ECOHYDRAULICS
ÉCOHYDRAULIQUE
2000

2nd INTERNATIONAL SYMPOSIUM ON HABITAT HYDRAULICS

2e SYMPOSIUM INTERNATIONAL SUR L'HYDRAULIQUE ET LES HABITATS

QUÉBEC, JUNE / JUIN 1996

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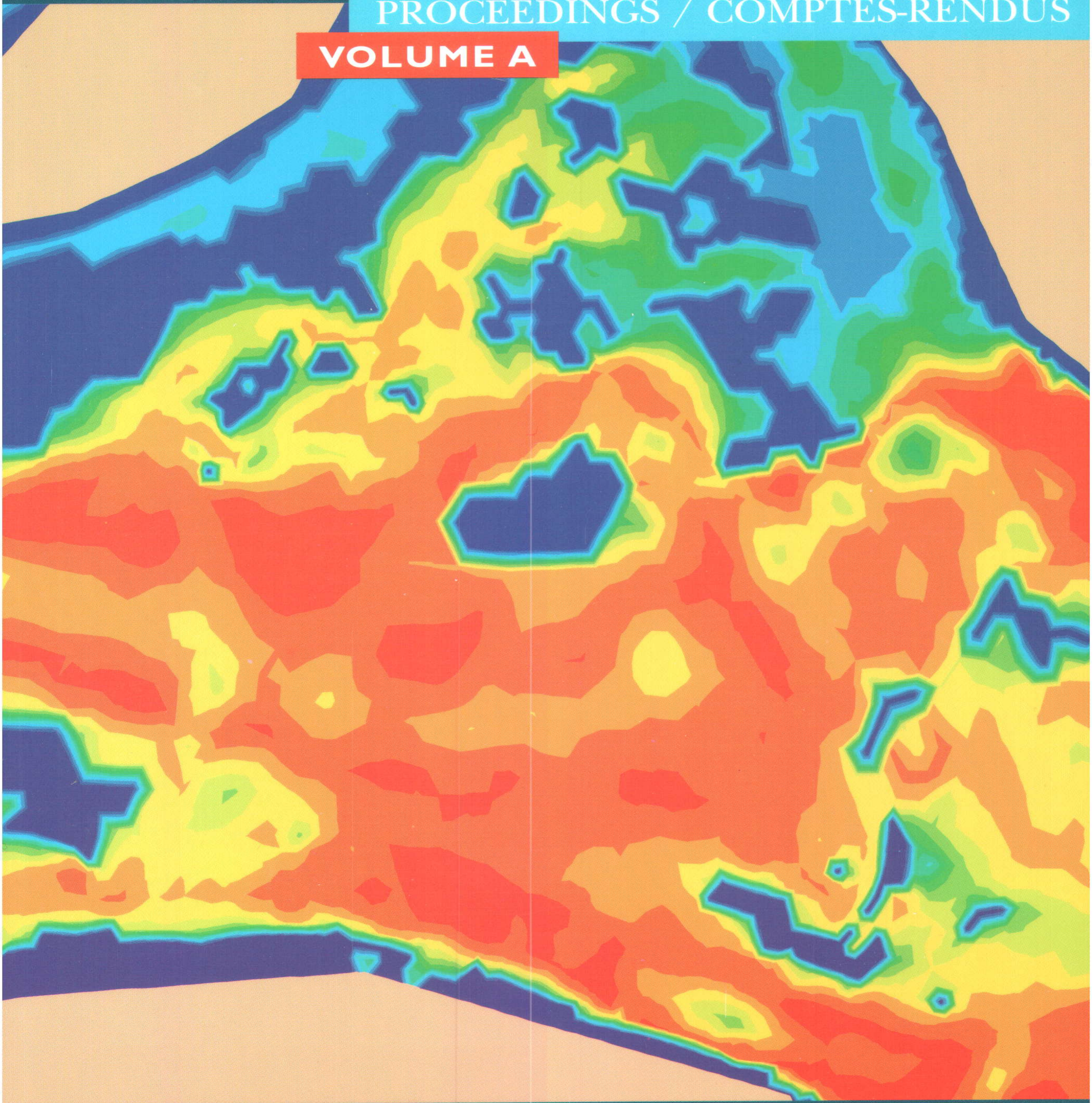
VOLUME A



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Two-Dimensional Map of Salmon Parr Habitat on Moisie River (Québec, Canada) for a severe low flow event (55 m^3/s) - See also the present Proceedings (Boudreau *et al*, Vol. B, pp. B365-B380)

Carte bi-dimensionnelle de l'habitat du saumon, stade tacon, sur la rivière Moisie (Québec, Canada) for un débit d'étiage très sévère (55 m^3/s) - Voir aussi les présents Comptes-rendus (Boudreau et al, Vol. B, pp. B365-B380)

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2ième Symposium international sur l'hydraulique et les habitats

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Volume A

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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*

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Effets directs et à distance des ouvrages de génie

THE EFFECT OF GAS SUPERSATURATION ON FISH HEALTH BELOW YACYRETÁ DAM (PARANÁ RIVER, ARGENTINA)

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Supersaturation below the spillways of dams can provoke Gas Bubble Disease (GBD) in fish and other riverine animals. This was the case of Yacyretá Dam, a low head (21 m), 1,600 km^2 hydroelectric project located on the Paraná River, at the international border of Argentina and Paraguay. The dam has two gated overflow spillways. Spillage of water over the dam entrains air and causes supersaturation in the stilling basins placed below each spillway. A massive fish mortality caused by GBD was observed in 1994, in a 100-km reach below Yacyretá Dam. After the event, modifications in the operation of spillway gates reduced the levels of supersaturation. Other improvements are presently being implemented, such as modifications in the structure of spillway gates to reduce supersaturation levels, as well as increases in discharge passing through turbines. An intensive monitoring of dissolved gases and fish sanitary conditions was carried out by an agreement between the public power utility of Yacyretá Dam (EBY) and the Institute of Ichthyology of the National University of the Northeast (UNNE). The first results of this survey are reported here. Gas supersaturation was measured daily at different sampling points in the river using a *Common Sensing* electronic saturometer. Fish health condition was examined in individuals caught by gillnets in two sampling points within the river; one of them placed close to the main spillway and the other located 70 km downriver, every 20-30 days, from March 1995 to March 1995. Each fish was examined for emphysema (bubbles) within the skin, eyes and fins, and some of them were dissected to permit internal examination and histological sampling of various tissues. Close to the spillway, in the main river channel, total gas supersaturation values oscillated in a range comprised between 125,4 and 153.3 %. In this sampling point, 13% to 58% of the fish caught were found with emphysema within the fins or other GBD signs. This high incidence of the GBD would have been artificially enhanced by the netting, because fish were caught at low depths (0.5-3.0 m), out of the hydrostatic compensation zone. Histopathological analyses confirmed the presence of lesions caused by

GBD in gills and liver in about 35% of the fish sampled. In the sampling point located 70 km downstream, supersaturation values varied in a range comprised between 107-130%, which was closer to the normal river conditions. Percentage of fish affected varied between 0 and 9%, but no histopathological lesions were found in gills or liver. These results indicate that the dam is still generating dangerous levels of supersaturation, and fish inhabiting the shallowest areas of the river (<3 m) can develop the GBD. Modifications being in execution will likely reduce the present levels of supersaturation.

KEY-WORDS: gas bubble disease/ fish/ gas supersaturation/ total dissolved gases/ dams/ large rivers/ Argentina.

INTRODUCTION

A major characteristic of the South American drainage systems is the presence of large rivers with extensive floodplains having a diversified fish fauna. The Paraná River belongs to the Del Plata Basin, the second largest drainage basin of the subcontinent after the Amazon Basin. Since 1960, considerable development of civil works in Del Plata Basin has transformed large reaches of the headwater zone in a succession of reservoirs, particularly in Brazil. In addition, two huge dams were constructed in the middle sections of the Paraná River during the 80's, named Itaipú at the frontier of Brazil and Paraguay, and Yacyretá in the border of Argentina and Paraguay. These dams together can supply a large proportion of the power requirements of these countries.

In 1994, a massive fish mortality below Yacyretá Dam was observed. This mortality was due to an acute gas bubble disease (GBD) produced by gas supersaturation in the stilling basins placed below the spillways (Domitrovic et al., 1994). This happened after the level of the reservoir was raised several meters to begin the energy production of the first installed turbine. A massive fish kill was also observed below Itaipú Dam several years before, under similar circumstances (G. Gavilán, pers. obs.).

After the event, modifications in the operation regime of spillway gates reduced the levels of gas supersaturation downriver Yacyretá Dam. Other improvements are presently being implemented, such as the modification of spillway design by adding structures to avoid spilled water to plunge deeply into the stilling basin. Besides, larger amount of water passing through an increasing number of installed turbines is also supposed to reduce the discharge in the spillways.

An intensive monitoring of total dissolved gases and fishes sanitary conditions were carried out by an agreement between the public power utility of Yacyretá Dam (EBY) and the Institute of Ichthyology of the National University of the Northeast (UNNE). The objectives of this study were, (i) to survey the impact of the fluctuating supersaturation levels on fish health and, (ii) to create a baseline reference to evaluate the success of the structural and operational modifications that are presently being implemented in the dam to reduce supersaturation levels. The first results of this study are reported here.

STUDY SITE

Yacyretá Dam and Reservoir are located a few kilometers upstream Ituzaingó Town, in Corrientes Province, Argentina (Fig. 1). The reservoir has a surface of approximately 1,600 km^2 and a volume of 21,000 hm^3. The dam has a length of 63.7 km, with two gated overflow spillways, one of them placed on the Añá Cuá channel (Añá Cuá Spillway, 16 gates), and the remaining on the main channel (Principal Spillway, 18 gates). Spillways have mobile valves to control flow and air ducts to reduce cavitation effects. However, both features produce a substantial increase in the amount of air incorporated to the water that plunge 20 m depth in the stilling basins placed below each spillway.

The power house is located next to the Principal Spillway. All the 20 Kaplan turbines programmed will be operative by 1998. At the beginning of the survey, there were only three turbines running, while five other units were working at the end of the study period. Each turbine can discharge a maximum of 850 $m^3 s^{-1}$, summing together 3200 MW (160 MW for each turbine) of total capacity. Total dissolved gases of the turbinated waters have similar levels to those measured in the reservoir (<105%, G. Gavilán, pers. obs.). The dam operates on a near-run-of-the-river regime, with minor fluctuations due to the operation of turbines and spillways. There is also two fish scales equipped with elevators, located on each side of the powerhouse. Elevators operate continuously transporting migratory fishes from the river to the reservoir.

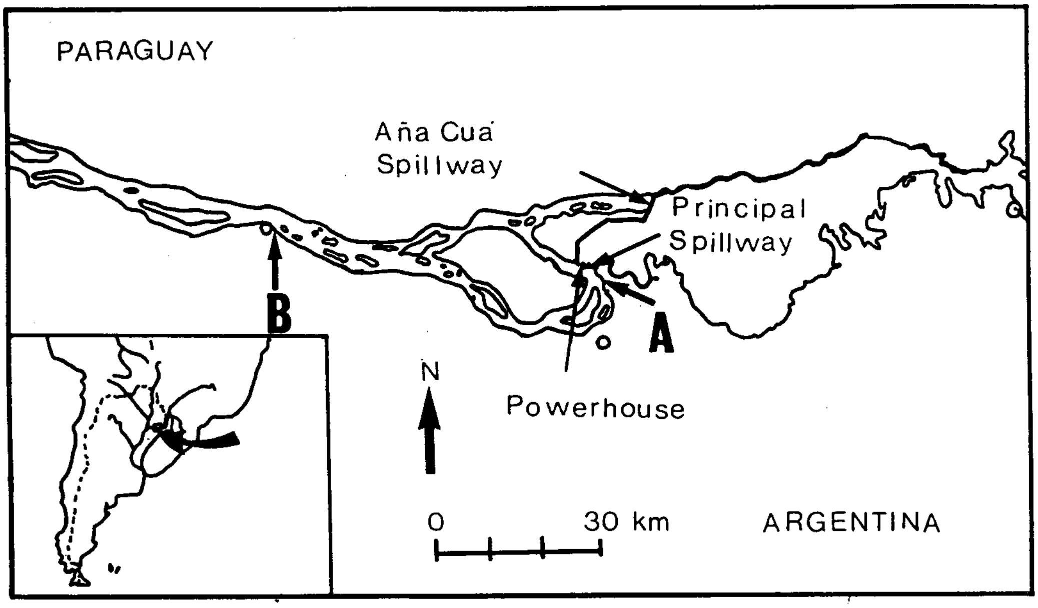

Fig. 1: Geographic location of the study area. A: Impact Station, B: Control Station.

The Paraná River at Yacyretá Dam has a mean discharge of 12,000 $m^3 s^{-1}$, and drains an area of about 840, 000 km^2. Along several kilometers below the reservoir, the river flows mainly through basalt bedrock and sandstone, changing downriver to moving sands bedloads, with isolated basaltic and sandstone outcrops. On the bedrock, there is massive growth of epiphytic algae and macrophytes, especially diatoms and the reophylic Podostemaceae.

MATERIAL AND METHODS

Field Studies

Fish were sampled at two locations, one of them placed about 2 km below the Principal Spillway on the left margin of the river, and it is named hereafter Impact Station. The other sampling station was placed 70 km downriver, at Itá Ibaté Town (Control Station) (Fig. 1). Maximum river depths at medium water levels are about 5 m in Yacyretá and 8 m in Itá Ibaté. A battery of nine gillnets was placed on each site every 20-30 days on 16 different occasions from March 1995 to March 1996. Each gillnet was placed perpendicularly to the coast, in contact with the bottom, and had the following mesh size (stretched from knots to knots) in cm: 4, 5, 6, 7, 8, 12 14, 16 and 20. Fish caught were collected every 6-8 hours during a 48 hours period. Fish were identified taxonomically to the specific level, weighted (g), measured (standard length in cm), and examined for the presence of macroscopic gas bubble disease (GBD) signs, such as emphysema in skin and fins, exophthalmia and haemorrhages. Liver and gill samples of the species of economical importance were preserved in 10% formalin to be later processed in the laboratory for histopathological analyses. A total of 9250 individuals was examined externally and 300 of them were dissected for microscopic examination.

At each sampling point, the total dissolved gases were measured using a *Common Sensing* electronic saturometer. Measures were taken every one to three days at the Impact Station and on each sampling date at the Control Station. Additionally, other routine environmental variables were collected on each sampling date, such as water and air temperature, pH, conductivity, transparency (Secchi disk), and dissolved oxygen. Also, current speed and maximum depth were measured at the site of each gillnet location.

Laboratory Analyses

Tissues fixed in 10% formalin were imbedded in paraffin, sectioned at 7 μm layers, mounted on microscope slides and stained with hematoxylin and eosin. Samples of a similar number of both healthy and affected fish were examined, and the following signs were associated to the GBD: presence of gas emboli and microemboli in vessels and capillaries, empty capillaries, emphysema in tissues, edema, congestion, and telangiectasia in capillaries. Other lesions caused by parasites, injuries, or microbes were also registered.

Data Analyses

The information was stored in a database and the total percentage of fish having macroscopic GBD signs was computed for every site date and sampling point. For each species, the percentage affected with GBD was also computed for each sampling point. The relationship between the percentage of all fish with GBD and some environmental variables (total dissolved gases (%TDG), water temperature (°C), and mean depth at gillnet location) was examined using standard multiple regression procedures (Tabachnick and Fidell, 1989). Data were previously transformed to log (x+1) (%TDG and temperature) or to sqr (x) (depth) to achieve a distribution of residuals close to normality, linearity and homoscedasticity. Durbin-Watson statistic was also applied to reveal the presence of serial correlation.

RESULTS

River Discharge And Total Dissolved Gases (%TDG)

The %TDG in the left margin of the river was highly associated with the discharge of the Principal Spillway (Fig. 2), as evidenced by a significant positive correlation between both variables ($r_{Pearson} = 0.769$, $p<0.001$, $n=186$). The amount of turbinated water increased gradually with time, as new turbine units were being included in the powerhouse, but %TDG did not show a clear tendency to decrease with time (Fig. 2).

As a matter of fact, %TDG were almost all the time over 130% at the Impact Station, reaching values as high as 153% during high water periods (Fig. 2). Values at the Control Station were much lower and varied between 107 and 126%. At Paso de la Patria, a locality placed 250 km downriver Yacyretá Dam, supersaturation values were of 108% during a high water period (February 1996).

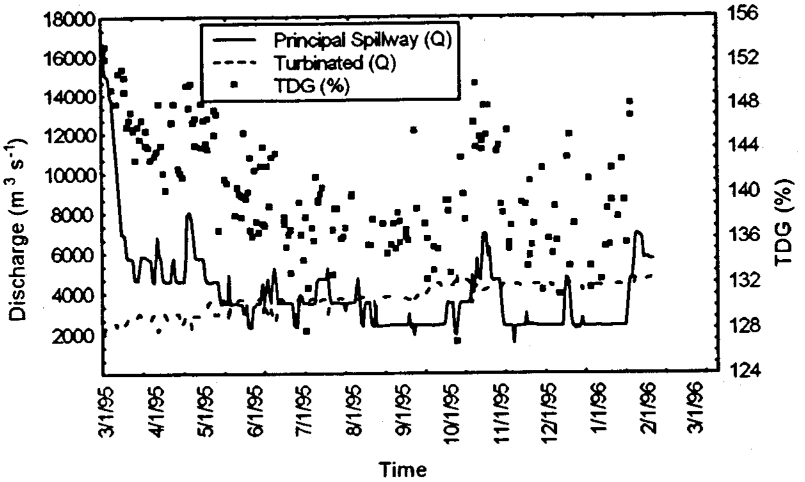

Fig. 2: Variations in discharge (Q) of the Principal Spillway and turbines, and total dissolved gases (%TDG) along the study period.

Total Dissolved Gases And Gas Bubble Disease (GBD)

Macroscopic Examination

About 73 different fish species were collected during the study period at each sampling site, and 51 of them presented macroscopic signs of GBD at the Impact Station, while only 19 were affected at the Control Station . The small to medium size species were the most frequently affected (Table 1). In this sense, a significant negative correlation between mean size of the species and the percentage of affected fish was observed ($r_{Pearson}$= -0.44, P< 0.001, N= 50). On average, 35% of the sampled individuals had macroscopic lesions caused by GBD in the Impact Station and only 3% in the Control Station. In spite of this high occurrence of the disease, there was not massive fish kills, although groups of dead fish were occasionally found at the higher %TDG levels.

There was a significant relationship between the %TDG and the percentage of fish with macroscopic lesions caused by GBD, as shown in Table 2. Most of the variability in the percentage of fish affected was explained by %TDG, but water temperature, and in minor extent, mean depth at gillnets location also showed significant effects. They explained together almost all the observed variability in %GBD. Furthermore, Fig. 3 shows that when %TDG surpassed 120%, the percentage of fish with GBD increased rapidly in both sampling sites.

Table 1. List of the most important species affected by GBD at each sampling station and percentage of individuals presenting macroscopic signs. Average size corresponds to standard length. Species of economical interest are indicated with asterisks (*).

SPECIES	Control Station		Impact Station	
	Average size (mm)	Ind. with GBD (%)	Average size (mm)	Ind. with GBD (%)
Acestrorhynchus pantaneiro	170	3.03	-	0
Astyanax (P.) bimaculatus	112.5	1.8	107	57.89
Auchenipterus nuchalis	178.33	2.21	178.36	53
Brycon orbygnianus *	-	0	385	40
Cochliodon cochliodon	370	2.7	218.33	46.15
Cynopotamus argenteus	214.17	4.38	181.08	43.02
Cynopotamus zettii	241.67	3.23	178.13	56.07
Cyphocharax platanus	150	1.11	133.15	61.54
Galeocharax humeralis	205.14	3.75	141.84	46.25
Hemiodus orthonops	199.17	1.94	205.14	46.67
*Hemisorubim platyrhynchos**	-	0	367.5	6.67
Hoplias malabaricus	-	0	160	50
Hypostomus alatus	-	0	325	14.29
Hypostomus luteomaculatus	-	0	242.94	19.77
Leporinus friderici *	231	2.22	215.9	35.04
Leporinus obtusidens *	280	1.18	298.2	24.75
Lycengraulis olidus *	128	4	-	0
Mylossoma orbignyanum	-	0	197.31	10.74
Oxydoras kneri *	-	0	492.54	11.02
Pachyurus bonariensis	-	0	60	27.27
Piaractus mesopotamicus *	-	0	290	10.81
Pimelodus (I.) labrosus	168	2.02	150.95	20.58
Pimelodus albicans *	-	0	120	16.67
Pimelodus clarias *	213	1.68	238.03	31.87
Plagioscion ternetzii *	-	0	0	12.5
Prochilodus lineatus *	-	0	427.5	12.9
Prochilodus scrofa *	417.5	1.26	373.91	40.83
Psectrogaster curviventris	-	0	163.6	69.78
Pterygoplichthys anisitsi	-	0	400	4.35
Rhaphiodon vulpinus	420	1.89	378.89	18
Roeboides bonariensis	250	5.88	124.33	40
Roeboides prognathus	239.23	37.14	178.25	66.67
Salminus maxillosus *	-	0	314.06	12.59
Schizodon fasciatus	-	0	256.74	50
Schizodon nasutus	-	0	198.42	16.44
Serrasalmus nattereri	-	0	260	1.16
Serrasalmus spilopleura	-	0	228.75	8
Sorubim lima *	-	0	343.52	33.33
Triportheus paranensis	175.5	10	181.92	76.47

Microscopic Examination

Histopathological analyses carried out in species of economical importance (Table 1) revealed that 35% of them presented microscopic lesions caused by GBD in the Impact Station and only 2% in the Control Station. All of them corresponded to individuals having also macroscopic lesions. On the other hand, those in healthy condition did not present microscopic lesions caused by GBD. In gills, these lesions were generally characterised by empty capillaries, edema, congestion, small-sized gas emboli, capillary and vessel dilatation, anemia, and telangiectasia in the secondary lamellae. In liver, only gas emboli were observed but in few occasions. Note that the species of economical importance have generally the lower percentage of external GBD lesions, except the detritus-feeder *Prochilodus scrofa*.

Table 2. Multiple regression analysis of some environmental variables and percentage of fish affected by GBD. sr^2 (incremental) represents the absolute contribution of each independent variable to the total explained variance of the dependent variable, as expressed in the adjusted R^2. Durbin-Watson Statistics evidence a low serial correlation (0.162), and redundancy is also low for all independent variables (<0.123).

Variables	%GBD (dep.var.)	TDG	Temp.	Depth	B	β	sr^2 (incremental)
TDG	0.91				70.75	1.05	0.77**
Temp.	0.15	0.34			-19	-0.4	0.14**
Depth	-0.28	-0.31	-0.37	-	-1.43	-0.58	0.03**
				Intercept= -117.44**			
Means	32.39	134.61	23.89	2.33			
Standard Deviation	22.42	11.51	3.87	0.39		R^2=0.994	
						Adjusted R^2=0.993	

*=P<0.05; **=P<0.001; N=29

DISCUSSION

The results of the present study suggest that Yacyretá Dam generates supersaturated levels of total dissolved gases that can potentially affect the health condition of the fish inhabiting areas close to the margins of the river in a reach of several kilometers below the dam. It is not possible to extrapolate these results for fish living in deepest areas due to the difficulty of sampling. Nevertheless, at depths greater that 3 m, the majority of the fishes should be out of danger of GBD most of the time due to the hydrostatic compensation. In contrast, in the Control Station placed 70 km downriver, macroscopic signs of the disease had a low frequency of occurrence and appeared in a smaller number of species.

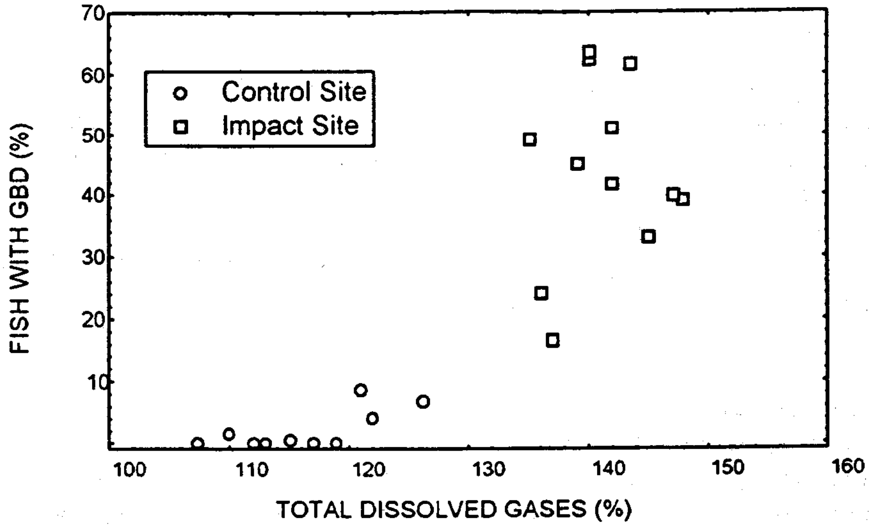

Fig. 3. Relationships between total dissolved gases (%TDG) and the percentage of fish presenting external signs of gas bubble disease (GBD) in the two sampling sites.

There was a strong relationship between the percentage of fish affected with external emphysema and the levels of total dissolved gases and water temperature. Given the high statistical significance of this empirical model, it could be eventually useful for the estimation of %TDG using the percentage of fish caught having GBD. The model must be previously validated for other independent situations. An exponential increase in the percentage of fish with external emphysema was observed once the levels of supersaturation surpassed 120% (Fig. 3). According to Bouck (1980), experiments carried out with steelhead trout (*Oncorhynchus mykiss*) revealed that after two to three days of exposure to supersaturated water, fish that survived develop chronic signs, such as large emphysema, vascular malfunction, and less frequently, exophthalmia. Large emphysema usually takes longer time to form than does gas emboli. This observation, along with the rareness of massive fish mortalities suggest that affected fish were exposed to chronic levels of supersaturation.

A number of questions can be raised concerning the sampling technique employed, since gillnet retention could produce erroneous results. First, fish caught in the gillnets are obviously unable to compensate hydrostatically by moving to lower depths, and being in supersaturated water they could rapidly develop the disease. In the present study, mean gillnet depth varied between 0.5 and 3 m in most cases. However, as mentioned above (Bouck , op. cit.), massive development of emphysema as observed in the present study takes usually longer than 8 hours (maximum time of exposure of fish caught in gillnets) to occur. Unpublished field bioassays by one of the author (H. Domitrovic) show that external emphysema can appear within 16 hours of exposition to %TDG higher than 115%, and without hydrostatic compensation. The supposed error caused by the nets should also produce a similar percentage of small and large fish affected, which was not the case as mentioned above. Therefore, the results can not be unequivocally attributed to the netting effects. Furthermore, the revision of Weitkamp and Katz (1980), mentions that bubbles in fins indicate chronic disease in many observed cases. Emphysema appears more frequently in caudal fins, which was also the case observed in the present research. A second sampling artefact may be due to the fact that after death, emphysema and gas microemboli tend to disappear as a result of the reduction in the intake of hyperbaric gases (Bouck, 1980). Since most fish were already dead when retired of the gillnets the actual number and magnitude of macroscopic and microscopic lesions may be higher than those observed.

Levels of supersaturation measured could have indirect effects on the structure of the fish communities below Yacyretá Dam. Particularly vulnerable could be the herbivorous/detritivorous fish guild, which are the most common group in the study area. According to the Secchi disk lectures (70-100 cm), and by employing an approximate calculation of the depth of the photic zone (2.5 times the Secchi disk) aquatic plants should growth on substrates placed between surface water and 1.75 to 2.5 meter depth. They are clearly out of the hydrostatic compensation zone that is over 3-5 m in the Impact Station. Therefore, fish would be forced to feed lower quality foods at deepest areas or being exposed to dangerous levels of supersaturation, and develop chronic or acute lesions caused by GBD near surface waters. Even supposing that they were able to detect the formation of gas emboli in blood and avoid supersaturated water, they had to move up and down continuously to feed, thus adding an additional stress factor, and an increased energy expenditure. Similarly, small sized fish that inhabit shallow areas would be also more affected than large fish (Table 1) by their preferential occupation of this habitat. In the short and mean term, this impact should be reflected at the individual level in low condition factors, fecundity, and growth, in comparison with fish inhabiting zones not exposed to supersaturated waters. However, the extensive migratory movements of many species within the river, and the heterogeneity of habitat conditions make these comparisons very difficult. The comparative approach would be then more appropriate in the long term and with species of more restricted movements.

Safe levels of supersaturation are difficult to establish with the present observational data due to the complexity of the environment and the large number of species affected. It is clear from this study that if a significant reduction in the number of fish with macroscopic GBD signs has to be achieved in coastal areas, maximum supersaturation levels should be always inferior to 120%. In this sense, the application of spillway structures as well as an increasing turbinated water will likely reduce the present values to more normal conditions, except during floods.

In a next stage of the study , a series of programmed field and laboratory bioassays will likely allow to better determine the safe levels of supersaturation necessary for the survival of the most important riverine species. It would be also useful to understand the mechanisms that produce the observed macroscopic and microscopic lesions of GBD in the field.

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DOWNSTREAM EFFECTS OF HYDROELECTRIC IMPOUNDMENT ON RIVER MACROPHYTE COMMUNITIES

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ABSTRACT

Macrophytes have been studied around two hydroelectric dams of the Massif Armoricain belonging to Electricité de France : Rophémel dam, located on the River Rance (Department of Côtes d'Armor), and Rabodanges dam, located on the River Orne (Department of Orne). Our aim was to assess the effects of hydroelectric functioning on river macrophyte communities. We studied mainly the first system, with 7 stations representative of different hydrological units, and to compare the results to those observed on the second one, only in the peaking flow stretches.

Two methods of macrophyte study were involved : floristic surveys on 50 m long stretches (28 for Rophémel dam, obtained in 7 sites with 2 stations, and 2 periods), and cover estimation (1026 quadrats, 3 periods). We considered genus of algae, species of bryophyta, hydrophytes and helophytes. Both methods gave convergent results.

Rophémel dams modified greatly macrophyte communities, specially the ratio between macro-algae, bryophytes, hydrophytes and helophytes. We observed a large extent of helophytes, *Phalaris arundinacea* and *Oenanthe crocata* and deposits in the main channel, which creates braided arms. There was an important development of algae (*Cladophora* sp., *Vaucheria* sp., *Spirogyra* sp., *Melosira* sp.) in all the disturbed parts, with a large cover and a great variety of taxa, while the rhodophyta *Hildembrandia* sp. disappeared. The presence of some species such as *Stigeoclonium sp.* assesses a degradation of water quality which is showed by ammonium contents. An eutrophication indicated for example by *Myriophyllum spicatum*, *Elodea canadensis*, *Potamogeton crispus* and *P. perfoliatus* was also noted. Seasonal changes were observed, affecting differently species of macrophytes, with a greater development of algae in summer and autumn and a decrease of *Ranunculus penicillatus* cover, with a delayed cycle just dowsntream from the dam. A progressive recovery of normal structures occurred about 3 km downstream from the dam.

In comparison, the peaking flow stretches dowstream from Rabodanges dam had a sparse vegetation, with much *Hildembrandia* sp., fewer green macro-algae. No deposits were observed. *Ranunculus fluitans*, *Myriophyllum spicatum* and *Potamogeton pectinatus* were the dominant hydrophytes.

These comparative results were discussed in a functional view. Chemical effects exist as a general context, but hydraulic effects seemed the striking ones to explain macrophyte composition, vegetation structure and its seasonal changes.

KEY-WORDS : River / Macrophytes / Impoudment / Regulated flow / Algae / Hydrophytes / Helophytes / Disturbance / Armorican Massif.

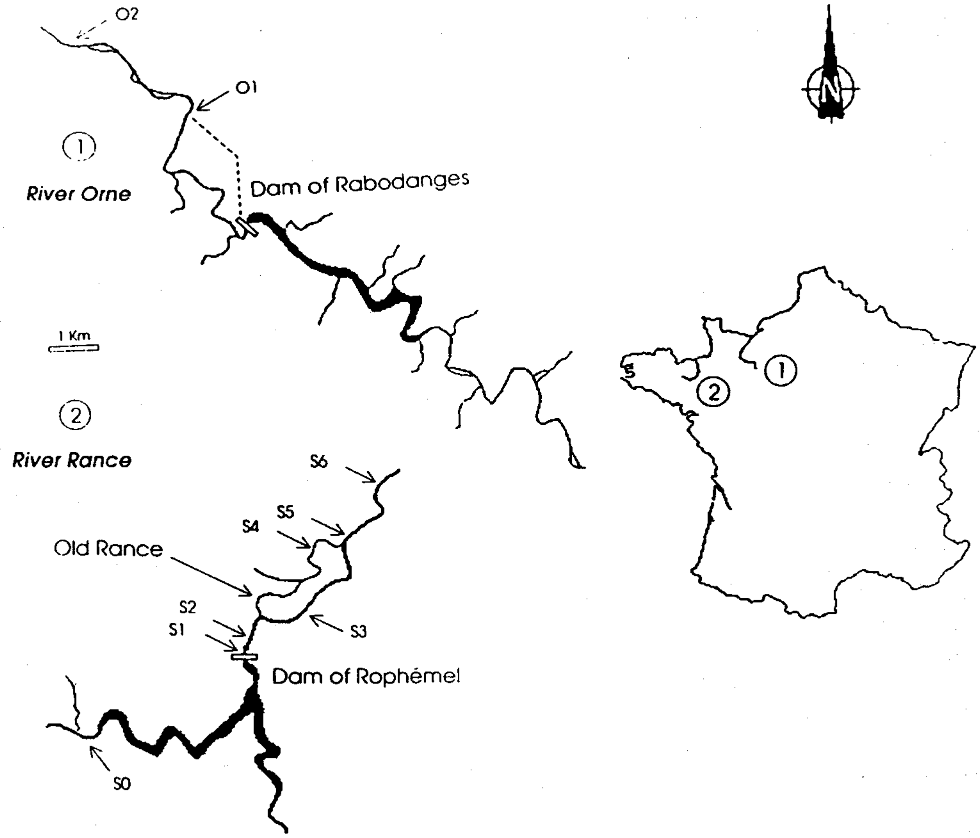

Figure 1 : Location of Rophémel and Rabodanges reservoirs and sampling sites

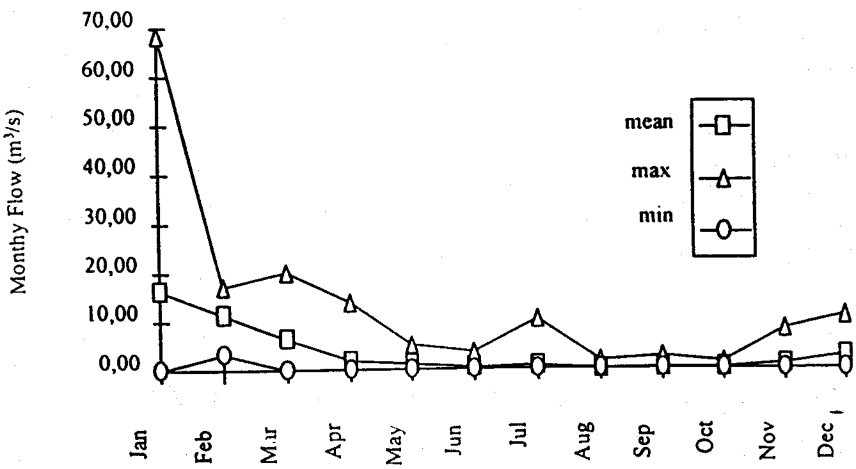

Figure 2 : Monthly flow pattern below Rophémel Dam for 1995

INTRODUCTION

The river macrophytes have been used for the assessment of water quality in rivers, but the different methods of the authors to collect data create comparison difficulties (Wiegleb, 1984). Moreover few studies investigate particular parts of the rivers, as the downstream stretches of the dams, while many downstream effects are described (Baxter, 1977). There is little knowledge in France about effects of hydroelectric dams on macrophytes, except some specific studies on biomass (Petitjean, 1981 and Khalanski *et al.*, 1990). Concerning impoundment, aquatic macrophytes are regarded as being a management problem from an engineering point of view, and many studies on macrophytes concern problems of weed control in, or below dams (Henriques, 1987). Regulatory works change completely aquatic environnements (Jongman, 1992) and comparison with floristic inventory before the regulation of river level and flow is the best way to assess their effects (Nilsson, 1978). When there are no previous data, the comparison of upstream-downstream sites remains the only possibility (Haury *et al*, 1996). Downstream effects of hydropower impoundment on flora were investigated by Rorslett *et al.* (1989) and De Jalon and Sanchez (1994). Irvine and Jowet (1987) showed that the hydraulic particularity of each site led them to adapt their methods of investigation. The non-natural habitats created by flow disturbance provide prime areas for opportunist species, which present particular responses (Rorslett, 1988). Some specific effects on river macrophytes can be assessed : this is the purpose of this paper.

Two hydroelectric impoundments were studied on two rivers. We focused on the general distribution of macrophytes around one of the two dams and compared the responses to the peaking flow conditions between the both. We set out to understand how the impoudment modifies the distribution of river macrophytes, and to assess the relative part of seasonal pattern versus effects of peaking flow management.

DESCRIPTION OF THE STUDY AREAS

General Features

The two dams of Rophémel and Rabodanges belong respectively to the River Rance and to the River Orne.
- The River Rance is located in Northern Brittany, in Western France. It joins to the English Channel between Dinard and St-Malo. Main underlain by siliceous rocks (e.g. granites), the River Rance crosses the calcareous "mer des Faluns" downstream from the hydroelectric dam of Rophémel. Locally, upstream from the dam, calcareous boulders were found in a zone described as homogeneous crystalline. The hydroelectric dam was built in 1938. We studied one representative site (S0) upstream from the dam. Different hydraulic conditions and distances downstream from the dam led us to choose six other sites (S1 to S6, see figure 1), whose particularities are summarised in Table 1.
- The River Orne (fig. 1), which flows in Basse-Normandie, is cut by the hydroelectric dam of Rabodanges just after leaving the calcareous Parisian Basin. The dam is located on granite at the entry of the Massif Armoricain. Two sites downstream from Rabodanges, O1 and O2, with peaking flow disturbance, were compared to the ones of Rophémel. This system was already described by Haury *et al.* (1996).

Flow Regulation

Rophémel is a reservoir of 5 million m^3, with an area of 80 ha and a maximum depth of 23 m.. The maximum monthly flow observed is 120 $m^3 \cdot s^{-1}$. The monthly flow pattern for 1995 is shown in figure 2. Maxima were obtained in January, and minima in August , for an average annual flow of 3.5 $m^3 \cdot s^{-1}$. The four turbines of the dam can discharge a maximal volume of 27 $m^3 \cdot s^{-1}$. Its minimal regulated regime is 0.9 $m^3 \cdot s^{-1}$.

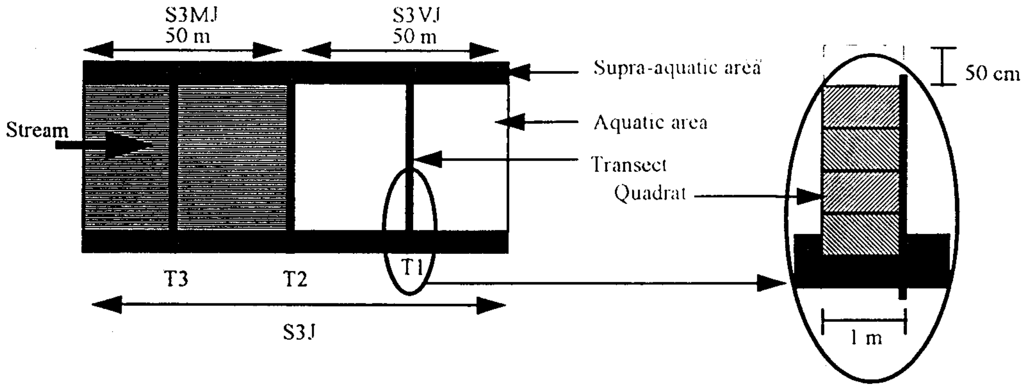

Figure 3 : Methods and sampling

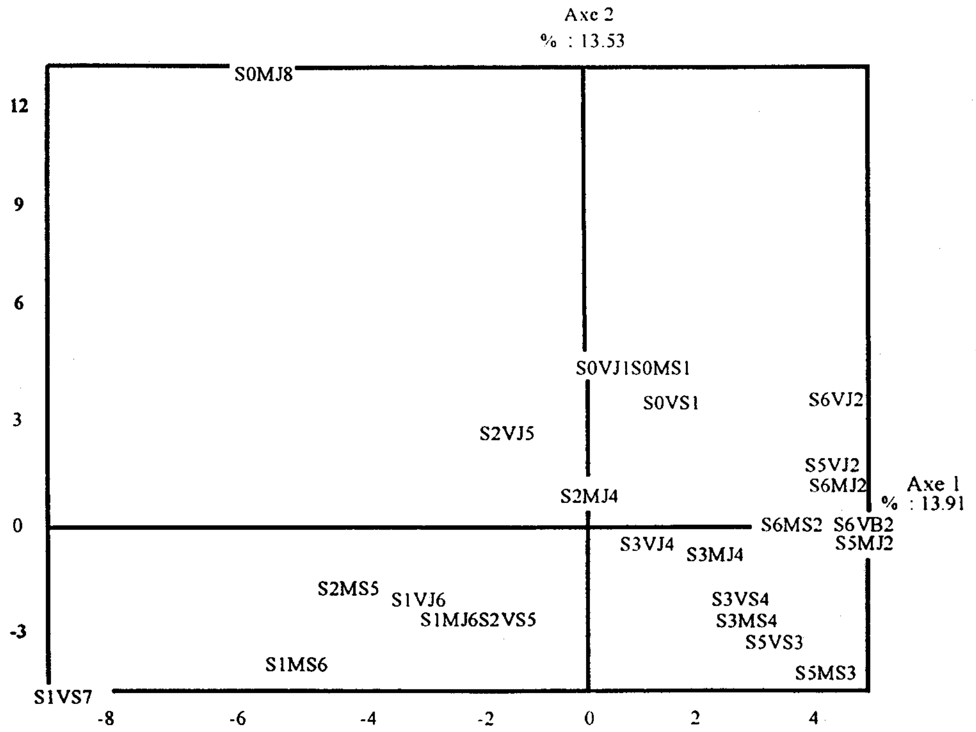

Figure 4 : F1 X F2 plan of PCA (24 individuals X 101 active variables. HCA, 8 classes. The coding is n°of site, upstream(M) or downstream(V)parts, month(J: june, S: september), and n°of classe of HCA.

