = 0.23$

From Eq. (31), $U/u_* = 1.18 \exp[1.1 \times 1.0 \times e^{-0.8 \times 0.15} \times e^{5.5 \times 0.01} (0.7)] = 2.39$

From Eq. (16), $Fr = (U/u_*) I^{0.5} = 0.24$, and $n = R^{1/6} / [g^{0.5} (U/u_*)] = 0.104$

Mean velocity $U = 2.39\ u_* = 0.42 m/sec$. This mean velocity ($U = 0.42$m/sec) is acceptable to fishways for *Ayu*. Finally, flow discharge $Q = A\ U = 0.882 m^3/sec$.

(2) Design of Ramp Fishway Design conditions are as follows: Fishway slope $I = 0.05$, design flow depth $d = 0.20m$, fishway width $B = 6m$, and fishway bed should be packed by cobbles and boulders with the basal roughness concentration $\lambda_b = 0.75$ (standard deviation $\sigma = 0.15$, and $D_{84} = 0.50m$). Find the values of ϵ, R_d, U/u_*, Fr, n, U, and Q.

Solutions:

From Eq. (27), $\epsilon = 2.28 \exp(-0.82 \lambda_b) = 1.23$, Relative submergence $(d/D_{84}) = 0.4$, and hydraulic radius $R = 0.19m$.

From Eq. (28), $R_d = 3.93 \exp[-4.4 \times 1.23 \times e^{-0.8 \times 0.15} \times e^{5.5 \times 0.05} (0.4)] = 0.31$

From Eq. (31), $U/u_* = 1.18 \exp[1.1 \times 1.23 \times e^{-0.8 \times 0.15} \times e^{5.5 \times 0.05} (0.4)] = 2.22$

From Eq. (16), $Fr = (U/u_*) I^{0.5} = 0.50$, and $n = R^{1/6} / [g^{0.5} (U/u_*)] = 0.11$

Mean velocity $U = 2.22\ u_* = 0.68 m/sec$, and finally, flow discharge $Q = AU = 0.816 m^3/sec$.

In the case in which the design discharge is given as a prerequisite condition, it should be noted that the trial and error method for obtaining a flow depth is needed.

CONCLUSIONS

The conclusions obtained from the results of this investigation are as follows:

1. Flow resistance equation for riffles in mountain rivers of large-scale roughness is newly developed.
2. It is found that the flow resistance equations is a function of water-surface slope, relative submergence, and standard deviation of bed-material distribution.
3. In order to test the applicability and reliability of newly developed resistance equation, published experimental and field data are used. Comparisons of observed values and predicted values show good applicability and reliability. However, further efforts should be made to produce a more reliable flow resistance equation, especially suitable for naturally-shaped fishway designs.
4. Expressions on the packing coefficient, ϵ, roughness density, R_d, and basal roughness concentration, λ_b, are presented.
5. Adjustment method of D_{84} for resistance computations is newly presented.

REFERENCES

(1) Bathurst, J. C. (1978). Flow resistance of large-scale roughness. *Journal of the Hydraulics Division*, ASCE, 104(HY12), pp. 1587-1603.

(2) Bathurst, J. C., R-M, Li and D. B. Simons (1981). Resistance equation for large-scale roughness. *Journal of the Hydraulics Division*, ASCE, 107(HY12), pp. 1593-1613.

(3) Bathurst, J. C. (1985). Flow resistance estimation in mountain rivers. *Journal of Hydraulic Engineering*, ASCE, 111(4), pp. 625-643.

(4) Hey, R. D. (1979). Flow resistance in gravel-bed rivers. *Journal of the Hydraulics Division*, ASCE, 105(HY4), pp. 365-379.

(5) Kellerhals, R. (1970). Runoff routing through steep natural streams. *Journal of the Hydraulics Division*, ASCE, 96(HY11), pp. 2201-2217.

(6) Komura, S., and K. Ozawa (1970). Bed-material sampling procedures and grain-size distributions for mountain stream. *Transactions of Japan Society of Civil Engineers*, 55(12), pp. 53-58. In Japanese.

(7) Miyamoto, S., M. Taki, M. Watanabe, and S. Komura (1995). Design method of boulders protruded slope fishways. *Proceedings of the International Symposium on Fishways in Gifu '95*, pp. 101-112.

(8) Thompson, S. M. and P. L. Campbell (1979). Hydraulics of a large channel paved with boulders. *Journal of the Hydraulic Research*, 17(4), pp. 341-354.

(9) Thorne, C. R., and L. W. Zevenbergen (1985). Estimating mean velocity in mountain rivers. *Journal of Hydraulic Engineering*, ASCE, 111(4), pp. 612-624.

(10) Wolman, M. G. (1954). A method of sampling coarse river-bed material. *Transactions*, American Geophysical Union, 35(6), Part 1, pp. 951-956.

FISH AND LAMPREY PASSAGE IN A COMBINED DENIL AND VERTICAL SLOT FISHWAY OF THE REGULATED RIVER KEMIJOKI

Anne Laine[1)], Riitta Kamula[2)] and Jussi Hooli[2)]
[1)] University of Oulu, Department of Biology
P.O.Box 333, FIN-90571 Oulu
[2)] University of Oulu, Hydraulic and Water Resources Engineering Laboratory
P.O.Box 191, FIN-90101 Oulu
Finland
E-mail: anne.laine@oulu.fi

ABSTRACT

The River Kemijoki, which runs from Northern Finland into the Baltic Sea, was one of the best salmon rivers in Europe. The construction of the Isohaara dam and hydropower plant in 1949 blocked the river. A fish lift was built at the dam a couple of years later but it did not work. Simultaneously with the enlargement of the power plant a new fishway was constructed in 1993. The total length of the fishway is 230 meters and its rise is 12.5 meters. There are two alternative entrances and both vertical slot and Denil sections in the fishway.

Almost a thousand adult salmonids passed through the fishway in 1993-95. Of them, salmon *(Salmo salar)* was most numerous. Almost all the salmon that entered the uppermost pool were young males weighing 1-2 kg. The trout *(S. trutta)* were on the average bigger with some weighing 6-7 kg. It took up to two weeks for the fish to swim from the tailrace into the fishway. The fish often visited the fishway several times before they finally passed through it. Of environmental factors water temperature, headwater level, i.e. fishway discharge, and discharge through the power plant explained most of the variation in numbers of salmon passing through the fishway. On trout numbers, they had a minor effect.

In addition to adult salmon and trout, rainbow trout (*Oncorhynchus mykiss*), several freshwater fish species and thousands of juvenile salmonids passed through the fishway. None of the migratory whitefish (*Coregonus lavaretus*) that entered the fishway were observed in its uppermost pool. Lamprey (*Lampetra fluviatilis)* entered the fishway through its vertical slot entrance. They were not able to get further than 2-3 pools above the sea level in the first study year, as slot velocities of 1.4 m/s exceeded their swimming capacity. After plastic bristles were fastened into the bottom of the slots, lamprey passed through the 120-meter-long vertical slot section. The Denil section in the upper part of the fishway is still an obstacle to them.

Numbers of salmon and trout that pass the Isohaara fishway are rather small mainly because of excessive fishing. As salmon, trout and whitefish have been planted in the estuary as juveniles and smolts, most of them may have no drive to pass the fishway. So far, the spawning and rearing habitats in the basins above the Isohaara dam are missing as well. A project dealing with possibilities to restrict fishing, move at least a part of the plantings above the dam, and improve fish habitats further in the river, is about to start.

KEY-WORDS: Combined fishway / Vertical slot / Denil / Fish passage / Atlantic salmon / Sea trout / Whitefish / Lamprey / Finland

INTRODUCTION

Overflow pool and weir fishways and vertical slot fishways are the most common fishways in Finland. Denil fishways and fish locks have been built nly recently,. Many of the coastal fishways do not operate well. Main reason for malfunction is lack of migrating fish entering the river or the river branch where the fishway is situated (Kamula *et al.*, 1992; Saura *et al.*, 1992; Mikkola and Saura, 1994). In some cases, the entrance of the fishway is not located properly (Saura *et al.*, 1992). As a whole, the reasons for fishway inefficiency are similar to other Nordic countries (Romundstad, 1990; Johlander and Sjöstrand, 1993).

The River Kemijoki was one of the most productive salmon rivers in Europe. The Isohaara dam and hydropower plant were completed close to the river mouth in 1949. This hindered the migration of Baltic salmon (*Salmo salar*), sea trout (*S. trutta m. trutta*), whitefish (*Coregonus lavaretus*) and lamprey (*Lampetra fluviatilis*) to their spawning areas further in the river. Thousands of salmonids were shoaling below the dam after its completion. Fish were transferred above the dam and a fish lift was constructed. In ten years, however, the anadromous fish stocks of the river were close to their extinction.

Nowadays, the River Kemijoki is totally harnessed for hydropower generation. To maintain the anadromous fish stocks of the river, salmon and trout smolts and whitefish juveniles are planted in the estuary and in the nearby sea area each year. The lamprey stock is maintained by transferring considerable numbers of them above the dam. In the basins, mainly lake trout (*Salmo trutta m. lacustris)* juveniles and catchable sized rainbow trout (*Oncorhynchus mykiss*) are planted. In spite of these actions, local people have strongly demanded a fishway at Isohaara. Simultaneously with the power plant enlargement, a fishway was finally built at the dam in 1993.

The Isohaara fishway is a combined fishway with vertical slot and Denil sections. The flow patterns of these fishway types differ drastically from each other. The aim of this paper is to evaluate this kind of fishway as a passage route for several migrating fish species. Also the importance of the fishway built in a large regulated river, with fish stocks that are mainly of hatchery origin, is assessed.

MATERIAL AND METHODS

Site And Structure Of The Fishway

The River Kemijoki, running from north into the Bothnian Bay of the Baltic Sea, is one of the largest rivers in Finland with a watershed area of 51,000 km^2 (Fig. 1). The mean flow of the 483-km-long river is 556 m^3/s and the mean flood flow is 3,373 m^3/s. The unbuilt western tributary, the River Ounasjoki, is preserved from water power construction. There are altogether 17 hydropower plants in the river system at the moment. The lowest, Isohaara hydropower plant, is situated 3.5 km from the river mouth. The distance between Isohaara and the dam above, Taivalkoski, is 18 km. Below the Isohaara dam, the river is almost 200 meters wide. The old power house is on the northern shore of the river. It has two machineries with vertical Kaplan turbines. The enlargement is on the southern shore and it has two machineries with horizontal Kaplan turbines. The spillways are situated between the power houses (Fig. 2).

The Isohaara fishway (Fig. 3) was constructed in connection with the power plant enlargement. Its total lenght is 230 meters and its rise is 12.5 meters. The fishway has two entrances which are used alternately. The vertical slot entrance, nearest to the shore, is in a tunnel. The so called Denil entrance begins as a Denil section and continues as a vertical slot section. After the junction pool, where the two entrances join together, the fishway continues as one vertical slot section for a distance of 65 meters. The 45-meter-long Denil section above it consists of five sloping channels and stilling basins between them. The uppermost part of the fishway is of vertical slot type, and it opens into an 18-meter-long exit tunnel.

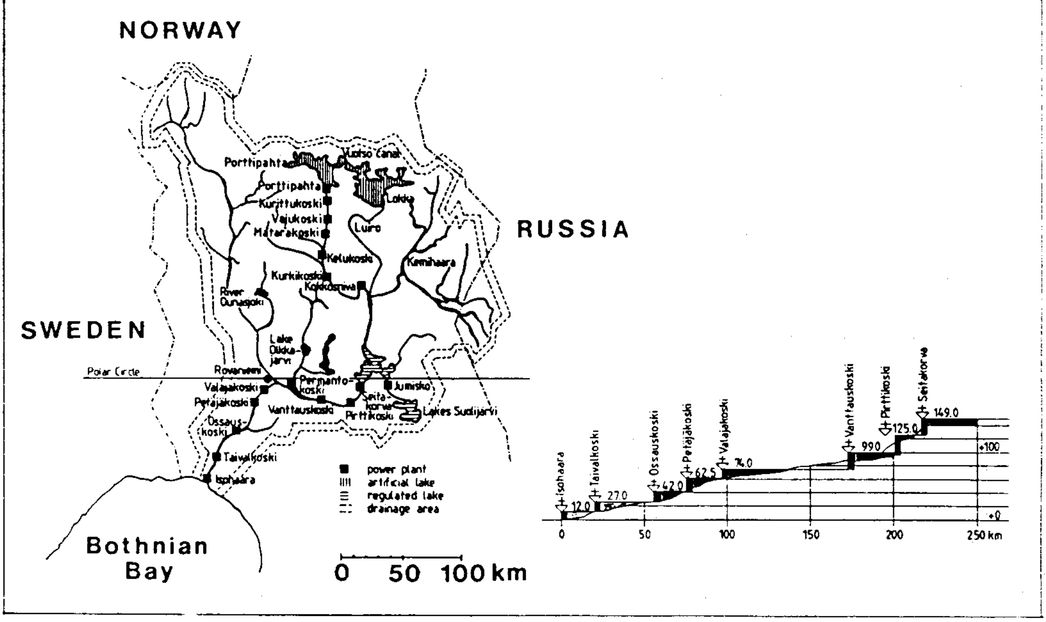


Figure 1: The watershed area and the profile of the River Kemijoki

In the vertical slot sections of the fishway, the pool width is 1.8 meters and the pool lenght is mainly 2.9 meters. The slope of the vertical slot sections is 7 %, with the exception of the vertical slot entrance where it is 4 %. The 0.3-meter-wide slots are in a straight line and they have no sills. The energy of the flowing water is dissipated with baffles that direct the water jet towards the center of the pool. The 0.15-meter-wide baffles are situated 0.3 meters below the slots.

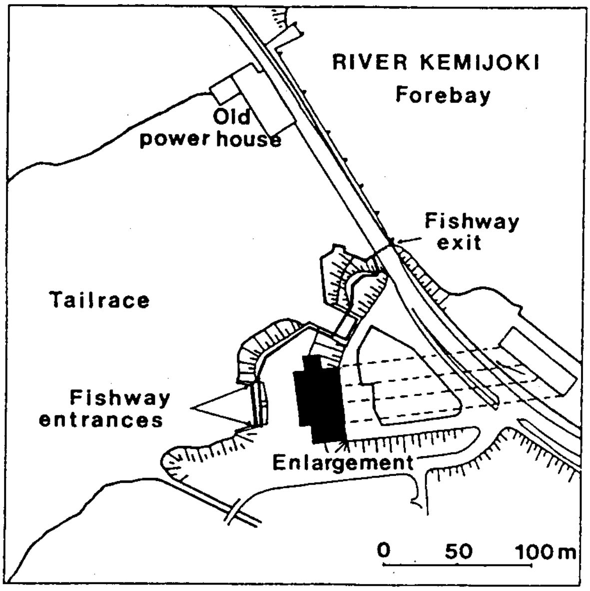


Figure 2: The location of the hydropower plant and the fishway at the Isohaara dam

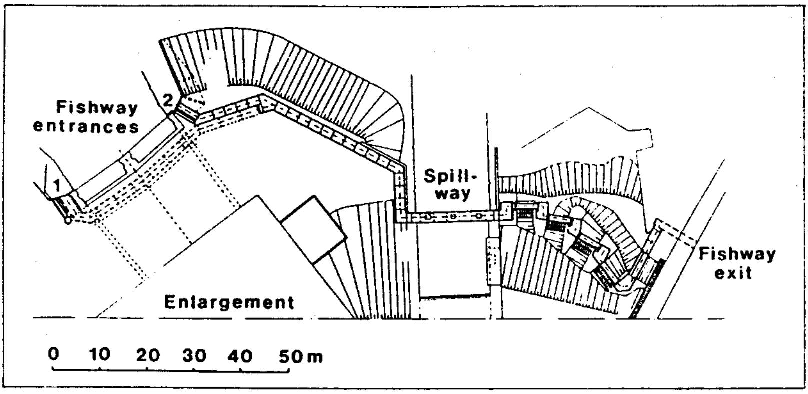


Figure 3: The layout of the Isohaara fishway (1 = vertical slot entrance, 2 = Denil entrance)

In the Denil sections of the fishway, the channel width is 1.0 meters and the opening is 0.5 meters. The energy of flowing water is dissipated with slotted baffles that have a V-shaped bottom. The angle between the baffles and the bottom of the channel is 45 degrees, and the baffle spacing is 0.6 meters. All the Denil channels in the upper part of the fishway were designed to be similar with a slope of 17 % and a length of 6.4 meters. Due to construction faults, the slopes are 16 % - 21 % and the lengths are 5.0 - 7.0 meters.

Physical Data

The fishway was opened in late May or early June, when water temperature reached 6-10°C during the research, in 1993-95. It was closed in middle October, when the water temperature decreased to 4-5°C. Data on water temperatures were collected from the waterworks reports. Comments on the weather and on the general operation of the fishway were recorded on site. Daily discharges of the machineries, discharge through the spillways, as well as head- and tailwater levels were collected from the power plant reports.

For studying the hydraulic operation of the fishway, water depths and water velocities in several depths were measured. In the Denil channels, measurements were made in the middle of the openings. In the vertical slot section, representative slots were selected. The measurements were made in the middle of the slots. For studying the flow pattern of pools, water velocities and depths were measured in several points in all pools whose shape differed from the general pattern, in a representative pool of the vertical slot section and in all stilling basins of the Denil section. In water velocity measurements, A. Ott's current meter was used.

Biological Data

The fishway was not designed to directly facilitate fish sampling. All the fish that passed through the fishway were gathered into the topmost pool of the upper vertical slot section. The fish were blocked from entering the upper channel by a mesh screen. Trap wires were installed in the downstream slot of the pool, to prevent the fish from escaping the pool.

The adult salmonids that entered the gathering pool were identified to species and their number was counted through a viewing window once or twice each day. Also an estimate of their weight was made. After counting the fish, the mesh screen was removed to let the fish swim upstream.. Adult salmon that entered the observation pool were marked by clipping off a piece of their caudal fin in September-October 1995. The numbers of smaller fish, e.g. cyprinids and juvenile salmonids, were only estimated as it was possible for them to escape from the pool through the mesh openings or through the trap wires. For catching lamprey in the fishway during their migration time in August-September, a special lamprey trap was used.

Two video cameras with water proof cases were used to observe fish and lamprey behavior at the fishway entrances and in the fishway. The cameras were mainly used below the vertical slot entrance, in the pools of the vertical slot section, and in the stilling basins of the Denil section. Excessive water velocities and air bubbles impeded video shootings in the Denil sections of the fishway. Occasionally, observations were made on fish numbers and on fish behavior in the tailrace.

HYDRAULIC FUNCTION OF THE FISHWAY

During the design of the fishway, scale model studies were performed to determine the pool dimensions and the location of slots and baffles in the vertical slot section. In the scale model, also the slope of the Denil channels and the dimensions and the spacing of baffles were determined. According to the results, the different sections of the fishway were dimensioned so that they work hydraulically well together.

In the vertical slots of the fishway, the water velocities and the depths in the pools corresponded to the velocities and depths in the model with an accuracy of ± 5 percent. Water velocities in vertical slots are approximately 1.4 m/s irrespective of the fishway discharge. The velocity is about the same in the whole height of the slot. The water depths in the pools are mainly 0.9-1.2 meters depending on the fishway discharge.

In the upper Denil section, the hydraulic function of the separate channels differed from each other. While the velocities of the fish swimming depths were less than 0.5 m/s in the three upper channels, they were 1-2 m/s in the two lowest ones (Fig. 4). In order to improve their hydraulic function, additional walls were installed in the stilling basins above the channels that operated poorly. Also an additional weir with a vertical slot was installed in the lowermost stilling basin, to increase its water depth. After these actions, the hydraulic function of the upper Denil section improved considerably (Fig. 4).

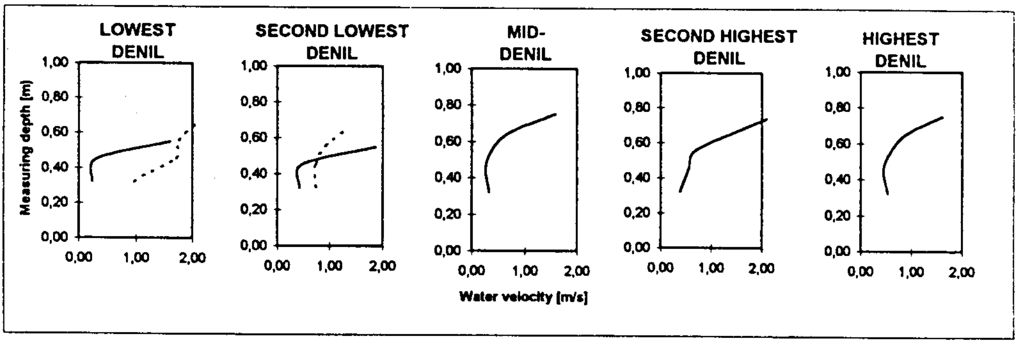


Figure 4: Water velocities in the Isohaara fishway Denil channels before (broken lines) and after (solid lines) structural improvements

Resting areas, with water velocities being less than 0.30 m/s, are found in almost every pool of the vertical slot section and in the stilling basins of the Denil section. Only the corner pool below the crossing of the spillway (see Fig. 3) lacks suitable resting areas. There the water velocities are greater than 0.40 m/s.

The discharge of the fishway depends on the headwater level. According to the duration curve, the headwater level is above NN+11.80 for more than 65 % of the time. The respective fishway discharge is 0.5 m^3/s. In addition to the fishway discharge, auxiliary water can be conducted into the vertical slot entrance. A pump, creating a flow of 0.4 m^3/s, is situated close to the entrance. Most of the auxiliary water is conducted into the second lowest pool through a perforated plate, which is placed close to the bottom of the pool. A part of the auxiliary water creates a small jet at the entrance.

The water velocities in the lowest slot of the vertical slot entrance are usually 0.7-0.9 m/s without and 1.0-1.2 m/s with auxiliary water. The tailwater level, however, affects the velocities. During the study, its fluctuation was more than a meter. The velocities of the Denil entrance were not measured owing to difficult working conditions there.

BIOLOGICAL FUNCTION OF THE FISHWAY

Fish Passage

Of the 984 adult salmonids that were counted in the gathering pool of the fishway salmon were most numerous (Table 1). Almost all the salmon that ascended the fishway were grilse having spent only one year at sea. Their weight was usually 1-2.5 kg while the largest salmon weighed 3-4 kg. The trout usually weighed 2.5-3 kg, largest of them being 6-7 kg. Approximately 40 % of the trout were brown trout that had been planted above the dam. The rainbow trout that ascended into the gathering pool of the fishway weighed 0.8-1.0 kg. They originated from yearly plantings in the basins above. The highest daily number of adult salmonids was found in September 1993, when 22 salmon, 2 trout and 1 rainbow trout had entered the gathering pool of the fishway.

Table 1: The numbers of adult salmonids in the gathering pool of the Isohaara fishway during the years 1993-95

	1993	1994	1995	Total
Salmon	304	121	163	588
Trout	97	101	115	313
Rainbow trout	55	8	20	83
Total	456	230	298	984

Based on observations through the window of the gathering pool, considerable numbers of roach (*Rutilus rutilus*) and dace (*Leuciscus leuciscus)* passed through the fishway each year. Also a few grayling (*Thymallus thymallus*), brook trout (*Salvelinus fontinalis*), perch (*Perca fluviatilis*) and altogether thousands of salmon and trout juveniles were observed in the gathering pool during the summer months. The length of the roach, the dace and the perch was 20-25 cm. The grayling were smaller, on the average 15 cm, and the brook trout were larger, on the average 25 cm. All the juvenile salmon (13-14 cm) had been planted downstream of the dam whereas juvenile trout (16-18 cm) had been planted both in the basins and below the dam.

There were doubts that some of the adult salmonids might have passed through the fishway more than just once. To confirm this, 46 salmon entering the gathering pool were marked in the fall of 1995 prior to their release upstream. Six of the marked salmon (13 %) were recaught in the gathering pool in a few days, after they had migrated downstream through the turbines.

After the spring flood, the yearly fluctuation of the headwater was 0.70-0.85 meters and that of the tailwater 1.2-1.4 meters. The average discharge through the enlargement (256 m^3/s in 1993 and 294 m^3/s in 1995) was slightly greater than that through the old power house (220 m^3/s and 209 m^3/s, respectively). The year 1994 was exceptional in this respect as, owing to reparations, the enlargement was closed from August 18th onwards. However, altogether 94 salmon and 8 trout entered the gathering pool of the fishway during the two months till the fishway was closed for the winter. The highest daily salmon number was seven, while that of trout was two.

Most of the trout and the rainbow trout passed through the fishway in July. The salmon, on the contrary, ascended mainly in September (Table 2). The first trout entered the gathering pool in June, when the water temperature exceeded 10°C while the first observations of salmon were made in July, when the water temperature was as high as 17-19°C. Each year, the peak water temperature (20-21°C) was recorded during the first week of August. The last salmon and trout passed the fishway in October, when the water temperature was approximately 5°C. The peak daily numbers of salmon were concentrated in considerably lower temperatures (8-11°C) than those of the trout (13-20°C, Fig. 5).

Table 2: Mean monthly water temperatures (±S.D) and monthly percentages of adult salmonids in the gathering pool of the fishway in 1993-95

Month	V	VI	VII	VIII	IX	X
Temperature °C	5.7 ± 2.1	13.1 ± 2.8	17.4 ± 1.6	17.9 ± 1.8	11.7 ± 2.3	6.3 ± 1.6
Salmon	0	0	5.1	31.8	50.2	12.9
Trout	0	19.2	39.3	16.6	16.0	8.9
Rainbow trout	1.2	9.6	39.8	20.5	26.5	2.4

A stepwise multiple regression was used for testing the effect of water temperature, headwater level, tailwater level and river flow at the dam on the salmon and trout numbers in the gathering pool of the fishway. Also the fishway entrance was included in the model as a binary factor. The test period was restricted to July 1st - September 30th when both salmon and trout were seen in the river and caught by nets. Before modelling, the salmon and trout frequencies were log(n+1) transformed.

For explaining the salmon frequencies, the analysis selected water temperature, river flow, and headwater level (Table 3). For the trout, only river flow was selected. Variation of these environmental factors explained 21 % of the variation in salmon numbers. For the trout, the corresponding percentage was as low as 11 %. Although tailwater level was omitted from the models it obviously has an optimal range of operation (Fig. 5).

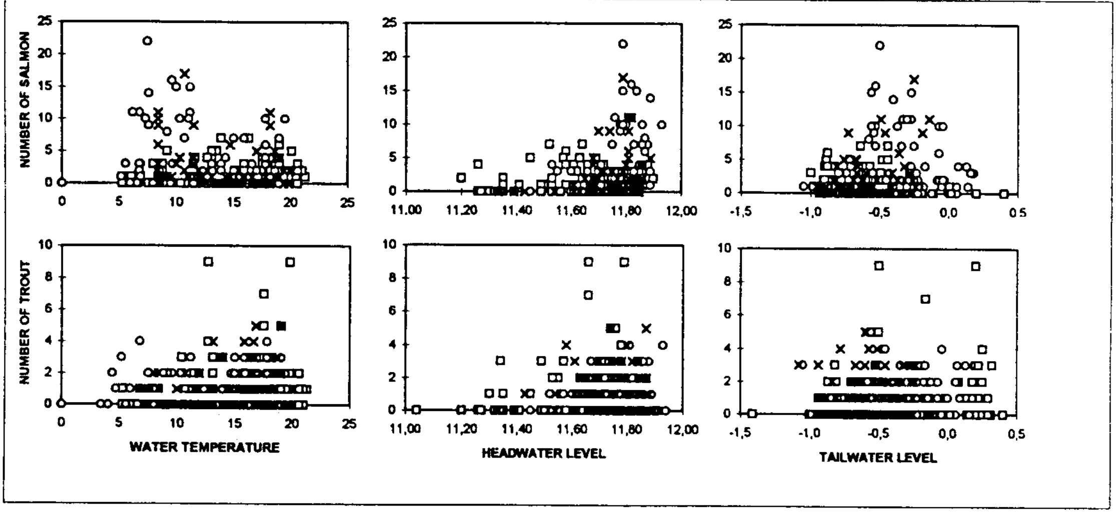


Figure 5: Numbers of salmon and trout plotted with water temperature (°C), headwater level and tailwater level at the Isohaara fishway in 1993-95 (circles = 1993, squares = 1994 and crosses = 1995)

Table 3: Variables entered in the model of the stepwise regression analysis (b = regression coefficient, SE = standard error) explaining salmon and trout frequencies in the gathering pool of the fishway

Variable	b	SE	t	p
Salmon				
River flow	-0.0020	0.0004	-5.577	0.0000
Water temperature	-0.0700	0.0130	-5.386	0.0000
Headwater level	1.6226	0.3254	4.986	0.0000
Intercept	-16.6078	3.7653	-4.411	0.0000
Trout				
River flow	0.0011	0.0003	4.584	0.0000
Intercept	0.0781	0.0856	0.913	0.3619

Fish And Lamprey Behavior

The first salmon and trout enter the River Kemijoki usually in early June. Soon they either reach the dam or are caught by nets below it. Salmon entered the River Kemijoki exceptionally late in 1993, owing to the prevailing north winds. It took more than two weeks until the first salmon ascended the fishway (July 21th). Trout had started to ascend the fishway more than a month earlier (June 19th). In 1994, the main flow was on the northern shore for two weeks in July and during this time salmon were caught below the old power house. After the main flow changed to the southern shore, below the enlargement, it took ten days till the first salmon entered the fishway (July 25th). Trout, on the contrary, had started to enter the fishway already two weeks after the first trout were caught at the river mouth (June 8th). In 1995, salmon and trout were observed close to the fishway entrances by the end of June. Simultaneously, thefirst trout passed through the fishway (June 23th). For the first salmon, the delay was almost a month (July 18th).

Downstream of the both power houses, salmon, trout and whitefish concentrated in the turbine flow which was closer to the shore. Of salmon only grilse were observed near the power houses. The fish swam closer to the outlet tunnels when the discharge was diminished and either further downstream or towards the boundaries of the main flow, i.e. closer to the fishway entrances, when the discharge was increased. Salmon and trout usually stayed in the fishway flow and in the boundaries of the turbine flows for several hours or for days. Occasionally they even visited the fishway. Three of the grilse that were swimming below the enlargement were identified individually in 1994. They spent two weeks below the powerhouse before they passed through the fishway.

According to small daily numbers of adult salmonids, practically no information was gained on the time that they use for passing through the fishway or through its different sections. Salmon were not observed resting in the fishway pools while trout and rainbow trout were occasionally seen there. Juveniles of salmon and trout, on the contrary, spent lots of time in the pools. Also cyprinids tended to gather there being most numerous in the lower parts of the fishway.

Whitefish usually enter the River Kemijoki in September and October. A few of the migrating whitefish were observed to swim into the fishway through the vertical slot entrance each year. None of them were, however, observed in the gathering pool or elsewhere in the fishway. Thousands of whitefish were migrating downstream from the Isohaara basin through the turbines and spillways in June 1995. Considerable numbers of them ascended up to the junction pool of the fishway. They, however, left the fishway in less than two days and continued their migration towards the sea.

Lamprey were not able to swim further than 2-3 pools above the junction pool in the first study year. They gathered behind the baffles and made several attempts to swim through the slots. The water velocity of 1.4 m/s was obviously too high for them in water temperatures of 13-14°C. In 1994, plastic bristles were installed on the bottom of the slots. Using them, lamprey were able to swim through the whole vertical slot section of the fishway. Also the water temperature was higher than in the previous fall, decreasing from 19°C to 14°C during the lamprey follow-up. Owing to the machinery reparations and thus lack of flow below the fishway entrances, the total number of lamprey was, however, small. When the vertical slot entrance was in operation, on the average 67 (±50) lamprey were caught in different parts of the vertical slot section during twelve nights. The highest nightly catch was 179, in one of the lowest pools. When the Denil entrance was in operation, the average catch in the pools above it was 2 (±3) lampreys during six nights. In the fall of 1995, it took three days for the lamprey to enter from the lower parts of the fishway into the lowermost stilling basin of the upper Denil section. Their nightly catch there averaged to 278 (±111) during three nights when the water temperature was 15-16°C. No lamprey were seen to enter the channels of the upper Denil section although several attempts were observed in its lowermost stilling basin.

DISCUSSION

In the unbuilt River Kemijoki, a part of the wild salmon started to migrate upstream already during spring floods. These salmon ascended to the upper reaches of the river system. The most important fishing grounds concentrated, however, on the lower course of the river, which also was the destination of the salmon that started to ascend later in summer and fall. The average annual salmon catch of the river was 375 tons (Vilkuna, 1975). Nowadays, the number of migrating fish below the Isohaara dam is greatly dependent on the amount and success of planting of juveniles and smolts in the estuary and in the sea. Fishing at sea and in some extent at the lower course of the river has a major effect on the number of migrating fish that can reach the Isohaara dam. From the peak year of 1990, when the salmon catch of the Baltic Sea was 2,000 tons and the trout catch 325 tons, they have decreased each year and were 1,100 and 125 tons in 1994. In proportion, also the numbers of salmon and trout entering the River Kemijoki have diminished.

When entering the River Kemijoki, salmon and trout use the whole width of the river (Jokikokko and Viitala, 1995). Having crossed the shallows, 500-750 meters below the Isohaara dam, they select the shore with a greater flow. In the Norwegian River Imsa, Jonsson *et al.* (1990) observed that sea-ranched salmon spent more time moving up- and downstream in the river than wild salmon. This type of migrating behavior was also observed in the River Kemijoki where most of the radiotagged fish were caught by nets while they were swimming around close to the shores (Jokikokko and Viitala, 1995). The greater vulnerability of hatchery reared salmon to fishing has been stated in the Swedish River Umeälven as well (McKinnell *et al.*, 1994).

Downstream of hydropower stations, the use of machineries has a major role in leading fish close to the fishway entrances where appropriate water velocities together with sufficient flow should attract them to enter the fishway. For Atlantic salmon, fishway discharges of 0.5-1.5 m^3/s are commonly used in Norway and Sweden (Grande, 1990; Laine *et al.*,1993). The River Kemijoki, with a mean discharge of 556 m^3/s and a width of 200 meters below the dam, is a large river and the fish may have problems in locating the fishway entrances with a maximal flow of 0.9 m^3/s. Adult salmonids spent plenty of time swimming below the fishway entrances. They also often visited the fishway several times before they finally passed through it. This may reveal either inefficiency of the entrance or reluctance of the fish to enter it. In British Columbia, several Pacific salmon species on their spawning run enter and ascend the vertical slot fishways without delay and showing no rejection, even those species that prefer the deepest parts of large rivers (Andrew, 1990). In Saskatchewan, on the contrary, of freshwater species white sucker (*Catostomus commersoni*) and northern pike (*Esox lucius*) stayed in the tailwater on the average 7.7 and 16.2 days, correspondingly, before ascending the Cowan Denil fishway (Katopodis *et al.*, 1991).

Almost a thousand adult salmonids passed through the Isohaara fishway in three years. The count obtained is probably an overestimate as at least some of the salmon migrated down from the basin and entered the gathering pool more than once. The adult salmon downstream migration from the Isohaara basin, however, concentrated into late September and early October, close to their spawning time (Jokikokko and Viitala, 1995). Almost all salmon that passed through the Isohaara fishway were grilse. The fishway has been suspected not to be suitable for larger salmon. As trout weighing 6-7 kg have passed through the fishway, it should not be too narrow for salmon of the same size either. Using video cameras, only grilse were observed close to the fishway entrances and the outlet tunnels of the power plant enlargement. Radiotelemetry studies revealed that the largest salmon usually stayed 30-60 meters below the outlet tunnels. They also often swam downstream for some hundred meters, even back to the estuary (Jokikokko and Viitala, 1995). Thus the motivation of old salmon, which as smolts have been released to the estuary, to migrate above the dam might be weaker than that of the grilse of the same origin.

In Norway, salmon and trout do not use the fishways when water temperature decreases to 6-7°C (Grande, 1990). At Isohaara, salmon and trout passed through the fishway in a wide range of temperatures from 5°C to 21°C. Salmon preferred lower temperatures than trout. Also high headwater level increased the salmon numbers indicating the importance of sufficient flow at the fishway entrance. Tailwater level, which fluctuates along with the seawater, did not explain salmon or trout numbers but it has an optimal range of operation. As salmon were not observed resting in the fishway pools and trout were seen there only occasionally, these fish most probably pass through the fishway rapidly and do not hesitate in the junction points of the different fishway types either. This indicates the proper function of these points. In addition to adult anadromous salmonids, also planted juveniles and fish of several freshwater species passed through the fishway. Unlike salmon and trout, they spent lots of time in the fishway pools.

None of the whitefish that entered the Isohaara fishway passed through it. In natural conditions, the whitefish that enter the rivers in fall, as in the River Kemijoki, spawn in the lower course of the river. The whitefish that have been planted at the estuary and at the sea area may also lack the motivation to pass the fishway. As whitefish are known to ascend in vertical slot and pool type fishways with low enough water velocities (Laine, 1990; Koponen, 1991) they should be able to pass through the vertical slot sections of the Isohaara fishway as well. For the so called 'fall whitefish', an additional factor to impede the ascent is the cold water in September-October. Whitefish may also be reluctant to use the Denil sections of the fishway. In some Denil fishways of Canada, lake whitefish (*Coregonus clupeaformis*) and cisco (*C. artedii*) have been either incidentally caught (Katopodis *et al.*, 1991) or their numbers have been much lower than predicted from their presentation in the catch downstream of the fishways (Schwalme *et al.*, 1985). Similarly, in Denmark only a few whitefish have been observed to use the Denil fishways (Lonnebjerg, 1990). In Sweden, however, whitefish ascend in several Denil fishways (Kamula *et al.*, 1994).

The lamprey, whose length is usually 25-32 cm, entered the Isohaara fishway merely through the vertical slot entrance. In the tunnel section the maximal slot velocities are 1.3 m/s. In the bottom of the slots, they may, however, be considerably less during high tailwater levels. The lamprey passed through this 60-meter-long tunnel without aiding structures. To be able to pass through the vertical slot section above, with a steeper slope and slot velocities of 1.4 m/s, they needed bristles which decreased the velocities at the bottom. In earlier experiments, lamprey were proven to swim through all 12 pools of a small experimental vertical slot fishway of Keminmaa, where maximal slot velocities were 1.1 m/s (Laine 1993). In the vertical slot fishway of the River Siikajoki, where slot velocities are higher than those at Isohaara, lamprey ascended 6-7 pools at the most (Kamula *et al.*, 1992). Generally, lamprey are not known to use the fishways that have been built for anadromous fish species (Ikonen, 1990). The Pacific lamprey (*Lampetra tridentata*), which is more than twice as long as lamprey, has been proven to pass through Denil fishways of several combinations of length and slope (Slatick and Basham, 1985). The longest tested channel was 20.1 meters and the slope 28.7%. At Isohaara, the lamprey did not enter the Denil channels of the fishway. Those specimens which were caught in the upper parts of the Denil entrance may have descended there from the junction pool.

Although whitefish and lamprey did not ascend the Isohaara fishway, adult salmonids, considerable numbers of salmon and trout juveniles and freshwater fish with a poorer swimming ability passed through it. This indicates the proper function of this kind of combined fishway with both vertical slot and Denil sections. Main reason for the relatively small numbers of salmon and trout that passed through the fishway is the excessive fishing at the sea, which threatens also the few wild salmon stocks of the unbuilt rivers of Finland. Additional problem of the built-up rivers, at the moment, is that salmon, trout and whitefish, are of hatchery origin and as smolts or juveniles they are usually planted at the estuaries and at the sea, not above the dams. Thus they may lack the motivation to use the fishways. Downstream of the dams, the so called terminal fishing is very effective and especially hatchery-reared fish are extremely vulnerable to it.

Constructing fishways and fish habitats in built-up and regulated rivers is insufficient for saving threatened anadromous fish species such as Baltic salmon, without simultaneously restricting fishing effectively. In the River Kemijoki, first steps are being taken to improve fish habitats and fish migration possibilities further in the river. In this connection, possibilities to restrict net fishing downstream of the dam and possibilities to move at least a part of the salmon and sea trout plantings above the dam are being considered.

REFERENCES

Andrew, F.J. (1990). The use of vertical-slot fishways in British Columbia, Canada. *Proc. Int. Symp. Fishways '90, Gifu, Japan, October 8-10,1990*, pp. 267-274.

Grande, R. (1990). Fish ladders in Norway. *Proc. Int. Symp. Fishways '90, Gifu, Japan, October 8-10,1990*, pp. 517-522.

Ikonen, E. (1990). Lamprey and fishways. *Vesi- ja ympäristöhallituksen monistesarja* 208 pp. 62-66. In Finnish.

Johlander, A. and P. Sjöstrand (1993). Fishways of southwestern Sweden. An overview. *Fiskeriverket, Utredningskontoret Jönköping. Projekt Fiskvägar. Delrapport* 1. 33 pp. In Swedish.

Jokikokko, E. and J. Viitala (1995). Radiotelemetry study on salmon at the the River Kemijoki in 1995. *Riistan ja kalantutkimus. Kala- ja riistaraportteja* 43. 11 pp. In Finnish.

Jonsson, B., N. Jonsson and L.P. Hansen (1990). Does juvenile experience affect migration and spawning of adult Atlantic salmon? *Behavioral Ecology and Sociobiology* 26(4) pp. 225-230.

Kamula, R., A. Laine, T. Pohjamo and J. Hooli (1992). The research in the Pöyry fishway of the River Siikajoki, Finland. Summary report. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory* A49. 57 pp. In Finnish with an English abstract.

Kamula, R., T. Pohjamo, and J. Hooli (1994). Fishways in Northern Sweden. Report on the excursion to Northern Sweden in 25.-27.8.1992. In: Kamula, R. (ed.), Travelling fishways. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory* A:59. 57 pp.

Katopodis, C., A.J. Derksen, and B.L. Christensen (1991). Assessment of two Denil fishways for passage of freshwater species. *Fish. Bioeng. Symp., Am. Fish. Soc. Symp.* 10. pp. 306-324.

Koponen, K. (1991). Study on fishways in the province of Kuopio, 1989-1990. *Kuopion kalastuspiirin tiedotus* 11. 55 pp.

Laine, A. (1990). The effects of a fishway model hydraulics on the ascend of vendace, whitefish and brown trout in Inari, northern Finland. *Aqua Fennica* 20(2) pp. 191-198.

Laine, A. (1993). The swimming behaviour of fish and factors affecting it in flowing water and fishways. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory.* A53. 74 pp. In Finnish with an English abstract.

Laine, A., R. Kamula and J. Hooli (1993). Fundamental concepts of fish passage in Scandinavian countries. *Fish Passage Policy and Technology. Proc. Symp. Bioeng. Sect. Am. Fish. Soc.* Portland, Oregon, September, 1993. pp: 81-85.

McKinnell, S., H. Lundqvist and H. Johansson (1994). Biological characteristics of the upstream migration of naturally and hatchery-reared Baltic salmon, Salmo salar L. *Aquaculture and Fisheries Management* 25 (suppl. 2) pp. 45-63.

Mikkola, J. and A. Saura (1994). Studies on migrating fish stocks of the River Vantaanjoki in 1987-1993. *Riista- ja kalatalouden tutkimuslaitos. Kalatutkimuksia* 84. 90 pp. In Finnish.

Romundstad, A.T. (1990). Norwegian experiences. *Finnish Water Power Association. Fishway Seminar, Karhula, January 16.-17.1990.*

Saura, A., J. Mikkola and E. Ikonen (1992). Studies on migrating fish stocks of the River Kymijoki in 1989-1991. *Riista- ja kalatalouden tutkimuslaitos. Kalatutkimuksia* 52. 60 pp. In Finnish.

Schwalme, K., W.C. Mackay and D. Lindner (1985). Suitability of vertical slot and Denil fishways for passing north-temperate, nonsalmonid fish. *Can J. Fish. Aquat. Sci.* 42 pp. 1815-1822.

Slatick, E. and L.R. Basham (1985). The effect of Denil fishway length on passage of some nonsalmonid fishes. *Mar. Fish. Rev.* 47(1) pp. 83-85.

Vilkuna, K. (1975). Lohi. 423 pp. In Finnish.

ACKNOWLEDGEMENTS

The authors wish to thank the Ministry of Agriculture and Forestry and the Academy of Finland for funding the studies and Jukka Viitala for his valuable help in the field work.

THE TIMING OF THE CYCLE IN BORLAND FISH LOCKS: OPTIMIZATION OF THE DURATION OF THE FISH PASSAGE PHASE

Jorge Bochechas
INSTITUTO FLORESTAL
Av. 5 de Outubro 52 - 6°D, 1050 LISBOA
PORTUGAL

ABSTRACT

The efficiency of a Borland fish lock is related with the ability to induce the fish to get into the reservoir as soon as possible. In general two main situations can occur. Many fish loiter in the upper chamber rather than pass directly to the reservoir due to the lack of attractiveness in the top sluice area. The duration of fish passage or discharge phase may not be adapted to the patterns of fish passage from the upper chamber to the reservoir. This results in general in a very long discharge phase, and its duration is not based on fish behavior and/or hydraulic parameters knowledge. This often results in a waste of time since during this period of time the fish lock is not "fishing". Two Borland fish locks were studied in order to know the distribution of fish passage to the reservoir during the discharge phase and its relation with water speed in the upper chamber and operation conditions of the top sluice. Monitoring with video cameras was done to count and identify all the fish and to register when within the discharge phase the fish got into the reservoir. Considering the species that used both fish locks, optimal duration for the discharge phase, operation conditions of the top sluice and water speed in the upper chamber were established. Modifications are suggested in order to improve the efficiency of both fish locks.

KEY-WORDS: Fishways / Borland fishpass / Fishlocks / Video monitoring / Fishlock cycle / Passage phase / Timing of the cycle.

INTRODUCTION

Between 1970 and 1986 six Borland type fish passes were installed in Portugal in dams built in rivers Douro and Tejo. The timing of the cycle, both the whole and the component phases, were based on the references for salmons from Scotland, and not based on the knowledge of the behaviour for the species present in both rivers.

Crestuma-Lever fish pass in river Douro and Belver fish pass in river Tejo were recently monitored with video equipment in order to know their level of efficiency and to identify the main causes for the alleged unsuccessful operation of both devices (Bochechas, 1995 and Bochechas, 1996).

It was found that the duration of the fish passage phase is not adapted to the patterns of fish passage from the upper chamber to the reservoir. Most of the fish leave the fish pass during the first minutes of the passage phase. in particular at Belver fish pass. This results in a waste of time as according with Travade and Larinier (1992a) during this period of time the fish pass in not "fishing" and any fish approaching the pass entrance meanwhile could leave the area without getting into the lower chamber.

GENERAL CHARACTERISTICS OF THE FISH PASSES

Both fish passes are classical Borland fish locks designed and made by the Scottish firm of Glenfield and Kennedy and are similar in design to those described by Clay (1961), Aitken *et al.* (1966) and Aitken (1980) for Scotland.

Crestuma-Lever fish pass comprises an upper and a lower chamber connected by a vertical shaft. There are two entrances at different levels that operate one or the other depending on the water level downstream. In general only the lower entrance operates for water levels downstream between 0.20m and 3.50m. At the channel connecting the upper chamber with the reservoir there are two windows equipped with glasses from where video observation of fish passage was made.

Belver fish pass comprises an upper and a lower chamber connected by a sloping shaft. There are also two entrances at different levels that operate one or the other depending on the water level downstream. As in Crestuma-Lever in general only the lower entrance operates for water levels downstream below 34.25m. Here there is no place suitable for easy observation of fish passage, so it was necessary to adapt a device to the upper gate allowing observation from above the water surface.

Both fish locks operate automatically with cycles in Crestuma-Lever having a duration of 3 hours (45 minutes for fish passage). Each cycle in Belver has a duration of 4 hours (40 minutes for fish passage).

During the passage phase the fishes are induced to swim into the reservoir by water which flows over the top sluice and is discharged through the bypass pipes out of the lower chamber. The flow during this period is dependent of the water levels upstream and downstream, as it is the water speed over the top sluice threshold.

The top sluices are similar in both fish passes. They are 1.5m wide, the same width of the channel connecting the upper chambers with the reservoirs.

At Crestuma-Lever during the passage phase the top sluice opens totally and its threshold is positioned at the same level of the channel bottom. On the other hand Belver top sluice is set to give an overfall of 0.16 to 0.22m, and the fishes are obliged to pass over the sluice within 0.50m from the water surface. This is a modification introduced during the present work. Before this study both fish passes operate as described for Crestuma-Lever.

METHODS AND EQUIPMENT

Crestuma-Lever and Belver fish passes were monitored continuously with video cameras (Panasonic, model WV-BP100, Matsushita Electric Industrial Co.,Ltd., Japan) and the passage of fish registered with videotape recorders (Panasonic, model AG-6720A, Matsushita Electric Industrial Co.,Ltd., Japan). At Crestuma-Lever the monitoring was made through the window installed in the upstream channel. The fishes were forced to pass within 0.50m from the glass, with a special device, as referred in Travade and Larinier (1992b), to allow observation since the channel is 1.50m wide.

At Belver as there is no window for observation the video camera was installed above the water surface. The fishes were filmed passing over a white board installed on the uppergate threshold as suggested by Travade and Larinier (1992b).

In both cases, the fishes were counted and identified, and the time of passage within the passage phase was registered.

RESULTS

Data from Crestuma-Lever were collected from 25 October 1994 to 17 August 1995 and during this period of time 714 cycles of the Borland fish pass were monitored. Only in 308 cycles (43%) the fish pass was utilised by at least one fish. A total of 109608 fish utilised the fish pass with an average of 356 fish per cycle (minimum of 1 fish and a maximum of 11397 fish per cycle). The fish pass was utilised mainly by *Mugilidae* (*Lisa ramada* or *Mugil cephalus*) which represents 90% of the fish passage (98804 individuals), and by eels (10729), barbels (35), largemouth blackbass (36) and Iberian nase (4).

The distribution of fish passage within the passage phase expressed in average percentage of fish present in the upper chamber in each cycle is shown in figure 1. Twenty minutes after the beginning of the passage phase 90% in average of the fish in each cycle have got into the reservoir.

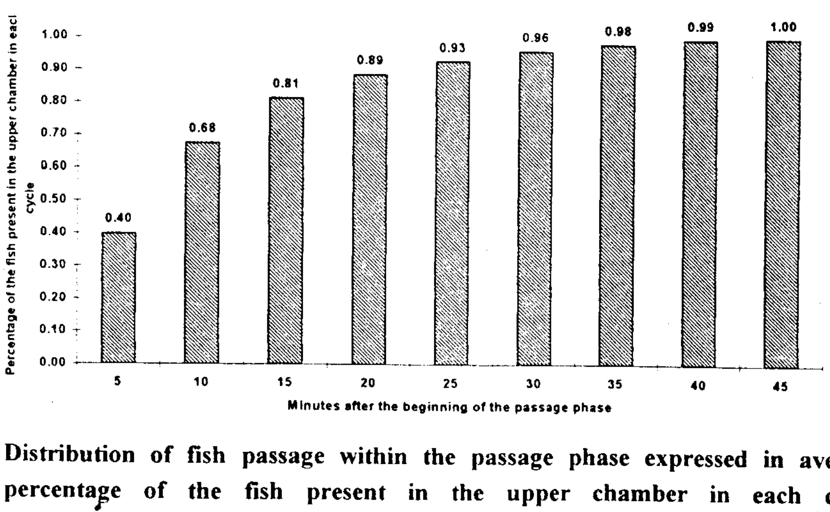


Figure 1: **Distribution of fish passage within the passage phase expressed in average percentage of the fish present in the upper chamber in each cycle. Crestuma-Lever fish pass.**

The patterns of fish passage within the passage phase are variable from cycle to cycle. In 10% of the cycles all the fishes leave the fish pass during the first 5 minutes of the passage phase. In 57% of the cycles all the fish in the cycle got to the reservoir after 30 minutes of the passage phase and in 88% of the cycles 90% of the fish passed during that period of time.

Although there is an evident good correlation between the number of fish in the upper chamber in each cycle and the number of fish that got into the reservoir during the first five minutes of the passage phase (ρ=0.96), no correlation was found between the number of fish in the upper chamber in each cycle and the percentage of these that got into the reservoir during the first five minutes of the passage phase (ρ=0.09). This suggests that the number of fish present in each cycle doesn't influence the speed of passage.

The water speed over the top sluice and in the channel connecting the fish pass with the reservoir ranged between 0.26 and 0.42ms^{-1} depending on the water levels upstream and downstream. This factor seems to have great influence in fish passage from the upper chamber to the reservoir. As shown in figure 2, higher water speeds over the top sluice are related with higher percentage of fish passed during the first 5 minutes of the passage phase (correlation coefficient ρ=0.82). After 30 minutes most of the fish have got into the reservoir, in particular when the water speed was higher than 0.37 ms^{-1} (figure 3).

No difference in the patterns of passage was found between the two main species (mugilids and eels) that passed by this fish pass.

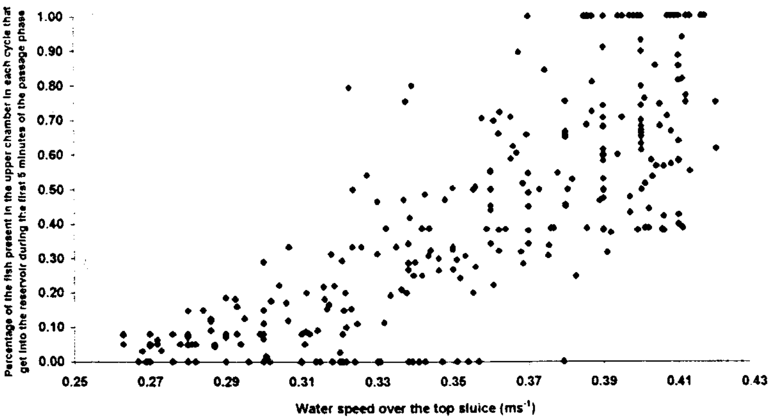


Figure 2: Variation of the percentage of the fish in the upper chamber in each cycle that got into the reservoir during the first 5 minutes of the passage phase with the water speed over the top sluice. Crestuma-Lever fish pass.

Data from Belver fish pass were collected from 5 January 1995 to 21 May 1995 and during this period of time 477 cycles of the fish pass were monitored. Only in 188 cycles (39%) the fish pass was utilised by at least one fish. A total of 15324 fish utilised the fish pass with an average of 81 fish per cycle (minimum of 1 fish and a maximum of 501 fish per cycle). The fish pass was utilised mainly by the Iberian nase (*Chondrostoma polylepis*) which represented 72% of the fish passage (10995 individuals), and by *Mugilidae* (2223), eels (1493) and 26 shads (*Alosa fallax*). It was not possible to identify 586 fish.

The distribution of fish passage within the passage phase expressed in average percentage of fish present in the upper chamber in each cycle is shown in figure 4. Ten minutes after the beginning of the passage phase 98% in average of the fish in each cycle have got into the reservoir.

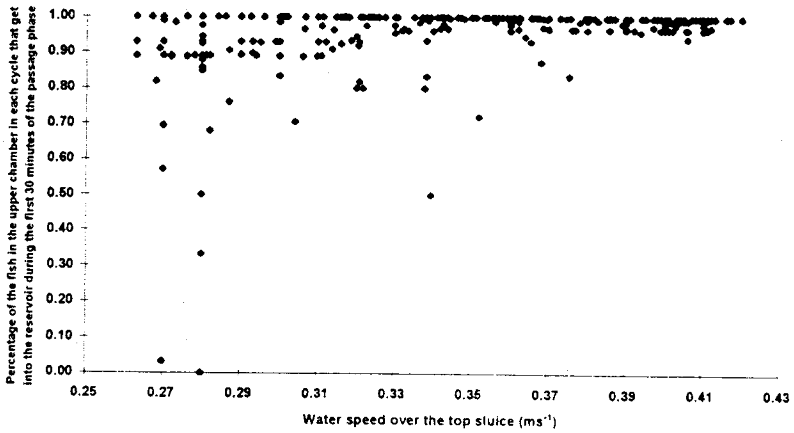


Figure 3 - **Variation of the percentage of the fish in the upper chamber in each cycle that got into the reservoir during the first 30 minutes of the passage phase with the water speed over the top sluice. Crestuma-Lever fish pass.**

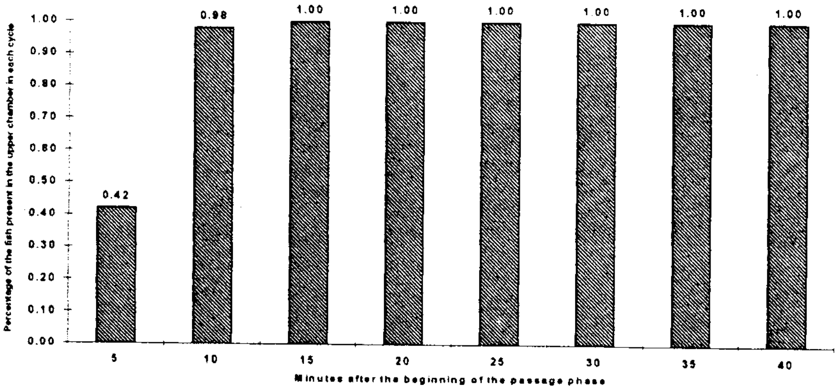


Figure 4: **Distribution of fish passage within the passage phase expressed in average percentage of the fish present in the upper chamber in each cycle. Belver fish pass.**

Like in Crestuma-Lever the patterns of fish passage within the passage phase in Belver are variable from cycle to cycle. But in this case the fish tend to get into the reservoir much more quickly. In 30 and 88% of the cycles all the fish leave the fish pass during the first 5 or 10 minutes of the passage phase respectively. In no cycle any fish was observed during the passage phase after 20 minutes.

The water speed over the top sluice ranged between 1.54 and 2.08 ms^{-1} depending on the water levels upstream and downstream. Although the variation within this range of water velocities seems to have no direct influence in the patterns of fish passage it was observed that when the water velocity over the top sluice is higher than

$1.95ms^{-1}$ a great number of fish make more than two tentative before they attempt to get into the reservoir.

The correlation coefficient between the water velocity over the top sluice and the percentage of the fish in each cycle that got into the reservoir during the first 5 minutes of the passage phase was equal to -0.04.

No difference in the patterns of passage was found between the two main species (mugilids and iberian nase) that passed by Belver fish pass.

DISCUSSION

According to Travade and Larinier (1992a) the optimal characteristics of the cycle in a Borland fish pass are related with the species present in each case. It is necessary to find a timing for the cycle which will have in consideration the behaviour of the species.

The results suggest that an increase in the water speed over the top sluice in Crestuma-Lever fish pass contribute to encourage the fish to leave the fish pass faster. But even when considering only water speeds over the top sluice higher than $0.38ms^{-1}$, the speed of fish passage into the reservoir is lower than in Belver. Only after 25 minutes of the passage phase in average 98% of the fish got into the reservoir (considering only the cycles with water speed over the top sluice higher than $0.38ms^{-1}$). In Belver this happens after 10 minutes.

As mentioned before, in no cycle in Belver fish pass any fish was observed during the passage phase after 20 minutes. This suggests a good efficiency in the passage phase and that the top sluice should operate in the future as it has been operating during the present work. The duration of the passage phase could be modified to 20 minutes or even 15 minutes as an average of 100% of fish got into the reservoir during this period of time.

On the other hand in Crestuma-Lever the passage of the fish into the reservoir should be tested with the top sluice operating as it was in Belver during the present work in order to find the best duration for the passage phase.

The modification in the operation in the top sluice in Belver allowed a reduction in 20 minutes in the duration of the cycle and similar results are expected in Crestuma-Lever.

The duration of the emptying phase can also be reduced in both fish passes with simple modifications. With the reduction in the duration of the passage and emptying phases it will be possible to reduce the non fishing periods in these Borland fish passes. This reduction will be an important contribution to increase the efficiency of the fish passes.

REFERENCES

Aitken, P.L. (1980). Dams and salmon in Scotland. *Water Power & Dam Construction*, Oct. 1980.

Aitken, P.L., Dickerson, L.H. and Menzies, W.J.M. (1966). Fish passes and screens at water power works. *Proc.Inst. Civ. Eng.*, 35:29-57.

Bochechas, J. (1995). Preliminary data on Borland fish passes efficiency for non-salmonids in two Portuguese large rivers. *Proceedings of the International Symposium on Fishways '95, Gifu (Japan)*, October 24-26, 1995.

Bochechas, J. (1996). Condições de funcionamento e de eficácia de eclusas para peixes: casos das barragens de Crestuma-Lever e de Belver. *Dissertação para obtenção do grau de mestre em Hidráulica e Recursos Hídricos, IST, Lisboa.*

Clay, C.H. (1961). Design of fishways and other fish facilities. *Department of Fisheries of Canada (Publ.), Otawa,* 301p.

Travade, F. and Larinier, M. (1992) Ecluses et ascenseurs à poissons. *Bull. Fr. Pêche Piscic.* 326-327: 95 -110.

Travade, F. and Larinier, M. (1992) Les techniques de contrôle des passes à poissons. *Bull. Fr. Pêche Piscic.* 326-327: 151 -164.

SMOLT BEHAVIOR AND DOWNSTREAM FISH BYPASS EFFICIENCY AT SMALL HYDROELECTRIC PLANTS IN FRANCE

Larinier* M., Travade** F.
*CSP-CEMAGREF
GHAAPPE - Institut de Mécanique des Fluides -
Avenue du Professeur Camille Soula 31400 TOULOUSE
France
**EDF-DER.
6 Quai Watier - 78401 CHATOU CEDEX
France

ABSTRACT

Surface bypasses can be an efficient solution to problems of downstream passage at small scale hydroelectric plants. Experiments were conducted for three years at a powerplant on the Aspe river to relate bypass efficiency to hydrodynamic conditions and fish behaviour in the canal intake.

Bypass efficiency in 1992 and 1993, evaluated by mark-recapture technique, was between 20% and 40%. Improving the hydrodynamic conditions (flow pattern, discharge) at and near the entrance of the bypass resulted in a significant increase in the efficiency of the bypass to between 50% and 80% in 1994. Radiotelemetry was used to monitor movement patterns of salmon and sea trout smolts in the powerhouse canal. The measured efficiencies of the bypass obtained with radiotracked smolts were consistent with those obtained with unmarked smolts. Behavior of fish in the vicinity of the trashrack and the bypass seemed to be largely influenced by the flow pattern. Poor hydrodynamic conditions (turbulence, strong acceleration, upwelling) were indentified through direct and video observations as beeing responsible for many aborted passages at the bypass entrance.

The results suggest the siting of such bypasses must be determined by carefully examining the flow pattern in the trashrack area and actual velocity conditions in the canal. Satisfactory entrance conditions at the bypass, as well as an appropriate flow discharge, are important in order to lead the arriving smolts into the attraction zone of the downstream bypass. It is suggested that surface bypasses can be occasionally a valuable way of diverting downstream migrants at small hydroelectric plants, taking advantage of the natural reluctance of fish to pass through conventional turbine protection screens.

KEY-WORDS : downstream migration/ small-scale powerplant/ surface bypass/ radiotelemetry/ smolt/ salmon/ sea trout/ efficiency/.

1. INTRODUCTION

Over the past fifteen years, programs have been launched on a number of French rivers to restore stocks of certain migratory fish species (salmon, sea trout, lamprey, etc.). Initially, these programs involved building fishpasses. More recently, attention has been given to problems of downstream migration of juveniles. The principal obstacles to downstream migration on rivers are small-scale hydroelectric power stations, and passage of juveniles through the turbines can lead to serious damage. Attemps are now being made to retrofit existing facilities to reduce these damages. It appeared unrealistic, however, to envisage systematic installation of physical barriers, such as the fine-mesh screens found in Scotland and the United States (Ruggles, 1980 ; EPRI, 1986 ; Clay, 1995) which would require resizing of most water intakes. For this reason, the first solution adopted was surface bypasses, making use of the reluctance on the part of fish to cross conventional turbine trashracks (Ruggles, 1992 ; EPRI, 1994). Experiments were undertaken to assess the efficiency of such bypasses and to optimize their positioning and sizing, with the long-term objective of defining optimum design criteria and determining the limits to their use. Following studies conducted at the Halsou power station and at the Poutès dam (Bomassi and Travade, 1985; Larinier and Boyer-Bemard, 1991a ; Larinier and Boyer-Bemard, 1991b), an experiment was launched at the Soeix plant on the Aspe river in 1992, with a view to optimizing the use of a downstream bypass and establishing the relationship between its efficiency and certain hydraulic parameters and patterns of fish behavior (Larinier and al., 1993 ; Larinier et al., 1994 ; Travade et al., 1996). In addition to the basic experiments, it was clear that there was a need to solve the problem of downstream migration on this site correctly before attempting to progress further with restocking programs upstream of the dam.

2. EXPERIMENTAL SITES

The Aspe river in southwest France is a salmon-bearing river with a drainage area of some 580 km2 and a pluvio-nival regime: high waters in winter and spring, low water from July to October. Mean annual discharge is on the order of 24 m^3/s on the study site. Mean monthly discharge in the period corresponding to downstream migration of smolts (March, April and May) is respectively around 33 m^3/s, 43 m^3/s and 54 m^3/s. The Soeix hydroelectric installation (Figure 1), some 100 km from the sea, consists in a gravity dam 28 meters long and 7 meters high which supplies the plant through a 450-m headrace canal; the plant has one vertical Kaplan wheel (nominal discharge of 24.5 m^3/s) and one propeller wheel (nominal discharge of 10.3 m^3/s). Head at the plant is around 9.5 m. The width of the headrace canal varies from 6 to 10 m, and its depth is close to 3.5 m. The water intake at the plant is some 15 meters wide and is fitted with a trashrack (bars 3.5 cm apart) with a section of some 50 m^2.

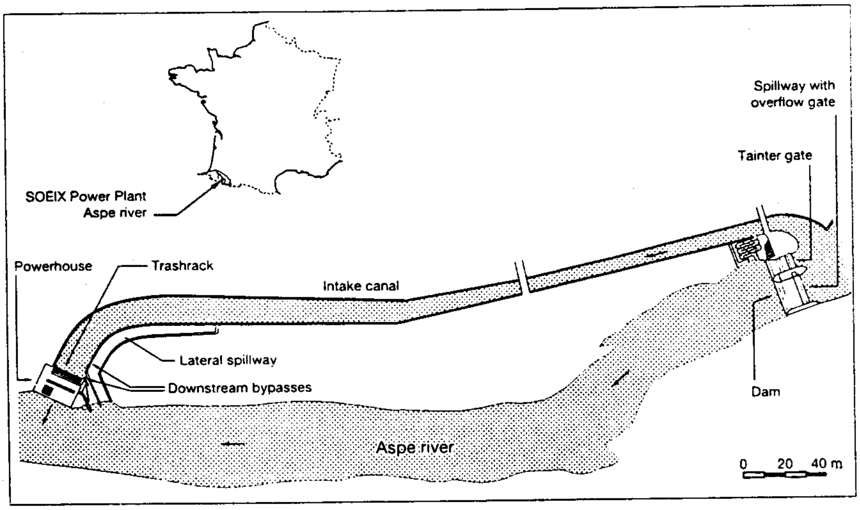


Figure 1 : Plan view showing the hydroelectric plant : dam, intake canal and powerhouse, location of downstream migration bypasses.

The dam is equipped (Figure 1) with one spillway with an overflow gate on the left bank, 15 m long and 1.73 m high, and one tainter gate 7 m long and 7 m high on the right bank. A lateral spillway some fifty meters long is located on the left bank immediately upstream of the power station. Spillage is relatively frequent during downstream migration period.

Flow velocities in the headrace are high (1.8 to 2 m/s) and their distribution throughout the sections of the canal relatively uniform. In the vicinity of the intake, the flow becomes dissymmetrical due to the broadening and change in direction of the headrace and to the uneven discharge from the two turbines. It is characterized by a tangential current from the right to left bank and by the presence on the left bank of a recirculation area covering some thirty meters in length and 2 to 3 meters in width upstream of the trashrack (Figure 2). In this area, upwellings were pointed up by fluorescein. Under the effect of this flow pattern, spherical buoys released in the headrace concentrate rapidly on the left bank, rolling along the trashrack. When both turbines are operating (discharge around 28 m^3/s), mean velocity measured just below the surface in front of the trashracks (between 0 and 0.5 m in depth) is between 0.3 and 0.9 m/s,the lowest velocity being located on the left bank.

3. DOWNSTREAM MIGRATION DEVICES TESTED

Three downstream migration bypasses were successively set up and tested (Figure 2) :

- in 1992, the bypass was located on the left bank of the headrace, 6 m upstream of the screens. It consisted in a 1x1-meter opening fitted with a flap gate maneuvered by a hand pulley. Mean discharge through the bypass was some 0.4 m^3/s, varying from 0.2 m^3/s to a maximum of 0.6 m^3/s (Figure 3).

- in 1993, the trash gate on the left bank immediately upstream of the screens (1.5 m) was turned into a bypass by dividing it into two sections 90 cm wide, only one of which was used. Mean discharge remained close to the 1992 value of 0.4 m^3/s, varying from 0.2 m^3/s to a maximum of 0.75 m^3/s.

- in 1994, discharge was increased significantly (mean of 1.2 m^3/s) by opening the two sections of the trash flap gate (width = 1.8 m); it varied between 0.4 m^3/s and a maximum of 2.2 m^3/s. A horizontal metal plate (Figure 2) was positioned upstream of the bypass, 25 cm below its axis so as to limit the effect of upwellings and to increase surface velocities near the bypass entrance.

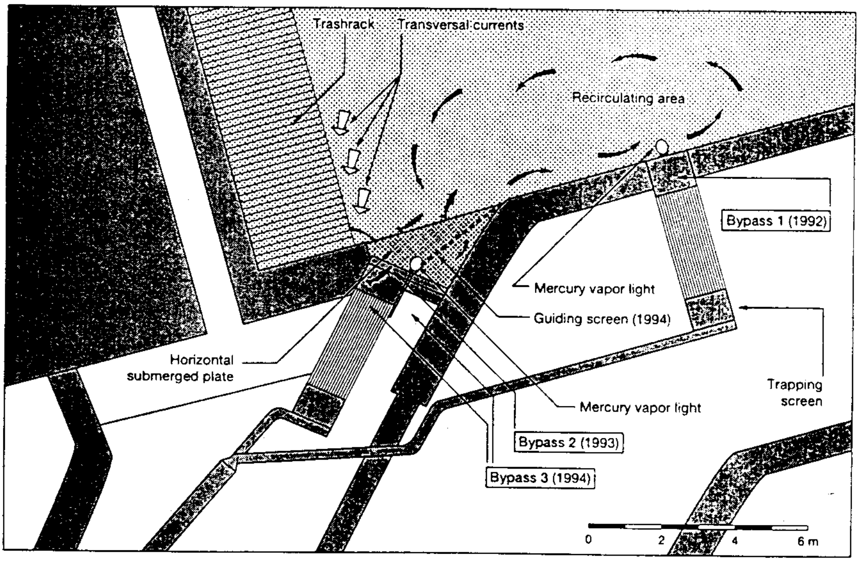


Figure 2 : Detail of trashrack area on the left bank, location of dowstream migration bypasses tested in 1992, 1993 and 1994. Schematic current patterns near the bypasses.

Flow in the immediate vicinity of the bypasses was characterized in 1992 and 1993 by upwellings resulting from the overall pattern of currents in the intake zone; the area within which the hydraulic conditions tend to draw fish toward the bypass was very small (less than 1 m upstream of the bypass). In 1994, under the effect of increased discharge and positioning of the metal plate, the surface velocities were perceptible within a 2 to 3-meter radius upstream of the bypass. They varied from 0.5 to 1 m/s upstream of the horizontal plate, from 0.8 to 1.4 m/s at the bypass entrance and from 2.0 to 2.3 m/s over the flap gate itself.

Recording of the upstream water level and the flap gate crest enabled recalculation of discharge through the bypass, which varies greatly with the constant variations in level upstream due to turbine operation. In addition, various environmental and plant operation parameters likely to influence the behavior of the migrants and the efficiency of the bypasses were obtained at Soeix powerhouse or recorded with automatic data collection stations (measurement every 15 minutes) : turbined flow, spillage, water temperature, conductivity, atmospheric pressure, solar radiation, turbidity...

The study covered a period of approximately two months each year during the period of peak downstream migration of salmonid juveniles, or from March 11 to May 13, from March 23 to May 12 and from March 11 to May 6 in 1992, 1993 and 1994 respectively.

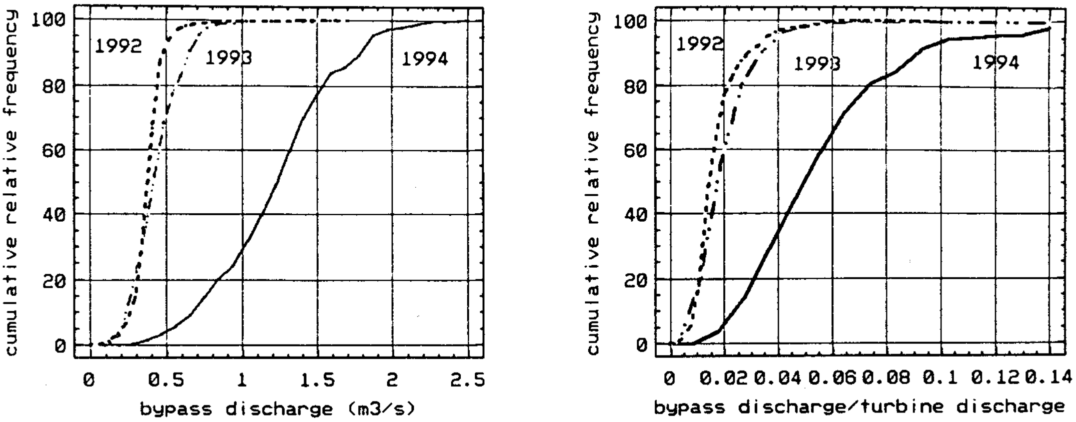


Figure 3 : Bypass discharge during 1992, 1993 and 1994 experiments.

4. ASSESSMENT OF BYPASS EFFICIENCY WITH MARK-RECAPTURE STUDIES

Downstream bypass efficiency was assessed each year using a mark-recapture method. The marked fish were either hatchery-reared salmon smolts or wild salmon or sea trout smolts captured in the downstream bypass at the Gurmençon hydroelectric plant 1 kilometer upstream of Soeix. It was necessary to include hatchery-reared smolts in the study as capture of wild smolts was too uncertain and dependent on as yet limited natural reproduction and on upstream restocking efforts which vary from year to year, as well as on the efficiency of the Gurmençon bypass. Salmon smolts varied in size from 14 to 24 cm (average: 18 cm) and weighed 50 g on average; sea trout smolts were significantly larger (20-30 cm).

Over the three years of the study, more than 3,500 fish were marked. Wild fish were generally released in the tailrace just downstream of Gurmençon power plant, except in the case of spillage over Soeix dam where, like the reared fish, they were directly released in the upstream part of the intake canal of the Soeix plant, about 450 m from the trashracks.

A system for trapping fish was installed downstream of the bypass : it consisted in an inclined screen (15% to 30% slope, depending on the degree of opening of the flap gate) to filter bypass discharge. The space between the bars

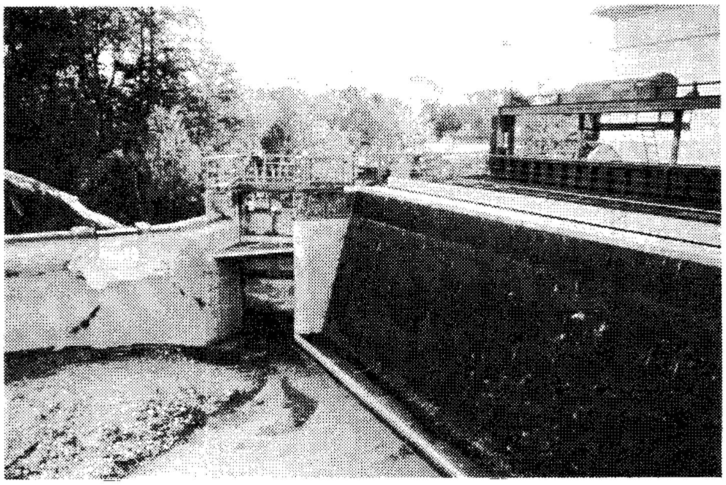


Photo 1 : Location of the two bypasses near the trashrack (forebay emptied). On the left 1992 bypass (closed). On the right 1993 and 1994 bypass with horizontal submerged plate.

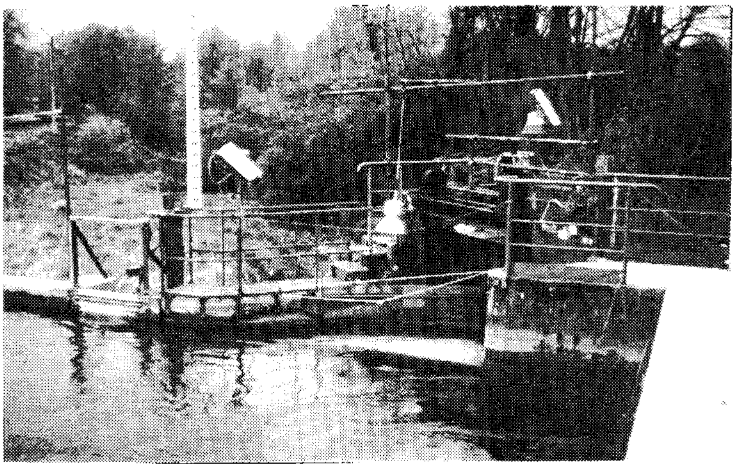


Photo 2 : Bypasses during operation with light above the entrance and video camera to count fish.

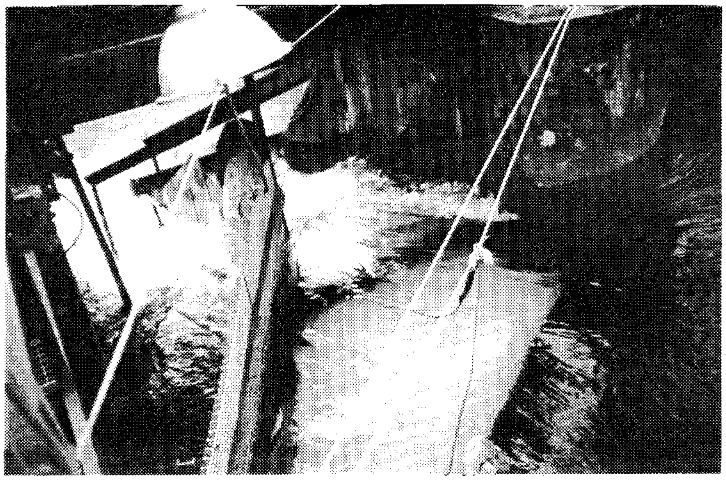


Photo 3 : Entrance view of the 1994 bypass with vertical screen to guide fish to the trap.

was 1 cm, and porosity was around 50%. A PVC plastic spout 0.2 m in diameter at the edge of the screen collected and guided fish toward a holding basin. Depending on the year, fish were trapped in this way 3 to 4 times a day (at 8 a.m., 2 p.m., 8 p.m. and midnight). To avoid mixing the batches, each batch was marked differently; most often this consisted in clipping a notch in one fin. Bypass efficiency was evaluated on the basis of the proportion of fish recaptured in the trap.

In 1992, 1,088 hatchery-reared salmon smolts (in 9 batches) and 121 wild smolts (in 4 batches) were released. The global rate of recapture in the bypass was 22% (from 12.5% to 48% depending on the batch) for hatchery-reared smolts and 22% (from 19.4% to 23.3% depending on the batch) for wild smolts.

In 1993, 305 wild salmon and 221 sea trout smolts were released. The rate of recapture in the bypass was 32% (from 13% to 39% depending on the batch) for the salmon smolts and 35% (from 6% to 41% depending on the batch) for sea trout smolts.

In 1994, 1,536 hatchery-reared salmon smolts (in 12 batches) and 88 wild smolts (in 4 batches), together with 149 sea trout smolts (3 batches) were released. The rate of recapture was 55% for hatchery-reared smolts (from 27% to 89% depending on the batch), 58% for wild smolts (46% to 77%) and 68% for sea trout smolts (65% to 72%).

Several factors might explain the wide variations within one year in the rate of bypass efficiency from one batch to another :

- the stage of smoltification of hatchery smolts released: in 1992 and 1994, the first batches released comprised individuals not fully smoltified, and not all ready for the downstream migration as can be seen from the significant delay (from 15 days to over a month) between release and recapture of the last individuals: a certain percentage of the fish in these batches may have passed over the dam at the time of their release,

- different sizes of fish from one batch to the other (especially between wild and hatchery-reared smolts),

- variable hydraulic conditions (discharge in the bypass, turbining conditions, velocities at the trashrack), even in the timespan of one batch release.

If we consider only the batches of wild fish together with the hatchery-reared fish theoretically ready to migrate downstream, the variability in efficiency is less marked; mean efficiencies are respectively 27% in 1992, 32% in 1993 and 59% in 1994 for salmon smolts; they are systematically higher for sea trout: 35% in 1992, 35% in 1993 and 68% in 1994.

Given the great number of factors influencing the test results, and the natural variability in the results, we can give only a relatively broad range for bypass efficiency, which we estimate at 20-35% in 1992, 25-40% in 1993 and 50-80% in 1994.

For the optimized bypass of 1994, we find that at the end of the migration period, when the fish are probably more ready to migrate, passage through the bypass follows very close on their release (< 24h) and bypass efficiency is at its highest (> 75%).

A certain number of differences were observed in fish size, both between batches released upstream and those recaptured at the bypass and between wild fish captured at Gurmençon and at Soeix. As a general rule, fish recaptured in the bypass are larger than those released upstream, and fish trapped at Gurmençon are larger than those trapped at Soeix. Furthermore, the proportion of sea trout (larger than salmon smolts) in fish captured in the bypasses is consistently higher in Gurmençon than at Soeix. The water intakes of the two plants are similar (size, flow velocity, etc.) with the exception of the spacing of the trashrack bars, greater at Gurmençon (4.5 cm) than at Soeix (3.5 cm). It therefore seems probable that the diversionary effect of the screens depends on the spacing between the bars, or more precisely on the relation between fish size and spacing. It was, however, not possible to examine this criterion more closely.

5. RADIOTRACKING TO MONITOR FISH BEHAVIOR

In each of the study years, radiotracking was performed to monitor displacement and behavior of smolts upstream of the trashracks and close to the bypass. This consisted in releasing fish tagged with transmitters upstream of the Soeix plant and localizing them in areas monitored by recorders in fixed locations. These automatic recordings were complemented by manual tracking. As in the case of the mark-recapture procedures, releases were either in the Gurmençon tailrace some 1 km upstream of Soeix dam (in the absence of spillage over the dam) or directly in the upper the headrace when there was spillage.

The material used was American-manufactured (Advanced Telemetry Systems). Transmitters (frequency band: 48-49 MHz) weighing from 1.5 to 1.9 g and measuring 19.6 mm in length and 6 mm in thickness, with a lifetime of from 11 to 15 days, were introduced into the fish stomach using a technique similar to that followed in France for several years with adult salmon and shad (Travade et al, 1989) and elsewhere on salmonid juveniles (Moser et al., 1990; Armstrong and Rawlings, 1993). Fish displacement was monitored with a set of digital and graphic recorders which, on a very small time step (a few seconds), detected and recorded the presence of fish in areas delimited by underwater antennas (1/4 wavelength wires) or aerial antennas (directional loops). The zones of detection, identical for the three years of the study, are shown in Figure 4. They were defined with a view to identifying the bank along which fish arrive at the plant, displacement in the vicinity of trashracks, passage in the zone of attraction of the bypass (a 3-meter radius upstream of the bypass gate) and passage through the turbines. For manual counting of fish, we used the aerial directional antennas and one underwater coaxial cable loop (12 to 16 cm in perimeter) for precise localization.

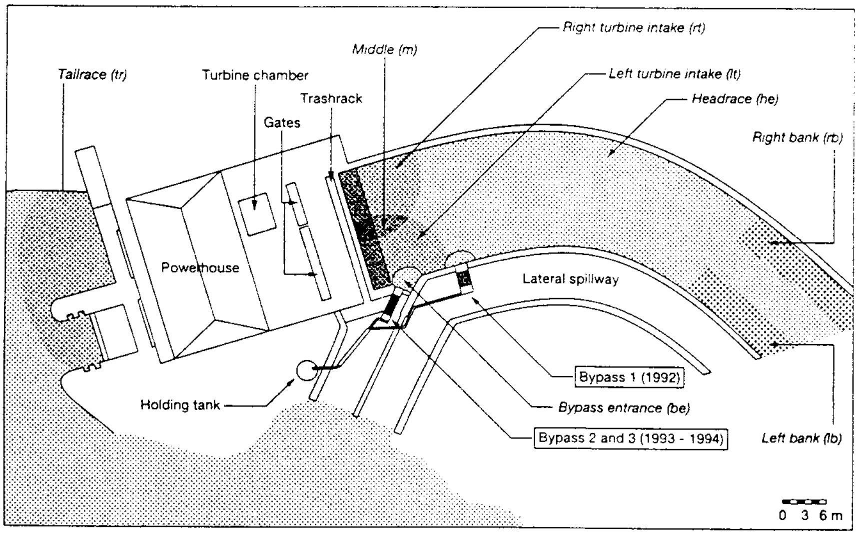


Figure 4 : Location of antennas areas for automatic radiotracking (with abbreviations used on figure 5).

A total of 100 fish (86 salmon and 14 sea trout) were tagged in the three years. For salmon, these were essentially hatchery-reared individuals in 1992 and wild individuals in 1993 and 1994.

In 1992, 27 hatchery-reared salmon smolts and 3 wild salmon smolts were tagged. 30% were recaptured in the bypass.

In 1993, 32 wild salmon smolts and 2 sea trout smolts were tagged. 28% were recaptured in the bypass.

In 1994, 24 wild salmon smolts, 2 hatchery-reared smolts and 12 sea trout smolts were tagged. 76% of the salmon smolts and 82% of the sea trout smolts were recaptured in the bypass.

The efficiency of the three bypasses, successively tested in 1992, 1993 and 1994 and estimated on the basis of radiotracked fish, is close to that measured by mark-recapture methods.

The principal observations with respect to fish behavior on the study site are as follows:

- downstream displacement (passage through the bypass and the turbines) is essentially nocturnal. Almost all passage occurs between 6 p.m. and 8 a.m., through some fish did move around the study site during the day;

- most salmon and sea trout (90%) pause in front of the trashracks and seek a passage, whereas their displacement upstream of the study site and in the headrace is relatively continuous and at a speed close to the flow velocity. This points up the deterrent effect of the screens with the present bar spacing (3.5 cm), but tends to indicate that the efficiency of bypasses located close to trashracks will always be limited by the proportion of fish (10% in this case) which show no reluctance in front of the trashracks;

- in general, fish tend to arrive in front of the screens from the left bank, and it is in this area that most of them stay. This is most certainly due to the general current patterns in the vicinity of the trashrack;

- almost all fish remain and move close to the trashracks (a few dozen meters maximum). A small proportion (2 salmon and 3 sea trout) returned more than 450 meters in the reservoir, once having reached the trashracks. Some even went back and forth several times between the power station and the reservoir. Most displacement of this kind occurred when turbined discharge was between 20 and 25 m^3/s;

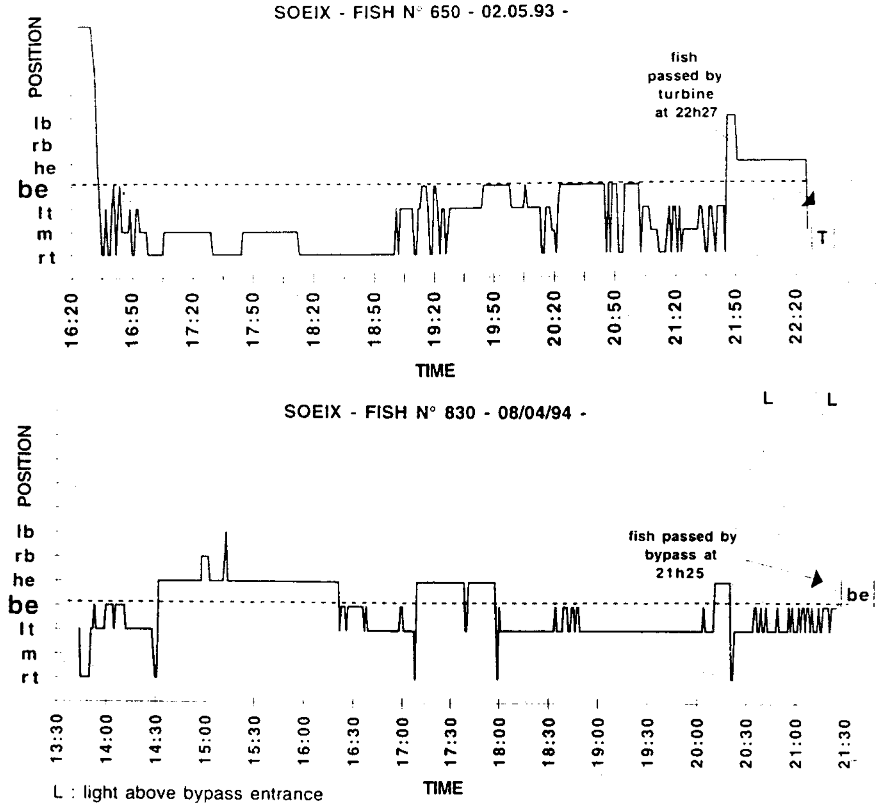


Figure 5 : Two examples of fish movement patterns in the headrace and trashrack areas

- the duration of presence in front of the screens is extremely variable (from 1 minute to 38 hours). There is a difference in this duration depending on the year: in 1994, fish stayed for a shorter time in front of the screens (50% stayed for less than 15 minutes) than in 1992 and 1993, when respectively 24% and 16% stayed for less than 15 minutes;

- fish passing through the turbines passed primarily through the Kaplan wheel (left bank). Most fish (respectively 88%, 50% and 60% in 1992, 1993 and 1994) had already passed in the zone of attraction of the bypass (less than 3 m in front), and some had passed there more than 30 times, indicating its low attraction in 1992 and 1993 and prompting the decision to modify it in 1994: higher discharge and improved hydrodynamics around the entrance,

- fish passing through the bypass generally did so after several passages in the near vicinity (up to 66 times). There is nonetheless a difference between the first two bypasses (1992 and 1993) and the 1994 version into which a higher proportion (40%) entered the first time they approached the bypass (compared with 14% and 20% in 1992 and 1993). In 1994, furthermore, 4 fish (14%) passed directly into the bypass without having reached the trashrack, a phenomenon not observed the preceding years. This again indicates its greater attraction in 1994,

- the effect of the rate of discharge in the bypass was not pointed up in 1992-1993 but, in 1994, we find that when fish pass through the bypass, the discharge in the bypass and the ratio of bypass discharge/turbined discharge are higher than when fish pass through the turbine. The flow discharge passing into the bypass when fish pass through is about 6% (from 4% to 8%) of the turbined discharge.

6. EFFECT OF LIGHT

Lighting the bypass at night to increase its attraction appears promising, as shown in a number of previous experiments (EPRI, 1986). Intermittent lighting was found preferable to continuous lighting, as it was observed in previous studies (Taft, 1988 ; Larinier and Bernard, 1991b) that while the bypass was lit, fish passage occurred essentially at the moment the light was turned on or off. During the 3 years of experiments at Soeix, therefore, intermittent lighting of the bypass was tested, with a 50-W mercury vapor lamp installed some 1.5 meters from the surface above the bypass.

The frequency and duration of lighting intervals changed over the 3 years in accordance with the results observed in the preceding year :

- in 1992, light phases alternating with dark phases lasting 1/2 hour,

- in 1993, light phases lasting 20 minutes followed by extinction for 10 minutes;

- in 1994, 10-minute light phases followed by 5-minute extinction.

The rates of fish passage through the bypass lit by the mercury vapor lamp and in the absence of any lighting at all were compared. While the frequency and duration of light phases changed over the three years, the results were nonetheless identical: fish react clearly and systematically to a change in lighting (most particularly to extinction). Figure 6 shows the results obtained in 1992 and 1993: in the minutes following extinction of the lamp, a significant increase in fish passage is observed. On the other hand, when we count all fish passage, there are statistically no more fish passing during intermittent lighting than during no lighting of the bypass. In fact, it appears that lighting concentrates passage through the bypass but does not globally enhance bypass efficiency.

With radiotracked fish, the attraction of light could not be demonstrated. Behavior differs widely from fish to fish. The effects of hydraulic attraction undoubtedly mask those of lighting.

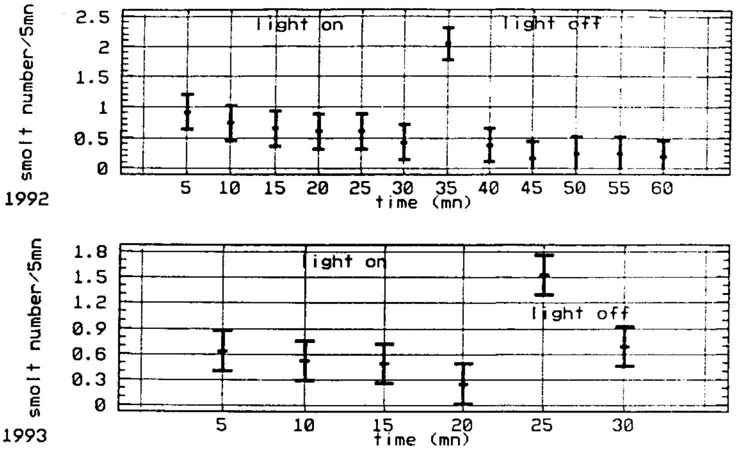


Fig. 6 : Rate of smolt passage under different light conditions (1992 and 1993)

7. DISCUSSION

Experiments conducted over a three-year period at Soeix enabled successive testing of three downstream bypasses designed to deter passage of salmonid juveniles through turbines. The devices tested were surface bypasses installed on the river bank near the trashracks of the power station. The use of two complementary techniques, mark-recapture to quantify the efficiency of the bypasses and radiotracking to monitor fish behavior, enabled modifying the configuration of the bypasses from one year to the next, and successively highlighting the parameters influencing their efficiency.

The first bypass (1992), located some distance (around 6 meters) upstream of the trashrack and functioning for 90% of the time with a discharge between 0.2 m^3/s and 0.5 m^3/s (or 0.5% to 3.5% of the turbined discharge), was revealed to have low efficiency (20% to 35%), due in part to its distance from the trashracks.

The second bypass (1993), located much closer to the screens (1.5 m) and functioning with much the same discharge (between 0.2 and 0.7 m3/s for 90% of the time, or 0.5% to 4% of the turbined discharge), was not much more efficient (25% to 40%) whereas radiotracking pointed up the presence of significant numbers of fish congregating in the immediate vicinity of the bypass. The low efficiency was then attributed to the insufficient flow velocities on the surface and to hydrodynamic disturbances near the bypass, particularly the presence of an upwelling that masked the entrance.

The third bypass (1994) was located in the same spot as in 1993 but modifications had been made prior to the 1994 study to improve the hydrodynamics:

- enlarging of the bypass to 1.8 m (double the original width),

- raising the discharge (mean discharge: 1.2 m^3/s; 90% of the time between 0.5 and 1.9 m^3/s, or between 2% and 10% of the turbined discharge),

- immersion of a horizontal plate in front of the bypass entrance so as to eliminate the upwellings that masked the entrance, and increased surface velocities in an attempt to draw fish toward the bypass.

The 1994 bypass was significantly more efficient in that it was practically double the 1993 efficiency (50% to 80%): on average, 59% of the salmon smolts and 68% of the sea trout (from 42% to 89%, depending on the batch considered).

This improved efficiency was confirmed by radiotracking, which showed that the fish find the bypass entrance more rapidly and pass fewer times in its zone of attraction.

It is, however, difficult to evaluate the relative importance of the improvement made by eliminating the upwellings masking the entrance and that of the increased bypass width and discharge; in our opinion, the two factors are complementary.

The discharge factor does appear significant insofar as, in 1994, radiotracking showed that fish passage through the turbines generally occurs when bypass discharge is lowest.

The attraction exerted by light appeared less significant in 1994 than in the two preceding years. Results obtained in the three years at Soeix seem to indicate that the hydrodynamic factor is most certainly the predominant factor to consider when designing a downstream bypass. Light does not noticeably affect bypass efficiency, whether the hydrodynamic conditions are favorable or severe. On the other hand, given the undeniable effects of light on fish, it is possible that light will be found useful in the absence of pronounced hydrodynamic factors, for example in attracting fish toward a downstream bypass on a reservoir with very low surface velocities.

8. CONCLUSIONS AND PROSPECTS

We can draw certain conclusions from the experiments at Soeix to help in designing downstream bypasses for small water intakes (width < 20 m).

The bypass must be both close to the trashracks (1 to 2 meters maximum upstream of the screen) and in an area where fish tend to congregate, which depends heavily on the hydrodynamic conditions upstream of the screens (tangential currents, zones of recirculation, upwellings, etc.). It is relatively clear that, having fought against entrainment through the trashracks, fish concentrate more or less passively in lateral recirculation zones close to the screens, which therefore appear to be the most favorable to installation of a bypass.

Discharge in the bypass must be proportional to the turbined discharge. As for upstream fishpasses, the minimum required discharge appears to be a few per cent of the turbined discharge. The good results obtained in 1994 were for a discharge that was from 2% to 10% of the turbined discharge (median 5%), while bypass discharges of 0.5% and 4% of the turbined discharge were revealed inadequate in the preceding years. In this respect, it appears very important to account for fluctuations in upstream water level when determining the level of the bypass crest, so as to maintain a constantly adequate discharge.

The hydrodynamic conditions in the immediate vicinity of the bypass entrance appear essential. They must be characterized by undisturbed surface flow (no detachment or turbulence) and be perceptible from as great a distance as possible from the bypass, with a gradual acceleration up to the bypass entrance. This acceleration must be found both on a horizontal and a vertical plane, hence the importance of the geometry of the bypass entrance (no sharp angles) and the preference for the use of surface flap gates rather than flat vertical gates. The head of water over the crest must be at least forty centimeters.

The characteristics of the trashrack (spacing of bars) and velocity patterns in the vicinity of the bypass entrance play an important role in determining the possibility for and duration of stay in front of the screens and consequently the probability that fish will find the bypass entrance. These two factors, related to the swimming ability of the fish (and therefore to their size), could not be quantified, but influence the size-selection of fish entering the bypass, which is all the greater when spacing between bars and velocities in front of the screens are great.

Following the monitoring at Soeix power station, experiments with several different focuses have been undertaken with a view to defining improved design of downstream bypasses.

The good results obtained at Soeix are largely due to the fact that fish are "hydraulically" concentrated near the bypass; it therefore appeared interesting to assess the efficiency of bypasses on less hydrodynamically favorable sites. This was done on another intake on the Aspe river at Bedous, some twenty kilometers upstream of Soeix, where bypass efficiency was found in 1995 to be no better than 17%.

The trashracks tested on the Gave d'Aspe are of moderate width (around fifteen meters). It appears indispensable to know the trashrack size above which one should install several downstream bypasses. Studies were thus undertaken in spring 1996 on the Garonne, upstream of Toulouse, to test the efficiency of downstream bypasses in the case of larger intakes and higher discharge. It would also be advisable to test devices to attract or guide fish which might improve bypass efficiency. Such tests were undertaken in 1995 on electric screens, and in spring 1996 on acoustic deflection devices.

Acknowledgments

These studies were carried out in collaboration between CSP (National Council of Inland Fisheries), CEMAGREF and EDF (Electicité de France). Funding was provided by the Regional Administration, CSP and EDF. We particularly thank D. Ingendahl, J.M. Bach , D. Pujo for their contribution to the three year field study.

REFERENCES

Amstrong, J.D. and Rawlings, C.E. (1993). The effect of intragastric transmitters on feeding behaviour of Atlantic salmon, *Salmo Salar*, parr during autumn. *Journal of Fish Biology* 43, pp. 646-648.

Bomassi P. and Travade F. (1985). Projet de réimplantation du saumon dans la partie supérieure de l'Allier : expériences sur les possibilités de dévalaison des saumoneaux au barrage hydroélectrique de Poutès en 1983 et 1984. Colloque pour la restauration des rivières à saumon, Bergerac, 28 mai-1er juin 1985.

Clay C.H. (1995). Design of fishways and other fish facilities.Lewis publishers, 248 p.

E.P.R.I. (1986). Assessment of downstream migrant fish protection technology for hydroelectric application. EPRI RP 2694-1, 420 p..

E.P.R.I. (1994). Research update of fish protection technologies for water intakes. EPRI TR-104122, 208 p.

Larinier M. and Boyer-Bernard S. (1991a). Dévalaison des smolts et efficacité d'un exutoire de dévalaison à l'usine hydroélectrique d'Halsou sur la Nive. *Bull.Fr.Pêche.Pisc.* 321, pp. 72-92.

Larinier M. and Boyer-Bernard S. (1991b). La dévalaison des smolts de saumon atlantique de Poutès sur l'Allier (43) : utilisation des lampes à vapeur de mercure en vue d'optimiser l'efficacité de l'exutoire de dévalaison. *Bull.Fr.Pêche.Pisc.* 323, pp. 129-148.