The comparative study of the effects of peaking flow management at Rophémel and Rabodanges allows a better understanding of the response of river macrophytes to regime flow disturbance. The two sites on the River Orne, O1 and O2, are submitted to similar hydraulic conditions, 0,8 $m^3.s^{-1}$ (versus 0,9 at Rophémel at the regulated flow), and to a similar peaking flow regime. The main difference is the bigger frequency of disturbance at Rabodanges. While daily powering is recorded in Normandy, there were only 2 dates for August at Rophémel.

Table 1 : Characteristics of the sites around Rophémel

sites	S0M	S0V	S1M	S1V	S2M	S2V	S3M	S3V	S4M	S4V	S5M	S5V	S6M	S6V
DEPTH (cm)	26,4	36,0	28,4	32,3	38,6	29,0	47,0	22,3	6,0	6,1	26,2	29,8	33,1	17,5
WIDTH (cm)	635	835	1837	1500	1006	968	1073	837	364	354	876	1134	1580	1448
LIGHTING	3	3,5	4	3	3,5	4,5	4	4,5	2,5	4,5	5	4,5	4,5	5
CONDUCTIVITY	445,5	445,5	311,5	311,5	317	317	306,5	306,5	338	338	331	331	412	412
NO3 (mg/l)	27,5	27,5	9	9	13	13	15,5	15,5	21	21	18,5	18,5	20	20
PO4 (mg/l)	0,195	0,195	0,11	0,11	0,09	0,09	0,065	0,065	0,085	0,085	0,195	0,195	0,055	0,055
NH4 (mg/l)	0,025	0,025	0,885	0,885	0,2	0,2	0	0	0,01	0,01	0,005	0,005	0	0
% Silt	0,0	10,0	0,0	5,0	0,0	0,0	0,0	0,0	20,0	2,5	0,0	0,0	0,0	3,3
% Sand	1,5	6,5	2,5	0,0	0,0	0,0	0,0	0,0	10,0	2,3	2,5	2,5	2,5	2,1
% Gravel	6,7	17,5	5,0	5,0	7,5	8,8	2,5	4,5	7,5	7,3	10,0	12,5	10,8	7,1
% Stones	19,0	18,2	17,5	7,5	15,0	7,0	32,5	27,1	7,5	21,0	35,0	27,5	34,1	36,4
% Boulders	22,2	2,3	22,5	32,5	22,5	32,5	12,5	17,5	10,0	18,3	2,5	7,5	2,0	2,8
% Blocks	0,7	0,5	2,5	2,5	5,0	1,8	2,5	0,9	5,0	0,0	0,0	0,0	0,7	0,0
Nb of aquatic species	32	28	42	42	34	32	27	30	34	40	30	31	31	33
Nb of aquatic & supra-aquatic species	74	71	81	100	69	63	67	90	81	88	86	72	68	83

METHODOLOGY

Macrophyte Field Studies

Two methods were applied : floristic relevés and quadrat sampling (figure 3). The whole period of vegetation study began in April for the spring period and finished at the end of October for the autumn period. Macrophytes studied were species of spermaphyta (taxonomy from Tutin *et al.*, 1966 to 1980), species of bryophyta (taxonomy from Augier, 1966) and genus of algae (taxonomy from Bourrelly, 1966, 1970, 1981). Aquatic and supra-aquatic compartments were considered (Holmes and Whitton, 1977).

Floristic relevés were performed on two 50 meter sections for each site (S0 to S6) around the dam. The coding is n° of the site, upstream (M) or downstream (V) parts, month (J: June; S : September). Species cover was estimated by the same surveyor. Sampling periods were June and September. The data set had 28 individuals.

In order to test changes of flora in relation to season and flow disturbance, the survey of permanent places (quadrats) is necessary (Henry *et al*, 1996). The two extremities of the cross-sections (transects) defined the lines of study (Wolff *et al.* , 1989). Thus, for the 6 sites, sampling sets along the transects were established : each 50 cm, one meter for the line, determines the unit of study, the quadrat (Wright *et al.*, 1981). On each 0.5 m^2 unit, the cover of each species was estimated by the same observer. We used an adapted Braun-Blanquet cover scale to avoid under-representing small species (Everitt and Burkholder, 1991) : seven classes form the scale : 0 as absent, + as ≤ 1%, 1 as 2-10%, 2 as 11-25%, 3 as 26-50%, 4 as 51-75% and 5 as 76-100%. Between 50 to 90 quadrats on 2 or 3 transect-lines per site at 3 periods (May, August and October) were sampled, which led to 1026 quadrats.

Mesological Data

Physical data were obtained together with floristic reievés (depth, current velocity, bed substratum, lighting conditions), and with quadrat analysis (measure of depth at each point, frequency of bed substratum size, measure of current velocity with an electro-magnetic "Flo'Mate" on one transect per site per period). Water samples were collected in June and September; we analyzed 15 parameters, but only NH3, NO3, PO4, and conductivity were used here(table 1).

Data Analysis

We used multivariate analyses to assess the general structure of macrophyte distribution : Principal Component Analysis (P.C.A.) followed by Hierarchical Cluster Analysis (H.C.A.) with the SPAD-N software (Centre International de Statistique et d'Informatique Appliquées, 1992). To prevent from the non linear scale used for the vegetation cover in the quadrats method, when the scale of analysis is the site, results needed to be converted into cover percentages before the statistical analysis : 87,5% for the 75-100% cover, class 5, for example, (Barrat-Segrétain and Amoros, 1995).

RESULTS

Mesological And Macrophytic Variations Around The Dam

Chemical effects and river bed modifications

The main chemical effect is an increase of ammonium in S1. A great variability in conductivity appears in S0 due to flow variations and sewage from the nearby town of Caulnes; such variability does not occur downstream from the dam.

Riverbed modifications are obvious (table 1). S1 is large and not very deep. In S2, S3 and S5 and even in S6, deposition of coarse particles (stones and gravel) occurs near the banks within clumps of *Phalaris arundinacea* and *Oenanthe crocata* : these deposits spread over more than 70 % of the channel section. In S4, located in the former channel called "Old Rance", the flow is low, and deposits are mainly silt. Such a difference in shape of substrate particles corresponds with the observations of Henry *et al.* (1994).

Former channel of Rophémel (S4)

Presence of meadow species in the former channel (S4) was observed, with an increase in August according to the decrease of the flow regime. These species were *Agrostis stolonifera, Poa trivialis, Symphytum officinale*, which are tolerant to short-time aquatic conditions. One transect located on a riffle was out of water during August and October; most of its macrophyte cover was due to *Apium nodiflorum*. Particular aquatic species occurred as well, mainly eutrophic and slow-flowing macrophytes such as *Potamogeton panormitanus, Spirodela polyrhiza, Callitriche obtusangula.* As the floristic data were very different from those of the other sites, they were not used in multidimensional analyses.

General Response Of The Vegetation downstream from the dam

The Principal Component Analysis involving six stations (S0 to S6 except S4) and periods (24 individuals), and all aquatic and supra-aquatic macrophytes (101 taxa) assesses main structures. The variability is fourth-

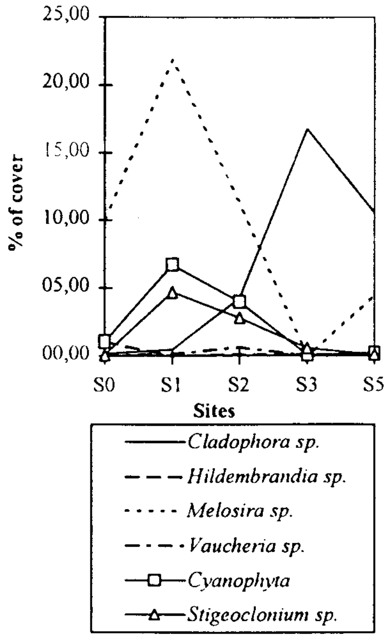

Figure 5 : % of cover of the main algae on the 5 sites

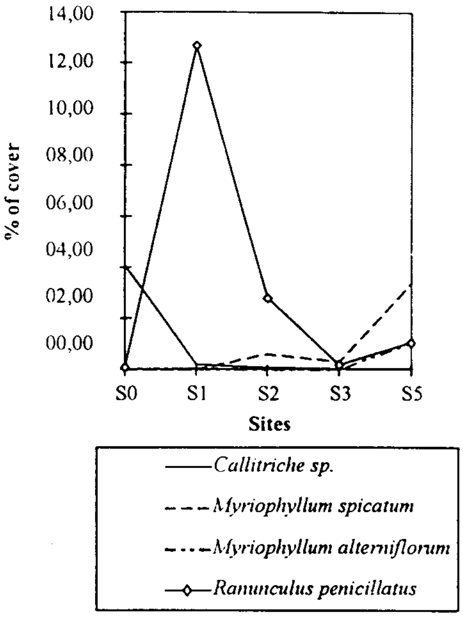

Figure 6 : % of cover of bryophyta and hydrophyta on the 5 sites

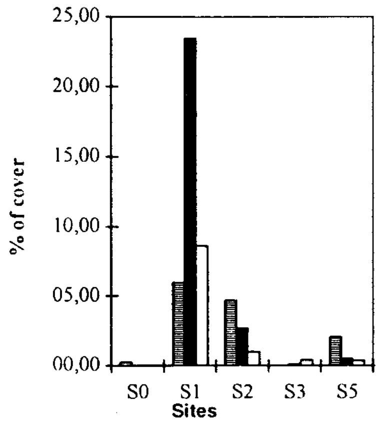

Figure 7 : seasonality of cover percentage of *Ranunculus penicillatus* on the 5 sites

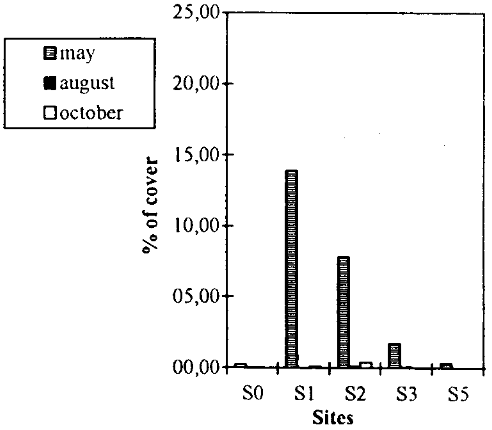

Figure 8 : seasonality of cover percentage of *Stigeoclonium sp.* on the 5 sites

dimensional (45,5 % of inertia). First axis, we have an opposition between species growing in the immediate area of the dam (*Callitriche sp.*, *Lemna minor*, *Nuphar lutea*, mosses as *Fontinalis antipyretica*, *Cinclidotus fontinaloïdes*, *Fissidens pusillus*, *Platyhypnidium rusciforme*) and most of the further downstream species characterising slow-flowing parts (*Potamogeton alpinus*, *P. perfoliatus*, *P. crispus*, *Elodea canadensis*, *Myriophyllum alterniflorum*, *Cladophora* sp.) : this axis assesses the downstream decreasing effect of the dam. The significance of the second axis is clearer, showing an opposition between species growing mainly upstream (*Porella pinnata*, *Hildembrandia* sp., *Lemanea* sp., *Apium inundatum*, *Callitriche platycarpa*) and downstream (*Ranunculus penicillatus*, *Myosotis scorpioides*) from the dam. The third axis cannot be explained simply. The fourth one distinguishes stagnant species such as *Nuphar lutea* and *Potamogeton natans* from bank species such as *Pellia epiphylla* and *Phalaris arundinacea*; it opposites S1 to S2. S1 has concreted banks. The distribution of individuals in the F1xF2 plane of P.C.A. and within the 8 classes (figure 4) obtained by a H.C.A. confirms the zoning downstream from the dam, the opposition between stations located upstream from, and just below the dam. Most downstream stations demonstrate macrophyte recovery, with a position of S6 close to S0. Data obtained with quadrats showed similar effects and precise seasonality.

Impacts Of Peaking Flow Management

Longitudinal zonation

The frequency of main algae (figure 5) showed different patterns in both percentage of cover and species composition. Upstream (S0), the filamentous diatom *Melosira* sp. was very frequent; in S1, its cover increased, and then decreased downstream from the dam. An inverse pattern appeared with *Cladophora* sp.. The third pattern showed by clumps of Cyanophyta and *Stigeoclonium* sp. corresponded to a locational increase below the dam and decreasing covers downstream. *Hildembrandia* disappeared between S0 and S1, and recovered in S5. The figure shows that a possible recuperation could occur, with a decrease of cover of algae in S5 where cover by algae is similar to that of the usptream S0.

The frequency of main bryophytes and hydrophytes (figure 6) also showed differences. Upstream from the dam, there was a typical community of Armorican rivers, with a co-dominance of *Callitriche* and *Ranunculus* species (Haury, 1985). In S1, the community is dominated (in May and August) by *Ranunculus* stands and *Callitriche* species such as *C. obtusangula*, *Leptodictyum riparium*, and is enriched with pond species such as *Nuphar lutea* and *Potamogeton natans*. Downstream from S1, an eutrophic and calcareous community appeared, with *Myriophyllum spicatum*, *Potamogeton alpinus*, *P. crispus*, *P. perfoliatus*. *Myriophyllum alterniflorum* grew in the same sites than *M. spicatum* in S5 and S6. *Fontinalis antipyretica* appeared to be opportunist in S2, where it benefited from the low cover by other macrophytes and a pebble substratum bed.

Seasonality and effects of peaking flow

Seasonality of cover percentages are presented for *Ranunculus penicillatus* and *Stigeoclonium* sp. (figures 7 and 8). A seasonal pattern of undisturbed populations of *R. penicillatus* appeared in S0, S2 to S6. But in S1, there was a delay between May and August before reaching its maximum cover.

Peaking Flow Bounded Sections Of Rabodanges.

When riverbed substrata are compared at the sites under peaking flow pressure, O1, O2, to S2, S3, S5 and S6, stones and boulders are the more common classes of granulometry for the two systems. In term of riverbed occupation, % of emergence could reach 70% on the river Rance, versus 5% in O1 and 14% in O2. It has a

direct impact on flow velocity, recorded in regulated flow with a mean of 0,12 $m{\cdot}s^{-1}$ in this 2 sites against 0,45 at Rophémel.
The vegetation described at Rabodanges using the same quadrat method by the same observer as at Rophémel, showed the significance of the cover of algae, 37% of *Cladophora*, 23% *of Hildembrandia*, 8% of *Melosira* and 6% of *Oscillatoria*. Main hydrophytes were *Ranunculus fluitans* and *Myriophyllum spicatum* covering respectively 2% and 1%, and *Potamogeton pectinatus*. *Phalaris arundinacea* at the bottom of the river banks covered 11% in O1 and 6% in O2. Bryophyta were represented by 5 main taxa, *Fontinalis antipyretica*, *Cinclidotus fontinaloïdes*, *Thamnium alopecurum*, *Brachythecium rivulare* and *Fissidens viridulus*, covering 3% altogether at O1.

DISCUSSION

Methodology

The use of complementary methods permitted us both to obtain comprehensive lists of macrophytes, with floristic surveys, and to assess seasonal and spatial changes, mainly with quadrats. For example, recuperation was shown by P.C.A. both in S6 and in S5 during autumn, but only suggested by analysis of algae cover within quadrats

Effects Of Water Quality

Many macrophytes growing in S1 indicate a degradation of water quality with fair ammonium contents (more than 0,10 mg/l $N\text{-}NH_4$). Clumps of Cyanophyta and *Stigeoclonium* sp. indicate pollution (Hawkes, 1964 *in* Whitton, 1970). The great development of *Cladophora* sp. is considered by Haslam (1987), Whitton (1970), Dodds and Gudder (1992) as an indication of eutrophication. In S2, S3, S5 and S6, such eutrophication is assessed by most of the algae and hydrophytes, while *Myriophyllum alterniflorum* indicates a recovery. Importance of phytobenthos in regulated rivers suggers Biggs (1987) to used it as indicator of water quality.

Effects Of Low Discharge

Siltation and colonisation by less hydrophilous species when discharge decreases have been previously described by Henzsey *et al.* (1991). Henry *et al.* (1994) observed also a replacement of aquatic species by terrestrial plants following a dry period. Thus, the use of the former channel when powering keeps it functional, and permits the permanency of a particular and interesting eutrophic vegetation.

Effects Of Peaking Flow Management

Modification of river-bed substratum

The flora invasion of the riverbed, especially in August when peaking flow was rare, promoted sediment retention in the " Old Rance "(S4). This phenomenon was described in the former channel of the River Rhône by Bornette and Large (1995). This site S4 has finer bed particles than the sites under the direct peaking flow pressure : such disturbed zones (S1, S2, S3, S5 and S6) have more stones and boulders and present a decreasing percentage of coarse particles.

Macrophyte vegetation

This gradual evolution of bed substratum according to the regime of disturbance and its consequences on the ecological response of flora have been already described by Henry *et al.* (1994). Three main effects were observed at Rophémel in these disturbed sites :

1 - A great increase of epilithic algae, for bed rocks permit the highest algae development (Biggs, 1987). Dense floating mats of filamentaneous green algae as *Spirogyra* sp., *Stigeoclonium* sp. and *Cladophora* sp. invading the downstream perturbed sites between boulders were also reported by Holmes and Whitton (1981). These mats could partly disappear after peaking flow. Interactions between the effect of ambient environmental conditions and various flow were studied by Peterson and Stevenson (1992) on algal communities. The importance of filamentous diatom *Melosira* sp. in S0 was probably in relation with fair contents of nitrates-Nitrogen in an intensive agricultural landscape, and downstream from the town of Caulnes. Its cover in S1 could be explained by a large river bed, a low current velocity without powering and much substratum offered by water-crownfeet; downstream, its decrease seems due to greater velocity.

2 - A great magnitude of bare substratum resulted from the regular disturbance on relative important silt, sand and gravel areas. This was in relation with the dominance of opportunist epilithic species such as *Cladophora* sp., obvious in the fast-flowing parts. The changes of cover and of the species composition of macrophytes could be due to such a disturbance (Henry *et al.*, 1994).

3 - Changes in relative importance of algae, hydrophytes and helophytes in the different sections were obvious too. Provided a great percentage of flamentous green algae indicated a disequilibrium (Haslam, 1987), at least in Armorican rivers where hydrophytes are usually dominant macrophytes, such changes in macrophyte vegetation are troublesome. We were surprised to find these developments of algae, with a large taxonomical richness (in regards to other sites studied in South-Eastern France - A. CAZAUBON, Univ. Marseille, pers. comm.). The relative importance of helophytes within the main channel indicates a discordance between a large bed and the erosive flow using it. When powering is frequent, helophytes cannot settle, while when it is scarce, colonization occurs, the macrophyte stands retain materials and stabilize deposits.

Comparison Of The Peaking Flow Effects Of Both Dams

In spite of comparable flow, three major differences appeared : 1 - the abundance of *Hildembrandia rivularis* at Rabodanges (23% of cover), 2 - a greater diversity of the bryophyta downstream for Rabodanges dam, 3 - the lack of emerged deposits in the channel of the River Orne.

These differences seem to be due to differing management between both dams : an almost daily frequency of peaking flow at Rabodanges, even in summer, does not allow the propagation of helophyte such as *Phalaris arundinacea*, which is responsible at Rophémel for the braided stretches within S2, S5 and S6. These daily peaking flows also have an impact on competition between algae : the partial destruction of the epilitic filamenteous green algae, permits the installation of *Hildembrandia rivularis* (Peterson and Stevenson, 1990 and 1992; Holmes and Whitton, 1981). The richness of the bryophyte flora in Rabodanges could be favoured by colder water and a weaker competition with algae, while downstream from Rophémel, warmer waters become unsuitable for bryophytes which cannot bear regulated flow for many days.

Macrophyte Communities And Species Biology

Model of Ranunculus penicillatus

Undisturbed pattern of *Ranunculus* development is similar to Haury's results (1985). The effects of peaking flow which delays the maximum extent could be due to an hydraulic cutting of the water-crownfoot in spring. It prevents its flowering, and increases its growth, as HAM *et al.* (1982) showed for R. *penicillatus* var. *calcareus*. Such an effect could explain the fair extent of *Ranunculus* downstream the dams (Petitjean, 1981).

Competition between species

The opportunist *Cladophora* sp. (Dodds and Grubber, 1992) presents an opposite pattern and seems to concur *Melosira* sp. : as it grows on stones and boulders, such an apparent concurrence could also be a change in favourable substratum. A concurrence and a succession between *Stigeoclonium* sp. and *Cladophora* sp. downstream pollution were described by Whitton (1970). *Hildembrandia* disappearance between S0 and S1 could be due to the dominant cover of other algae too (Holmes and Whitton, 1981).

CONCLUSION

Macrophyte vegetation changes between upstream and downstream reaches. The dams affect greatly the communities, but the effects of disturbance must be related to its frequency, to its intensity and to the distance from the dams. However, general trends appear. Opportunist species are favoured by disturbance, which explains the great importance of algae such as *Cladophora* sp.. Thus macro-algae appear as a particularly reactive compartment within macrophyte vegetation. The equilibrium between macro-algae, bryophyta, hydrophytes and helophytes seems to be a good bioindicator of the dam effects. Such disturbance due to peaking flow management should be studied with much more frequent observations, to assess recolonisation patterns, in the framework of the patch dynamic concept (Barrat-Segrétain and Amoros, 1995). Thus, improvement on methology should be necessary both to precise the mechanisms of recolonisation and to measure downstream recovery : hydroelectric impoudment gives field laboratory for such studies.

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ANTHROPOGENIC IMPACTS, HYDROLOGICAL VARIATIONS AND VEGETATION CHANGES IN STREAM CORRIDORS OF NORTHERN ALPS

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ABSTRACT

The valleys of the Alps are marked by increased anthropogenic impacts since the Neolithic epoch. Forest removals on hillslopes and alluvial terraces, in conjunction with climatic changes, induced by the Little Ice Age (15th-19th centuries), have caused an instability of the river beds triggered by frequent floodings and transits of great water-, energy- and matter-fluxes. Furthermore, the need for farmed soils and woods have increased the instability and contributed to the establishment of braided channel floodplains occupied since the 16th century by gravel deposits, sofwood forests and meadows. It has been shown how engineering-works have transformed the landscape during the last 200 years. Diking, channelization, warping, irrigation and drainage have modified the dispersion patterns of the fluxes and, new disturbances such as dredging, gravel mining, damming, filling, water pumping... have added their effects. Using archive documents and field data recorded for 20 years in the Rhône river hydrosystems, it is shown that the present landscape exhibits characteristics acquired from both old and recent land uses. The evolution trends are highlighted and allow to draw some lessons concerning dynamics and management of the systems. Finally, the concept of alluvial ecocomplexe is defined and it is underlined that it might be a useful tool in order to propose management planning with respect to the major functions of a stream-corridor: production (energy, crops, gravel...), conservation (biodiversity), circulation (transit of human, water, species...) and urbanization.

KEY WORDS: Landscape Ecology / floodplains / deforestation / water-flows / sediments / disturbance / land-use history / Alps Range / France

INTRODUCTION

Unlike North American catchments and floodplains, which were severely influenced by human disturbances only since the European settlement two centuries ago, (Foster, 1992), Alpine stream corridors have been marked by increased anthropogenic changes since the late Neolithic Period (Bravard, 1987; Moore and Evans, 1991). Evolutionary patterns of hydrology and vegetation are presented at three different time spans: (i) the millenium (since 5000 BP, when established the early Farming Communities), (ii) the century (since the end of the 18th century when civil engineering-works were developed) and (iii) the decade (after the development of the hydroelectric projects).

THE MAIN STEPS IN THE COLONIZATION IN THE EUROPEAN VALLEYS

The early periods: from the Late Neolithic Epoch to the Roman Period

For the last 40 years, Archeologists, Geographers and Biologists are interested by the early farming developments (Smith, 1995). The early farmers got settled by 5500/5000 BC on the most fertile loess and alluvial deposits, when Europe had a warmer (+2°C) climate than today. According to Archeologists (Bogucki, 1991), the farming developed along large rivers because of a favourable environment: presence of fertile soils that were easy to plough, water in the vicinity and the existence of gaps. It must be noted that in a heavily forested landscape, stream channels form what some have called "lines of weakness" These are belts between the dense vegetation of the bottomland communities (willows, alders) and the pristine climax hardwood forests (oaks, elms and limes) of the adjacent watershed. These zones are attractive for the early agriculturalists in that they are the places in which the process of clearance can be initiated more easily, from gaps created by beavers for example. Agriculturalists could spread over European lowlands from the Danubian region. They arrived with their crops (cereals) and herds (cattle, sheep, horse and goat) so, the forest clearing was carried out to create croplands and more importantly pasturelands. In order to establish large fodder meadows on alluvial terraces and floodplains, English lowlands (e.g. Thames, Trent and Severn rivers), were mainly cleared during the Iron Age and the Roman periods (Knight. and Howard, 1995; Peterken and Hughes, 1995). In the French Alps Range (Figure 1), the early agriculturalists probably came from the Mediterranean belt by 4500 BP (Bocquet, 1983). Then, five or six centuries later, arrived a second wave of colonization composed of two distinct populations (i) a culture (probably Danubian) coming from Switzerland where which established itself on the lake shores and (ii) a Mediterranean culture, coming from Provence and Languedoc regions which spread through the valleys on the warm and sunny slopes.

The Medieval Epoch

In continental Europe, farming and forest clearance progressively extended along the secondary valleys as the human population increased. The last farmed stream corridors were deforested during the Medieval Epoch by Cistercian and Benedictan monks. Many internal valleys where occupied from only a varying period between 600 and 1250 (Peterken and Hughes 1995). According to Palaeoenvironmentalists (Falinski, 1986) two levels must be distinguished in the Alps Range valleys (i) a lower zone (corresponding to the piedmont) exhibiting evidences of clearance before the 10th century and (ii) a upper zone (corresponding to the upper stretch of large rivers and to the tributaries) first poorly marked by cultivation, then highly deforested between the 10th and 17th centuries.

<u>The last land clearings</u>

In France and Piemont-Sardinia kingdoms, during the 17th century, some laws were enacted by governments in order to protect Alpine forests. Unfortunately, the deforestation started again from the end of the 18th century. The increased population in the countryside explained great needs for pastures, arable soils and wood (construction and fuel). The last alluvial primary hardwood forests, belonging to states, parishes and rich landowners (clergy, nobility) disappeared during the French Revolution period. Thus, for example, in the Isère river valley, three forests (covering more than 3000 ha) spreading over alluvial terraces and floodplain along the "France-Savoie" boundary, were cut down and replaced by croplands (Pautou and Girel, 1994)

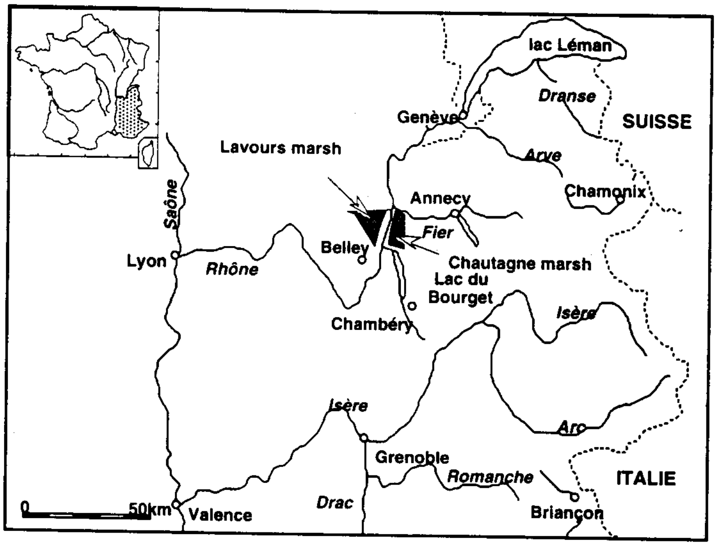


Figure 1: The Rhône river watershed

CLIMATIC CHANGES, HYDROLOGY AND FOREST REMOVAL

<u>Impacts of the Neolithic deforestations on hydrology and alluvial landscapes</u>

In England, the Neolithic clearances and farmings (tillage) caused the establishment of valley mires (Fig.2). Clay sedimentations and water table raisings were recorded in the bottomland. They triggered the initiation of peat-forming processes in the large wetlands which were established over the alluvial deposits (Moore and Evans, 1991). The presence of charcoal added to the high rate of alder and mire plants pollens in the clayed sediment are evidences showing the clearance by fire and the shift from a hardwood forest (oak, elm and lime) to a wet tall-grass prairie (sedges, rushes) and softwood communities (alders, poplars, willows). In the Alps Range, along the upper Rhône river, the Neolithic deforestations of the hillslopes and terraces have probably caused a filling of the alluvial floor and a rising of the water level in the Lac du Bourget. The presence of a deep peat layer (5 to 8 metres covering coarse alluvial deposits) in the Lavours and Chautagne marshes (Fig.1) as well as the discovery of several Neolithic housings lying under 5 metres of water near the lake shores are explained by this hydrological change (Bravard, 1987)

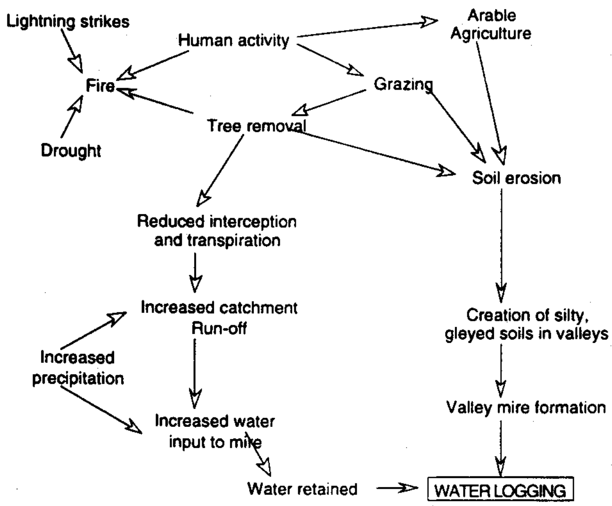


Figure 2: Scheme to show relationship between humain impact on a landscape and waterlogging of valleys leading to wetlands formation (modified after Moore and Evans 1991)

<u>The succession of warm/dry and cold/wet periods</u>

Dendroclimatological studies (Bradley and Jones, 1992; Lamb, 1995) have allowed to highlight the possible succession of warm and cold periods. In Europe, for example, the Warm Medieval Period (1000-1300) characterized by warm and dry summers, soft winters, low rainfalls came before the Little Ice Age (1570-1850) which was by contrast, characterized by soft summers, cold winters, high rainfalls and deep snow covers. The latter was marked by an annual temperature 1°C lower than today and a high frequency of westerly surface winds which tended to give rainfall. It has been shown by the presence of charcoal and ash layers in sediments that warm periods were marked by frequent fires These probably occured during summer storms (lightnings) and were favoured by the presence of large stocks of fuel (dead trees, dry scrub). After the 1988 fires, in the Yellowstone Park (USA), it has been shown (Meyer *et al.*, 1992) that the climatic fluctuations could have indirectly affected the water-flow, the matter-flow and, hence, the geomorphological response of the streams in mountain environments. After intense fires, the sediment transport was enhanced. Ashes and charcoal were swept first by stormy rainfalls, carried down to the alluvial fans and rapidly buried by sediments (abundant analogous deposits were found in older alluvial fan sequences, showing the frequency of past fire-related sedimentation events in Yellowstone mountains). Therefore, alluvial fans aggraded during periods of frequent fire; downstream, the sediment transport was reduced and the rivers were characterized by narrow active floodplains, incising the alluvium. Factors which inhibit local redeposition (such as transport predominantly in suspension) or constrain the channel laterally (such as wooded vegetation and cohesive sediment) favoured the meandering processes. Vegetation was constituted mainly of mature communities growing on old terraces (oak, elm, lime forests) and narrow belts of softwood (willows, alders, poplars) along the banks. During the cold/wet following period, high rainfall and high snowmelt probably caused erosion through fan channel incision and fan toe undercutting. Downstream, the floodplains became widened by strong aggradations and frequent floodings. The new environmental conditions favoured the metamorphosis of the channel patterns, so that braided and anastomosed models replaced the meandering system. Woody terraces were destroyed and hence, coarse deposits with pioneer Buckthorn/Tamarisk communities (*Hippophae rhamnoides, Myricaria germanica, Salix eleagnos* bushes), White Willow thickets and Grey Alder woodlands took the place (Pautou and Girel, 1994; Pautou, *et al.*, 1996). In the Northern Alps region, three periods of aggradation have been identified in the Isère valley (i) the Iron Age,(ii) the post-Roman period and (iii) the Little Ice Age (Salvador, 1991); each one is known as cold and wet. Along Rhine river tributaries old housing dated from the 14th century were discovered buried under 5 metres of alluvial material (Gauckler, 1868). It is currently thought (thanks to documentary evidences) that, before the Little Ice Age, Alpine valleys such as the river Drac and the river Isère near Grenoble corresponded to anastomosed and meandering channels occupied by hardwood forests, orchards, fodder meadows and various crops (Salvador, 1991).

Consequences of Little Ice Age climatic and anthropogenic factors

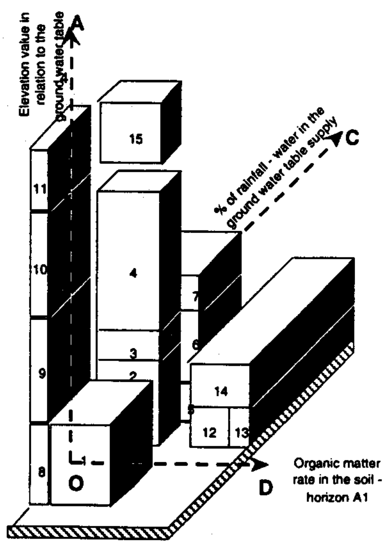
The climatic change of the Little Ice Age has been proved by documentary evidences concerning rainfall, snow cover, flooding or plant and animal biogeographical areas. For example, it has been shown that the Rhone river was marked by frequent and severe floodings during spring (linked to snowmelt) and autumn (high rainfalls); furthermore it was frequently spoken of streams such as Rhône, Isère... carrying ice blocks in winter (Pichard, 1995). Research in vegetation history have shown that the European forest area had strongly decreased before 1650, stayed steady to the end of the 18th century then decreased rapidly between 1800 and 1825 and finally, more slowly by the second half of the 19th century when reafforestation operations began (Peterken, 1977). Various data supplied by "experimental research watersheds" (Freedman 1995) have highlighted that vegetation acted at three levels (i) at the microclimatic level (the trees intercept rainfalls, control snowmelt rates and wind movements), (ii) at the soil level (vegetation anchors the deposits, favours porosity, controls the biological activity and the organic matter) and (iii) at the physiological level (vegetation eliminates water through evapotranspiration processes). In the Alps Range, deforestation was produced by strong needs for farming areas and various resources: wood (fuel), barks (tanin for leather industry), twigs and litter (manure); furthermore, the permanent presence of cattle, sheep and goat prevented the saplings regrowth. On the alpine grasslands, sheep folds were so numerous that they caused overgrazing and trampling. Because of these major perturbations erosion and water run-off were highly increased.
At the end of the 18th century, the upper Rhône river and tributaries floodplains were characterized by braided and anastomoded channels. The islands constituted mainly of coarse material were occupied by pastures and thickets of softwood communities.

Vegetation in Alpine stream corridors at the beginning of 19th century

Along upper catchments, floodplains were subjected to catastrophic and frequent overfloodings triggered by great coarse material and water flow inputs. The raising processes of the valley floor have replaced the entrenchment processes and, as a result, the instability of the geomorphological landforms significantly increased. According to the slope value, a shift from a meandering model to an anastomosed or to a braided model occured and occupied the whole floodplain. Therefore, a large array of plant communities, from aquatic to terrestrial, was probably represented. These communities could be known by reference to upper Rhône river sections described before the construction of hydroelectric dams and reservoirs (Pautou, *et al.* 1979). They were organized in a 3rd dimensions space corresponding to three variables: (i) a vertical dimension (water table levels), (ii) a longitudinal dimension (water supply regimes) and (iii) a transversal dimension (sedimentation and organic matter gradients). The Figure 3 (Pautou, *et al.*, 1996) illustrates the biodiversity of wooded communities; in the same way, it could be possible to place pioneer Buckthorn/Tamarix bushes (*Hippophae rhamnoides, Myricaria germanica, ...*) and herbaceous communities (*Phalaris arundinacea, Typha minima, ...*) characterizing rejuvenated habitats (new channels and islands). Nevertheless, it seems necessary to underline that during the 18th and 19th centuries, anthropogenic factors prevented the natural succession. Farmers tried to cultivate the silty deposits and used islands as pastureland. In addition to, they regularly cleared the woodlands (fuel and fodder production) and mowed sedges in the river mires (to get greater volume of manure), so that the most developed communities were mainly the 5 to 7 years old grey alder woodlands.

TOWARD THE CONTROL OF THE FLUXES: THE 19TH CENTURY ENGINEERING-WORKS

At the beginning of the 19th century, alluvial landscapes of many alpine stream corridors were as follow: (i) The active channels and the poorly wooded river islands sprawled over the whole valley; arable soils and grasslands were threatened by floodings which often buried the crops under the alluvium; (ii) on the margins, the backswamps and the former channels were flooded by over bank flows and water table risings; so they constituted numerous sources of malaria (Pautou, Girel *et al.*, 1995) In order to struggle against poverty and migration, eradicate malaria, and develop farming and modern roads, governments of Alpine countries decided to control the flow in the valleys areas.



The origin corresponds to the lines of highest flow velocities (active main channels); the pole "D" corresponds to the soils having the highest organic matter contents; the pole "A" corresponds to the deepest water table levels; the pole "C" corresponds to the backswamps locations.

1- various willow (Salix triandra, S. Purpurea, S. alba...) bushes
2- White Willow woodlands (Salix alba with Impatiens glandulifera in the herbaceous stratum).
3- Grey Alder woodland (Alnus incana with Equisetum hiemale in the herbaceous stratum)
4- Oak-Ash forest (Quercus robur,-Fraxinus excelsior presence of Populus nigra; Aegopodium podagraria in the herbaceous stratum)
5- Salix triandra bushes; White Willow woodland (Salix alba, with Carex pendula in the herbaceous stratum)
6- Elm woodland (Ulmus minor)
7- Oak-Ash forest (Quercus robur -Fraxinus excelsior + Populus alba; Carex pendula in the herbaceous stratum)
8- Salix eleagnos thickets
9- Black Poplar-Ash open communities (Populus nigra, Fraxinus excelsior)
10- Black Poplar-Ash woodlands (Populus nigra-Fraxinus excelsior with a dense understorey of thorny shrubs)
11- Locust open woodland (Robinia pseudacacia)
12- Black Alder woodland (Alnus glutinosa with Carex elata in the herbaceous stratum); Black Alder-Ash forest (Alnus glutinosa-Fraxinus excelsior)
13- Frangula alnus thickets
14- Ash-Black Alder forest (Fraxinus excelsior-Alnus glutinosa with Carex remota in the herbaceous stratum); Ash forest (Fraxinus excelsior with Allium ursinum in the herbaceous stratum)
15- Oak-Hornbeam forest (Quercus robur, Carpinus betulus)
N.B: The level (in metres) of the ground water table is precised along the vertical axis "OA"; the zero means that the roof of the water table showed on the soil surface for the major part of the year.

Figure 3: position of the major alluvial woody communities in braided / anastomosed hydrosystems.
(After PAUTOU *et al.* 1996)

The control of water and energy fluxes: diking/channelization

The aims of the procedure were bank protection and flooding limitation. The technics consisted in making an artificial bed, deep and large enough to convey highflows downstream (Figure 4). Diking operations deleted the connectivities, so that the former braided channels were isolated from the channelized river in which are focused the fluxes. Hence, the braided floodplain became significantly reduced to the narrow band located between the levees. In the case of high and unsinkable levees, vegetation could only grow upon the alternating bars (see Fig. 5) where new dynamics processes occured. During high flow periods, gravel bars were destroyed at the upstream end by erosion processes and constructed at the downstream end by aggradation processes.Thus, there was a permanent rejuvenation of the habitats; vegetation was characterized by pioneer communities (*Phalaris arundinacea, Calamagrostis littorea* on sandy deposits) and young softwood brushes (*Alnus incana, Salix triandra, Salix alba, Populus nigra* on highest levels). In the case of the so-called "golene system" (Fig. 4), the floodplain was less reduced. Sedimentation basins located between the dikes allowed the trapping of alluvial material during highflow periods. The raised deposits were colonized by reedmaces, reeds, sedges (*Typha minima, T. latifolia, Phragmites australis, Carex gracilis, C. elata*)...which could increase the rate of trapped sediments. Once the ground was raised enough, willows and poplars were planted. The aim of the operation was to constitute artificial banks supporting wet meadows or arable soils. After diking development, on the protected areas, vegetation was no longer influenced by floods, so the main part of the water table was supplied by rainfalls. Consequently, Alpine river species linked to fresh and oxygenated water, such as *Salix eleagnos, Salix triandra, Salix daphnoides, Alnus incana, Myricaria germanica, Hippophae rhamnoides...* were replaced by backswamp species such as *Alnus glutinosa, Frangula alnus, Salix cinerea, Viburnum opulus* or*Betula verrucosa*

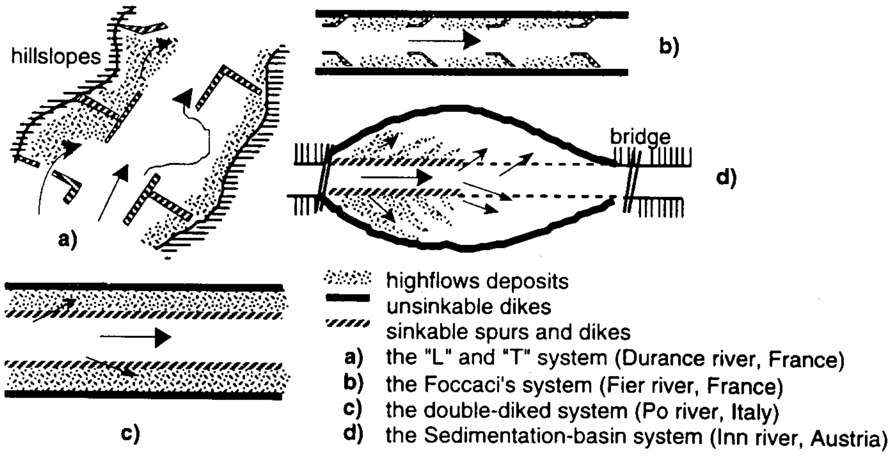


Figure 4: Four methods used in the Alps to channelized braided rivers
(after Girel 1994a ; 1994b)

The control of matter-fluxes: warping

One of the goals of the channelization was the reclamation of large areas for cultivation. Braided and anastomosed models are usually linked to the establishment of coarse material deposits and numerous channels which are not really fertile or easily farmed. For this reason it became necessary to use various technics, wellknown one millenium ago in the Bolivian Andes (Zimmerer, 1995), widely developed in Tuscany since the 16th century(Alexander, 1984) and currently used in the European floodplains since the beginning of the 19th century (Girel, 1994a; Girel, 1994b). The so-called "warping" technics consisted in using the river silty water flows to fill and to raise the coarse alluvial deposits. The area to reclaim was divided into warping basins by gravel levees. Decantation processes were favoured by a slow movement of the silty water which flowed from upstream to downstream through the basins (Fig.5). After several years, channels were filled and coarse alluvial deposits were covered by a layer of fine material (fine sand, silt and clay). It is important to underline that new soil parameters occured after warping operations. These were depended on (i) the duration of the operation (a long-lasting warping produced a deep layer), (ii) the used warping method (simultaneous feeding or successive discharges, see Fig.5, produced different types of soil) and (iii) the location of the basin (the grain size was depending on the place of the basin in the warping system: heavy material such as fine sand were deposited first). It has been shown (Girel, 1994a) that the new environmental conditions (grain size gradients, depth of the fine deposit) plaid a major role for the constitution of the alluvial landscape. Many basins were cultivated; some were plowed but many other were colonized by spontaneous vegetation and were changed into wetlands.