Larinier M., Travade F., Ingendahl D., Bach J.M. and Pujo D. (1993). Expérimentation d'un dispositif de dévalaison pour les juvéniles de saumon atlantique (Usine hydroélectrique de Soeix - Gave d'Aspe année 1992). Rapport GHAAPPE 93.02/EDF-DER HE.31/93/012, 28 p.

Larinier M., Travade F., Ingendahl D., Bach J.M., Pujo D., Barracou D. and Nouel F. (1994). Expérimentation d'un dispositif de dévalaison pour les juvéniles de saumon atlantique (Usine hydroélectrique de Soeix - Gave d'Aspe année 1993). Rapport GHAAPPE 94.02/EDF-DER HE.31/94/043, 29 p.

Moser M.L., Olson A.F. and Quinn T.P. (1990). Effects on dummy transmitters on juvenile coho salmon. *Am. Fish. Soc. Symp.* 7, pp. 353-356.

Ruggles C.P. (1980). A review of the dowstream migration of atlantic salmon. *Can. Tech. Rep. Fish. Aquat. Sci.* 952, 1-39.

Ruggles C.P. (1992). What's new in downstream fish passage ? Fourth International Atlantic Salmon Symposium, St Andrews, New Brunswick, Canada, June 14-17, 1992, 22 p.

Taft N. (1988). Evaluations of fish protection systems for use at hydroelectric plants. Hydro Review VII(2) pp. 54-62.

Travade F., Bomassi P., Bach J.M., Brugel C., Steinback P., Luquet J.F., Pustelnick G. (1989). Use of radiotracking in France for recent studies concerning the EDF fishway program. Hydroécol. Appl. 1(2) pp. 33-51.

Travade F., Larinier M., Ingendahl D., Bach J.M., Pujo D. and Barracou D. (1996). Expérimentation d'un dispositif de dévalaison pour les juvéniles de saumon atlantique (année 1994). Usine hydroélectrique de Soeix (Gave d'Aspe - 64). Rapport GHAAPPE 96.02/EDF-DER HE-31/96/006, 34 p.

Fish community structure, distribution patterns and migration in the vicinity of different types of fishways on the Enz River (Germany).

Jansen, W., Kappus, B., Böhmer, J., Beiter, T., Rahmann, H.
Institute for Zoology, University of Hohenheim
Garbenstr. 30, D-70593 Stuttgart
Germany
jansen@uni-hohenheim.de

ABSTRACT

In order to improve the longitudinal continuum of the many dammed streams, increasing numbers of new fishways have been constructed in southern Germany within the past few years. Although it has been generally recognized that these structures must allow the migration of all aquatic biota and more 'natural' designs are frequently adopted, fishways often differ greatly in their structural characteristics. Furthermore, the effectiveness of fishpasses, particularly with regard to artificial flow regulation or other disturbances encountered in regulated streams, have rarely been assessed. We evaluated the effects of fishway design, stream morphometry, and hydrological regime within the corresponding river section on fish distribution patterns and migrations. On 20 different occasions between August 1994 and September 1995, we collected fish upstream, downstream, and inside three different types of fishways on the Enz River (108 km length, 16.6 $m^3 \cdot s^{-1}$ mean annual discharge), a second order tributary of the Rhine River. The first fishpass (site I) is a 75 m long, 2 m wide concrete channel filled with gravel and boulders, the second (site II) is a 83 m long, 2-4 m wide artificial stream, and the third (site III) is a step and pool fishway. Fish were collected by electrofishing. On most sampling occasions, species and size classes were recorded separately for specific habitat types or sections within the fishways. In addition, traps were installed for 3-5 week long periods at the upstream outlets of all three fishways. More than 3550 fish (>5 cm) of 28 species were captured and measured. Approximately 770 fish were marked by subcutaneous injection of alcyan blue dye and were released downstream of a fishway. Of these, 45 fish were recaptured. Almost 1150 fish, including 5 marked individuals, were captured in the traps, providing direct evidence for migration through the fishway. Of the 28 species found, 11 occurred at all three locations, 11 were captured at only one site, and three were represented by a single individual. The total number of species recorded decreased downstream from 21 at site I to 18 at site II, and 17 at site III. There were pronounced changes in the abundance of most of the dominant species downstream and upstream of the fishway at site III, and, to a lesser degree at sites I and II, indicating species (and size) specific differences in the effectiveness of the three designs for fish passage. Furthermore, marked differences in relative fish abundance between downstream locations and inside the fishway itself, and in the relative numerical composition of the trap catch at sites I and II, clearly demonstrated that some species (*Cottus gobio*, *Barbatula barbatula*, *Salmo trutta*, *Leuciscus cephalus*) use these fishways as a habitat, whereas others (*Phoxinus phoxinus*, *Gobio gobio*) seem to simply migrate through. Only inside the artificial stream (site II) did we find early life stages of at least seven species, indicating that passage of the other fishways is probably restricted to older fish.

KEY-WORDS: southern Germany / fishway / migration / marking / traps / species distribution / community composition / habitat use / young-of-year / abundance.

INTRODUCTION

The typical nature of most streams as longitudinal ecosystems showing a more or less pronounced continuum in characteristic changes of physical, chemical, morphometrical, or hydrological properties along their length has been profoundly disturbed, particularly in the so called 'developed' countries (e.g. Weaver and Garman, 1994; Lignon *et al.*, 1995). Over the last few hundred years, increasing numbers of weirs, gauge structures, dams, and other horizontal barriers have been constructed to obtain, for example, hydroelectric power or drinking water, to promote ship transportation, to supposedly prevent flooding, or to reduce erosion. Some of the ecological consequences of such barriers which have the potential to substantially affect fish communities include the shifts in thermal regimes (Spence and Hynes, 1971), excessive sedimentation (Moog *et al.* 1993), a reduction in the areal extent and duration of floodplain inundation (Zalumi 1970), alterations of channel morphology (Simons 1979), shifts in the size of organic particles and in trophic relationships (Ward and Stanford, 1983), and fragmentation or complete loss of habitats (Kirchhofer, 1995). Although many of these changes in the ecological processes of rivers are difficult to reverse without incurring immense costs for the large scale removal of existing barriers, recently many politicians and administrators appear to have embraced the idea that in those cases in which artificial barriers prevent the upstream (and downstream) migration of fishes (and macroinvertebrates), the construction of fish passages is the preferred solution to improve, if not restore, the river continuum. Thus, in southern Germany, increasing numbers of new fishways have been constructed within the past few years. Although it has been more frequently recognized that these structures must allow the migration of all aquatic biota and more 'natural' designs are often adopted, fishways may differ greatly in their structural characteristics. Furthermore, the effectiveness of fishpasses, particularly with regard to artificial flow regulation or other disturbances encountered in regulated streams, have rarely been assessed, especially for nonsalmonid species (for exceptions see Schwalme *et al.*, 1985; Schmutz *et al.*, 1995). The aim of this study was to evaluate the effects of fishway design, stream morphometry, and hydrological regime on fish distribution patterns and migrations in the vicinity of three different types of fishpasses on a medium sized river in south-west Germany. Furthermore we tried to relate fish community composition along the 50 km river section under study to possible barrier effects, habitat features, and the concept of river zonation.

MATERIAL AND METHODS

Study sites

Fish were captured upstream, downstream, and inside three different types of fishways on the Enz River, a second order tributary of the Rhine River in southern Germany (Fig.1). The Enz, a carbonate rich stream of 108 km length has a mean annual discharge of 16.6 $m^3 \cdot s^{-1}$ and a basin area of 326 km^2 at the first sampling site 58 km upstream from the mouth. The second and third sites are located 29 km and 56 km, respectively, further downstream (Fig.1). The first two fishways are relatively long bypasses. The first one (site I) is a 75 m long, 2 m wide concrete channel filled with coarse gravel and regularly placed large boulders. The second one (site II) is a 83 m long, 2-4 m wide artificial stream, and at the third site, a classical step and pool fishway with four concrete basins has been installed. Importantly, the upstream end of the concrete bypass exits abruptly - more than 4 m above the river bottom - into an almost stagnant, 1 km long, impounded section. At the exit, the concrete channel is split into two <1-m wide openings by a central railing into which tightly fitting wooden beams are slid at different heights for the two openings. On the one side, these beams leave a 40 cm high submersed opening immediately above the channel bottom, and on the other side, the beams start from the bottom up, leaving a surface opening of 20-30 cm

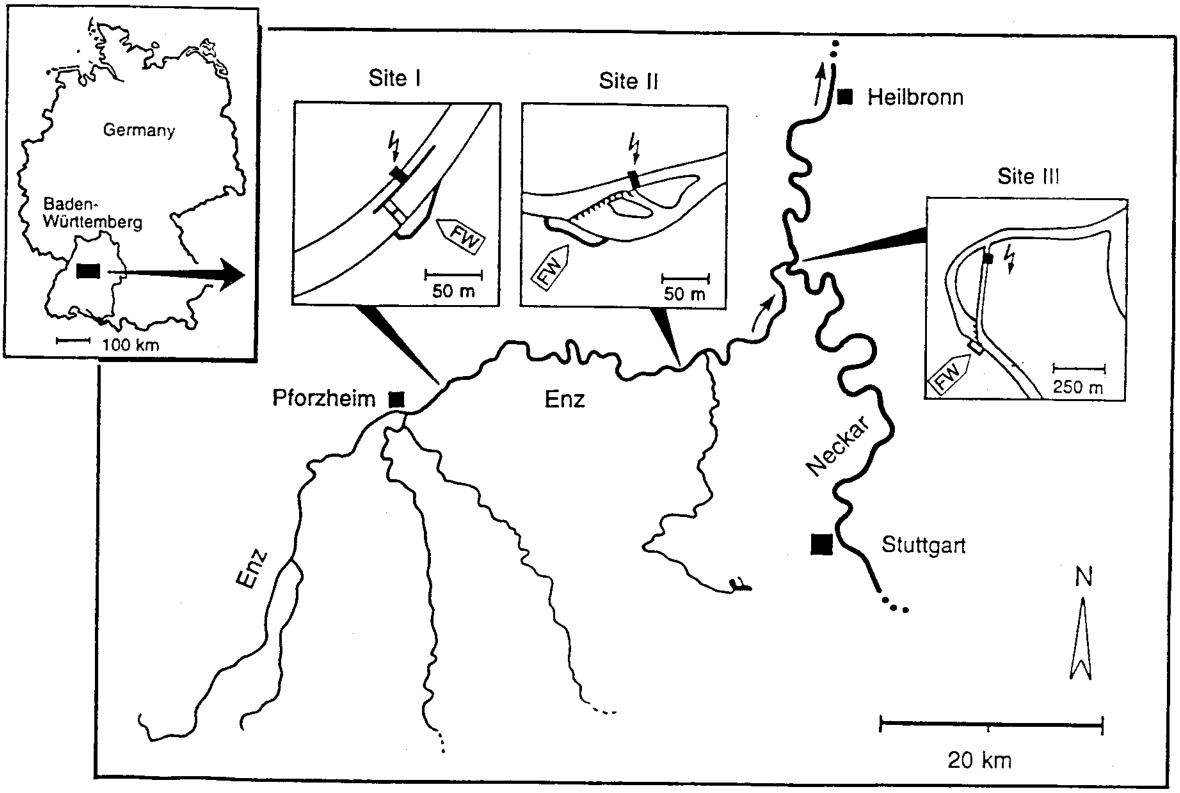


Figure 1: Map showing the location of the three fishways (FW) at Eutingen (site I), Oberriexingen (site II), and Besigheim (site III) on the Enz River.

height depending on overall water level. For the step and pool fishway, the basins are separated by wooden boards into which 10 x 10 cm wide openings are cut at opposite sides on the bottom and on the top. Other characteristic features of each fishpass are given in Table 1. At each site, two (site III) to four (site II and I) 20-100 m long river sections were identified as actual fish sampling locations. Since the length of the impounded river sections upstream of each weir differed between sites, and comparable downstream habitats were sometimes difficult to find, for sites II and III, the upstream locations were identical to the first downstream location of site I and II, respectively. For site I, the upstream sampling location was almost 3 km above the weir.

To assess fish habitats on a smaller scale upstream and downstream of the fishways and inside the artificial stream, distinct habitat types were characterized based on substrate, local current speeds and water depth. Inside the concrete bypass, 37 consecutive 2-m sections were designated as sampling units, and for the classical fishpass, each chamber was considered a unit. On most sampling occasions, species and size classes were recorded separately for each specific habitat type or sampling unit.

Fish sampling

On at least two different occasions each between August 1994 and September 1995, we collected fish by electrofishing, trapping, or with pond nets (Table 2). Nets were used once in all three fishways when water levels were normal and at three other times when levels were much reduced because of repairs to the Eutingen power plant.

Table 1: Characterization of the Enz River fishways at Eutingen, Oberriexingen, and Besigheim. Height Δ = Height differential between downstream entrance and upstream exit. Ranges for water depths and current speeds at each fishway are from 20-150 measurements taken with a 'Mini-Water 2' meter (Fa. Schiltknecht, Swizerland) at various levels of discharge. * Maximum current speed measured close to the downstream entrance before it was remodelled on August 7, 1995. Thereafter, maximum speeds were reduced to 1.4 m/s.

Parameter	Eutingen	Oberriexingen	Besigheim
Type	Bypass channel	Artificial stream	Step and pool fishway
Design	Concrete lining filled with coarse gravel and boulders	diverse; locally present natural materials	3 concrete basins, wooden dividers with 10x10 cm slots
Date build	June 1992	Dezember 1994	July 1980
Length (m)	75	83	10
Width (m)	2	2-4	1
Water depth, range (m)	0.2-0.6	0.2-1.0	0.5-0.8
Height Δ (m)	4.3	1.8	0.8
Slope, range (%)	5-10	2-9	20 cm high steps
Current speed, range (m/s)	0.2-1.3	0.1-2.1*	0.6-1.1
Discharge, range (L/s)	120-150	600-900	20-40

Apart from a routine, weekly plant maintenance, which resulted in a transient (4-6 hours) approximately 80% reduction of water flow into the fishpass, on these three occasions (September 7 and 30, 1994; May 8, 1995), the movable weir was lowered to the point where river water level dropped below the exit of the bypass channel, causing the fishway to fall dry for 3-23 days. On September 30, 1994, the rubber gaskets of the weir were replaced and two gravel and sand dams were built across the Enz River 10 m upstream and 25 m downstream of the weir while the complete flow was diverted through the turbine canal and a spillway. Before the water between the two dams was pumped out completely, we quantitatively collected the remaining fishes with nets, which had congregated in a depression of a 25 m long, 1.5-2 m deep concrete basin which sloped towards the base of the weir. On October 22, the dams were left to be washed away and water supply to the fishway was restored. However, frequent construction work was carried out inside the river on a 200 m long section immediately downstream of the weir until January 1995. This greatly affected water turbidity and flow conditions at two of our sampling locations, and permanently created a 120 m long section with reduced current below the downstream entrance of the fishway, thus completely separating its attractant current from the main stem flow.

Traps were installed for 3-5 week long periods at the upstream outlets of all three fishways (Table 2). The trap for the concrete channel consisted of two aluminum frames each fitted precisely to cover one of the two openings of the fishway exit (see above). Because the 1 mm square mesh net used as screen during the first trapping period clogged rapidly causing reduced flow condition inside the fish pass, and apparently facilitated the escape of captured fishes, the trap screen was changed to 4 mm stainless steel wire for the second trapping period and funnels made out of 4 mm mesh netting were installed into the trap entrances. As a test of capture efficiency, two individuals from each of the six most common fish species at site I were marked and placed into the two traps. After three days, no fish had escaped, and it was assumed that the traps would quantitatively catch fish migrating upstream. The commercially bought hoop net traps (5 mm stretched mesh; Fa. Kappel, D-87452 Altusried) used at sites II and III, either completely covered the upstream exit of the fishpass, as in the case of the concrete fishway, or were installed just inside the exit, the wings leaving a minimal distance to the water surface to reduce the risk of clogging or damage. Unfortunately, the trap set inside the artificial stream was stolen 5 days into the first trapping period. For the second period, a custom made box trap (stainless steel frame with 2 m long wings, 4 mm wire mesh) was installed in place of the hoop net trap. All traps were checked daily.

Table 2: Dates of electrofishing (EF), net sampling (NS), and traping periods (TP) at the fishways (FW) in Eutingen, Oberriexingen, and Besigheim on the Enz River in 1994 and 1995. See text for descriptions of traps. 'FW entrance' refers to the sampling location immediately below the downstream entrance of the fishway.

Date/Period	Eutingen	Oberriexingen	Besigheim
18.08.94	EF, all locations	-	-
25.08.94	-	-	EF, all locations
07.09.94	NS, FW only	-	-
01.10.94	NS, weir basin only	-	-
07.08.-07.09.94	TP, box traps	-	-
31.10.94	EF, FW + FW entrance	-	-
07.12.94	EF, FW + FW entrance	-	-
09.03.95	-	EF, all locations	-
26.04.95	NS, FW only	NS, FW only	NS, FW only
05.05.95	EF, all locations	-	-
08.05.95	NS, FW only	-	-
11.05.95	-	EF, all locations	-
11.-15.05.95	-	TP, hoop net trap	
31.05.95	EF, FW + FW entrance	-	-
20.07.-11.08.95	TP, box trap	-	-
11.08.-18.09.95	-	TP, box trap	-
07.-24.08.95	-	-	TP, hoop net trap
22.08.95	-	EF, FW only	-
24.08.95	-	EF, all locations	EF, all locations
18.09.95	-	EF, FW + FW entrance	-

All fish caught were identified to species and their length was estimated to the nearest centimetre. All trapped fish, and approximately 70% of the fish captured by electrofishing and with nets were transferred to holding tanks in the field. After anaesthesization with MS-222, all fish were measured (±0.1 cm) and most were weighed (±0.2 g) on a digital field balance (PESA, model BR 100). Depending on the apparent health status and ambient temperatures, a variable portion of all fishes >5 cm was marked by subcutaneous injection of alcyan blue dye, and was released after complete recovery, at different distances (maximally 300 m) downstream of a fishway. This marking technique is known to cause only negligible mortality and effectively marks even small fish for periods of over a year (Hill and Crossmann, 1987; Bridcut, 1993). We used small hypodermic needles (0.6 mm opening), which could be used more accurately and were less damaging to small fish than jet injection. Specific fin markings (e.g. left pectoral fin) were used to indicate the exact location and/or the date of release. Fish recapture during subsequent electrofishing or trapping periods was taken as direct evidence of upstream migration including some information on approximate distances covered and on the minimum time period spent at a particular site.

RESULTS

Exactly 3554 fish (>5 cm) of 28 species were captured by electrofishing or netting. In addition, approximately 4800 young-of-year (YOY) fish of 11 species were caught. More than 4000 of the YOY fish were gudgeons (*Gobio gobio*; approximately 75%), riffle minnow (*Alburnoides bipunctatus*; 8%), stone loach (*Barbatula barbatula*; 6%), and barbels (*Barbus barbus*; 4%), netted from the concrete basin below the weir at Eutingen on October 31, 1994. Of the species present in the step and pool fishway, no individuals smaller than 5 cm were caught either during the trapping period or when electrofishing in late August of 1994 and 1995 (Table 3). At the same time, YOY barbels, gudgeons, chub (*Leuciscus cephalus*), roach (*Rutilus rutilus*) dace (*Leuciscus leuciscus*), and bleak (*Alburnus alburnus*) measuring 2-5 cm were abundant immediately downstream of the fishway.

Table 3: Length (cm) of the smallest specimen captured downstream of the fishway, inside the fishway, or inside the traps for the more abundant fish species at three sites on the Enz River. Only sampling times between August and December are considered. Values without a decimal place are estimates. [*] = single juvenile fish, next smallest specimen measured 22.0 cm; [b] = in addition, larger numbers of (unmeasured) fry were present at the sampling location. DS=downstream, FW=inside the fishway.

Species	Eutingen			Oberriexingen			Besigheim		
	DS	Trap	FW	DS	Trap	FW	DS	Trap	FW
Salmo trutta	10.0	9.7	8.6	4.7	-	9.0	-	-	-
Alburnoides bipunctatus	2.2	7.3	8.0	5.6	8.1	-	-	-	-
Barbus barbus	3.7	5.3[*]	4.9	2.5	8.2	3.6	2	8.5	6
Gobio gobio	2.2	6.7	13.2	3.5	9.4	3.2	2	8.0	-
Leuciscus cephalus	4.0	12.0	13.0	5.6[b]	9.0	4.6[b]	4	8.0	5
Phoxinus phoxinus	3.0	5.3	7.1	3.0	-	3.2	-	-	-
Rutilus rutilus	10.7	22.6	15.3	8.5[b]	12.5	7.8	4	6.0	5
Barbatula barbatula	3.7	6.2	4.1	4.4	9.1	5.1	3	-	-
Cottus gobio	4.7	9.7	5.1	6	8.2	4	-	-	-

Of the 386 fishes caught inside the bypass channel at Eutingen, only five, three bullheads, one stone loach and one European minnow (*Phoxinus phoxinus*), were YOY fish. All of these were captured within the first 8 m of the downstream entrance. At this point, the gravel had been washed out, and a 1.2 m long section of sheer concrete remained, over which current speeds were always in excess of 1 m/s. Actually, less than 10% and 15% of all the 121 bullhead and 21 stone loach caught inside the fishway occurred upstream of this area. No YOY fish were found inside the traps during both trapping periods (see section on trap catches below). At Oberriexingen in late August, substantial numbers of YOY chub and European minnow were found at the only "pool" area, approximately two thirds up the length of the artificial stream. Other species for which YOY fish were caught both inside the artificial stream as well as downstream of the fishway, included - in descending order of abundance - barbel, gudgeon, roach, bullhead, and dace. Only two species, riffle minnow and bleak, of which approximately 10 and 40, respectively, YOY fish were caught downstream of the artificial stream, were never observed as YOYs inside the fishway.

Of the 28 species found in the Enz River, 11 occurred at all three sites, 11 were captured at only one of the three sites, and brook trout (*Salvelinus fontinalis*), goldfish (*Carassius auratus gibelio*), rudd (*Scardinius* *erythrophthalmus*), and pikeperch (*Stizostedion lucioperca*) were represented by just a single specimen (Table 4). The total number of species recorded decreased downstream from 21 at site I (Eutingen) to 18 at site II (Oberriexingen), and 17 at site III (Besigheim; Table 3). These differences remained, if species for which only individual finds existed were excluded from the analysis. Community similarity, expressed by Soerensen's Index (SI; Soerensen, 1948), showed a clear pattern of decreasing similarity with increasing distance downstream. Whereas the SI values for the comparison between Eutingen and Oberriexingen and between Oberriexingen and Besigheim were 80% and 71%, respectively, the SI for Eutingen vs. Besigheim was 54%.

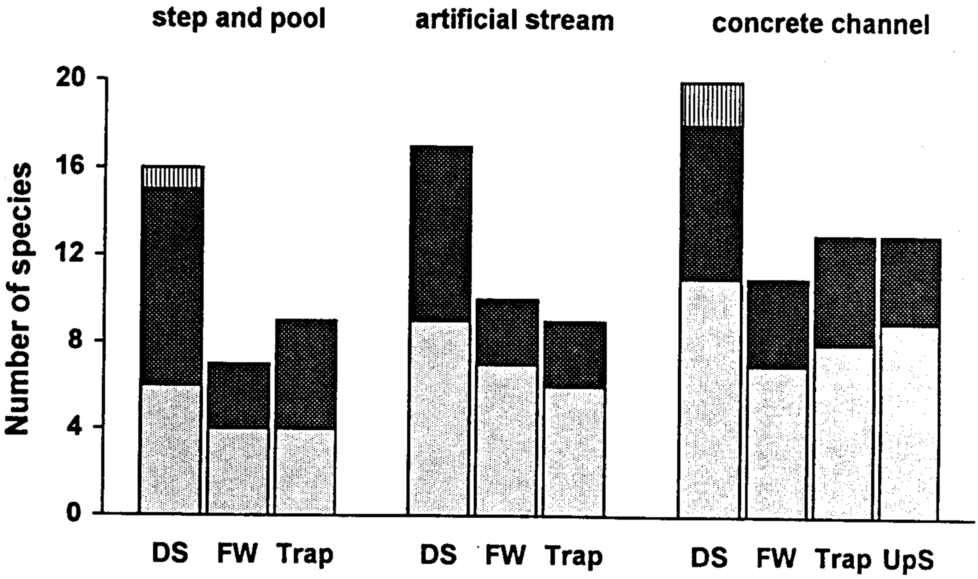


Figure 2: Number of fish species caught downstream (DS), within (FW), with traps at the upstream exit, and upstream (UpS) of three types of fishways on the Enz River between August 1994 and September 1995. Species are categorized according to their habitat preferences with regard to current and substrate conditions: rheophilic (light stipling), indifferent (dark stipling), and limnophilic (vertical lines).

When the different sampling locations and the abundance of fishes were considered in an analysis of species distribution, some additional trends became obvious. Both absolute numbers and relative proportions of rheophilic species (see Table 4) caught at each location were lowest at the step and pool fishway near the mouth of the Enz River (Figure 2). In contrast, the location specific total number of species was always highest near the concrete channel at Eutingen. Importantly, for all three sites, the number of species at locations downstream of the fishway was considerably higher than the number of species found inside the fishway (Figure 2), and this large difference persisted even when the species caught with traps were combined with those captured by electrofishing or net sampling. In addition, the relative numerical abundance of species differed profoundly between sites and among locations within sites. The bullhead, for example, was abundant at Eutingen, occured in small numbers at Oberriexingen, and was absent in Besigheim (Figure 3). An almost identical pattern of declining abundances downstream, although at much lower absolute numbers, existed for the grayling. In contrast, the bleak showed almost the reverse pattern of strongly declining abundances upstream. This species, together with roach and gudgeon, dominated the fish community downstream of the weir at Besigheim (Figure 3), and, together with roach, contributed almost 90% to the catches from the fishway, either by electrofishing or trapping (Figure 3).

Table 4: List of fish species caught in the vicinity of three fishways on the Enz River between August 1994 and September 1995. Habitat preferences (HP) with regard to current and substrate conditions (L=Limnophilic, R=Rheophilic, I=Indifferent; modified after Jungwirth 1984) and relative abundances (●=common, >5% of all fishes caught; ○=rare; E=individual find) are indicated. Oberriex. = Oberriexingen, B'heim = Besigheim.

Species	Common name	Eutingen	Oberriex.	B'heim	HP
Lampetra planeri (BLOCH)	Brook lamprey	○	-	-	R
Anguilla anguilla (L.)	Eel	○	○	○	I
Abramis brama (L.)	Bream	○	-	-	L
Alburnoides bipunctatus (BLOCH)	Riffle minnow	●	○	-	R
Alburnus alburnus (L.)	Bleak	E	○	●	I
Barbus barbus (L.)	Barbel	●	●	○	R
Carassius auratus gibelio (BLOCH)	Goldfish	-	E	-	I
Gobio gobio (L.)	Gudgeon	●	●	●	R
Leuciscus cephalus (L.)	Chub	●	●	●	R
Leuciscus leuciscus (L.)	Dace	○	●	●	R
Leuciscus souffia agassizi (RISSO)	Soufie	○	-	-	R
Phoxinus phoxinus (L.)	Minnow	●	●	-	I
Pseudorasbora parva (SCHLEGEL)	Asiatic minnow	E	E	-	I
Rutilus rutilus (L.)	Roach	●	●	●	I
Scardinius erythrophthalmus (L.)	Rudd	-	-	E	L
Tinca tinca (L.)	Tench	○	-	-	L
Barbatula barbatula (L.)	Stone loach	●	●	○	R
Esox lucius (L.)	Pike	-	○	○	I
Thymallus thymallus (L.)	Grayling	●	○	-	R
Salvelinus fontinalis (MITCHELL)	Brook trout	-	-	E	R
Salmo trutta morpha fario (L.)	Brown trout	●	●	○	R
Onchorhynchus mykiss (WALBAUM)	Rainbow trout	●	-	-	I
Lepomis gibbosus (L.)	Bluegill	-	-	○	I
Gymnocephalus cernuus (L.)	Ruffe	-	-	●	I
Perca fluviatilis (L.)	Eurasian perch	E	○	●	I
Stizostedion lucioperca (L.)	Pikeperch	-	-	E	I
Gasterosteus aculeatus (L.)	3-spined-stickleback	○	○	○	I
Cottus gobio (L.)	Bullhead	●	●	-	R
Number of species		21	18	17	28
Number of species excluding individual finds		18	16	14	23

At Oberriexingen, species specific abundances inside the fishway closely matched those from the sampling locations downstream, barbel and stone loach contributing over 65% and 75%, respectively, to the total number of fish caught. Particularly, the latter species was almost absent in the trap catches, whereas less abundant species like roach and European perch were found in relative high proportions (Figure 3). This discrepancy between the abundance of species inside the fishway compared to their numerical presence in the trap catches was even more pronounced at Eutingen. There, more than 430 gudgeon and 340 European minnows were caught inside the trap, while both these species were practically absent in the fishway. In contrast, only a singele bullhead, the most abundant species inside the fishway, which also occured in substantial numbers downstream and upstream, was ever caught inside the traps. The upstream and downstream locations at Eutingen showed the most numerically balanced community composition of all locations sampled. However, after the extention of the the turbine canal and the concomitant loss of an adequate attractant current connecting the fishway entrance with the main flow in spring of 1995, no graylings and only a few barbels were captured within a 150 m long section downstream of the fishway.

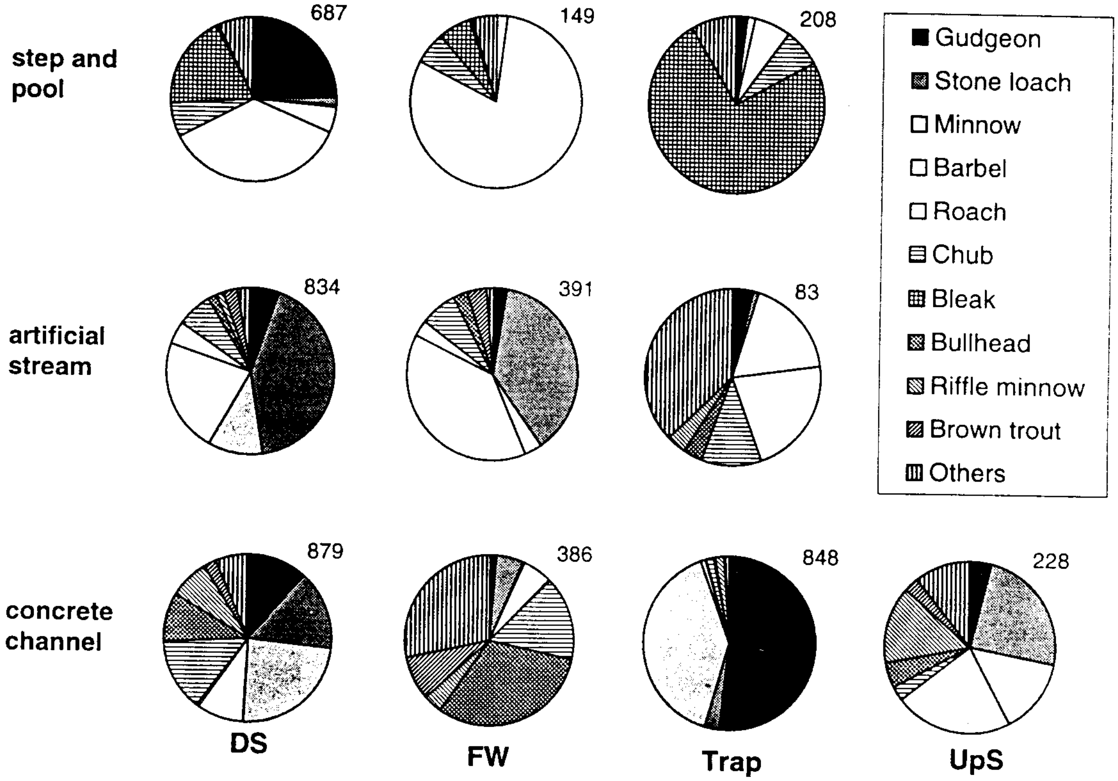


Figure 3: Percentage distribution by numbers of fish species caught downstream (DS), within (FW), with traps at the upstream exit (Trap), and upstream (UpS) of three types of fishways on the Enz River. The numbers beside each diagramm indicate the total number of fish caught.

A total of 1139 fish were captured inside the traps, 848 in Eutingen, 83 in Oberriexingen, and 208 in Besigheim (Figure 3). Barbel, gudgeon, chub, roach, and eel were the only species caught in all types of traps, indicating that these species successfully ascended the three fishways. In addition, bullhead, riffle minnow, and stone loach, which were never caught downstream of the step and pool fishway or were only present in very small numbers (stone loach), were, however, caught with the traps at the concrete bypass and the artificial stream. Based on the length-frequency distribution of the trap-caught individuals, no YOY fish, with the questionable exception of a single 5.3 cm long barbel caught in mid August in Eutingen (Table 3), swam through the upstream exit of either fishway, at least not during the respective trapping periods.

Of the 3554 fish >5 cm caught by electrofishing or net sampling, 772 were marked and released downstream of the fishways at Eutingen (n=273) and Oberriexingen (n=499). In addition, 226 and 38 individuals of the fish caught with traps at the concrete channel and the artificial stream, respectively, were also specifically marked and released just below the fishway entrance. None of these previously trapped fish were ever recaptured. In fact, of all the marked fish released downstream of the fishway at Eutingen, only 3 chubs, 1 rainbow trout, and 1 brown trout were recaptured in the 2%-slope section of the fishway. One specimen each of chub, rainbow trout, and brown trout were caught by net on May 5, 1995, three days after the initial capture. Of all the marked fish in Oberriexingen, 45 were recaptured. Of these, 38 fish were caught by electrofishing inside the fishway and two less than 20 m downstream of the fishway. In addition, two bullheads, one chub and one barbel were recaptured inside the traps. The two bullheads had been marked 82 days earlier and were released, like all the other recaptured marked fish, immediately downstream of the fishway. Relatively high recapture rates were recorded for brown trout (5 of 11 marked) and barbel (23 of 122), whereas only 8 of 267 marked stone loach were caught again. However, apart from the two bullheads, with 25 days (maximum 127 days) the median time until recapture was the highest for this species. Brown trout had the same median value as stone loach, but the longest time until recapture was 103 days. Barbels were recaptured after only 2-27 days.

DISCUSSION

With the exception of the nase (*Chondrostoma nasus*), among the 28 fish species recorded in this study, we found the full complement of taxa to be expected for this section of the Enz. Between Pforzheim and Besigheim, the river is transitional between the grayling and barbel zones of Huet (1959), who has successfully refined earlier schemes of fish-based zonal classifications for western European rivers. The nase has not been reported from the Enz River for many decades (Berg *et al.*, 1989), however, it can be assumed that, similar to most other comparable rivers, it once was the most abundant species. From a conservation standpoint, the frequent catches of substantial numbers of the riffle minnow, including many hundred YOY fish at the Eutingen site, and the presence of the soufie (*Leuciscus souffia agassizi*) is of considerable importance. Both species have been classified as 'strongly endangered' for the state of Baden-Württemberg (Hoffmann *et al.*, 1995), and, although they have been reported from the Enz in recent years, these reccords were mainly individual finds (Berg *et al.*, 1989; Steineck and Schöffel, 1994).

Apart from this larger scale view of community composition, we found pronounced changes in species distribution and abundances along the lower Enz River. For example, bullhead and grayling showed a strong pattern of declining abundances downstream and were not found near the mouth of the river. However intriguing, it is difficult to explain this distribution pattern with the above zonal distribution patterns, i.e. the physical habitat requirements of bullhead and grayling. Although the bullhead is commonly described as a typical species of the trout zone, preferring well oxygenated, cool, clean, and rapidly flowing water at shallow depth and rocky substrate (Huet, 1959; Lelek, 1987), this species is obviously considerably more plastic with regard to substrate conditions and water quality (Köhler *et al.*, 1993). One possible reason for the zero captures of bullhead and grayling at Besigheim is the considerably lower sampling effort compared to Eutingen and Oberriexingen. Perhaps more importantly, our sampling sites downstream of the weir and fishway were located in the natural river bed. During low flow condition, the bed receives less than 1/10 of the river's discharge which is perhaps not enough to stimulate grayling to move into this river section or to remain there. Furthermore, an outflow of a chemical company empties into the natural river bed, which may cause water quality to drop below tolerable levels, particularly for bullhead and grayling.

The percentage of marked fish that were recaptured in this study is quite low (6.5%). In particular, none of the more than 300 fish marked and released after their initial capture inside a trap were ever recaptured. From the results of laboratory studies, there seem to be no ill effects on fish survival and growth from alcian blue injections (Hill and Crossmann, 1987). However, to our knowledge, there are no field data to support this view and information on possible effects on migratory behaviour are also lacking. Therefore, the possibility must be considered that subcutaneous injection, or any other technique that requires handling and some degree of injury, causes sufficient metabolic stress to reduce individual upstream migratory activity for prolonged periods. This situation may exist at least in those cases were the migration through fish passes approaches a fish's physical limits. Severity of exercise differs with fishway design (Schwalme *et al.* 1985), and perhaps the fishway in Eutingen is very challenging. In a parallel study on an in-stream boulder ramp, we recaptured considerable numbers of previously marked fish with upstream traps, and some individuals ascended this fishway twice within a few days and after receiving a second mark (Kappus *et al.*, 1994).

The pronounced differences in the abundance and size distribution of many of the dominant species downstream compared to the fishway itself and to the trap catches at all sites, indicate species (and size) specific differences in the effectiveness of the respective designs for fish passage. Although our traps were not suitable for catching fish smaller 40 mm quantitatively, the complete lack and the very small numbers of YOY fish caught from within and at the upstream exit of the fishways in Eutingen and Besigheim, respectively, suggests that this age group is unable to ascend these fishways successfully. At least for the step and pool fishway, this is not surprising. Under normal operating conditions, we measured minimal current speeds at the fishway entrance with >0.8 m/s. From flume experiments, Stahlberg and Peckman (1986) have calculated critical swimming speeds for some forage fish, including adult stone loach and gudgeon, at 0.55 m/s. Although gudgeon were abundant downstream, only four specimen were ever caught inside the fishway or with the trap at Besigheim. At the same time, gudgeons were captured in large numbers inside the trap at the concrete bypass channel, suggesting that this species was actually migrating during this period. Again, the size spectrum of the trap caught gudgeons (as well as of European minnows and stone loach) was strongly biased towards the larger size classes compared to the specimen captured downstream of the fishway. Furthermore, the marked differences in relative fish abundance between downstream locations and inside the fishway itself, and in the relative numerical composition of the trap catch at Eutingen, clearly demonstrated that some species (bullhead, stone loach, chub, brown trout, and rainbow trout) use this fishways also as a habitat, whereas others (European minnow and gudgeon) seem to simply migrate through in relatively short time.

Although the artificial stream at Oberriexingen had been in full operation just a few weeks prior to our first electrofishing survey, 10 of the 16 species found regularly at this site were already present inside the fishway. Six months later, all species, with the exception of grayling, had been found inside the fishway. A similarly fast natural recolonization of desiccated stream habitats after reflooding has been documented by Bayley and Osborne (1993). Contrary to the other two fishways, we found early life stages of at least seven species inside the artificial stream. Most of these juveniles were captured in a rather deep pool area with armouring made out of overhanging willow branches. The importance of resting areas and shelter for the successful passage of fishways is well known (Jungwirth and Pelikan, 1989; Schmutz *et al.*, 1995), and their presence enables small species or juvenile fish to pass fishways which average current velocities largely exceed the respective sustainable swimming speeds (Schwalme *et al.*, 1985). Our results strongly reinforce this view and argue for more natural designs of bypass fishways. This at least in those cases were the more desirable solutions of avoiding barriers altogether or the construction of an in-stream ramp are not feasible.

REFERENCES

Bayley, P. B., and L. L. Osborne (1993). Natural rehabilitation of stream fish populations in an Illinois catchment. *Freshwater Biology*. 29 pp. 295-300.

Berg, R., S. Blank and T. Strubelt (1989). Fische in Baden-Württemberg. *Ministerium für ländlichen Raum, Ernährung, Landwirtschaft und Forsten Baden-Württemberg*. Stttgart, 158 pp.

Bridcut, E. E. (1993). A coded alcian blue marking technique for the identification of individual brown trout, *Salmo trutta* L.: an evaluation of its use in fish biology. *Biology & Environment: Proceedings Royal Irish Academy*. 93B pp. 107-110.

Hill, J., and G. D. Grossman (1987). Effects of subcutaneous marking on stream fishes. *Copeia*. 1987 pp. 492-495.

Hoffmann, R., R. Berg, , P. Dehus, R. Grimm and R. Rösch (1995). Fische in Baden-Württemberg - Gefährdung und Schutz. *Ministerium für Ländlichen Raum, Ernährung, Landwirtschaft und Forsten, Baden-Württemberg*. Stuttgart, 92 pp.

Huet, M. (1959). Profiles and Biology of Western European streams as related to fish management. *Transactions of the American Fisheries Society*. 88 pp. 155-163.

Jungwirth, M. (1984). Die fischereilichen Verhältnisse in Laufstauen alpiner Flüsse, aufgezeigt am Beispiel der österreichischen Donau. *Österreichische Wasserwirtschaft*. 36 pp. 103-111.

Jungwirth, M. and B. Pelikan (1989). Zur Problematik von Fischaufstiegshilfen. *Österreichische Wasserwirtschaft*. 41 pp. 80-89.

Kappus, B., J. Böhmer, W. Jansen, A. Müller, T. Beiter and H. Rahmann. Fischwanderungen im Bereich von naturnahen Aufstiegshilfen im Neckareinzugsgebiet. *Deutsche Gesellschaft für Limnologie* (Publ.): Erweiterte Zusammanfassung der Jahrestagung 1995, 24.-29. September in Berlin, *In Press*.

Kirchhofer, A. (1995). Schutzkonzept für Bachneunaugen (*Lampetra planeri*) in der Schweiz. *Fischökologie*. 8 93-108.

Köhler, C., A. Lelek and W. G. Cazemier (1993). Die Groppe *(Cottus gobio)* im Niederrhein. Merkwürdigkeit oder etablierter Bestandteil der Fischartengemeinschaft ? *Natur und Museum*. 123 pp. 373-404.

Lelek, A. (1987). *The Freshwater Fishes of Europe. Vol.9 Threatened fishes of Europe*. AULA-Verlag, Wiesbaden. 343 pp.

Ligon, F. K., W. E. Dietrich and W. J. Trush (1995). Downstream ecological effects of dams. *BioScience*. 45 pp. 183-192.

Moog, O., M. Jungwirth, S. Muhar and B. Schönbauer (1983). Berücksichtigung ökologischer Gesichtspunkte bei der Wasserkraftnutzung durch Ausleitungskraftwerke. *Oesterreichische Wasserwirtschaft*. 45 pp. 197-210.

Schmutz, S., H. Mader and G. Unfer (1995). Funktionalität von Potamalaufstiegshilfen im Marchfeldkanalsystem. *Österreichische Wasser- und Abfallwirtschaft*. 47 pp. 43-58.

Schwalme, K., W. C. Mackay and D. Lindner (1985). Suitability of vertical slot and denil fishways for passing north-temperate, nonsalmonid fish. *Canadian Journal of Fisheries and Aquatic Sciences*. 42 pp. 1815-1822.

Simons, D. B. (1979). Effect of stream regulation on channel morphology. In: "*The Ecology of Regulated Streams*", Ward, J. V. and J. A. Stanford, eds., Plenum Press, New York. pp. 95-111.

Soerensen, T. (1948). A method of establishing groups of equal amplitude in plant sociology based on similarity of species content. *Biol. Skr., K. Danske Vidensk. Selsk.* 5 pp. 1-34.

Spence, J. A. and H. B. N. Hynes (1971). Differences in fish populations upstream and downstream of a mainstem impoundment. *Journal of the Fisheries Research Board of Canada*. 28 pp. 45-46.

Stahlberg, S. and P. Peckmann (1986). Bestimmung der kritischen Strömungsgeschwindigkeit für einheimische Kleinfische. *Wasserwirtschaft*. 76 pp. 340-342.

Steineck, W. and S. Schöffel (1994). Limnologische Untersuchungen zur Effizienzkontrolle an der umgestalteten Enz in Pforzheim hinsichtlich der Entwicklung der Fischfauna, Wirbellosenfauna sowie der Gewässergüte. *Veröffentlichungen Projekt Aangewandte Ökologie*. 8 pp. 245-264.

Ward, J. V. and J. A. Stanford (1983). The serial discontinuity concept of lotic ecosystems. In: "*Dynamics of Lotic Ecosystems*", Fontaine, T. D. and Bartell, S. M., eds., Ann Arbor Science, Ann Arbor, MI. pp. 29-42.

Weaver, L. A. and G. C. Garman (1994). Urbanization of a watershed and historical changes in a stream fish assemblage. *Transactions of the American Fisheries Society*. 123 pp. 162-172.

Zalumi, S. G. (1970). The fish fauna of the lower reaches of the Dnieper: its present composition and some features of its formation under conditions of regulated and reduced river discharge. *J. Ichthyol.* 12 pp. 249-259.

GUIDING SALMONOIDS IN A BIG RIVER TO FIXED FISH TRAPS OR TO FISH PASSAGES

Matti Herva
Keminsuuntie 91, FIN-94400 Keminmaa or
University of Oulu, [illegible] and Water Resources Engineering Laboratory
Box [illegible] FIN-90101 [illegible]

ABSTRACT

[illegible] been characterised by the spring flood rising from minimum discharges to 3000 and even 4500 m^3/s and summertime flows prevailing above the mean of 530 m^3/s and less towards fall. The water temperature increases in May, when ice breaks, and crosses [illegible]s. The midsummer wa[illegible] below [illegible]

For the fish there were no impassable falls in the Kemijoki River. Cold water and strong flow during the high flood [illegible] impeded ascending [illegible] of salmonoids had to be [illegible] nets [illegible]

The big salmon weirs extended to the middle of the broad river. They were conventially placed and constructed for guiding and trap[illegible] fish during several [illegible]tion of the salmon [illegible] several means to catch fish. As the season advanced they moved from the shore to the middle of the river while they changed the type and the size of the traps to the extensive undertaking of the big salmon weir. But all the time they caught active individuals of the salm[illegible]s surpassing the prevailing flow. The fish [illegible]his can be controlled by guiding in flow velocities over it's threshold level and more successfully at the critical velocity. For the fishermen it was crucial to catch enough salmon from their river to cover the costs of the fish weirs, to pay taxes and still have enough left over to make [illegible]ng. This pressure to make return from fishing refined the practical skills of salmon weir making.

The fishways at natural waterfalls may have various entrances along the shore. Many fishways at dams, barrages or power plants have been provided and accomplished with several entrances as well. Some barriers of the anadromous fish migration have parallelly several fishways e.g. for the fish of very different st[illegible]s and locomotory skills. They may be situated at the banks and in the middle section of the river. Extra discharge as a call and cue for the fish may be indispensable. Because the fishways have been built on the banks of the river, the entrances are not far from the shoreline; here extra volume and velocity of water is needed to produce an approach area and make the entrance sensible to the fish. At the entrance the fish behaves in the same way as when swimming into the fish trap. The entrances in the middle lane of the river to the Borland fishway and as a part of the fish collection installation of a wide wall of the power plant have been apparent enough for the ascending fish. It seems quite obvious that similar principles for guiding fish and analogical human activities for managing discharge could be adapted usefully in the riverfishing and in providing proper passages for salmonoid fishes.

Keywords: Aboriginal salmonoid fishing / Fish rheotaxis / Flow / Temperature of water / Swimming capacities of fish / Open mainstream / Fixed salmon weirs / Fishways / Fish passage

INTRODUCTION

The purpose of this paper is to review fish rheotaxis in rivers and complicated flow fields. Furthermore, some conceptual pictures, especially of the migration speed and distribution of migratory fish in the river channels, will be presented.

The paper will also describe fixed trap constructions for rheotactical fish and the ways of creating flow conditions to concentrate fish with salmon weirs. These methods were commonly used in the River Kemijoki in Northern Finland from the early ages until 1930's. This art of modifying flow conditions suitable to the fish weirs to match the natural rheotaxis of the fish during their river migration, is part of the ancient Finnish know-how.

The aboriginal methods and ways of salmonoid fishing off the river were developed basing on the knowledge of how to modify flow conditions for the fish to make them swim into a trap which then was to be emptied from time to time (Sirelius 1906, Naskali 1993). This applies to various fish traps situated in streams (Fig. 1 and 2). In 16th and 17th century, the so called pen weirs (fig. 2 and 3) (karsinapadot) were taken in to use in river fishing (Vilkuna 1975 a, b).

With the help of these weirs it was possible to consistently catch a considerably grate numbers of salmon (Fig. 5), trout and whitefish from big, wide rivers without hindering these fish from passing along the deepest part of the river i.e. in the mainstream. It was not allowed to close the migration route of mainstream by barriers of the ascent of the Baltic Salmon in it (Fig. 3).

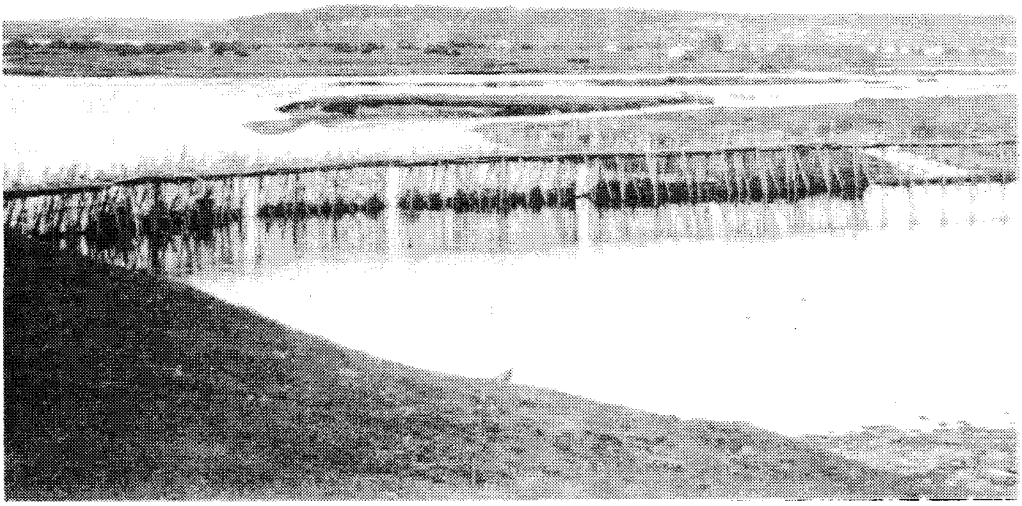


Figure 1: The salmon weir of the Hervanpudas in the Kemijoki River, 1920's

The purpose of a wall in a fishweir was to interrupt and slow down migratory fish from ascending in the river. The effect of a big fish weir on the whole in this sense resembles that of a natural flow obstacle. Unlike technical dam constructions and natural migration obstacles, such as rapids and waterfalls, the purpose of an old fish weir was not to dam up the water. A weir as a construction in the water creates flow conditions which then have an effect on the fish ascending behaviour. Pen fish weirs do not merely serve a function of being traps situated on a meaningful spot in a river channel to ensure good catches, but rather they are there to make flow conditions ideal for this trap to catch even more fish. Simultaneously the water velocities in the open mainstream become slightly higher.

The present strive to restore the migration cycle of salmonoids in the River Kemijoki with the help of fish passages benefits from this review on the subject of how to match and use the rheotactical need of the fish at old fisheries.

INTERACTION BETWEEN THE MIGRATORY FISH AND THE FLOW

The ascent of migratory fish in the river is not like an even flow of traffic. The fish are distributed unevenly in the river both horizontally and in the cross section of the river flow. The individual fish within different species of rheotactical and anadromous schools of fish are affected by physical factors, such as factors connected with flow conditions, light conditions, water temperatures and the weather, as well as biological aspects related to fish and school of fish. The velocity of the water flow, the turbulence and the gradients in it as well as the differences in water velocities in different parts of the cross section are qualities affecting the fish. Naturally, the direction of the velocity and strong and wide whirls in the flow have an effect on the progress of the ascending fish (Pavlov 1980).

The temperatures suitable for ascent swim depends on the fish species. Pelagic stream fish get the visual marks vital for their ascent in the river by seeing formations of river bed materials in certain depths and in transparent lit water. The fish who orientate according to their tactile senses swim in unlit conditions above the river bed formation with the help of some pressure related perceptions. The salmon swims without the help of light in certain levels of turbulence in the flow, e.g. when ascending in darkness through fishways.

The swimming ability of fish based on sensonal and motorical skills is related to their species. Yet, the swimming ability of an individual fish is dependent on ontogeny and actual temporal, seasonal events and changes. In big rivers the salmon stocks which have differentiated from the main population may ascent during characteristic times which deviate from the usual seasons.

All fish have a basic inborn rheotaxis in them i.e. a facility to manage in various flow conditions of a water system (Pavlov 1980). The old salmon fishing took heed of this rheotaxis of the salmonoids (Sirelius 1906, Hoikkala 1948, Vilkuna 1975 a,b, Naskali 1993, Herva 1995).

Rheotaxis comes forth in the form of orientation to and in the flow. The sense perceptions transmitted by the sense organs of the fish direct the orientation, and the events the fish can sense while swimming, launch swimming reactions. As a basic reaction in swimming behaviour the rheotaxis comes forth in the velocities between threshold and critical velocities which are typical for fish species (Pavlov and Pahorukov 1978) (Fig. 5).

Within this range of water velocities the other swimming reactions of fish also vary considerably. In other words, they are relative to the velocity of the water. The intensitivity of the reactions to avoid a particular flow and to seek for a stronger flow can be defined in relation to those velocities prevailing in the water within the velocity zones of threshold velocity, travelling speed and critical velocity. On the other hand, the fish avoids the chinky wall of the weir when approaching it from upstream (Pavlov and Pahorukov 1978). In river fishing these principles are important aspects.

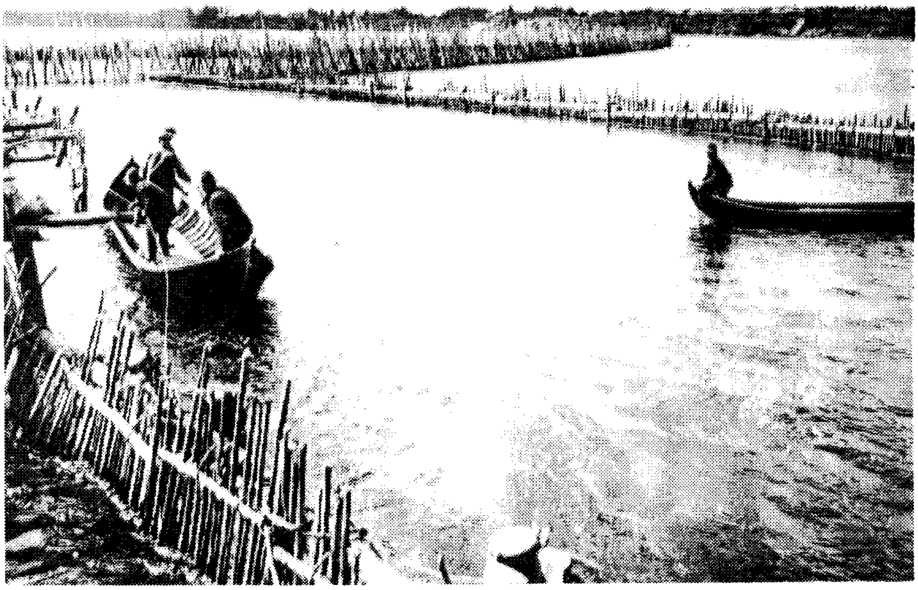
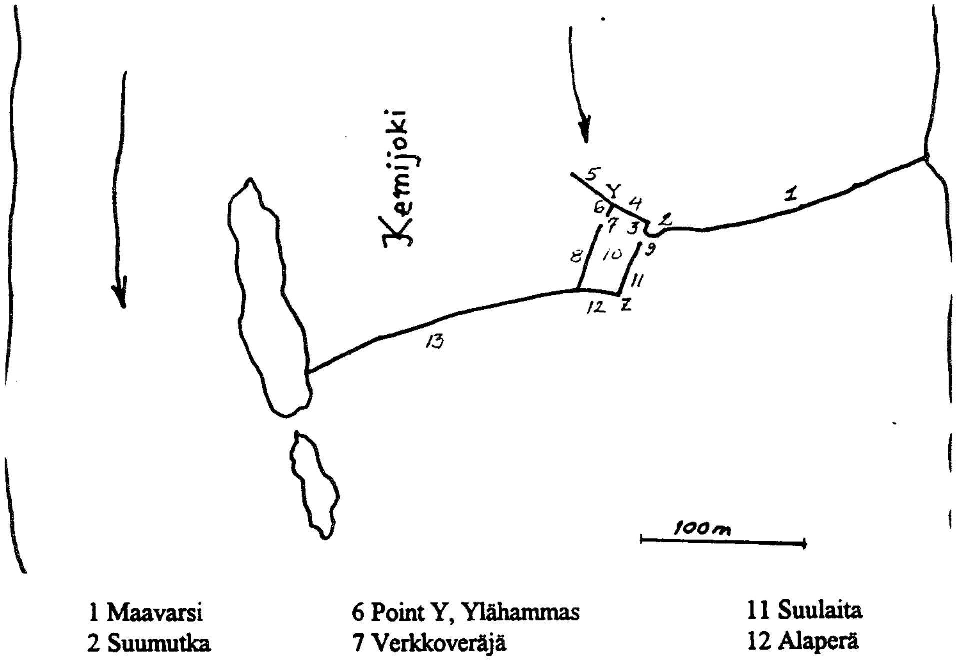


Figure 2: The salmon pen weir of Korva in the Kemijoki River (Photo by T.H. Järvi 1922).



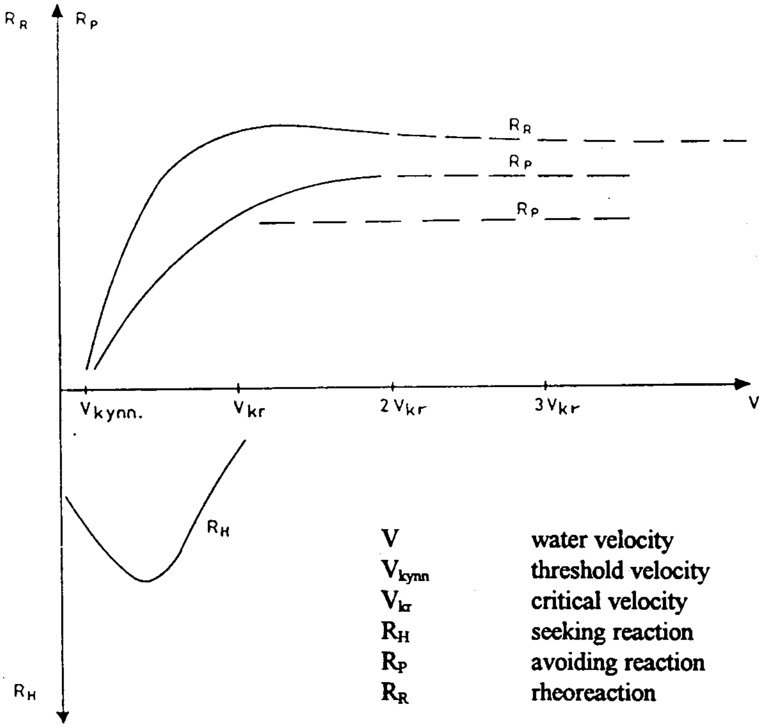
1 Maavarsi
2 Suumutka
3 Alahammas
4 Yliperä
5 Yliperän sarvi

6 Point Y, Ylähammas
7 Verkkoveräjä
8 Selkälaita
9 The Salmon gate, Lohiveräjä
10 The pen trap, Karsina

11 Suulaita
12 Alaperä
13 Vääravarsi
14 Point Z

Figure 3: The functional parts of the pen weir of Korva. The Kemijoki River flows southwards having the main channel on the right (west) side of the river (Hoikkala 1948).

Figure 4: The catch of the year 1922 in Korva was 3500 salmon making total of 39000 kg (Photo by T.H. Järvi).



V	water velocity
V_{kynn}	threshold velocity
V_{kr}	critical velocity
R_H	seeking reaction
R_P	avoiding reaction
R_R	rheoreaction

Figure 5: The intensity of the swimming reactions of the fish against water velocities of the positive rheotropism (Pavlov & Pahorukov 1983).

Above water velocities are referred to as if they were even and evenly distributed factors in cross sections. But in natural flow channels water velocities vary. In the water space the fish are able to sense velocities which have gradients. From them the fish choose a velocity zone. In these flow conditions fish find it stable to exceed the velocity of the flow, i.e. fish swim to the direction of the gradient. Thus, the higher the velocity of the flow, the faster the fish swim, naturally within the range of it's swimming capacity. This kinesis, i.e. reaction of the animal in proportion of the quantity of the stimulus, emerges alongside with the rheoreaction (rheotaxis) of turning into the flow.

APPLIED RHEOTAXIS IN THE SALMONWEIRS

The wall of a fish weir stops the fish which have ascended to the flow below it from migrating further in the river. The weir has a wall with chinks (Fig. 1, 2 and 6) that are narrow enough to prevent the fish from escaping through them. The structure of the wall changes the flow conditions below. There are rods which have been placed horizontally against the frame of the wall. This framework was stabilised by heavy stoneweights against the pressure of the current. The birch trees with their crowns left untouched had been pushed into the flowing water to stand upside down against the river bottom. They were supported by the horizontal rods and created flow conditions in which the water velocity and the turbulence are suitable for the fish. In places where the velocities are near the lower threshold velocity the fish seek for stronger flows. When coming to a slot of the wall (Fig. 1) where the horizontal basket traps ('lana') (Fig. 7) have been situated to catch, the fish ascend to the trap lead by the high velocity water flows. The etological concepts 'taxis' (i.e. turning into the flow) and 'kinesis' (i.e. by swimming exceeding the velocity in the flow) are factors describing a strong swimming performance. On the other hand, under the tendency to seek into the flow the salmon in susceptible to fall into the J-shaped sack nets ('potku') (Fig. 8) below the wall of the weir.

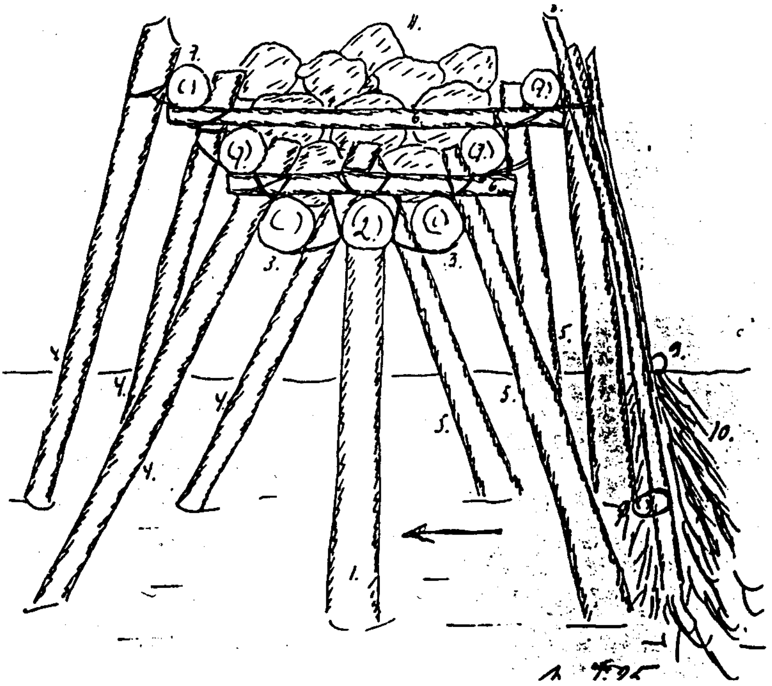


Figure 6: The cross section of the structure of the wall of the weir of Korva (Hoikkala 1948).

Figure 7: The horizontal basket trap from the salmon weir of the Figure 1. (The Rural Museum of Keminmaa. Photo by Jaakko Heikkilä 1996).

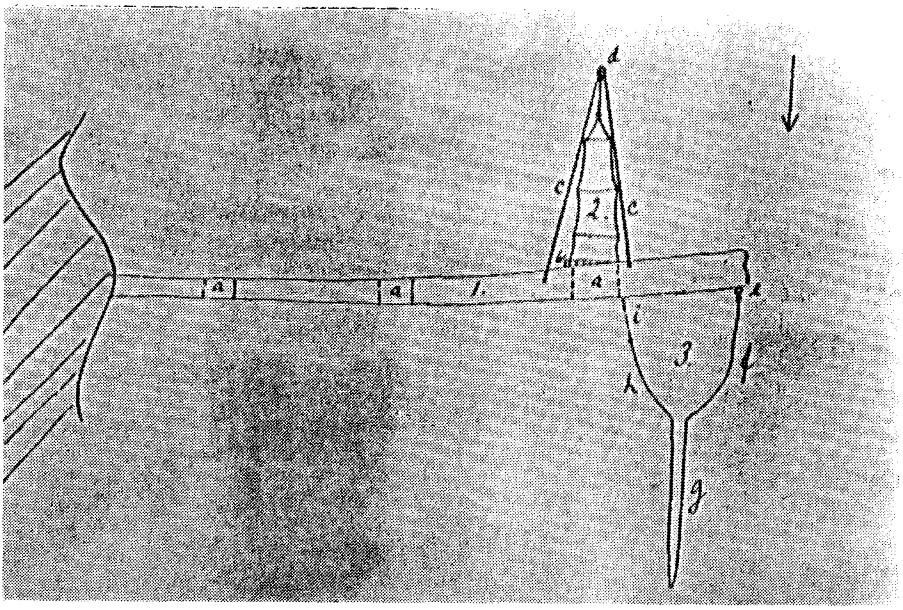


Figure 8: The J-shaped sack net for salmon together with a basket trap in a shore weir (Hoikkala 1948).

The entrance of the pen weir ('karsinapato') in Korva (Figures 3 and 9) was built every summer on the very same fixed spot. The weir consisted of two walls ('varsi') and a pen trap ('karsina') which imprisoned the fish. The weir also included other constructions effecting the flow conditions (Fig. 3). 'Yliperän sarvi' was a special wall structure which lead the water flow past 'alahammas' to bent formation in the weir called 'suumutka'. In this formation the chinked wall of the weir was bent and a high discharge went through it to the salmon gate entrance ('lohiveräjä' Fig. 3 and 10). It was said that the side walls of the pen ('maavarsi' and 'vääråvarsi') acted as pulling and pushing devices to get the salmon in to the pen trap. In other words, a rheotactical salmon when coming below a big salmon weir, were directed in to the mouth of the pen trap with the help of the flow modified by these means in the weir. The velocity of the water was at its highest in the area where the pen trap was located (Fig. 10 and 11). This large trap (25x40 m) (Fig. 3) was placed in the river by using a chip of wood which was put in the water to float downstreams from 'ylähammas', point Y, as far as the length of the diagonal beam of the pen trap. Then point 'Z' i.e. the opposite corner to point 'Y' was fixed on to that spot in the stream. The fences of the pen trap ('selkälaita' and 'suulaita') were made of series of vertical rods in a form of a lattice and of salmon nets respectively. The lattice as well as all the supporting poles and the rods of the framework were made of fir.

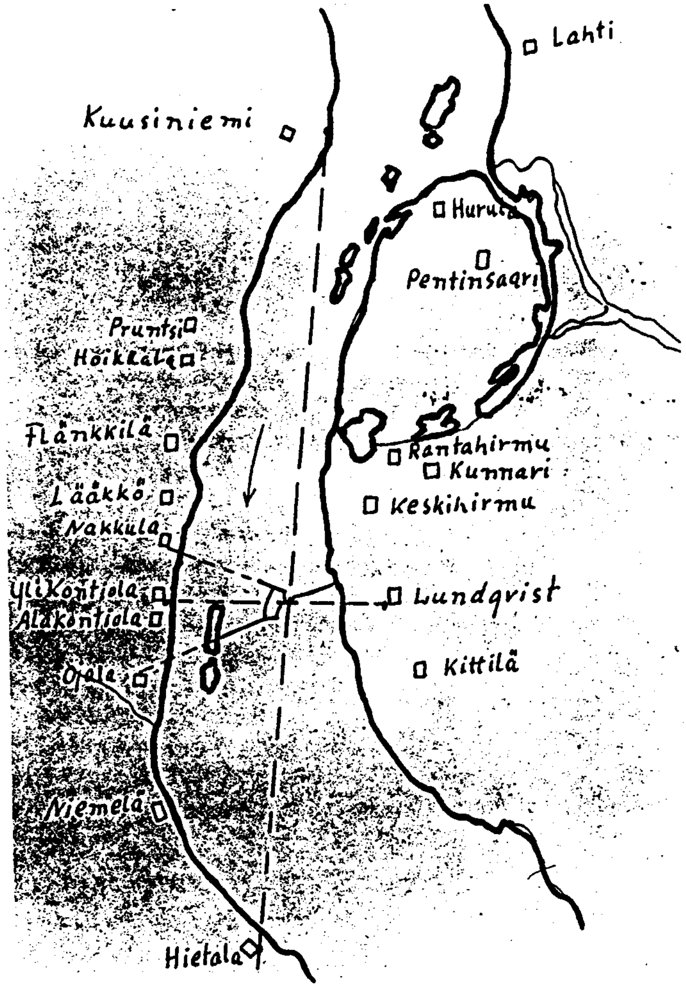


Figure. 9. The Korva salmon weir was lined and built after fixed landmarks. Scale 1:20000 (Hoikkala 1948).

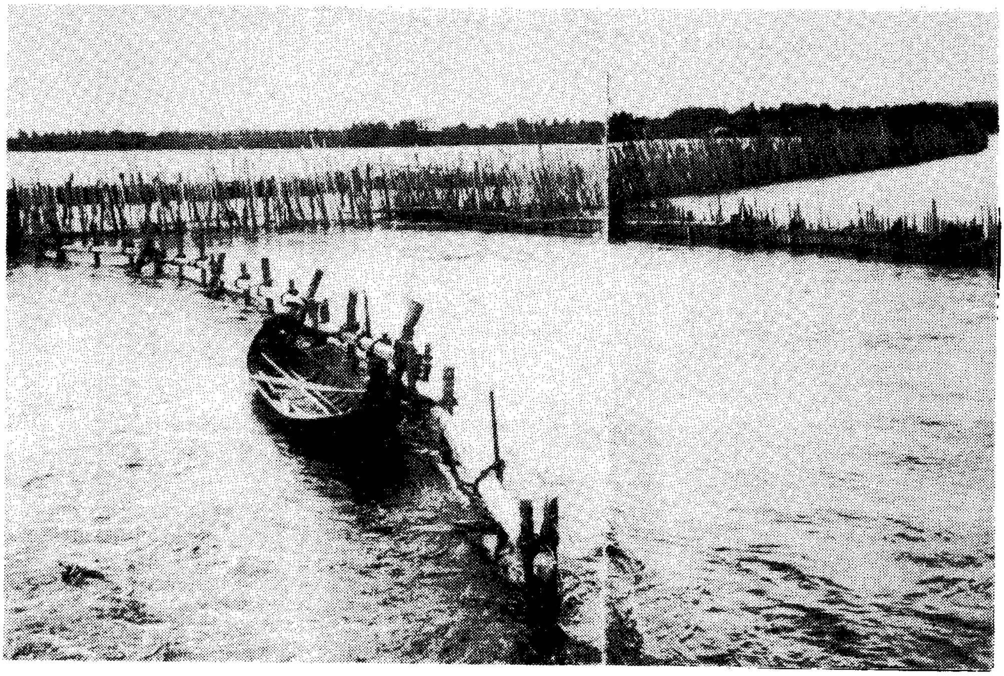


Figure. 10. The strong flow at the salmon gate guides the fish into the pen trap. (Photo by T.H. Järvi 1922)

Thus, the flow in the pen was increased with the help of the weir structures located in the stream above the trap. The bent formation in the wall called `suumutka` created a strong flow and a big whirl outwards from the pen. Fish swimming into this pen was based on their ascent in the strong flow.

What is it then that makes a pen weir a trap eventhough its mouth is wide open and there are no gate constructions? The entire explanation lies in the fish themselves. During the ascent migration the fish use their most powerful swimming capacity and they will not stay to be lead by the mouth streams. On the contrary, the fish approaching the entrance stream from the inside of the pen trap are lead by their rheotaxis upstream towards the upstream part fence and the wall of the pen. They are kept in the pen trap by their rheotaxis.

Individual fish which have been swimming in the penstream to the vicinity of the entrance and near the downstream wall of the pen, must escape and swim upstream to avoid the chinky downstream wall of the pen.

A salmon weir was a trap formed by its constructions and created and modified flow by those constructions.

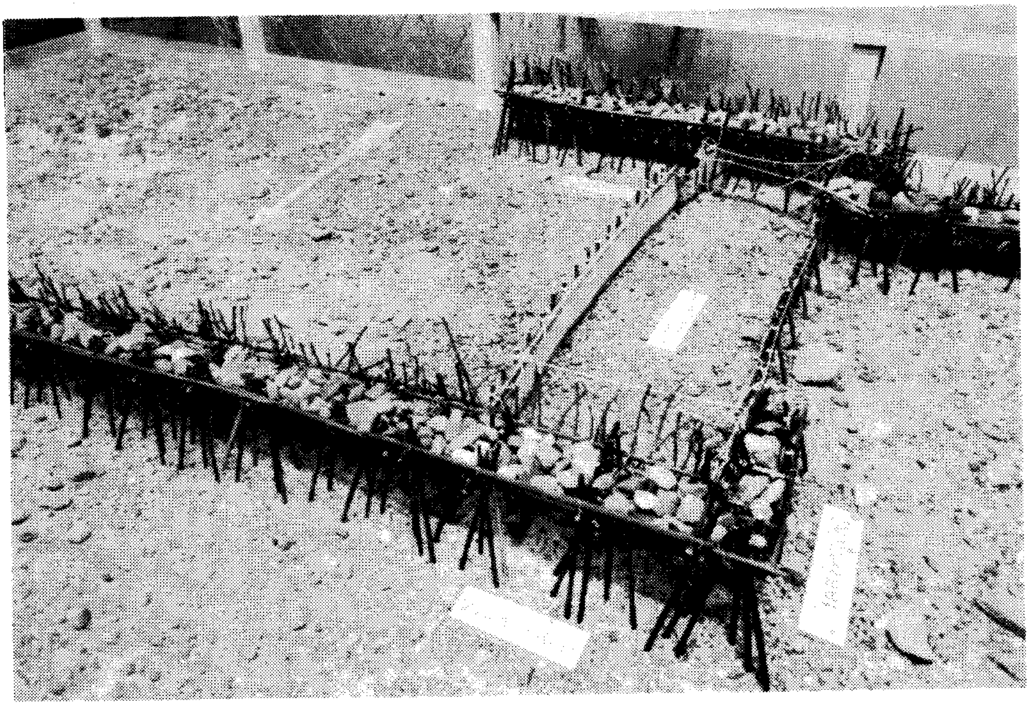


Figure. 11. The salmon pen trap illustrated by a scale model. (The Municipal museum of Kemi; photo by Jaakko Heikkilä 1996)

IMPORTANCE OF RIVER FLOW FOR FISHERIES AND FISHWAYS

The fishery management policy has of old ordered all the rivers of migratory fish in Finland to be left open according to the 600 years old Swedish law. Later, from the 16th century, particular restrictions were stipulated for concerning the open channel in the mainstream when building big salmon weirs. The free access of fish was vital for fisheries aswell, as it had a major importance for the Kemijoki River in the legth of 120 km.

Modern concession of dam building on streches of river which carry migratory fish may be permitted in Norway, if the constructions do not prevent fish from migrating upstream. Flows for fish swimming are stipulated. Power companies pay all the needed measures. Among the countries which have a long tradition and experience in building fishways Norway has solutions in locating one or, if needed, several entrances of the fishway to the flowing approach area below the obstruction (Grande 1995, Herva 1994).

Scotland is the home of the Borland fishway. In the River Beanly there are the entrances to the Borland lift at the middle lane of the river amidst the turbines (Fig. 12). The success in guiding and passing Atlantic salmon over the two power plants has proved to be fine (Herva 1994).

Canadian and American fishways with hydraulic fish collection installations have proved to be effective in trapping and passing salmon aswell (Fig. 13). There are the entrances of the fishway in the middle of the power canal.

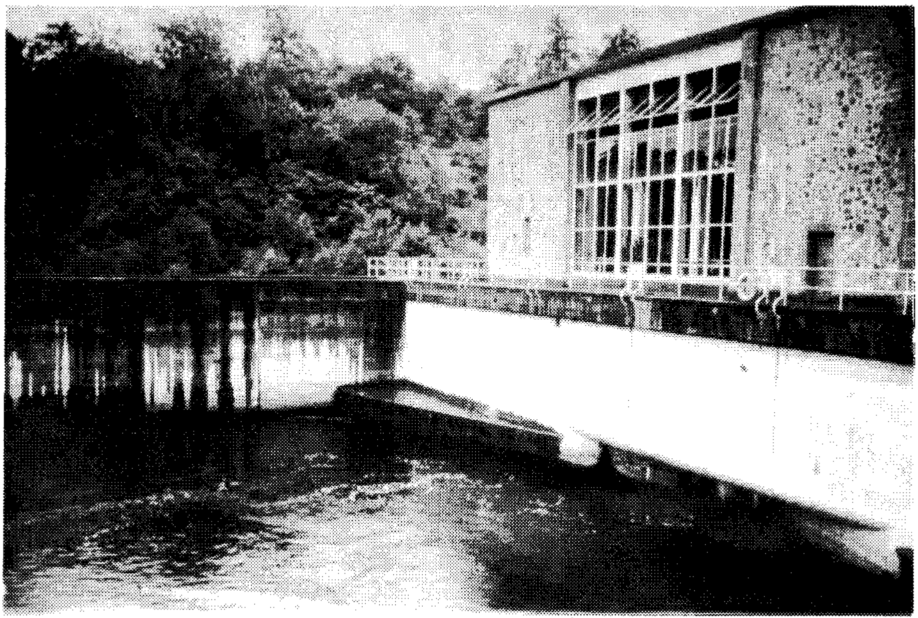


Figure 12: The entrance of the Borland fishway between the two turbines of the powerplant Kilmorack, River Beauly, Scotland (Photo: Matti Herva 1979).

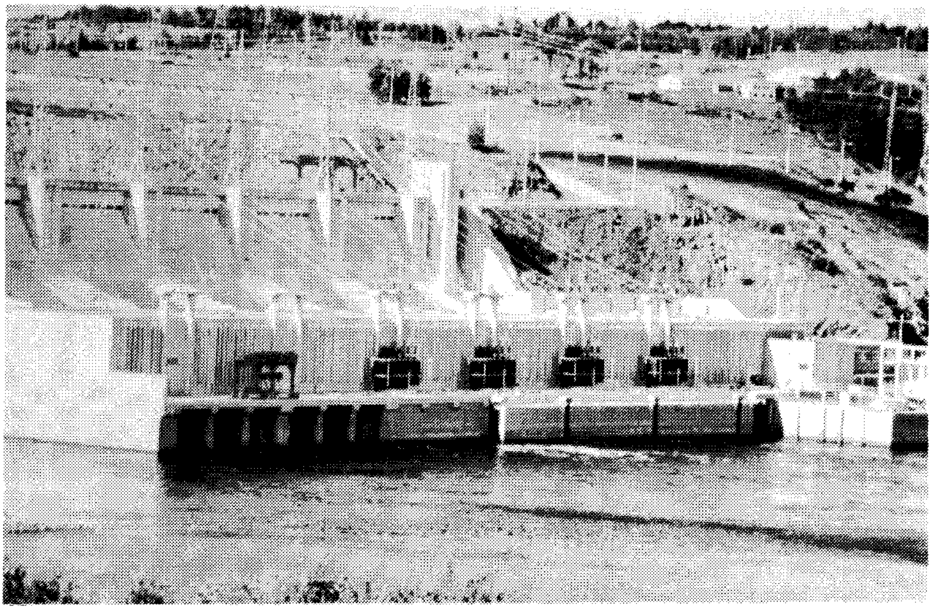


Figure 13: The entrances of the fish collection installation in the downstreem face of the power plant of Mactaquac, River St. John, N.B. Canada (Photo: Matti Herva 1974).

Constructional elements of the old salmon weir were functional factors for guiding fish, too. The whole building was made to achieve certain flow for the traps and the fishing itself.

The construction and the function of the fishway are analogic to these of the salmon weir.

CONCLUSIONS

For salmon fishermen it was crucial to catch enough salmon from the river to cover the costs of the fish weir, to pay the taxes and still have enough left over to make a living. This pressure to make money from fishing refined the practical skills of salmon weir making.

The old skills together with the knowledge of flow conditions in the river can still be applied today. To build a passage through a dam with a fishway, we need knowledge based on fish behaviour and information on flow patterns as well as other vital properties of the target location. Naturally a sufficient discharge and other flow conditions are needed (Fig. 14), so that the fish will swim into the passage and go through it (Fig. 15). Thus, the fish will be passing the migration obstacle.

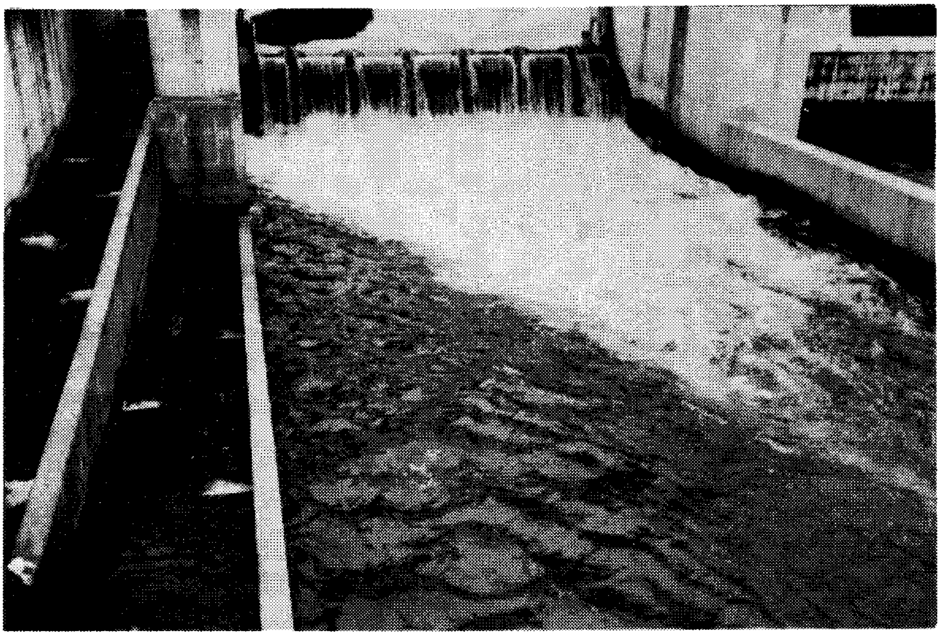


Figure. 14. The technical fishway is a functional element of the dam in the Natori River, Sendai Japan, and provides passage for the fish. (Photo by Matti Herva 1990)

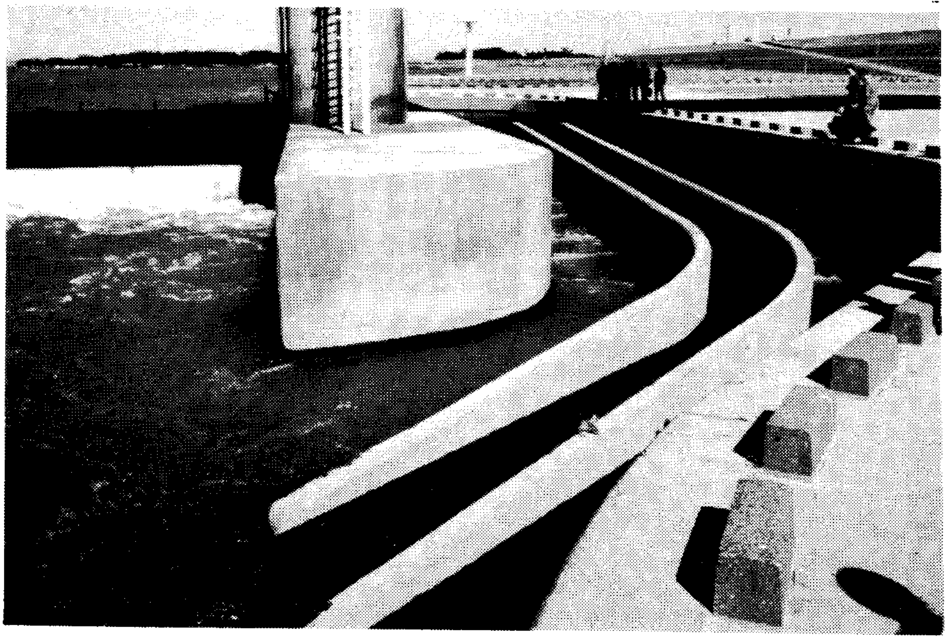


Figure. 15. At the eastern abutment of the barrage in the River Kiso the guiding of fish through the passage is accomplished by discharge and flow arrangements for a sufficient approach area. (Photo by Matti Herva 1990)

EPILOGUE

Traditionally both movable and fixed fish traps have been used in river fishing. In the River Kemijoki weir fishing, in particular, became the method to catch Baltic Salmon. From the 16th century all the way to the 20th, until the era of water power stations, most of the salmon (over 100 tons/year) was caught with weirs. A considerable number of the salmon weirs were situated along the 120 km long downstream of the 500 km long river.

Despite the fact that the systematic salmon weir fishing, following the regulations (Malmgren 1869), continued for 350 year, the salmon population of the river survived and remained fairly stable. There was an official mainstream in the river in which fixed traps were not allowed.

The rheotaxis of the salmonoid in general, as well as their ability to ascent to their spawning areas along the open mainstreams in high water velocities, can be seen as the explanation to the fact that during 350 years, over 50 generations of salmon and 10 generations of fishermen lived from the same river. The era ended in 1950's as the power stations took over the river and destroyed both the salmon population and the river fishing. The salmon stocks which survived are nowadays threatened by overfishing and adverse effects of pollution in the Baltic Sea. Let's hope that the story of the Baltic Salmon will not end during the last decade of this century. Building fishways would help in restoring the migration of salmonoids to their now unpopulated production areas in rivers.

BIBLIOGRAPHY

Grande, R. (1995). Fishway solutions in connection with Hydro-Electric plants and regulated rivers in Norway. The *International Symposium on Fishways 1995 in Gifu, Japan*, 7 p. In English.

Herva, M. (1994). Fiskevandring og systematiske Fiskeveiobservasjoner. In: Kamula, R. (ed.) Travelling fishways. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory*. A:59. pp. 10-20. In Norwegian.

Herva, M. (1995). Vanha jokikalastus käytti hyväkseen lohikalojen virtahakuisuutta. *Suomen Kalastuslehti 3/1995*, 3 p. In Finnish.

Hoikkala, A (1948). Kemijoen alajuoksun lohenkalastuksesta. Laudaturtyö, *Helsingin yliopisto. Sanakirjasäätiön arkisto*. Käsikirjoitus. In Finnish.

Malmgren, A.J., (1869). Handlingar och förordningar angående Finlands fiskerier. Helsinfors. 288 p. In Swedish.

Mantejfel, B.P., Pavlov, D.S. & Pahorukov, A.M. (1978). Kalansuojauslaitteiden ja kalateiden rakentamisen biologiset perusteet, In: Kalojen virtakäyttäytyminen neuvostoliittolaisten tutkimusten valossa. Käännöksiä neuvostoliittolaisista julkaisuista. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory*. A:33. 1988. In Finnish.

Naskali, E. (1993). Kadonnut jokipyynti. In: Manninen, K ja Westman, K. (ed.). Pyyntitavat ennen ja nyt. *Kalastusmuseoyhdistys ry. Helsinki*. pp. 21-34. In Finnish.

Pavlov, D.S. (1980). Kalojen virtakäyttäytymisen ohjaaminen. In: Kalojen virtakäyttäytyminen neuvostoliittolaisten tutkimusten valossa. Käännöksiä neuvostoliittolaisista julkaisuista *University of Oulu, Hydraulic and Water Resources Engineering Laboratory*. A:33. 1988.

Pavlov, D.S. & Pahorukov, A.M. (1983). Biologiset perusteet kalojen suojaamiseksi joutumasta vedenottolaitteisiin. In: Kalojen virtakäyttäytyminen neuvostoliittolaisten tutkimusten valossa. Käännöksiä neuvostoliittolaisista julkaisuista. *University of Oulu, Hydraulic and Water Resources Engineering Laboratory*. A:33. 1988. In Finnish.

Sirelius, T. (1906). Über die Sperrfischerei bei den Finnisch Ugrischen Völkern. *Sociètè Fenno-Ougrienne. Helsinki*, 525 p. In German

Vilkuna, K (1975). Lohi. Kemijoen ja sen lähialueen lohenkalastuksen historia. Otava. Keuruu, 432 p. In Finnish

Vilkuna, K (1975). Unternehmen Lachsfangst. Die Geschichte der Lachsfischerei im Kemijoki. *Suomalaisen kirjallisuuden seura. Studia Fennica 19. Helsinki*, 545 p. In German.

INTEGRATING STRUCTURAL AND BEHAVIORAL FISH PROTECTION AT A MAJOR PUMPED-STORAGE HYDROPOWER DAM

John M. Nestler, Gene R. Ploskey, and L. Toni Schneider
U.S. Army Engineer Waterways Experiment Station
3909 Halls Ferry Rd
Vicksburg, MS 39180
USA
nestlej@ex1.wes.army.mil
gene.r.ploskey@NPP01.usace.army.mil
schneil@ex1.wes.army.mil

Gary Weeks
AScI Inc.
Trotters Shoals Research Facility
Calhoun Falls, SC 29628
USA
gweeks@sunbelt.net

Abstract

A large scale, fully operational, integrated fish protection system was developed for Richard B. Russell Dam, a Corps of Engineers pumped-storage hydropower facility with 640 MW conventional generation capacity and 340 MW pumping capacity, on the Savannah River between Georgia and South Carolina. The fish protection system, designed to operate during pumping operation only, combines: 1) knowledge of seasonal and diel movement patterns of fishes to develop guidelines to restrict pumping to periods of minimal fish entrainment potential; 2) detailed physical and 2-dimensional numerical hydraulic modeling to identify high velocity entraining flow zones, low velocity zones, and slack water zones; 3) an acoustic repulsion system employing high-frequency sound to divert blueback herring out of the entraining zone and into low velocity or slack water zones; 4) banks of high pressure sodium incandescent lights located in the low velocity-slack water zones to attract and hold fishes during pumping operation; and 5) a veneer made of 0.32-cm wedge wire on 5.08-cm centers that is placed directly over the trash racks to divert fishes larger than about 35-cm in length from the trash racks. Yearlong full recovery net monitoring supplemented by fixed aspect hydroacoustics sampling using two of the four pumped-storage units demonstrates the effectiveness of the fish protection. Tests compared numbers and lengths of entrained fish collected in a full-recovery net in the forebay under several treatments: (1) day versus night; (2) sound on versus off; (3) strobe-light on versus off; and (4) before versus after installation of the bar-screen. Attracting-light tests compared relative densities of fish in lit and adjacent unlit tailrace areas. In four nights of sound and one night of strobe-light tests, two 15-minute "on" treatments were randomly selected in each of six 1-hour strata. Bar-screen tests compared hourly entrainment and length-frequencies of fish collected in night pumps (7 before installation; 9 and 45 after). The total cost of the system was less than one million dollars. Integrating separate fish protection technologies into a comprehensive fish protection system can be used to increase fish protection at hydropower dams.

KEY-WORDS: Fish Entrainment/ Pumped-storage Hydropower/ Physical Fish Barriers/ Fish Protection/ Turbine Mortality/ Behavioral Barriers/ Fish Passage/ Richard B. Russell Dam/ Bar Screen

INTRODUCTION

Pumped-storage hydropower dams have reversible turbines that can generate like conventional hydropower dams during periods of peak demand. During periods of lower demand pumped-storage dams will use power to pump water back upstream into the forebay. Water pumped back is used later to generate power during periods of peak demand. Pumped-storage dams can have substantial impacts on afterbay (tailrace) fish communities (ASCE 1995). Many fishes concentrate in the dam afterbay either because the dam blocks seasonal migrations to historical spawning sites or because of attracting flow or water quality conditions associated with generation. In the afterbay, reduced conveyance area in the headwaters of the downstream reservoir will result in relatively high water velocities in the general area of the dam increasing the entrainment risk for fishes. The U. S. Army Corps of Engineers, state and Federal resource agencies, and several private natural resource advocacy groups were concerned that pumpback operation and increased conventional generation could severely impact the afterbay fish community. A successful fish-protection system (FPS) was considered to be a necessary part of pumpback operation.

This paper describes the planning process used to select and design a fish protection system for the pumping phase of pumped-storage operation and the design of the FPS. The planning led to the design of a fish protection system that integrates operational restrictions, hydraulic modeling results, and structural and behavioral protection technologies. Evaluation of system effectiveness had two objectives First, we wanted to test those components that could be subjected to on and off treatments (strobe lights and high-frequency sound) or those affecting different areas (attracting lights) to quickly decide whether to modify the installation. Second, we sought to evaluate diel timing of pumpback by comparing mean entrainment rates during night and day tests and bar-screens by comparing mean rates before and after installation.

Behavior modification methods for protecting fish at hydropower intakes by altering fish movements or distributions have received considerable attention. Lights have been used to control fish movements at power plants (Hocutt 1980) by excluding them from intakes (Patrick et al. 1985) or guiding them toward a bypass (Haymes et al. 1984). Lights have been used to direct fish movements in open water (Wickham 1973) and to attract commercial species including anadromous Alosa spp. (Loesch et al. 1982). Avoidance responses of fish to strobe lights with or without bubble curtains have been used to reduce entrainment at water intakes (Patrick et al. 1985). High-frequency sound has been used to repel anadromous clupeids (Alosa spp.) including alewives (Dunning et al. 1992; Ross et al. 1993), blueback herring (Nestler et al. 1992), and American shad (Kynard and O'Leary 1993; Ploskey et al. 1994). A wide variety of screen types has been used to guide or to protect fish at intakes (Richards 1977).

Dam Description and Fishery Characterization

Richard B. Russell (RBR) Dam and Lake, begun in 1976 and reaching full pool in 1985, is the most recent Savannah River impoundment (Figure 1). RBR dam has four pumped-storage (reversible) turbines and four conventional turbines for a total conventional generation capacity of 640 MW using all eight units and a pumpback capacity of 340 MW using the four pumped-storage units. Capacity generation discharge for all units is approximately 1700 m^3 sec^{-1}, and capacity pumpback is approximately 850 m^3 sec^{-1}. Each draft tube of the four pump turbines is separated into two equal-sized bays by a vertical splitter wall. Each draft tube bay opening is protected by a 9-m wide by 6-m high trash rack. Extensive background fishery data were collected from 1986 to the present using a variety of collection methods to characterize the fish community of the downstream reservoir (Nestler and Ploskey 1990). Some one-unit operation entrainment data were available to optimize the fish protection system prior to the beginning of systematic entrainment data collected during two-unit operation.

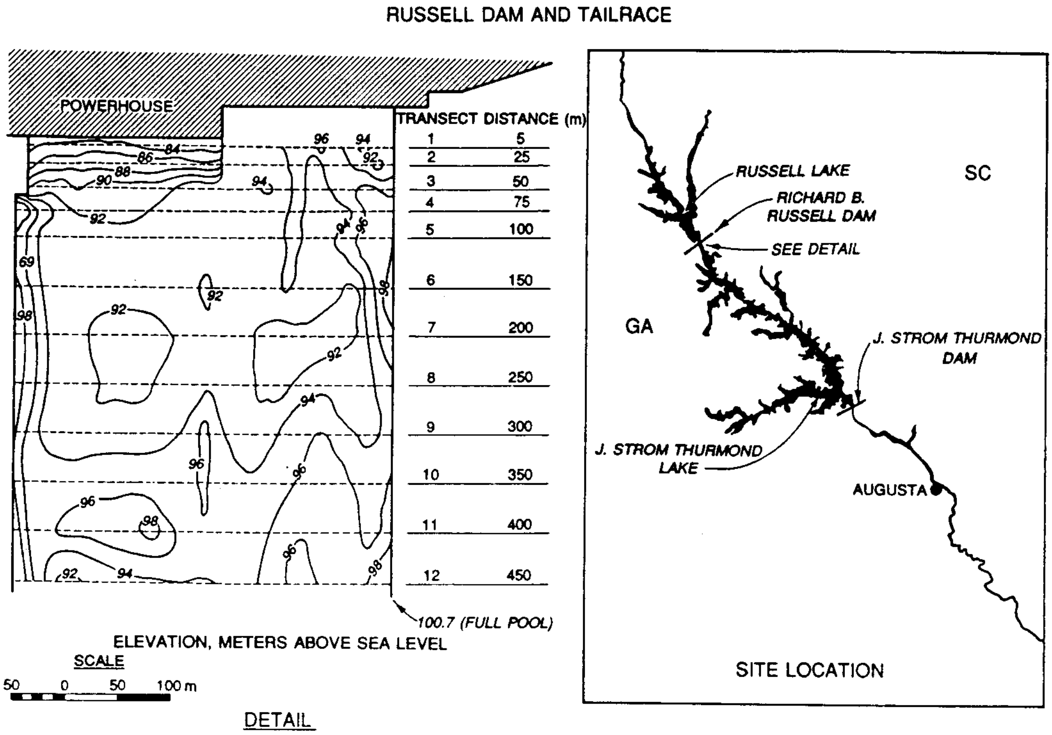


Figure 1. Location map (right) of Richard B. Russell Dam and plan view (left) showing transect locations for mobile hydroacoustic surveys and depth contours.

Assessing Impact Potential with the "Four-Zones" Concept

A "Four Zones" assessment approach was used to predict the numbers and species of fish entrained during pumped-storage operation. Each of the four zones is of different spatial scale, located at varying distances from the dam, and characterized by different hydraulic characteristics (Table 1). Species and size classes of fish most likely to be impacted by pumped-storage operation can be identified by evaluating diel and seasonal species abundance data from each zone. With this knowledge, we blended together muliple fish protection technologies into an integrated fish protection system.

Draft Tube Zone

Certain fishes are attracted to cover provided by the draft tubes and interior physical structure of the dam. These fishes will be entrained into the units during start-up and initial operation. Suckers, catfishes, bullheads, and sturgeon located in this zone are particularly susceptible to entrainment. The potential for entrainment for fish in this zone was estimated using fish recoveries from monthly draft tube dewaterings. The numbers of fishes recovered during draft tube dewaterings were small and usually dominated by small catfishes. Candidate fish protection technologies must be able to operate within confines of draft tubes (very unlikely) or used to prevent entry of fishes into draft tube during non-generation periods.

Near Trash Racks

Some fishes are attracted to shadows and external structure of the dam. These species may be particularly susceptible to

Table 1. Evaluation of entrainment potential by species life stage and zone.

ZONE	TIME PERIOD[1]	ABUNDANT SPECIES[2]	SIZE & HARDINESS	SPORT/COMM VALUE[3]	HABITAT UNIQUE[4]	PROTECT. SYSTEM SCALE[5]
1	DIEL	NONE	NA	NA	NA	NONE
1	SEASON	NONE	NA	NA	NA	NONE
2	DAY	BLUEBACK HERRING	SMALL & FRAGILE	BAIT FISH & FORAGE	NONE	NONE
2	NIGHT	NONE	NA	NA	NONE	NONE
2	SUMMER/ FALL	BLUEBACK HERRING	SMALL & FRAGILE	BAIT FISH & FORAGE	RESTRICTED	NEAR DAM
3	DAY	BLUEBACK HERRING THREADFIN SHAD	SMALL & FRAGILE	BAIT FISH & FORAGE	NA NA	LARGE & FAR FIELD
		STRIPED BASS HYBRID BASS	LARGE & HARDY	RECREAT. FISHERY	NA NA	NEAR DAM NEAR DAM
3	NIGHT	BLUEBACK HERRING THREADFIN SHAD	SMALL & FRAGILE	BAIT FISH & FORAGE	NA NA	LARGE & FAR FIELD
		STRIPED BASS HYBRID BASS	LARGE & HARDY	RECREAT. FISHERY	NA NA	NEAR DAM NEAR DAM
3	SPRING	STRIPED BASS HYBRID BASS	LARGE & HARDY	RECREAT. FISHERY	NONE NONE	NEAR DAM NEAR DAM
3	SUMMER FALL	BLUEBACK HERRING THREADFIN SHAD	SMALL & FRAGILE	BAIT FISH & FORAGE	RESTRICTED NONE	LARGE & FAR FIELD
		STRIPED BASS HYBRID BASS	LARGE & HARDY	RECREAT. FISHERY	RESTRICTED RESTRICTED	NEAR DAM NEAR DAM
4	DIEL	NA	NA	NA	NA	NA
4	SUMMER	BLUEBACK HERRING	SMALL & FRAGILE	BAIT FISH & FORAGE	RESTRICTED	LARGE & FAR FIELD
		STRIPED BASS HYBRID BASS	LARGE & HARDY	RECREAT. FISHERY	RECREAT. FISHERY	NEAR DAM
4	NON-SUMMER	BACKGROUND ABUNDANCES	FULL RANGE	ALL	RESTRICTED	ALL

[1] Time of day or season

[2] Most abundant species by biomass or number

[3] Value: RECREATional FISHERY, commercial BAIT FISH, FORAGE (food) for other fish, no threatened/endangered species present

[4] Uniqueness of habitat - None - habitat not unique, same habitat or water quality conditions occur elsewhere in lake, Restricted - habitat occurs only in this zone

[5] Spatial scale required for fish protection system: NEAR DAM - within several meters of the trash racks, FAR FIELD - must influence fish distribution in Zone 3 or 4, NONE - no system required.

entrainment if they concentrate near the depth of the intakes. Yellow perch, bluegill, and crappie are commonly found near dams and can potentially be entrained into the units. Fishery data to assess the potential for entrainment in this zone were collected using gill nets set monthly immediately in front of the trash racks and mobile hydroacoustics surveys that included transects immediately in front of the trash racks (Figure 1). Gill net catches and hydroacoustic surveys indicated that few fish occurred at trash rack depths but that fish were abundant at the surface 1/3 of the water column. Physical barriers may protect larger fish if they are strong swimmers and have "hydraulic refuges" near the barrier. Small, fragile fishes must be repelled before they reach this zone using behavioral technology.