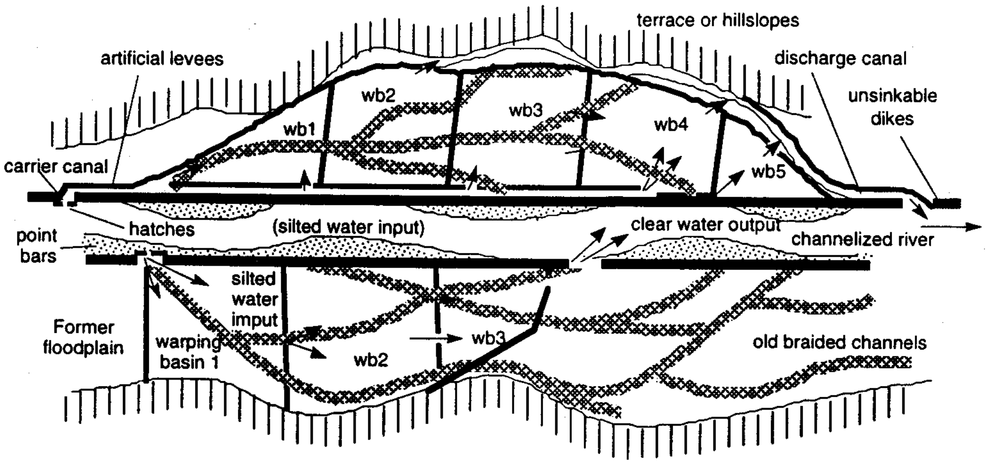


Figure 5: The new channelized river and the warping systems (19th century)
- on the left bank : a "simultaneous feeding" system
- on the right bank : a "successive discharges" system

<u>The control of the nutrient flow: irrigation</u>

During the 19th century, irrigation systems using urban waste water were currently developed in the European floodplains. These technics were wellknown since the 17th century in South England, Sweden (Emanuelsson and Möller, 1990) and Lombardy (Soresi, 1914). Irrigation canals provided a fertilizing flow to crops and meadows while drainage networks collected purified water downstream (Fig.6). In irrigated cultivated - meadows (ray grass, clovers) the nutrient flux allowed to get 3 to 5 yields per year, even during the cold season. Irrigation processes which were used to treat sewage for several decades (Ronna 1890) likely have induced change in alluvial soils (filling of the superficial layer by organic matter) and in the plant composition. High nitrate and phosphate contents favoured some grasses such as *Lolium perenne* or *Dactylis glomerata*, and clovers at the expense of oligotrophic species such as *Ononis repens, Polygala vulgare, Succisa pratensis, Onobrychis sativa, Fritillaria meleagris*,... It has been shown (Broyer and Prudhomme, 1995) that fertilization has caused a high decrease in biodiversity of flooded meadows (7 species were present in a fertilized meadow, 40 in a grazed natural meadow, 70 to 95 in a mown natural meadow).

<u>The control of the water table depth: drainage</u>

This operation aiming to lower the ground water table was realized by two different periods along European valleys. The first one was developed since the second half of the 19th century. It is the so-called "underdraining" which consisted in putting tile-drains (0.60 to 0.80 m deep) in the ground (Phillips, 1989). Its role was the elimination of surplus water (seeping from hillslopes and dikes) and the improving of waterlogged soils resulting from warping. Drainage allowed to increase productivity of meadows and crops. Its impact on plant communities was a decrease of the biodiversity: wet pastures beeing replaced by fertilized mown meadows.

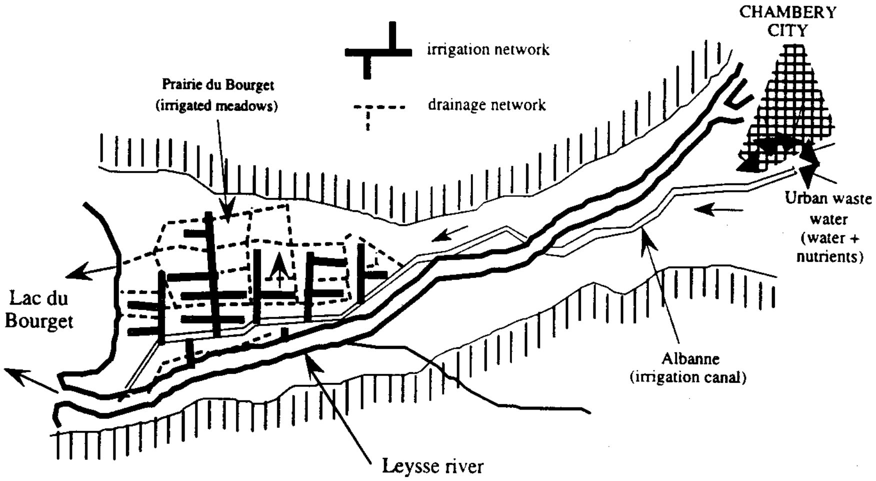


Figure 6: Irrigation / Fertilization of the "Prairie du Bourget" downstream from Chambery (1780 - 1900)

The second drainage period was recent (1970 to 1980). It was performed in order to drop significantly the water tables. Canals and ditches networks were dug in wetlands; the goal of the operation was to increase the arable soil area. Many warping basins occupied before by wet bushes and meadows were cleared and changed into maize crops. This drastic drainage influenced the adjacent marshlands. Significant water table lowering and eutrophication processes were recorded; they were characterized by the development of nitratophilous species (such as *Urtica dioïca, Solidago canadensis, Clematis vitalba, Galeopsis tetrahit*) and by a decrease in the biodiversity. It has been shown that drainage operations had decreased highly the biodiversity of marshes since the 19th century: 123 species (to 526) have disappeared in English valley mires (Mountford, 1994) and 115 (to 417) have disappeared in the upper Rhône valley mires (Giugni, 1985)

EVOLUTION TRENDS IN THE ALPS RANGE VALLEYS

The disturbances induced by hydroelectric power development, dredging and gravel mining

Since the 1950's, the Rhône river and its main tributaries are impounded by damming along their upper course. The impacts downstream from hydroelectric reservoirs are linked to change caused to the water and sediment transfers. The discharge regimes of the streams are changed; the flow is regulated during the year by the storage in the reservoirs. Nevertheless, flooding periods may be marked by highflows in the channelized river; furthermore highflows may also be produced at any moment when great volumes are released by the dams. The sediment load decreased because of reafforestation of the drainage basins and because of trapping in reservoirs. Nowadays, the river load downstream from reservoirs is constituted mainly of fine sediments; this induces geomorphological change such as fine deposits on the bars. Since the 1970s, dredging and gravel mining have removed a great volume of alluvial material; this volume was greater than the volume which could be brought by the river. Consequently, there was creation of pits which were filled by a regressive erosion of the bed floors.

Impacts on vegetation

Dams and reservoirs which trap sediment and store water, in conjunction with watershed reafforestation and gravel extractions, are indirectly or directly rsponsible for erosion processes of the bed floors. Hence, entrenchment processes (incision) are initiated and cause various perturbations on the environmental conditions. Bridges and embankments are threatened by undermining. The processes of destruction/construction which allowed the plant dynamics on the bars are no longer produced; the destruction of willows/grey alder bushes occurs rarely, so that, the succession does not stop at the shrubby stages but carries on to the woodland stages. For 10 years, it is noted the establishment in by-passed channels and river islands of trees which can be uprooted by undermining processes during flooding periods. Fine material deposits and "tidal zones" produced daily by dam releases contitute new habitats. It has been recorded the spreading of tall-grass communities (*Typha minima, Calamagrostis littorea, Agrostis stolonifera, Phalaris arundinacea...*) on the Isère river bars. The entrenchment processes in conjunction with pumping operations (for irrigation and drinking water) have induced a lowering of the adjacent water tables (e.g. more than 1.50 m along the Drac river near Grenoble). The channelized stream plays the role of a draining canal. Subsequently, wetland areas are replaced by mesic thickets; therefore, farmed areas and fragmentation increase. Alluvial hardwood forests (Oak and Elm) are invaded by a mesic vegetation coming from the hillslopes. Trees such as Hornbeam (*Carpinus betulus*), maples (*Acer pseudoplatanus, A. platanoides, A. campestre*), Locust (*Robinia pseudacacia*), Walnut (*Juglans regia*)... become established. In wet woodlands, Ash (*Fraxinus excelsior*) take the place of alders (*Alnus glutinosa, A. incana*). On permeable and dry deposits, Black Poplar open woodlands are colonized by characteristics of xeric communities such as *Quercus pubescens, Acer monspessulanum, Pistacia terebinthus* and by alien species such as *Buddleja japonica*.. The Alpine component of the Flora (*Salix eleagnos, Hippophae rhamnoides, Salix daphnoides, Alnus incana...*) is slowly replaced by a Supramediterranean component, hence, the river Rhône corridor and its main tributaries valleys seem to have now many similarities with the Atlantico-European valleys such as the river Garonne for example.

DISCUSSION AND CONCLUSION

Presently, the landscape mosaic of the Northern Alps stream corridors results from cumulated impacts induced by anthropogenic perturbations, both old and recent. It constitutes an ecocomplexe (Pautou *et al.*, 1996), in other words a set of ecosystems (=subsystems) which are interdependant in the way of hierarchized conduits (main channels, secondary channels, tributaries, canals, ditches...). Water, energy, nutrients, sediments, organic matter, and diaspores flow in these conduits which are also connected to reservoirs (lakes, ponds, water-tables, humus layers...). A floodplain and its adjacent areas constitute an ecocomplexe that the structure, the functioning and the dynamics depend on large gradients of hydrology and energy. The specificity and the biodiversity of the ecocomplexe are hight because of a great diversity of hydrological situations linked to the spatio-temporal patterns. In the landscape, the ecocomplexe can be highlighted and delineated by spontaneous vegetation, crops and other land-uses. Consequently, it can be easily mapped. The ecocomplexe allows to propose predictive scenarios (Pautou and Girel, 1992) concerning the responses of the various compartments in case of catastrophic events such as a 50 or a 100-year flooding. It is easy to take into account of new land-uses (such as waterproofed and filled areas created by industrial activity and urbanization), reclaimed wetlands (which can no longer store floods), or barriers (such as highways which can diverte the flows to particular sites) in the development schedules. The catastrophic impacts of floodings which affected many French watersheds for the last years are not only explained by rainfalls. The concentration of flows on reduced areas during a short time could be explained by a bad management of the drainage basin. The ecocomplexe may be a useful tool allowing to propose simulation patterns taking into account of hydrological parameters.

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HYDROBIOLOGICAL ASPECTS OF IMPOUNDING AT PETIT SAUT (FRENCH GUIANA)

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ABSTRACT

Petit Saut dam was built in an equatorial zone 65 km from the mouth of the Sinnamary river. It created a 3550 hm^3 reservoir that floods 310 km^2 of primary forest; maximum depth is 35 m. Impounding took 18 months, from January 1994 to July 1995. The time required on average to renew the reservoir water is of the order of 5 months. Air temperatures remain constant year-round in the area (28-30°) and the differences between the seasons mainly concern rainfall: rainy season from January to July, dry season from July to December. In the very first months after impounding, oligomictic conditions developed with a well-oxygenated epilimnion 1 to 3 m thick and a totally anoxic hypolimnion. This pronounced stratification was compounded by a longitudinal gradient in the epilimnion that creates a transition zone between river and reservoir. On the other hand, no pronounced transversal gradient has developed. From a biological standpoint, no shore zone has been detected.Detritivorous Bosminidae cladocerans colonised the reservoir before the phytoplankton developed. The lake ecosystem matured with a succession of ever-larger dominant zooplankton groups: Bosminidae, followed by Daphnidae, Cyclopidae copepoda, and Diaptomidae. The last two families accompanied the development of the phytoplankton. Despite great diversity, rotifera never represented more than a tiny portion of the zooplankton biomass. On the other hand, a single species of reputedly benthic ostracoda (*Physiocypria affinis*) dominated the zooplankton on several occasions. Chaoboridae diptera progressively developed in the mass of water. It was only toward the end of impounding, and in the epilimnion, that the zoobenthos (diptera, tricoptera, oligochaeta) colonised the plant supports provided by the submerged forest. The abundance of food provided by the development of the zooplankton and then the zoobenthos encouraged a multitude of fish fry, concentrated near the surface. Oxygen in the mass of water comes almost exclusively from photosynthesis. Because of this, a series of rainy spells caused brief reductions in the epilimnion's thickness by mixing the different layers. Those fleeting but difficult conditions resulted in a high death rate for the zooplankton, which developed anew each time in the space of a few weeks, with the same succession of species. Presently, there is a slow trend towards improved living conditions in the mass of water, manifested by a thickening of the oxygenated surface layer at the same time as the reduced element content (especially methane) stabilises in the bottom of the reservoir.

KEY-WORDS: Hydroelectric dam / Impounding / Reservoir / Regulated river / Tropical zone / Invertebrate / Zooplancton / Biodiversity / Distribution / Food resource

I. THE SINNAMARY RIVER BEFORE IMPOUNDING

The Sinnamary river is 250 km long and springs up in the centre of French Guiana at an altitude of 125 m (figure 1). Its catchment area stretches over 7000 km² of crystalline formations overgrown by uninhabited primary forest. Hydrology in the area is marked by mean annual flow of 260 m^3/s, with a high water period from April to June in the great rainy season (flow ≈ 400 m^3/s) and a low water period from September to November (75 m^3/s) in the great dry season. In general slopes are gentle; a longitudinal profile of the area shows sections with rapids (called "sauts") and sections of very gentle slopes (the "entre sauts") in succession until the Petit Saut site where the dam was built, 35 km as the crow flies from Sinnamary village, close to the river mouth.

The water's physical and chemical characteristics (Richard and Horeau, 1996) change little along its course. The water is warm (26°C), slightly acid (pH 6-6.3), with a low mineral content (24 µS/cm), well-oxygenated (7 to 8 mg/l), turbid (13 NTU), lacking in transparency (secchi 0.50 m), poor in suspended matter (5 to 25 mg/l), essentially composed of organic matter (500-800 µg C/l) and poor in nutrient salts (nitrates 15-30 µ mol N/l, orthophosphates 0.2-6 µ mol P/l).

Aquatic invertebrates that were monitored for the three years preceding impounding seem fairly limited in numbers and it is difficult to assess how diverse they are. In fact, this type of fauna has still not been thoroughly researched in South America and there are still thorny taxonomic problems, despite the invaluable work of several specialists. However, there seems to be a wealth of different species in the groups that have been most thoroughly investigated, i.e. the rotifera, cladocerans, ephemerals and odonates, of which 98, 43, 36 and 76 species or taxona respectively have been identified to date.

They are divided in strictly delimited zones in relation with the strong differentiation between the 2 major types of biotope: the rapids and stretches between them ("sauts" and "entre sauts") are populated by insect larva, while the temporarily submerged forest and slow-flowing creeks are also rich in microcrustaceans and insect larva. At the bottom of the "entre sauts" live decapod crustaceans and in their sand live harpacticoid copepoda. The mass of water itself carries small invertebrates of which many come from other aquatic environments (creeks, submerged forest, marshes, etc.) and from both the land and the gallery forest. The density of the aquatic invertebrates population depends on flow variations and on the activity of predatory fish. The highest figures are recorded in the dry season, except for cladoceran crustaceans.

- The fish caught in the rapids mostly eat aquatic invertebrates, while terrestrial invertebrates that fall into the water are a major food supply for the fish caught in the "entre sauts" and the creeks (Horeau et al., 1996).

- The upstream part of the reservoir comprises zones of submerged forest with a wealth of microcrustaceans (rotifera, cladocerans, copepoda) and zones of stagnant water that are likely to contribute to the reservoir populations through drifting.

II. RESERVOIR IMPOUNDING

1. The Petit Saut Hydroelectric Scheme

The Petit Saut dam stands 35 m high and 740 m long. A 20 m high cofferdam built 150 m upstream from the water intakes prevents water from the reservoir bottom flowing directly to the turbines.

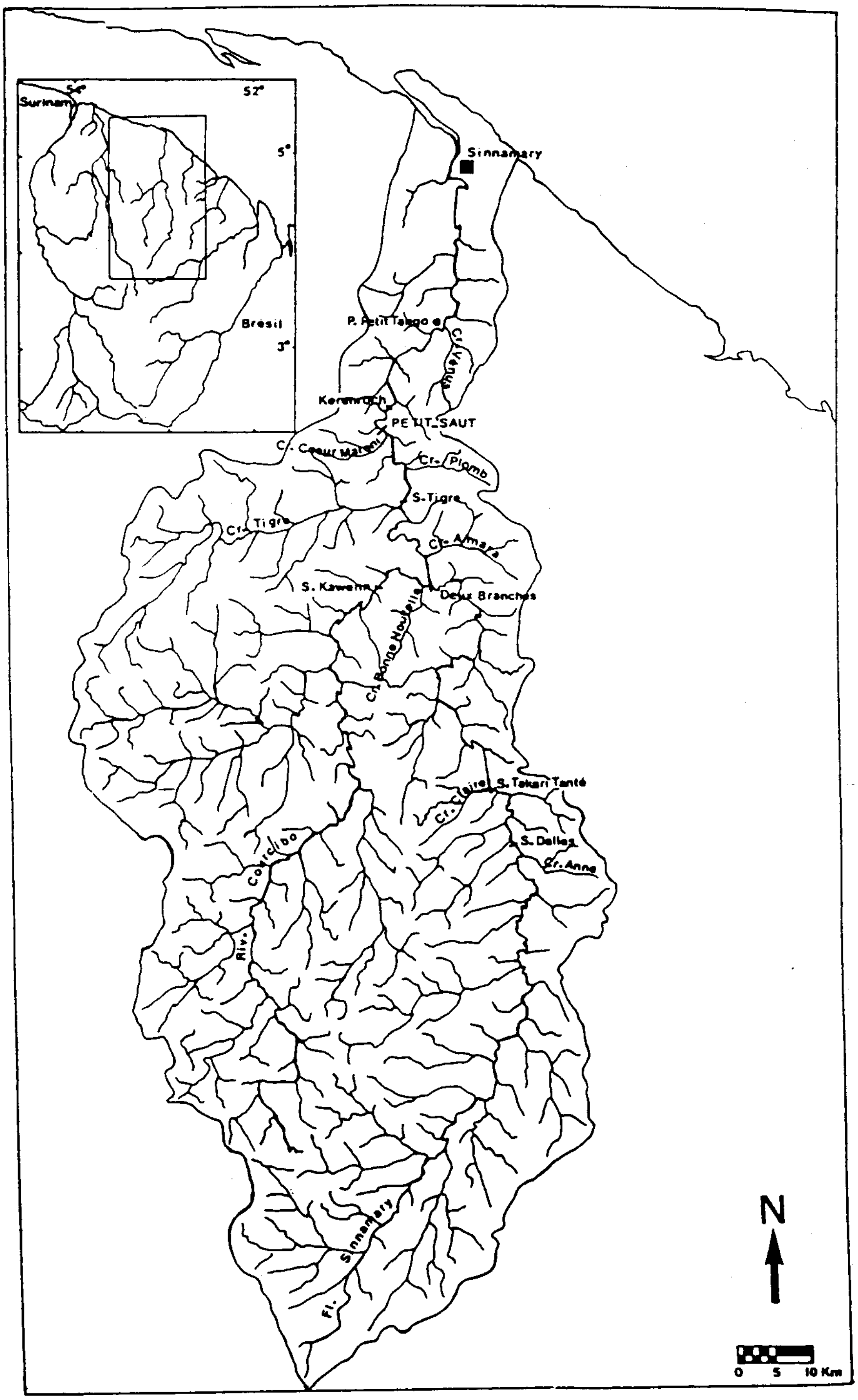

Figure 1 - River system in the Sinnamary catchment area

Poor hydrological conditions resulted in 3 stages in impounding:

- from January to July 1994 water level rose constantly by a few centimetres a day, up to El.31 m;

- from July to December 1994 the reservoir remained stable at El.31 m;

- from December 1994 to June 1995 impounding was resumed until F.S.L. at El.35 m was reached.

During that 18-month period, riparian outflow of 100 m^3/s was released through the bottom outlets.

At F.S.L., reservoir storage capacity is 3 billion cubic metres and its surface area 310 km^2 for an average depth of 10 m. At the dam, maximum depth is 35 m. The time the water is retained in the reservoir has been evaluated at 6 months. The reservoir's shape is extremely complicated.

2. Water Quality

Monitoring of water quality has shown that the various processes are determined by the decomposition of organic matter from the uncleared forest. Thermal stratification occurred rapidly. A few occurrences of partial destratification can be attributed to rainfall or major water releases at the dam. Only the top layer (between 1 and 4 m) was oxygenated.

Anoxic conditions in the hypolimnion mean that the organic matter continues to mineralise by anaerobic means, producing carbon dioxide, methane and sulphur hydrogen. As depth increases, pH decreases, conductivity increases and so does the ammonium concentration.

3. Invertebrates

Monitoring of invertebrates proved especially interesting in 3 fields: biodiversity, functioning of the lake ecosystem, and the food available for fish.

3.1 Biodiversity

Reservoir impounding diminished the diversity that had previously been observed in the Sinnamary and its associated hydrological system. However, there is still greater diversity than in reservoirs in temperate regions.

The reservoir's populations are original for the discovery of species new to scientists (e.g. the copepoda Notodiaptomus nsp), for original populations within a species (e.g. the rotifera *Filinia terminalis*), and for species that had already been identified elsewhere but that hitherto had not been known to exist in the area (many species of rotifera, cladocerans and copepoda).

3.2 Functioning of the Lake Ecosystem

3.2.1 During impounding

Monitoring of invertebrates has shown that the running water biocenosis disappeared upon impounding but the lake-type biomass developed very rapidly, in just a few weeks, with abundant and varied zooplankton. A succession of small, largely detritivorous species was observed before the phytoplankton bloom; then came larger species after

Many fish stomachs have been examined, of many species. It can therefore be established that, 2 years after impounding, most of the Sinnamary's fish species have survived and bred in the reservoir thanks to the abundant food supply offered by the invertebrates that have developed there and that form a dense and varied population.

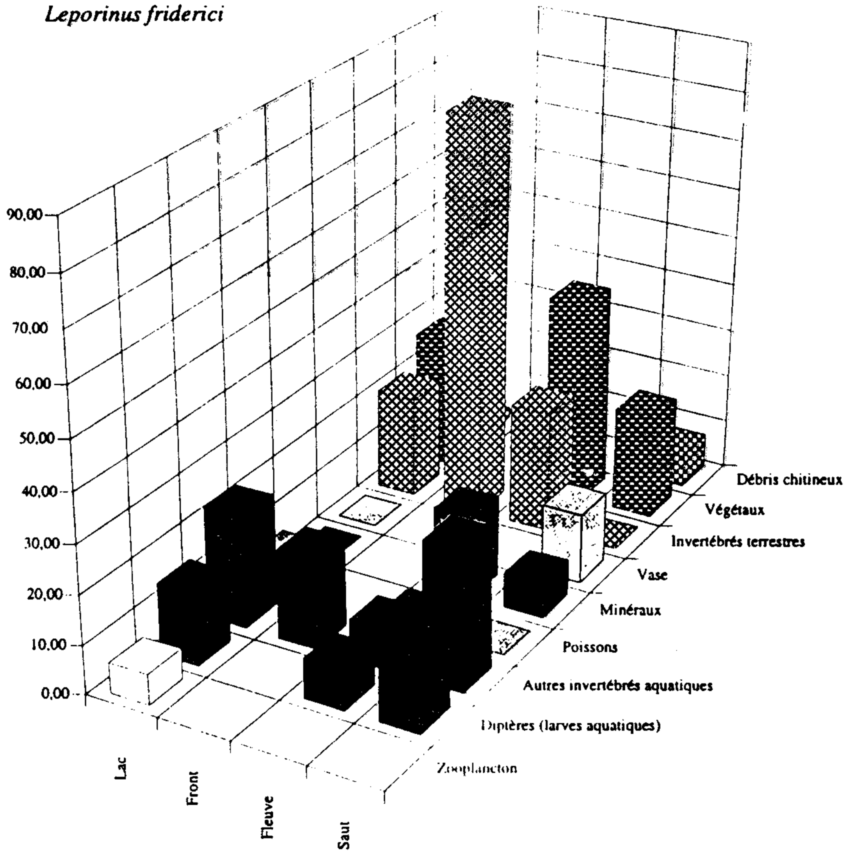

Figure 4 - Comparison of feeding habits of *Leporinus friderici* in the different biotopes

III. THE SINNAMARY RIVER DOWNSTREAM FROM THE RESERVOIR

3.1 Water Quality

Flow in the river consists of reservoir water discharged through the bottom outlets and the turbines, and overspill.

Immediately downstream from the bottom outlets, the water is well oxygenated with a low methane content. The strong turbulence at this level does encourage re-oxygenation and de-gassing.

Immediately downstream from the turbines, the lack of any such turbulence precludes any transfers of gases between the air and the water. Discharge from the turbines is completely lacking in oxygen and highly charged with methane and organic matter. Therefore, to create the same situation as at the bottom outlets, a scale model study was done to design a weir with optimal characteristics to generate as much eddy, and therefore re-oxygenation and de-gassing, as possible.

In any case, it should be noted that as a general rule, not enough oxygen comes into the water downstream (from the atmosphere or from tributaries) to compensate for oxygen consumption by residual reducing elements such as methane. A closed curve is observed that reaches its low point about 40 km downstream from the dam, in the Pointe Combi area.

3.2 Invertebrates

In the mass of water in the river, downstream populations essentially consist of zooplankton organisms (cladocerans, copepoda) drifting from the reservoir. In relation with the various taxona or with hydrological conditions, the lake zooplankton continues to be found at a more or less great distance and may even still be abundant (much more so than before the reservoir was impounded) all the way to the estuary.

The fauna in the sand, dominated by small cyclopoida before reservoir impounding, has evolved considerably, in two stages (figure 5):

- multiplication of harpacticoid copepoda, with the increase in organic particles settling on the bottom, before the sand became anoxic;

- multiplication of the oligochaeta when the sand became oxygen-poor.

No anadromous migrations of decapod crustaceans has been observed.

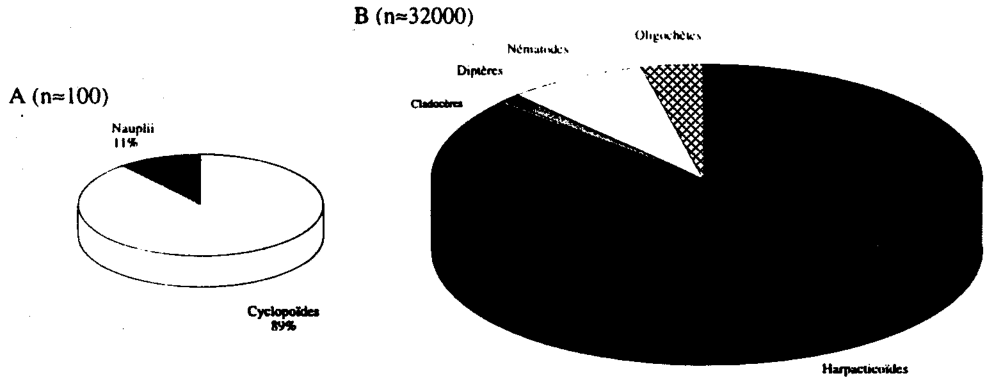

Figure 5 - Comparison of populations in the sand observed downstream from the works at Kerenroch. A = before reservoir impounding, B = since impounding, n = total number of individuals in the sample

the phytoplankton had developed: small rotifera ⇒ bosminidae cladocerans ⇒ cyclopoid copepoda ⇒ daphnidae cladocerans ⇒ calanoid copepoda ⇒ chaoboridae larva ⇒ ostracoda.

3.2.2 After impounding

A) Zooplankton

At present the situation at the middle station (GENIPA) represents the most advanced, stable state in the reservoir. There we find a calanoid domination that indicates good functioning of the lake ecosystem, in the well-oxygenated top layers (epilimnion) where the zooplankton is confined (figure 2). As we move away from that middle station, study of zooplankton development in time and space faithfully and rapidly reflects the changes in how the ecosystem functions consistently with the changes in the water's physical and chemical qualities. Accordingly, the level of structuring or maturity in the lake ecosystem decreases as we move:

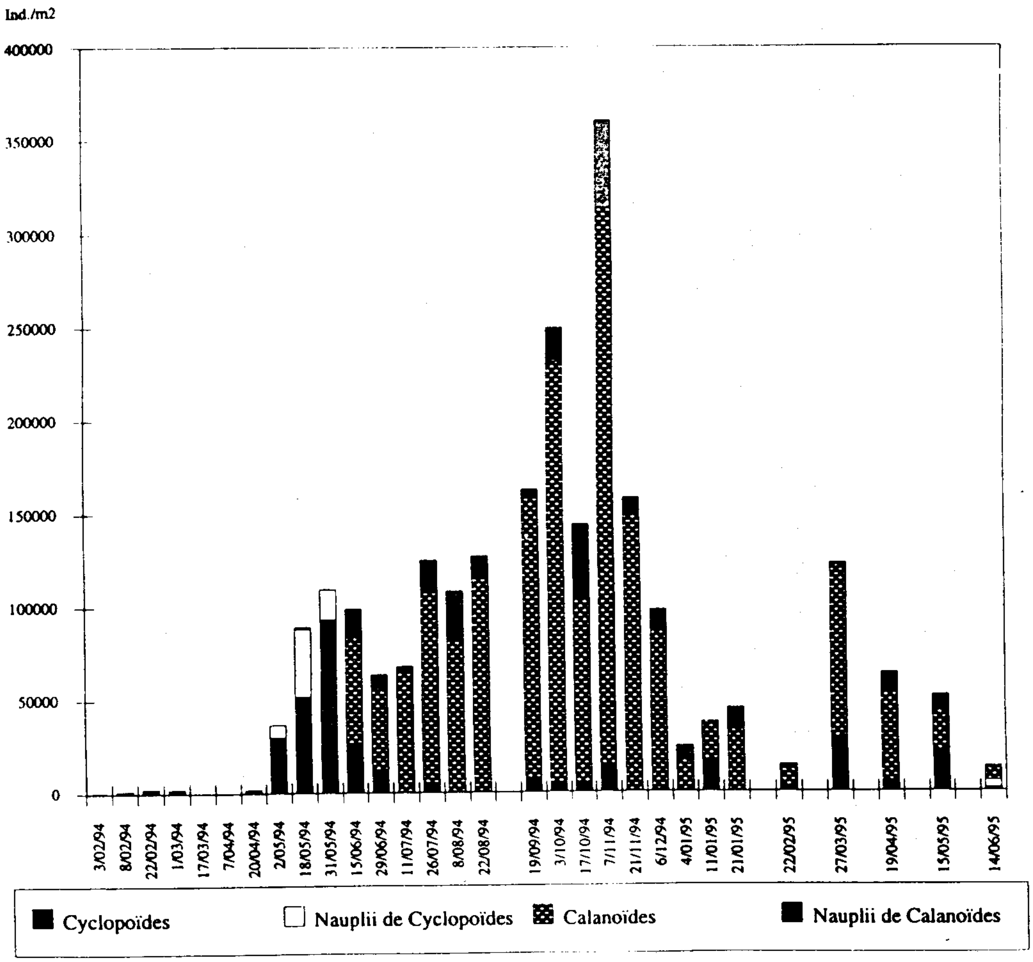

Figure 2 - Fluctuations in copepode numbers at Roche Genipa 1, during impounding

- upstream, where either cyclopoida or cladocerans dominate in the zone of transition between the river and the reservoir;

- downstream, where cladocerans and ostracoda dominate near the dam in reaction to discharges through the bottom outlets, over the surface, or through the turbines;

- sideways, where cladocerans dominate in the submerged forest.

It should be noted that there are no heleoplankton species yet that would characterise a shore zone.

The system's maturity can also be seriously disturbed by heavy rainfall that mixes the mass of water to a certain extent, lowering the dissolved oxygen content and stirring up organic "debris". In these brief incidents the zooplankton is partly wiped out but soon reforms from its most resistant stages, following the same succession of species as during impounding.

B) Zoobenthos

Benthic invertebrates have not been able to develop below the anoxic hypolimnion, at the bottom of the reservoir where aerobic life is not possible. At a slower pace than the zooplankton (the species in question are bigger), they have flourished in the epilimnion thanks to the multitude of supports available in the submerged trees. At present they represent a very large biomass dominated by diptera (figure 3).

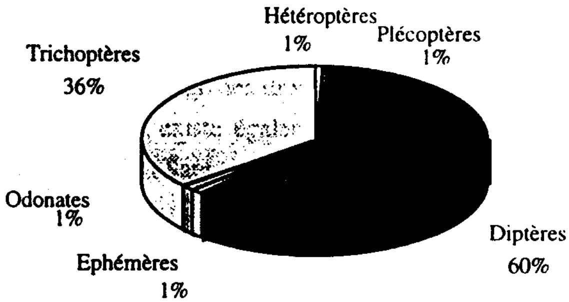

Figure 3 - Percentages of the various insect larva gathered on the man-made substrata placed in the reservoir

3.2.3 New Foods for Fish

Taken together, the zooplankton and the macroinvertebrates in the submerged forest represent an increase in potential food resources. But will the fish like this new prey?

As concerns the macroinvertebrates, it has been observed that the stomachs of fish in the reservoir contain both the larva and the imagos of the limnophilous aquatic insects that have recently appeared, and the larva and imagos of tree insects that have been progressively drowned by the rising waters. Those prey do not represent a fundamental change in the eating habits of many fish in the Sinnamary, already identified as insectivores and omnivores.

As concerns the zooplankton, observations indicate that many species of fish have had to change their diet and have become at least partially zooplankton-eaters in the reservoir, for instance *Leporinus friderici* (figure 4).

CONCLUSION

The physico-chemical characteristics of the Petit Saut reservoir result first and foremost from the decomposition of the submerged primary forest; they are also influenced by weather conditions (rain, wind, cloud or sunshine) and finally by operation of the hydroelectric scheme. The reservoir fills up during the rainy season to the maximum water level at El. 37 m and then is partially emptied during the dry season to supply power to Guiana. That means reservoir water level varies by at most 4 metres over an annual cycle. In the stratified lake zone, rainfall, which is colder than the surface water, helps to mix some of the water in the epilimnion with that of the hypolimnion. Finally, overcast weather reduces photosynthesis, leading to a decrease in dissolved oxygen contents in the daytime, and also in the density of the phytoplankton, a food staple for many taxona.

In the first weeks of impounding only the top two metres of the reservoir's surface layer were oxygenated. Since then the thickness of the oxygenated layer has increased, to 4 m today.

Development of a vertical gradient, whether physical (thermocline) or chemical (oxycline), influences the vertical distribution of the zooplankton. The depth to which light penetrates defines a layer of high production (photosynthesis), where herbivores and carnivores concentrate. The greatest abundance of zooplankton therefore often occurs above or at the level of the thermocline. With the exception of a few chaoboridae, there is no zooplankton in the anoxic hypolimnion.

As for the horizontal distribution of the zooplankton, two zones can be identifies: an axial zone where the Daphnidae and the calanoida are well represented, and a submerged forest zone where, like the reservoir headwaters, Bosminidae are better represented. However, at Petit Saut, unlike the situation in natural lakes and many reservoirs, we have not identified any shore zone, either in terms of water quality or in terms of zooplankton.

To sum up, the most important points that draw our attention are:

- the rapidity of lake-type biological production, since zooplankton was very dense in the reservoir as for the initial months of impounding;

- the diversity of the zooplankton as of impounding;

- the high density of the zooplankton, in relation to the quantities of organic debris and development of the phytoplankton: at the beginning of impounding we witnessed colonisation by detritivorous cladocerans, followed by the development of plankton-eating species;

- good utilisation of this new lake-type food resource by fish.

However, thanks to the re-oxygenating weir placed just downstream of the turbines, the characteristics of the river water between the dam and the river mouth are good enough to support aquatic life.

The types of water-borne aquatic invertebrates living in the river downstream from the dam, especially the microcrustaceans, are directly related to those in the reservoir. In the sediment of the Sinnamary, it has been observed since impounding that the cyclopoida are being replaced by Harpacticoida. Monitoring of the environment, for at least three years after impounding, will provide details on the present trends.

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Biological motivation of water level regulation in the large shallow Lake Võrtsjärv (Estonia)

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ABSTRACT

Lake Võrtsjarv is a shallow and turbid eutrophic lake in Central Estonia with an area of 270 km^2, mean depth of 2.8 m and maximum depth of 6 m. In spite of shallowness the amplitude of water level fluctuations is large: the annual mean 1.38 m, annual maximum 2.20 m and the absolute range 2.92 m. Besides seasonal fluctuations, the annual mean water level of dry and rainy years differs by more than one meter. Long-term water level measurements (starting from 1885, daily measurements from 1921) show a sinusoidal alternation of low and high water states within 30-year periods. Due to the shallowness of the lake low-level periods are accompanied by several negative phenomena like fish kills, intensive resuspension of bottom sediments and cyanophyte blooms.

Since 1964 phyto-, zoo- and bacterioplankton, hydrochemical variables and water transparency have been monitored at monthly intervals. This period coincided with the rising phase of the mean water level (1.5 m during 1964-1993) that allowed to follow the effect of the water level on plankton and hydrochemistry. In spite of the increased Secchi depth for the growing season from 0.6 to 0.9 m, bottom irradiance dropped 7-fold and water column irradiance by 1/3 during these years. As vernal excystation and germination of resting stages of several planktonic organisms is triggered by light, changes in light intensities at the bottom were probably among factors which changed the plankton species composition. Deterioration of mean water column irradiance increased dark losses of light-limited phytoplankton and reduced its biomass. Bacterioplankton for which detritus and phytoplankton exudates serve as the main substrate, decreased in number more than twofold. The increased mean depth of the lake reduced significantly the rate of resuspension. Although nutrient concentrations increased during the period studied, the rising water level had a positive effect on the ecological state of the lake. A decrease in phytoplankton and bacterioplankton biomass and in detritus amount, resembling the effect of a reduction of the trophic status, was related to the strengthening of the light limitation effect on phytoplankton and a decline in sediment resuspension.

Hydraulic calculations have shown that an outflow regulator would generally maintain the natural water level regime, but at a higher mean level and reduced amplitude. During low water periods the level would be elevated by approximately one meter, the annual mean amplitude would decrease from 140 to 110 cm but the annual maximum amplitude by more than 1 m.

KEY-WORDS: Water level / Shallow lake / Light limitation / Secchi depth / Phytoplankton biomass / Bacterioplankton (total count) / Long-term investigations / Sediment resuspension / Level regulation

INTRODUCTION

The word *eutrophic* generally refers to nutrient-rich conditions and, in a narrow sense, eutrophication can be understood as an increase in the nutritional level of waterbodies. This increase is usually sensitively reflected in biota by increased production and standing stock and by reduced biological diversity. In a broader sense, the concept of eutrophication implies both its initial causes as well as responses of the ecosystem and, as a fact, often focuses upon the latter. Quantitative responses of the biota to increased nutrient loading are adequate in a nutrient-limited environment but can differ in a wide range in the case of other limitations. Optical properties of water as well as the mixing depth and resuspension rate of sediments, all determined by the water level, come to the forefront in the case of shallow (mean 2.8 m) and turbid L. Võrtsjärv where light limitation plays a major role in controlling phytoplankton growth (Nõges, 1995; Nõges *et al.*, submitted). Despite the rather stable nutrient content the biological indices of the trophic state fluctuate strongly depending on water richness. The influence of changes in the water level on different trophic levels is either direct or mediated by cascading effects in the food chain. The shallowness of the lake and the fluctuating level restricts the use of the lake for recreation during low-level periods. The shoreline recedes in places up to a kilometer in the second half of summer, leaving large littoral areas dry. This area cannot be used as a beach due to the high humidity content of sediments. Stony reefs become dangerous for boating.

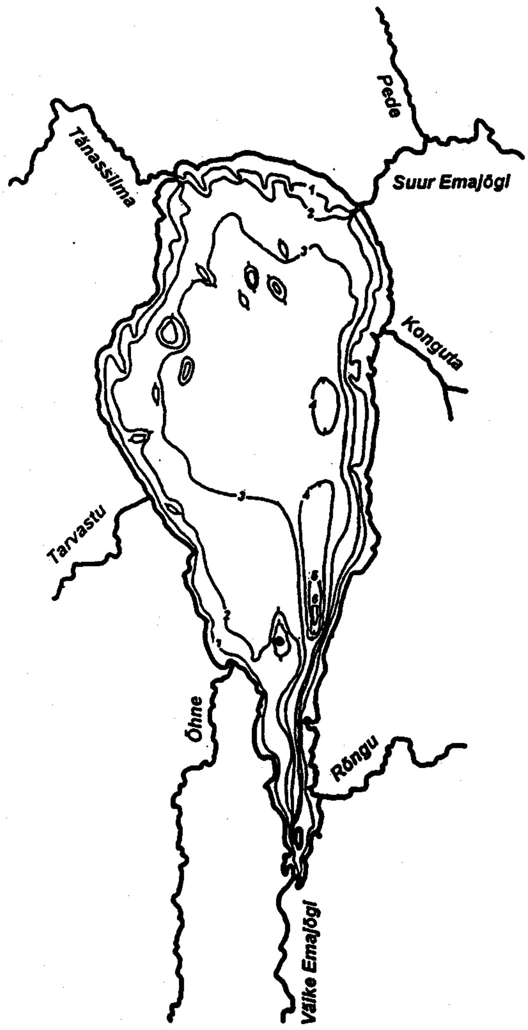

Figure 1: L. Võrtsjärv and connected rivers

The height of the water level has been recorded daily since 1921, the discharge, since 1961 in the mouth of the R. Suur Emajõgi, the outflow of the lake. The foundation of the Võrtsjärv Limnological Station at the beginning of the 1960s has enabled all-year-round investigations of the lake. L.Võrtsjärv is one of the few lakes in the world supplied with a continuous monthly data series covering a period of 30 years. At present, continuous monthly records of physical and biological parameters are available since 1964, those of chemical parameters, since 1968.

DESCRIPTION OF THE LAKE WITH AN EMPHASIS ON HYDROLOGY

Lake Võrtsjärv (Fig. 1) is a shallow lake in Central Estonia at 58°05'- 58°25' N, and 25°55'- 26°10' E. Its area is 270 km^2, mean depth 2.8 m and maximum depth 6 m. Eighteen rivers and streams collect their water from the 3104 km,[2] mostly intensively cultivated, drainage basin. The lake is highly eutrophic, characterized by mean concentrations of total N 2 mg l^{-1} and total P 53 µg l^{-1}. The water is alkaline (pH 7.5-8.5) with a great buffering capacity and a high seston content. During the ice free period, transparency does not exceed 1 m. The only outflow of L. Võrtsjärv is the River Suur Emajõgi which has a small slope, only 0.038‰. Its bankfull discharge depends on the level of the main tributary, the River Pede, due to opposite flow occurring in the R. Suur Emajõgi in the reach between the mouth of the R. Pede and L. Võrtsjärv almost every spring. This bifurcation influences the outflow regime and the lowering of the flood level in the lake (Järvet, 1995).

The lake depression is of preglacial origin. About 2/3 of the lake bottom is covered with mud lying on marl having a total thickness of up to 7.6 m. (Veber 1973). Sediment accumulation is intensive and the lake becomes ever more shallow from year to year (Kaljumäe and Koskor, 1980). Short-term increases in the water level, observed in the middle of the last century and at the beginning of this century (Sievers, 1854; Mühlen, 1919), intensified shore erosion and transport of eroded material towards the outflow, and promoted further clogging of the outflow (Orviku, 1973).

Increase in the water budget of L. Võrtsjärv is mostly accounted for by inflow (73-83 %) and decrease by outflow (80-91 %) (Jaani, 1990). The renewal of water takes 240 to 384 days, occurring, on an average, once a year. The balance between net inflow and outflow clearly controls the water level. Due to rather flat shores and the small water depth, changes in the water level are expressed in the mobile shoreline and in large relative differences in volume. The long-term mean level of L. Võrtsjärv, computed over a 73-year period (1922-1993) is 33.66 meters according to the Baltic Level System. At the mean level the area of the lake is 270 km^2, the volume 0.754 km^3. The absolute long-term range of level fluctuations at the Rannu-Jõesuu station is 292 cm, which corresponds to the 84 km^2 difference in the lake area and to the 0.799 km^3 difference in its volume. The shores are flat, except in the east. At the highest natural water level (35.28 m) in November 1926, 57 km^2 of the shore area was flooded. The linear approximation fitted well for the relation between the water level (L) and the mean depth of the lake (z_{av}) in the recorded range:

(1) $$z_{av} = 0.741 \times L - 22.29 \quad (R^2 = 0.976)$$

The absolute range of the mean depth of the lake extended from 1.71 m (09.11.1964) to 3.71 m (26.11.1923).

Intensive and prolonged high water in spring, low water in summer and winter, and a noticeable rise in the water level in autumn are the characteristic seasonal features of the lake (Fig. 2). The annual amplitude of fluctuation in the water level has been 1.38 m on an average, ranging from 0.76 to 2.20 m in different years (Jaani 1990). In 78 % of years the highest water level has occurred in spring, and in 22 % of years in late autumn.

A long-term sinusoidal fluctuation of the water level with a period of about 30 years is best revealed by using a 7-year moving average (Fig. 3). Water levels for the period 1885-1921 have been calculated by Jaani (1990) on the basis of data on the R. Suur Emajõgi. A rather smooth continuous decrease (1928-1940) or increase (1940-1957; 1965-1990) within the period resembles a long-term trend and can be distinguished from it only in the context of a longer time series. Apparent periodicity is probably associated with large-scale fluctuations in solar activity (Jaani 1973) and atmospheric processes, since it appears in the same way in large geographic regions. Similar periodicities with a spectral density maximum of 28-32 years have been found in the water level dynamics of the lakes of Saimaa, Ilmen and Onega (Masanova and Filatova, 1985), as well as for different hydrological elements of Lake Ladoga (20-

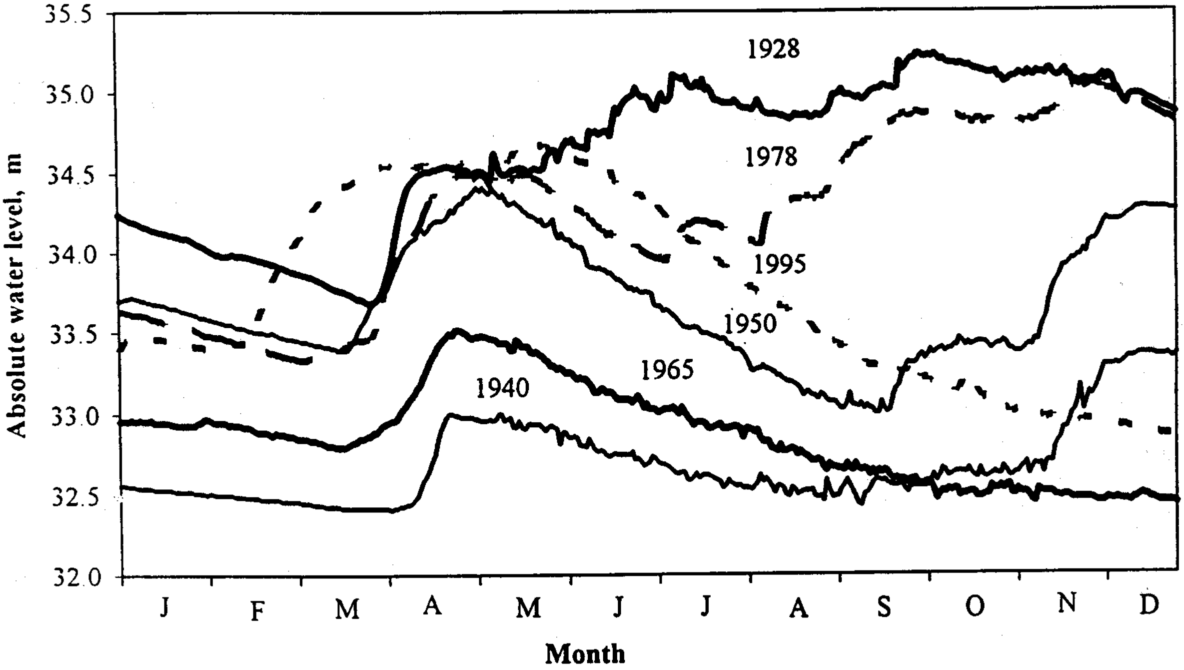

Figure 2: Seasonal dynamics of the water level in L. Võrtsjärv
(After Jaani, 1990 supplemented by data on 1995)

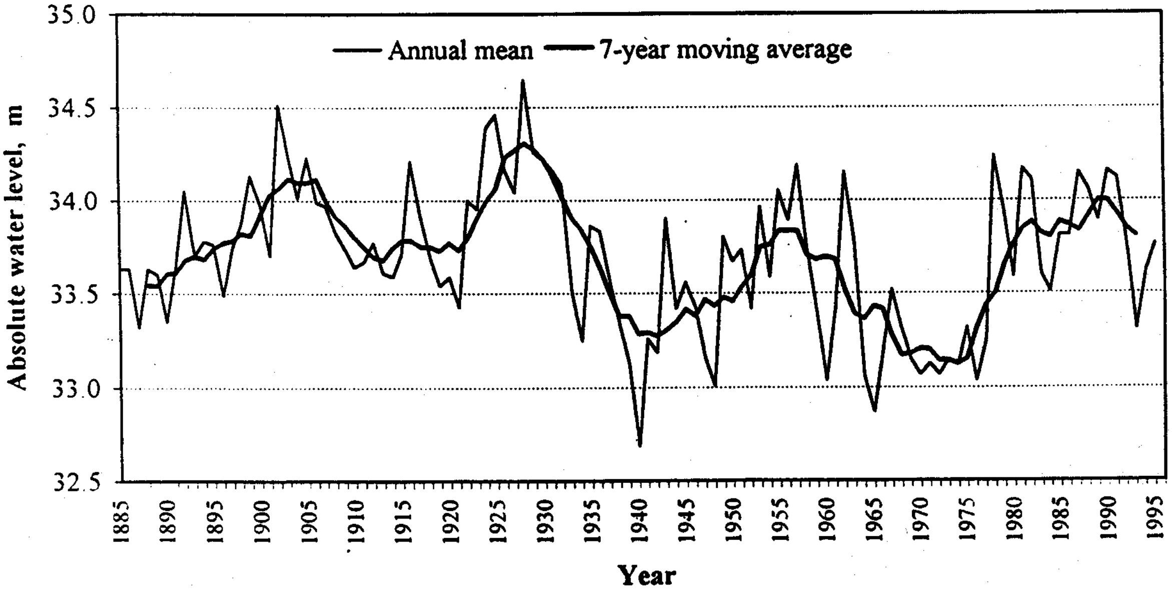

Figure 3: Long-term changes in the water level in L. Võrtsjärv
(After Jaani, 1990 supplemented by latest data)

30 years) (Malinina *et al.*, 1985) and Lake Müggelsee (Behrendt and Stellmacher, 1987). The last increase in the water level can be firmly attributed to the 25 % increase in the annual amount of precipitation in Central Estonia (from 526 to 652 mm) between 1964-1993 (Kivi & Russak, 1990).

The period 1964-1993, overlapping with the biological time series, can be divided into two parts each characterized by a significantly different water level. A more detailed analysis has been presented in Nõges and Järvet (1995). The years 1964-1977 were dry with a mean water level of 33.14 m and a mean lake depth of 2.38 m. The period 1978-1993 can be referred to as rainy years. The fifth and eighth highest water levels during the whole 72-year measuring period correspond to the years 1978 and 1981. Due to the long dry period the average water level during the whole period 1964-1993 appeared to be 10 cm lower compared with the long-term average. Still, the constructed linear trend line for this period revealed an increase in the mean water level by 1.5 m and the corresponding increase in the mean depth of the lake by 1.1 m.

MATERIAL AND METHODS

The data set used in the present paper consists of water transparency measurements, hydrochemical and plankton analyses made in 1964-1993 at the Võrtsjärv Limnological Station. Daily water level data for the years 1922-1993 were obtained from the hydrological station on the outflow. The other measurements and analyses pertain to the surface layer of the main monitoring station. Routine sampling was performed once a month, in some periods more frequently, 2-6 times per month.Underwater light measurements with an 4-π PhAR collector (Williams and Jenkinson, 1980) were made in 1989. Bottom irradiance (I_B) was calculated according to Lambert-Beer's law

$$I_B = I_0 \exp(-kz) \tag{2}$$

where irradiance in the surface layer (I_0) was taken equal to 100 %, the attenuation coefficient (k) was replaced by its relation to Secchi depth (Eq. 4), and the mean depth of the lake was denoted by z. Average irradiance in the mixed water column was calculated according to Riley (1957):

$$I_{mix} = I_0[1-\exp(-kz)]/kz \tag{3}$$

assuming that in L. Võrtsjärv the mixing depth increases simultaneously with the mean depth of the lake z

RESULTS AND DISCUSSION

Water level effects on lake biota

Phytoplankton

The phytoplankton of L. Võrtsjärv is abundant and rich in species. Owing to shallow water and continuous wind-induced mixing the number of benthic and periphytic forms is large. In a few cases phytoplankton biomass reached 100 g m^{-3} in the early 1970s. At the end of the 1970s, biomass decreased, and several changes took place in the species composition: it became generally poorer, while a new species *Limnothrix redekei* (Van Goor) Meffert appeared among dominants. Nowadays the algal community is represented by an association of filamentous algae, the species of the genus *Aulacoseira* in spring, *Planktolyngbya limnetica* (Lemm.) Kom. et Cronb. and *Limnothrix redekei* in summer and autumn, accompanied by a low variable biomass of small algae, mostly chlorophytes and chrysophytes. In spite of a slight increase in winter nitrate levels and summer phosphate levels during recent decades, phytoplankton biomass has seldom exceeded 30 g m^{-3}. The annual average chlorophyll *a* concentration is 26.5 mg m^{-3}, integral primary production 0.67 g C m^{-2} d^{-1} (243 g C m^{-2} y^{-1}). The mean values of destruction and sedimentation rates in 1991 were 0.47 and 2.42 g C m^{-2} d^{-1}, respectively.

Several circumstances support the hypothesis of the prevalence of light limitation in phytoplankton growth in L. Võrtsjärv (Nõges, *et al.*, submitted). In natural waterbodies light limitation is an universal phenomenon, as light-limited phytoplankton is always found below a certain depth or in the dark season of the year (Sakshaug, 1980). In shallow polymictic lakes where the mixing depth is determined by the physical depth of the lake, light climate is strongly affected by seasonal and long-term changes in the water level. This is a dual effect since not only the mean depth (z) but also the optical properties of water are affected by the water level. Secchi depth is the only optical parameter belonging to the set of routine monthly measurements in Lake Võrtsjärv. During the last thirty years water transparency has shown an increasing trend which is especially clearly expressed in autumn (September - November) at the time of the annual maximum of phytoplankton biomass and detritus content. Autumn Secchi depth increased from 0.6 m in the 1970s to 0.9 m in the 1990s. According to light measurements in L.Võrtsjärv in 1989 the relation between the attenuation coefficient (k) and Secchi depth (S) can be described by the reciprocal function:

(4) $$k = 1.48 \pm 0.05 / S \quad (\pm \text{ standard error})$$

In productive shallow waterbodies, phytoplankton and detritus gain a leading position among optically active substances determining the under-water light field. In these conditions, the key position of phytoplankton becomes obvious as most of the particulate matter originates directly or indirectly from primary production like phytoplankton itself, detritus (mainly dead phytoplankton), bacteria (consumers of phytoplankton) or zooplankton (consumers of phytoplankton and bacteria). By vertical mixing phytoplankton is transported through a light gradient and adapted to an average light intensity in the mixed layer which Riley (1957) has defined as "effective light climate". This average level is controlled by surface irradiance, vertical light attenuation and mixing depth (Eq. 3). For L. Võrtsjärv surface irradiance was taken equal to 100 %since its changes in a 30-year period could be neglected. The attenuation coefficient (k) was replaced by its relation to Secchi depth, and the mixing depth was determined by the mean depth of the lake. In spite of improved water transparency, the calculated water column irradiance dropped from the average 28% to 19% i.e. by about 1/3 during the period 1964-1993. In these years, phytoplankton biomass maxima were inversely correlated with the water level and remained at a rather low level during the rainy period (Fig. 4). A similar process has been observed in Lake Beloe (Rumyantsev *et al.*, 1985) where a 1.5 m increase in the water level affected much stronger the underwater light field than an increase in trophic parameters like mineral phosphorus and nitrates did.

An increase in the water level had a much stronger effect on bottom irradiance than on water column irradiance. Relative bottom irradiance decreased from about 2.8% in the middle of the 1960s to 0.4 - 0.5 % in the 1990s. The 1% surface PhAR criterion, commonly used to assess the lower boundary of the euphotic zone (Tilzer, 1987), was overstepped while large bottom areas fell outside the euphotic zone. Although most of the water column remained within the euphotic zone, deterioration of light conditions resulted in a decrease of primary production which was reflected in reduced phytoplankton biomass and pH values. During the ice-free period, relative bottom irradiance was the lowest in May (due to the high water level coinciding with large algal biomass) and increased gradually towards the end of the year. As vernal excystation and germination of resting stages of several planktonic organisms is triggered by light, changes in light intensities at the bottom were probably among factors which changed the plankton species composition. The invasion of *Limnothrix redekei*, a blue-green alga, whose development starts in early spring in the shade of diatoms, is obviously connected with light conditions as well. Nicklisch *et al.* (1981) demonstrated a specific light adaptation of this species, which allows its mass development, compared with other blue-greens, in more turbid and cool water.

Bacterioplankton

Bacterial counts in L. Võrtsjärv have ranged from 1.2 to 14.0 · 10^6 cells ml^{-1} with a mean value of 4.2 · 10^6. Their maxima occurred in the first half of the 1970s just as those of phytoplankton biomass, and a similar relation could be observed between the total number of free-living bacteria and monthly mean water levels (Fig. 4). Three mechanisms may be responsible for a simultaneous decrease in phyto- and bacterioplankton abundance: 1) limitation of bacterioplankton growth by algal substrate under conditions of decreasing primary production; 2) limitation of bacterial numbers by the smaller amount of resuspended sediments in the case of a rising water level; 3) increase in zooplankton grazing pressure on bacteria under conditions of the growing proportion of filamentous blue-green algae in phytoplankton. The first two mechanisms can be joined under a common denominator "dilution effect". On the basis of available data it is hard to determine the role of each of these mechanisms, though the leading role of the water level is obvious. Fondén (1969) found a similar negative correlation between bacterial density and the depth of the

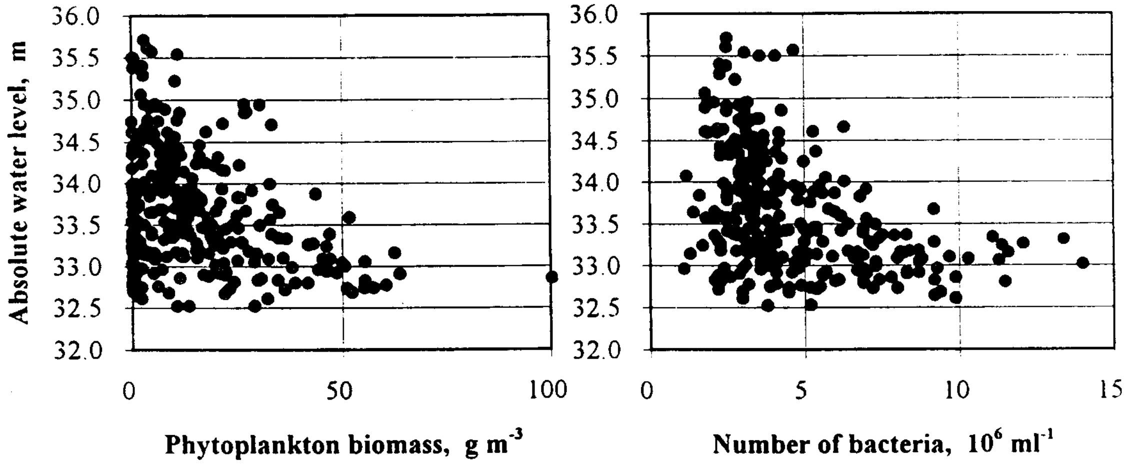

Figure 4: The scatter plot of plankton abundance *versus* water level in L. Võrtsjärv
Monthly mean values for the period 1964-1994

water column in the large Swedish lakes of Mälaren and Hjälmaren, and associated it with the vertical dilution effect of the deeper water column. During summer, bacterioplankton was influenced by both filter-feeding zooplankton and the amount of available organic carbon. There exists a close link between pelagic and benthic processes in large shallow lakes. On windy days resuspended POC exceeded 15-20 times carbon fixation by phytoplankton in Loosdrecht Lakes (Gons *et al.*, 1986). Wainwright (1987) and Ritzrau and Graf (1992) have shown that resuspension not only passively increased bacterial abundance but also stimulated bacterial growth. Intensive resuspension occurs when water depth is less than half of the wavelength. Due to the short wavelength (3-5 m), characteristic of Lake Võrtsjärv (Jaani, 1973), the percentage of the lake area affected by resuspension ought to be strongly dependent on the mean depth of the lake. Malve *et al.* (1994) demonstrated on the example of the large shallow Lake Pyhäjärvi that 1.7-fold increase in wind speed was necessary to deepen the resuspension front by one meter.

Zooplankton

The zooplankton of L. Võrtsjärv is dominated by small cladocerans, mainly *Chydorus sphaericus* (Müller), and rotifers which are specialized in feeding in patches between filaments. Recent studies made in the dry year of

1995 revealed an important role of protozoa in the lake, among which ciliates formed a mean biomass of 0.68 g m^{-3} (Zingel, unpublished) which is almost equal to the biomass of crustacean and rotifer zooplankton (0.80 g m^{-3}) (Põllumäe, unpublished). The high detritus content of water, resulting from continuous mixing and wave action, has been considered the main factor determining the small biomass and mean body size of filtering zooplankters, as it clogs their filtering apparatus (Haberman, 1984). Revival of the ecosystem through the increasing water level at the beginning of the 1980s was reflected in a decrease of rotifer biomass (Haberman, 1983). However, no significant trends were observed in the biomass of crustaceans and rotifers in a 30-year time series, though it established several changes in the species composition.

Zoobenthos

Among the bottom fauna of L. Võrtsjärv *Chironomidae* dominate in all zones of the lake. In 1973-1978 big larvae of *Chironomus plumosus* L. formed on the average 60.8 % of macrozoobenthos biomass (Kangur, 1982). During the rising phase of the water level in 1973-1993, a general tendency towards increasing the abundance and biomass of *Chironomidae* and *Oligochaeta*, as well as decreasing the abundance of *Mollusca* was noticeable. In 1973-1993 the annual mean abundance of bottom animals fluctuated from 520 to 1600 ind m^{-2}, and biomass from 1.6 to 26.4 g m^{-2} (Kangur *et al.*, in press). As annual differences in zoobenthos amount are strongly influenced by changes in *Chironomidae* whose abundance depends on meteorological conditions during the reproduction period and on fish pressure (Kangur, 1982), no significant correlations occurred between zoobenthos and the water level. As a rule, the littoral area has been the richest in species diversity, as well as in abundance and biomass. Merely *Chironomus plumosus* achieved its highest biomass in the muddy profundal area (Kangur *et al.*, in press). In dry years the shoreline recedes in places up to a kilometer in the second half of summer, leaving large littoral areas, where the benthic fauna cannot survive, dry. A more stable water regime would gain these areas of high bottom fauna production available for feeding the basically benthivorous fish community.

Aquatic macrophytes

Wind-exposed northern and eastern coasts of the lake are bordered with a mostly continuous belt of *Phragmites australis* (Cav.) Trin. ex Steud., followed by submerged *Potamogeton perfoliatus* L. at a depth of about 2 m. Only in places exposed to the strongest wave action, the reed-beds become narrow or fragmentary, often separated from the shore by a shallow open water area. The western coast is richer in vegetation. The wide zone of *Phragmites australis* and *Schoenoplectus lacustris* (L.) Palla), starting right on the shore, has dense vegetation. The zone of floating-leaved plants (*Nuphar luteum* (L.) Smith) is distinct in places, while the zone of submerged vegetation is everywhere well formed. The southern corner of the lake is fully overgrown with macrophytes. All types of aquatic macrophytes are abundantly represented, and become often merged. This description by Mäemets (1973) rendered briefly here, is still valid, though the area and density of macrophyte stands has increased. Depending on fluctuations in the water level the abundance and distribution boundaries of different species vary greatly. Besides eutrophication, also lasting low-level periods in dry years contribute to the broadening of reed-bed areas. The formation of macrophyte stands along the shoreline northward of the mouth of the R. Tänassilma has been associated with the extremely low water level in 1939-1940 which enabled emergent macrophytes (*Phragmites australis*, *Schoenoplectus lacustris*) and submerged vegetation to fix and extend to the previously sandy areas (Pihu, 1959). More intensive shore abrasion occurring at higher water level would prevent rapid overgrowing of shallow areas, especially in the southernmost part of the lake.

Fishes

The most abundant fish in Lake Võrtsjärv is bream (*Abramis brama* (L.)), its mean biomass in 1981-1990 was 47.9 kg ha^{-1}, production 31.1 kg ha^{-1} y^{-1}. Pike-perch (*Stizostedion lucioperca* (L.)and pike (*Esox lucius* L.) with biomasses 11.5 kg ha^{-1} and 7.8 kg ha^{-1}, respectively, occupy the leading position among predators. The production of pike is 5.6 kg ha^{-1} y^{-1}. The abundance of the eel (*Anguilla anguilla* (L.)) population depends on the introduction of glass eels. About 35 million glass eels have been regularly introduced into the lake during 1956-1993 (Nõges *et al.*, in press).

In the 1950s and 1960s L. Võrtsjärv was regarded as a ruff (*Acerina cernua* (L.)) lake because the bulk of fish caught here consisted of ruff, young perch (*Perca fluviatilis* L.) and roach (*Rutilus rutilus* (L.)). In the 1970s commercial trawling was stopped, glass eels were regularly introduced into the lake and the protection of commercial fishes was improved. These measures resulted in a rapid growth of stocks and catches of commercial fishes (mainly pike-perch and eel) and in a sharp decline in the abundance of less valuable small fish (Pihu, in press).

Several winter fish kills (in 1939, 1948, 1967, 1969, 1978, 1987) have been documented during this century (Kirsipuu *et al.*, 1987). Most of them (1939, 1948, 1967, 1969) coincided with low-level periods and, hence, with a higher primary production in the preceding summer and oxygen depletion during winter. The winter of 1995/1996 has been another example of the disastrous consequences of the low water level. After the highly productive summer of 1995 the lake was frozen at an extremely low level. The winter has been cold and the lake has been covered by thick ice (~0.6 m) and snow (~0.3 m). There have been no thaws from mid-December till the beginning of March. The lowest oxygen concentrations during the studied 30 years (2.3 mg l^{-1} just below the ice, 0.4 mg l^{-1} in the bottom layer) were registered on 1st March.

The flocding of lakeside meadows starts at the water level of 34.47 m (Pihu, 1959). In dry springs the flood plain remains dry, and the spawning area of pike is restricted. Strong winds coinciding with low water in May endanger the spawn of pike-perch, which can be buried under sediments (A. Kangur, oral communication).

Prospects in water level regulation

Lake Võrtsjärv with its large drainage area, bad outflow conditions and flat shores is a complicated object for level regulation. The requirements of fishery, water economy, agriculture, recreation and water transport must be taken into account while designing the regulation scheme. The following goals can be set:

- to reduce both the annual and long-term water level fluctuations;
- to prevent low level and to keep the level as high and as stable as possible;
- to raise the flood level so as to form water-meadows, i.e. the water level must exceed 34.4 m;
- to prevent the regulated maximum level from exceeding natural maxima in order to avoid flooding of buildings and roads;
- to guarantee the sanitary minimum flow in the River Suur Emajõgi.

Outflow calculations (Kaljumäe and Koskor, 1980) showed that a constraint level regulation of L.Võrtsjärv is impossible as it would requires far better outflow conditions. Nineteen regulation scenarios were tested in which the minimum levels varied from 33.0 to 33.7 m, the maximum levels from 34.0 to 34.8 m, and the difference between minimum and maximum levels from 0.6 to 1.8 m. By blocking the outflow it would be possible to prevent dropping of the water level below the set minimum value, however the level might be higher than desired in years of large runoff from the drainage area. Lowering the set minimum level is not recommended, as it would result in a decrease of maximum levels by 30 - 50 cm in dry years, but only by 5 cm in rainy years. The regulated maximum level can be lower than the set maximum in dry years and exceed the latter in rainy years. All requirements would be satisfied in the best way when using the following regulation scheme: filling of the lake proceeds in April-May when the regulator is closed. Lowering of the water level starts in the second half of May at the maximum flow rate determined by the outflow conditions, until it reaches the set minimum (usually in September). The water level would increase slightly in November-December. In years when the lake cannot be regulated due to the high level in the R. Pede, or due to excessive inflow, the regulator would be opened and the natural regime established. It would be reasonable to keep the minimum level at 33.7 m which is the long-term average and to which the level can be lowered in most cases (75% of years). The regulated maximum water level is to be 34.8 m in which case forced maxima would reach 35.6 m.

As a result of regulation measures water level dynamics does not change much. It will generally follow the natural pattern but will proceed at a higher level and will have a smaller amplitude. The level will be about 1 m higher during low-water periods. Extraordinary high levels do not exceed much natural maxima, as the natural level equals at 1% probability 35.50 m and the regulated level 35.6 m. The annual mean amplitude of level fluctuations will decrease by 0.3 m (from 1.4 to 1.1 m), the annual maximum amplitude by more than 1 m. The sanitary minimum flow in the R. Suur Emajõgi will be guaranteed, while fishery requirements can be satisfied partially: meadows will not be flooded only in exceedingly dry years

Arguing against this water level regulation plan Raukas and Tavast (1990) have pointed out intensification of shore abrasion in the eastern and northern shores and the expansion of continuously flooded areas.

CONCLUSIONS

Lasting low-water periods in L. Võrtsjärv are accompanied by a number of adverse biological phenomena consisting generally in destabilizing of the ecosystem. An increase in phytoplankton and bacterioplankton biomass deteriorates the transparency and gas regime of the lake. Low-level periods accelerate the overgrowing of shallow areas with macrophytes and deteriorates the spawning conditions for pike by restriction of spawning places and for pike-perch by the spawn being buried under sediments. By means of the regulation of the water regime, by raising the minimum and maintaining the optimum level, conditions in the lake can be improved. Greatest success would be achieved by reducing sediment resuspension and by strengthening light limitation on phytoplankton, which would control primary production and the amount of particulate matter in the water.

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Integrated System Analysis:
Effects of water management on the habitat suitability for key species in the Sea of Azov

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ABSTRACT

The Azov Sea is a shallow inland sea on the North-Eastern side of the Black Sea, bordered by Russia and the Ukraine. The main tributaries of the Azov Sea are the rivers Don and Kuban, which together provide 80% of the fresh water inflow. A small outlet provides exchange with the Black Sea, which results in a salinity gradient across the Azov Sea of 0 to 18 promilles. The Azov Sea is used for shipping and tourism, while valuable wetlands are situated on the eastern and western sides. However, the sea's most striking feature is its high productivity. In the early 1930's the ecosystem was able to sustain fish catches of up to 300,000 tons per year, including economically valuable species like sturgeon. Due to changes in its drainage basin, the ecosystem of the sea has suffered severe impacts on its functioning. The habitat suitability of key species of the area was evaluated using water quality model results and an inventory of species distribution and species habitat requirements data. For a number of management scenarios based on water-use in the area, changes in habitat suitability could be quantified. Habitat suitability was recalculated into potential carrying capacity for each species. From the results is was concluded that changed salinity gradients within the sea and low oxygen concentration are a major cause of changed habitat suitability of species. Other causes, such as loss of spawning habitats for migratory fish in bordering rivers were not taken into account.

KEY-WORDS:

Habitat Evaluation Procedure/ Integrated System Analysis/ Azov Sea/ Ecology/ HEP/ water management/ modeling/ impact assessment/ ecotope/ carrying capacity/ habitat suitability

INTRODUCTION TO THE PROJECT

The ecosystem of the Sea of Azov is impacted by anthropogenic activities in its drainage basin. Impacts such as use of fresh water for irrigation, industrial and agricultural chemical pollution, use of fertilizers and fisheries have led to major changes in the species composition within the Sea. The changed characteristics of this ecosystem are illustrated by frequent occurrence of benthic anoxia, the almost total decline (2% of reported catch in 1930's) of fisheries and the blooming of new species. In 1994-1995, an evaluation of habitat suitability for key species within the Azov Sea was conducted on top of an integrated study of relations between water resources, water quality and water management based on the standard Delft Hydraulics Framework for Analysis. The objective of the ecological evaluation was to gain an understanding of the present ecological status of the Sea of Asov, its natural functions and the relationships between physico-chemical characteristics and the impact upon the life-cycles of key-species within the study area. However, it is emphasized that this study aimed to, at most, illustrate the potential of the methodology by identifying and quantifying major impacts and provide quantitative links to effects within the ecosystem of the Sea. This paper therefore, serves to illuminate the methodology rather than to provide a calibrated and validated habitat evaluation for the study area (Delft Hydraulics, 1996). Finally, an important objective was to provide an integrated management tool to local experts in order to enhance the region's expertise for execution of integrated water management studies. This tool is called the Asov Sea Decision Support System.

Integrated System Management

The management of ecosystems deals with a variety of user functions and environmental aspects. In the past, water resources and water quality used to be carried out separately. More and more, however, the approach to both water resources and water quality management are integrated to enable the assessment of impacts of managerial actions on the quality of (surface) water, sediment and biota in rivers, estuaries and coastal zones. This integrated approach of water management increases the complexity of the decision making, as can be seen from figure 1 that shows the Delft Hydraulics framework for analysis.

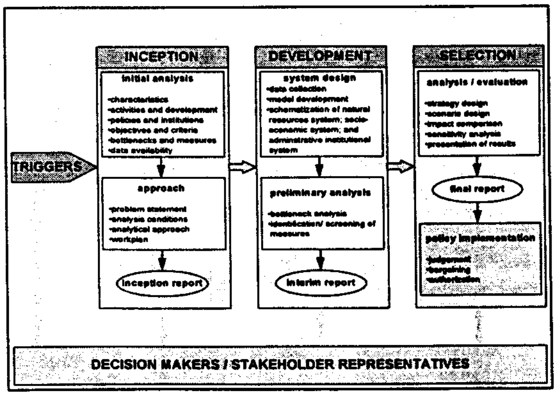


Figure 1 Delft Hydraulics Framework for Analysis

Complex water management decision problems can be made more manageable by including all the available knowledge in a Decision Support System (DSS). A Decision Support System should be considered as a learning device that allows for experimentation, interpretation and communication so as to find creative solutions and new ideas. It implies an implementation of a computer program that:

- assists people in their decision process; serves as a learning device that allows for experimentation, interpretation and communication so as to find creative solutions;
- supports rather than replaces judgement of these people;
- improves the quality of the decision making process, rather than the quality of the final decision.

The Azov Sea Decision Support System

Development of the Azov Sea DSS was started as an illustration of the power of integrated management. The system is limited to the Azov Sea itself. The rivers Don and Kuban and more than 20 small rivers in its drainage basin, and coastal emission sites of polluted substances were treated as point sources on the boundary of the system. This set-up limits the type of strategies and measures that can be analysed, but decreases the complexity of the DSS. A modular approach for the DSS was chosen, which facilitates later improvement or replacement of parts of the system. The basis for the Azov Sea DSS is formed by:

- a simulation of the system hydrodynamics; a database on pollution loads, organized by a Waste Load Management tool;
- a simulation of physical, chemical and basic biological processes;
- an ecological evaluation tool;
- a map-based presentation and evaluation tool;
- a user friendly shell which integrates the different modules (a suite of so called Case Definition, Case Management and Case Analysis tools).

HABITAT SUITABILITY EVALUATION

The evaluation of habitat suitability within the scope of this study is based on standard Habitat Evaluation Procedure (HEP) as described by the US Fish and Wildlife Service (1980). In this study ecotopes (comparable with cover types in HEP), are not pre-defined in the study area. Instead, species habitats are identified and located in the study area (therefore including all ecotopes that provide support to the species). On the basis of distribution of habitats, identification of species' habitat requirements and determination of suitability defining factors such as the hydrodynamics, morphology and quality of the water system, a Habitat Suitability Index (HSI) is derived. Habitat requirements are derived from field observations and life history studies. The HSI is expressed as an index rating between 0.0 and 1.0, expressing the suitability of each environmental factor in the habitat. The overall habitat suitability is determined by combining separate indices per environmental factor.

The potential carrying capacity (PCC) of the habitat for a species is derived from the overall HSI in combination with the habitat area (A), expressed in habitat units (HU) and a given actual carrying capacity (CC) of the species (Duel et al., 1995).

pHSI = Function { field value, habitat requirements}

$$HSI = \text{minimum}\ (pHSI_1, pHSI_2, pHSI_{...}, pHSI_n)$$
$$HU = HSI * A$$
$$PCC = (HU_c / HU_a) * CC$$

where

HSI = habitat suitability index
$pHSI_1$ = partial HSI for environmental factor 1
HU = habitat units (ha)
HU_c= habitat units of a case
HU_a= habitat units of the actual situation
A = habitat area (ha)
PCC = potential carrying capacity (numbers or mass)
CC = carrying capacity of the actual situation

THE AZOV SEA PROBLEM

The Azov Sea is an integral part of the Black Sea water system. The Black Sea is semi-enclosed, the only exchange with the Mediterranean and the worlds oceans being the Bosporus. The catchment basin of the Black Sea (figure 2) covers an area of 2.2 million km^2, includes territories of more than 17 nations and is inhabited by approximately 162 million people. Economic developments in its drainage basin have affected the quantity and quality of the freshwater flow into the sea. Economic developments include the development of agriculture and industry in the drainage basin and on its sea shores, the exploration and transport of oil and increases in fishery efforts on the sea itself. Because of the enclosed nature, these developments have resulted in a severe stress of Black Sea ecosystems including the Azov Sea system.

Water balance and water quality of the Sea of Azov

Figure 2 Black Sea and Asov Sea drainage basins

The Azov Sea is a shallow inland sea on the north-east corner of the Black Sea. Its size is approximately 300 x 150 km, with a surface area of 35,000 km^2 and an average depth of no more than 9 meters. The total area the drainage basin is approximately 570,000 km^2. The average natural flow of fresh water into the Azov Sea is 43 km^3 per year, with large yearly fluctuations between 30 - 50 km^3.
The exchange with the Black Sea is mainly wind driven and can only take place through the Kerch Strait. The estimated residence time is between 10 - 20 years.

The quality of the Azov Sea water system is very much dependent upon the quantity and quality of the fresh water runoff from its drainage basin. The main influencing river is the Don, with an average natural flow of 28 km^3 (65% of total in flow) per year. The Kuban contributes approximately 12 km^3 (27%). The remaining inflow comes from more than 20 small rivers. Both the seasonal variations and the yearly variations of fresh water runoff are very high, causing frequent shortages of fresh water in the region. On the other hand, flooding of the lower reaches of the river Don and its estuary is caused by high runoff in spring, due to melting snow and rain (figure 3).

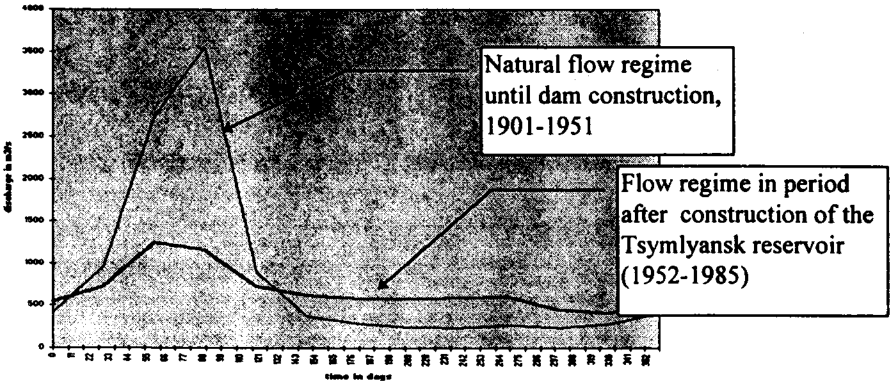


Figure 3: Historic and present flow regime of the river Don

In the 1920's and after 1945, agriculture, industry and urban settlements were developed at a large scale in the region. The need for a steady water supply for irrigation, drinking water, industrial use and navigation called for the construction of reservoirs. At present there are 130 reservoirs in the basin, containing up to a maximum of 35 km^3 of fresh water.

It is estimated that 50% of the annual runoff in the Azov Sea Basin is used for economic activities. The nonretrievable use has resulted in a reduction of the freshwater flow into the sea of more than 20% and has caused a steady increase in salinity from an average of 10‰ in the 1930's to almost 14‰ in the 1970's. The remaining flow is highly regulated, decreasing the frequency and severity of spring flooding. The quality of the remaining flow has been affected by emission of insufficiently treated or untreated waste water from industrial, domestic and agricultural sources. Concentrations of heavy metals, pesticides and oil products have been steadily rising until the economic decline of the 1990's.

Present status of habitat quality

The ecology of the Azov Sea is characterised by the very high spatial and temporal variability in physical, chemical and morphological conditions. The system in the sea itself is linked to bordering estuarine and riverine ecosystems. The coastal zone has extensive wetland ecosystems which form the transitional interface connecting the terrestrial drainage basin and the Sea of Asov itself.

The Sea of Asov coastal wetlands include habitats such as reed dominated marshes, forested riverine flood plains, inland lakes and lagoons, limans (a coastal lagoon with a salinity gradient), deltas, coastal lagoons and bays, and associated mud and sand flats, as well as artificial wetlands such as fish ponds, rice paddies and salt ponds (Wilson, 1994). Main wetlands are situated in the mouth of the river Don and Kuban totaling approximately 200,000 ha of wetlands, salt lakes and foredelta. The human impact on the water resources of the drainage basin has affected the habitats of species in the ecosystem. For many migrating fish species access to important spawning areas has been cut off by the construction of dams (e.g. Sturgeon species). Other spawning areas have decreased due to the decrease of frequently flooded land. The species diversity of primary producers and higher trophic levels has decreased and the ecosystem seems more vulnerable to disturbances like the introduction of the ctenophore *Mnemiopsis leidyi* from the Black Sea. The habitat conditions for this jellyfish have apparently become more favorable and have resulted in periods of total dominance, in which it outcompetes all other planktivorous species for food sources.