Zone of Immediate Hydraulic Influence

At start-up the hydraulic environment in this zone is characterized by highly turbulent, localized flow patterns that persist until the long-term flow patterns associated with pumping operation establish (usually about 15 minutes). Consequently, there are no well-defined flow patterns on which fish can orient. This zone should be carefully evaluated for species that are migratory or exhibit pronounced in-reservoir movement patterns. Blueback herring, alewives, striped bass, white bass, hybrid bass, sturgeon, and paddlefish can be found in large numbers near dams particularly at certain seasons or times of day. Entrainment potential from this zone was estimated using monthly mobile hydroacoustic surveys, monthly gill nets set in the tailrace, monthly electrofishing of the shoreline of the tailrace, seasonal purse seining, and ichthyoplankton surveys. Results from these gears indicated that fish abundance varied seasonally, with high abundances of striped bass and hybrid bass occurring in the spring and summer (although they were present year-round) and high concentrations of blueback herring occurring during the reservoir stratification period, particularly when the releases from the dam were the only source of well-oxygenated, cool water in JST Lake. Fish protection strategies for this zone center on a redistribution of small, fragile resident fishes prior to startup of pumping. Larger fishes can be protected with zone 2 physical barriers.

Extended Zone

The last zone encompasses the maximum volume of water pumped back in one pumpback event uninterrupted by generation. Weak swimming pelagic species such as threadfin shad, juvenile gizzard shad, blueback herring, and alewives are particularly susceptible to entrainment from this zone. Entrainment potential was evaluated using monthly mobile hydroacoustics surveys, monthly gill net sets, and monthly electroshocking surveys. The results indicated that fish concentrations in this zone were generally consistent with fish species composition and abundance in the rest of the lake.

Fish Protection System Design

Findings suggested that fish in zones 2 and 3 would be most likely subjected to entrainment. Blueback herring, the most abundant fish in zones 2 and 3, were considered the fish species most likely to be impacted by pumped-storage operation because this genus of fish, which includes alewives and American shad, are commonly entrained at power plant intakes. The distribution of adults of this species appears to be concentrated in zones 2 to 4 during late summer and early fall so that the entire population of blueback herring could be jeopardized by pumped-storage operation. Striped bass and hybrid bass also concentrate in zone 3 in the spring and may be restricted to zones 3 and 4 in the late summer and early fall when they are attracted to the releases of RBR Dam. We concluded that fish in Zones 2 and 3 exhibited the greatest need for protection followed by fish in zone 4. Zone 1 did not appear to require fish protection.

A combination of operational restrictions and behavioral modification technology was used to protect blueback herring in zones 2 and 3. Daytime pumped-storage operation was eliminated to protect blueback herring in zone 2. This restriction had minimal impact on project economics because most pumping operation was conducted at night. Sound to repel blueback herring was not considered feasible because their proximity to the bottom would result in severe echoing preventing the fish from directionalizing on the acoustic signal. Physical barriers for blueback herring protection were not considered feasible because this species is very fragile and would not survive contact with a physical barrier. The nighttime protection strategy for blueback herring in zone 3 was to move them away from the intake plume using underwater sound prior to and during pumping and to use light to attract them to slack or slow water areas. At night blueback herring move up into the water column where a relatively predictable, high energy acoustical field can be generated. The spatial scale over which behavioral modifiers had to work was dictated by the size of zone 3. To adequately protect blueback herring their behavior had to be influenced over distances of 50 to 100 m (the portion of the powerhouse containing the pump units is approximately 100 m wide - a fish in the middle of the intake plume at the powerhouse would have to swim 50 m to reach slower velocity water on either side). The repelling sound had to be effective over a sufficient distance to allow blueback herring enough swimming time to escape the intake plume prior to their entrainment into the trash racks.

Siting and configuring behavioral parts of the fish protection system required characterizing hydraulic patterns in zone 3 (Figure 2) using STREMR, a steady state, 2-dimensional (lateral and longitudinal) numerical hydraulic model to identify boundaries of the intake plume and adjacent eddies or slack/slow water areas (Schneider 1991) and verified with scaled physical hydraulic models. A high-frequency sound field (Nestler et al. 1992), generated using an 8-transducer array, was used to divert blueback herring out of the intake plume. Four transducers covered each draft tube. One transducer was located in the center of each trash rack and aimed parallel to the bottom. A single transducer was located on the side of each trash rack and aimed at an angle of 30 degrees from the face of the powerhouse across the entraining flows. The transducers

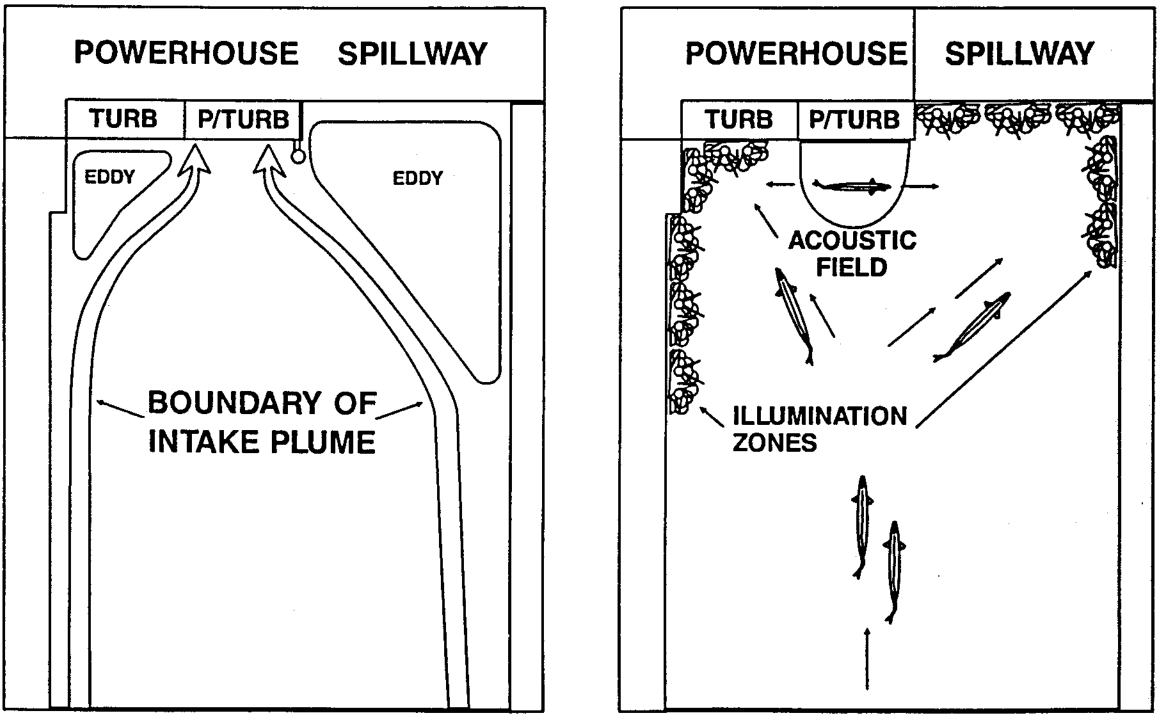


Figure 2. Deployment strategy for components of fish protection system. Left plot shows approximate boundaries of intake plume stream lines and locations of lower velocity zones. Right plot shows approximate locations of acoustic field and attracting lights. TURB=conventional turbines. P/TURB=pump turbines.

generated 118-130 kilohertz (KHz) signal bursts of 5 milliseconds every 50 milliseconds at a sound pressure level of 200-212 dB // 1.0 μPascal at 1.0 m. Decibels (dB) are referenced to 1 micro-Pascal at 1 m here and elsewhere. Transducers were fired sequentially to avoid constructive and destructive interference, and the signal frequency was automatically changed by about 10% every 15 minutes to prevent acclimation of the blueback herring.
High pressure sodium lights (1000 Watts) were used to attract and hold blueback herring to slow water areas located near the spillway (9 lights spaced at 15-30 m intervals long the spillway wall) and near the four conventional units (seven lights spaced about 40 m apart either supported by light poles or attached to the dam (Figure 2)). Preliminary studies of three types of lights and literature information (EPRI 1986) indicated that blueback herring and other fishes would attract to high pressure sodium or mercury vapor lights. Predators such as striped bass and hybrid bass may be indirectly influenced by the light fields as they follow prey fishes such as threadfin shad and blueback herring attracted by lights. Local fishermen routinely fish margins of illuminated areas for striped bass and hybrid bass. A bar screen veneer of 0.32-cm wide wedge wire on 5.08-cm centers (4.8-cm X 15.2-cm openings) was placed directly over the trash racks to divert fishes larger than about 35-cm in length from the trash racks. This spacing allowed smaller fishes that would impinge to pass through the bar screen and into the pump units. The probability of their survival was higher if they passed though the units than impinge on a screen. We felt it reasonable that large, strong-swimming fishes such as striped bass and hybrid bass would be able to avoid impingement at velocities of nearly 2 m sec^{-1} at the bar screen surface because extensive videoimaging has shown that much smaller salmon smolts can escape impingement from bypass screens at velocities over 1 m sec^{-1}. Two strobe lights (Flash Technology - Model AGL) per trash rack aimed into the entraining flow were initially installed on the bar screen veneer to repel fishes.

METHODS

Fish passage was estimated using a 55-m long full recovery net in the forebay that sampled the entire volume of the pumpback jet as it exited the penstock. The distal end of the net was attached to a pontoon boat modified to collect fish. The net was fished during all pump tests, most of which lasted about 6 hours, at least two times per month from November through March and four times per month from April through October. The efficiency of the net for catching fish of different shapes and length classes was determined by introducing replicate samples of marked fish and determining percent recovery. Fixed-aspect hydroacoustics was used to monitor fish passage into all afterbay draft tubes during pumpback, including the draft tube leading to the forebay recovery net. Pumpback net catches were regressed upon acoustic counts from afterbay draft tubes of the same unit to derive an equation for predicting passage through unnetted units from acoustic counts. Equipment included a BioSonics ES-2000 echosounder calibrated with cables and one dual-beam (6 x 15 degree) and one single-elliptical-beam (6 x 12 degree), 420-KHz transducer looking downward from near the surface to the bottom immediately in front of each draft-tube opening. Other equipment included a BioSonics Echo-Signal-Processing (ESP) board, Model 111 Chart Recorder, 486 Gateway computer, and Hitachi Oscilloscope. Receiver gains for fixed-aspect work were set to equalize outputs among transducers, after assuring that the most sensitive transducer did not saturate from the largest echoes. We set a threshold to detect -50 dB targets (fish > about 64 mm long) while avoiding detection of small non-fish targets. The same type of acoustic equipment was used to determine the distribution of fish in the afterbay (tailrace) with mobile surveys of transects. The behavior of fish approaching the bar-screen veneer was recorded during spring pumps using five Simrad Osprey underwater video cameras (Model OE1359) and several video-camera recorders.

Tests of FPS components compared numbers and lengths of entrained fish collected in the forebay net and acoustic counts of tracked fish under several treatments: (1) day versus night; (2) strobe-light on versus off; (3) sound on versus off; and (4) before versus after bar-screen installation. Attracting-light tests compared relative densities of fish in lit and adjacent unlit tailrace areas. In four nights of sound tests and 1 night of strobe-light tests, two 15-minute "on" treatments were

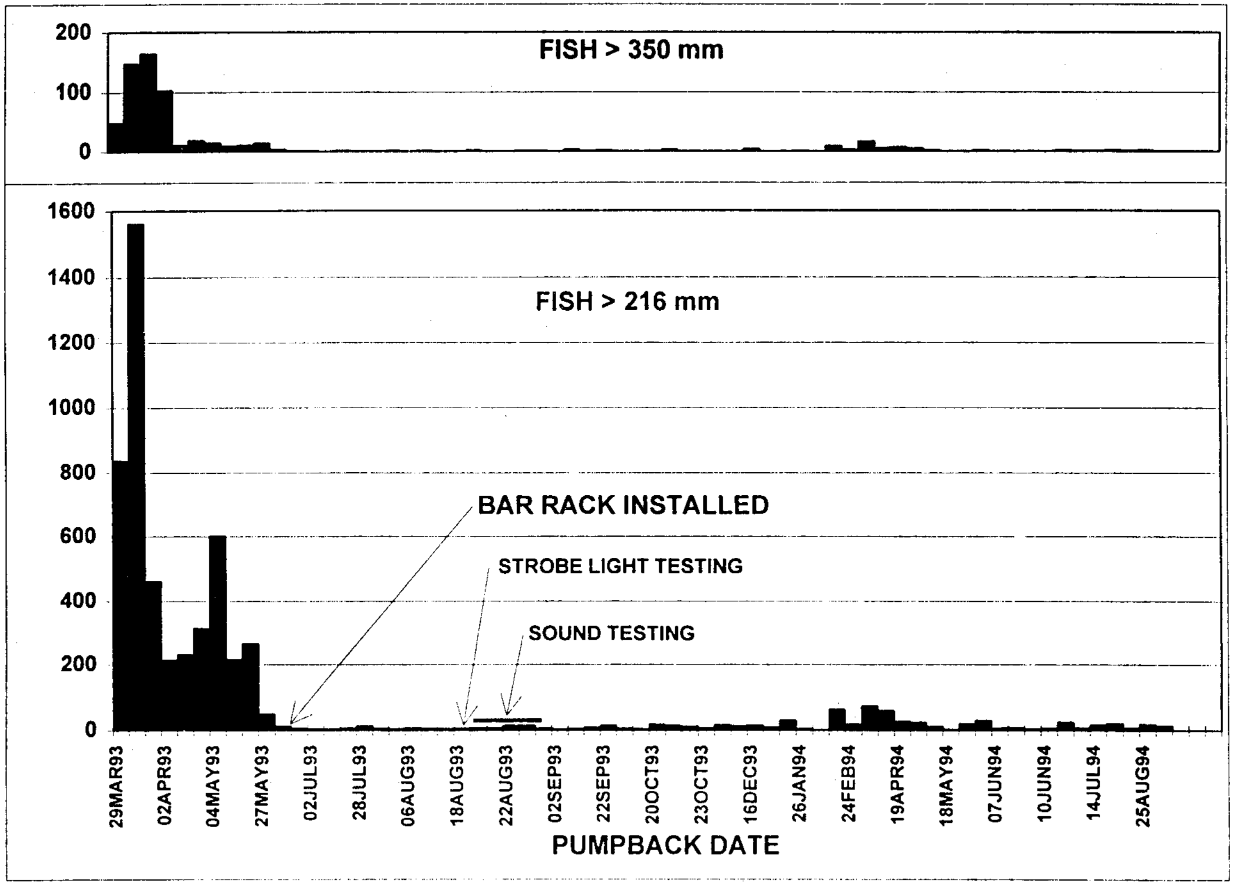


Figure 5. Plots of numbers of large fish entrained in pumpback flows before and after bar-screen installation.

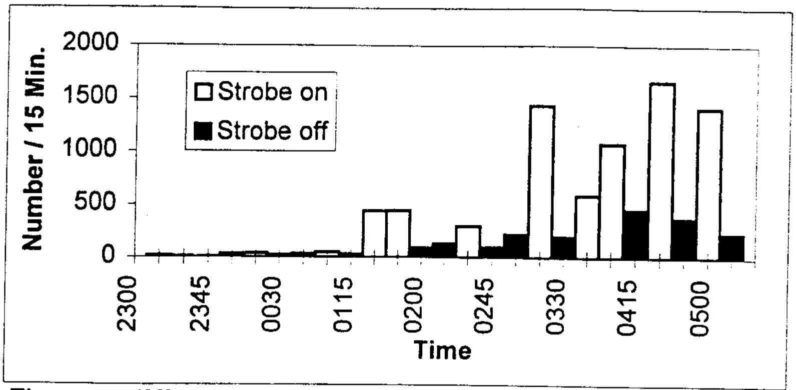


Figure 3. Effect of strobe lights mounted four per unit on fish passage during pumpback at Richard B. Russell Da

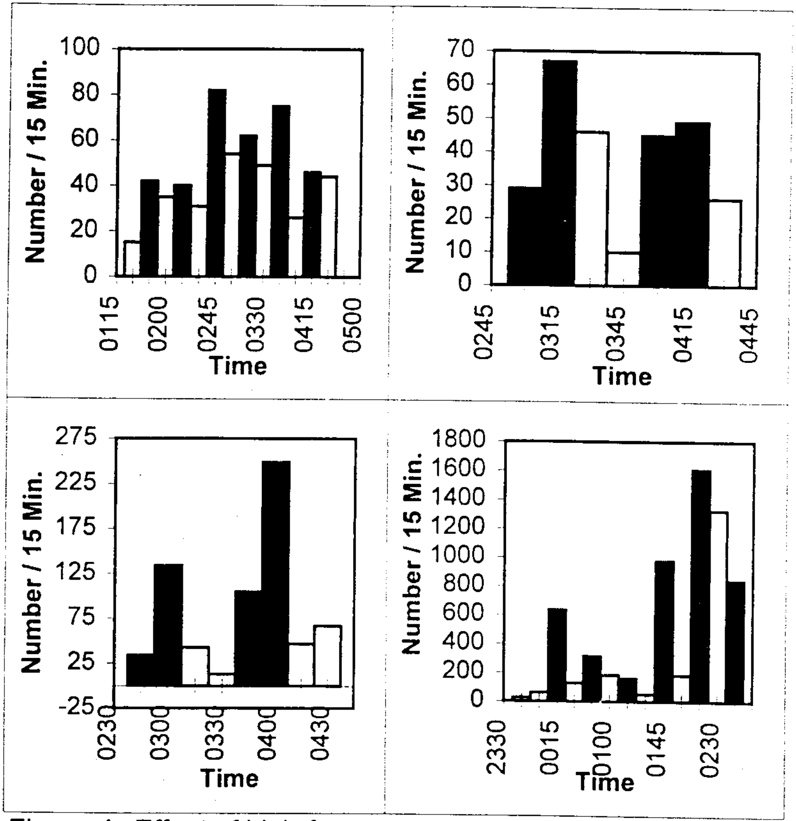


Figure 4. Effect of high-frequency sound on blueback herring passage during four successive nights of pumpback testing. Black = sound off. White = sound on.

randomly selected each hour. Sound or strobe lights, depending upon the type of test, were off during the remaining 15-minute periods each hour. High-frequency sound was not transmitted on the night of strobe-light tests, and strobe lights were off during sound testing, but bar-screens were present and high-pressure sodium lights in the afterbay were lit throughout strobe-light and sound testing. We only performed statistical tests on data collected when background passage rates (those measured during off treatments) exceeded 25 fish / 15 minutes. Deterrence cannot be detected when fish vulnerable to entrainment are scarce or absent. Tests of bar-screens compared hourly entrainment and length-frequencies of fish collected in night pumps (7 before installation; 9 and 45 after). Tailrace fish were sampled monthly with gill nets and mobile hydroacoustics to help account for entrainment differences resulting from changes in tailrace fish populations. Gill nets were 76.2 m long and 2.44 m deep, with ten 7.62-m panels of 9.8, 12.7, 19.1, 25.4, 38.1, 50.8, 63.5, 76.2, 88.9, and 101.6-mm bar mesh.

RESULTS

Mean daytime entrainment rates (3,622 fish/hour) were significantly higher than nighttime rates (294 fish/hour). Blueback herring made up over 98% of all fish passed during day pumps but only 5% during night pumps. Fish passage was significantly higher ($P=0.0004$; $N=8$) during 15-minute treatments when strobe lights were flashing than when they were off (Figure 3). About 96% of the 43,678 fish entrained during one night of strobe-light testing were blueback herring. The four-fold increase in fish passage during treatments when strobe lights were flashing also was detected by a t-test on fixed-aspect hydroacoustic counts. Mean hourly rates of blueback herring passage during sound-on treatments were significantly lower (56%) than rates when sound was off ($P=0.0083$; $N=20$; Figure 4). About 75% of the 92,000 fish entrained in four nights of sound testing were blueback herring, whereas 24% were threadfin shad (Dorosoma petenense). Acoustic counts indicated that treatment differences were only significant at $P=0.2$.

The bar screen significantly reduced fish passage, particularly of fish >216 mm long (Figure 5). It reduced mean entrainment of striped bass and white-bass x striped bass hybrids from 31.1/hour to 2.8/hour (91%). However, mean relative densities of these fish in the tailrace also decreased 64% during the same period. Nevertheless, variance of density estimates was high relative to consistently low entrainment for 12 months after installation. After bar-screen installation, entrainment decreased 47% for all fish >254 mm and 75% for all fish >165 mm. Only fish >350 mm should have been mechanically excluded, but fish >200 mm were observed (by remote video) avoiding passage after approaching the bar-screen. Most entrained blueback herring were 150-254 mm long. Passage in all pumps after September 22, 1993, was dominated (85%) by threadfin shad 38-100 mm long. The remainder consisted of small (<152 mm) yellow perch (5%), other species (3%), bluegills (1%), and spottail shiners (1%). Blueback herring made up only 5% of the total number. Before the bar screen was installed, entrained fish consisted mostly of blueback herring followed by yellow perch, striped bass, and white-bass x striped bass hybrids. Fixed-aspect acoustics showed that mean density of fish under a single attracting light ($7.03 / m^2$) was significantly higher ($P<0.0001$) than mean density in an unlit area of comparable depth (20 m) 24 m away ($0.10 / m^2$). Mean densities of fish along partially lit transects sampled by mobile hydroacoustics were 3.2 times higher than those sampled along unlit transects ($P = 0.01$).

DISCUSSION

Numbers of blueback herring entrained were 92% lower at night than they were during the day due to diel changes in vertical distributions. Dense schools of blueback herring layered near the bottom and close to afterbay draft tubes during the day were particularly vulnerable to entrainment. After differences in day and night rates were quantified. the U.S. Army Engineer District, Savannah, declared a moratorium on all day pumping.

Strobe lights were only tested one night, but statistical tests and observations of acoustic echograms left little doubt that blueback herring were attracted from near the surface to strobe lights mounted on afterbay trash racks at a depth of about 16.8 m. Our findings differ from findings of other researchers such as Patrick et al. (1985) who evaluated strobe lights and found avoidance by most species. We know that blueback herring at RBR Dam are strongly attracted to constant light. Perhaps blueback herring in RBR Tailrace respond positively to light, flashing or not. Strobe lights were removed from the FPS based upon strong evidence of attraction of blueback herring, which made up over 98% of total passage during testing.

High-frequency sound was retained in the FPS after test nettings indicated that it reduced mean passage rates of blueback herring by 56%. This reduction was not as pronounced as that observed by Ross et al. (1993), who observed an 87% reduction in alewife impingement at a nuclear power plant with approach velocities of 0.43 m/sec. Approach velocities very near afterbay trash racks at RBR were 1.52-2.1 m/sec. It is possible that greater transducer coverage could increase deterrence at RBR Dam, but very low rates of blueback-herring entrainment after the last sound test and removal of strobe lights on August 22, 1994, make irrelevant any additional effort. Fixed-aspect acoustics did not strongly corroborate the deterrence indicated by net catches, probably because 25% of the fish passed during sound testing were threadfin shad. Acoustic estimates of blueback-herring passage were obtained by multiplying acoustic counts of tracked fish / 15 minutes by the blueback herring fraction of all fish netted 5 to 20 minutes later in the forebay. Threadfin shad present a problem because most were 25-100 mm long and could not be consistently detected with acoustics using a threshold of -50 dB. High variation in acoustic estimates of blueback herring passage among treatments likely results from poor matching of acoustic counts with species-composition data from netting. There was significant variation in both acoustic detectability of threadfin shad and in the time between the entry of fish in the afterbay draft tube and the time of capture in the forebay net (it averaged 5 minutes).

Only fish >350 mm would be mechanically excluded by the bar screen veneer, so reduced entrainment of fish 165-350-mm fish could not be explained by the screen alone. Observations of 200-350-mm fish approaching and avoiding the bar screen suggest that more than just mechanical exclusion was involved, although other possible factors such as screen vibration or visibility of the stainless steel screen were not evaluated. Whatever the reason, the apparent 75% reduction in passage of fish >165 mm long and 92% reduction in striped bass and hybrid bass passage were ample justification to retain the bar-screen veneer. Head loss due to the bar-screen was about 0.305 m relative to the average 42.7 m head at RBR Dam.

The fixed-aspect acoustic evaluation of differences in densities of fish in lit and unlit areas was much more robust than mobile acoustic evaluations. Fixed-aspect acoustics showed 70 times more fish in lit than in unlit areas, whereas mobile acoustics suggested only a 3-fold difference. Observations of schools of fish under lights scattering as the hydroacoustic boat approached provide a likely explanation for observed differences.

The 81% reduction in hourly entrainment of blueback herring suggested by statistical models derived in a few nights of strobe-light and sound testing was close to the observed 78% reduction in 18 pre-test pumps versus 30 post-test pumps. Most blueback herring were 150-254 mm long, so the bar-screen probably also made a contribution to reducing their entrainment rates.

ACKNOWLEDGEMENTS

The tests described and the resulting data presented herein, unless otherwise noted, were obtained from research conducted for the U.S. Army Engineer District, Savannah, of the U.S. Army Corps of Engineers. Permission was granted by the Chief of Engineers to publish this information. We gratefully thank the U.S. Army Engineer District, Savannah, for coordination and operations that made testing possible, and the National Biological Survey, Cooperative Fish and Wildlife Research Unit, University of Georgia, Athens, Georgia, for all fish sampling during pumpback.

REFERENCES

ASCE. 1995. Guidelines for the Design of Intakes for Hydroelectric Plants. prepared by ASCE Energy Division, Hydropower Intakes Task Committee, Published by ASCE, New York.

Dunning, D. J., Q. E. Ross, P. Geoghegan, J. J. Reichle, J. K. Menezes, and J. K. Watson. 1992. Alewives avoid high-frequency sound. N. Am. J. Fish. Manage. 12: 407-416.

EPRI. 1986. Assessment of Downstream Migrant Fish Protection Technologies for Hydroelectric Application. EPRI AP-4711, Electrical Power Research Institute, Palo Alto, CA.

Haymes, G. T., P. H. Patrick, and L. J. Onisto. 1984. Attraction of fish to mercury vapour light and its application in a generating station forebay. Int. Revue ges. Hydrobiol. 69: 867-876.

Hocutt, C. H. 1980. Behavioral barriers and guidance systems. Pages 183-205 in C. H. Hocutt, J. R. Staufer, J. Edinger, L. Hall, and R. Morgan, eds. Power Plants: effects on fish and shellfish behavior. Acad. Press, New York.

Kynard, B. and J. O'Leary. 1993. Evaluation of a bypass system for spent American shad at Holyoke Dam, Massachusetts. N. Am. J. Fish. Manage. 13: 782-789. Loesch, J. G., W. H. Kriete, Jr., and E. J. Foell. 1982. Effects of light intensity on the catchability of juvenile anadromous Alosa species. Trans. Am. Fish. Soc. 111: 41-44.

Nestler, J. M. and R. Davidson. 1995. Imaging Smolt Behavior on Bypass Screens and a Vertical Barrier Screen at McNary Dam in 1992. Technical Report EL-95-00 (in press), US Army Engineer Waterways Experiment Station, Vicksbury, MS.

Nestler, J. M. and G. R. Ploskey, eds. 1990. Workshop Proceedings: Results of the Richard B. Russell Fish Entrainment Study. Miscellaneous Paper EL-90-3, US Army Engineer Waterways Experiment Station, Vicksbury, MS.

Nestler, J. M. G. R. Ploskey, J. Pickens, J. Menezes, and C. Schilt. 1992. Responses of blueback herring to high-frequency sound with implications for reducing entrainment at hydropower dams. North American Journal of Fisheries Management 12:667-683.

Patrick, P. H., A. E. Christie, D. Sager, C. Hocutt, and J. Stauffer. 1985. Responses of fish to a strobe light/air bubble barrier. Fish Res. 3:157-172.

Ploskey, G. R., G. N. Weeks, and J. L. Pickens. 1994. Behavioral responses of anadromous blueback herring and American shad to high-frequency sound, flow, and a wing wall in St. Stephen Tailrace, South Carolina. Tech. Rep. EL-94-7, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS, USA.

Richards, R. T. 1977. Fish screening - the state of the practical art. Pages 296-303 in Proceedings 25th Annu. Hydraulics Div. Specialty Conference, Texas A&M University, Am. Soc. Civil. Eng., New York, NY, USA.

Ross, Q. E., D. J. Dunning, R. Thorne, J. K. Menezes, G. W. Tiller, and J. K. Watson. 1993. Response of alewives to high-frequency sound at a power plant intake on Lake Ontario. N. Am. J. Fish. Manage. 13: 291-303.

Schneider, M. L. 1991. Clarks Hill Fish Entrainment Study: Hydraulic Analysis. in J. M. Nestler, ed., Preliminary Results of the Richard B. Russell Fish Entrainment Study: Proceedings of a Workshop. Miscellaneous Paper EL-91-15, US Army Engineer Waterways Experiment Station, Vicksburg, MS.

Wickham, D. A. 1973. Attracting and controlling coastal pelagic fish with nightlights. Trans. Am. Fish. Soc. 4:816-825.

HYDRO-ELECTRIC DEVELOPMENT IN IRELAND: PROBLEMS FOR ATLANTIC SALMON (SALMO SALAR) AND SOME SOLUTIONS.

M.O'Farrell,
Zoology Department, University College, Galway, Ireland
E-mail: mofarrel@iol.ie

M. Murphy, W.O'Connor, L.O'Sullivan, D.Quigley,
Fisheries Conservation Unit, Electricity Supply Board, Ardnacrusha, Co. Clare, Ireland

J. Browne, N.O'Maoileidigh,
The Marine Institute, Fisheries Research Centre, Department of the Marine,
Abbotstown, Dublin 15, Ireland.

ABSTRACT

Four major Irish catchments have been harnessed for the production of hydro-electricity, namely, the rivers Shannon, Liffey, Erne and Lee. The design of each scheme is described in terms of percentage of catchment area affected, impoundment of water upstream of generating stations, number of generating stations, type and number of turbines, nature of spillway gates and fish-passes, discharge regime, generating protocols and statutory minimum flows. These hydro-electric schemes were constructed between 1924 and 1957. In most cases provision for the upstream passage of salmon was made while no consideration was given to the downstream passage of smolt. The nature and timing of the impact on salmon differed for each river. In the case of the River Shannon the impact was gradual. The large multi-sea-winter component of the Shannon stock was lost while the abundance of fish migrating upstream past the generating station and regulating weir fell to unacceptable levels. In the case of the River Lee the impact was immediate and was caused by deoxygenated water discharging to the lower river from filling reservoirs upstream of both generating stations. On the River Erne the impact was also gradual with the numbers of salmon ascending the river falling to very low levels some 20 years after construction. Mitigation in the form of annual juvenile salmon releases on the Shannon, Lee and Erne have not proved to be lasting solutions to declining salmon runs. These releases have in some cases masked the decline of wild salmon. The River Liffey scheme comprises three generating stations, only one of which lies in the path of salmon and it's impact appears to have been negligible. At this generating station the locations of the Kaplan turbine and Borland - MacDonald fish-lock intakes, coupled with the operation of a spillway gate trash flap, have facilitated successful downstream passage of smolt. Solutions on all rivers are in the form of modified generating protocols which, within the confines of the original hydro-electric scheme designs, take into account the requirements of migrating smolt. The solutions arrived at for the various stations which facilitate downstream passage of salmon smolt have also involved consideration of the upstream migration requirements of juvenile European eel. In the case of the Shannon and Erne, generating protocols during the months of April and May which facilitate the migrations of both species have been implemented.

KEY-WORDS: Ireland / Hydro-Electric Generating Stations / Atlantic Salmon / Shannon / Liffey / Lee / Erne / European Eel / Turbine Passage / Generating Protocol / Borland - MacDonald Fish-Lock

INTRODUCTION

This paper collates all published work carried out on Irish rivers harnessed for the generation of hydro-electricity and sets them in the context of current understanding of the impact of hydro-electric schemes on fisheries. During the first half of the twentieth century Ireland's economic growth depended on hydro-electric development and it was national policy that fisheries interests should not be allowed to interfere with economic prosperity. In the mid-1930's one hydro-electric generating station (Ardnacrusha on the River Shannon) produced 87% of the electricity requirements of what was then the Irish Free State. Today Ardnacrusha accounts for 3% of national output while hydro-electric generating stations produce only 6% of the total.

The present paper reports on fisheries work carried out on the major Irish hydro-electric schemes on the rivers Shannon, Liffey, Erne and Lee. Most fisheries work carried out before and after the construction of these hydro-electric schemes related to the characteristics of adult salmon returning to the rivers concerned (Southern, 1928; Newman, 1959; Twomey, 1957,1959,1964; Went, 1938,1940,1942,1943,1946,1950,1953,1964,1970). Some work was published on the effects of turbine passage on smolt in the River Lee (Twomey, 1965a) and on the impact of predation on smolt delayed in reservoirs upstream of generating stations on the River Lee (Twomey, 1965b). Pyefinch (1966) carried out a review of the impact of hydro-electric developments in Scotland on fisheries. The present paper, the first overview for Ireland, describes the hydro-electric schemes, their impact on salmon and some solutions which form integral parts of current fisheries management programmes on these rivers.

HYDRO-ELECTRIC SCHEMES IN IRELAND

Description Of Hydro-Electric Schemes

The island of Ireland covers 82880 km^2. A total of four major rivers, namely, the Shannon, Liffey, Erne and Lee have been harnessed for the generation of hydro-electricity (figure 1). The combined catchment areas total 18696 km^2 while the combined catchment areas upstream of generating stations total 16382 km^2 (22.6% and 19.8% of the island of Ireland, respectively). Table 1 describes the characteristics of harnessed rivers in terms of catchment area upstream of generating stations, influence of tides at generating stations, average annual flow and year of commissioning.

On the River Shannon a hydro-electric scheme was constructed between 1925 and 1929 which involved harnessing 10,400 km^2 of the catchment area upstream of a regulating weir at Parteen which diverted water into the River Shannon proper (statutory minimum discharge of 10 m^3 sec^{-1}) and into a 12.6 km headrace canal with a capacity of 400 cumecs supplying a 30 m head, 85MW generating station at Ardnacrusha. Water is then returned to the River Shannon via a 2.4 km tailrace close to the river's tidal limits. When the River Shannon hydro-electric scheme was constructed no provision was made for the downstream passage of Atlantic salmon smolt at either the regulating weir or the generating station while provision for the upstream passage of adult salmon was made at the regulating weir in the form of a conventional pool and traverse fish pass. In 1959 an unique 34m vertical Borland - MacDonald fish-lock was constructed at Ardnacrusha to facilitate upstream passage of adult salmon.
On the River Liffey three hydro-electric generating stations were constructed during the 1940's. The two upstream stations, Pollaphouca and Golden Falls are situated above and at the upper limit of salmon distribution in the catchment, respectively. Only Leixlip lies in the path of salmon and it is the only Liffey generating station that will be considered in this paper. The construction of Leixlip resulted in the creation of a 2 km^2 reservoir immediately upstream. On the River Erne two generating stations were constructed during the late 1940's. The hydro-electric scheme involved major drainage works in the mid-catchment area. Assaroe Lake (2.3 km^2) was created between the hydro-electric generating stations. Fish passes of the White submerged orifice type were constructed at Cathaleen's Fall and Cliff to facilitate the upstream migration of adult salmon. A tidal tailrace about 1.5 km long discharges to the Erne estuary. The construction of the River Lee hydro-electric scheme

commenced in 1953 and both Carrigadrohid and Inniscarra generating stations were commissioned in 1957. The scheme involved the creation of two reservoirs (5 and 9.3 km^2) upstream of both generating stations. The tidal limit of the River Lee is about 14.5 km downstream of Inniscarra.

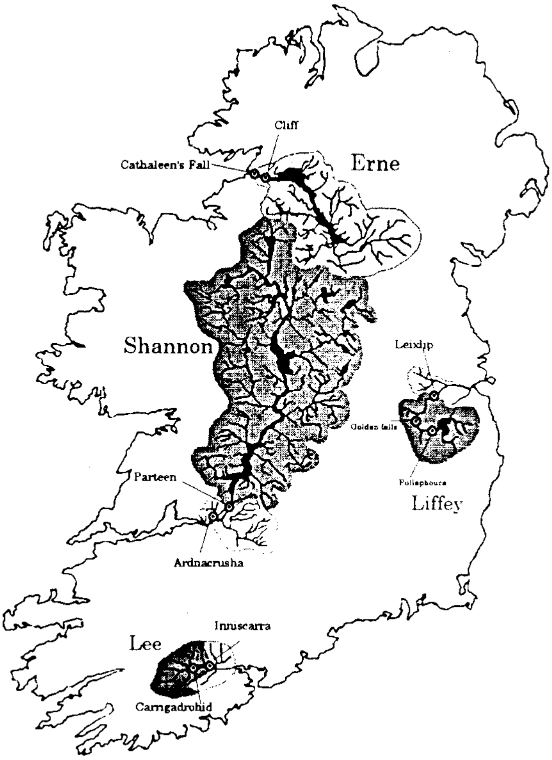


Figure 1: Map of Ireland showing river catchments and generating stations.

Table 1 describes the catchment and discharge characteristics of each hydro-electric scheme on the rivers Shannon, Liffey, Erne and Lee. Almost the entire River Erne has been harnessed while significant percentages of the Shannon, Liffey and Lee are located downstream of the hydro-electric generating stations. The characteristics of turbines and the preferred smolt passage routes at each hydro-electric generating station are given in tables 2 and 3, respectively.

Table 1: Characteristics of hydro-electric schemes on the rivers Shannon, Liffey, Erne and Lee.

River	Catchment area (km^2)	Generating Station	Year of Commission	Upstream catchment area (km^2) (%)	Average Annual Flow (m^3 sec^{-1})	Tidal Influence
Shannon	11700	Ardnacrusha	1929	10400 (88.9)	179.4	+
Liffey	1369	Leixlip	1949	843 (61.6)	13.1	-
Erne	4374	Cliff	1950	4320 (98.8)		-
Erne	4374	Cathaleen's Fall	1951	4349 (99.4)	98.1	+
Lee	1253	Carrigadrohid	1957	616 (49.2)	21.4	-
Lee	1253	Inniscarra	1957	790 (63.0)	27.1	-

Table 2: Characteristics of turbines at hydro-electric generating stations on Irish rivers

River	Gen. Stn.	Set	Turbine	rpm	No. Blades	Head (m)	Discharge ($m^3 sec^{-1}$)	MW	Runner D. (m)
Shannon	Ardnacrusha	G1	Francis	150.0	13	28.5	100	20	4.09
	Ardnacrusha	G2	Francis	150.0	13	28.5	100	20	4.09
	Ardnacrusha	G3	Francis	150.0	13	28.5	100	20	4.09
	Ardnacrusha	G4	Kaplan	167.0	4	28.5	100	25	4.10
Liffey	Leixlip	G5	Kaplan	300.0	4	17.5	31	4	2.00
Erne	Cliff	G1	Kaplan	115.3	4	10.0	120	10	4.30
	Cliff	G2	Kaplan	115.3	4	10.0	120	10	4.30
	Cathaleen's Fall	G3	Kaplan	187.5	4	28.5	93.5	22.5	3.85
	Cathaleen's Fall	G4	Kaplan	187.5	4	28.5	93.5	22.5	3.85
Lee	Carrigadrohid	G3	Kaplan	167.0	5	13.6	73.5	8	3.37
	Inniscarra	G1	Kaplan	214.0	6	29.4	62	15	3.06
	Inniscarra	G2	Kaplan	428.0	6	29.4	15	4	1.55

Table 3: Fish passes, spillways and preferred smolt passage route at hydro-electric generating stations on Irish rivers

River	Generating Station	Fish Pass	Spillway Gates (over/undershot)	Preferred smolt passage route
Shannon	Ardnacrusha	Borland - MacDonald	Under	via turbines
Liffey	Leixlip	Borland - MacDonald	Under + Over	via fish-pass & trash flap
Erne	Cliff	White submerged orifice	Under + Over	via turbines & fish pass
Erne	Cathaleen's Fall	White submerged orifice	Under + Over	via turbines & fish pass
Lee	Carrigadrohid	Borland - MacDonald	Under + Over	via turbine & fish pass
Lee	Inniscarra	Borland - MacDonald	Under + Over	via fish pass

Long-term average monthly inflows and temperatures are shown for each river in figures 2 and 3, respectively. During the period of salmon smolt migration (April - May) discharges are falling on each river while water temperatures are rising. Water availability for specific generation protocols and / or operation of fish passes as downstream passes is a key factor during the critical period of smolt migration. During years of low inflow e.g. 1995, low water availability meant that proposed generating protocols etc. could not be implemented with the result that smolt accumulated upstream of generating stations.

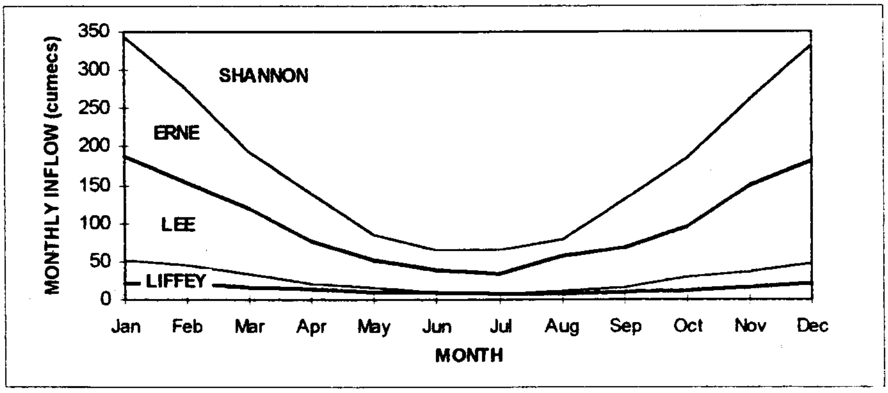


Figure 2: Long-term average monthly inflows (Shannon data for 1930-1969; Erne data for 1900-1941; Lee data for 1957-1991; Liffey data for 1950-1995) for Irish rivers

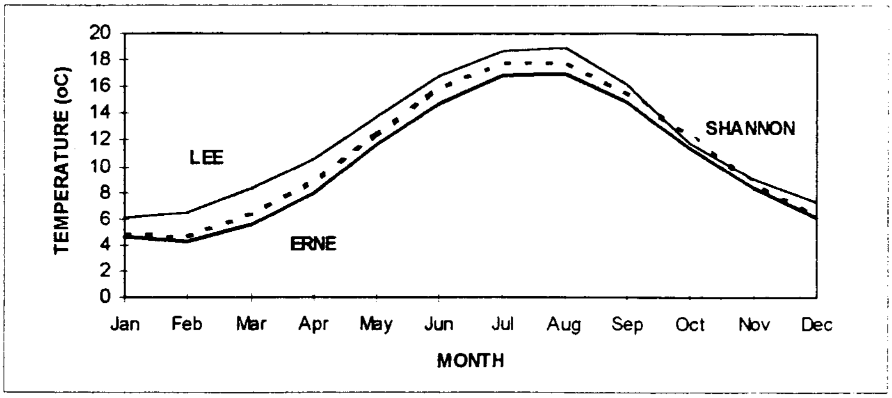


Figure 3: Long-term monthly average temperature (Shannon data measured daily at Parteen Hatchery 1976-1995; Lee data measured daily at Carrigadrohid Hatchery 1991-1995; Erne data measured daily at Ballyshannon Hatchery 1991-1995) for Irish rivers.

IMPACT OF THE HYDRO-ELECTRIC SCHEMES ON SALMON

Case Histories

The River Shannon was renowned as a producer of large multi-sea-winter salmon to commercial and recreational fisheries located on it's lower reaches. Prior to the advent of the scheme salmon spawned in the tributaries of the uppermost lake, Lough Allen and a commercial salmon fishery operated on Lough Key in the upper catchment (Went, 1970). The hydro-electric scheme resulted in the shrinkage of the bypassed River Shannon downstream of Parteen regulating weir with the resultant loss of spawning and nursery habitat and the large multi-sea-winter (MSW) salmon (Went, 1970). The hydro-electric scheme assumed that ascending fish would use the pool and traverse fish pass at Parteen regulating weir. However, the majority of fish appeared to enter the tailrace which is a sanctuary area. This situation continued until 1959 when an unique Borland - MacDonald fish-lock was constructed at Ardnacrusha. While 30 years had elapsed since the commissioning of the station the impact on salmon in the system was never measured. Salmon continued to ascend the Shannon using either the pool and traverse fish pass at Parteen regulating weir or the navigation locks at Ardnacrusha. This was confirmed by Went (1962) who intercepted a total of 5461 salmon smolt, for tagging purposes, in the navigation locks during the years 1953-1955.

In the case of the River Liffey the impact of Leixlip on salmon appears to have been minimal. The fact that significant numbers of adult salmon ascend at Leixlip each year proves that the fishery installations (fish-lock and spillway gates) are working satisfactorily and that significant numbers of smolt are not using the Kaplan turbine (300 rpm) as a downstream passage route.

On the River Lee the impact was immediate and related to impaired water quality discharging to the lower river from flooding reservoirs upstream of both Carrigadrohid and Inniscarra. The stock of salmon in the Lee collapsed within five years of construction. The collapse of salmon stocks upstream of Inniscarra has been attributed to the following factors (Twomey 1991):

deoxygenation of tailrace water due to the presence of large quantities of decomposing vegetation in the reservoirs during the filling of these reservoirs (filling of the reservoirs took place during the latter half of 1957); predation on descending smolt initially by pike (*Esox lucius* L.) and subsequently by brown trout (*Salmo trutta* L) which was associated with the failure of descending smolt to pass downstream at Carrigadrohid and Inniscarra and their subsequent accumulation in the reservoirs.

On the Erne the numbers of adult salmon entering the system remained at a high level during the 1950's and 1960's. Thereafter numbers fell to unacceptably low levels. The reason(s) for the delayed impact on the Erne are not understood at present.

Mitigation, Restoration And Continuous Assessment of Long-Term Impact on Fisheries

Hatchery Operation And Restocking

Large scale salmon hatcheries were constructed on the Shannon in 1959, on the Lee in 1970 and on the Erne in 1983. Small scale hatcheries were also operated on the Lee since 1960 and on the Erne during the years 1959-1966. The output from these hatcheries in terms of eyed ova, unfed fry, parr and smolt released into the rivers Shannon, Lee and Erne is described in table 4 and figure 3.

Table 4: Numbers of salmon (millions) of various life stages released into the rivers Shannon, Lee and Erne

River	Eyed ova	Unfed fry	Parr	Smolt	Duration
Shannon	0.566	11.227	11.349	4.937	35 years
Lee	0.458	4.127	2.292	3.059	36 years
Erne	0.956	1.437	1.655	0.537	13 years

During the years 1961-1990 mitigation measures on the Shannon involved annual large scale releases of parr and smolt. The parr were usually lower mode 'thinnings' averaging 2-4 g while smolt were S1's. Annual releases of juvenile salmon were carried out as a matter of course with little concern for the status of naturally spawned salmon in the tributary streams planted. Mitigation measures on the Lee involved, initially, restocking with large numbers of fingerling salmon of River Shannon origin upstream of Carrigadrohid. During the 1960's serious attempts were made to monitor the number of smolt migrating from the system using smolt traps located in the upper pool of the Borland - MacDonald fish-locks at Inniscarra and Carrigadrohid. When it became clear that restoration of the upstream salmon population was not succeeding it was decided to abandon the catchment area upstream of Inniscarra in terms of natural smolt production and to construct a large hatchery immediately downstream of Carrigadrohid with the objective of producing smolt and lower mode 0+ parr for release downstream of Inniscarra. This mitigation measure was designed to offset lost smolt production in the catchment area upstream of Inniscarra. This situation continued through the 1970's and 1980's and in the 1990's it was decided to attempt to restore salmon in the upper catchment area by initially restocking with unfed fry of Lee and Shannon origin. When the Erne hatchery was constructed in 1983 ova of River Lee origin were used to initiate smolt production. The policy at this hatchery until 1993 was also to produce a predetermined number of smolt while lower mode 0+ parr (thinnings) were planted in selected tributaries of the Erne.

Restocking procedures changed in 1991 with a move to unfed fry planting and retention of limited numbers of unfed fry for smolt production the following year. Baseline electrical fishing surveys were also carried out which identified suitable planting habitat and also the presence / absence of naturally spawned salmon. Genetic screening of the hatchery stock at Parteen Hatchery on the River Shannon showed it to be as genetically variable as other salmon stocks in Irish rivers and therefore suitable for reseeding rivers which did not support naturally spawned salmon (Galvin *et al.*, 1994). In the Shannon, Lee and Erne catchments, rivers planted with unfed fry / parr have been surveyed and the performance of planted fish evaluated. In the case of the Shannon, unfed fry planted in streams in the upper catchment area have been captured as 1+ parr, using electrical fishing equipment, adipose fin-clipped and coded-wire tagged so that their contribution to commercial and recreational fisheries can be assessed as part of Ireland's national coded-wire tagging programme.

Though a hatchery was constructed in the Liffey catchment area in the mid-1980's it was used solely for commercial purposes and for the production of brown trout for stocking into the reservoir upstream of Pollaphouca Generating Station. A small hatchery is located on the lower River Liffey downstream of Leixlip but the number of juvenile salmon released into the river each year is very low.