APPLICATION OF THE HABITAT EVALUATION PROCEDURE

The careful selection of key-species is important, based on the assumption that the selected species represent the status of the ecosystem as a whole. Species selection was based upon data availability of its habitat requirements and sensitivity to pollution, its geographical distribution and its commercial value. From the groups phytoplankton, zooplankton, zoobenthos and fish, a number of species were selected for further assessment. Life histories, habitat requirements and field data on actual distribution were collected for each species. The species represent typical marine/brackish and typical fresh water types. For phytoplankton, zooplankton and zoobenthos the characteristics of habitats are not further defined ("all-functions"). In principle, the whole sea area is available as potential habitat for these species. The selected benthos species constitute a large percentage of total biomass of zoobenthos in the sea of Azov. For fish species, important habitats in relation to its life-cycle (spawning, feeding and wintering habitats) were identified as much as possible.

The habitat quality of the following species is studied:

Phytoplankton

- *Sceletonema costatum*
- *Microcystis pulverea*

Zooplankton

- *Oithona nana*
- *Calanipeda aqua-dulcis*
- *Mnemiopsis leidyi*

Zoobenthos

- *Balanus improvisus*
- *Cerastoderma lamarci*
- *Mytilaster lineatus*
- *Nereis succinea*

Fish species

- Fluvial anadromous species: Bream (*Abramis brama*), Roach (*Rutilus r.*), Pike-perch (*Stizostedion l.*)
- Migratory species: Sturgeon, *Acipenser stellatus, A. guldenstadti*
- Marine species: Turbot *(Scophtalmus m.), Anchovy (Engraulis e.), Kilka (Clupeonella sp.), Gobius sp.*

Input of hydrodynamics and water quality model data

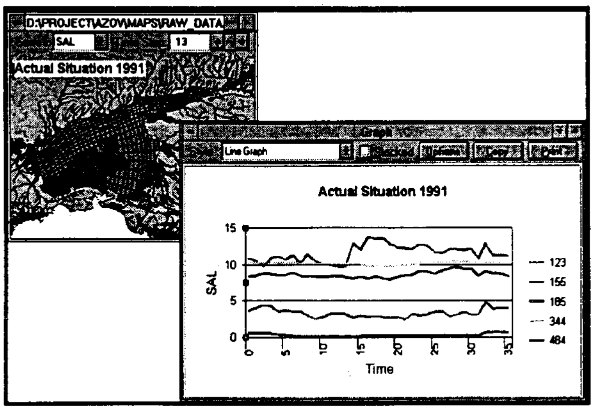
Results of the assessment are based upon the output of hydrodynamic and water quality models for three modelled cases. The following cases are studied:

1. Actual situation 1991 ("actual"), see figure 3
2. Best case with natural river flow ("pristine"), see figure 3
3. Worst case with 20% further reduction of river inflow ("20%rrf")

On the basis of the description of species requirements, the crucial spawning and larval feeding period (approx. the second week of April to end of August for most key species) was used to subselect data from the model dataset. The following model-parameters are used in the ecological assessment:

- Salinity
- Daily Averaged Oxygen concentration
- Night time Minimum Oxygen concentration
- Dissolved Copper[1]
- Dissolved Zinc[1]
- Total Oils[1] ([1] =model underestimates pollutant concentrations due to lacking loads)

The geographic figure shows a result of the actual situation for salinity at decade 13. The time series show clear salinity gradients in the Azov Sea for 5 gridcells going from the strait connecting the Azov Sea to the Black Sea (cell 484) to the mouth of the river Don (cell 123). Similar results are available for other environmental factors (figure 4).



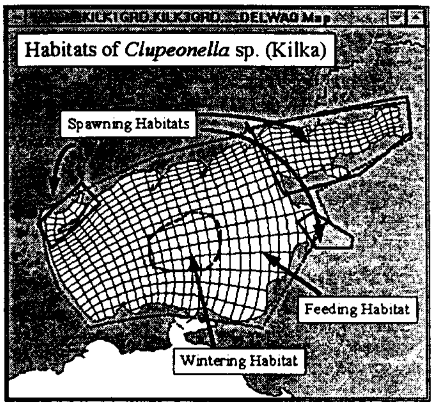


Figure 4: Salinity distribution derived from the model application

Figure 5: Distribution of *Clupeonella* habitats in the Azov Sea

Habitat mapping, habitat requirements and sensitivity to pollution

In order to perform a habitat suitability evaluation, habitats were mapped for each species. In figure 5 the distribution of the fish species *Clupeonella* is shown. In figure 6, habitat requirements for environmental factors available from the model are listed.

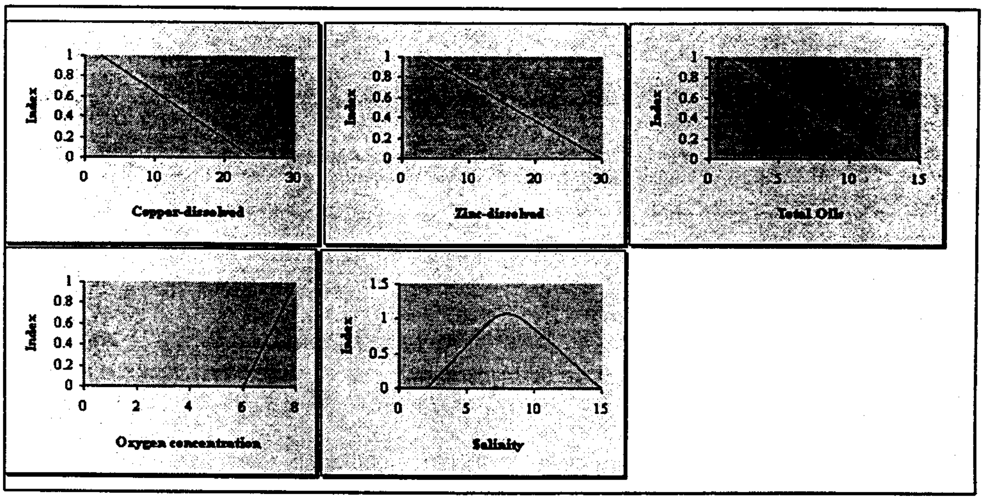


Figure 6: Spawning habitat requirements and sensitivities of *Clupeonella* for available environmental factors (Copper, Zinc and Total Oils in ug/l, Oxygen in mg/l, Salinity in 0/00)

Data on sensitivity to pollution were derived from Mance (1987) for copper and zinc, using lowest chronic LC50's found for marine fish larvae. An arbitrary factor of 0.10 was used to establish a lower limit. Total Oils, Oxygen and salinity data were provided by local experts.

Calculation of HSI's

For each combination of a species and its habitats a HSI was calculated. The HSI is the aggregation of indices for individual parameters over all gridcells within a theme. Figure 7 shows the partial HSI's per gridcell within a habitat for each environmental factor. Depending on the habitat requirements of species, changes in the environment will improve or deteriorate the suitability of the habitat. Figure 8 shows the impact of changing oxygen concentrations on habitat suitability.

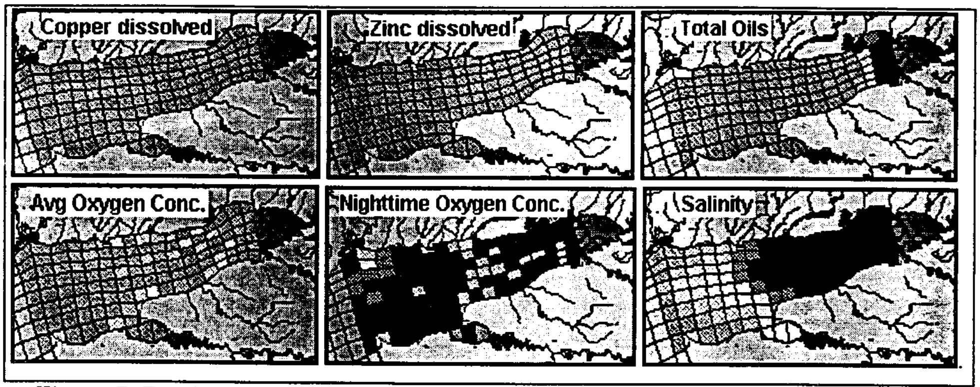
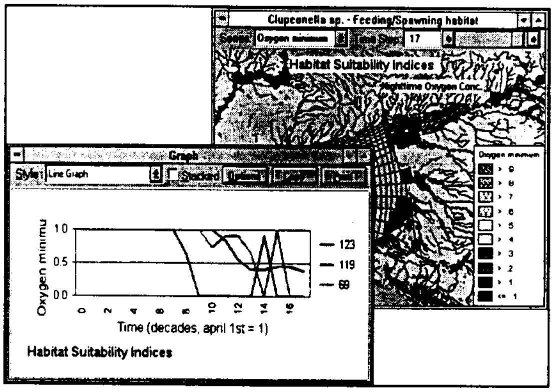


Figure 7: Partial HSI's per gridcell within Clupeonella spawning habitat for each environmental factor (decade 17, April 1st = 1). For legend see figure 8.



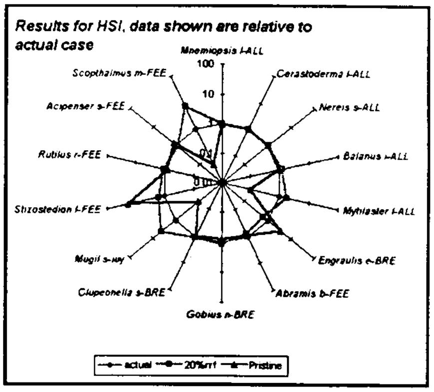


Figure 8: Impact of night time oxygen concentrations on habitat suitability for Clupeonella
Figure 11: Resulting HSI's for each species habitat

RESULTS OF THE ASSESSMENT

Given a set of environmental conditions, described within a case, some species will benefit, and others will be negatively impacted. From the three cases studied, "20%rrf" will tend to increase salinity by reduced river flow (when compared to actual). The case "Pristine" will result in a reduced salinity for large parts of the study area by increased river flow. This reasoning clearly indicates that impacts to typical fresh or marine species may be expected if salinity is the most limiting parameter.

Furthermore, it should be noted that the distribution of the habitats for each theme is fixed to the present known distribution. Results of three cases are shown in Figure 9. Figure 10 shows that oxygen concentrations have improved considerably in the pristine case. Habitat suitability is now limited only by salinity. See figure 11 for results of all species habitats.

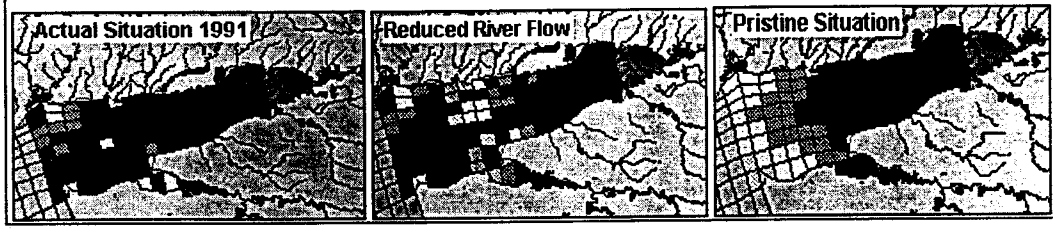


Figure 9: HSI's calculated for each case for spawning habitat of *Clupeonella*

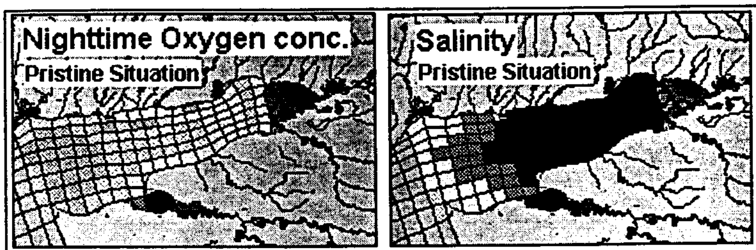


Figure 10. Most limiting factors in the pristine situation

From the results shown in figure 7 it can be seen that salinity and oxygen concentration are the factors most strongly affecting the habitat suitability for *Clupeonella*. Considering micro-pollutants, oil pollution resulted in somewhat reduced habitat suitabilities. Copper and zinc did not affect the habitat suitability of any species (not shown). However, concentrations of all pollutants are underestimated by the water quality model possibly at least by a factor 2 due to an at present incomplete set of waste loads data. From the results shown in figure 10 it is concluded that significant impact of management scenarios might be expected on the habitat suitability for a number of species. Within the limitations and assumptions of the approach, this results in increased or decreased carrying capacity expressed as biomass or catch (Table 1). Going from the actual situation to a less saline pristine situation, catch of fresh water species such as *Stizostedion*/Pike-perch increases. The migratory species *Acipenser*/Sturgeon clearly profits. On the other hand, for the marine species *Clupeonella*/Kilka the pristine case improves habitat suitability due to improved oxygen conditions and for *Scophtalmus*/Turbot, this case leads to a strong decrease of catch due to freshening of the system. In contrast, marine species profit if the river flow is further reduced (case "20%rrf"). Catch of Turbot is even increased to over 600% of the actual situation.

DISCUSSION AND CONCLUSIONS

The present selection of environmental factors within this study is based on the limited availability of parameters from the hydrodynamic and water quality modelling. In reality, other parameters will influence the suitability of habitats to species. For instance, it is well known that temperature ranges are of relevance to spawning of species.

Table 1: Calculated Habitat Units and Potential Carrying Capacity for Species Habitats

Species Habitat	habitat area (ha)	actual HU (ha)	20%rrf HU (ha)	pristine HU (ha)		actual CC	20%rrf PCC	pristine PCC
Calanipeda aqua-dulcis-ALL_FUNCTIONS	3692589	1749313	1235560	2291605	B	**5800**	4097	7598
Heterocope caspia-ALL_FUNCTIONS	3692589	600077	340519	1540913	B	**1000**	567	2568
Balanus larvae-ALL_FUNCTIONS	3692589	3434971	3487379	3328204	B	**1000**	1015	969
Acartia (Azov Sea)-ALL_FUNCTIONS	3692589	3422863	3329199	3325055	B	**300000**	291791	291428
Oithona nana-ALL_FUNCTIONS	3692589	1254282	2880005	50202	B	**1000**	2296	40
Mnemiopsis leidyi-ALL_FUNCTIONS	3692589	3367272	3438672	3233750	B	**3000000**	3063613	2881042
Cerastoderma l.-ALL_FUNCTIONS	929704	758790	742491	740204	B	**4500000**	4403336	4389777
Nereis succinea-ALL_FUNCTIONS	3692589	3163190	3135574	3067258	B	**500000**	495635	484836
Balanus improvisus-ALL_FUNCTIONS	1057360	768259	943801	293640	B	**1000000**	1228492	382214
Mytilaster lineatus-ALL_FUNCTIONS	438281	217470	383817	21370	B	**3500000**	6177224	343938
Engraulis encrasicolus-FEEDING	3443844	177824	112306	642188	C	**15000**	9473	54171
Engraulis encrasicolus-BREED/SPAWN	3443844	177824	112306	642188	C	**15000**	9473	54171
Abramis brama-WINTERING	893997	475985	390421	447002	C	**1500**	1230	1409
Abramis brama-FEEDING	893997	475985	390421	447002	C	**1500**	1230	1409
Gobius niger-WINTERING	120823	90292	102741	75487	C	**250**	284	209
Gobius niger-FEEDING	929704	598201	695578	482641	C	**250**	291	202
Gobius niger-BREED/SPAWN	21674	14640	17065	11604	C	**250**	291	198
Clupeonella sp.-WINTERING	303959	244647	186780	302265	C	**2000**	1527	2471
Clupeonella sp.-FEEDING	3692589	2712223	2225592	3140414	C	**2000**	1641	2316
Clupeonella sp.-BREED/SPAWN	671498	228566	266529	210008	C	**2000**	2332	1838
Mugil so-iuy-FEEDING	802303	47268	192787	4926	C	**435**	1774	45
Stizostedion lucioperca-WINTERING	1442969	1111722	972334	1401478	C	**1000**	875	1261
Stizostedion lucioperca-BREED/SPAWN	21420	13741	11157	19951	C	**1000**	812	1452
Rutilus rutilus-WINTERING	488277	333317	347143	276441	C	**200**	208	166
Rutilus rutilus-FEEDING	488277	333317	347143	276441	C	**200**	208	166
Acipenser guldenstadtii-WINTERING	447261	216139	165414	275264	C	**750**	574	955
Acipenser guldenstadtii-FEEDING	3692589	1908385	1555376	2295569	C	**750**	611	902
Acipenser stellatus-WINTERING	306900	146036	111215	187395	C	**300**	228	385
Acipenser stellatus-FEEDING	3692589	1908385	1555376	2295569	C	**300**	245	361
Scopthalmus m.-FEEDING	3256068	351803	2478106	15871	C	**300**	2113	14
Scopthalmus m.-BREED/SPAWN	262380	32427	199224	32427	C	**300**	1843	300

HU - Habitat Unit, CC- Carrying Capacity, PCC - Potential Carrying Capacity, Bold = '90-'94 averaged data obtained from local experts, B - biomass (ton), C - Catch (ton/yr), underlined PCC's give lowest value per species over its known habitats

Furthermore, additional contaminants such as pesticides and PAH's will induce extra stress to species. For the latter substances, no data were available to justify their modelling. For studied pollutants loading data are incomplete, resulting in an underestimation of ambient concentrations. At present calculated low levels, impact of oil is already seen, therefore worse pollution effects on habitat suitability may be expected. When looking to distribution data, it is now assumed that no changes in distribution occur in the future under different hydrodynamic regimes as identified in the cases that were studied. In effect, this will probably not be the case. For some migratory species (such as *Acipenser*/Sturgeon) a part of its habitat is not included in the study area. Improvement of habitat suitability within the study area does not necessarily imply improvement of habitats outside of the study area.

The calculated PCC could be affected by food limitations and other species interactions that are excluded in this approach. Improved PCC for the pristine case shows increases of a factor 6 for some species. However, reportedly increases of a factor 50 should be possible for some species. This indicates that (1) some limiting factors are not included and (2) habitat area is probably not a constant factor. These aspects need to be studied in more detail in future.

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CHANGES OF FLOOD DYNAMICS REGULATING FLOODPLAIN SYSTEM FUNCTIONING ON THE GARONNE RIVER, SOUTH-WEST FRANCE

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ABSTRACT

Floodplain - ecosystem functioning is directly linked to hydrology, hydraulics and geomorphology and therefore to flood dynamics, the driving force for floodplain - ecosystems. If it is well established that flood dynamics are controlled by climatic and/or morphodynamic changes, human activity also has to be considered in Western Europe drainage basins.

This study focuses on the causes of flood dynamic changes that alter floodplain - ecosystem functioning. It has been conducted on an upstream reach of the Garonne River (South-West France), where training works are moderate. Changes of flood dynamics which correspond to a decrease in flood-peak discharge and regularly flooded areas might be linked to climatic changes, but are certainly linked to Man's impact on the channel and the upstream drainage basin.

Two main human impacts have accelerated channel incision: gravel extraction and dam construction after 1960. The channel volume increased up to 30 to 40 % causing a recent increase of the mean bankfull discharge.

Bankfull discharge is usually associated with floods with a recurrence interval (T) of 1.58 years. In the studied river reach, discharges for a 1.58 yr flood were calculated for three time periods of 30 years between 1827 and 1993. For the last 30 years a value of 1 200 $m^3 s^{-1}$ was found. It represents the lowest value among the 1.58 flood discharges calculated among the three time series.

The lag between cross-section increase and flood-discharge decrease for a given return period indicates a perturbation of the dynamic equilibrium of the Garonne River and outlines Man's impact on the fluvial system.

One of the effects of channel capacity increase is demonstrated by flood peak propagation times. Flood peak propagation in the reach under study can be considered according to two phases. Below bankfull stage, flood peak propagation attains 11 km h^{-1} whereas during overbank flooding the flood peak propagation may slow down to 5 km h^{-1}. Thus, if high frequency floods don't produce overbank flooding, flood peak propagation times are generally higher.

Contemporary flood dynamic changes are determined and examined. Since these hydrological and closely related geomorphological changes are altering the floodplain - system, they have to be known for a further understanding of the ecological functioning of these river ecotones.

KEY-WORDS: Garonne River, human impacts, channel incision, channel adjustment, bankfull discharge, runoff fluctuations, overbank flow, flood dynamics, floodplain, flood peak propagation, roughness coefficient

INTRODUCTION

Most river systems in Western Europe have endured direct or indirect human impacts (Petts, 1984). The interrelationship between flow regime, sediment load transport and channel response has been recognized for quite a long time (Fargue, 1868; Leopold and Maddock, 1953; Schumm, 1969). Especially the importance of single flood events of high magnitude and low frequency, as well as bankfull discharge or dominant discharge, for channel and flood plain changes is well known (Tricart, 1960; Wolman and Miller, 1960; Beven and Carling, 1989; Gurnell and Petts, 1995).

Case studies show the different adjustments of fluvial systems to changes of the river channel control variables (e.g. Babinski, 1992; Sear, 1995; Ibanez, Prat and Canicio, 1996). The hydrological regime, stream channel and flood plain morphology is a function of the characteristics of the whole drainage basin including geology, geomorphology, vegetation, land use and sediment sources and availability (e.g. Gregory and Walling, 1973; Lyons and Beschta, 1983). There was always a strong interference of human activities, the natural conditions of the environment and the related channel adjustments (Darby and Thorne, 1992; Thoms and Walker, 1992; Ligon et al., 1995). For example, land use changes due to agricultural practices can increase erosion of fine particles and, consequently, increase the particulate matter transport in the draining river (Schlosser and Karr, 1981; Loughran *et al.*, 1986; Quine and Walling, 1993).

The geomorphological implications of channel and floodplain changes affect not only the physical, but also the biotic functionning of the aquatic-terrestrial transition zone or ecotone. Therefore, a holistic approach of the geomorphological and ecological processes of the whole system are necessary.

The historical changes of the Garonne River, the urban, industrial and agricultural developments are outlined by Décamps *et al.* (1989). This paper focuses on more recent hydrological, geomorphological and related ecological effects due to Man's activity.

THE UPPER GARONNE RIVER

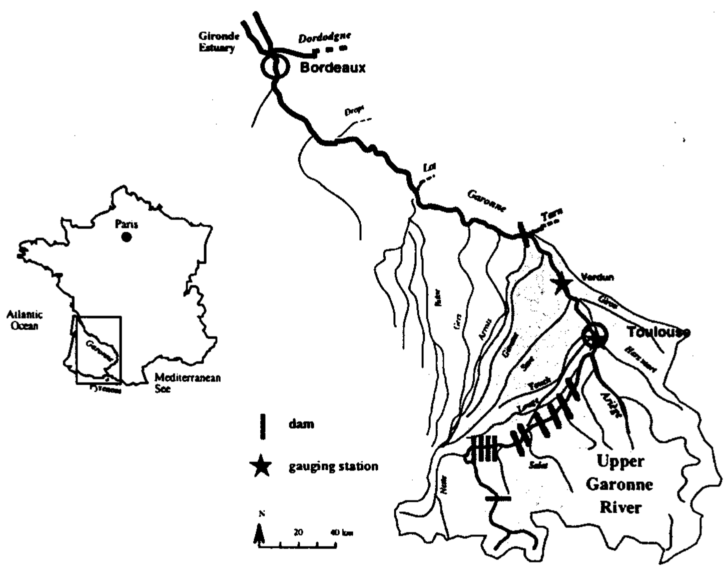

Figure 1: The Garonne River and the study area.

The Upper Garonne River, SW France covers a drainage basin of about 32 350 km^2 upstream the confluence with the Tarn River (Fig. 1). The total length of the river from the spanish Pyrenees to the Gironde estuary north of Bordeaux is 478 km, 247 km drain the upper basin. Steep gravel bed mountain rivers with gradients higher than 0.003 drain forested watersheds. The Upper Garonne River is either a rather straight or a low meandering river where no distinguished braided zones can be recognized. The river dams on the stream channel of the Upper Garonne River were mainly constructed to produce hydro-electricity.

Flow Regime

The mean abundance at Toulouse is 194 m^{3}s^{-1} (19.4 l^{-1}s^{-1}km^2). More than 30 % of all floods which attained at least two meters at Toulouse and those with highest peak discharges occured during the two spring months of May and June (Fig. 2), when precipitations are high.

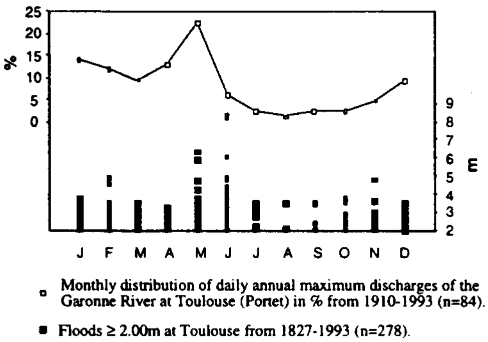

Figure 2: The general flow regime of the Garonne River at Toulouse

Comparing the lowest discharge ever measured (17.1 m^{3}s^{-1}) and the highest discharge estimated from the catastrophic flood of June 1875 (8 000 m^{3}s^{-1}), the excessive character of the regime becomes evident. The hydrographs from 1977 and 1989 at Verdun (Fig. 3), situated in between Toulouse and the Tarn tributary (Fig. 1), show the high flow variability of the Garonne River. Sudden increases of discharges by factors of up to seven are observed.

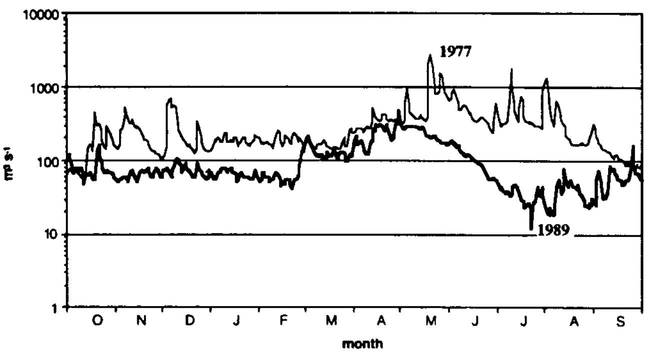

Figure 3: Daily discharges at Verdun during a high water level year and a low water level year.

RUNOFF FLUCTUATIONS

During the 19th century more natural catastrophes such as avalanches and flood events generated in the Pyrenees (Métailié, 1991; Steiger, 1990; 1991). However, their main causes are still subject of controversy.

Changes in the precipitation regime as well as the deforested and overexploited Pyrenees landscapes at this period are accused. Most probably though, a combination of these two factos is responsible for this phenomen.

During the 20th century, a rather humid period from 1910 till 1940 can be distinguished in reference to the interannual mean of 194 m^3s^{-1}, followed by a dry period during the 1940s till the end of the 1960s (Fig. 4). From 1970 until 1982 higher annual discharges have been observed. During the 1980s the trend to a dry period was as distinguished as during the 1940s. The general cyclic trend of higher and lower annual discharges has to be considered as natural (Probst, 1989).

Irrigation of water indigent cultures such as corn augmented significantly after the 1960s, and therefore does not significantly affect low flow nor annual discharges prior to this period. After this period, low discharges during the 1980s were accentuated by an increased water consummation. However, these low discharges are also related to low precipitations and therefore to a decrease of water storage in aquifers and a decrease of snow and ice storage in the high Pyrennes mountains (Lambert *et al.*,1989).

Pumping in the river and aquifer, as well as the functioning of hydropower plants, may influence daily discharges, especially at low water flow during the summer and irrigation season. Recent estimations for the Garonne River upstream Toulouse suggest that only 5.5 % of the total annual discharge are prelevated by drinking water, industries, irrigation and water transferts to canals (Agence de l'Eau, 1994). However, the percentage increases drastically during low water periods. 30 to 50 % of the discharge might be removed. Thus, even monthly abundance may be perceptibly changed.

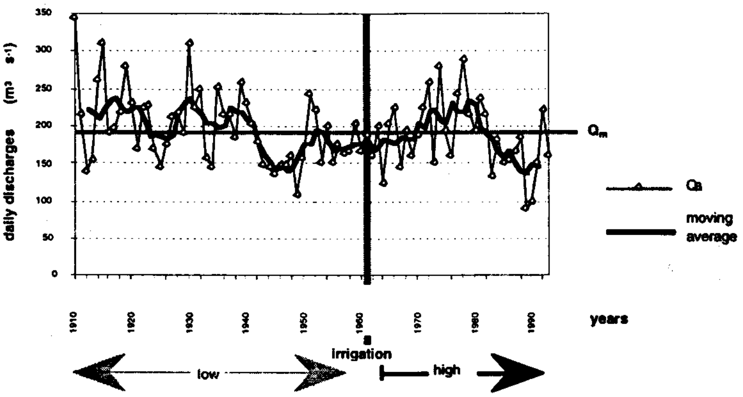

Figure 4: Moving averages (5 years) for annual discharges at Toulouse between 1910 and 1993. Water intake due to irrigation practices was low before the 1960s and increased since.

BED LOAD DEACREASE AND CHANNEL INCISION

At least since the end of the last century, the channel bed of the Upper Garonne River was never completely covered by a coherent layer of gravels and cobbles. Harlé, Serret and Denizot observed respectively in 1895, 1900 and again in 1953, a few years before the beginning of industrial extraction of gravels in the river bed, that the calcareous molasse, the bedrock, appeared in the river channel.

These observations confirm the fact, that no braiding of the Upper Garonne River has been documented. Braiding of a river is correlated to steep gradients and also to high bedload charges (Leopold and Wolman, 1957). The small part of the Garonne's catchment in the Pyrenees mountains (<15 %), relative low relief energies and little glacial activity are certainly responsible for low bed load supply. The first locally introduced artificial bed load retention artifices in altitudes from about 1 500 m to 2 600 m have been constructed in 1870 and in the 1920s and 30s on natural tams, deep lakes formed in abandoned cirques. To our knowledge no studies have been published to estimate the alteration of the bedload transport of the upper Garonne River by these first constructions on the river network in the Pyrennes Mountains. The major hydroelectric power plants on the river in between the Pyrenees and Toulouse have only been constructed after 1960. This cascade of dams stops the bedload supply to the downstream reaches.

Intensive industrial gravel extractions in the main channel downstream Toulouse started in the middle of the 1960s. Thus, on one hand bed load was extracted artificially and on the other hand bed load transfer from the upstream sections was interrupted by the construction of dams. Today, the only bed load sources consist on bank failures. However, even though that the Garonne River has never been canalized in this section, at least 30 % of the river banks, especially concave banks, were stabilized after the high magnitude flood of February 1952 between Toulouse and the Tarn River (S.M.E.P.A.G., 1989).

A net increase in channel depth during the last 20 to 30 years was observed. A channel incision of 1.50 m in only 22 years, from 1959 to 1980 (66 mm y^{-1}) and 0.78 m from 1970 to 1984 (55 mm y^{-1}) were estimated by Beaudelin (1989)). Furthermore, field observations after a series of floods in the first half of the 1990s revealed a tendancy to bank erosion. Local bank failure and therefore channel width increase up to several meters has been observed after the last high magnitude floods in 1991, 1992 and 1993. Channel capacity could increase by 30 to 40 %.

BANKFULL DISCHARGE

Bankfull discharge, which can be associated to the dominant discharge concept, plays an decisive role in fluvial morphology. It is considered as an important parameter controling channel and flood plain morphology (e.g. Wolman and Miller, 1960). Dury (1973) and other authors found a mean return period of 1.58 yrs for this specific discharge. Despite some controversials about this value (e.g. Williams, 1978) it is still used as an indicative value. Furthermore, it is well accepted now, that the river bed is a product of a range of discharges rather than only of one single discharge (Biedenharn and Thorne, 1994).

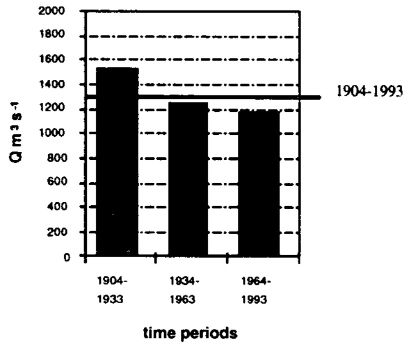

Figure 5: Discharge with a return period of 1.58 years calculated for different time periods at Toulouse.

To better understand the evolution of the theoretically discharge with a return period of 1.58 yrs, it has been calculated for several time periods (Fig. 5). The discharge associated to a return period of 1.58 yrs decreases from the beginning of the century till today. The lowest value of 1 200 m^3s^{-1} was obtained for the period from 1964 to 1993. This means, that the calculated bankfull discharge decreased by about 30 % from 1 500 m^3s^{-1} at the beginning of the century to 1 200 m^3s^{-1} nowadays.

CHANNEL ADJUSTMENTS

According to the conceptual model of channel adjustments to runoff and bedload changes, it would be expected that a decrease of a representative discharge (Q_w) and a similar decrease in bed load (Q_{sb}) induces a decrease in channel width, in the width-depth ratio and in meander wavelength (e.g. Knighton, 1984). Contrarily, sinuosity should increase, but changes to depth and slope would remain constant.

(1) $$Q_w^-, Q_{sb}^- \approx w^-, d^\pm, (w/d^-), \lambda^-, S^+, s^\pm$$

(Knighton, 1984)

Q_w	= a representative discharge	Q_{sb}	= bed load
w	= channel width	d	= channel depth
w/d	= width-depth ratio	λ	= meander wavelength
S	= sinuosity	s	= channel gradient
+	= increase	-	= decrease

Observed channel adjustments on the Garonne River to discharge and bed load decrease don't correspond to the adjustments proposed by the general model. This situation translates Man's impact on the fluvial system. The main causes are dam construction hampering bed load transfer downstream and gravel extraction in the river channel. The river adjusts its stream power by stream bed erosion. Stabilization of concave banks stopped natural meandering, but lateral erosion occurs on oversteepened and fragilized banks, which have not been protected.

If channel incision is higher than the channel width increase, the width-depth ratio actually increases. Otherwise it might stay stable. Meander wavelength and sinuosity don't change due to the stabilization of the concave banks. Channel gradient probably increases, but no topographic survey could be undertaken.

OVERBANK FLOW

The increase of the cross-sectional area, due to channel incision and channel widening during the last three decennaries has modified the frequency of flood plain inundation downstream Toulouse (Steiger and Gazelle, 1994). Especially low frequency floods are concerned and not as much high magnitude floods. For a same discharge of 1 250 m^3s^{-1} the gauging height at Verdun decreased by 84 cm between 1961 and 1992. Hence, today higher discharges are needed to cause overbank flow. Consequently, the lowest flood plain level is inundated less frequently.

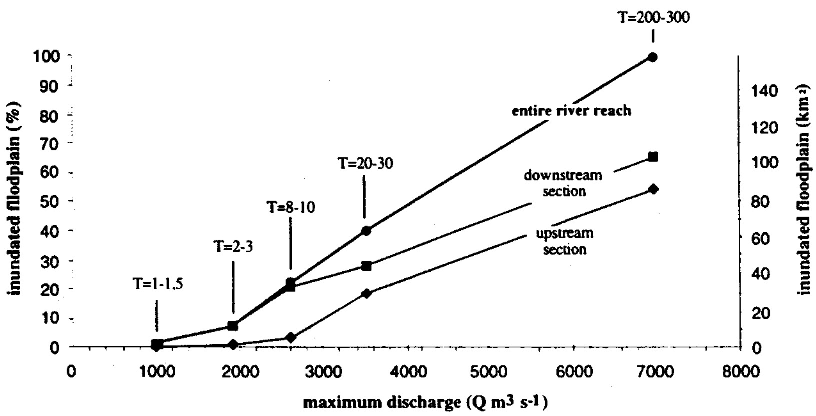

Figure 6: Relationship of the inundated floodplain area and the maximum discharge of different floods with varies recurrence intervals (T) of the Garonne River between Toulouse and the Tarn affluent.

In Fig. 6 the relationship of peak discharge and the inundated flood plain area has been established for several floods of different magnitudes. The entire river reach between the Toulouse and the Tarn tributary of 70.6 km channel length has been divided into an upstream reach (23.0 km) and downstream reach (47.6 km). Several differences between the increase of the inundated surface and the increase of the maximum flood discharge are observed. The first floodplain levels of the upstream reach are more constrained than on the downstream reach. Therefore, a net increase of the inundated floodplain area occurs between 1 900 m^3s^{-1} and 2 600 m^3s^{-1} on the downstream reach, while a comparable increase occurs only between 2 600 m^3s^{-1} and 3 500 m^3s^{-1} on the upstream reach.

The graph for the entire river reach shows a threshold value at 1 900 m^3s^{-1}, which corresponds to the mean annual flood with a recurrence interval of 2 to 3 years. From this discharge on, inundated floodplain surface increases almost linearly with an increasing discharge. A linear regression model was established (2). Thus, the inundated floodplain surface (ISF) corresponding to discharges higher than 1 900 m^3s^{-1} can be estimated for the Garonne River between Toulouse and the Tarn River.

(2)
$$ISF(km^2) = -38.721+0.028\ Q\ (m^3s^{-1})$$
$$r^2 = 0.999;\ P = 0.0006$$

Overbank Flooding Index

In order to compare inundated floodplain areas of different river reaches, an Overbank Flooding Index (OFI) was calculated for seven sub-reaches.

OFI: inundated floodplain area / channel length

$L^2 / L = L$

It exists a statistically significant correlation between the Overbank Flooding Index for the annual most probable flood and the channel gradient (Fig. 7). An increase in channel gradient induces a decrease in inundated floodplain area. Water velocity increases and flood water is evacuated more quickly.

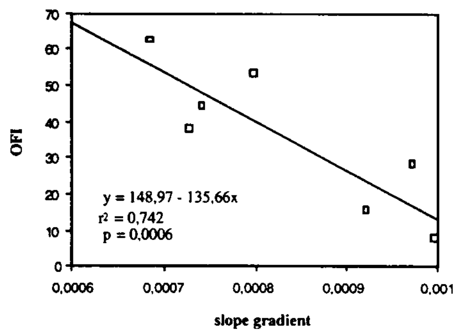

Figure 7: Relationship between the Overbank Flooding Index (OFI) and the river bed slope gradient.

However, no significant correlation between the Overbank Flooding Index for the mean annual or less frequently floods and the channel gradient exists. Therefore, channel gradient has an influence on floodplain inundation during low magnitude floods, but not during high magnitude floods. The sinuosity coefficient does not have a significant influence on the inundated floodplain surface. No statistically significant correlations between the Overbank Flooding Index and the sinuosity coefficient were found.

Flood Wave Propagation

For a long time men were intrigued by the spontaneous, rapid and often catastrophic floods of the Garonne River at Toulouse (Serret, 1900; Pardé, 1935; Gazelle, 1984; Lambert, 1982). The dams built on the Garonne River during the 1960s for hydroelectricity production (Fig. 1) do not significantly influence flood peak propagation. The size of the reservoirs was not calculated for this purpose. Besides, their retention capacity is highly reduced due to an important upfilling by sediments.

Downstream Toulouse, where significant overbank flooding may occur, the influence of the floodplain on flood peak propagation (FPP) was examined. The gauging stations are situated at Toulouse and Verdun (Fig. 1).

$$FPP = \frac{L}{\Delta tp} \ (\mathrm{km\ h^{-1}})$$

FPP = flood peak propagation (km h^{-1})
L = distance between two gauging stations (km)
Δtp = flood peak propagation time between two gauging stations (h)

Flood peaks of high frequency floods without important overbank flooding (800 - 1200m^3s^{-1}) progress with celerities of 8 to 10kmh^{-1} between Toulouse and Verdun (Tab.1). Flood peak celerities are two times slower for flood events with peak discharges of 1 800 to 2 700 m^3s^{-1}, covering large parts of the adjacent alluvial plain. The threshold discharge is about 1 200 m^3s^{-1}, corresponding to the bankfull stage.

Table 1 Flood peak celerity times for flood events below and above bankfull discharge between Toulouse andVerdun (50.7 km channel length).

Month	Year	max. flood gauge height at Verdun (m)	Flood peak celerity Toulouse-Verdun (km h^{-1})
March	1991	3.54	8.5
April	1992	3.95	10.1
Mai	1992	3.05	9.2
December	**1993**	**5.42**	**10.1**
Mai	1991	5.24	5.1
Juin	1992	5.51	5.1
September	1993	5.88	5.1
October	1992	5.91	5.5

In general flood peak propagation celerity depends as well on water velocity, which is affected by vegetation, as on the presence of floodplains. The presence of the flood plain changes the cross-sectional area between floods without and with overbank flooding. During overbank flooding the cross-sectional area increases and thus, flow velocity may decrease ($v = \frac{Q}{A}$). Flow velocities may also decrease due to a higher roughness coefficient on the floodplain.

However, flood peak celerity of the flood of December 1993 is twice as fast as the other above bankfull stage floods with maximum flood peak heights of more than 5.00 m at Verdun. Assuming that the cross-sectional area between the floods of December 1993, Mai 1991 and June 1992 did not change significantly, other parameters affecting flood peak celerity have to be examined.

The spatial distribution of precipitation on the drainage basin might have favoured higher flood peak celerities during the flood of December 1993. But even though precipitations were slightly higher in the Pyrenees comparing the flood of December 1993 to those of Mai 1991 and June 1992, differences were not high enough to explain the observed phenomen.

Change of the roughness coefficient of the floodplain may have played a role. Vegetation is largely developing in May and June, while winter interrupts the vegetative period in December. Manning's roughness coefficient (n) may change on floodplains with medium to dense brush in summer from 0.160 to 0.045 in winter (Gregory and Walling, 1973). Using these values, mean flow velocity on a floodplain with a flow depth of 1.00 m and a width of 100 m, estimated with Manning's equation could therefore increase from 0.2 m s^{-1} to 0.7 m s^{-1} according to the season.

(3)
$$v = \frac{1}{n} r^{2/3} s^{1/2} \ (\mathrm{m\ s^{-1}})$$

v = mean velocity
r = hydraulic mean radius
s = slope of river bed
n Mannning's roughness coefficient

The differences between the inundating flood of December 1993 with high flood peak progression times and the other inundating floods, can also be explained by the time sequence of floods. Time sequence is another important factor in controlling the varying rates of recovery from flood-produced changes. Beven (1981) points out that the effectiveness of floods of a given magnitude will vary significantly depending on preceding events, or 'event ordering'. That is, effectiveness or efficiency of an event will be more efficient if it follows a preceding event quickly (Kochel, 1988). Geomorphic effectiveness is defined as the modification of landforms (Wolman and Gerson, 1978). In the case of accelerated flood wave propagation during a high magnitude flood we might call effectiveness higher progression times and therefore changes of the hydraulics and morphological processes. The flood of December 1993 has been preceded by a high magnitude flood (T=8 years) in September 1993. This preceding event has modified the roughness coefficient by weighing down the vegetation which could not recover before the December flood event. Thus, in relation with the seasonal effect, the event ordering also influenced directly the flood peak propagation time of the largely inundating flood of December 1993.

CONCLUSION

The Upper Garonne River supports less river training works compared to the downstream section and other european rivers. Main human interferences on the river channel are experienced only after the middle of this century. Therefore, the phenomena observed today on the Garonne River lag behind others rivers, such as north alpine rivers. However, these anthropogenic interferences on the Upper Garonne River alter directly fluvial dynamics which control ecological processes on the aquatic-terrestrial interaction zone or ecotone.

The flow regime is subject to natural cyclic trends, but water discharges may significantly be altered especially during low water periods. None of the major constructions on the Upper Garonne River hamper significantly flood peak propagation. But channel adjustment to bed load decrease and river training works by channel incision and widening has an influence notably on the flood peak propagation time of high fequency floods.

The most probable annual flood (T=1-1.5 yrs) covers less than 5 % of the whole floodplain surface. The mean annual flood (T=2-3 yrs) does not even submerge 10 % of the total floodplain. These surfaces have a tendancy to decrease, affecting floodplain system functioning. Especially the most dynamic and fragile riparian zone boarding immediately the river is mostly disturbed. Agricultural impacts and the decrease of overbank flooding reduced the riparian vegetation and only incoherent patches of riparian zones sustain. A significant riparian forest decline during the last 10 to 15 years was observed (James, 1996).

The anthropogenic impacts such as bank stabilization and gravel extraction favoured the recent channelization of the upper Garonne. In the river system vertical accretion processes are now the dominating floodplain construction processes. A gradual decrease of accumulation rates might be expected, altering nutrient cycling on the floodplain.

This study shows some of the closely linked geomorphological and ecological interactions of modern human impacts on a fluvial system. Even though river training works on the Upper Garonne River have to be considered as moderate compared to other European Rivers, irreversible changes are taken place.

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A PRELIMINARY TELEMETRY INVESTIGATION ON THE OBSTACLES TO ANADROMOUS SALMONIDS MIGRATION IN SPAWNING STREAMS OF THE BELGIAN ARDENNES (RIVER MEUSE BASIN)

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ABSTRACT

In the course of the 'Meuse Salmon 2000' programme aiming at the restoration of the Atlantic salmon *Salmo salar* and sea trout *Salmo trutta* in the River Meuse Basin, most large dams are progressively equipped with fishways to restore the free circulation of spawners between the North Sea and the first major spawning streams, the River Ourthe and its tributaries. Spawners entering the River Ourthe would still be confronted to so-called minor obstacles, aiming at water regulation for tourism purposes but of which the actual impact on fish migration has never been investigated. In order to test for the actual free-circulation of salmonid spawners in the upper River Ourthe and to locate potential spawning grounds, a probe fish (489 mm FL male sea trout) was tagged with an intraperitoneally implanted radio-transmitter. From the 18th of ovember 1995 onwards, the trout was tracked in a part of the river (44 km upstream of the confluence) which was thought to be devoid of any major obstacle to fish migration. Three days after its release, the trout had migrated over 6 km up to a small weir (1.8 m high). During four consecutive days, the trout was consistently located downstream of the weir but no successful climbing was observed, reflecting the poor efficiency of the central fishpass under dry weather conditions during summer and autumn. The trout then settled in a deep run habitat, 150 m downstream of the weir and no upstream excursion was recorded until the first major rise of water level, four weeks later, even when the weir was opened for water regulation purposes. When the water level was maximum (24th of December), the trout moved upstream of the weir and migrated over 28 km during the next 72 hours up to a spot identified as a potential spawning redd from habitat features, and where it was consistently located till the 31st of December. These results, though most preliminary, clearly indicate that even minor obstacles may cause a substantial lag in trout migration of which the impact on spawning success remains to be determined. Since similar minor man-made obstacles are most frequent in the salmonid spawning streams of the Belgian Ardennes, it is thus uncertain that migratory trout having successfully climbed the major obstacles since the North Sea would find their way to the spawning redds. As a corollary, it is suggested that more detailed case studies should be undertaken, ideally via the use of telemetered probe-fish, in order to provide management policies that would represent a suitable compromise between users of water resources with apparently conflicting interests (water regulation, tourism, nature conservancy).

KEY-WORDS: Hydraulic Works / Dam influence / *Salmo trutta* L. / Migration / Radio-tracking / Belgium.

INTRODUCTION

The Atlantic salmon *Salmo salar* and the sea trout *Salmo trutta trutta* of the river Meuse Basin, which once represented an important natural resource in Belgium, The Netherlands and France, have totally disappeared from this river basin since the 1930's. This extinction was mainly due to water pollution and the building of weirs and dams for navigation, hydropower generation and river regulation purposes. Significant efforts have been made to improve the water quality and the restocking of juveniles gave encouraging results. However huge weirs and dams still compromise the successful recolonisation of the river basin by adults. In the course of the 'Meuse Salmon 2000' programme aiming at the restoration of these species in the River Meuse Basin (Philippart, 1985; Philippart *et al.*, 1990; Philippart *et al..*, 1994, Philippart *et al.*, 1996), most large dams are progressively equipped with fishways to restore the free circulation of spawners between the North Sea and the first major spawning streams, the River Ourthe and its tributaries (figure 1). Spawners entering the River Ourthe would still be confronted to so-called minor obstacles, aiming at water regulation for tourism purposes but of which the actual impact on fish migration has never been investigated. The aim of this preliminary study was to describe the behaviour of trout facing these so-called minor obstacles in order to test for their real impact on migration patterns.

METHODS

<u>Study Area</u>

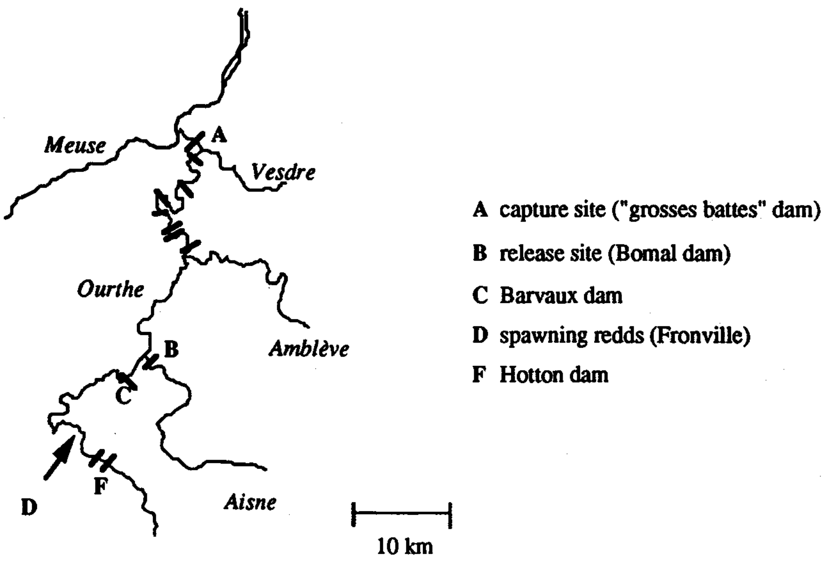


Figure 1: Map of the study area, the River Ourthe sub-basin

The study was conducted in the Belgian Ardennes, in the main tributary of the River Meuse, the River Ourthe and one of its sub-tributaries, the Aisne stream (figure 1). The study site was located in between the villages of Bomal and Hotton even thought to be devoid of any major obstacle to fish circulation. Water temperature and level fluctuations were measured daily (0.1°C and 1 cm reading accuracy, respectively). Water flow data were communicated by the SETY (Ministry of Equipment and Transport, Walloon Region).

Fish tagging

On the 17th of November 1995, a wild sea trout (male, 489 mm FL) was captured by electrofishing (EPMC generator, 2.5 KVA, DC) just downstream of the "Grosses Battes" dam on the River Ourthe. The fish was then transferred 45 km upstream, equipped with a 40 MHz activity radio transmitter (BE-512 model; ATS, Inc.) then released 200 m downstream of a small dam in the Aisne stream (figure 1).

The implantation procedure of the transmitter went as followed. The trout was anaesthetised with a 0.25ml l^{-1} solution of 2-phenoxy-ethanol. Once it had reached the tolerance stage (± 5 min), it was placed upside down into a support made of wet paper, that was adjusted to its shape (R.S. McKinley, Univ. Waterloo, Canada, pers. comm.), always making sure its gills were kept under the anaesthetic solution . A midventral incision was made between the pelvic girdle and the papilla, with its length minimised (±3 cm in average) to enable the passage of the alcohol sterilised transmitter with a slight external pressure. The incision was closed by two separate stitches, 9-10 mm apart, using sterile catgut (2.0 Dec) on cutting needles. The trout recovered its posture and swimming around 3 min after surgery and was transferred to the study area .

The trout could be detected at a maximum distance of 500 m (depending on local environment). It was located every day since the 21st of November 1995 with a Fieldmaster radio receiver and a loop antenna (ATS) and its position was determined by triangulation. It was located to an accuracy of $2m^2$ by reference to labelled marks on the banks of the river (at the time the trout was downstream of the Barvaux dam). Locations were carried out as frequently as every 2 to 10 min during the tracking.

RESULTS AND DISCUSSION

Once the trout was released, no obvious deviance from normal behaviour was observed as long as it was located (± 2 hours). The next day, the trout had left the tributary and started migratory upstream the river Ourthe. On the 21st of November 1995, the trout was found just 400 m downstream of a small mobile dam (Barvaux dam, see figure 1). This small dam (1.3 m high), constituted of two separate parts which can be elevated or lowered via hydraulic arms, is equipped with a central fish pass consisting of 7 successive 1 m^3 pools (plate 1).

During the next three days, the trout was located below the dam and on two occasions, it was observed trying to clear it, but in vain. Precise locations were undertaken in order to understand the behaviour of the trout as it faced the obstacle and to detect whether the trout would use the fish pass or not to clear it. It was obvious that the fish pass did not attract the fish which was mainly located near the left weir or near a sewer on the left hand bank (figure 2). In addition, if the trout would have found the entrance to the fish pass, it would further had to leap into it as the first pool was about 20 cm above the river level due to extremely low autumnal rains. Similar observations were made on each tracking day until the 24th of November 1995. From the 25th to the 28th of November, the trout was located about 100 m downstream of the dam and no more attempts to clear the dam were observed. On the next day, the trout moved another 100 m downstream into a deep run habitat (suitable habitat for adult trout). During the following weeks, it was consistently located in this habitat, only leaving it for short excursions (± 30 m) which were regarded as feeding behaviour. The situation remained unchanged until the 22nd of December despite the mobile dam had been lowered 9 days earlier for water regulation purposes. As a matter of fact, the trout did not move until the water level started increasing on the 23d of December 1995 when we located it 60 m downstream of the dam and just below the dam on the next day (figure 3).

Plate 1: Downstream view of the Barvaux dam

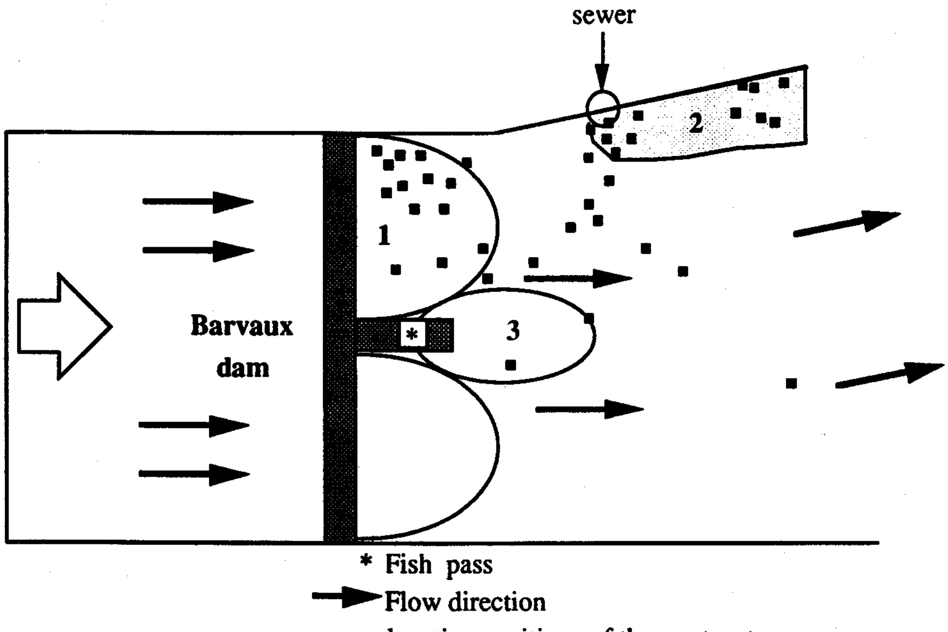
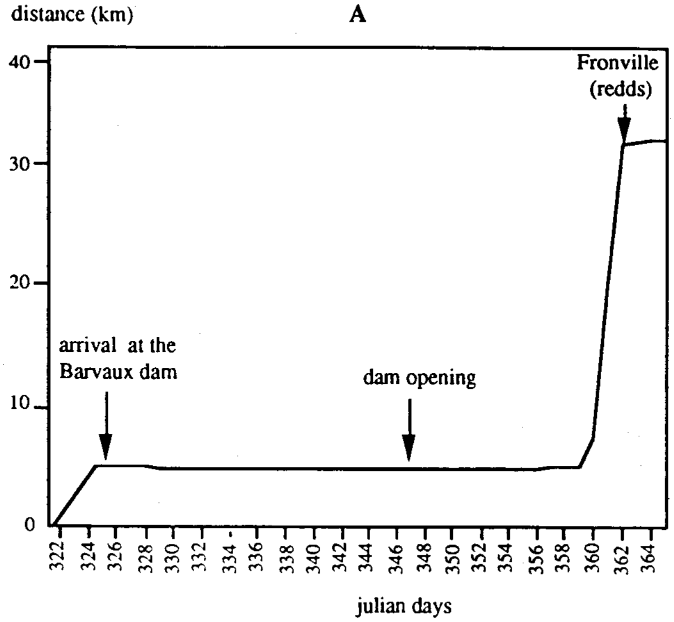


Figure 2: Locations of the radio tracked sea trout below Barvaux Dam on the 23d of November 1995 (Julian day 326) in between 13:07 and 15:06 h. The influence areas of spillways, sewer and fish pass are represented by ellipses and numbers 1, 2 and 3, respectively.

On Christmas day, it cleared the dam and was located 2.5 km upstream. The mean water flow increased from 5.3 m^3s^{-1} on the 20th of December to 65.8 m^3s^{-1} on the 24th of December then decreased to 61.8 m^3s^{-1} on the 25th of December. From the moment the dam was cleared, the length of upstream daily journeys increased substantially: 2.5, 12.5 and 13 km on the 25, 26 and 27th of December respectively. On the next day, the trout was located on the edge of a potential spawning redd (in Fronville, see figures 1 and

3) where it remained until the 31st of December 1995. No spawning activity was detected at the time of the day (13:00-17:40 h) when the trout was located.



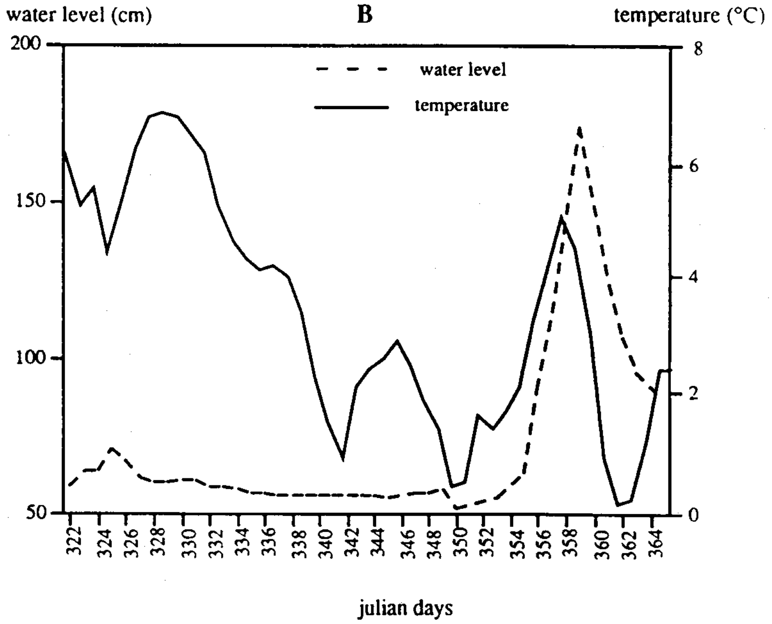


Figure 3: A. Daily movements of the radio tracked trout related to its release site, from the 18th of November 1995 (julian day 321) to the 1st of January 1996 (julian day 1). Point (321,0) corresponds to the release site (Bomal dam). B. Variations of water level (cm) and temperature (°C) in the River Ourthe during the study.

Further tracking (January-late March) indicated that the trout resumed its upstream migration from the moment water temperature was above 3°C. It travelled to a second dam (1 m high) in Hotton (figure 3) which it did not clear, probably due to low water levels. It then moved 2 km downstream and settled in this area until the third decade of March 1996, when it started a downstream migration down to Barvaux. Most movements were favoured by high water levels but their lengths were substantially less at low temperature.

CONCLUSIONS

This preliminary study using a probe fish (*Salmo trutta* L.) clearly shows how a small dam, thought to be insignificant towards the free movements of fish, can disrupt and/or enable the upstream spawning migration of anadromous Salmonids. These first results also show that this is not a local problem (limited to the Barvaux dam) as a similar problem was observed at the Hotton dam during the study. Therefore, the locations of the trout near the dam made by a precise telemetry tracking technique, enabled us to understand that the interruption of the migration would be due to a mismatch in the conception of the fish pass, at least in its functioning under low water levels. Since Salmonid spawners enter tributaries or resume their spawning migration in early autumn, it would be crucial that these mobile weirs be lowered as early as the end of October (if the meteorological conditions would permit it) to enable the free access to spawning redds. The creation of all these small dams in the study area is a result of increasing tourism activity, with dams mainly aiming to maintain minimum levels for water sports during spring and summer, when the water levels are usually low. The natural richness of the region developed its attraction for tourism activity but the increasing success of tourism nowadays imperils nature conservancy, essentially because all users and managers of water resources are not conscious of their own impact on these resources. In order to provide comprehensive information to resources users, detailed investigations should be undertaken to analyse interactions which were thought to be insignificant at first sight but could prove more serious in the long run, as suggested by this telemetry study on the impact of small dams on migration patterns of trout.

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COLONISATION DYNAMICS OF A CATCHMENT AREA BY EEL. CHARACTERIZATION OF MIGRATING POPULATIONS IN A FREE ACCESS RIVER

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ABSTRACT

European eel is an amphihaline sea pawning fish. It colonizes continental waters, its growth area, by anadromous migration from the estuaries and progressive sedentarisation. Characteristics of the migrating population change in relation with the distance to the estuary because migrating eels are in a growth phase. Potential obstacles to the migration of fish in general, and the colonization of eels in particular, are more and more numerous in French rivers. To assess the effects of those obstacles, it was essential to describe the process of the colonization on a river which would not be affected by obstacles.

The Loire river, in its low part, is characterized by the lack of obstacles to the anadromous migration. The hydroelectric dam of " *Maison Rouge*" is located on the Vienne river, one of its main tributary ; it represents the first obstacle to migration at 200 km from the estuary. A trap was set immediately downstream the dam. The monitoring of catches was realized in 1994 and 1995 from April to November. The collected data allowed to describe the migration process of eels during the season. This paper presents seasonal tendencies of the migration and characteristics of migrating eels.

The migration season was long, from April to October. The migration intensity varied during the season and eels' sizes ranged between 80 and 540 mm. Several cohorts were observed in the migrating population. One year old migrating eels were observed, but they were scarce.

The heterogeneity of eel size distribution characterizes two behaviors in the migration: a diffusive migration which would correspond to a progressive colonization of the water basin; and a focused migration of eels directly towards the upper zones of the water basin. Obstacles to the migration could have quantitative but also qualitative effects on eel populations in a water basin.

Keywords : eel / upstream migration / fresh water

INTRODUCTION

Eel is an amphihaline sea spawner fish. It colonizes continental waters, its growth area, by anadromous migration from estuaries. The first phase of this migration has often been studied, either in estuaries and tidal areas, or in river areas located immediately upstream estuaries (table 1). Those studies mainly described the process and the effect of abiotic factors during early phases of the migration. Depending on sampling strategies, results referred to the whole population of eels or to part of it.

Table 1 : Various study sites for eel migration

Author	Distance from the estuary	Number of obstacles in the downstream part
Moriarty , 1986	15 km	0
Legault, 1994	17 km	3
White et Knights, 1994	0 à 36,2	0 à 5
Naismith et Knights, 1988	0,5 km à 15 km	0 à 1
Dahl, 1983	35 km	0

However, because eels can colonize the entire catchment area, it seemed essential to describe the anadromous migration on a bigger spatial scale. This study was conducted in the river Vienne, which joins the Loire basin 200 km upstream the estuary without any obstacle to fish migration. It provided comparative elements to assess the influence of the factors of degradation of migrating ways on eel distribution in continental waters.

METHODOLOGY

Study site

The dam of Maison Rouge is located on the Vienne river, a major tributary of the Loire river. Both rivers join upstream the city of Saumur, at 195 km from the Atlantic ocean, and 142 km upstream the city of Nantes which corresponds to the upper limit of tide. The dam is the first obstacle to fish migration. It is located 60 km upstream the junction of both rivers, 255 km from the ocean (fig.1) and 202 km upstream Nantes. The catchment area upstream the dam covers 19 600 km².

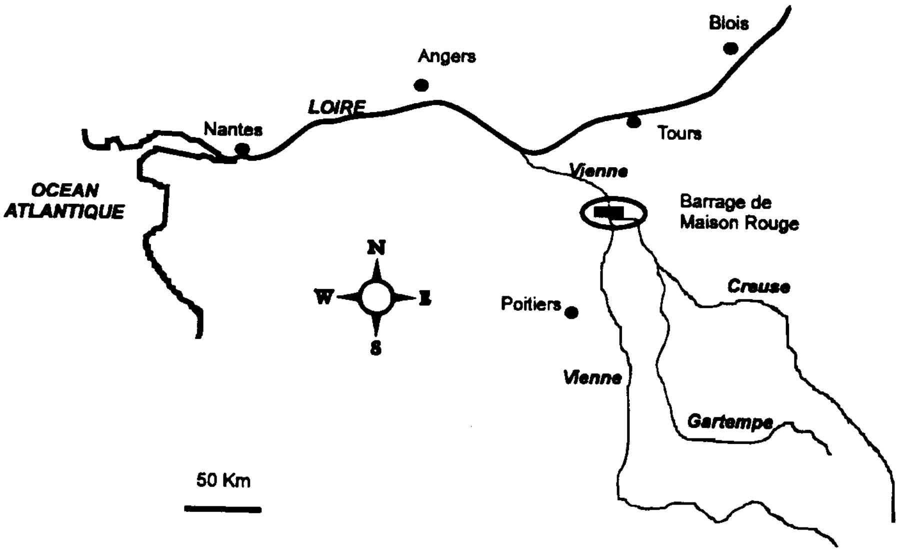

Figure 1 : The lower river Loire and location of Maison Rouge dam

The dam is composed of a weir crest and a power plant. The hydro production is supported by 3 Francis turbines ; the maximal water flow is 105 m3/sec. This obstacle to fish migration is equipped with 3 fish ways, among which only one is functional. The latter is composed of 2 successive pass with baffles separated by a pool. It is equipped with a trap in the upstream part and a lift-net in the medium pool.

Trap system

The trap was located on the left side of the river, above the turbine floor, beside the fish way. It was composed of 2 crawling ramps separated with a resting pool. At the top of the system, the trapping and irrigation system open into the stocking tank. The ramp substrate is of the "mixed type" defined by Legault (1992) in the Dordogne river.

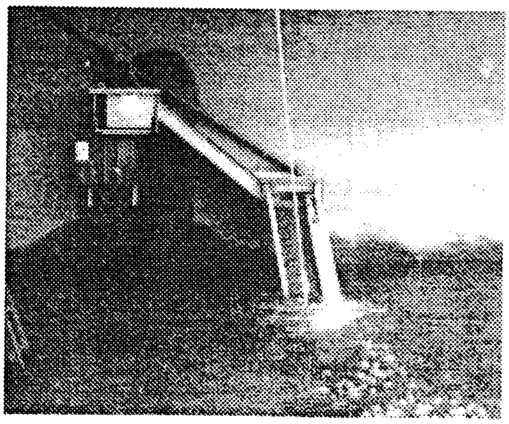

Figure 2: The trapping system

Technical characteristics of the trap allow it to be functional when river flow is comprised between 16 m^3/sec and 284 m^3/sec. However, the stocking tank is not accessible when the flow is over 199 m^3/sec.

Water was pumped up from the river and stocked in a water tower. A hose-pipe was installed by E.D.F. (French hydro company) to constantly supply the trap system with water.

Experimental design

All eels going through the trap were caught and stocked in a tank. Qualitative and quantitative data of migrating populations were collected each time the trap was hold up, then eels were released upstream the dam.

In 1994 and 1995, eels were collected between April and November. The number of trap holding by week was determined according to the migratory intensity (table 2).

-periods of low or medium migratory intensity : 2 to 3 collections a week
-periods of high migratory intensity : 5 to 6 collections a week.

Table 2 : Duration between 2 trap holding in 1994 and 1995.

Year	1994	1995
Duration of yields	Number of samples	Number of samples
0 day *	0	1
1 day	64	38
2 days	34	44
3 days	13	25
4 days	1	4
5 days	0	1
Total	112	113

Collected data

The trap was visited in the morning. The whole batch of eels caught was weighed, then individual biometrics data were collected on the whole sample, or on a subsample when more than 200 eels were caught.

Abiotic data were collected during the study but will not be presented in this paper. They will be used when the whole study be completed.

RESULTS

The study has been planned on 3 years, from 1994 to 1996. Only results of the two first years of monitoring of the trap (1994 and 1995) are presented here.

Weekly captures

Weekly catches were calculated from one or several trap sampling. When the trap was visited after several days of collecting, the mean daily catch was calculated as the ratio of total catches to the number of days of catching. This ratio was then allocated to each day of the catching period and weekly catches were determined using standard weeks as defined by Lewis and Taylor (figure 3).

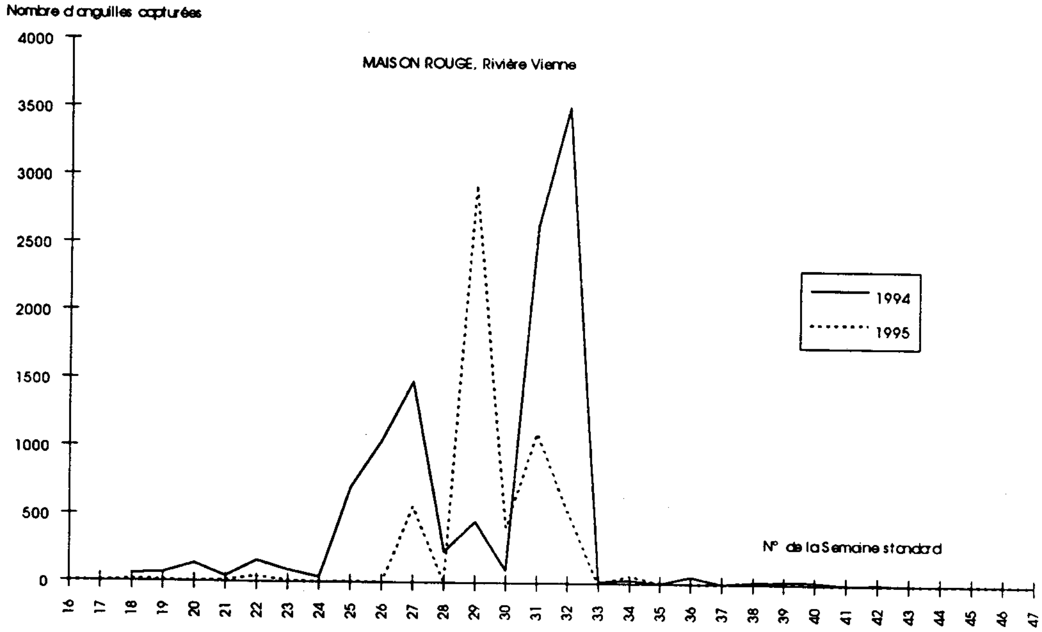

Figure 3. Weekly catches in 1994 and 1995.

The period of monitoring extended on 26 weeks in 1994, and 31 weeks in 1995. Total catches reached 10 921 eels in 1994 and 5 720 eels in 1995. Weekly catches varied from 0 to more than 3 500 eels a week. Catches were low until week 23 and most eels were caught between weeks 24 and 34 (table 3 and figure 3). After week 34, catches decreased to 1,2% of total catches in 1994, and 0,3% in 1995.

Table 3. Seasonal tendency of catches.

	% annual yields	
Period	1994	1995
Before June 11 week N° 24	5,1	1,7
Between 11/06 and 27/08	93,7	98,0
After August 28, week N°34	1,2	0,3

However, seasonal tendency differed slightly between both years. In 1994, two periods of high catches were separated by a period of low catches. In 1995, only one peak of catches was observed.

Eels length.

The size distribution of eels caught in 1994 and 1995 was determined from length measurements (figure 4).

The size range of eels was wide, from 80 to 540 mm. A first mode was observed in the small sizes, between 80 and 160 mm; it was sharp in 1994, smoother in 1995. A second mode was observed between 170 and 540 mm.

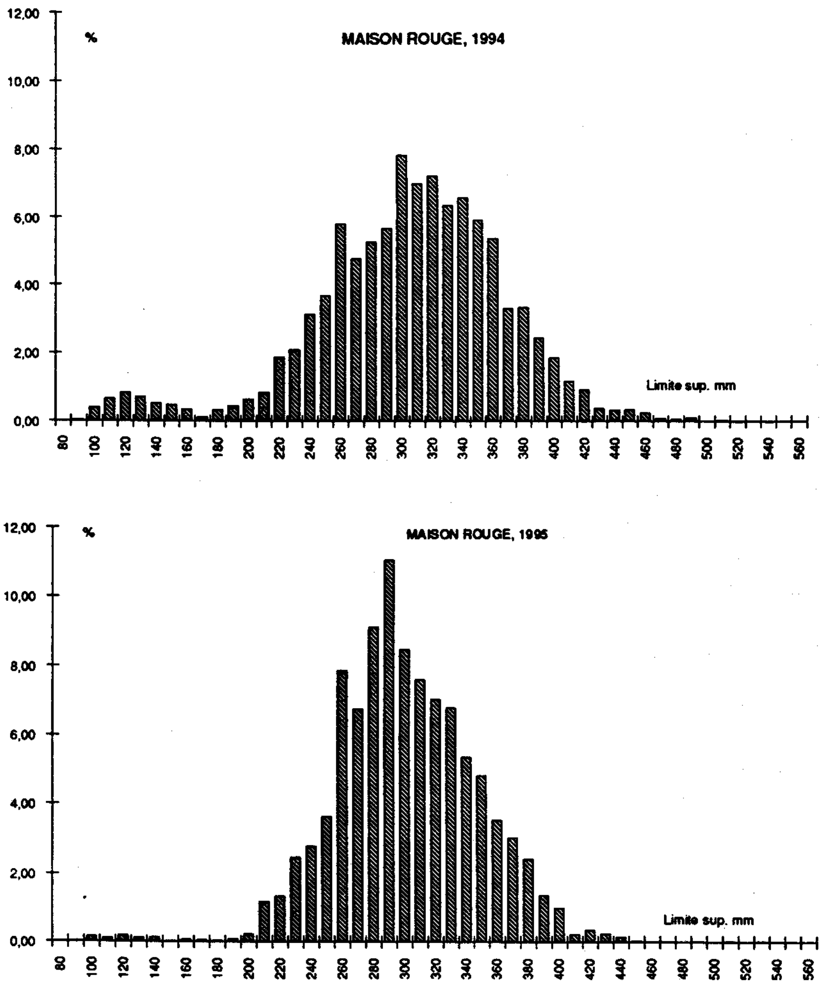

Figure 4. Size distribution of eels caught in 1994 and 1995.

DISCUSSION

Recruitment index

The trap system acted as a new fish way on the dam. The sampling strategy we used permitted to catch eels going upstream. As they crawled through the system, we supposed that those eels were in an anadromous migration process.

The other available fish ways seemed inefficient for eels. Crawling zones exist on the weir crest, but they have vertical parts which are not usable by eels longer than 100 mm. Moreover, they cannot be used by small eels when they are covered by an homogeneous water layer or when they are dry (Legault, 1988). One of the fish way on the dam is not suitable for eels : to pass upstream, eels have to jump a step of 40 cm. On the other fish way, current speed is too high. Sampling realized in the resting pool did not permit to catch more than 7 eels in 1994 (Andrieu, Bessy, Delbreihl, 1994). The trap corresponded to the preferred way to get over the dam. Thus, our results are representative of eels colonizing the upstream water basin.

The amount of catches we realized was low compared to the catchment area upstream the dam. The recruitment index was 0.56 eels/km^2 in 1994 and 0.29 eels/km^2 in 1995. Those indexes are lower than the one measured in the Arguenon river (561 eels/km^2: Legault, 1994) with the same catching system. They are also lower than recruitment indexes calculated in the Shannon river (Moriarty, 1986) or in the Gudena river in Denmark (Dahl, 1983). The indexes we calculated are also low compared to annual catches of glass-eels in the Loire estuary (several tenth of metric tons). Our data would then indicate a very low recruitment in the water basin of the Vienne river and more generally in upstream zones of the Loire basin. Because the area we studied was free of obstacle to migration, we think that this low recruitment could indicate the sedentarization of part of the population migrating from the estuary; it could also result of mortality. However, we do not have any assessment of the number of migrating eels reaching the dam, neither the initial assessment of the number of eels going through the estuary. It is then impossible to assess the sedentarization rate of eels downstream the dam.

Migration intensity during the season.

Our data give a first insight into the seasonal dynamic of eel anadromous migration. They indicate a long season of migration from April to October with variations of intensity during the season. Indeed, more than 90% of the annual migration was concentrated in less than 10 weeks although the migration season extends over 22 weeks. The tendency of the migration intensity indicates one main period for migration. The date of this period differed slightly for both years and occurred between the end of June and the middle of August. In the Vienne river, migration ended later than in the Thames river (Naismith and Knights, 1988) where the migration was achieved mid-July. The achievement of the monitoring study will allow to assess the effect of abiotic factors on anadromous migration and to characterize inter-annual variations of the peak of the migration. However, our results yet permit to define priority periods of efficiency for fish pass when working on the restoration of migrating ways.

Migrating eels

The age interpretation of eels we caught was not realized during this study. However, we used length to age keys calculated on eels of the river Vilaine (Mounaix, 1992) and eels of the river Rhine (Meunier, 1994).

Table 4 : Length to age relationships for eels.

Continental age (year)	1	2	3	4	5	+
Mounaix, 1992	140	235	296	306	335	451
Meunier, 1994	168	195	281	307	346	

The polymodal decomposition of eel size distribution was realized using length to age relationship described in the Vilaine river. They have been corrected to integrate the shortest growth period (table 5). Normsep software was used to characterize cohorts, to calculate their mean size and to determine their proportion to total catches (table 6).

Table 5: Parameters of polymodal decomposition (Normsep method).

troncature point	160	250	290	310	340	510
lower limit of the mean.	110	170	240	280	320	340
upper limit of the mean	130	230	270	305	335	440

Table 6 : Results of the polymodal decomposition.

	1994			1995		
Age	Lengths	Number	Frequency	Lengths	Number	Frequency
0+	122.2	424	3.9	118.9	45	0.8
1+	217.3	631	5.8	215.0	121	2.1
2+	258.2	2342	21.6	259.5	1337	23.3
3+	298.6	1208	11.1	282.2	1735	30.2
4+	333.0	2730	25.1	324.9	1183	20.6
5+ et +	340	3537	32.5	340	1318	23.0

Calculated lengths to age were lower than the ones observed in the Vilaine water basin at the end of the growth period. Ninety percent of our catches were realized between June and August. Our data would then correspond to a shorter period of growth than the study in the river Vilaine; this would explain the difference in length to age. However, our results remain comparable with those observed in the river Vilaine and in the river Rhine. Age composition of catches in the Vienne river differed from those calculated in the rivers Severn and Avon (White and Knights, 1994) and in the river Gudena (Dahl, 1983). However, those authors used a different method to assess the age; also growth parameters could be affected by the northern location of those rivers.

The age composition indicates the heterogeneity of migrating populations of eels. More than 5 age groups were observed, from eels in their first year of continental life to eels living in freshwater for several years. The O+ group was the scarcest, from 0,8 to 4% of annual catches. Proportions of older age groups varied with the year. In 1994, the 5+ group was prominent and 4+ group was abundant. In 1995, their proportion diminished as groups 2 and 3 increased in number. This could be related to the opening of the trap in 1994. Big eels gathered at the bottom of the dam could have gone through the first year the trap and so their proportion have decreased in 1995. This would conclude on the efficiency of the trap as a fish way. The continuity of the study in 1996 will allow to check this efficiency, as well as to determine mean ages of eels arriving on a river site when no obstacle affect the migration.

Our observations illustrate the specificity of anadromous migration of eels. To the opposite of other fish migration, eel migration is a several year process, and migrating eels are in a growth phase. The variety of age groups indicate the heterogeneity of migration speed. As some eels cross the 200 km distance in a few months, other ones reach the dam after several years. Then, migration speed would vary from more than 200 to 50 km per season. The heterogeneity of migration speed was also observed on the river Severn and the river Avon where it varied from a few meters to more than 2 km a day (White and Knights, 1994). Two different migration behaviors can be described. On one hand, a slow diffusion migration which would lead to the colonization of the entire water basin ; on the other hand, an focused migration where eels rapidly reach upstream zones. However, the ability of eels to get through obstacles varies with fish size (Legault, 1988). Only small eels can get through vertical obstacles. So, the capacity of eels to get through obstacles will vary according to their size and to their progression speed. Obstacles to migration could then affect not only the number of migrating eels, but also their progression speed and the migrating behavior of cohorts.

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THE LA GRANDE HYDROELECTRIC PROJECT AND EELGRASS MONITORING IN COASTAL JAMES BAY

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ABSTRACT

The habitat monitoring program along the northeastern coast of James Bay was implemented more than 10 years ago in order to validate the projected impacts described in the La Grande-2A and La Grande 1 hydro project environmental impact assessment. The modified hydrograph of the La Grande River, resulting from the commissionning of the two generating stations, was expected to cause the impacts. Before the monitoring progam could be drawn, it was essential to study in some detail the various components of these coastal ecosystems. This characterization led to the selection of a single component and of a sampling technique adequate for an efficient monitoring after the progressive commissionning of the hydroelectric system.

Coastal eelgrass beds are the only environmental component retained for long-term monitoring. These beds constitute an important habitat for a variety of organisms, most of which are mobile and may be influenced by factors other than the changes created by the hydroelectric projects. Since eelgrass beds are fixed and are submitted to salinity changes in the zone of influence of the projects, they were the only component selected.

The dry leaf biomass and the shoot density in eelgrass beds were monitored at six permanent sampling stations; their annual variations were assessed, and increasing or decreasing trends were highlighted. Climate conditions and water level have a direct influence on the annual production of eelgrass. In the longer term, the isostatic land-rise has a dominant influence on the distribution of eel-grass beds; its impact is modulated by the physical characteristics of the sampling stations. Aerial photographs (scale 1:10 000) taken in 1986 and 1995 help compare the distribution of eelgrass before and after the commissionning of the power stations.

With the results obtained thus far, it is now conceivable to develop a predictive model of the evolution of eelgrass beds which takes into consideration the isostatic land-rise rate, eelgrass densities and biomass and physical parameters such as water level, foreshore slope and climate.

KEY-WORDS: Monitoring / Coastal habitats / Discharges / Eelgrass beds / Salinity / Impacts / Hydroelectricity / Isostatic land-rise / Biomass / Shoot density

INTRODUCTION

The eastern James Bay coast harbours large marine eelgrass beds (*Zostera marina* L.) which are ecologically important to migratory birds. Prior to the regulation of the La Grande River., little was known on the distribution and ecology of these habitats. In fact, only a preliminary mapping of eelgrass beds along eastern James Bay had been conducted by the Canadian Wildlife Service in 1974-1975 (Curtis, 1975).

The operation of the La Grande phase 1 complex, the overequipment at the La Grande 2A station and the commissionning of the La Grande 1 station have an influence on the La Grande River hydrograph and, as a consequence, on the winter freshwater plume along eastern James Bay. As mentionned in the impact assessment report of the La Grande 2A and La Grande 1 hydro projects (SEBJ and HQ, 1985; 1986; 1987), lower salinities in coastal waters could have negative impacts on the distribution and abundance of marine eelgrass, a strict halophyte species.

For this reason, the James Bay Energy Corporation (SEBJ) initiated in 1982 a research program on the eelgrass beds, which later became the basis for the on-going eelgrass monitoring program (Lalumière *et al.*, 1994; Lalumière and Lemieux, 1995).

This paper reviews the monitoring program over the years and highlights the major trends observed to this day.

ENVIRONMENTAL MONITORING: BACKGROUND INFORMATION

The design of the research program responds to the basic objectives of the environmental monitoring which are:

- to follow the actual evolution of eelgrass beds in relation with predicted changes;
- to detect rapidly the unpredicted impacts that may occur;
- and to improve the impact prediction methods and the design of future environmental monitoring programs.

During the last decade, the La Grande River hydrograph was the object of rather important modifications (Figure 1). The main feature of the new hydrograph is a gradual increase of winter discharges, as a result of increased power production in winter. The main consequence of these new flow characteristics is mostly felt on the extent of the La Grande River winter plume (SEBJ and HQ, 1985; 1986). Research has shown that, under ice cover, the freshwater plume extends much further than in open water (Messier *et al.*, 1989; CSSA, 1987). Several factors may influence the mixing of freshwater and saltwater: the extent of the ice cover, tide cycles, river discharges, ocean floor topography, certain exceptional meteorological events, and coastal currents (Messier and Anctil, 1996). The result is a great variability of winter salinities in coastal habitats. In summer, the La Grande River discharges are similar to natural discharges and the extent of the plume is limited by the intense mixing caused by ocean swell and tidal currents (SEBJ, 1990).