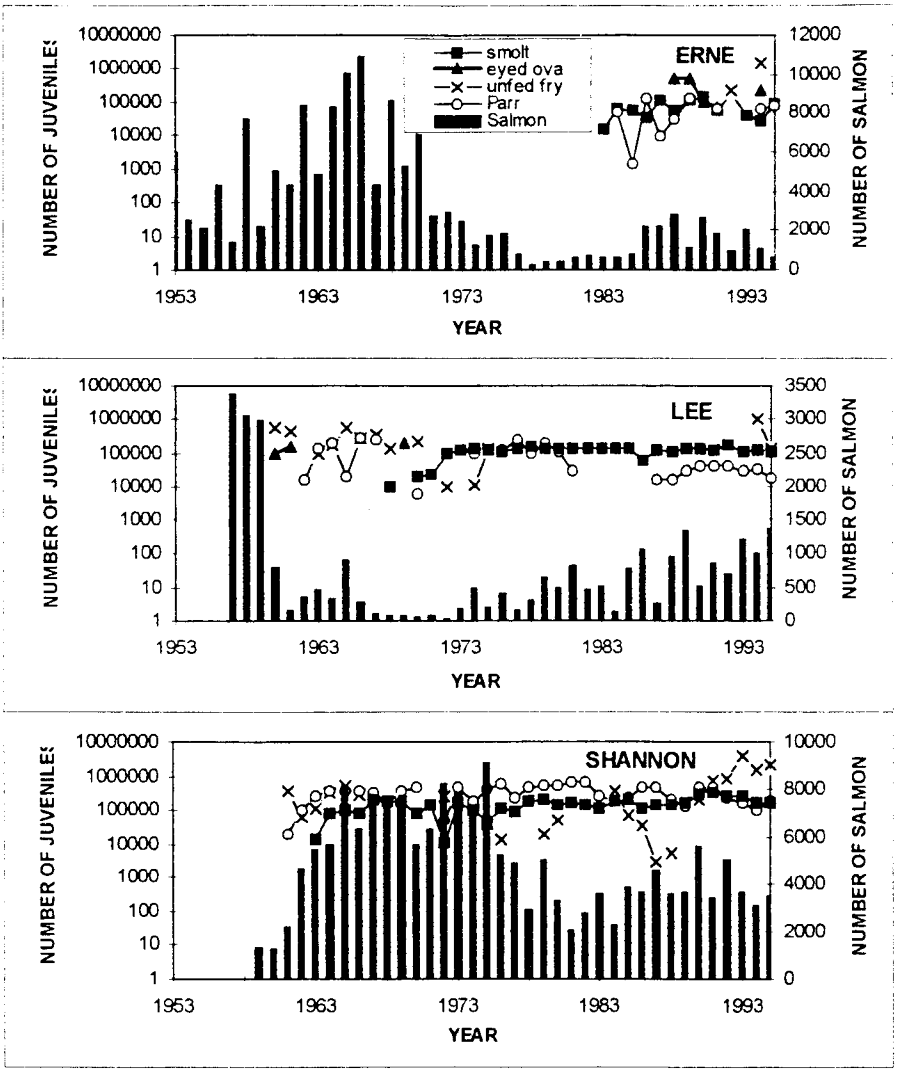


Figure 4: Numbers of salmon eyed ova, unfed fry, parr and smolt released in the Shannon, Lee and Erne catchment areas (log scale) together with adult salmon census data for these rivers.

Generating Protocol And Operation Of Fish Passes For Downstream Passage Of Smolt

The fish passes associated with hydro-electric schemes in Ireland were designed for upstream passage of adult salmonids. In the past electricity generated during daylight hours and particularly during peak demand periods, was about four times as valuable as that generated during the hours of darkness. Consequently hydro-electric generating stations were biased towards peak demand generation - except during the winter months when stations often remained on load over 24 hour periods due to the availability of water. Thus when smolt were migrating on the Shannon their migration was interrupted by the absence of night-time generation. McGrath (1975) stated that smolt were allowed to accumulate in the headrace, and 'when a sufficient number had been assembled', the ship-lock was used to pass them downstream. It is unlikely that this was good practice as delayed smolt were easy prey for piscivorous birds and fish and were also liable to physiological damage associated with this delay. The peripheral velocity of the blades of the Francis turbines at Ardnacrusha (32.1 m sec^{-1}) and the relatively low rpm value for the Kaplan turbine mean that smolt mortality during turbine passage at this station is low. Tests carried out using large groups (c.10.000) of coded-wire tagged reared smolt released by helicopter upstream and downstream of Ardnacrusha during the years 1991-1993 have shown that turbine passage mortality has averaged 8.5% (O'Farrell *et al.*, 1995).

On the Erne, both generating stations are required to remain on load until mid-night each day during the months April and May. All four turbines are relatively benign to smolt during passage (particularly the Cliff turbines). Both generating stations have overflow gates capable of discharging significant quantities of water (up to 4 cumecs) to the upper pools of the White submerged orifice fish passes. However, because of their locations, they also have the potential to cause flooding to the stations and for this reason their use as smolt by-passes has been curtailed. On the Erne and Shannon, the generating protocol during the months of April and May has to take into consideration the requirements of descending smolt and ascending elvers / juvenile eels. An attempted compromise involves continuous generation until mid-night (approximately) each day during this period to facilitate smolt with the stations coming off load between mid-night and 08.00 hrs (approximately) to facilitate elvers / juvenile eels. Figure 5 shows the 1995 generation protocol at Cathaleen's Fall during the months April through June together with the smolt index trap catch at Cliff (trap operated during May only) and the elver catch made using traps located immediately downstream of Cathaleen's Fall.

During the period 1963 - 1969 smolt traps were operated in the Inniscarra (1963-1969) and Carrigadrohid (1969) Borland -MacDonald fish-lock upper pool. Table 5 describes the numbers of smolt intercepted during these years.

Table 5: Smolt census work carried out on the River Lee during the 1960's (Electricity Supply Board of Ireland, unpublished report)

Location	1963	1964	1965	1966	1967	1968	1969
Carrigadrohid							1003
Inniscarra	7534	12212	16936	1192	499	1614	719

This was achieved by cracking open the lower gate and discharging one cumec while at the same time ensuring that the upper pool and reservoir levels were equal. A large basket was used to intercept descending smolt which were released downstream after counting. Both Kaplan turbines at Inniscarra, particularly G2 (Twomey 1965a) have the potential to kill a high percentage of smolt during passage. The intakes for both are located at depth and the preferred generating protocol during the period of smolt migration is to have this station off load during the hours of darkness. At present the Borland-MacDonald fish-locks associated with Carrigadrohid and Inniscarra discharge 2 cumecs throughout April and May in an attempt to pass smolt downstream. The lower gate is opened fully and the level in the upper pool is lower than that in the reservoir. The focus is now on passing smolt downstream with less emphasis on census work. However, it is planned to use a dewatering screen in association with a commercial smolt counter to carry out smolt census work with minimum impact on migrating fish.

At Leixlip on the River Liffey the intake for the Kaplan turbine is located at depth while the intake for the Borland-MacDonald fish-lock, the first of it's kind ever constructed (Aitken *et al.*, 1966) and reported on at the time by O'Meallain (1951) and by Murphy and Dooge (1951), is at the surface and almost directly overhead the turbine intake. Smolt use the Borland - MacDonald fish-lock as a downstream fish pass. The trash flap on the centre spillway gate (1.5 cumecs) in addition to the fish-lock (0.5 cumecs) have been used to discharge the statutory minimum flow when the station is off load and this combined surface discharge has facilitated the downstream passage of kelt and smolt.

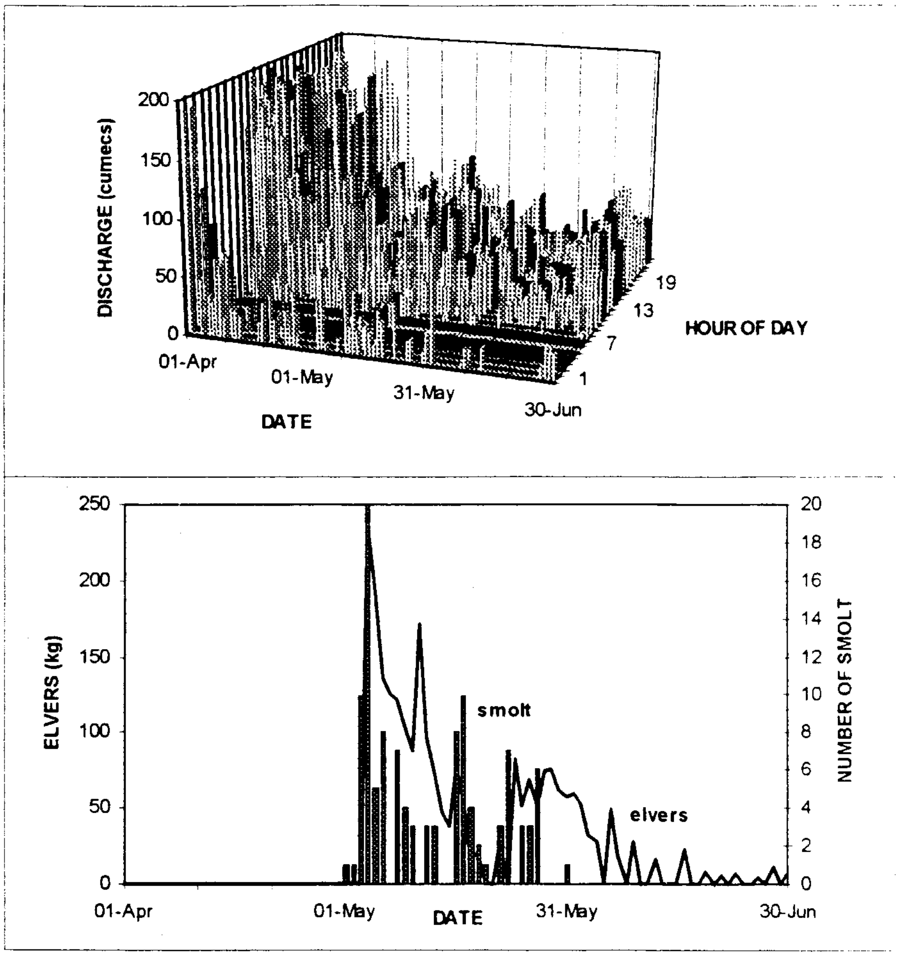


Figure 5: Discharge at Cathaleen's Fall (top panel) and index smolt catch (n=113) and elver catch (2506 kg) (bottom panel) during the period 1 April - 30 June 1995.

Assessment Of Juvenile Salmon Populations

During the years 1990-1992 a 362 site baseline electrical fishing survey was carried out in the River Shannon with the objective of describing the distribution and relative abundance of naturally spawned juvenile salmon throughout the catchment area. Salmon were recorded at 110 sites and with the exception of a single tributary stream salmon were absent from the upper catchment (3300 km^2). Many of the sites where salmon were recorded supported only nominal salmon populations and the statistics support the view that upstream of the generating station the distribution of juvenile salmon in the catchment is patchy and that where they occur they do so, with some exceptions, at low densities (Electricity Supply Board of Ireland, unpublished report).

An electrical fishing survey was carried out in the Lee system during August 1993. Results showed that downstream of Inniscarra juvenile salmon occurred at all sites surveyed and at satisfactory densities. However, upstream of Inniscarra juvenile salmon were present only at sites close to the reservoirs (Electricity Supply Board of Ireland, unpublished report).

In the River Erne catchment several electrical fishing surveys have been carried out in tributaries located in Northern Ireland (Vickers, 1969; Cragg-Hine 1972; Kennedy and Strange 1978). These surveys showed that the distribution and abundance of juvenile salmon throughout the tributaries investigated were unsatisfactory. During the period 1994-1995 approximately 150 sites were electrically fished in tributaries throughout the Erne catchment area. In both years naturally spawned 0+ and 1+ salmon were recorded at only a handful of sites (Patrick Boylan, personal communication).

Assessment Of Adult Salmon Populations

The assessment of the numbers of salmon ascending via fish passes at generating stations and regulating weirs is carried out by trap interception (Parteen, Inniscarra), resistivity counter linked to VCR system (Ardnacrusha, Leixlip) or a combination of resistivity counter and trap (at Cathaleen's Fall an upstream trap is operated from 1 October each year). The results of adult census work on these rivers are shown in figure 3.

In some years census work was not complete due to resistivity counter breakdowns. At Leixlip, the average number of adult salmon counted upstream for the period 1969 - 1995 has been 574 (data for five years omitted due to resistivity counter malfunction). In the past, policy relating to adipose fin-clipping / tagging of reared smolt had been variable with the result that the contribution of reared fish to commercial and recreational catches and hatchery broodstocks has not been measured accurately. This scenario masked the true relationship between the numbers of wild and reared adults returning to each river. In the case of the River Lee all reared smolt were adipose fin-clipped in selected years and it was determined that virtually all adults returning to Inniscarra were of hatchery origin. Since 1991 on the Shannon and 1993 on the Erne and Lee all reared smolt have been adipose fin-clipped. Selected lines have also been coded-wire tagged. This has facilitated the separation of reared and wild adults. The numbers of wild and reared adults ascending at Ardnacrusha and Parteen on the River Shannon during the years 1992-1995 are given in table 5 (Electricity Supply Board of Ireland, unpublished report). On the Lee a total of 1371 adults returned to Inniscarra during 1995 of which 361 (26.3%) were determined to be wild fish.

Table 5: Numbers of wild and reared salmon recorded at Ardnacrusha and Parteen during the years 1992-1995.

Year	Ardnacrusha		Parteen		Total
	Wild	Reared	Wild	Reared	
1992	1639	792	229	2376	5036
1993	595	252	887	1943	3677
1994	773	231	886	1184	3074
1995	1395	729	430	962	3516

The performance of the fish pass at Cathaleen's Fall has been described by Jackson and Howie (1967) who used an electro-mechanical fish counter (the Irish Harp). Census work at this location has been particularly difficult in recent years because of the presence of reared fish without adipose fin-clips and the probable presence of non-natal fish in the fish pass (when the trap is operated after 1 October each year most fish are collected upstream of the trap - indicating their wish to descend the pass).

Fish passes at hydro-electric generating stations and regulating weirs have worked satisfactorily in terms of upstream passage of adult salmon. However, early run MSW fish have experienced delays at these passes but the consequences of such delays for MSW Atlantic salmon on these relatively small rivers are not understood at present.

DISCUSSION

Traditional mitigation measures (hatchery operation) have been expensive in the long term and do not appear to have been successful. Since 1991 large numbers of unfed fry have been planted in the Shannon, Erne and Lee catchments in an attempt to boost smolt production. In the case of the Shannon, naturally spawned 0+ salmon were recorded in the upper catchment area (Boyle and Feorish rivers) during 1995 (Electricity Supply Board of Ireland, unpublished report) indicating that the switch to unfed fry planting coupled with smolt friendly generating protocols has achieved positive results. It is possible that previous generating protocols on the Shannon which delayed smolt in the headrace, coupled with the fact that most of the restocking effort was concentrated in the lower half of the Shannon catchment, combined to eliminate salmon from the upper catchment area. It is also possible that fish passes, navigation locks and sluice gates associated with water level control for navigation purposes in the mid-Shannon area may also have played a part in the elimination of salmon from the upper catchment area. In this paper the impact of Irish hydro-electric schemes on salmon was discussed in terms of numbers of returning adults and distribution and relative abundance of naturally spawned juveniles in tributary streams. The impact of the River Shannon hydro-electric scheme and of hatchery operations on the genetic makeup of salmon has recently been investigated (Galvin *et al*, 1994, 1995, in press). The evidence from both protein electrophoresis and single locus minisatellite DNA analysis suggests that genetic drift is the principal force affecting the genetic structure of the salmon populations upstream of Ardnacrusha; low levels of adult escapement to many of the upper tributaries in the past is likely to be the main cause. This may be directly attributable to Ardnacrusha and the regulating weir at Parteen acting as a barrier to migration; alternatively it may be due to the fact that the enhancement stock originated from a tributary located downstream of Parteen (Mulcair River), or that the hatchery itself is located at Parteen, resulting in strains which have no tendency to migrate upstream past the generating station or the regulating weir.

It is clear that the restoration of the River Erne salmon stock represents a serious challenge. The proximity of the generating stations to the sea is largely responsible for this situation in that vary little salmon production occurs downstream of Cathaleen's Fall. The probable entry of non-natal adults to the system also masks the true number of Erne salmon counted / trapped each year.

Today there is also a greater awareness of fisheries matters and the need to operate hydro-electric generating stations in a manner which, within the confines of the original hydro-electric scheme designs, takes into account the requirements of migratory fish. Though the construction of hydro-electric schemes was only one of many factors (arterial drainage, pollution, abstraction, occurrence of ulcerative dermal necrosis disease in Ireland during the 1960's which decimated MSW salmon stocks in affected rivers, Irish drift-net fishery, high-seas fisheries) to impact on salmon in these rivers, it was seen by many as the main factor. Clearly, many of the early practices were inappropriate and in most cases ineffective. As the twenty-first century approaches large scale efforts are being made in Ireland to improve the management of salmon stocks in rivers harnessed for the generation of electricity.

ACKNOWLEDGEMENTS

The work reported on in this paper is part of ongoing fisheries management programmes initiated and funded by the Fisheries Conservation Unit, Electricity Supply Board of Ireland. The authors would like to thank John Cantwell and Cyril O'Dowd for their continuing support and encouragement. The authors also wish to acknowledge the support of colleagues in the Electricity Supply Board, the Marine Institute and other state fisheries agencies and universities throughout Ireland.

REFERENCES

Aitken, P.L., L.H. Dickersen and W.J.M. Menzies, (1966) Fish passes and screens at water power works. *Proc. Inst. Civil Eng.*, Vol 35, pp 29-57.

Cragg-Hine, D. (1972) Further note on the salmonid populations of the Lough Erne tributaries in Northern Ireland. *Journal of Fish Biology* 4, 17-20.

Galvin, P., T.Cross and A.Ferguson, (1994) Genetic differentiation and gene flow in Atlantic salmon, *Salmo salar* L.: a case study of the River Shannon system in Ireland. *Aquaculture and Fisheries Management*, 25 (Supplement 2), 131-145.

Galvin, P., S.Mckinnell, J.B.Taggart, A.Ferguson, M.O'Farrell and T.Cross (1995) Genetic stock identification of Atlantic salmon (*Salmo salar* L.) using single locus minisatellite DNA profiles. *Journal of Fish Biology*, 47 (Supplement A), 186-199.

Galvin, P., J.B.Taggart, A.Ferguson, M.O'Farrell and T.Cross. (in press) Population genetics of Atlantic salmon (*Salmo salar* L.) in the River Shannon system in Ireland: an appraisal using single locus minisatellite (VTNR) probes. *Canadian Journal of Fisheries and Aquatic Sciences*, 53(9).

Jackson, P.A. and D.I.D.Howie, (1967) The movement of salmon (*Salmo salar*) through an estuary and a fish-pass. *Irish Fisheries Investigations* Series A (Freshwater) No. 2.

Kennedy, G.J.A. and C.D.Strange, (1978) Seven years on - A continuing investigation of salmonid stocks in Lough Erne tributaries. *Journal of Fish Biology* 12, 325-330

Murphy, A.M. and J.C.I. Dooge, (1951) The hydraulic fish lift at Leixlip. *Institute of Civil Engineers of Ireland*

McGrath, C.J. (1975) Review. Symposium on the methodology for the survey, monitoring and appraisal of fishery resources in lakes and large rivers. *EIFAC Technical Paper* No. 23, Supplement 1, Vol II, pp 438-446.

Newman, H. (1959) Salmon of the River Lee 1944 and 1945 *Proceedings of the Royal Irish Academy* 59B, 53-69

O'Farrell, M., N. O'Maoileidigh, M. Keatinge, J. Browne and W. O'Connor,(1995) Management of Atlantic salmon (*Salmo salar*) smolt in the River Shannon, Ireland. *Proceedings of the International Symposium on Fishways '95* in Gifu, Japan, 24-26 October.

O'Meallain, S. (1951) Fish pass at Leixlip dam. *Journal of the Department of Agriculture* 48,3-10. Dublin. Stationery Office

Pyefinch, K.A, (1966) Hydro-electric schemes in Scotland - biological problems and effects on salmonid fisheries. *Symp. Inst. Biol.*, Vol 15, pp 139-147.

Southern, R. (1928) Salmon of the River Shannon, 1924, 1925 and 1926. *Proceedings of the Royal Irish Academy* 28B: 38-64

Twomey, E. (1957) Salmon of the River Shannon. *Journal du Conseil* 23(3): 89-96

Twomey, E. (1959) Salmon of the River Erne. *Fisheries Report, Department of Lands.*

Twomey, E. (1964) Salmon of the River Shannon (1957-1962). *Irish Fisheries Investigations Series A(Freshwater)* No. 1. pp 26-35

Twomey, E. (1965a) Investigations into the passage of smolts through a small Kaplan turbine on the River Lee. *ICES Salmon and Trout Committee* CM 1965: M/85

Twomey, E. (1965b) Predation by pike on the River Lee. *ICES Salmon and Trout Committee* CM 1965: M/68

Twomey, E. (1991) The effects of hydro-electric development on salmon stocks in the River Lee. *In: Steer.M.W.(ed.) 1991 'Irish Rivers - Biology and Management'* pp 151-162, Royal Irish Academy, Dublin.

Vickers, K.U. (1969) Observations on the salmonid populations of the Lough Erne tributaries in Northern Ireland. *Journal of Fish Biology* 1, 297-309.

Went, A.E.J. (1938) Salmon of the River Shannon. *Proceedings of the Royal Irish Academy* 44B : 261-322

Went, A.E.J. (1940) Salmon of the River Shannon. *Journal of the Department of Agriculture, Dublin* 37(2):254-310

Went, A.E.J. (1942) Salmon of the River Erne. Results of the examination of a small collection of scales and data. *Scientific Proceedings of the Royal Dublin Society* 22, 471-480.

Went, A.E.J. (1943) Salmon of the River Shannon. *Proceedings of the Royal Irish Academy* 49B : 151-175

Went, A.E.J. (1946) Salmon of the River Shannon - 1944 and 1945. *Journal of Animal Ecology* 15(2): 155-169

Went, A.E.J. (1950) Salmon of the River Shannon in 1946 and 1947. *Journal du Conseil* 16(3): 341-357

Went, A.E.J. (1953) Salmon of the River Shannon (1948-1951). *Journal du Conseil* 18(3): 326-332

Went, A.E.J. (1962) Smolt taggings in Ireland (1952-1955). *Department of Lands report on sea and inland fisheries for the year 1962*, 4pp.

Went, A.E.J. (1964) Irish Salmon. A review of investigations up to 1963. *Scientific Proceedings of the Royal Dublin Society Series A*. 1. No 15, pp 365-412

Went, A.E.J. (1970) Salmon investigations on the River Shannon. *Atlantic Salmon Association Centennial Award Fund.* Series of Lectures pp 17-22

DESIGN OF SIMPLE DENIL AND DEEPENED ALASKA STEEPPASS FISHWAYS

Mufeed Odeh and Alex Haro
Anadromous Fish Research Center
U.S. National Biological Service
One Migratory Way
Turners Falls, Massachusetts 01376
United States of America
odehm@external.umass.edu

ABSTRACT

Coupling of hydraulic and biological (fish behavior) data is important for evaluation and design of fishways. However, these measurements are rarely taken simultaneously or under controlled conditions. Flow and fish behavior was studied in a large Simple Denil (122 cm wide x 122 cm deep) fishway and a deepened Alaska Steeppass (56 cm wide x 102 cm deep) fishway installed in a large controlled flume environment. Discharge capacities, water velocities, and turbulence of both fishways were measured at slopes ranging from 10% to 45% at varying hydraulic heads. The Denil fishway passed a maximum of 0.85 $m^3 \cdot s^{-1}$ at 114 cm head and a slope of 1:4, and a minimum of 0.27 $m^3 \cdot s^{-1}$ at 76 cm head and a slope of 1:10 . The Steeppass fishway discharged a maximum of 0.54 $m^3 \cdot s^{-1}$ at 100 cm head and a slope of 1:2.2 and a minimum of 11 $m^3 \cdot s^{-1}$ at 46 cm head and a slope of 1:10. Water velocities ranged between -0.7 and 2 $m \cdot s^{-1}$ in the Denil and between 0.3 and 1.85 $m \cdot s^{-1}$ in the Steeppass. Discharge versus head rating curves and velocity profiles were developed for both fishways. Behavior of actively migrating adult American shad (*Alosa sapidissima*) and blueback herring (*A. aestivalis*) collected from the Connecticut River was monitored within each fishway. Percent passage of introduced groups of fish and transit times of individuals through each fishway were determined using passive integrated transponder (PIT) tag technology. Fish were tested at fishway slopes of 1:8 and 1:6 and low (76cm-Denil, 61cm-Steeppass) and high (107cm-Denil, 91cm-Steeppass) operating headponds. For both species, percent of fish passed was greater at 1:8 slope than 1:6 slope in the steeppass fishway. Percent passage of both species was greater at low headpond than high headpond in the Denil fishway. Percent passage of shad was greater at high headpond than low headpond in the steeppass fishway. Transit time of shad was longer at low headpond than high headpond in both fishways. Also, transit time was longer at 1:8 than 1:6 for both species in the steeppass and for shad only in the Denil. Water temperature was positively correlated with percent passage and negatively correlated with transit time under certain conditions. Efficiency of fish passage for Denil-type fishways is dependent on hydraulic operating head, temperature and species-specific behaviors in addition to fishway type and slope.

KEY-WORDS: Fishways/Denil/Alaska Steeppass/American shad/Blueback herring/rating curves/fish passage/PIT tags

INTRODUCTION

Fishways are hydraulic structures that assist upstream migrating fishes in overcoming obstacles blocking the way to spawning, feeding, or other habitat. Fishways can be divided into three types: Pool and weir, vertical slot, and Denil (Clay 1995). All fishway designs are based on the principle of energy dissipation to reduce water velocities and facilitate ascent of fish. This means the flow inside the fishway is very turbulent and includes secondary flows and reverse flow patterns around the energy dissipating baffles, vanes, or weirs. Therefore, understanding the flow behavior inside fishways and optimizing their hydraulic design is essential for successful biological performance.

Simple Denil and Alaska Steeppass fishways are essentially straight rectangular channels with energy dissipating baffles or vanes placed on the sides and bottom, Figure 1 shows typical cross sections of the fishway baffle. Both of these fishways are descendants of the original type that was developed by G. Denil in Belgium in 1908. The present form of the Simple Denil is modelled after the one recommended by the Committee on Fish Passes (1942). Ziemer (1962) developed the existing Model A Alaska Steeppass fishway. McLeod and Nemenyi (1940) performed a thorough investigation of design of fishways using over 40 models of Denil type and other channel type fishways by testing full scale versions of promising designs with fish. Katapodis and Rajararatnam (1983) give a good historical perspective on the development of the Simple Denil and Alaska Steeppass designs used in this research project.

Although understanding the hydraulic characteristics by engineers and biologists is essential, biological evaluation of fishways is a task that must not be overlooked. To date, studies have concentrated on either the hydraulics or the biological aspects of a fishway, but rarely on both simultaneously under controlled conditions. The literature is replete with reviews on the development, design and construction of these fishways and their uses for passing salmonid and nonsalmonid fishes (Clay 1995; Larinier 1992; Rajaratnam 1992; Rajaratnam and Katopodis 1991; Beach 1984; Katopodis and Rajaratnam 1983; Ziemer 1962; and McLeod and Nemenyi 1940).

The main objectives of this research project were to develop the hydraulic characteristics of the two fishways and simultaneously evaluate their biological performance using two actively migrating anadromous species; American shad (Alosa sapidissima) and blueback herring (A. aestivalis)

METHODS AND EXPERIMENTAL APPARATUS

Our approach to studying flow behavior of the Denil and steeppass fishways was to use the headpond elevation upstream of the fishway rather than flow depth inside of it. The hydraulic characteristics would be described by rating curves and equations relating upstream headpond elevations with the discharge through each fishway at various slopes. Upstream headpond elevation is stable and can be determined within a given range when constructing a fishway. It is usually available from hydrologic data for the region.

Two experimental fishways, a Simple Denil and an Alaska Steeppass , 7.62 m long each were tested at the Conte Anadromous Fish Research Center (CAFRC). The steeppass fishway was a deepened model A prefabricated aluminum structure similar to that developed by Ziemer (1962). The fishway had a 56 cm overall width (35.5 cm clear width) and 102 cm depth (conventional fishway depth is 68.6 cm). The baffles are placed 25.4 cm apart at 30° into the flow. The Denil fishway was fabricated at CAFRC using the form of Simple Denils recommended by the Committee on Fish

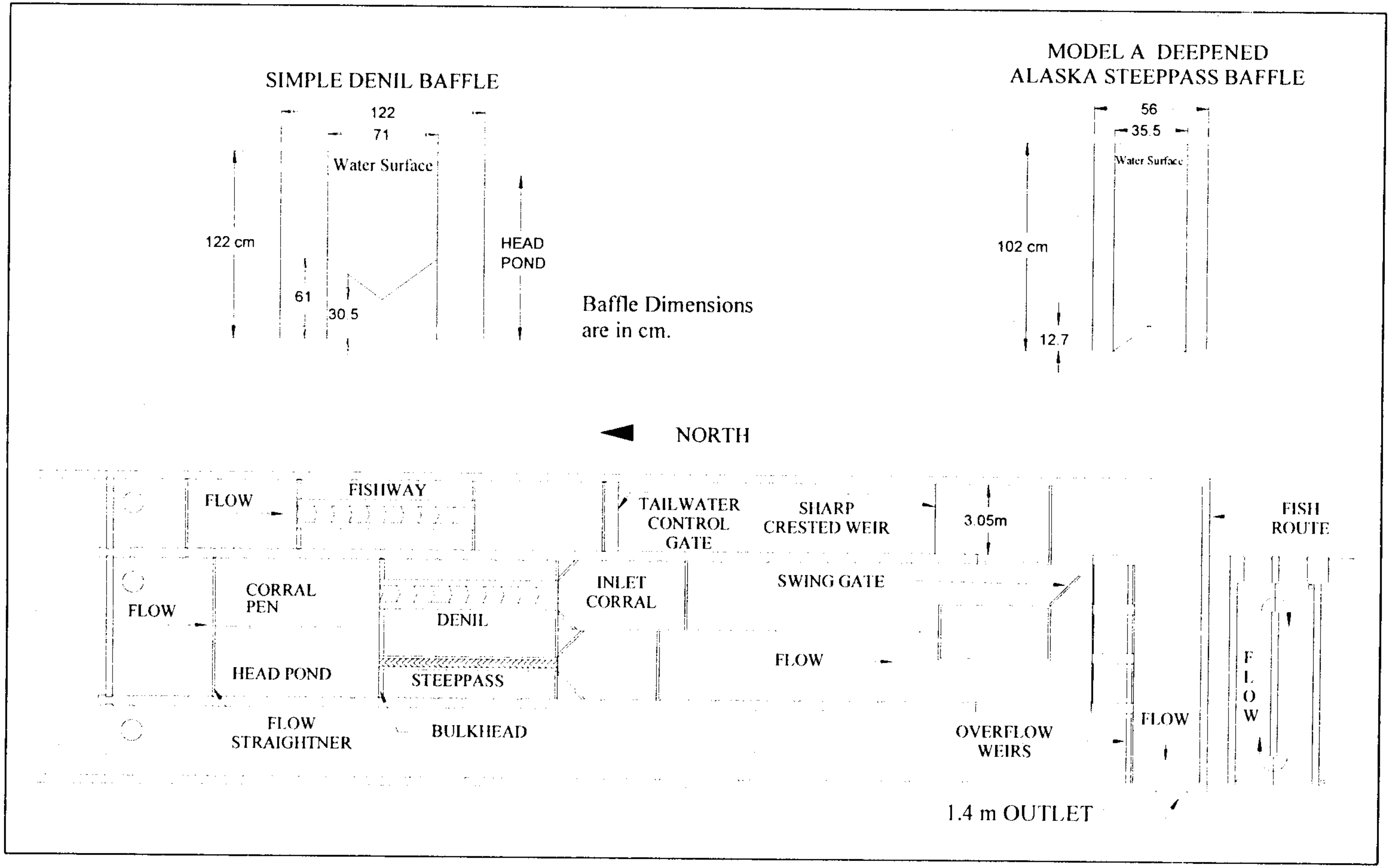


Figure 1. Hydraulic and Biological experiments setup - CAFRC flumes

Passes (1942) using timber, steel, and plywood. It was 1.22 m wide (0.71 m clear opening) by 1.22 m deep. The baffles were placed 76 cm apart and at 45° inclination to the bottom.

The two fishways were hydraulically rated separately in the 3.05 m wide flumes at CAFRC (Fig. 1). A head pond 3.05 m wide by 5 m long and 6 m deep was used to simulate a dam forebay upstream. Water was supplied by a 91 cm diameter opening through the concrete floor of the flume. Water at CAFRC is obtained by gravity flow from the Cabot Hydropower Station canal, Turners Falls, Massachusetts. The canal is adjacent to the CAFRC flume building and provides a 7 meter gravity head.

The hydraulic experiment setup used for both fishways is shown in the east flume at CAFRC (Fig. 1). A flow straightener upstream between the fishways and the inlet ensured calm forebay water with uniform depth and a gate downstream was used for tailwater depth adjustment. Downstream of the fishway, a calibrated sharp-crested weir was used to measure the flow. Fishway flow was discharged back into the river via a 1.4 m pipe at the southwest end of the flume building.

Hydraulic rating curves were developed by taking at least 12 measurements of flow versus headpond elevation at each fixed slope. The headpond elevation refers to the depth measured from the water surface to the fishway invert (floor) at the upstream end. Compared with the depth of water inside the fishway, this did not fluctuate and was easier to measure. However, an attempt was made to determine a hydraulic grade line in the Denil fishway to show an average water surface profile. This was done by placing piezometer taps at various locations along the fishway and reading the static head using differential pressure cells. Headpond elevation upstream of the fishways was kept constant at each flow and slope. The Denil fishway was rated for slopes ranging between 1:8 (12.5%) and 1:4 (25%), and the steeppass was rated at slopes between 1:10 (10%) and 1:2.2 (45%).

Velocity profiles were obtained at 10 cross sections (19 positions each) along the Denil and 7 cross sections (25 positions each) along the steeppass. Combinations of two headpond elevations and two slopes in each fishway were used for velocity measurements. These were high headpond (107 cm - Denil, 91 cm -steeppass) and low (76 cm - Denil, 61 cm -steeppass) at slopes of 1:6 (16.7%) and 1:8 (12.5%). The same combinations of headpond elevations and slopes were used during the biological evaluation. Velocities in the fishways were measured with a Marsh-McBirney two-directional electromagnetic velocity meter (Model 521) with a 13 mm diameter spherical probe . Time averaged values including flow direction were recorded by means of an analogue/digital board interface between the meter and a personal computer.

We used an application of passive integrated transponder (PIT) technology (Texas Instruments TIRIS Series 2000 readers and model RI-TRP-WB2B-03 read/write PIT tags) to monitor volitional movements of upstream migrant shad and blueback herring. For details of the system design and operation, see Castro-Santos et al. (in press). Upstream migrant shad and herring were collected from a nearby fishway, transported to the flume facility, and held for testing in open circulating ponds. Figure 1 shows the biological test setup in the 6.1 m wide flume. Fish were placed in the inlet corrals at the beginning of the test. At the end of each test fish were removed from the inlet corral (if not passed) or from the upstream corral pen (if passed).

PIT antennas were constructed on-site, and consisted of wire coils bent to the shape of fishway baffle openings. Pairs of antennas were connected in series with one antenna of each pair installed in each fishway. This arrangement allowed both fishways to be monitored by four antennas, each connected to a separate reader. PIT tags were fitted with #6 hooks and attached to fish by insertion of the hook through the cartilage at the base of the dorsal fin. At the start of each trial, groups of 20 tagged fish were crowded into the flume below each fishway and allowed to volitionally ascend for 3 hours. Fish were then removed from the flume and their tag numbers, passage status (above or below fishway), and length were recorded. A minimum of six trials were run for each treatment condition (slope, headpond, and fishway type). Water temperature was recorded at the beginning of each trial.

PIT tag passage data were recorded on four computers (one computer per reader). Data were compiled into a single database with each record marked with trial, reader, and species identifiers. The data were then sorted on trial, fishway, identification code, and passage time, yielding a complete time series data set on the passage of each individual. The passage data was then analyzed to generate percent of fish passed per trial, and transit time (duration of time required to ascend from the bottom of the fishway to the top) for each fish passed.

RESULTS

Hydraulics

The Simple Denil fishway passed a maximum operational flow of 0.85 $m^3 \cdot s^{-1}$ at 114 cm head and a slope of 1:4 (25%), and a minimum operational flow of 0.27 $m^3 \cdot s^{-1}$ at 76 cm head and a slope of 1:10 (10%). The steeppass fishway passed a maximum operational flow of 0.54 $m^3 \cdot s^{-1}$ at 100 cm head and a slope of 1:2.2 (45%), and a minimum operational flow of 0.11 $m^3 \cdot s^{-1}$ at 46 cm head and a slope of 1:10 (10%).

In the steeppass fishway a flow versus headpond elevation curve was distinct for each slope, (Fig. 2). Experimental data was found to best fit an equation of the form $Q = a + bh^c$, where Q is flow in $m^3 \cdot s^{-1}$, h is headpond elevation in meters, and a, b and c are constants. The value of c was found to be approximately the same for all curves $c = 1.508$. The slope b, however, changed with the fishway slope. Equation (1) gives the value of b as a function of the fishway slope s.

$$b = (0.826 + 0.115 \ln(s))^2 \tag{1}$$

Therefore, in a model A deepened Alaska Steeppass fishway, the discharge can be predicted by using equation (2) below, given the upstream headpond elevation, h, and the value of b from equation (1) above.

$$Q = -0.12 + bh^{1.508} \tag{2}$$

Fishway discharge capacities at slopes other than those in Figure 2 can be estimated by interpolation.

In the steeppass fishway, flow was faster towards the bottom and slower at the surface, Fig.3. Velocities ranged between 1.85 $m \cdot s^{-1}$ in the first half of the fishway, towards the bottom, at high headpond and slope of 16.7% and 0.3 $m \cdot s^{-1}$ close to the water surface at low headpond and slope of 12.5%. Flow throughout the fishway, downstream of the vena contracta, was fully developed and velocities stayed reasonably uniform. Also, despite the severe turbulence at the surface, main flow velocities remained in the downstream direction throughout the fishway. Figure 3 shows three different presentations of the flow velocities inside the steeppass fishway. The centerline velocity profile was taken in a plane passing through the invert of the baffle. It clearly demonstrates the higher velocities at the bottom of the channel.

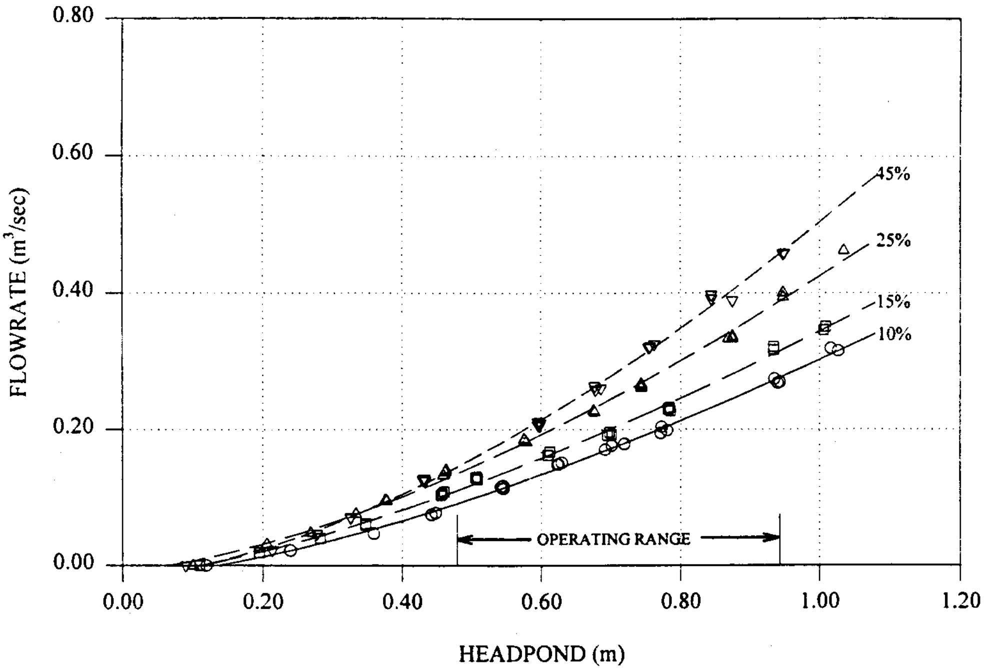
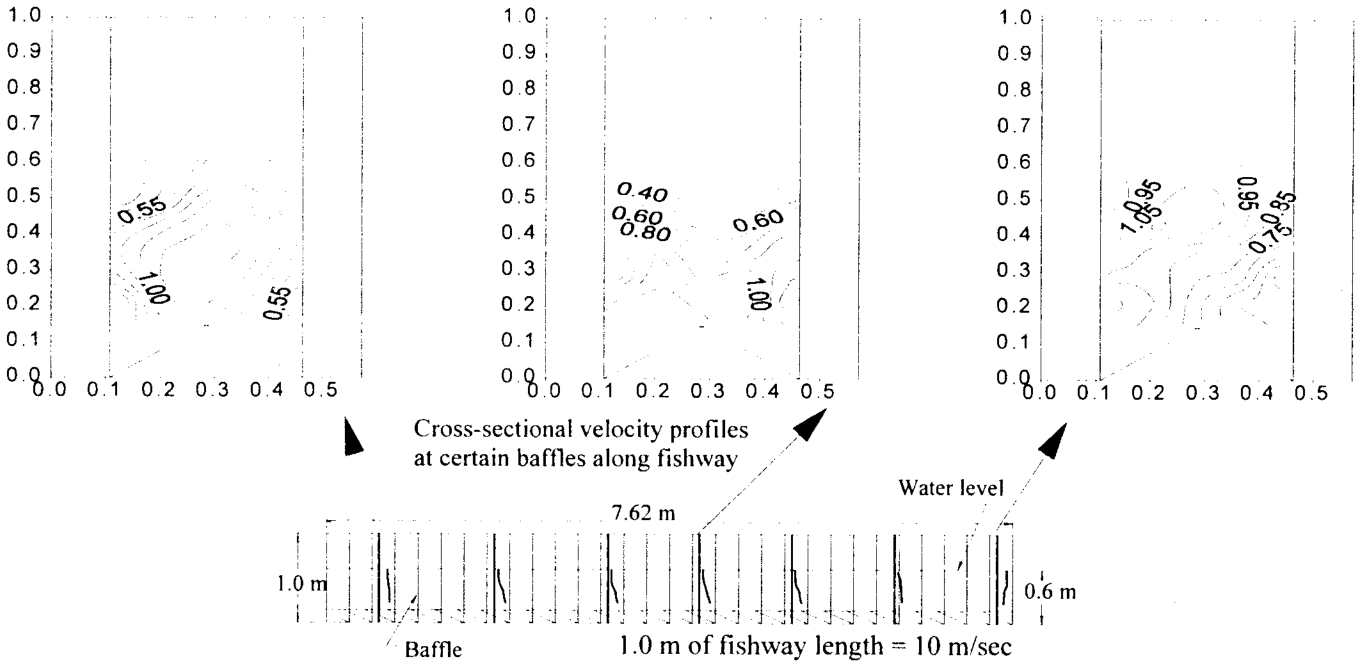
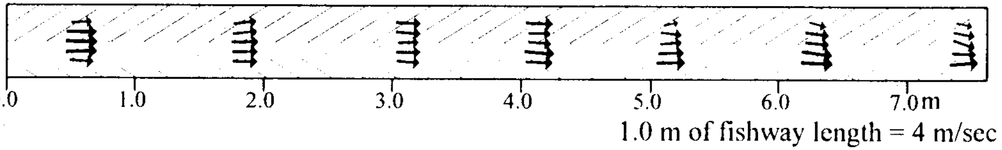


Figure 2. Hydraulic rating curves for Model A Deepened Alaska Steeppass fishway



Cross-sectional velocity profiles at certain baffles along fishway

Centerline velocity profiles along the fishway



Velocity profiles across the baffle at mid-depth (0.21 m from water surface)

Figure 3. Velocity presentations in a steeppass fishway (1:8 slope & 0.61 headpond)

The hydraulic rating curves for the Denil fishway were not as distinguished for the different fishway slopes. At the same headpond elevation a variation of about ±5% was found in the flow at different fishway slopes. A statistical analysis of the residuals between experimental data and predicted values proved this was true. Therefore, only one equation of the form ($Q=a+bh^{c}$) was derived to predict Denil fishway discharge given the desired headpond elevation and fishway slope, Fig. 4.

$$Q=-0.056+0.684h^{2.132} \tag{3}$$

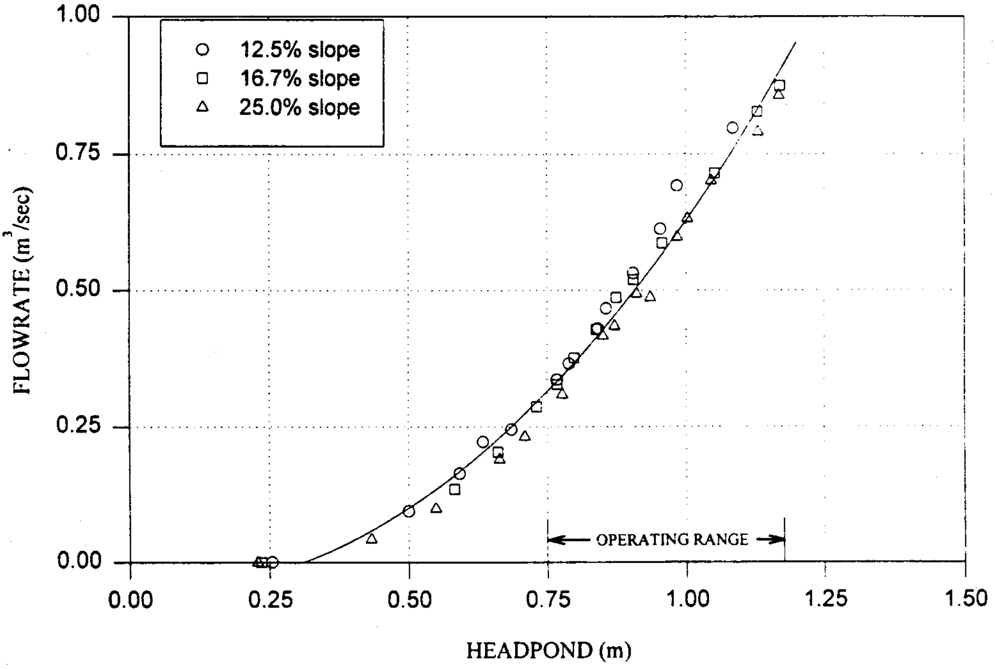


Figure 4. Hydraulic rating curve for a simple Denil fishway

Flow velocities in the Denil varied in cross section from top to bottom as well as along the length of the fishway. Velocities were positive and higher at the top of the water column, and lower and sometimes negative close to the baffle. At the upstream end (vena contracta), where flow is not yet fully developed, velocities were about 30 to 40% higher than the rest of the Denil fishway. Downstream of the vena contracta, after the second or third baffle, flow became very turbulent and velocities decreased. Between baffles the velocity was lower than at the baffle itself, where the flow contracted to pass through a smaller area (Fig. 5).

Water velocities in the Denil fishway varied from a maximum of about 2 $m \cdot s^{-1}$ in the upstream section (close to the vena contracta at high headpond and slope of 1:6) to a minimum of -0.70 $m \cdot s^{-1}$ in the fully developed turbulent flow, in the middle and downstream sections (close to the baffle invert and at all headpond slope combinations).