Figure 2 illustrates the projected modifications of the 5‰ surface isohaline contour along the James Bay east coast. In order to understand the modifications caused by an extended winter plume, SEBJ initiated a coastal salinity monitoring program (CSSA Consultants ltd, 1987; 1989; 1993; 1995).

In theory, all coastal habitats and resources located inside the zones of plume changes could be more or less influenced by a decrease of winter salinity below 5‰ (SEBJ and HQ, 1986). The selection of a component to be monitored had to take into account the sheer size of the study area, the logistics constraints associated with research in remote areas, and the restrained circulation along the coast during the goose hunting season.

Two criterias led to the selection: the component had to be linked directly to the projected impacts described in the impact assessment study, and the natural variability of the biological component had to be considered. We had to be able to separate the mid and long term changes caused by natural variations from the changes induced by the hydroelectric project.

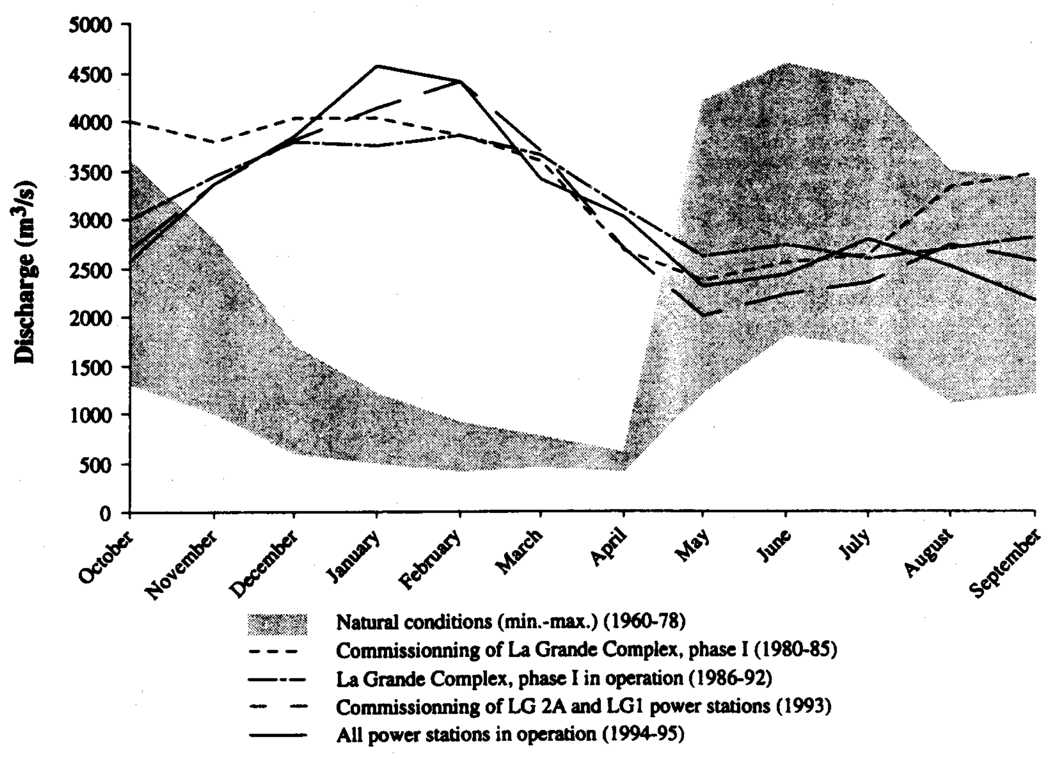


Figure 1. Mean monthly discharges of the La Grande River (1960-1995).

Marine eelgrass beds answered the requirements best and were selected for the environmental monitoring program. They constitute essential habitats for migrating waterfowl (Reed *et al., 1996);* they also host a large variety of aquatic organisms. While several organisms are highly mobile and thus are influenced in other areas by factors other than those related to the hydroelectric projects, eelgrass beds are fixed and are directly affected by salinity changes induced by the project. It must be noted that marine eelgrass is a strict euryhaline halophyte species which cannot tolerate freshwater in permanence.

In a first phase, the eelgrass beds were the object of a bioecological characterization (Lalumière, 1988) to assess their ecological value and to better understand the eventual impacts of changes caused by the operation of generating stations.

Eelgrass monitoring addresses two separate aspects: eelgrass production at permanent sampling stations, and eelgrass distribution along the northeast coast of James Bay. Indeed, large-scale changes could occur along the coast without being detected at permanent sampling stations.

METHODS

Permanent stations

Six permanent stations (Figure 2) were established according to projected changes of the La Grande River winter plume. Stations Attikuan I and Attikuan II, located north of the La Grande River, and Station Dead Duck, south of the river, are outside of the zone of influence of the new winter plume and are used as reference stations. The stations at Kakassituq, Bay of Many Islands and Tees Bay are all within the zone of

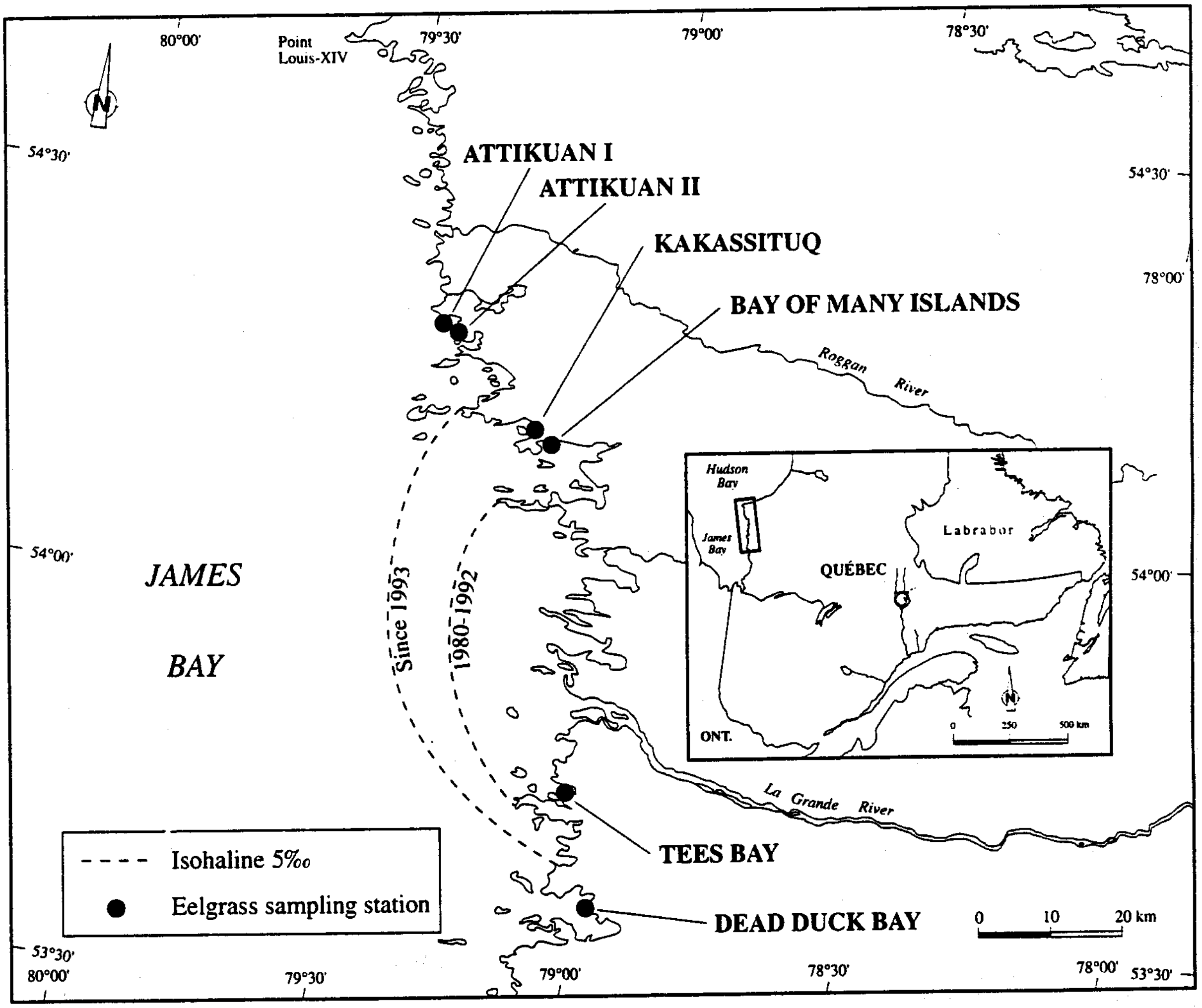


Figure 2. Predicted 5‰ surface isohaline contour after the La Grande Complex commissionning and location of eelgrass permanent sampling stations.

The three most commonly used parameters to measure eelgrass production are the shoot density and dry biomass, and the rhizome biomass (Thayer *et al.* , 1984; Kentula and McIntire, 1986; Jacobs, 1979; and McRoy, 1970). Shoot density and biomass are easy to measure, but rhizome biomass is not; indeed it is hard to separate clearly dead rhizomes from live ones, which may introduce a manipulation bias from year to year. Since sexual reproduction intensifies as a result of environmental stress (Philipps, 1980; Philipps *et al.* 1983), an increase in the density of reproduction shoots may reflect changes in environmental conditions.

Therefore, the parameters selected to monitor eelgrass production at six permanent stations are: shoot density and dry biomass, and the proportions of vegetative and reproductive shoots.

Sampling methodology has evolved between 1982 and 1989. From 1982 to 1986, techniques used in similar studies were followed (Grontved, 1957; McRoy, 1970): eelgrass was collected with a Grontved sampler operated from the surface along transects parallel to shore (0.5 and 1.5 m deep). Handling the sampler was an uneasy task and sorting the samples proved rather difficult; experimental sampling by divers rapidly demonstrated greater efficiency and reliability (Lalumière, 1986). This technique has been used systematically since 1988.

Also, instead of sampling eelgrass along transects parallel to shore at 0.5 and 1.0 m depth, sampling has been conducted since 1988 along five equidistant transects perpendicular to shore and at three distinct depths (0.5, 1.0 and 1.5 m and 2.0 m at Attikuan I). Each sampling point is located by surveying equipment from fixed landmarks. The statistic validity of the sampling design was verified by Scherrer (1990).

Finally, to account for the influence of climate on the growth of eelgrass, a weather station is in operation since 1992 near the mouth of the La Grande River, recording water temperature, wind speed and direction, and solar radiation.

Eelgrass distribution

There are various tools available to map the distribution of eelgrass beds. A classical approach is to use aerial photographs combined with ground-truthing (Jacob 1979; Nenhuis, 1983; Orth and Moore, 1983; Orth *et al.*, 1986). Satellite imaging (Landsat, Spot), airborne sensing (e.g. MEIS), and echosounding (Spratt, 1989) are more and more often used to map vegetation cover on a small scale (Ackleson and Klemas, 1987); but these techniques have rarely been used to map eelgrass distribution. They also require ground-truthing.

Eelgrass distribution monitoring is based on mapping from aerial photo-interpretation of about 2000 colour photographs, scale 1: 10 000, taken in 1986 and covering approximately 150 km of coastline. Eelgrass bed boundaries were determined and the beds were grouped under two classes of shoot density: high density (> 50% cover) and low density (< 50% cover). Helicopter surveys were combined with validation dives to fix the limits more accurately.

The resulting map, scale 1:125 000, shows that eelgrass distribution is not continuous along the coast. Beds are not found at the mouth of tributaries where substrate is unstable and salinity is low (Skinner, 1974).

In 1996, a new eelgrass distribution map will be produced from a new set of aerial photographs, scale 1:10 000, taken in 1995. Comparison between the 1987 and 1996 maps will reveal the evolution of eelgrass distribution along the whole northeast coast of James Bay over the last decade.

RESULTS AND DISCUSSION

Monitoring at six permanent stations

Eelgrass production (shoot density and dry biomass per m^2) is not the same at all stations, nor at all depths (Tables 1 and 2). Monitoring at the six permanent stations indicate that the annual variability observed in dry biomass production may be high at a given station (Table 1). In addition, the annual variations recorded do not necessarily follow the same pattern from one station to the other, nor even at the three depths at a given station.

Shoot density also shows a high annual variation at a given station and between stations (Table 2). The variation decreases with depth which reflects a greater environmental stability in deeper waters. At a depth of 0.5 m, shoots are usually more abundant but shorter.

In the early years of monitoring, it was difficult to outline any increasing or decreasing trends in eelgrass production and even more difficult to pinpoint the environmental causes of these high variation. In addition, there were no obvious differences between reference stations and stations located within the reach of the winter plume.

In parallel to the eelgrass monitoring program, SEBJ was also monitoring the La Grande River winter plume (CSSA Consultants ltd., 1987; 1989; 1993; 1995; Messier and Anctil, 1996) on both sides of the river mouth. So far, the results of this program cannot be linked to any biological data.

However, field observations strongly suggested the occurrence of good and bad eelgrass production years. Obviously, the climate conditions and the annual fluctuation of water level in coastal bays, during the growth season, seem to have a determining influence on eelgrass growth. Result interpretation should also take into consideration the relatively fast isostatic land-rise (= 1cm/year) observed along the coast (Tushingham 1992).

Table 1. Variations in mean dry leaf biomass ($gDWm^{-2}$) at the six sampling stations.

Depth (m)	1985 $\bar{x}$	1985 CI	1986 $\bar{x}$	1986 CI	1987 $\bar{x}$	1987 CI	1988 $\bar{x}$	1988 CI	1989 $\bar{x}$	1989 CI	1990 $\bar{x}$	1990 CI	1991 $\bar{x}$	1991 CI	1993 $\bar{x}$	1993 CI	1994 $\bar{x}$	1994 CI	1995 $\bar{x}$	1995 CI
Attikuan I																				
0,5	-	-	216	±126	-	-	116	±46	109	±67	-	-	74	±47	87	±52	68	±48	60	±47
1,0	-	-	-	-	-	-	234	±46	218	±69	-	-	198	±57	187	±37	120	±27	134	±40
1,5	-	-	104	±62	-	-	180	±122	140	±45	-	-	107	±50	277	±175	106	±25	188	±95
2,0	-	-	-	-	-	-	83	±67	93	±19	-	-	-	-	190	±26	130	±15	148	±43
Attikuan II																				
0,5	-	-	-	-	-	-	-	-	189	±39	-	-	139	±37	-	-	178	±26	137	±26
1,0	-	-	-	-	-	-	-	-	241	±69	-	-	208	±48	-	-	192	±42	252	±60
1,5	-	-	-	-	-	-	-	-	164	±54	-	-	212	±39	-	-	111	±36	165	±83
Kakassituq																				
0,5	254	±87	344	±117	-	-	194	±74	263	±30	234	±63	189	±85	382	±82	354	±74	456	±44
1,0	-	-	-	-	-	-	472	±192	358	±51	272	±96	294	±79	415	±54	295	±61	559	±132
1,5	244	±134	229	±123	229	±77	399	±109	387	±42	317	±28	392	±109	361	±72	298	±58	548	±548
Baie of Many Islands (2A)																				
0,5	-	-	-	-	-	-	-	-	22	±14	-	-	202	±60	-	-	36	±44	37	±55
1,0	-	-	-	-	-	-	-	-	63	±30	-	-	175	±67	-	-	54	±23	198	±83
1,5	-	-	-	-	-	-	-	-	119	±65	-	-	124	±37	-	-	64	±42	211	±88
Baie Tees																				
0,5	-	-	-	-	-	-	73	±46	82	±33	-	-	31	±13	-	-	57	±37	37	±23
1,0	-	-	-	-	-	-	38	±21	48	±15	-	-	15	±10	-	-	8	±6	9	±9
1,5	-	-	114	±31	-	-	34	±5	53	±21	-	-	18	±6	-	-	18	±11	31	±27
Dead Duck																				
0,5	-	-	-	-	-	-	-	-	154	±37	-	-	111	±10	-	-	5	±8	0	±0
1,0	-	-	-	-	-	-	-	-	140	±30	-	-	172	±55	-	-	151	±59	175	±31
1,5	-	-	-	-	-	-	-	-	135	±28	-	-	141	±20	-	-	152	±45	177	±23

CI : Confidence interval of the mean ($P < 0,05$)

TABLE 2. Variations in mean shoot density (shoot m^{-2}) at the six sampling stations.

Depth (m)	1985 $\bar{x}$	1985 CI	1986 $\bar{x}$	1986 CI	1987 $\bar{x}$	1987 CI	1988 $\bar{x}$	1988 CI	1989 $\bar{x}$	1989 CI	1990 $\bar{x}$	1990 CI	1991 $\bar{x}$	1991 CI	1993 $\bar{x}$	1993 CI	1994 $\bar{x}$	1994 CI	1995 $\bar{x}$	1995 CI
									Attikuan I											
0,5	-	-	**1475**	±626	-	-	**1123**	±526	**923**	±204	-	-	**403**	±312	**1053**	±330	**659**	±304	**536**	±243
1,0	-	-	-	-	-	-	**583**	±235	**694**	±185	-	-	**583**	±131	**670**	±213	**445**	±123	**419**	±141
1,5	-	-	**345**	±166	-	-	**329**	±142	**457**	±123	-	-	**378**	±177	**508**	±104	**375**	±74	**369**	±137
2,0	-	-	-	-	-	-	**227**	±107	**284**	±51	-	-	-	-	**486**	±58	**399**	±60	**248**	±58
									Attikuan II											
0,5	-	-	-	-	-	-	-	-	**1761**	±206	-	-	**1156**	±254	-	-	**1373**	±247	**1582**	±281
1,0	-	-	-	-	-	-	-	-	**614**	±99	-	-	**513**	±120	-	-	**554**	±95	**702**	±108
1,5	-	-	-	-	-	-	-	-	**421**	±77	-	-	**451**	±70	-	-	**301**	±76	**390**	±158
									Kakassituq											
0,5	**1465**	±174	**715**	±326	-	-	**982**	±309	**1640**	±246	**836**	±195	**931**	±406	**1737**	±411	**1039**	±166	**1751**	±307
1,0	-	-	-	-	-	-	**689**	±206	**880**	±55	**578**	±123	**699**	±154	**887**	±139	**665**	±140	**1108**	±255
1,5	**420**	±257	**290**	±264	**350**	±108	**512**	±174	**834**	±149	**516**	±66	**757**	±190	**787**	±150	**617**	±185	**948**	±235
									Baie of Many Islands (2A)											
0,5	-	-	-	-	-	-	-	-	**157**	±53	-	-	**552**	±124	-	-	**165**	±174	**100**	±138
1,0	-	-	-	-	-	-	-	-	**320**	±103	-	-	**484**	±108	-	-	**233**	±85	**545**	±252
1,5	-	-	-	-	-	-	-	-	**548**	±171	-	-	**455**	±99	-	-	**187**	±110	**490**	±189
									Baie Tees											
0,5	-	-	-	-	-	-	**339**	±251	**376**	±144	-	-	**228**	±107	-	-	**295**	±158	**274**	±194
1,0	-	-	-	-	-	-	**129**	±52	**205**	±37	-	-	**62**	±30	-	-	**45**	±41	**41**	±42
1,5	-	-	**316**	±58	-	-	**88**	±27	**206**	±51	-	-	**89**	±30	-	-	**78**	±52	**106**	±88
									Dead Duck											
0,5	-	-	-	-	-	-	-	-	**931**	±178	-	-	**936**	±103	-	-	**199**	±349	**0**	±0
1,0	-	-	-	-	-	-	-	-	**405**	±132	-	-	**463**	±220	-	-	**426**	±241	**547**	±246
1,5	-	-	-	-	-	-	-	-	**290**	±72	-	-	**226**	±43	-	-	**247**	±62	**326**	±48

CI : Confidence interval of the mean (P < 0,05)
$\bar{x}$: mean
- : no data available

Considering the weather data available, results show that there is a linear relationship between the number of degrees-days of growth and the shoot density. At the Kakassituq station, the number of degrees-days of growth accounts for 45% of annual variations in shoot density at a 1.5 m depth. In shallower waters, this relationship is also statistically significant, but R^2 decreases, indicating that other factors are also acting strongly (possibly water level, foreshore dynamics, ice action or isostatic land-rise).

After 10 years of monitoring, a simple linear regression highlighted increasing or decreasing trends in the mean dry biomass produced at each of the six stations (Figure 3). At depths of 0.5 m and 1.0 m, decreasing trends are recorded at all stations with the exception of Kakassituq but not all trends are statistically significant. At depths of 1.5 m and to 2.0 m at Attikuan I, increasing trends are observed at all stations with the exception of Tees Bay. In general, production has decreased in shallower waters and has increased at deeper locations. Reproductive shoot density at all depths has remained quite stable throughout the monitoring program.

Data analysis from the tide gauge at Churchill, Manitoba (MDES, 1992), along Hudson Bay, reveals high water level variations during the growth season, and between various growth seasons (Figure 4). From one year to the next, water depth during the growth season varies and probably influences greatly eelgrass production, particularly leaf length.

The results of the monitoring program strongly suggest that the annual variations of eelgrass production are likely induced, in large part, by the combined action of climate conditions and water level variations during the growth season. Isostatic land-rise would account for longer term increasing or decreasing trends. The foreshore slope at sampling stations would modulate the response of eelgrass production to isostatic land-rise. The gentler the slope, the more intense the effect. For example, the slope at station Attikuan I is gentler than at station Kakassituq and eelgrass production has decreased more markedly in shallow water at that station.

Assuming that isostatic land-rise has a slow but continuous influence, it is likely that eelgras will gradually disappear from shallow waters and that eelgrass beds will slowly move offshore.

A small scale map monitoring becomes an interesting tool to follow this effect.

<u>Small scale mapping</u>

Small scale mapping done in 1986-1987 was compared with similar mapping by Curtis (1976). A difference in methodology between the two studies limits the comparison; but it still reveals that the concentration zones of eelgrass beds have pratically remained the same over that period. Some changes in boundaries and extents of eelgrass beds are evident; but, it remains difficult to assess the actual causes (natural, man-induced, methodological bias).

However, a comparison of aerial colour photographs, scale 1:10 000 taken in 1995 and 1986, using the same methodology, reveals marked changes in the distribution of eelgrass in some coastal bays with gentle slopes. These beds are located either inside or outside of the zone of influence of the La Grande River plume and reflect a process affecting the whole coastline.

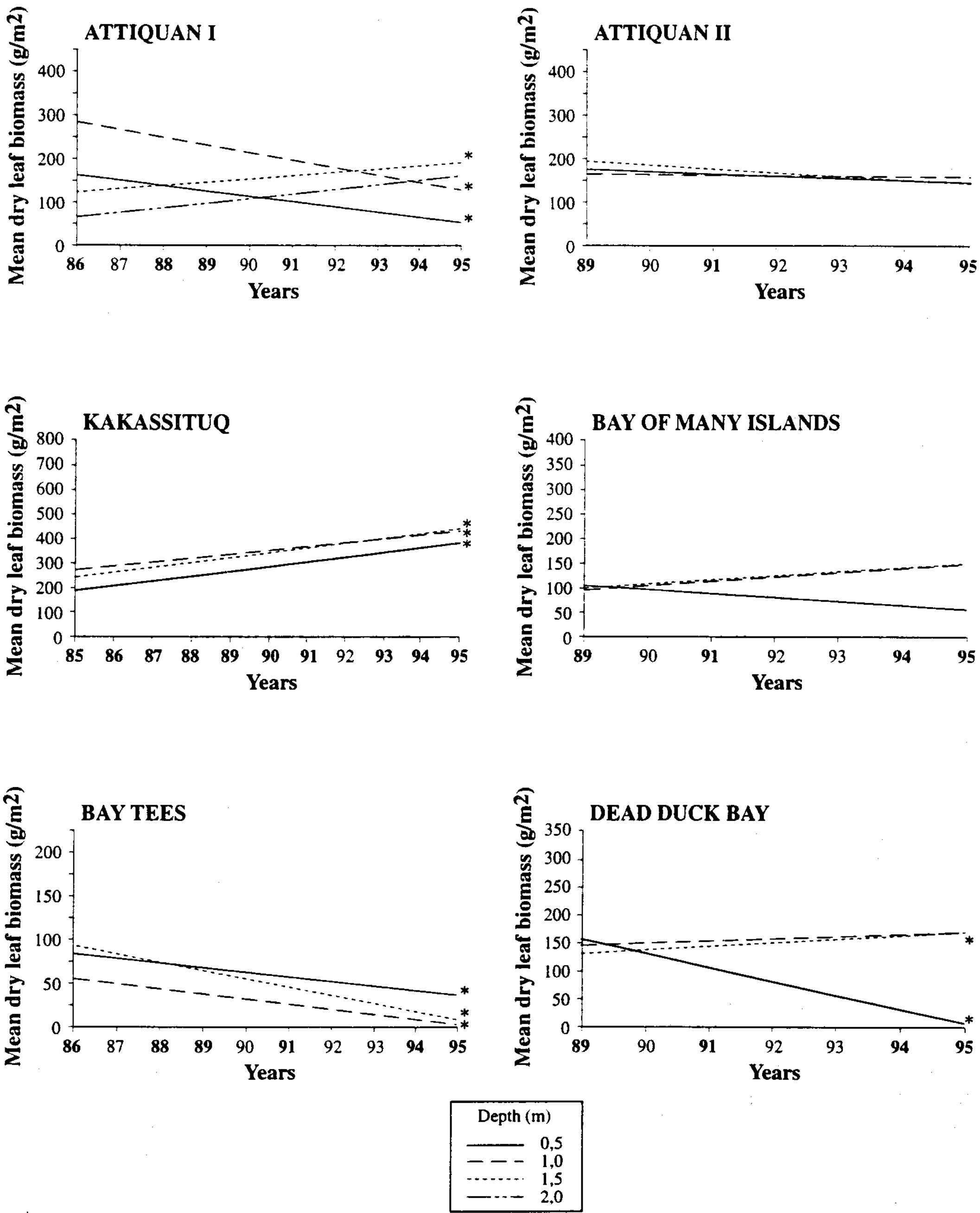
<u>Lessons drawn from the monitoring program:</u>

After 10 years of eelgrass monitoring, the main conclusions are :

1. Since there were no references to orient the monitoring of eelgrass, it was necessary to design and develop a methodology appropriate for both types of surveys (permanent stations, and overall study area), in parallel to the actual monitoring;

2. Permanent stations were selected on the basis of projected changes of the river plume. Looking back, a different selection could have been made. For example, with the commissionning of additional power stations, the Dead Duck station has become included inside the La Grande River freshwater plume.

3. We must remain cautious when interpreting annual data, especially for a short time series. Certain influence factors, such as isostatic land-rise, have only a mid to long-term impact and several years of data are needed to detect any trends;



* Significant at 0,05 level
Note: Sampling years are in bold.

Figure 3. Evolution of mean dry leaf biomass at the six sampling station.

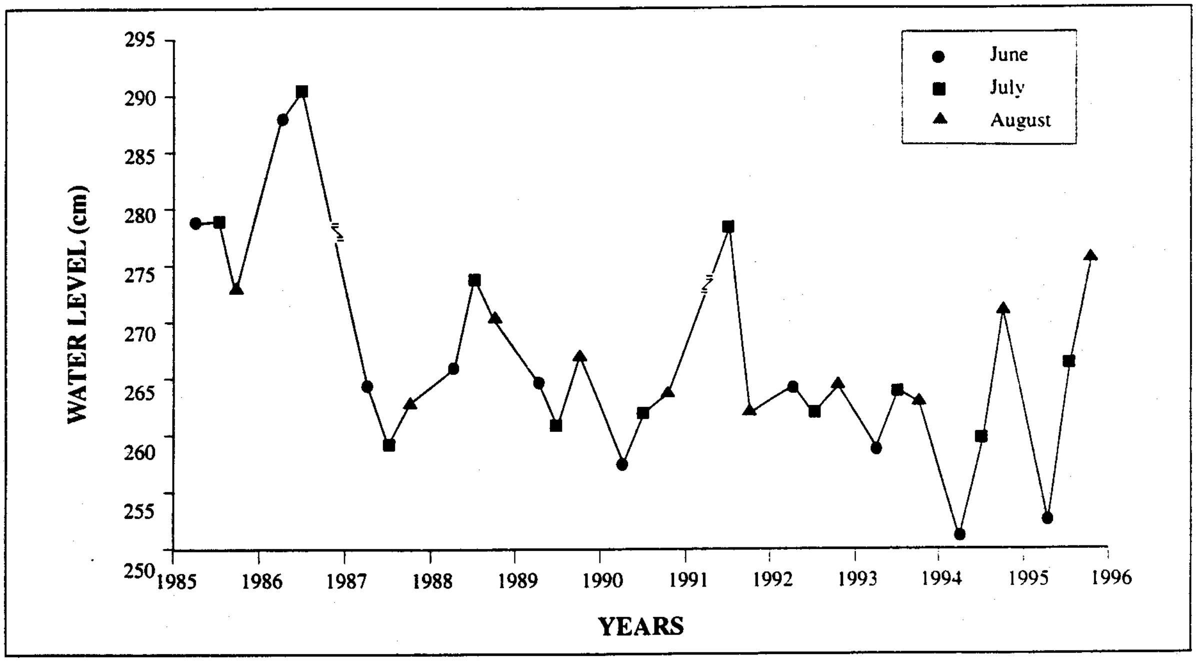


Figure 4. Mean water level during the growth season (June-July-August) at Churchill, Manitoba (1985-1995) (Churchill tide gauge data from MDES,1992).

4. A large-scale monitoring of eelgrass beds (permanent stations) combined with a small scale monitoring (mapping) has produced useful complementary data;

5. Decrease in winter salinity has little, if any, negative impacts on eelgrass for the following reasons:

- during the growth season, coastal water salinities are comparable to natural conditions;
- eelgrass has always survived in Tees Bay, a station located immediately to the south of the mouth of the La Grande River, certainly influenced by the river plume;
- at deeper locations, eelgrass production is still comparable, if not better, to the early years of monitoring.

Isostatic land-rise has a dominant influence on the growth of eelgrass in shallow water and its effects would largely mask the potential impacts of lower salinities.

Only a long term follow-up will confirm these hypotheses and will verify to which extent a decrease of winter salinity will be felt only in the long term by eelgrass.

6. Finally, the results for shoot biomass and density suggest the possibility of developing a predictive model for the evolution of eelgrass beds taking into account the landrise rate, the climate, the sea level and the foreshore slope.

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Coastal - estuarian - fluvial interactions

Interactions fluviales - estuariennes - côtières

MORPHOSEDIMENTOLOGICAL EVOLUTION OF THE SHORELINE IN POINTE-TAILLON CONSERVATION PARK. LAC SAINT-JEAN, QUEBEC

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ABSTRACT

The present study deals with the morphosedimentological evolution of lake shores in Pointe-Taillon Conservation Park, which is located in the Northwestern part of the Lac Saint-Jean, some 250 km North of Québec City. The present study is a part of a study on shore erosion requested in 1991 to the Consortium formed by Le Groupe LMB Experts-Conseils 1992 Inc. and Le Groupe-Conseil LaSalle Inc. (1992) by the Ministère de l'Environnement et de la Faune du Québec. This study was requested in order to help the Ministry to ensure an optimal management of the Park. As the shores within the limits of the Park are active, the Ministry wanted to know if the existing infrastructures and the ones planned could integrate in the long term together with the local natural heritage. Following a review of litterature, an analysis and interpretation were carried out on the basis of aerial photographies taken from 1959 to 1991. A division of the 16 km long shoreline into 7 homogeneous zones was undertaken considering morphosedimentology and wind exposition features.

Pointe-Taillon is a delta constructed by the Péribonka River into the Laflamme Sea, which was replaced by the Lac Saint-Jean after the last glaciation period (10 000 years B.P.). Lac Saint-Jean became a reservoir after a generating station called Isle Maligne was built up in 1927 at its outlet by Alcan. On the Péribonka River, three generating stations of a total capacity of 1 117MW were constructed between 1952 and 1960. The high water levels contributed to erosion, so that 95% of the total length of the Pointe-Taillon lake shoreline is active. Shrubs and trees stratas were removed by waves activity and recent cycling track and a peat bogs are close to the edge of the terrace. The photos from 1959 up to 1985 show little erosion along the shoreline. But from 1985 to 1991, the total volume of sand eroded from the shores was 200 x $10^3 m^3$, which means an average annual shore retreat of 0,55m. It could be related to the exceptional mid-november 1989 storm, when the water level was at 101.5 m, nearly the maximum level of operation during summer and fall, which coïncides with the foot of the shores.The prevailing Westerly winds produced high energy waves that undermined the bottom of cliffs. Since then, slope equilibrium has not been reached and erosion is still going on. These winds have become also the main transforming factor of the beach during the free ice period., since the sediment load input from the Péribonka River has decreased.

Soft remedial measures using faggots, shrubs and herbaceous plants are proposed. Betwen littoral and pre-littoral zones, setting booms attached with cables to concrete blocks could break high energy waves. These measures would contribute to shore stabilisation and vegetation and habitat protection. In order to slow down shore drift, the construction of a groin is also suggested. A monitoring program should be implemented in view of following evolution of shoreline and to evaluatuate the efficiency of remedial mesures set up.

KEY-WORDS: Morphosedimentology, photointerpretation, evolution, shore, beach, littoral, erosion, littoral drift, waves, remedial measures, habitats, vegetation, monitoring.

CONTEXT

The present study deals with the morphosedimentological evolution of lake shores in Pointe-Taillon Conservation Park, which is located in the Northwestern part of the Lac St-Jean, some 250 km North of Québec City (Figure 1) (Denis, 1992). This lake occupies a huge depression of more than 1,000 km^3. Its outlet is the Saguenay River which flows into the St-Lawrence estuary, some 180 km to the South-East of the Lac Saint-Jean. The present study is a part of a study on shore erosion requested in 1991-1992 to the Consortium formed by Le Groupe LMB Experts-Conseils 1992 Inc. and Le Groupe-Conseil LaSalle Inc.(1992) by the former Ministère du Loisir, de la Chasse et de la Pêche du Québec[1]. This study was requested in order to help the Ministry to ensure an optimal management of the Park. As the shores within the limits of the park are active, the Ministry wanted to know if the existing infrastructures and the ones planned could integrate in the long term together with the local natural heritage.

METHODS

Following a review of litterature relevant to the region and to the object of study, an analysis and interpretation were carried out on the basis of aerial photographies taken from 1959 to 1991, the scale of which varied from 1:13 000 to 1:40 000. A preliminary division of the 16 km long littoral zone was undertaken taking into account morphology and wind exposition features. Distances were subsequently calculated three times, in order to obtain a mean value, from points located at the edge of the littoral terrace (upper part of the shoreline-Figure 2) to fixed points such as crossroads found on aerial photographs. This work was carried out with the assistance of a Hilger & Watts lens (x 5) including a micrometric scale (0.1mm). Errors were reduced by using reference maps. By comparing distances measured on aerial photos of different years, taking into account their scales, evolution of the shores was evaluated. On aerial photos, accumulation features were also observed both in the littoral and pre-littoral zones.

At the date at which the aerial photos were taken, the water levels of Lac Saint-Jean, were also obtained from Alcan and were considered throughout the study. Wind directions, speeds and frequency were those recorded by Environment Canada at the Roberval meteorological station (Figure 1).

Field observations were carried out between August 31 and September 1, 1991. The height of the shores was measured with a scale and the slope with a clinometer. A Brunton compass was used to get measurements of the orientation of the shoreline and of the direction of some of the layers of sediments. Geological and morphological features were also recorded. All these observations contributed to increase the accuracy of shoreline divisions and descriptions (Figure 2).

POST-GLACIAL HISTORY

Pointe-Taillon is a deltaïc accumulation which was built up at the contact point between the Péribonka river and the Laflamme Sea (Laverdière et Mailloux, 1959) at the end of the last glaciation. After that period, the Lac Saint-Jean gradually replaced this brackish water body. The Laflamme Sea was an extension of the Atlantic Ocean which invaded that inland topographical depression, some 10 000 years B.P. (Lasalle et Tremblay, 1978). The Laflamme Sea incursion occurded after the ice remaining in the depression and the gacial tongue that scoured the Saguenay Fjord had melted away. The marine transgression into these lowlands occurred before the isostatic rebound forced it to regress and to be replaced by fresh waters.

While the earth crust was recovering from ice pressure, the Péribonka River fed by fluvioglacial waters was building up a delta of luniform shape whose front was eroded and transported by littoral drift. Many deltaïc terraces were built up at differents levels from 198m to 105m (Tremblay, 1985). With the lowering of the base level, the

1 Now called Ministère de l'Environnement et de la Faune du Québec.

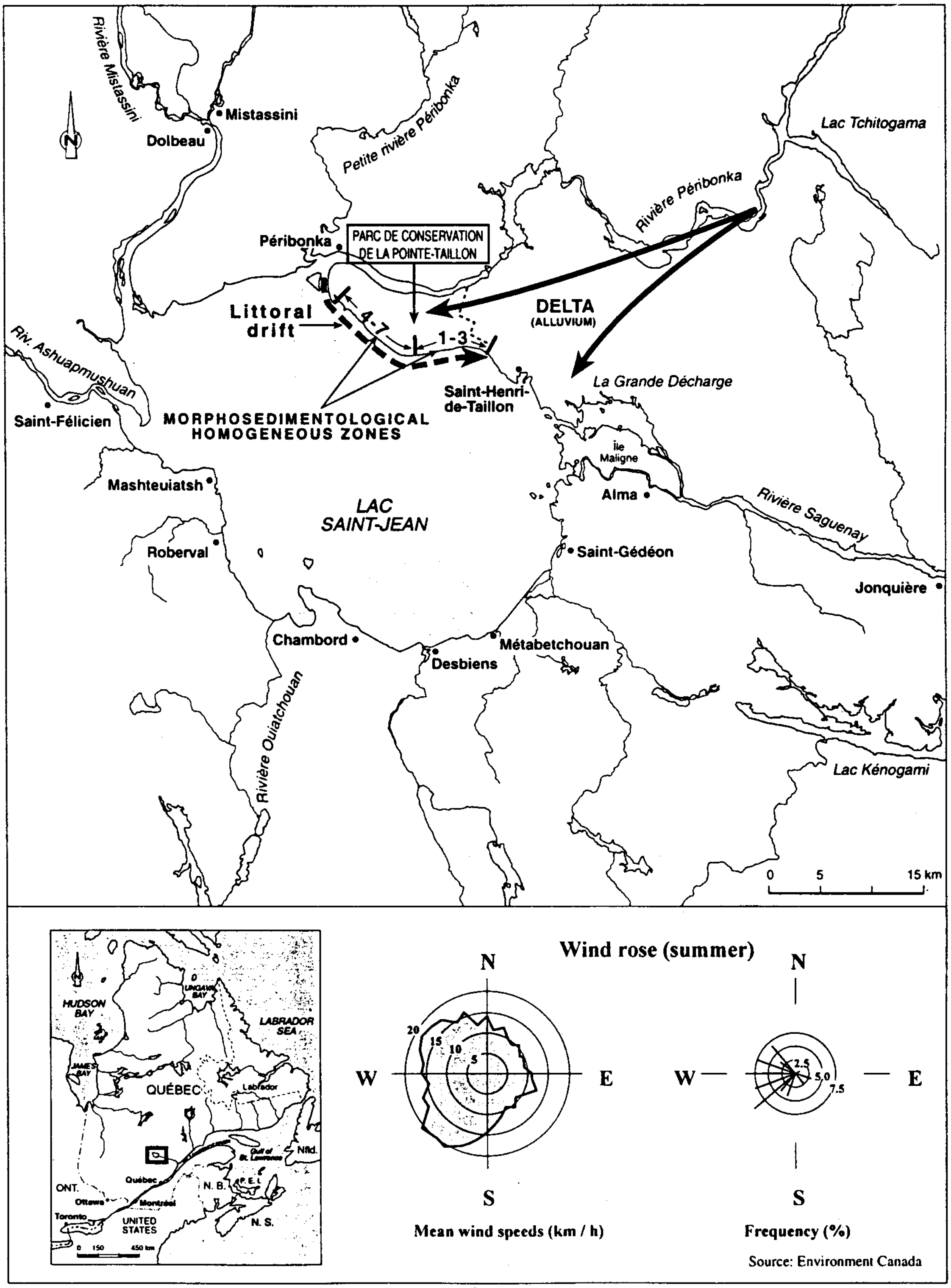

Figure 1: Location map of Pointe-Taillon

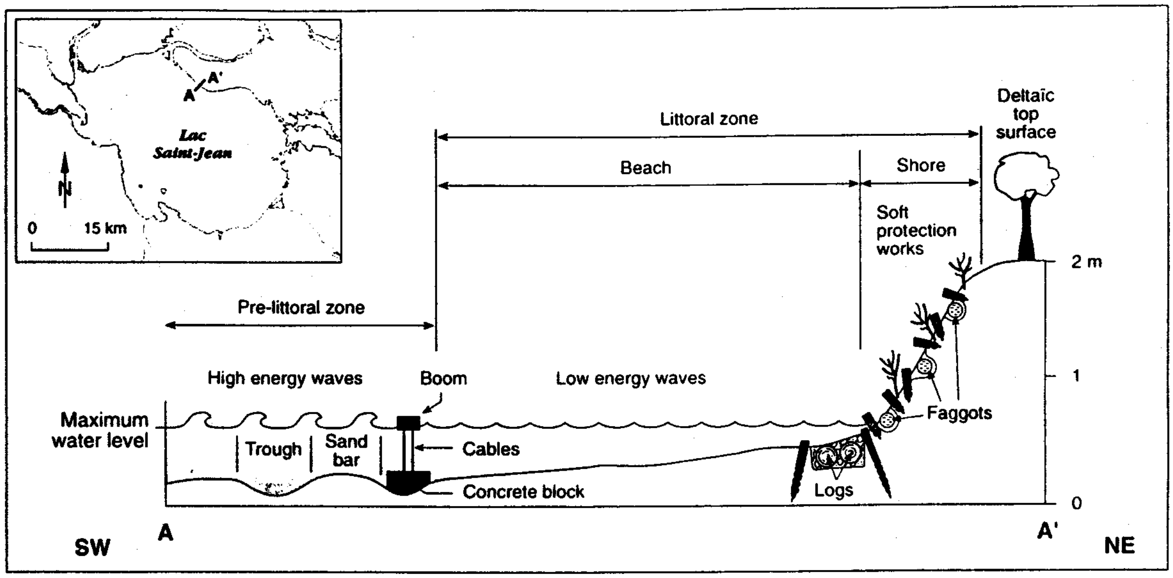

Figure 2: Main components of the littoral zone and remedial works

Péribonka River has eroded its previous deposits and embanked. The coarser particles have accumulated in the littoral zone as frontal and top layers prograding over the basal layers of finer materials that extended in the pre-littoral zone which was deeper and less turbulent.