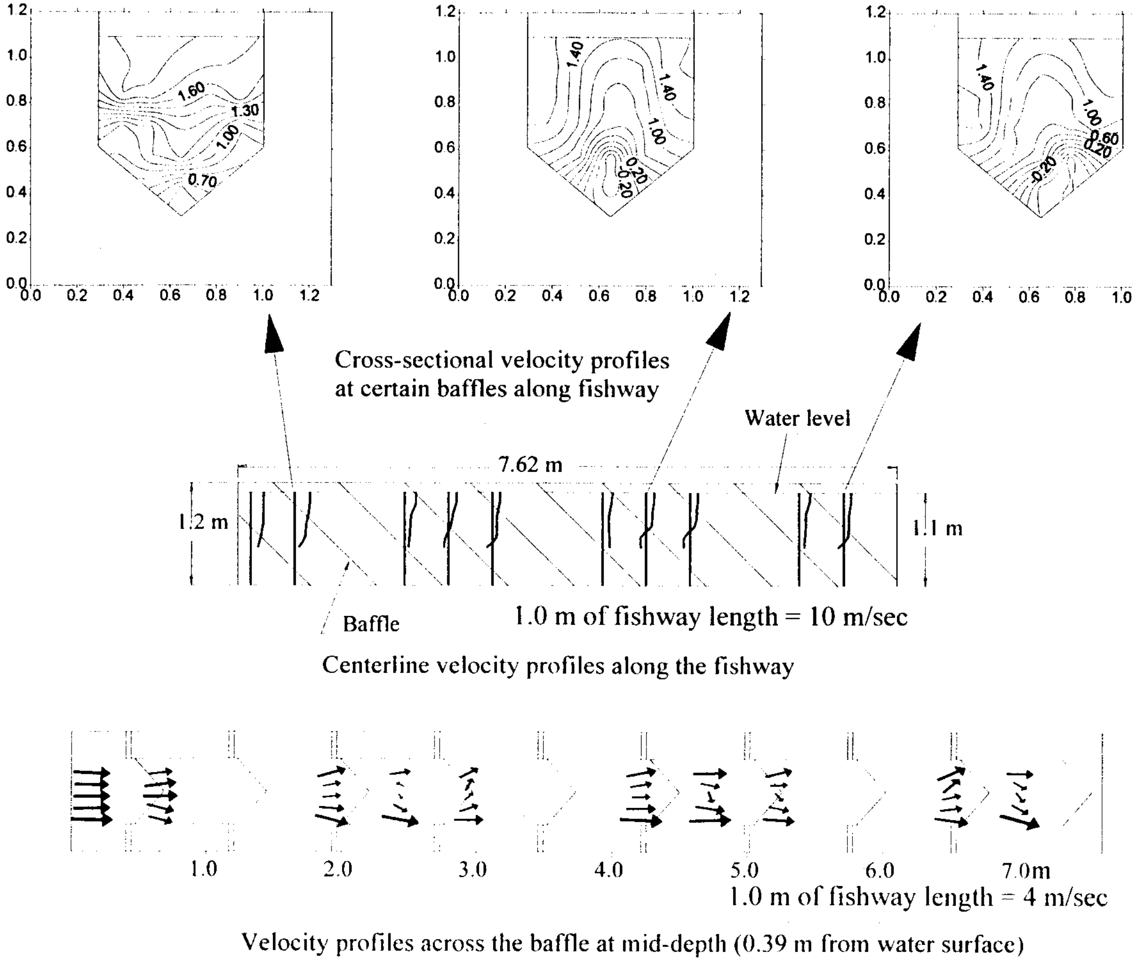


Figure 5. Velocity presentations in a Denil fishway (1:6 slope & 1.07 headpond)

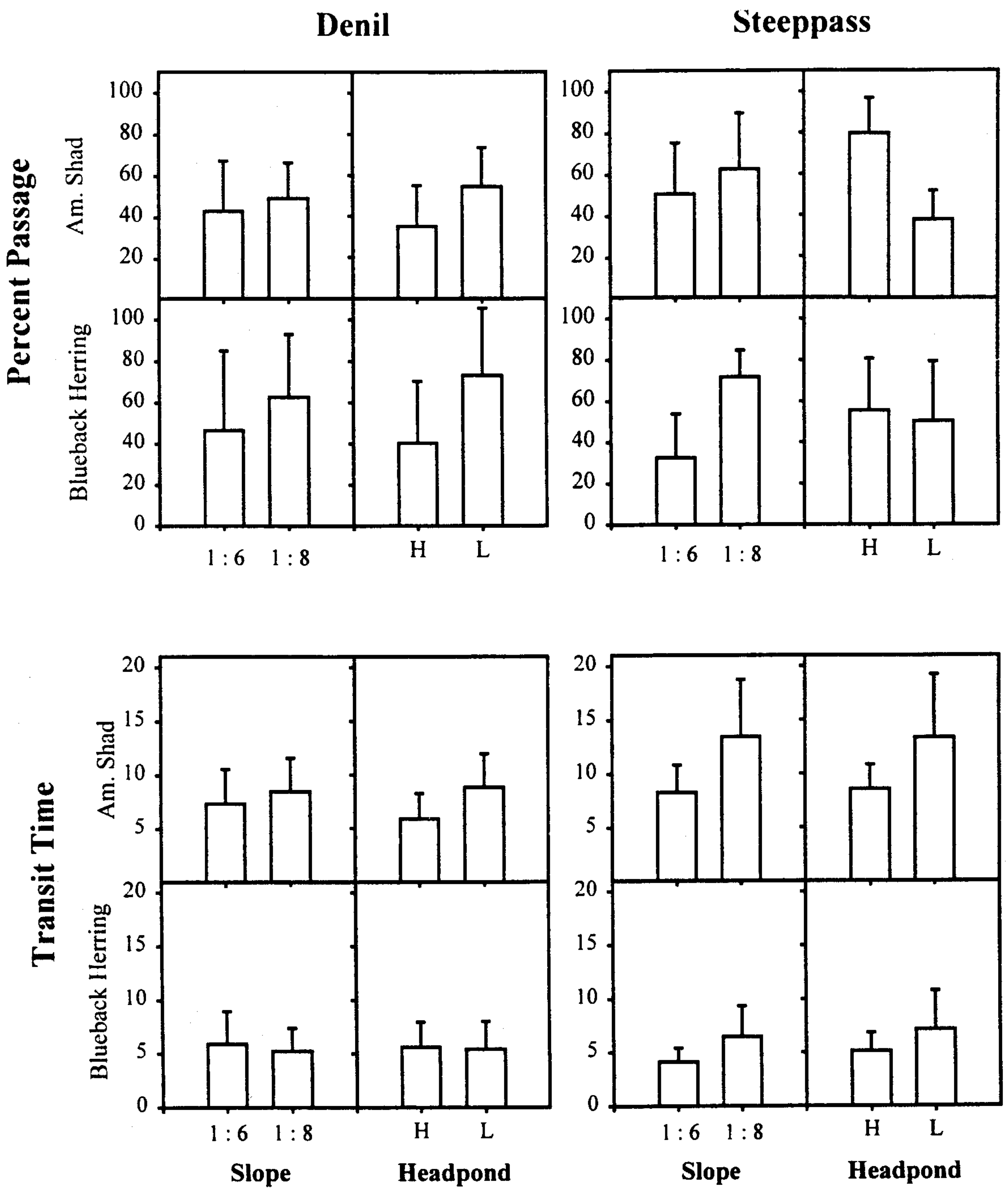


Figure 6. Means and standard deviations of percent passage and transit times of American shad and blueback herring in Denil and Steeppass fishways. Means are presented by slope and Headpond level ('H' = high, 'L' = low). Data for percent passage refer to trial means; data for transit time refer to individual fish.

Fish Passage

Percent passage and transit time results for each fishway, species, and treatment are given in Fig. 6. For both species, percent passage was greater at 1:8 slope than at 1:6 slope in the steeppass fishway. Percent passage of both species was greater at low headpond than at high headpond in the Denil fishway, but percent passage of shad was greater at high headpond than at low headpond in the steeppass fishway.

Transit time of shad was longer at low headpond than at high headpond in both fishways. Also, transit time was longer at 1:8 than 1:6 for both species in the steeppass and for shad only in the Denil.

Water temperature was positively correlated with percent passage and negatively correlated with transit time under certain conditions (Fig. 7).

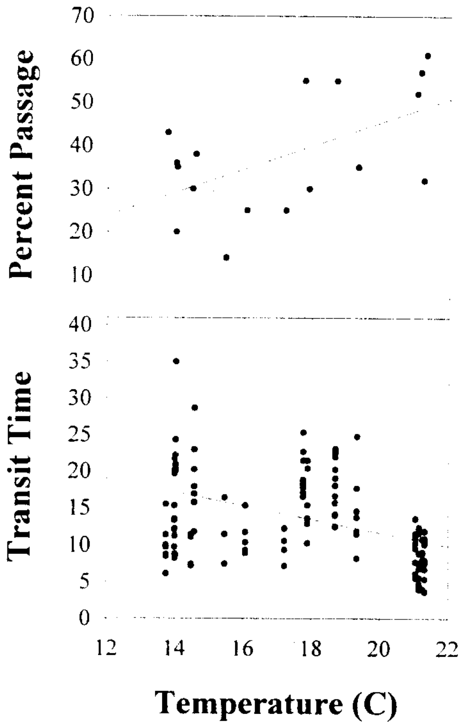


Figure 7. The effect of temperature on percent passage and transit times of American shad. Data are from the Steeppass at low headpond.

DISCUSSION

A hydraulic operating range, based on the upstream headpond elevation, for each fishway was defined, 0.46 to 0.91 m in the steeppass, and 0.75 to 1.0 m in the Denil (Figs. 2 & 3). The limits indicated take into consideration depth and flow necessary to pass fish. Water depth must be adequate to accommodate fish swimming up it. Denil fishways are normally operated with a minimum water depth of 76 cm and steeppass fishways with 46 cm depth above the invert. Also, fishway discharge needs to be sufficient to induce the secondary flows behind the baffles that ultimately cause the reduction in velocity to below fish burst speeds. At lower water depths the baffles act as weirs upstream and downstream of a resting pool, rather than energy dissipators.

Past studies (White and Pennino 1980; Katopodis and Rajaratnam 1984) proved that Froudian laws of similitude can be applied in the case of flow through fishway structures of varying sizes. Both fishways had free surface flows that were fully turbulent. Therefore, Froude number similitude between model and prototype applies. Using the proper relationship from Froude laws of similitude (French 1985), flow through different size fishways can be predicted. All fishways must be, however, geometrically similar to those used in our research project.

In both fishways the water velocities and turbulence intensities increased as the headpond elevation and/or the slope increased. However, the present lack of knowledge of the spatial position of the fish as they ascend each fishway prohibits correlating the effect of velocity and turbulence to passage rates.

A low slope resulted in higher percent passage for both species in the steeppass fishway. The inconsistent effect of slope and headpond on percent passage and transit times make other generalizations independent of fishway type and species difficult. Variation in slope and headpond affects both absolute water velocity and turbulence within both fishways. The individual and additive effects of velocity and turbulence on passage behavior and success are as yet unknown. Reduction of headpond level also results in a decreased area through which fish can ascend, which may restrict passage of larger species through small fishways.

We conclude that the effects of fishway slope and headpond on percent passage and transit time are complex, and are influenced by the inherent hydraulic characteristics of each fishway type and their effects on swimming speed and control (maneuverability and stability) of each species. Further detailed analysis of hydraulic characteristics of both fishway types and finer resolution of movements and position of fish within the fishway under a wider range of slopes may yield additional information about these relationships.

ACKNOWLEDGMENTS

The authors thank John Noreika and Theodore Catro-Santos for their experimental assistance and data collection and analysis. Steve Walk and Philip Rocasah provided valuable assistance with construction, instrumentation and fish collection. Joe Teter of the U.S. Forest Service provided the Alaska steeppass structure and supplemental funding.

REFERENCES

Beach, M.A. (1984). Fish Pass Design. Fisheries research Technical Report No. 78, Ministry of Agriculture, Fisheries and Food, Lowestoft, England. 46 pp.

Castro-Santos, T., A. Haro and S. Walk. (in press). A Passive Integrated Transponder (PIT) Tag System for Monitoring Fishways. Fisheries Research.

Clay, C.H. (1995). Design of Fishways and Other Fish Facilities. 2nd ed. CRC Press, Inc. Florida, USA.

Committee on Fish Passes (1942). Report of the Committee of Fish Passes, the Institution Research Committee, Institution of Civil Engineers, William Clowes and Sons, Limited. London, England. 59p.

French, R.H. (1985). Open-Channel Hydraulics. McGraw-Hill, Inc. New York. 739 p.

Katapodis, C. and N. Rajararatnam (1984). Similarity of Scale Effects of Denil Fishways. Symposium on Scale Effects in Modelling Hydraulic Structures. Technische Akademie Esslingen, W. Germany. pp. 2.8-1 - 2.8-6.

Katapodis, C. and N. Rajararatnam (1983). A Review and Laboratory Study of the Hydraulics of Denil Fishways. Canadian Technical Report of Fisheries & Aquatic Sciences No. 1145. Western Region Dept. of Fisheries and Oceans, Canada. 181 p.

Larinier, M. (1992). French Journal of Fishing and Fish Breeding - Aquatic Resource Management. French Fisheries Board. Nos. 326-327 - 3rd and 4th quarters of 1992, 65th year. 199 p.

McLeod, A.M. and P. Nemenyi (1940). An Investigation of Fishways. University of Iowa, Iowa Institute of Hydraulic Research. 63 p.

Rajaratnam, N. (1992). Recent Contributions to the Hydraulics of Fishways. Dept. of Civil Engineering, University of Alberta, Edmonton, Canada.

Rajaratnam, N. and C. Katopodis (1991). Hydraulics of Steeppass Fishways. Canadian Journal of Civil Engineering. 18(6) pp. 1024-1032.

White, D. and B. Pennino (1980). Connecticut River Fishways: Model Studies. ASCE Journal of the Hydraulic Division. 106(HY7) pp. 1219-1233.

Ziemer, G.L. (1962). Steeppass Fishway Development. Informational Leaflet No. 12, Alaska Department of Fish and Game, Division of Engineering and Services. 35 pp.

FISH MIGRATION FROM TWO REGULATED LAKES TO OUTCOMING RIVERS MONITORED BY HYDROACOUSTICS

Teppo Vehanen, Pekka Hyvärinen and Aki Mäki-Petäys
Finnish Game and Fisheries Research Institute, Kainuu Fisheries Research and Aquaculture
Manamansalontie 90, FIN-88 300 Paltamo
Finland
kainuun.kvl@rktl.fi

ABSTRACT

Fish migration from two large northern Finnish lakes, Lake Oulujärvi and Lake Inarijärvi, to their outcoming rivers were studied by echo-sounding, exploratory fishing and fish marking. Both lakes are regulated for hydroelectric purposes. The area of Lake Oulujärvi is 928 km^2. The mean depth of the lake is 7.6 m, and the average annual amplitude of the water level regulation is 1.9 m. The mean flow (MQ) in the outcoming River Oulujoki is 229 m^3/s. The surface of Lake Inarijärvi is 1071 km^2. The mean depth of the lake is 14 m. The annual mean amplitude of water level fluctuation is 1.48 m. The mean flow (MQ) in the outcoming River Paatsjoki is 157 m^3/s. In both rivers two sonar stations with stationary up- and down-looking transducers were used in data collecting for one year. Exploratory fishing was done for species identification. Fluorescent pigment marking was done to separate the stocked and naturally reproducing brown trout in Lake Inarijärvi.

The fish migration rate in the River Oulujoki was larger than in the River Paatsjoki. In the River Oulujoki the fish migrated mainly downstream and in the River Paatsjoki both down- and upstream. In the River Paatsjoki, large fish showed active migration in the spring and autumn, whereas in the River Oulujoki the increase in the migration occurred simultaneously in all the size groups. The different species composition and the different nature of the lakes together with the different regulation practises was proposed to result in the varying migration and behavioral patterns.The importance of downstream migration was greater for the fish community in Lake Oulujärvi, whereas in Lake Inarijärvi the importance of the migration in the River Paatsjoki can have a positive effect through the spawning and feeding areas in the river. We conclude that no barriers to prevent fish migration are needed in these rivers.

KEY-WORDS: Hydroacoustics/Exploratory fishing/Fish marking/Fish migration/Migration rate/Lake regulation/Fish community/Stocking/Whitefish/Brown trout

INTRODUCTION

Many lakes in northern Finland are regulated for hydropower production. Due to the decrease in natural fish reproduction in those lakes fish stocking is carried out as a compensatory process. The most important species stocked are whitefish (*Coregonus lavaretus* L. sl.) fingerlings and brown trout (*Salmo trutta* (L.) m. *lacustris*) smolts. The results from the stocking have been widely varying. One possible reason for these variations is the descending of stocked fish from the regulated lakes to the rivers (e.g. Heikinheimo-Schmid and Huusko, 1988). In order to prevent the migrations, demands have been made to establish physical barriers in the outlets of several rivers.

The application of hydroacoustics theory is currently accepted in fisheries research (Forbes and Nakken, 1972; Johannesson and Mitson, 1983; Thorne, 1983; Bodholt, 1990). With hydroacoustics it is possible to conduct absolute *in situ* measurements of fish abundance. The major disadvantage of the method is the lack of biological samples.

Portable hydroacoustic instruments have been used to detect pelagic fish abundance in marine environment and in lakes. The riverine conditions are often shallow and acoustically very noisy. Stationary transducers improve signal to noise characteristics and eliminate the problem of resolution near the boundaries (Thorne, 1983). Several single beam transducers can produce information of fish behaviour eg. the direction of migration. Stationary transducers have been used in studies solving problems related to fish migrations in hydroelectric dams (Johnston *et al.*, 1993), but they have also been used in studying the trout distribution in a lake environment (Stables and Thomas, 1992).

Fish migrations from two large regulated lakes in northern Finland to their outcoming rivers were studied with hydroacoustics using stationary transducers in 1990-1991 and 1993-1994. The purpose of the study was to evaluate the need for any physical barriers to prevent the fish migration.

MATERIAL AND METHODS

Study area

The area of Lake Oulujärvi is 928 km^2 (Fig 1). The mean depth of the lake is 7.6 m, and the maximum is 36 m. The lake has been regulated since 1951. The average annual amplitude of the water level regulation has been 1.9 m during 1959-89. This exceeds the preregulation average with 0.8-0.9 m. High water level has been decreased by 60 cm from the natural. In spring the water level of the lake is rapidly lowered to collect the meltwater. This is why the water level in April is about 1 m lower than in the unregulated lake. In the outcoming River Oulujoki, which flows into the Gulf of Bothnia, the flow varies greatly due to the regulation (Fig. 2A). The first power plant is situated in the outlet of the River Oulujoki and followed by six more in the main basin of the river. Two sonar stations were placed into the river in 1990-1991.

The surface of Lake Inarijärvi is 1071 km^2 (Fig. 1). The mean depth of the lake is 14 m and the maximum 96 m. The lake has been regulated since the 1940's. According to average values (from 1959 to 1989) the annual mean amplitude of water level fluctuation has been 1.48 m, which is 0.30-0.35 m more than in the unregulated lake. Compared to the natural situation the upper high water is on average 52 cm higher and the low water about 18 cm higher than in the unregulated lake. The outcoming River Paatsjoki, drains into the Barent Sea. Because of smothering regulation the variations in the flow of the River Paatsjoki are smaller than those in the River Oulujoki (Fig. 2B). There are seven power plants in the river. The first is situated five kilometres from the river's outlet. Two sonar stations were placed in the River Paatsjoki and they collected data from 1993 to 1994.

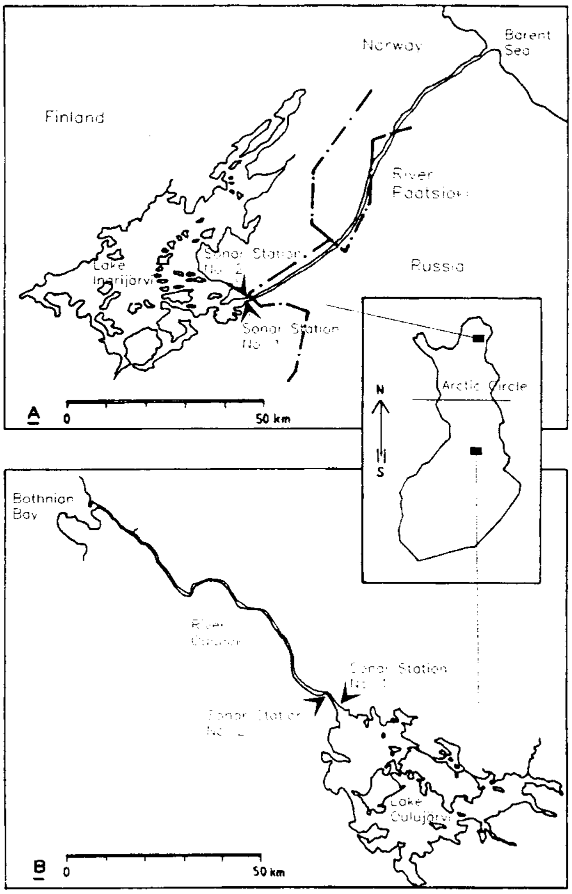


Figure 1: Location of the study areas and the sonar stations.

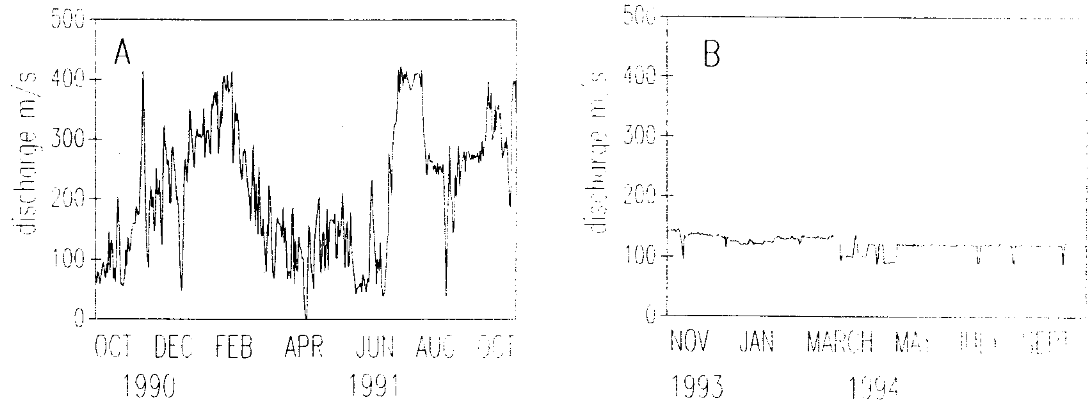


Figure 2. The mean daily discharge of the River Oulujoki (A) during the study from Oct 10,1990 to Oct 18, 1991, and the mean daily discharge of the River Paatsjoki (B) during the study from Nov 12, 1993 to Oct 24, 1994.

The greatest difference in the fish community between the lakes is the presence of smelt and a variety of cyprinid species in Lake Oulujärvi. Lake Inarijärvi is a typical subarctic lake, where salmonid stocks are strong and the cyprinids are not precent, with the exception of minnow. Stocking of whitefish fingerlings and brown trout smolts is carried out intensively in both lakes.

Hydroacustic sampling

The hydroacoustic sampling was conducted by a River Sonar Station (RSS) equipped with stationary down- and up-looking transducers (Echo Research Co. 1990) and controlled by specific program (HAS, Henk Co. 1993). The system was especially designed for hydroacoustic surveys in rivers less than 30 m deep. It was based on independent 192 kHz echo sounding units with 400W transmitters which were both located in waterproof container. The time varied gain (TVG) was 40lgR and the dynamic range was 120 dB. 4-10 pings per second were transmitted according to the depth of the site. Lowrance THS-1992-20 transducers were used, the beam angles were 20° (-3db), and the tilted 10 degrees against the river flow (up-looking transducers) or along the flow (down-looking transducers).

The calibration of the system was done twice in both locations with a standard copper sphere (Foote, 1982). The first calibration took place after deployment, and the second after a period of six months' continuous activity. The calibration was done separately to each sonar.

The hardware equipment used obtained echo strength (ES) instead of target strength (TS), which are known to differ in directivity:

$$TS=ES-20\log(D), \tag{1}$$

where: D = transducer directivity towards the target. To compensate for the lack of directivity information a precise modification of the Craig and Forbes (1969) method (Forbes and Nakken, 1972) was applied to provide an estimate of the target strength (TS). The detection of fish echoes begun 1 m from the transducer. The fish lengths (L) were calculated from the target strength (Lindem, 1983; Jurvelius, 1991):

$$\log L = (TS + 67)/20 \tag{2}$$

The equation was originally developed for smelt and whitefish, but in this study all fish species were expected to reflect the same target strength with respect to the length of the specimens. Only fish larger than 7 cm (-49.5 dB) were taken into consideration, because of a lack of biological samples from smaller fish and also the impurities in the acoustic data (gas bubbles, leaves etc.).

The direction of fish movement was calculated from the entrance and exit ranges of the targets as they passed through the beam. The transducers were placed at an angle of 10 degrees upstream or downstream. Target tracking technique (Johnston, 1985; Johnston and Hopelain 1987) was applied to all detected single targets on every sonars range to estimate angle aspects of moving targets towards transducer:

$$a = \arctan\frac{R1 * \cos(\theta/2) - R2 * \cos(\theta/2)}{R1 * \sin(\theta/2) + R2 * \sin(\theta/2)}, \tag{3}$$

where: a = angle of fish passage, R1 = entrance range of target, R2 = exit range of target and θ = transducer beam width at -3 dB.

Two sonar stations were constructed in the River Oulujoki and also in the River Paatsjoki (Fig 1., Table 1). The up-looking sonars were anchored on the bottom and the down-looking sonars were installed on a cable stretched across the river. The sonar units in the River Paatsjoki were installed in the same manner as in the River Oulujoki with the exception that the units located at the surface were anchored to a small metal bridge built across the river.

Table 1: Information of the two sonar stations constructed in the River Oulujoki and the River Paatsjoki to detect the number of fish passing the stations.

River	Sonar station	Data collection time	River width (m)	Depth of sonar locations (m)	Distance between sonars (m)	Number of down-looking sonars	Number of up-looking sonars	Coverage of the cross section by sonar beams %
Oulujoki	No.1	10.10.1990-18.10.1991*	140	3-8	5-15*	16*	6*	40*
	No.2	10.10.1990-18.10.1991	40	3-12	5-6	6	2	50
Paatsjoki	No.1	12.11.1993-24.10.1994	380	4-12	12-27	24**	0	15
	No.2	12.11.1993-24.10.1994	375	4-12	12-27	0	16**	12

*Ice floe broke the parts at the surface and from April 9, 1991 to April 25, 1991 the station did not collect data. After repairs station consisted of eight up-looking transducers covering 15 % of the rivers cross-section.

**One unit located at the surface and one at the bottom were shut down during the whole survey, because of the reflections causing noisiness in the other sonars.

To detect the possible noise of the data, visual echogram monitoring for each sonar was conducted on at least five days per month, and twice a week for minor instances of monitoring. Noisy data was excluded from the analysis by the monitoring program. The control program filtered out hydroacoustic noise resulting from strong echoes from stable obstacles. The data sometimes showed sporadic noise which may have been due to some non-predictable or occasional factors such as strong wind, rain or snowfall. These echoes clearly differed from fish echoes, and they were filtered from the data.

Each sonar collected daily data for 24 hours, and the data was stored on a hard disk. The daily numbers of upstream and downstream migrants in each size group were stored on disk. Only single fish targets were observed. Only few multiple (less than 3 % of targets) echoes were observed in both rivers. The direction of migration for multiple echoes could not be calculated, and they were left out of the calculations.The data from the sonars was expanded to cover the entire cross-section of the river.

Exploratory fishing

Gill net, fyke net and purse seine were used in the exploratory fishing in both rivers to identify the fish species which migrated through the acoustic beams. The fish caught were identified and their total length was measured. In the Oulujoki River the fishing was carried out in the area first above Station No.1, second in the area between the two stations and third in the area about one kilometer downstream from Station No.2. Monofilament gill nets with variable mesh sizes (30 x 1.8 m, 12-75 mm stretched mesh) were used. The nets were lifted daily and there were altogether 2448 net-days during the study. The trap nets were also lifted daily, the total effort being 418 fyke-days. Altogether 482 hauls were carried out with purse seine during the study.

In the River Paatsjoki the fishing took place above the sonar station only because no fishing was allowed downstream from the sonar stations due to vicinity of the Russian border. Gill nets with stretched mesh sizes of 12, 15, 20, 25, 35, 45, 60 and 75 mm and lengths of both 10 m and 30 were used. The height of the net panel was 1.8 m. Also some 3.0 m high panels were used. Altogether 5109 net-days were fished. Altogether 304 hauls were carried out with purse seine nets during the study. Fishing with fyke-nets was rather difficult in the River Paatsjoki conditions and only 114 fyke-net days were fished.

Hydroacoustic results were converted to fish species according to the exploratory fishing. The conversion was done on a monthly basis in the River Oulujoki. In the River Paatsjoki the conversion was based on a two-month catch. Six different length groups were used in both rivers: 7-11 cm, 11-15 cm, 15-21 cm, 21-30 cm, 30-42 cm, and over 42 cm fish.

The following conversion equation was used:

$$Aij = (Cij/(TOT(Ci)j))*(Xjx) \qquad (4)$$

where Aij = the total number of i-species and its j-size group fish migrating upstream or downstream. Cij = the number of i-species and its j-size group fish in the catch. TOT(Ci)j = the number of j-size group fish of all the species in the catch and Xjx = the number of j-size group fish during a period of one month (the River Oulujoki) or two months (the River Paatsjoki) migrating either downstream or upstream according to hydroacoustic sampling.

Fish marking

In order to estimate the number of stocked and natural brown trout, a fluorescent pigment marking (Jackson 1959, Strange and Kennedy, 1982) was used in Lake Inarijärvi in a mass-marking of stocked brown trout smolts. Altogether 39300 were marked (see also Table 4). The marked fish were kept at a hatchery for nine months, after which the marks were checked and the fish were released to Lake Inarijärvi. All the captured brown trout in the River Paatsjoki were checked for possible fluorescent pigment marks under a UV-light.

RESULTS

The number of fish detected in Station No.1 in the River Oulujoki was 438334 fish and at Station No.2 it was 265357 fish. The majority of the fish at Station No.1, (84.5 %), and No.2, (91.7 %), were migrating downstream.

In the River Paatsjoki the fish numbers detected by the sonar were smaller than in the River Oulujoki. At Station No.1 152522 fish were estimated to have passed the station, while the number at Station No.2 was 128438 fish. The direction of the migration in the River Paatsjoki was more evenly divided between the upstream and downstream movement from the division detected in the River Oulujoki. At Station No.1 70106 (46 %) fish were discovererd to be swimming downstream, whereas the corresponding figure at Station No.2 was 69300 (54 %).

In addition to the larger number of fish migrating downstream in the River Oulujoki, also the variations in the daily numbers of the fish observed at the sonar stations were distinguishably greater in the River Oulujoki in comparison with the numbers obtained from the River Paatsjoki (Fig. 3). However, a somewhat similar timing in the migration rate was found, so that a period of a relatively low migration rate of fish from November to March was found at both sites. In the spring a period indicating a slightly higher migration rate was found in the River Oulujoki in March and in April-May in the River Paatsjoki with a more northern location. After a period of a relatively low migration rate the numbers of the fish observed by the sonar stations began to increase and a relatively high migration rate was reached in October in both rivers. The cycle of the migration rate between the size groups did not differ noticeably in the River Oulujoki, the visible difference being the small amount of large fish (>35 cm long) migrating compared to the fish in the smaller size groups (Fig. 4A). The increase in the migration rate occurred simultaneously in all the size groups. In the River Paatsjoki the cycle of the migration rate varied between the different size groups (Fig. 4B).

The total number of the fish caught by the exploratory fishing was 33461 fish in the River Oulujoki and 4533 fish in the River Paatsjoki. On the basis of the species and size distribution in the catches of the sample fishing (see equation 3) the results of the hydroacoustic sampling were converted into species composition (Table 2 and 3). In the River Oulujoki the most abundant species migrating were perch, whitefish, roach, ruffe, burbot, brown trout, vendace, and dace. In the River Paatsjoki whitefish was the most abundant species, which constituted more than a half of the estimated number of the migrating fishes. At both stations in the River Paatsjoki both the downstream and upstream migrating fish were observed quite evenly.

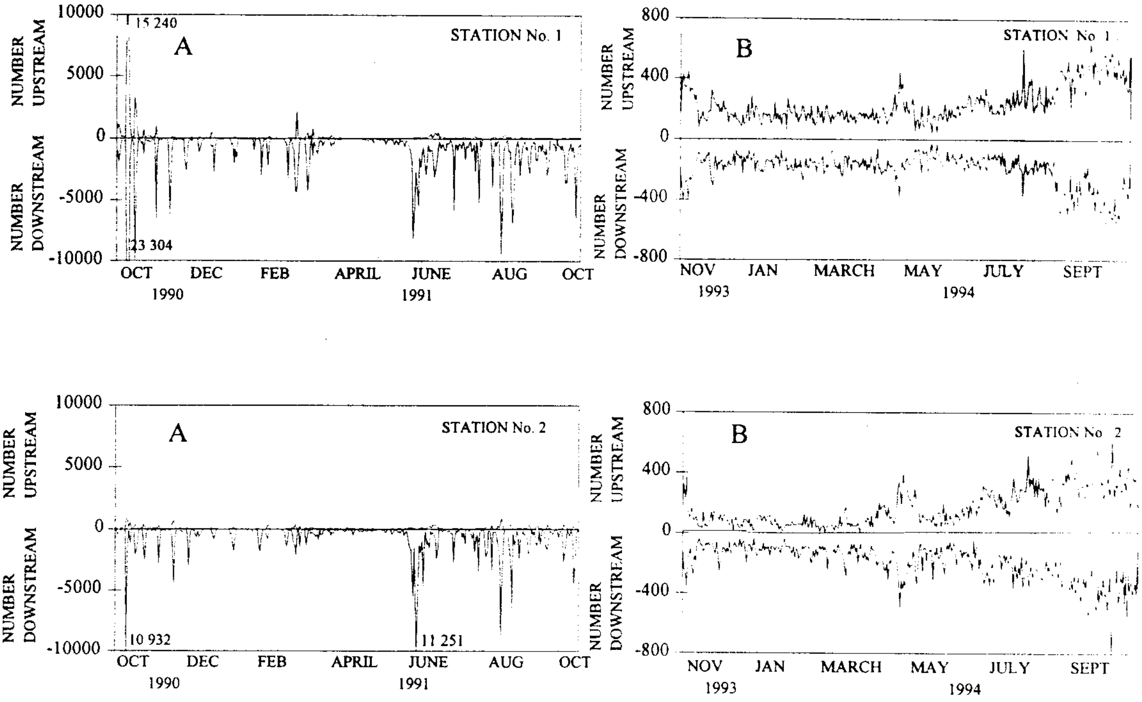


Figure 3. The daily numbers of the fish migrating in the River Oulujoki (A) and the daily numbers of the fish migrating in the River Paatsjoki (B). Notice different scales.

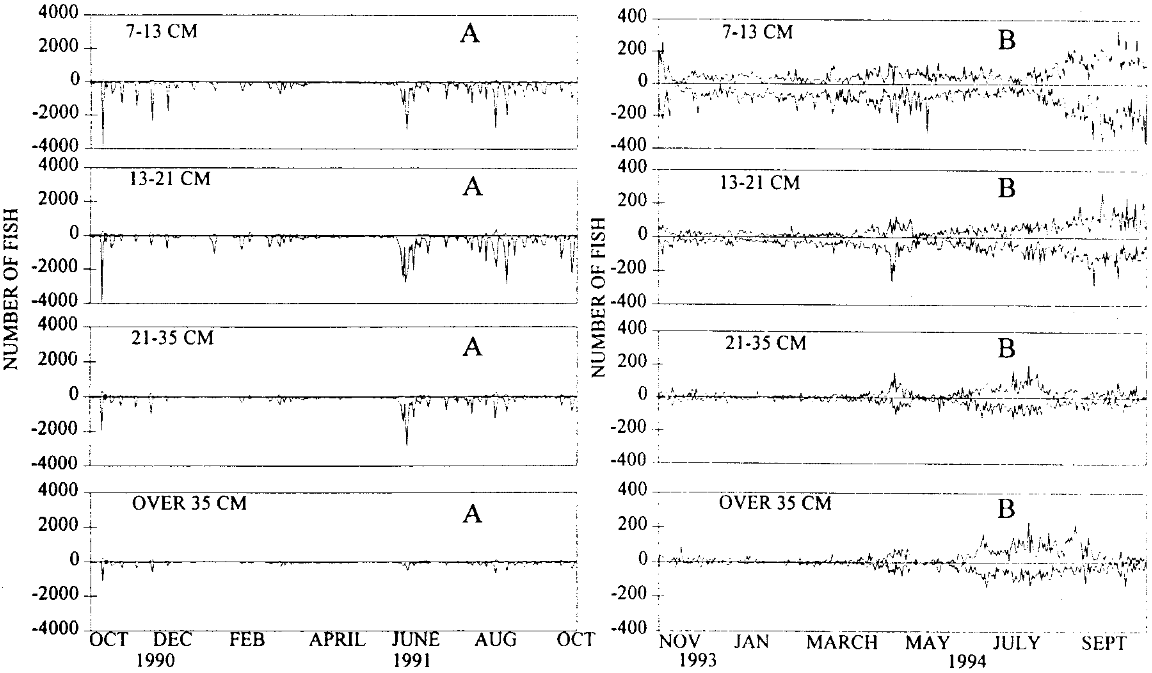


Figure 4. The daily numbers of the fish migrating in the River Oulujoki divided into four size groups at Sonar Station No.2 (A) and in the River Paatsjoki at Sonar Station No.2 (B). Notice different scales.

Table 2. The estimated numbers of the eight most abundant species migrating downstream or upstream at the two sonar stations in the River Oulujoki.

	Station 1.			**Station 2.**		
Species	Upstream	Downstream	Difference	Upstream	Downstream	Difference
Perch	8743	95357	-86614	2453	45234	-42781
Whitefish	28389	129637	-101248	3355	30653	-27298
Roach	5715	28259	-22544	9374	113418	-104044
Ruffe	4921	61213	-56292	630	18009	-17379
Vendace	6328	12929	-6601	69	871	-802
Burbot	6951	13583	-6632	3540	18469	-14929
Brown trout	2478	6457	-3979	240	3840	-3600
Dace	418	10446	-10028	59	947	-888

Table 3. The estimated numbers of the eight most abundant species migrating downstream or upstream at the two sonar stations in the River Paatsjoki.

	Station 1.			**Station 2.**		
Species	Upstream	Downstream	Difference	Upstream	Downstream	Difference
Whitefish	42933	37106	5827	32532	39665	-7133
Burbot	15142	14774	367	9189	11284	-2094
Grayling	14086	10272	3814	9915	9809	106
Perch	3540	3009	531	2306	3079	-773
Brown trout	3022	2055	968	2802	2426	376
Pike	2481	1841	639	2355	1964	391
Vendace	499	620	-121	767	678	88
Arctic char	423	424	-2	391	355	57

In the River Oulujoki the species composition at the upper (Station 1) and lower (Station 2) sonar stations was quite different (Fig. 5). Different species showed clear migrating periods. The majority of the brown trout migration in the River Oulujoki occurred in June-July. However, according to the results of the exploratory fishing, the peak in the brown trout migration took place during a very short period from the end of June to the beginning of July, when the brown trout catch of the test fishing was high. Because the numbers of the fish migrating were calculated on the basis of the monthly figures of the migrating fish, a migration peak of such sorts can lead to an underestimation of the migrating brown trout.

Whitefish was the most abundant species throughout the year in both stations in the River Paatsjoki, the estimated period of abundance being September-October (Fig. 6). Migrating periods of different species were also found in the River Paatsjoki.

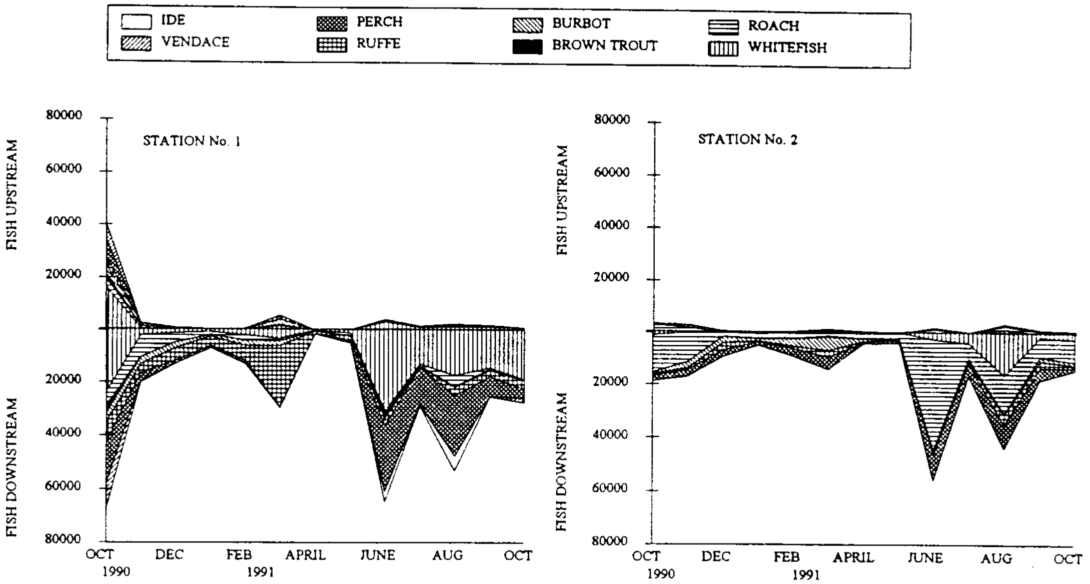


Figure 5. The results of the hydroacoustic sampling converted into various fish species on the basis of the sample fishing in the River Oulujoki. The eight most abundant species are shown.

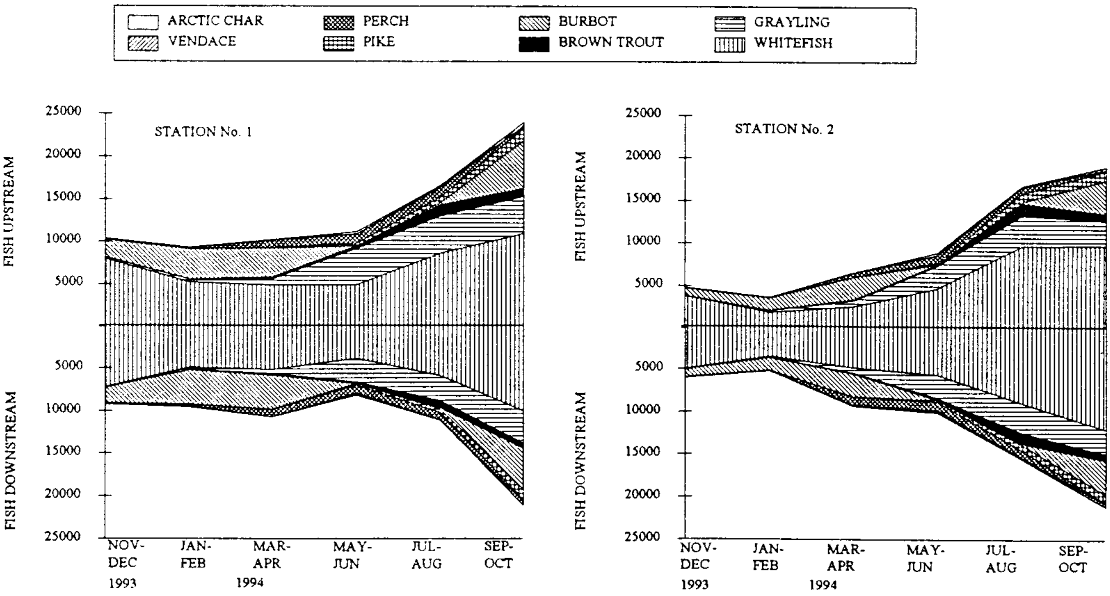


Figure 6. The results of the hydroacoustic sampling converted into various fish species on the basis of the sample fishing in the River Paatsjoki. The eight most abundant species are shown.

In the River Oulujoki 2145 whitefish were randomly sampled in order to measure the proportion of the stocked densely rakered whitefish from the naturally reproducing whitefish forms in the lake. According to the gillraker frequency, 41.2 % of the whitefish obtained migrating from the River Oulujoki were densely-rakered whitefish and thus they originated from stockings. In Lake Oulujärvi the brown trout catch is based on the stocking. On the basis of the age and size distribution the majority of the migrating brown trout in the River Oulujoki, 93 % (altogether

744 brown trout were sampled), were from the stocking made during the study in 1991. This would mean that 3571-6005 of the brown trout stocked in 1991 were migrating downstream. The number of the ascending brown trout was 6.1-10.3 % of the annual number stocked in 1991 (58 066 fish were stocked in 1991)

In the River Paatsjoki altogether 176 brown trout were caught during the study and out of them 145 were 2-4 years old, possibly originating from the stockings (2-4 year-old brown trout were stocked in 1994). Out of these 145 brown trout 25 were marked with fluorescent pigment (Table 4). The assumption was that the relation between marked and unmarked stocked fish was the same in the River Paatsjoki as it was in Lake Inarijärvi (34005 brown trout marked/116120 brown trout stocked). According to this relation more than a half (85 fish, 58.9 %) of the 2-4 year old brown trout detected in the River Paatsjoki were from stockings. On the basis of these figures the estimated numbers of the brown trout stocked in Lake Inarijärvi in 1994 and observed in the River Paatsjoki were 986-1451, suggesting that the total amount of the stocked brown trout in the River Paatsjoki was very small in respect to the total amount stocked in 1994.

Table 4. Results of brown trout groups marked with fluorescent pigment, stocked to Lake Inarijärvi and catched from River Paatsjoki. Tag retention was checked after nine months in a hatchery.

Tag-group	Age	Date of stocking	Number marked	Tag retention %	Marked fish stocked	Mean size of fish mm	Tagged fish catched
3	4	June 9-15,1994	10900	77,4	8437	266	1
4	4	June 24-29, 1994	14570	94,8	13812	264	20
5	3	June 20-22, 1994	13830	85	11756	238	4
TOTAL			39300		34004		25

DISCUSSION

Two largely different migration rates and also clear differences in the direction of travel between the lakes studied were found. The migration from Lake Oulujärvi to the River Oulujoki was larger and more varying when compared to the movement from Lake Inarijärvi to the River Paatsjoki. Also the direction of travel was clearly downstream migration in the River Oulujoki, whereas in the River Paatsjoki the direction of travel was almost equally both upstream and downstream. The direction of travel of each fish was measured according to the assumption that fish swim parallel to the bottom or the surface. This was obviously true in the majority of the cases, and the difference between the two rivers monitored by the same method was clear.

The possible reasons for the differences in the magnitude and direction of travel between the lakes can be attributed, besides the possible effect of the difference in the species composition, to the dissimilarities in the nature of the lakes. Lake Oulujärvi is the central lake in the Oulujoki water system. Two large water courses flow into the lake and drain via the River Oulujoki into the Bothnian Bay, Lake Oulujärvi being a typical water course lake. Although many rivers flow into Lake Inarijärvi, none of them are large ones and Lake Inarijärvi is not as typically a part of a water course as Lake Oulujärvi. Another difference between the lakes studied are the dissimilar regulation practices carried out in the lakes. Due to heavier regulation the variations in the water level of Lake Oulujärvi, and especially the variations in the flow of the River Oulujoki, are larger when compared to those in Lake Inarijärvi and the River Paatsjoki. Also the shallowness of Lake Oulujärvi increases the effect of the water level regulation on the physical environment of the lake in comparison with the deeper Lake Inarijärvi. All this can be the contribute to the larger and more varying amount of fish migrating downstream from Lake Oulujärvi.

The more southern location of Lake Oulujärvi is seen in the species composition of the River Oulujoki in the larger numbers of cyprinid and percid fishes when they were compared with those in the River Paatsjoki. In both rivers the spawning period was apparent in the increased movement involving the majority of the species. However, when the different size groups in the River Oulujoki were observed, an increase in the migration rate was seen to take place simultaneously in all the size groups. In the River Paatsjoki a clear migration peak in the spring and late summer was seen in the larger fish. The free flowing section of the River Paatsjoki serves as a feeding site and also a spawning site for the naturally reproducing fish, such as grayling, whitefish and brown trout. The results suggest that in the River Paatsjoki the up- and downstream migration results partly from the spawning fish actively migrating to the river. Also feeding migrations are possible in the River Paatsjoki. In the River Oulujoki, where the first power station is situated in the outlet of the river, the migration can be more passive and a larger proportion of the migrating fish were small. However, feeding migrations are possible also in the upper part of the River Oulujoki, above the first power station. A clear difference in numbers and species of migrating fish was found between the two sonar stations in the River Oulujoki. This difference can partly be attributed to fish returning to Lake Oulujärvi before descending below the first power station into the river, fishing and also turbine loss. Local river reservoir fishes (eg. roach) were detected in greater numbers at the second sonar station.

The total catch in Lake Oulujärvi has varied between 350-700 tonnes with whitefish, vendace, pike, roach, burbot, and perch forming more than 90 % of the total catch. According to the data the total number of the whitefish descending from Lake Oulujärvi exceded 100000 fish at Station No.1 and some 30000 fish at Station No.2, about 40 % of the fish originating from the stockings of the densely-rakered whitefish. However, when compared to the size of the natural and stocked whitefish stocks in Lake Oulujärvi (which is estimated to 1500000 two-year old and older fish in 1989, Salojärvi (1991)) the number of the ascending fish was not large. In addition to this, a substantial part of the descending whitefish were small fish, 7-15 cm long. Because the relationship between the stock size and growth of the whitefish has been found to exist in Lake Oulujärvi, the descending of the whitefish can be compensated by the better growth of the whitefish. Nevertheless, the effect of the downstream migration on the whitefish catch is fairly small. The ascending trout, 6.1-10.3 % of the total amount of the annually stocked brown trout, does have some significance for the brown trout yield obtained from the lake. However, the annual brown trout yield in Lake Oulujärvi was in 1990, for example, 9600 kg, and the loss in the yield due to the descending of brown trout is not crucial. The majority of the other migratory fish (mainly vendace, perch, roach, ruffe, burbot, and dace) were small, 7-13 cm fish. Although the amount of these migrating fish was fairly large their influence to the total yield of the lake was small and migrating fish do not present any threat for the natural stocks in the lake.

In Lake Inarijärvi the annual total catch in 1987-1993 has varied between 160-560 tonnes. Vendace, which was introduced into the lake in the 1980s, has been the strongest factor behind the variation. The vendace catch was at its greatest in 1989, but since then it has rapidly decreased. Only few vendace were found to migrate to River Paatsjoki in this study. The other commercially important coregonid fish, whitefish, was migrating almost evenly both up- and downstream in the River Paatsjoki and the importance of the migrations to the Lake Inarijärvi fishery was very small. In fact, the significance of the free flowing section of the River Paatsjoki can be positive for whitefish, and also for grayling and brown trout, for it serves as a spawning and feeding area. Also brown trout were found migrating evenly both up- and downstream. Even if all the stocked brown trout detected in River Paatsjoki were assumed to be migrating downstream, the amount found in the river was, with respect to the annual stocking amount, so small that the downstream migration had no or very little effect on the brown trout yield in Lake Inarijärvi. Also the amount of burbot, grayling, perch and, pike were relatively small and the direction of travel both up- and downstream indicated that the importance of the migration for the yield is neglible.

Hydroacoustic employing stationary transducers together with intensive exploratory fishing and fish marking could, with a reasonable error rate, give answers to the questions conserning the amount, size, and species of migrating fish and also about the timing of the migration. The fact that the same relation between fish length and target strength was employed, does, due to the different backscattering properties of the different species, create an error rate of some extent in the species identification. Our assumption, however, is that the size grouping used in the study was accurate enough for the species identification relevant for the purposes of this study. Two largely different migration patterns were found between the two lakes studied. The importance of the downstream migration ranged from moderate in Lake Oulujärvi to almost zero in Lake Inarijärvi. In addition, on the basis of the results, the significance of the migrations in either lake was not large enough to make barriers preventing migration recommendable.

ACKNOWLEDGEMENTS

This work was sponsored by the Finnish National Board of the Waters and the Environment. The work of Kai Kaatra and the late Kalervo Salojärvi started this project. Thanks are extended to the staff at the Echo Research Co.: Sergei Tigunov, Konstantin Tserkov, Andrei Kapanov, and Sergei Pushkin for the development of the hydroacoustic system. Mikko Ahonen and Erja Konttinen assisted in the exploratory fishing and Tuomas Friman in the fish marking. The help of Ari Kauttu was essential in the field work. Eero Aro, Juha Jurvelius and Raimo Parmanne gave valuable comments on the manuscript.

REFERENCES

Bodholt, H. 1990. Basic theory of underwater acoustics. Simrad Subsea Technical Paper. Horten, Norway. p. 31.

Craig, R. E. and Forbes, S.T. 1969. Design of a sonar for fish counting.*Fiskeridirektorateta Skrifter (Havundersokelser).* 15 pp. 210-219.

Echo Research Co. 1990. Introduction to RSS320 - Scientific Sonar System for fish stock assesssment in rivers. Echo Research Co., Helsinki, Finland. p. 13.

Foote, K. G. 1982. Optimizing copper spheres for precision calibration of hydroacoustic equipment.*Journal of the Acoustical Society of America.* 71 pp. 612-616.

Forbes, S.T. and Nakken, O. 1972. Manual of methods for fisheries resource survey and appraisal. Part 2. The use of acoustic instruments for fish detection and abundance estimation.*FAO Manuals in Fischeries Science.* No. 5 p. 138.

Heikinheimo-Shmid, O. and Huusko, A. 1988. Management of coregonids in the heavily modified Lake Kemijärvi, Northern Finland. *Finnish Fisheries Research.* 9 pp. 435-445.

Henk Co. 1993. Hydro-Acoustic System HAS version 4.10. Software System Operation Manual. Henk Co., Helsinki, Finland. p. 27.

Jackson, C.F. 1959. A technique for mass-marking fish by means of compressed air. New Hapshire Fish and Game Department, *Technical circular* No. 17. p. 8.

Johannesson, K.A. and Mitson, R.B. 1983. Fisheries acoustics, A practical manual for aquatic biomass estimation. FAO, Rome. *FAO Fisheries Technical Paper* 240. p. 249.

Johnston, S.V. 1985. Experimental testing of dual-beam acoustic fish size classifier for Alaska Department of Fish and Game. Report to Alaska Department of Fish and Game. Biosonics Inc., Seattle, Washington.

Johnston, S.V. and Hopelain, J.S. 1987. The application of dual-beam target tracking and Doppler shifted echo processing to assess uppstream salmonid migration in the Klamath River, California. *International Symposium on Fisheries Acoustics*, June 22-26, 1987, Seattle, Washington. p. 32.

Johnston, S.V., Ransom, B.H. and Bohr, J.R. 1993. Comparison of hydroacoustic and net catch estimates of fish entrainment at Tower and Kleber dams, Black River, Michigan. *Waterpower '93, Proceedings of the International Conference on Hydropower.* August 10-13, 1993, Nashville, Tennesee. pp. 308-317.

Jurvelius, J. 1991. Distribution and density of pelagic fish stocs, especially vendace (*Coregonus albula* (L.)), monitored by hydroacoustics in shallow and deep southern boreal lakes. *Finnish Fisheries Research.* 12: pp. 45-63.

Lindem, T. 1983. Successes with conventional in situ determinations of fish target strength. In: Nakken, O. & Venema, S.C. (eds.), Symposium on fisheries acoustics. Selected papers of the ICES/FAO Symposium on fisheries acoustics. Bergen, Norway, 21-24 June 1982. Rome. *FAO Fisheries Report.* 300 pp. 104-111.

Salojärvi, K., 1991. Recruitment mechanisms of the vendace (*Coregonus albula* (L.)) in Lake Oulujärvi, northern Finland.*Aqua Fennica.* 21 pp. 163-173.

Stables, T. B. and Thomas, G. L. 1992. Acoustic measurement of trout distributions in Spada Lake, Washington, using stationary transducers. *Journal of Fish Biology.* 40 pp. 191-203.

Strange, C.D. and Kennedy, G.J.A. 1982. Evaluation of fluorescent pigment marking of brown trout(*Salmo trutta* L.) and atlantic salmon (*Salmo salar* L.). *Fisheries Management.* 13 pp. 89.

Thorne, R.E. 1983. Hydroacoustics. In Nielsen, L. and Johnson, D. (eds.) Fisheries Techniques. Bethesda, American Society. pp. 239-260.

Index of Subjects

Index des sujets

A

- Aboriginal fishing, B915
- Acoustic Current Profiler, B323
- Acoustics, A215
- Aeration, A737
- Agriculture, A811, A167
  - Alabama River, B107
- Alberta, A559
- Algae, A13
  - Control, A725
- Alleghenis River, A665
- Almonda River, B767
- Alpine region, B175
- Alps range, A25
- American shad, B953
- Ammonium, A177
- AMOEBA approach, A619
- Andrle, B263
- Arctic bulhead, B45
- Argentina, A3
- Armorican Massif, A13
- Artificial Flood, A685
- Aspe River, B891
- Assessment
  - Habitat changes, A281
  - Habitat restoration, B607
  - Impact, A59, B215, B251, B275
  - Water quality, A779
- Atlantic salmon, A471, B15, B365, B381, B585, B797, B873, B891, B941
  - Habitat, A533
  - Juvenile, B67, B477
  - Migration, B81
  - Parr, A471
  - Parr and fry habitat, A305
  - Spawning habitat, B3
  - Spawning substrate, A305
  - Suitability index, B33
  - Swimming performance, A227
  - Unstable habitat, A305
- Atlantic tomcod, A379
- Aude River, B283
- Australia, A203, A215, A571
- Austria, A191, A685, A797, B55, B487
- Azov sea, A59

B

- Bacterioplankton (total count), A47
- Bank
  - Design, B741
  - Erosion, A305, B753, B775
  - Fauna, A743
  - Protection, B683, B729
  - Seepage, A121
  - Stabilization, B717
- Bankfull discharge, A71
- Bar screen, B929
- *Barbus barbus*
  - Spawning habitat, A367
- Barents sea, A145
- Bathymetry, B299
- Beach, A113
  - Habitat, A343
  - Profile, A525
  - Replenishing, B783
- Bed-load transport, A389
- Bed material, B861
- Behaviour, B341
- Behavioural barriers, B929
- Belgium, A83, A367, A835
- Belice River, A779
- Benthic
  - Habitat, A203, B163
- Benthos
  - Biomass, A145
- Bioclimatology, A797
- Biodiversity, A37, A501, A665, A865, B27
- Bioenergetic, B487
  - Model, B447, B465, B519
- Bioengineering, A501, B683, B717, B729, B741, B767
- Biomass, A239
  - Benthos, A145
  - Brook trout, A485
  - Eelgrass, A99
  - Hydrobionts, A145
  - Macrophytes, A191
  - Phytoplankton, A47
  - Trout, B507
- Biotic
  - Index, A665, A779
  - Response, A355
- Birds, A851
- Birse River, B683
- Bivalve
  - Habitat model, A343
- Blackwater River, B595
- Blueback herring, B953
- Boat waves, B729
- BOD5, A177
- Bonaventure River, B585
- Borland, B941
- Bottom baffle, B843
- Boulder structure, B619
- Breakwater, B783
- Bregenzerach River, A685
- British Columbia, A261
- Brook trout
  - Biomass, A485
- Brown trout, B45, B175, B283, B353, B395, B465, B619, B967
  - Hydropeaking, A673
  - Juvenile, A673
  - Population dynamics, B531
- Buffalo River, A281
- Burst swimming performance, A227

C

- Cache la Poudre River, B149
- Camargue, A167
- Canada, A99, A113, A133, A155, A227, A251, A261, A305, A379, A409, A435, A447, A459, A485, A501, A533, A559, A643, A753, A759, A769, A829, A851, A865, B15, B33, B67, B81, B187, B365, B381, B477, B555, B585, B655, B669, B705, B717, B775, B783, B797
- Cantor set, B263
- Capture success, B487
- Carrying capacity, A59, A619
- CASIMIR
  - Model, B163
- Catamaran Brook, A409, B381, B477
- Cellular motion, A293, B843
- CEQUEAU
  - Model, A409
- Channel, A797
  - Adjustment, A71
  - Change, A355
  - Distributary, A121
  - Geometry, A583
  - Incision, A71
  - Morphology, A281, A379
  - Substrate, B227
  - Vegetation, A389
- Chondrostoma nasus, B55, B487
- Coastal
  - Habitat, A99
  - Monitoring, A513
  - Morphology, A525
  - Outfall diffuser, A759
  - Processes, A145
  - Realignment, A513
  - Zone, A759
- Cobble-Bed

  Stream, A239
Colonization, A797
  Eel, A89
Comal River, B251
Community
  Fish, A871, B107, B967
  Macrophyte, A13
  Restoration, B543
Comparative evaluation, B395
Comparison, B307
  Hydrological models, A409
  Suitability curves, B519
Computer
  Geometry, B263
Conservation, A665, A865, B783
  Flow, B239
Consumption, B487
Continuous under threshold Control system, A759
Coregonus lavaretus, B873
Cottus gobio, B341
Cover, B175
  Ice, A435
Creation
  Habitat, A513, B655
Curtain
  Algae control, A725
Cyprinids, B95, B487
  Nursery, A367
  Spawning habitat, A367

D

Dam, A571, A737, B655
  Estuary, B855
  Impact, A3, A83, A331, B133
Danube River, B55, B487
Deadman River, A261
Dee River, B3
Deforestation, A25
Degradation, A811
  River-bed, A389
Demersal fish, B855
Denitrification, A177
Depth, B55, B631
  Averaged flow analysis, A293
  Width ratio, A583
Design, A743
Diatom, A239
Diel shifts, B341
Diffusers, A737
Discharge, A25, A99, A133, B15, B119
  Depression, A459
  Duration, A421
  Fluctuation, A389, A711, B283
  Habitat, A281, A409
  Hydropeaking, A699
  Low, A595
  Reference, B283
  Regulated, B543
  Station, A421
Dissolved Oxygen, A737
Distinct element method, A343
Disturbance, A25
  Reservoir, A13
Diversion, B163
Diversity, B107, B643
Drainage, A167
Dredging, A367, B767
Drift, B283, B465
Drift feeding, B487
Drought, A121

E

Echosounding, B829
Ecohydrology, A409, A421
Ecological
  Engineering, A711, A835
  Enhancement, A595, A619, A711
  Framework, A619
  Impact, A595
  Indicators, A609
  Information system, A653
  Management, A643
  Mapping, A643
  Niche, B27
  Restoration, A609, B705
  Status, B323
Ecology, A59
  Landscape, A25
Eel, B941
  Colonization Dynamics, A89
Eelgrass
  Beds, A99
  Biomass, A99
Efficiency
  Fishway, B891
Egg
  Salmo salar, B477
Embankments, B717
Emigration, B477
Endangered species, B251
Energy balance, B487
Enhancement
  Dissolved Oxygen, A737
  Ecological, A711
  Habitat, A501, B15, B619
  Invertebrate community diversity, B643
  Retention quality of pools, B585
Entrainment, A725
Environmental
  Emergency, A753
  Factors, B27
  Flow, A571
  Impact, A525, A811, A829
  Impact assessment, A619, A753, B655
  Instream flow, A547
  Planning, A653
  Regime, A711
Enz River, B903
Epilimnion, A725
Erne River, B941
Ernegger, AT., A797
Erosion, B579, B669, B717, B783
  Bank, A305, B753, B775
  By waves, A501
  Control, B683
  Remedial measures, A113
  Shore, A113, A501, B705
Estonia, A47, A177
Estuary, A155
  Ice-covered, A435
  Mixing power, A133
  Stratification, A133
Eutrophication, A725
Evaluation, B607
  Management of banks, B741
EVHA procedure, B307, B323
Expertise integration, A829
Exploitation, A485, A811
Exploits River, B15
Exploratory fishing, B967

F

Fish
  Swimming, B915
Feasability study, B655
Feeding, B283
  Drift, B465
Fencing, B669
Fertilizers
  Impact, A167
Field
  Measurement, A191, B631
  Sampling, B307
  Validation, B519
Finland, B45, B619, B873, B915, B967

- Fish, A559, A685, A797, A865, B27, B133, B555
  - Abundance, B903
  - Acoustic habitat, A215
  - Classification, B107
  - Cold habitat, A435
  - Community, A871, B107, B323, B341, B543, B903, B967
  - Distribution, B365
  - Entrainment, B929
  - Gas Bubble Disease, A3
  - Habitat, A261, B323, B631
  - Hearing, A215
  - Importance of hydrology, A409
  - Larval, B543
  - Life stage, B407
  - Long-term survey, A421
  - Marking, B967
  - Migration, A215, A379, B829, B855, B967
  - Passage, B555, B873, B915, B929, B953
  - Physical barriers, B929
  - Population, A533
  - Protection, B929
  - Rheotaxis, B915
  - Standing crop, B507
  - Swimming, B487
  - Young of the year, B55
- Fishlock
  - Cycle, B885
- Fishway, A293, B555, B797, B829, B861, B903, B915
  - Alaska steeppass, B843, B953
  - Borland fishpass, B885
  - Combined, B873
  - Denil, B843, B873, B953
  - Guide-flow, B855
  - Hydraulics, B843
  - Lock-type, B855
  - Naturally-shaped, B855
  - Surface, B891
- Flexibility
  - Vegetation, A251
- Flood, A395, B175
  - Artificial, A685
  - Channel geometry, A583
  - Current, B729
  - Disaster, A583
  - Frequency, A685
  - Peak propagation, A71
  - Protection level, A583
- Flooding, B729, B753
- Floodplain, A71, A653
- Floodplain, A25
- Flow, B915
  - Attraction, B829
  - Basic, A547
  - Boulders, B861
  - Ecological reference, B133
  - Gradual variation, B239
  - Guaranteed, B419
  - Influence of macrophytes, B187
  - Influence of vegetation, A293, A395
  - Limiting conditions, A421
  - Maintenance, A871
  - Micro-structure, A293
  - Nonuniform, A583
  - Open channel, B395
  - Pattern, A191
  - Recommendations, A559
  - Regulated, A13, A533
  - Requirement, A871, B275
  - Resistance, A191, B861
  - River, A447
  - River-bed, A203
  - Under-ice, A447, A459
  - Velocity, A379
- Fluvial
  - Geomorphology, A121, B3
  - Hydraulics, A293, A395
  - Species, B543
- Food resource, A37
- Fractals, B263
- Framework, A643, B215
- France, A13, A25, A71, A89, A167, A421, A631, A653, A673, A699, B95, B133, B283, B307, B323, B341, B419, B531, B829, B891
- French Guiana, A37
- Fresh water, A89
- Friction factor, B861
  - Vegetation, A251
- Froude number, B861

G

- Gabion, B767
- Gap Analysis, A665
- Garonne River, A71, B323
- Gas Bubble Disease (GBD), A3
- Gas Supersaturation, A3
- Gatineau River, B655
- Generating protocol, B941
- Geomorphology, A281, A533
  - Changes, A331, A355
  - Fluvial, A121, A261, A355, B3
  - *Salmo salar*, A305
  - Sedimentology, A113
- Germany, B163, B753, B903
- GIS, A595, A653, A665, B45
- Gjengadalselva River, B353
- *Gobio gobio*, B55
- GPS, B251, B299
- Graelva River, B579
- Gravel-bed river, A305
- Gravel-bed stream
  - Roughness, A293
- Gravel bars
  - Spawning areas, A367
- Grayling, B175
- Great Whale River, A447
- Grensmaas, A711
- Groin, B783
- Growth
  - Brown trout, B465
  - Invertebrates, A699
  - Periphyton, A239
- Groynes, B595
- Gunnison River, A319

H

- Habitat, A355, A485, A851, A865, B215, B263, B555, B631
  - Acoustics, A215
  - Analysis, A871, B275
  - Assessment, A559, B67
  - Atlantic salmon, A305, A533
  - Availability, B407, B619
  - Beach, A343
  - Benthic, A203, B163
  - Changes, A261
  - Classification, B107
  - Coastal, A99
  - Creation, A513, B655
  - Day and Night, B33
  - Edge, A471
  - Enhancement, A501, B15, B619
  - Environmental flow, A571
  - Evaluation, B619
  - Evaluation Procedure (HEP), A59, A619
  - Features, B263
  - Fish, A261, A409, A435, B323
  - Functional, B341
  - Generalists, B543
  - Hydraulic, A239, A281
  - Macrophytes, A191
  - Management, B585
  - Mapping, B203
  - Model, A743, B631

  Modeling, A59, A319, A533, A559, A871, B67, B95, B107, B133, B187, B215, B251, B275, B307, B365, B381, B395, B419, B447, B519, B531, B585, B607, B643
  Physical, A685, B607
  Prediction, B227
  Quality, B643
  Restoration, B579, B607, B643, B767
  Riffle-run, A239
  Riparian, A743
  River, B175, B507
  River channel, B299
  Selection, A471, B67
  Shore, A113
  Spawning, A367, B3, B133, B655
  Stream, A665
  Suitability, A59, A595, A619, B33, B119, B163, B239
  Suitability index, A743, B365, B407, B431, B607
  Usable, B431
  Use, B107, B903
  Variables, B407
HABITAT procedure, B307
Hearing
  Fish, A215
Helophytes, A13
Holly Fork Creek, B643
Hudson Bay, A447
Hydraulic
  Habitat, A239, A281
  Model, B187, B307, B655, B843
  Performance, B595
  Stream ecology, B643
  Works, A367
Hydraulic model
  Test, B855
Hydraulics, A203, A215, A319
  Coastal, A759
  Near-bed, A239
  Open channel, A191
  Spawning habitat, B3
  Statistical, B95
  Variables, B81
Hydroacoustics, B251, B967
Hydrobionts
  Biomass, A145
Hydrodynamic
  Model, A155, A513
  Sound, A215
Hydrodynamics, A215, A525, B187
Hydroelectric
  Dam, A37, B543
  Reservoir, A13, A37
Hydroelectricity, A447
  Impact, A99
Hydrological
  Likelihood events, A421
Hydrology, A177, A355, A485, A533, A797
  Environmental regime, A711
  Importance for habitat, A409
  Natural regime, A711
Hydropeaking
  Disturbance, A673
  Impact, A685
  Influence on Brown trout, A673
  Influence on invertebrates, A699
  Mitigation, A685
  Temperature, A699
Hydrophytes, A13
Hydropower, A699, A737, B655
  Plant, B163, B419, B797
Hydropower plant, A789, A829, B941
  Pumped-storage, B929
  Small scale, B891
Hydroturbine venting, A737

I

Ibex River, A459
Ice, A447
  Atlantic salmon, A471
  Cover, A435
  Formation, A471
IFIM (Instream Flow Incremental Methodology), B203, B239, B251, B275, B307, B407, B419
Impact, B669
  Agriculture practices, A167
  Assessment, A59, A753, A871, B215, B275, B717
  Dam, A3, A83, A331
  Ecological, A595
  Environmental, A525
  Fertilizers, A167
  Human, A71
  Hydraulic works, A83
  Hydroelectricity, A99
  Hydropeaking, A673, A685
  Legislation, A789
  Logging, A485
  On fish population, A167
Impoundment, A37, A533
Index
  Stress, B175
  Suitability, B175
Infrastructure work, A835
Instream Flow, B163, B239, B395, B447, B555
  Needs, A281, A559, A871
Instream Flow Needs, B275, B431, B797
  Macrophytes, B187
  Sedimentation, A319
Integrated data management, A769
Integrated management, A631, B741, B797
Integrated System Analysis, A59
Interdisciplinarity research, A797
International Aquatic Modeling Group (IAMG), A845
International cooperation, A845
Invertebrates, A37, A571, A685, A711, B45, B431
  Community, B643
  Drift, B465
  Growth, A699
  Influence of hydropeaking, A699
  Life history, A699
  Population structure, A699
Ireland, B595, B941
Irrigation, A167
Islands, A851
Isostatic land-rise, A99
Italy, B175, B507

J

Jacques-Cartier River, B797
James Bay, A99, A447
Japan, A293, A343, A389, A395, A583, A725, B27, B239, B631, B729, B843, B855, B861
Juvenile
  Atlantic salmon, B67, B365, B477
  Brown trout, A673

K

K-e model, B843
  Gravel-bed stream, A293
Kemijoki River, B873, B915
Kerlegan Brook, B341
Kislaya Bay, A145
Kocher River, B163
Kruger National Park, A121
Kutinjoki River, B619
Kuusinkijoki River, B45

L

La Grande River, A447
Laboratory modeling, A203
Lake, A47, A485
    Regulation, B967
Lake Constance, B753
Lake Ijsselmeer, A619
Lake Inarijärvi, B967
Lake Oulujärvi, B967
Lake Russell, B929
Lake Saint-François, B187
Lake Saint-Jean, A113, B783
Lake St-Francis, B187
Lake Vortsjärv, A47, A177
Lakeshore
    Restoration, B753
Lampetra fluviatilis, B873
Lamprey, B873
Land-use, A25
Land use, B775
Landscape ecology, A25, A653
Landslides, B579
Lee River, B941
Legislation, A789
Life history
    Invertebrates, A699
Liffey River, B941
Light limitation, A47
Limiting conditions, B133
    Flow, A421
Limnology, A797
Little Whale River, A447
Littoral
    Drift, A113
    Processes, A145
Lock, B829
Logan River, B395, B519
Logging operation, A485
Loire River, A89, A631
Long-term
    Changes, A583
    Investigations, A47
Low stone weirs, B595

M

Maas, A711
MacDonald Fish-Lock, B941
Macrohabitat, B175
    Variable, B239
Macrophytes, A177, B187
    Biomass, A191
    Influence on flow, B187
    Influence on river-bed, A389
    Resistance, A251
    River, A13
    Roughness, B187
    Turbulence, A191
    Velocity, A191
Man-made habitat, B655
Management
    Discharge, B133
    Ecosystem Resources, A769
    Habitat, B585
    Knowledge quality, A829
    Riparian habitat, A743
    River, A389, A395, A547, A571, A789
    Rules, B829
    Water, A59, A595, A711
    Water quality, A753
Mandlebrot, B263
Manning's coefficient, B861
Mapping
    Ecological, A643
    Habitat, B203
Marking, B903
    Fish, B967
Marsh, A851, B783
Mass budget, A177
Mean velocity
    Periphyton, A239
Measurement
    Swimming performance, A227
Mechanical damage, B753
Mesohabitat, A631, B203
    Brown trout, A673
Meuse River, A83, A367, A595, A711
Microhabitat, B15, B107
    Suitability index, B119
    Utilization, B55
Microposition choice
    Salmonids, B353
Migration, B903
    Atlantic salmon, B81
    Downstream, B891
    Fish, A215, A379, B967
    Rate, B967
    Salmo trutta, A83
    Survey, B855
    Upstream, B829
Migratory behaviour, B81
Minimum flow, A547, A737, B95, B163, B419
Miramichi Estuary, A133, A409
Missouri River, A871, B275
Mitigative measures, B655
Model
    Bioenergetic, B447, B465, B519
    CASIMIR, B163
    CEQUEAU, A409
    Deterministic, A409
    Finite element, A155
    Granular material, A343
    Habitat, A743, B619
    Habitat use, B95
    Hydraulic, B187, B307, B655
    Hydrodynamic, A155
    Hydrological, A409
    Multivariate, B507
    N-dimensional, B447
    Numerical, A753, A759, B585, B631, B843
    One-dimensional, B149, B395
    Physical, A203, B15
    Population, B477, B531
    Prediction, B175
    Shore restoration, B753
    Simulation, B163
    Stochastic, A343, A409
    Three-dimensional, B353
    Transport-diffusion, A759
    Two-dimensional, B149, B187, B251, B365, B395, B519
    Validation, B419, B519
Modeling
    Aquatic system, A845
    Changes, A261
    Future trends, B447
    Geomorphological response, A331
    Habitat, A59, A533, A559, A871, B67, B95, B107, B133, B187, B215, B251, B275, B307, B381, B395, B419, B447, B519, B531, B585, B607, B643
    Hydrodynamic, A513
    Riparian vegetation response, A331
Moisie River, B81, B365
Momentum absorbing area, A251
Monitoring, A99, B595, B741, B753
    Coastal realignment, A513
    River plume, A447
Morphological
    Change, A261
Morphology, B683
    Channel, A379
    Coastal, A525
    River, B175

Mosquitos control, A653
Multi-disciplinary assessments, B447
Multiple regression, B507
Multivariate statistics, B283, B507
  Habitat, B95, B107

N

N-dimensional
  Model, B447
Nagara River, B855
Nastapoka River, A447
Naturalness, B227
Near-bed
  Hydraulics, A239
Netherlands, A59, A595, A619, A711, A743, B741
Network, A851
Neuronal network, A409
New-Brunswick, A133, A409
New Zealand, A239, B465
Newfoundland, A533, B15, B33, B67
Niger Delta, A811
Nigeria, A811
Nitrification, A177
Nitrogen, A167
Nocturnal
  Habitat use, B33
Nocturnal activity, A471
North Arbour River, B67
Northern Pike, B133
Norway, A421, A789, B215, B307, B353, B579
Nouvelle River, A305
Nursery, B595
  Gravel bars, A367
Nutrient, A167, A725

O

Object-oriented, B215
Obstacle, B829
Oil, A811
One-dimensional
  Hydraulic model, B149
  Model, B395
Ontario, A251, A501, A769
Ontogeny, B487
Operational oceanography, A759
Order
  River, A631
Oregon, B299
Oriège River, A673, A699
Orne River, A13
Oto River, B27
Ottawa River, B717
Overbank flow, A71

P

Parana River, A3
Partnership, A865
Passage phase, B885
Pattern, B263
Periphyton
  Growth, A239
  Mean velocity, A239
  Riffle-run, A239
  Turbulence, A239
PHABSIM, A595, B149, B203, B275, B381, B407, B431, B519, B607, B619, B643
  Extension, B323
Phosphorus, A177
  Impacts on fish populations, A167
Photointerpretation, A113
Phragmites australis, B753
Physical
  Habitat, A685
  Model, A203
Phytoplankton, A725
  Biomass, A47
Pilot project, A835
PIT tags, B953
Plume
  River, A447
Policy analysis, A619
Pool, B341, B631, B595
  Retention, B585
Pool-rapid, A355
Population
  Dynamics, A485
  Invertebrates, A699
  Model, B477
Population dynamics
  Brown trout, B531
Portugal, B767, B885
Predator-prey interactions, B45
Prediction
  Habitat components, B227
  Model, B175
Preferences, B33, B365, B431, B519, B607
  Properties, A293
Principal component analysis, B27
Production, B477
Program development, B215
Protection
  Bank, B683, B729
  Ecosystem, A759
Proterorhinus marmoratus, B55
Pyrenees, A699

Q

Québec, A99, A113, A155, A305, A379, A435, A447, A485, A643, A753, A759, A829, A851, A865, B81, B187, B365, B381, B585, B655, B669, B705, B717, B775, B783, B797

R

Radio-tracking, A83
Radio-transmission, A227
Radiotelemetry, B891
Rance River, A13
Randomization, B67
Rating curves, B953
RCHARC, A871
Reach
  Scale, B203
Recession, A459
Recommendations
  Flow, A559
Reed deeline, B753
Reeds, B767
Reflective structure, A525
Regionalization, A631
Regulated
  Discharge, B543
  Flow, A13, A533
  River, A547
Regulated river, B873
Regulation, A533
  Lake, B967
  Level, A47
  River, A699
Rehabilitation, A829
Remote sensing, B447
Representativity
  PHABSIM sites, B203
Republic of South Africa, A121, A281, A331, A355
Reservoir, A725, A737
  Hydroelectric, A13, A37
  Releases, A121
Restoration, B555
  Ecological, B705
  Habitat, B579, B607, B643, B767
  Lakeshore, B753
Resuspension
  Sediment, A47

Retention potential, B585
Retrospective assessment, A533
Reverse grading, A343
Rhine River, A619
Rhodamine dispersion, A759
Rhône delta, A167
Rhône River, A25, A653, B683, B829
Rice cultivation, A167
Richard B. Russell Dam, B929
Rideau River, A769
Riffle, B341, B631, B861
    Artificial, B643
Riffle-run
    Habitat, A239
    Periphyton, A239
Riparian
    Vegetation, A121
Riparian vegetation response
    Modeling, A331
River, B299
    Artificial, A797
    Bed, B579
    Bottom, B163
    Braiding, A305
    Channelized, B619
    Flow, A447
    Fluvial-fan, A395
    Habitat, B175, B507
    Ice-covered, A435
    Large, A3, B323
    Macrophytes, A13
    Management, A389, A395, A547, A571, A789
    Meandering, A305
    Morphology, B175, B585
    Mountainous, B175
    Order, A631
    Plume, A447
    Reconstruction, B579
    Regulated, A37, A547
    Regulation, A699
    Restoration, A261
    Slope, A631
    Structure, B227
    Vegetation, A251
    Water quality assessment, A779
    Watershed, A609
River-bed
    Degradation, A389
    Flow, A203
River System Simulator, B215
River Works, B683
Riverbanks, A743, B717, B729, B741
Riverine fish faunas, B543
Rock River, A471
Root, B683
Roughness, B595
    Coefficient, A71
    Gravel-bed stream, A293
    Large-scale, B861
    Macrophytes, B187
    Vegetation, A251
Rubber gate, B855
Running waters, A367
Runoff fluctuation, A71
Russia, A145
Rutilus rutilus, B55

S

Sabie River, A121, A331, A355
Sai River, A293
Saint-Lawrence estuary, B705
Sainte-Anne River, A379, A435
Salinity, A99
Salmo salar, A533, B15, B797, B873, B891, B941
    Geomorphology, A305
    Habitat use, B67
    Migration, B81
Salmo trutta, A673, B341, B531, B873, B891, B967
    Migration, A83
Salmon
    Parr, B353
Salmonids, B407, B419, B507
    Microposition choice, B353
    Migration, A83
    Weirs, B915
Salt water, A155
Saltmarsh, A513
Salvelinus fontinalis, A485
Sampling
    Equal-area, B119
Sand
    Coastal morphology, A525
    Deposition, A395
Saskatchewan River, A559
Sata River, A725
Scale, B45
    Migration, B891
    Reach, B203
    Sector, B203
    Spatial, B67, B263
Scotland, B3
Sea trout, B873, B891
Seasonal shifts, B341
Seawall, A525
Secchi depth, A47
Sector
    Scale, B203
Sediment, B3
    Balance, B705
    Geomorphological response, A331
    Habitat modeling, A319
    Influence of vegetation, A395
    Resuspension, A47
    Sheetflow transport, A343
    Suspended, A395
    Transport, A319, A525, B631
Sedimentary characteristics, A533
Sedimentation
    Instream flow needs, A319
    Park, B669, B705
Sedimentology/, A797
Sediments, A25
Seine River, B133
Sensitivity analysis, B381, B507
Shad, B829
Shannon River, B941
Shear stress, A191
    Bottom, B163
Shoot density, A99
Shore, B753
    Erosion, A113, A501
    Processes, B705
    Protection, A501
Shoreline, A113
    Stabilization, A501
Sicily, A779
Simulations
    Test, B353
Sine wave, B263
Sinnamary River, A37
Slope
    River, A631
Smallmouth bass, B119
Smolt, B891
Socioeconomic issue, A643
Sonar, B299
Spain, A547
Spatial
    Distribution, A471
    Error analysis, B395
    Heterogeneity, B45, B67
    Reference, A653
    Scale, B263
    Variability, B163
Spatial pattern

  Patchiness, B67
Spatial validation, B365
Spawning, B595
  Barbus barbus, A367
  Gravel bars, A367
  Habitat, B655
  Salmon, A305
  Substrate, A305
Spawning habitat
  Atlantic salmon, B3
Species
  Distribution, B903
  Endangered, B251
Species diversity, B543
Spectral Analyses
  Hydrological regime, A711
Sport fishing, A485
Spur dike, B631
St. Lawrence River, B775
St-Charles River, A643
St-Lawrence River, A759, A851, A865, B187, B669
Stabilization, B669, B783
  Bank, B717
  Shoreline, A501
  Slope, B683
Standing crop, B507
Stiffness
  Vegetation, A251
Stocking, B967
Storage depletion, A459
Stratification, A725
  Estuary, A133
Stream, A409, B555
  Acoustic, A215
  Cobble-Bed, A203, A239
  Flood, A583
  Habitat, A665, B45
  Hydraulics, A203
  Montainous, A699
  Mountainous, A583
Stress
  Index, B175
  Riparian tree, A121
Submergence, B861
Substrate, B365, B861
  Channel, B227
  Habitat modeling, A319
  Near-bed flow, A203
  Salmon spawning, A305
Succession, A797
Sudano-Sahelian zone, A811
Suitability
  Curves, B519
  Habitat, A59, A743, B33, B163, B239, B365, B407, B607
  Index, B55, B175
Suitability index
  Habitat, B431
Support system, A769, A829
Surf zone, A525
Surface flooding, A121
Survey
  Fish, A421
  Shore erosion, A113
Survival, B477
Suspended solids, B175
Susquehanna River, B119
Swimming
  Critical speed, B487
  Fish, B915
  Performance of salmon, A227
Switzerland, B683

T

Tahsis River, A261
Takhini River, A459
Tama River, B729
Tänassilma River, A177
Target species, A595
Tavy River, B203
Tejo River, B767
Telemetry, A83, B555
  Swimming performance, A227
Temperature, B465
  Related to hydropeaking, A699
  Water, B915
Temporal
  Variability, B163
Tenryu-gawa River, A583
Test
  Minimum flow, B419
  Simulations, B353
Three-dimensional
  Model, B353
Tidal
  Cycle, A379
  Ice cover, A435
  Marsh, B705
  Power station, A145
Tide, A155
Total dissolved gases, A3
Trame River, B683
Transect, A571
  Sampling, B119
Transferability, B407
Transport-diffusion
  Model, A759
Transverse profiles, B175
Traps, B903
Trophic value, B283
Tropical zone, A37
Trout
  Biomass, B507
A395, B843
Turbine mortality, B929
Turbine passage, B941
Turbulence
  Acoustics, A215
  K-e model, A293
  Macrophytes, A191
  Periphyton, A239
Turbulent boundary layer
  Under ice, A435
Two-dimensional
  Hydraulic model, B149
  Model, B187, B251, B365, B395, B519

U

United Kingdom, A513, A525, B203, B227, B407, B431, B607
Upstream Migration
  Eel, A89
Urban water environment, A769
USA, A319, A471, A609, A665, A737, A845, A871, B107, B119, B149, B251, B263, B275, B299, B307, B395, B447, B519, B543, B643, B929, B953
Ushizu River, B239, B631

V

Validation
  Biological, B149
  Model, B519
  Strategy, B365
Variability
  Spatial, B163
  Temporal, B163
Vegetation
  Bank, B729, B741
  Flow, A293
  Friction factor, A251
  Growth, A395
  Influence on flow, A395
  Influence on river-bed, A389
  Revetment, B729
  Riparian, A121, B669
  River, A251
  Roughness, A251
  Shore, A113

Velocity, B55, B487
  Distribution, B631
  Diverse, B855
  Macrophytes, A191
  Wetted perimeter, A571
Vertical slot, B873
Video monitoring, B885
Vienne River, A89

W

Walleye, B655
Wastewater treatment, A177
Water
  Density, A155
  Diverson, A571
  Level, A47
  Management, A59, A595, A619, A711
  Municipal, A769
  Quality, A665, A737, A753
  Quality index, A779
  Resources potential, A811, A331
  System, A643, B741
  Temperature, B915
  Transfer, B431
Watercourse, A835
Waterfowl, A851, A865
Watershed, A665
  Ecological management, A643
Wave, A525, A113
  Erosion, A501
Wear River, B431
Weighted Usable Area (WUA), B55, B239, B407
Weighted Usable Volume (WUV), B55
Weirs, A737, B595
  Salmon, B915
West Salmon River, A533, B33
Wetland value, B669
Wetlands, A167, A177, A653, A811, A851, A865, B783
Wetted perimeter
  Infection point, A571
Wey River, B607
Whitefish, B873, B967
Wildlife, A851
Willamete River, B299
Willow, B683
Winter, A459, A471, B477
  Flood, A583
  Operations, B717
  Processes, A435
Withdrawal, A725
Wooden cribs, B669

XYZ

Young of the year, B903
  Rheophilic cyprinid, B487
Yukon, A459
Zooplancton, A37

Index of Authors

Index des auteurs

A

Acreman, M.C., B607
Addison, D.S., B643
Alcazar, J., A547
Alfredsen, K., B215
Allen, M.A., B119
Anctil, F., A155, A447
Andriamahefa, H., A631
Annoni, P., B507
Aoki, M., B631
Asaeda, T., A725
Asakusa, H., A583

B

Baglinière, J.L., B531
Bain, M.B., A665, B107, B543
Bakken, T.H., B307
Balogun, I., A811
Baras, E., A83, A367
Bardonnet, A., B341
Bariteau, L., B717
Bartsch, N., B251
Beaulieu, C., B797
Beaumont, P., B705
Bechara, J.A.,A3
Beiter, T., B903
Belaud, A., A673
Belzile, L., B365
Bergeron, N., A379, A435
Bernert, J.A., B299
Bernez, I., A13
Bertrand, P., B705
Bérubé, M., B717
Bérubé, P., A485
Biggs, B., A239
Bird, D., B203
Bird, D.J., B407
Birkhead, A.L., A121
Birtles, C., A83
Bochechas, J., B885
Boët, P., B133
Böhmer, J., B903
Booth, D., A133
Boudreau, A., B365
Boudreau, P., B187, B365
Bourgeois, G., B365, B381
Bovee, K., B149
Breil, P., A421, B133
Browne, J., B941
Bult, T.P., B67, B353

C

Caissie, D., A409, B381
Capra, H., A421, B95
Carpenter, K.E., A513
Carrel, G., B283
Castonguay, M., A753, B585
Cerdan, P., A37
Cereghino, R., A699
Champeau, A., A37
Chaumont, D., A379
Chauvelon, P., A167
Chesher, T.J., A513
Claus, K., A835
Cohen, P., A631
Colavecchia, M.V., A227
Corfa, G., B81
Courtenay, S.C., A133
Crawford, D.J., B643
Crivelli, A.J., A167
Crosato, A., B175
Cunjak, R., B477

D

De Vries, M.B., A59
Denis, R., A113
Domitrovic, H.A., A3
Ducruc, J.P., A643
Duel, H., A619
Dumont, B., B283
Dupuis, S., B783

E

Eilers, J.M., B299
El-Jabi, N., A409, B381
Elliott, C.R.N., B431, B607
Eon, J., B419
Ernegger, T., A797

F

Fathi-Mogadam, M., A251
Fedorov, M.P., A145
Ferreira-Machado, M.T., B767
Fisher, K., B595
Flore, L., B487
Fournier, J.J., A99
Fox, P., B227
Fukuoka, S., B729

G

Gagnon, B., B655
Gagnon, G., B717
Galbreath, G.H., A665
Garguet-Duport, B., A25, A653
Gendron, M., B655
Gentili, G., B175, B507
Gerardin, V., A643
Gibbins, C.N., B431
Gidas, M.B., A759
Gidas, N.K., A759
Gippel, C., A215, A571
Girel, J., A25, A653
Goosney, R., A227
Gore, J.A., B643
Goring, D., A239
Gotoh, H., A343
Gotoh, K., A725
Gouraud, V., B531
Grande, R., A789
Grégoire, A., A37
Gubala, C.P., B251, B299
Guzzi, L., B507

H

Hadjerioua, B., A737
Haedrich, R.L., B67
Hall, K.R., A501
Hamilton, S.A., A459
Hara, T., B855
Harby, A., B307, B353
Hardy, L., B717
Hardy, T.B., A845, B251, B395, B447, B519

Haro, A., B953
Harshbarger, E.D., A737
Haury, J., A13
Hauser, G.E., A737
Hayes, J., B465
Heggenes, J., B307, B353
Heritage, G.L., A121, A331, A355
Herva, M., B915
Holmes, P., A525
Hooli, J., B873
Horeau, V., A37
Huusko, A., B619
Hyvärinen, P., B967

I

Ishikawa, Y., B631

J

Jacobo, W.R., A3
James, C.S., A121, A331
Jansen, W., B903
Janssens, L., A835
Jeffries, M.J., B431
Johnsen, M., B579
Johnson, P., A871
Jorde, K., B163
Jouve, D., B323
Julien, M., A99

K

Kamula, R., B873
Kappus, B., B903
Katopodis, C., A227, B555
Kawahara, Y., B239, B631
Kazusa, S., B855
Keckeis, H., B487
Kellerhalls, R., A261
Kerkhofs, M.J.J., A595, A711
Killingtveit, A., B215
Kim, H.R., B239
Kitamura, T., A389, A395
Klein Breteler, J.T., A743, B741
Klein, J.D., A711
Knight, J.G., B107
Komura, S., B855, B861
Koutitonsky, V.G., A759
Kouwen, N., A251
Kreivi, P., B45
Krummscheid-Plankert, P., B753
Kubota, M., A583
Kutsumi, T., A583

L

Laane, W.E.M., A619
Lachat, B., B683
Laine, A., B873
Lalumière, R., A99
Lamouroux, N., B95
Lapointe, M.F., A305
Larinier, M., B891
Latka, D., A871
Lavandier, P., A699
Le Coarer, Y., B283, B323
Leclerc, M., B187, B365
LeDrew, L.J., A533, B33
Legault, A., A89
Lehoux, D., B669, B705
Lek, S., A673
Lemieux, C., A99
Lévesque, F., A485, B81
Liebig, H., A673
Lim, P., A673
Locke, A., A133
Locke, A.G.H., A559
Ludlow, J., B519
Lynch, D., A769
Lyness, J., B595

M

Maddock, I., B203
Mader, H., B55
Mäki-Petäys, A., B45, B967
Manné, S., B531
Marche, C., A753, B585
Martin, N., A435
Massicotte, B., B705
Massicotte, M.C., B775
Matsuzaki, H., B27
Matumoto, A., B631
McKinley, R.S., A227, B15, B33
Meixler, M.S., A665
Menella, J.Y., B829
Mercier, Y., A851, A865
Merle, G., B419
Messier, D., A447
Miles, M., A261
Milhous, R.T., A319
Mitchell, J., B15
Moir, H., B3
Monfort, O., B133
Moog, O., A685
Moore, R.D., A459
Morin, G., A409
Morin, J., B187
Muotka, T., B45
Murakami, S., A395
Murphy, M., B941
Myers, R., B595

N

Nakamura, S., A583
Nasello, C., A779
Naura, M., B227
Neshaei, A.L., A525
Nestler, J.M., A871, B263, B275, B929
Nicolson, C.R., A331
Nicosia, S., A779
Nindaba, J., A367
Nõges, P., A47, A177
Nõges, T., A47
Nottage, A.S., A513

O

O'Connor, L., B941
O'Farrell, M., B941
O'Maoileidigh, N., B941
O'Neill, I., A215
O'Sullivan, L., B941
O'Sulliven, J., B595
Odeh, M., B953
Ostendorp, W., B753
Ouellet, Y., A155
Ovidio, M., A83

A

Acreman, M.C., B607
Addison, D.S., B643
Alcazar, J., A547
Alfredsen, K., B215
Allen, M.A., B119
Anctil, F., A155, A447
Andriamahefa, H., A631
Annoni, P., B507
Aoki, M., B631
Asaeda, T., A725
Asakusa, H., A583

B

Baglinière, J.L., B531
Bain, M.B., A665, B107, B543
Bakken, T.H., B307
Balogun, I., A811
Baras, E., A83, A367
Bardonnet, A., B341
Bariteau, L., B717
Bartsch, N., B251
Beaulieu, C., B797
Beaumont, P., B705
Bechara, J.A.,A3
Beiter, T., B903
Belaud, A., A673
Belzile, L., B365
Bergeron, N., A379, A435
Bernert, J.A., B299
Bernez, I., A13
Bertrand, P., B705
Bérubé, M., B717
Bérubé, P., A485
Biggs, B., A239
Bird, D., B203
Bird, D.J., B407
Birkhead, A.L., A121
Birtles, C., A83
Bochechas, J., B885
Boët, P., B133
Böhmer, J., B903
Booth, D., A133
Boudreau, A., B365
Boudreau, P., B187, B365
Bourgeois, G., B365, B381
Bovee, K., B149
Breil, P., A421, B133
Browne, J., B941
Bult, T.P., B67, B353

C

Caissie, D., A409, B381
Capra, H., A421, B95
Carpenter, K.E., A513
Carrel, G., B283
Castonguay, M., A753, B585
Cerdan, P., A37
Cereghino, R., A699
Champeau, A., A37
Chaumont, D., A379
Chauvelon, P., A167
Chesher, T.J., A513
Claus, K., A835
Cohen, P., A631
Colavecchia, M.V., A227
Corfa, G., B81
Courtenay, S.C., A133
Crawford, D.J., B643
Crivelli, A.J., A167
Crosato, A., B175
Cunjak, R., B477

D

De Vries, M.B., A59
Denis, R., A113
Domitrovic, H.A., A3
Ducruc, J.P., A643
Duel, H., A619
Dumont, B., B283
Dupuis, S., B783

E

Eilers, J.M., B299
El-Jabi, N., A409, B381
Elliott, C.R.N., B431, B607
Eon, J., B419
Ernegger, T., A797

F

Fathi-Mogadam, M., A251
Fedorov, M.P., A145
Ferreira-Machado, M.T., B767
Fisher, K., B595
Flore, L., B487
Fournier, J.J., A99
Fox, P., B227
Fukuoka, S., B729

G

Gagnon, B., B655
Gagnon, G., B717
Galbreath, G.H., A665
Garguet-Duport, B., A25, A653
Gendron, M., B655
Gentili, G., B175, B507
Gerardin, V., A643
Gibbins, C.N., B431
Gidas, M.B., A759
Gidas, N.K., A759
Gippel, C., A215, A571
Girel, J., A25, A653
Goosney, R., A227
Gore, J.A., B643
Goring, D., A239
Gotoh, H., A343
Gotoh, K., A725
Gouraud, V., B531
Grande, R., A789
Grégoire, A., A37
Gubala, C.P., B251, B299
Guzzi, L., B507

H

Hadjerioua, B., A737
Haedrich, R.L., B67
Hall, K.R., A501
Hamilton, S.A., A459
Hara, T., B855
Harby, A., B307, B353
Hardy, L., B717
Hardy, T.B., A845, B251, B395, B447, B519

Haro, A., B953
Harshbarger, E.D., A737
Haury, J., A13
Hauser, G.E., A737
Hayes, J., B465
Heggenes, J., B307, B353
Heritage, G.L., A121, A331, A355
Herva, M., B915
Holmes, P., A525
Hooli, J., B873
Horeau, V., A37
Huusko, A., B619
Hyvärinen, P., B967

I

Ishikawa, Y., B631

J

Jacobo, W.R., A3
James, C.S., A121, A331
Jansen, W., B903
Janssens, L., A835
Jeffries, M.J., B431
Johnsen, M., B579
Johnson, P., A871
Jorde, K., B163
Jouve, D., B323
Julien, M., A99

K

Kamula, R., B873
Kappus, B., B903
Katopodis, C., A227, B555
Kawahara, Y., B239, B631
Kazusa, S., B855
Keckeis, H., B487
Kellerhalls, R., A261
Kerkhofs, M.J.J., A595, A711
Killingtveit, A., B215
Kim, H.R., B239
Kitamura, T., A389, A395
Klein Breteler, J.T., A743, B741
Klein, J.D., A711
Knight, J.G., B107
Komura, S., B855, B861
Koutitonsky, V.G., A759
Kouwen, N., A251
Kreivi, P., B45
Krummscheid-Plankert, P., B753
Kubota, M., A583
Kutsumi, T., A583

L

Laane, W.E.M., A619
Lachat, B., B683
Laine, A., B873
Lalumière, R., A99
Lamouroux, N., B95
Lapointe, M.F., A305
Larinier, M., B891
Latka, D., A871
Lavandier, P., A699
Le Coarer, Y., B283, B323
Leclerc, M., B187, B365
LeDrew, L.J., A533, B33
Legault, A., A89
Lehoux, D., B669, B705
Lek, S., A673
Lemieux, C., A99
Lévesque, F., A485, B81
Liebig, H., A673
Lim, P., A673
Locke, A., A133
Locke, A.G.H., A559
Ludlow, J., B519
Lynch, D., A769
Lyness, J., B595

M

Maddock, I., B203
Mader, H., B55
Mäki-Petäys, A., B45, B967
Manné, S., B531
Marche, C., A753, B585
Martin, N., A435
Massicotte, B., B705
Massicotte, M.C., B775
Matsuzaki, H., B27
Matumoto, A., B631
McKinley, R.S., A227, B15, B33
Meixler, M.S., A665
Menella, J.Y., B829
Mercier, Y., A851, A865
Merle, G., B419
Messier, D., A447
Miles, M., A261
Milhous, R.T., A319
Mitchell, J., B15
Moir, H., B3
Monfort, O., B133
Moog, O., A685
Moore, R.D., A459
Morin, G., A409
Morin, J., B187
Muotka, T., B45
Murakami, S., A395
Murphy, M., B941
Myers, R., B595

N

Nakamura, S., A583
Nasello, C., A779
Naura, M., B227
Neshaei, A.L., A525
Nestler, J.M., A871, B263, B275, B929
Nicolson, C.R., A331
Nicosia, S., A779
Nindaba, J., A367
Nõges, P., A47, A177
Nõges, T., A47
Nottage, A.S., A513

O

O'Connor, L., B941
O'Farrell, M., B941
O'Maoileidigh, N., B941
O'Neill, I., A215
O'Sullivan, L., B941
O'Sulliven, J., B595
Odeh, M., B953
Ostendorp, W., B753
Ouellet, Y., A155
Ovidio, M., A83

P

Palau, A., A547
Parasiewicz, P., A685
Parrish, D.L., A471
Pautou, G., A25, A653
Payne, B., A305
Pedroli, B., A619
Peviani, M., B175
Philippart, J.C., A83, A367
Pier, A., B753
Ploskey, G.R., B929
Poizat, G., A167
Pouilly, M., B95, B323
Power, G., B15, B33
Priyantha, D.G.N., A725
Proctor, W.D., A737
Proulx, M., B81

Q

Quigley, D., B941

R

Rahmann, H., B903
Raven, P., B227
Reinhold, J.O., A743, B741
Rivier, B., B283
Robert, B., A829
Robert, J.L., A155
Rochette, A., A829
Roussel, J.M., B341
Rowntree, K.M., A281
Roy, A.G., A379, A435

S

Sabaton, C., B531
Saccardo, I., B175, B507
Saint-Hilaire, A., A133, A409
Saitoh, S., A725
Sakai, T., A343
Salverda, P., A711
Sandnæs, T.O., B579
Sandoz, A., A167
Schmutz, S., A685, B55
Schneider, D.C., B67
Schneider, L.T., A871, B275, B929
Scruton, D.A., A227, A533, B15, B33, B307
Secretan, Y., B187
Sekine, M, A583
Semmekrot, S., A595
Shen, H.W., A609
Shilin, M.B., A145
Shimizu, Y., B843
Siegler, L., B531
Souchon, Y., B323
Soulsby, C., B3, B431
Stark, H., B753
Steiger, J., A71
Stephan, U., A191
Stewardson, M., A215, A571
Storset, L., A789
Suard, G., B283
Sutton, V., B263

T

Tamai, N., B27, B239, B631
Tarbet, K.L., B395
Telhado, A., B767
Thériault, I., B655
Therrien, J., B477
Travade, F., B891
Travnichek, V.T., B543
Tsujimoto, T., A293, A389, A395, B843

U

Unfer, G., B55

V

Valentin, S., B307
Van Coller, A.L., A121
Van Niekerk, A.W., A331, A355
Van Liebergen, J.C.G., A595
Vehanen, T., B967
Verbraak, P.J.J., A711

W

Wadeson, R.A., A281
Wallingford, H.R., B595
Wasson, G., A631
Watanabe, A., B729
Weeks, D., A355
Weeks, G., B929
Whalen, K.G., A471
Wiebe, K., A459, A769
Willis, D.J., B607
Wychera, U., A191

XY

Yasuda, M., B631
Young, W.J., A203
Youngson, A., B3
Yrjänä, T., B619

Z

Zylberblat, M., B829