Pointe-Taillon was formed by the combined action of the Péribonka River and the Laflamme Sea, at the end of the marine transgresion, that is between 7 000 years and 5 000 years B.P. (Lasalle et Tremblay, 1978). On the surface of the delta, channels and troughs between sand bars and levees were formed. Before vegetation developed, these accumulation landforms were transformed under wind action into sand dunes. Some of the oxbow lakes gradually became peat bogs.

As to the present watershed of the Péribonka river, it covers an area of 27 000 km^2. Its waters run into the Lac Saint-Jean, in combination with four other major rivers. The mean annual discharge at its mouth is 589 m^3/s. Lac St-Jean became a reservoir after a generating station, called Isle Maligne, was built up in 1927 at its outlet, by Alcan. Water level goes down to 97.73m in April and then it rises up to 101.84m in June and remains at 101,54m from July to December (Figure 3). Half of Lac Saint-Jean is about 10m deep, especially at the outlet of the main rivers, and the central part of it is 40m deep. The deepest zone is more than 65m. As far as the hydroelectric development of the Péribonka River is concerned, three generating stations of a total capacity of 1 117 MW were built up across this river, between 1952 and 1960.

FIELD OBSERVATIONS

More than 95% of the total length of the Pointe-Taillon lake shoreline is active. All of it is composed of fine to coarse-size sands (0.075mm- 5.0mm) except in the last zone to the NW (No.7) which is made up of gravel and pebbles (Figure 1). The sand layers are often cross-bedded or truncated. Their mean height ranges from 0,75m to 3,0m and their slopes from 25^0 to nearly 90^0. In some places, erosional scars were observed, such as undercutting at the foot of the banks, abrasion scarps. Elsewhere, there was a small debris talus whose slope was about 35^0 and whose height was about one third of the total height of the bank. Above, a nearly vertical wall could be seen. The

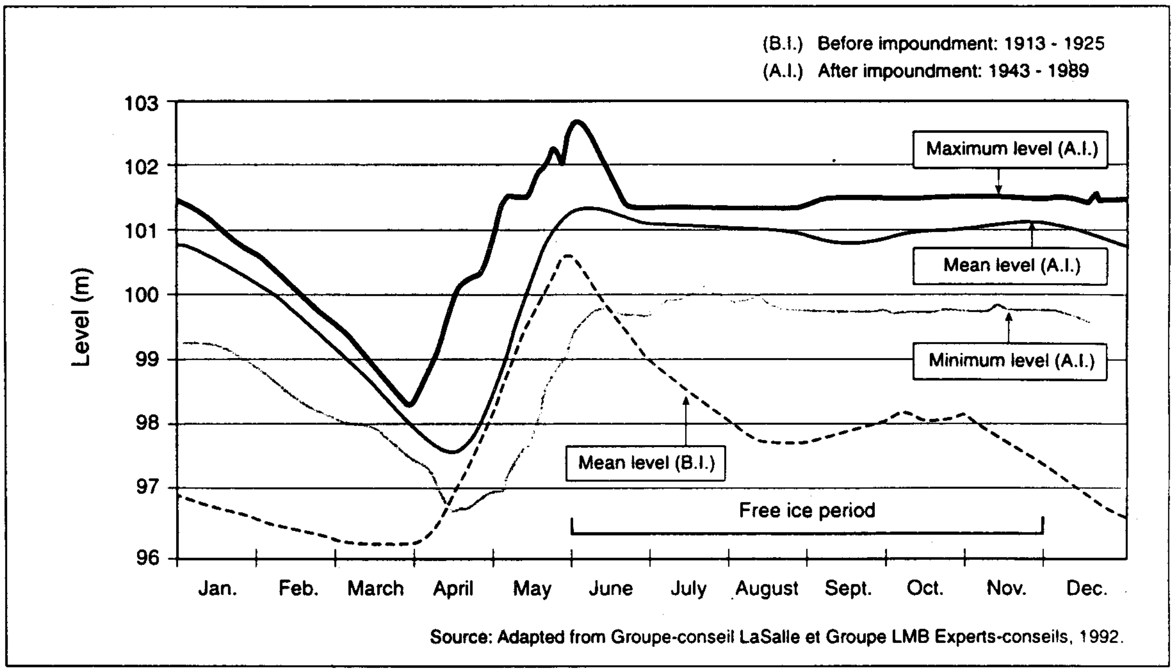

Figure 3: Lac Saint-Jean daily water levels before (B.I.) and after impoundment (A.I.)

only zone where the shoreline is not all active is zone No.1: about two thirds of it is active. A poorly developed berm is present in the upper part of the beach. In that zone, bushes and trees are still in place in the one third uneroded portion of it. Elsewhere, the bush strata has been eroded and the trees, mainly Poplars and Jack Pines have fallen or are falling down.

During field observations, the water level was at 100m above mean sea level, so the beach was extending from 65m at the Southeastern limit of the Park to 300m at the Northwestern part of it, with a mean slope of 5^0 to 8^0. The beach is composed with fine sands overlying silt and clay sediments.

Rock-fill protective berms with a geotextile underneath, were constructed during the spring of 1991 in the most active zones which are zones Nos. 2, 3 and 4. The total length of these remedial works is 1,87 km. They are 1 to 2m high and 10 to 15m wide. As to the anorthosite angular blocks used, their mean diameter is 1m. The decision to build up such structures was linked to the shores retreat following the development of a cycling track the year before along the edge of the deltaïc terrace. Sometimes, this track follows the boundaries of large bogs where special devices allow for birdwatching and plant observation.

The first three homogeneous zones from a morphosedimentological point of view, extended from the eastern limit of the Park to a point where the shore changes from an East-West to a Southeast-Northwest direction. Zones No.4 to No.7 extend in this last direction. Accordingly, the first group of zones face directly winds blowing from the South while the last group face winds from the West.

RECENT EVOLUTION OF THE LITTORAL ZONE

The littoral zone comprises both the shore and the beach (Figure 2). First, I will focus on shore erosion and then on beach evolution.

Bank erosion

By comparing aerial photographies of different dates and measuring distances from the edge of the deltaic terrace to fixed points as previously mentioned, the evolution of shores in every homogeneous zone has been followed from 1959 to 1991, that is over a period of 32 years.

The photos from 1959 up to 1985 show little erosion along the shoreline. Of course, it was different from one zone to the other and more or less important, from time to time. During this 26 years period, the shoreline was almost stable, except in zone No.2 where the average annual retreat was about 0.4m. Zone No.1 was the only zone that seemed to have advanced.

Table 1 shows results of erosion that occurred between 1981-1991. Aerial photos taken on June 18 and 19, 1981 and the ones taken on August 25 and September 3, 1991 are of the same scale (1:15 000), so errors from width measurements are reduced. The total volume of sediments eroded from the shores during that period was 200x 10^3 m^3, which means an erosion of 13m^3 of sediments per one meter of length for the total shoreline studied and an average annual shore retreat of 0.55m. Zone No. 5 was the most eroded, with 48% of the total volume of eroded sediments. Zone No.2 follows with 25%, while zones No.3 and No.4 attain respectively 11% of that volume. In these last three zones, rock-fill protection berms were constructed in 1991.

Table 1: Volume of eroded sediments (1981-1991)

					Erosion			
Homogeneous zone	**Length (m)**	**x**	**Width* (m)**	**x**	**Height (m)**	**=**	**Volume (m^3)**	**Volume / length (m^3/m)**
1	640	x	1.00	x	0.75	=	480	0.7
2	2,849	x	11.25	x	1.50	=	48,076	16.9
3	2,690	x	2.85	x	3.00	=	22,999	8.6
4	2,043	x	8.35	x	1.25	=	21,323	10.4
5	5,179	x	6.00	x	3.00	=	93,222	18.0
6	1,018	x	5.95	x	1.00	=	6,057	5.9
7	970	x	3.70	x	1.00	=	3,589	3.7
							Total: 195,746 m^3	

* Measured on aerial photographies.

This important erosion occurred between 1985 and 1991, since conditions observed on air photos from 1959 and 1985 were not that deteriorated. It could be related, in all probability, to the exceptional mid-november 1989 storm, when the water level was at 101.5m, nearly the maximum level of operation during summer, which coincides with the foot of the shore (Figures 2 and 3) according to the agreement signed between the Ministère de l'Environnement et de la Faune du Québec and Alcan in 1986. The prevailing winds with a fetch of 25 to 30km produced high energy waves that undermined the bottom of cliffs. Since then, slope equilibrium has not been reached and erosion is still going on. Gravity processes act at the foot of parts of the cliffs overhanging scars and landslides can occur.

Beach evolution and littoral drift.

According to Tremblay,1985, with the help of oblique photos taken in 1927 (one year after impoundment), sedimentation forms were observed in the channel between the island and the Northwesternmost part of Pointe-Taillon.

But since then and since the Péribonka River was harnessed, sediment load input has decreased. Waves raised by prevailing winds from Westerly directions (60% of the time during summer) that can blow at a mean speed of 20 km/h have become the main transforming factor of the beach during the period free of ice. They induce littoral drift by shore currents. On different aerial photos series, one can observe forms vanishing, such as small islands and shoals in the channel mentioned above. Towards the Southeast, series of sand bars and troughs, parallel to the shore, are easily seen on vertical photos. These subaquatic forms end one kilometer after the shoreline changes to a West-East direction. From that point to the Eastern entrance to the Park, they are replaced by oblique short bars inclined towards the Southeast. In the pre-littoral zone, sand-bars and troughs can also be seen on aerial photograhs. They are parallel to the shoreline. The shore drift will not last, since the Péribonka river load input has stopped. On the other hand, during summer, when water level is low and the weather is dry, fine sediments of the beach can also be transported by the wind. Such conditions occurred in 1968 and in 1991 when the water level was very low at 100.28m and 100.0m, respectively.

REMEDIAL MEASURES

During the spring of 1991, rock-fill protective berms were built up in three of the four most active zones of Pointe-Taillon Park. It was urgent to protect the shores themselves, as well as the recent bicycle track and the peat bogs located just beyond the shoreline. In other sensitive places around Lac Saint-Jean, Alcan has been using these hard techniques for several years. Elsewhere, gravel with sand on top of it was spread over levelled slopes or groins were constructed to provoke sedimentation (Marsan, A. & Ass., 1983).

At Pointe-Taillon, in order to protect other parts of the shore, namely in the most eroded zone No. 5, faggots together with shrubs and herbaceous plants fixed in between can be used in the slope, while logs can be put at the foot of the talus, parallel to the shoreline (Figure 2). Such techniques and others we could call soft measures are quite common measures proposed both by the Ministère de l'Environnement et de la Faune du Québec and by Environment Canada. They were also recommended by Le Groupe-Conseil LaSalle inc. and by Le Groupe LMB Experts-Conseils 1992 Inc.(1992). Wood debris already on place or fallen from the deltaïc top surface could be used for such a purpose. In addition to these techniques, we suggest to install booms with cables attached to concrete blocks, at the boundary of littoral and pre-littoral zones (Figure 2). Such booms could be found in the neighbourhood, since lumber and pulp and paper companies were using them to gather logs. By setting up these booms on a line parallel to the shoreline, erosion activity of high energy waves could be stopped along the most sensitive shores. Low energy waves left behind the booms will not undermine the foot of these shores. Thus, riparian vegetation will be able to develop and contribute itself to shore stabilisation.

While setting up booms, an adequate distance must be left between each of them, so that boats can access lake. Cables that link booms and concrete blocks must be adjusted to fluctuate with water levels. If ice conditions bring too much problem, booms could be removed just before the formation of the ice cover.

Another measure could also be considered. It deals with shore drift. If Park managers want to slow down the sand transportation from the beaches, they could construct a groin of wooden pilings, transverse to the shoreline, at the Eastern boundary of the Park. This work would help to have beaches broadened upstream and narrowed dowstream. Recreational uses of the beach will take advantage of this measure. However, there is a need of studies to be done on a scale-model in order to know if erosion problems among others can come out dowstream .

A monitoring program should be implemented in order to follow the evolution of the littoral zone (shoreline and beaches) and to evaluate the efficiency of remedial works set up. On many reservoirs operated by utilities and pulp and paper companies, such remedial measures could be initiated in order to protect sensitive shores together with vegetation communities and habitats.

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MODELLING WATER AVAILABILITY TO RIPARIAN VEGETATION IN AN IMPACTED RIVER SYSTEM

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ABSTRACT

Two processes determining water availability to riparian vegetation along the Sabie River in Kruger National Park are addressed, including, bank seepage in extensive alluvial deposits, and surface flooding in bedrock dominated morphologies. A bank seepage model relating water availability for transpiration to river flow conditions is presented, together with validation and verification provided by analytical and numerical solutions, and field data, respectively. An illustrative application shows that reservoir releases may be specified to optimise the spatial distribution of bank storage. Surface flooding provides recharge of local subsurface storage in bedrock dominated areas of the River. An aerial survey of riparian tree mortalities shows a strong correlation with bedrock influence, indicating the importance of local bedrock topography in determining water availability to riparian vegetation in these areas. This is confirmed by the distribution of drought stress levels amongst a tree species across the macro-channel in a bedrock dominated channel type. Construction of the Injaka Dam on a tributary of the Sabie River in the upper catchment will provide the means to regulate winter base-flows and associated water availability to riparian vegetation in alluvial areas, but may impact severely on the vegetation in bedrock influenced areas by reducing surface flooding.

KEY-WORDS: Kruger National Park / Sabie River / Fluvial Geomorphology / Riparian Vegetation / Bank Seepage / Reservoir Releases / Surface Flooding / Distributary Channels / Drought Conditions / Riparian Tree Stress

INTRODUCTION

The Kruger National Park (KNP) in the Mpumalanga Province, South Africa, (Fig. 1) is one of the world's fourteen major nature conservation areas, covering an area of 19 485 km² (Paynter and Nussey, 1986) and provides refuge to an abundant diversity of wild animal and plant life. Seven major rivers flow through the Park (from west to east) and are critical for the existence of unique biota within the riparian zones of the rivers. As a result of increasing human population pressure for land and water resources in the upstream catchments west of the Park, the flow regimes of a number of rivers have been modified from perennial to seasonal and even ephemeral systems such as the Letaba River (Venter and Deacon, 1994), while other rivers such as the Sabie are experiencing reduced flows giving rise to increased stress levels amongst the natural riverine biota. There is consequently an urgent need to effectively manage the water supplies to the rivers for the maintenance of the riparian ecosystems. In particular, the provision of adequate water supplies to meet the consumptive (transpiration) and non-consumptive (habitat) demands of the riparian vegetation needs to be addressed, since the vegetation contributes to the habitat of invertebrates, fish, reptiles, amphibians, birds and mammals. The riparian vegetation also influences fluvial geomorphology by reducing energy gradients and enhancing sedimentation (Hicken, 1994). This paper focuses on the availability of water to the riparian vegetation along the impacted but relatively pristine Sabie River in the KNP.

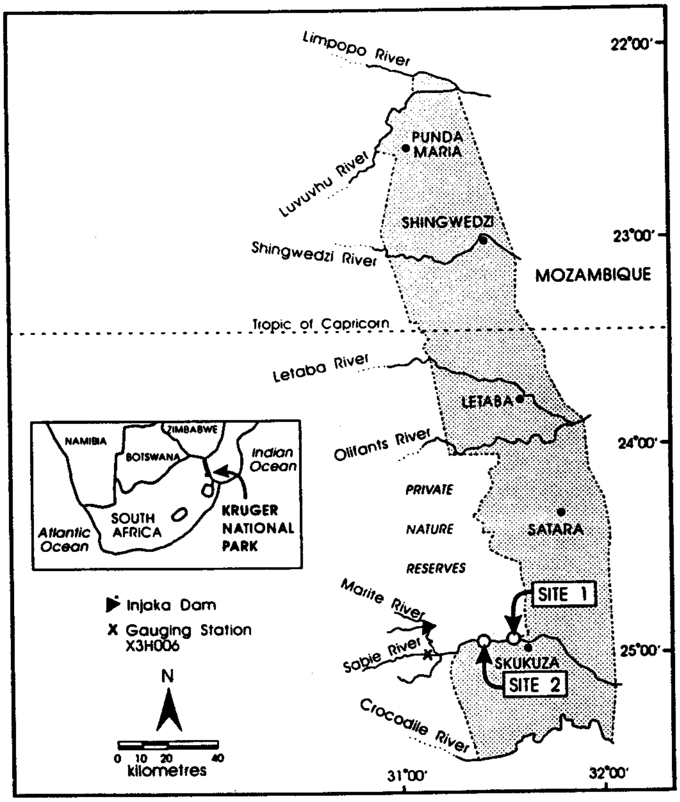

Figure 1: Location of the Injaka Dam, gauging station X3H006 and study sites along the Sabie River in the Kruger National Park, South Africa

The natural flow regime of Sabie River has been modified by well developed afforestation and increasing abstractions for irrigation, resulting in continued depletion of winter base-flows. This is confirmed by the lowest average daily discharge rates recorded in the Sabie River at a station in the upper catchment (station X3H006 in Fig. 1) during 1992 (0.774 m^3/s) and 1983 (0.839 m^3/s). A reduction in the expected annual rainfall and increase in variability over the period 1910/11 to 1985/86 have also been reported (Mason, 1995). Further modifications to the flow regime will result from the construction of the Injaka dam on the Marite River, a tributary of the Sabie River (Fig. 1). The Injaka Dam has a design gross storage capacity of 123 x 10^6 m^3/a (16% of the mean annual runoff of the Sabie River catchment in South Africa) and construction commenced in 1995. The primary objective is to supply domestic water to a large rural population, whilst providing higher assurance supplies for agriculture (Department of Water Affairs and Forestry, 1994). The dam will result in substantial flow reduction during the summer, but provides the means to regulate flows during winter thereby avoiding the impending flow degradation of the Sabie River to a seasonal system.

The Sabie River is incised into a macro-channel that has contained all recorded flows within its banks. The interaction of the underlying bedrock, sediment, channel hydraulics and riparian vegetation has resulted in a river system displaying five principal channel types, including, multi-thread bedrock and mixed anastomosing; mixed pool-rapid; braided and single-thread (van Niekerk *et al.*, 1995). Two distinct processes distribute water to the woody riparian vegetation along the Sabie River depending on the degree of bedrock and alluvial influence, including, (1) bank seepage in predominantly alluvial channel types; and (2) surface flooding in predominantly bedrock influenced channel types.

Bank seepage from active channels in laterally extensive fluvial deposits provides recharge of subsurface storage. Laterally extensive macro-channel deposits typically occur in single-thread, braided, and isolated alluvial sections of the predominantly bedrock influenced pool-rapid and anastomosing channel types. In these hydraulically connected surface-subsurface systems, the consumptive water use by vegetation (potential transpiration) is limited only by plant physiology and climatic conditions, since groundwater is freely available provided adequate base-flow is assured.

In the predominantly bedrock influenced channel types, irregular bedrock topography obstructs seepage from surface water. These channel morphologies are typically characterised by irregular cross-sectional geometries and an intricate network of active and seasonal distributaries. Surface flooding is considered an essential process in these areas of the Sabie River, supplying flow to seasonal channels thereby recharging local subsurface storage and riverine vegetation. The construction of the Injaka Dam may impact severely on the riparian vegetation and related ecosystems in bedrock dominated areas by significantly reducing or eliminating high-flows that periodically inundate cut-off distributaries.

These two processes providing recharge of subsurface water, required to meet transpirative demands, are addressed here. A model relating the temporal and spatial availability of subsurface water to surface flow has been developed for use in extensive alluvial deposits. Code validation and field verification of the model are presented. An application considers the response of phreatic surface levels to reservoir releases of varying magnitude and duration, illustrating that release policies may be optimised using ecological criteria. Signs of drought stress and in several cases fatalities amongst a number of riparian tree species following a recent drought are used to show the importance of surface flooding in morphological channel types with substantial bedrock influence.

MODELLING SURFACE-SUBSURFACE SEEPAGE IN ALLUVIAL DEPOSITS

Numerical Model of Bank Storage Dynamics

The temporal distribution of water in river banks, or bank storage dynamics, may be modelled by numerical solution of the governing partial differential equations describing flow in the saturated and unsaturated zones, subject to the appropriate boundary conditions. The input data required include the temporal distribution of river stage; cross-sectional surface and bedrock topography; alluvial-hydraulic characteristics; climatic conditions (eg.

evaporative demand and rainfall). Incorporating vegetation water use in the model requires additional data, including actual transpiration (function of potential transpirative demand and soil-water potential) and the spatial distribution of roots. Deterministic modelling of bank storage is therefore relatively data intensive, particularly when consumptive water use by riparian vegetation is included.

Two Dimensional Vertical Saturated Flow Model

The equation describing two-dimensional transient flow through a vertical cross-section of unit width is given by

(1) $$\frac{\partial}{\partial x}(K_x\frac{\partial h}{\partial x}) + \frac{\partial}{\partial z}(K_z\frac{\partial h}{\partial z}) = S_s\frac{\partial h}{\partial t} + q_v$$

where x is the horizontal direction, z is the vertical direction, h is the piezometric potential, K_x and K_z are the horizontal and vertical saturated hydraulic conductivities respectively, S_s is the specific storativity, q_v is the sink per unit volume (eg. transpirational extraction), and t is time. Saturated flow dynamics is frequently modelled by including a phreatic surface boundary condition that assumes an instantaneous response of the unsaturated profile. The phreatic surface boundary condition for two-dimensional vertical flow may be expressed as

(2) $$\frac{\partial}{\partial x}(K_x\frac{\partial h}{\partial x}) + \frac{\partial}{\partial z}(K_z\frac{\partial h}{\partial z}) = (S_s+\frac{S_y}{dz})\frac{\partial h}{\partial t} + q_v$$

where S_y is the specific yield. Backward-difference time approximations may be used to express equation (1) in finite-difference form for each node covering the domain of interest. A standard direct solution procedure (LU decomposition with partial pivoting) is used to solve the simultaneous set of finite-difference equations. The assumption of a constant specific yield (equation (2)) is appropriate for long-term hydrogeological investigations and for free-draining materials, but may be inadequate for time-dependant absorption and drainage conditions. Contemporary studies are directed at coupling saturated and unsaturated flow models (Zarandy, 1993), to adequately describe unsaturated flow dynamics.

One-Dimensional Unsaturated Flow Model

The one-dimensional (Richard's) equation describing vertical flow in the unsaturated zone is given by

(3) $$\frac{d\psi}{dt} = -\frac{d\psi}{d\theta}[\frac{\partial}{\partial z}K_z(\psi)(\frac{\partial\psi}{\partial z} - 1)] - q_v\frac{d\psi}{d\theta}$$

where ψ is the soil suction, θ is the volumetric water content, and $K_z(\psi)$ is the unsaturated hydraulic conductivity. Representing the unsaturated zone by a number of vertical models (neglecting horizontal flow) allows a integrated saturated-unsaturated flow model to be developed by coupling equations (1) and (3) across the phreatic surface.

Code Validation and Model Verification

Bank storage response to a sinusoidal stage hydrograph (Fig. 2, inset) is modelled using the saturated (equations (1) and (2)) and coupled saturated-unsaturated (equations (1) and (3)) flow models. A homogeneous and isotropic bank zone is used; the flow system is confined laterally by an impervious boundary (Fig. 2, inset); and the phreatic surface is initially horizontal. The response is also modelled using the one-dimensional analytical model of Cooper and Rorabaugh (1963) and numerical model of Hornberger *et al.* (1970) (Fig. 2). The model of Cooper and Rorabaugh (linearisation results from the approximation of constant saturated thickness) underestimates bank seepage, with the numerical models producing higher estimates and showing close agreement. The adequacy of assuming a constant specific yield (equation (2)) is confirmed by comparison with the coupled saturated-unsaturated model.

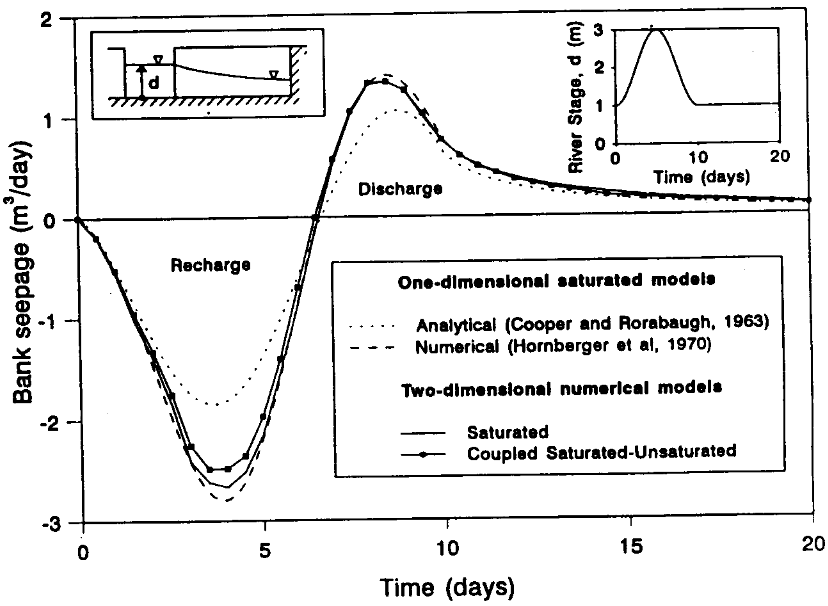

Figure 2: Bank seepage response to a sinusoidal fluctuation of river stage

A study site (site 1 in Fig. 1) on the Sabie River was instrumented to collect data for parameterisation and application of the bank storage dynamics model (Birkhead *et al.*, 1995). The site was selected based on the presence of considerable alluvium (Fig. 3, inset) and suitable biological characteristics. Local hydraulic conditions at the site produce significant longitudinal seepage, and resulted in the extension of the model to quasi three-dimensions. The predicted groundwater response to a sequence of stage hydrographs using the calibrated model are plotted against measured data in Fig. 3. The rapidity of the groundwater response is acceptably replicated by the model, with peak attenuation and lag increasing with distance from the active channel. The model is also used to synthesize the groundwater response to nine distinct hydrographs recorded over the period December 1992 to January 1995. The measured and modelled peak levels are plotted in Fig. 3 (inset) and show excellent correlation, yielding a regression coefficient $R^2 = 0.996$.

Spatial Distribution Of Maximum Phreatic Levels Resulting From Reservoir Releases Of Varying Duration.

The two-dimensional saturated flow model is applied to illustrate the implications of different release hydrographs on the spatial availability of water in a river bank. The release volume, channel geometry and alluvial-hydraulic data used in the analysis are given in Table 1. Manning's resistance equation is used to establish the uniform rating curve. A fixed volume of water is released from storage at different constant rates of rise in river stage. The model is used to predict the maximum phreatic surface level at different distances from the channel resulting from the releases of varying peak stage and duration (Fig. 4). The analysis shows that at a given distance from the channel, a maximum level is obtained which is associated with a specific release hydrograph. The locus of maxima vary spatially, with lower magnitude (higher duration) events resulting in higher levels further from the channel. The maxima are less distinct with increased distance, however, due to the effects of flow attenuation (as observed in Fig. 3), and marginal benefits are achieved by varying the rate of rise for $x > 10$ m.

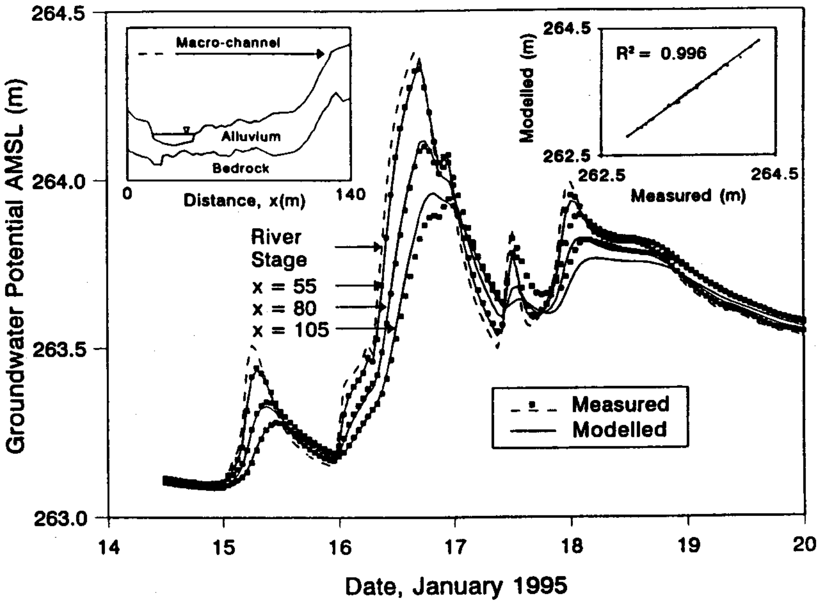

Figure 3: Measured and modelled groundwater response at site 1 on the Sabie River

Table 1: Release volume, channel and alluvial characteristics

Release Volume (m^3)	**100 000**
Channel bed width (m)	**1**
Channel side slope	**1:1**
Manning's resistance coefficient	**0.03**
Base-flow river stage (m)	**1**
Lateral extent of deposit (m)	**100**
Hydraulic conductivity (m/day)	**10**
Specific yield	**0.3**

Storage releases may therefore be defined to satisfy specific ecological requirements. For example, assuming a riparian plant species located 5 m from the river bank requires periodic inundation of the root zone up to a level of at least 1.45 m. This may be achieved for the allocated release volume (Table 1) by providing rates of rise in stage level in the range 3 to 40 m/day (Fig. 4, inset). Values outside of this range will fail to satisfy the ecological requirement. Spatial optimisation of bank storage for different channel geometries has shown that maximisation is a consequence of non-linear rating relationships (ie. discharge rate increases exponentially with stage) that are generally characteristic of natural river systems. Hydrographs released from upstream storage are modified with distance travelled (unsteady flow and temporal bank storage effects), resulting in different bank storage responses downstream. For the illustrative example, the event at the upper limit of the range (40 m/day) must be specified as the release to ensure that attenuation does not reduce the rate of rise in stage below that needed to satisfy the downstream ecological requirements.

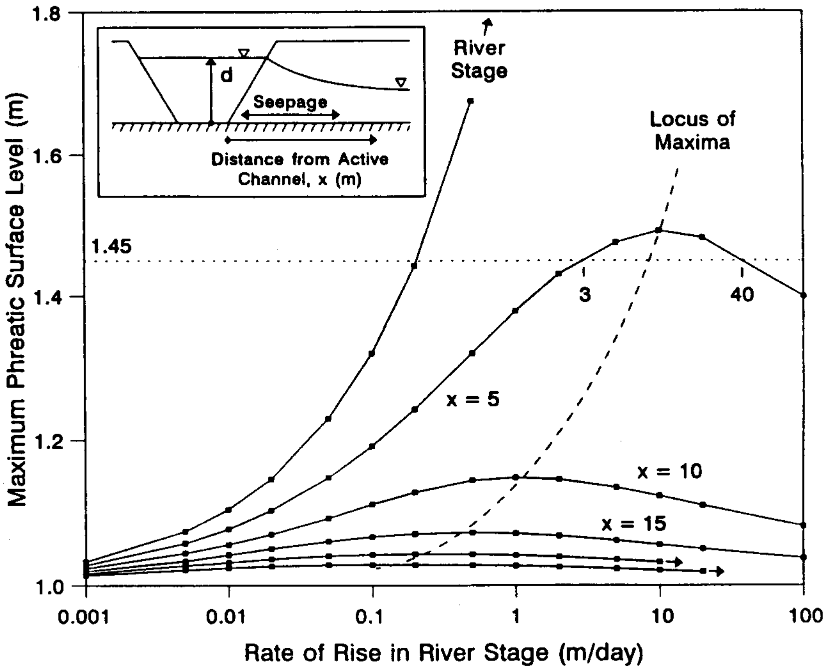

Figure 4: Spatial distribution of maximum phreatic surface levels resulting from different releases

SURFACE FLOODING, DROUGHT CONDITIONS AND RESULTANT STRESS AMONGST RIPARIAN TREE SPECIES

In predominantly bedrock influenced morphologies, recharge of local subsurface storage and associated water availability to riparian vegetation depends on the activation of seasonal distributaries by surface flooding. A severe drought from 1990 to 1992 provided an opportunity to observe the effects of drought stress and the importance of surface flooding in bedrock influenced morphologies. The distribution of stress levels and mortalities amongst riparian tree species were recorded along the length of the Sabie River in the KNP.

Distribution of Riparian Tree Mortalities

An aerial survey of dead riparian trees covering the extent of the Sabie River in the KNP was undertaken in March 1993. The number of individual mortalities and species were noted, together with their location along the River. The number of dead individuals were weighted according to the density of trees, and the relative proportion of each species, in each of the channel types. The results of the analysis are presented in Fig. 5, and show a clear correlation of higher mean species mortality with increased bedrock influence. The above average mortality for certain species is attributed to the effect of low replicate numbers on the weighting procedure. An exception are the higher occurrences of *Ficus sycomorus* and *Combretum erythrophyllum* in bedrock and mixed anastomosing channel types, which may possibly be related to higher consumptive water requirements. Spatial distributions of tree mortalities in bedrock dominated morphologies were observed to be patchy during the aerial survey. This patchiness was attributed to the influence of local bedrock topography, preventing subsurface seepage from active distributaries. To test this hypothesis, the distribution of stress levels across various channel types were surveyed in the field. The results from a bedrock anastomosing morphological channel type are presented.

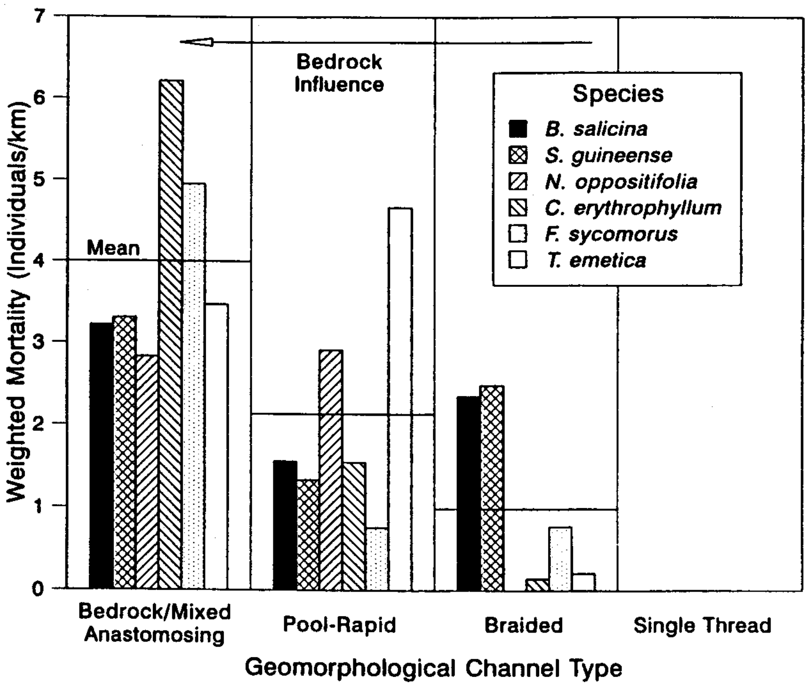

Figure 5: Distribution of weighted mortalities of the riparian tree species *Breonadia salicina*, *Syzygium guineense*, *Nuxia oppositifolia*, *Crombretum erythrophyllum*, *Ficus sycomorus* and *Trichilia emetica* according to geomorphological channel type

Distribution of Tree Stress Across a Bedrock Anastomosing Channel Type

The condition of riparian tree stems, bark and leaves within a 20 m wide belt transect across the macro-channel were recorded as visual indictors of drought stress. The proportional occurrence of dead tree stems was found to provide a suitable measure of stress conditions. Figure 6 shows the distribution of stem conditions for *Breonadia salicina* superimposed on a cross-sectional profile through the bedrock anastomosing channel type (site 2 in Fig. 1). The 125 m section of the 300 m wide macro-channel contains one seasonal and two active distributaries. High stress levels observed in *B. salicina* are associated with individuals located along the seasonal channel and also, to a lesser degree, on the north bank of active distributary 1. 27% and 12% of the trees (6 individuals within an area of 20 m x 20 m) colonising the north and south banks of the seasonal channel, respectively, showed between 75% and 100% stem mortality. The stage levels in the active channels corresponding to a discharge rate of 6 m^3/s are indicated, with flow activated in the seasonal distributary at this discharge rate (Broadhurst *et al.*, 1996). The spatial distribution of tree stress in *B. salicina* illustrates the importance of surface flooding for recharging local subsurface storage along cut-off channels. Consequently, flow conditions over the drought period, relative to historical flows, are examined.

River Flow Conditions

Since surface flooding activates flow in the seasonal distributaries, where riparian vegetation stress levels are noted to be highest, the maximum average daily discharge recorded over the summer periods is relevant. The discharges at the gauging station (station X3H006 in Fig. 1) located in the upper catchment, with the longest available record, are examined (Fig. 7). The flow reductions resulting from abstractions along the intervening stretch of river are neglected. The lowest recorded maximum average daily discharge occurred during the 1991/92 season (6.4 m^3/s),

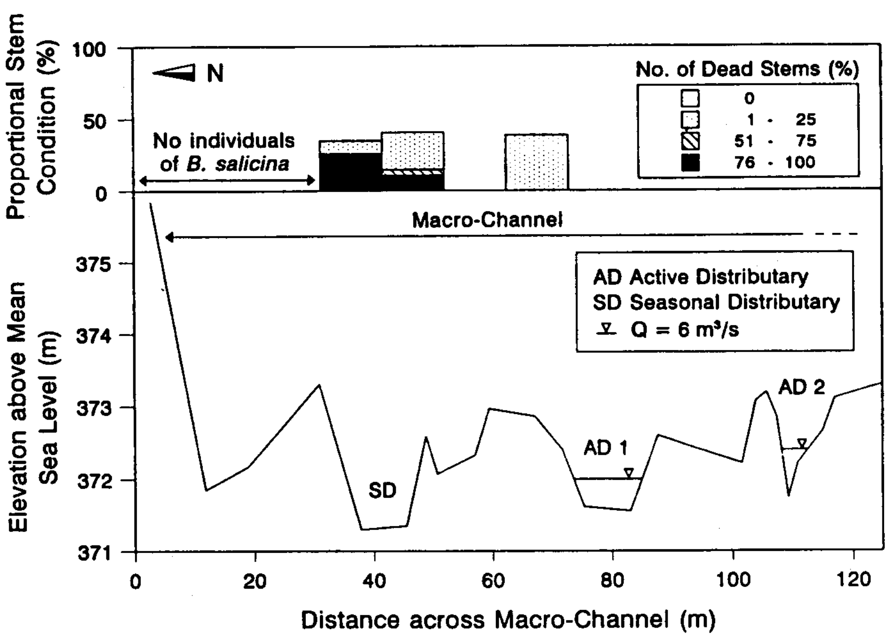

Figure 6: Distribution of proportional stem condition for the species Breonadia salicina across a bedrock anastomosing channel type

with the second lowest (8.2 m^3/s) recorded during the 1982/83 season. The 1981/83 period also experienced the lowest successive two and three seasonal flows in terms of peak discharge, as may be inferred from Fig. 7. It is possible that the maximum flow during 1991/92 dropped below a critical level, but this is unlikely since the differences between the flows during the 1982/83 and 1991/92 seasons are not markedly different; the lower summer flows during the early 1980's were more persistent and no tree mortalities were observed during this period. Although surface flooding has been shown to be an essential process for recharging subsurface storage in bedrock influenced morphologies (Fig. 6), analysis of the flow record does not provide an adequate explanation for the widespread tree mortalities observed in 1992. The relative local climate at Skukuza over the drought period is therefore also examined for a possible reason.

Climatic Conditions at Skukuza

The annual rainfall and Symons pan evaporation recorded at Skukuza are presented relative to mean annual values in Fig. 8 and Fig. 9, respectively. The lowest rainfall and highest evaporative demand were recorded over the 1991/92 season, with marginally higher rainfall and lower evaporative demand during 1982/83. 1991/92 may also be shown to be the season ending the successive two and three most severe dry periods on record.

Based on these data, the widespread riparian tree mortalities are attributed to the below average rainfall and above average evaporative demand which intensified over three seasons, coupled with the low summer-flows experienced in 1991/92. The accumulated effect and timing of these severe environmental conditions are likely to have depleted the energy reserves of tree species to critical levels, whence stress and mortalities were noted.

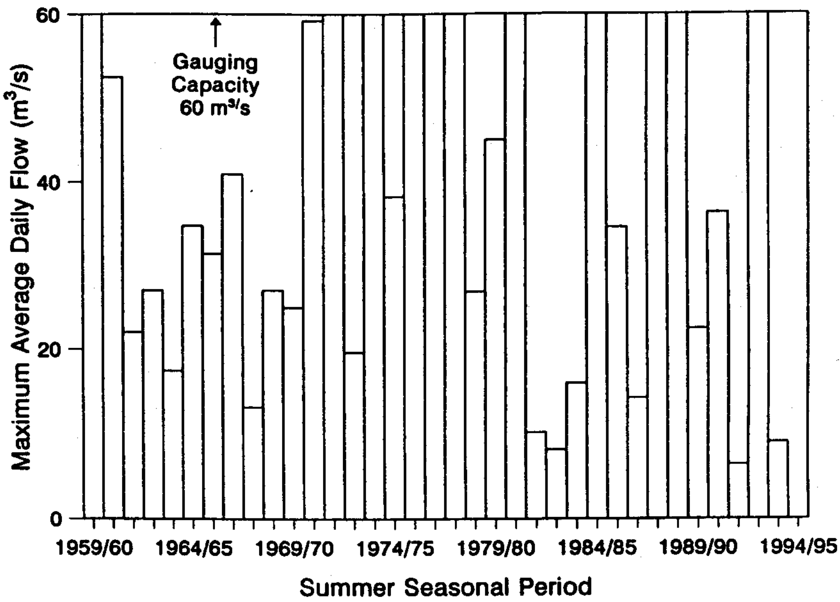

Figure 7: Maximum average daily summer flow recorded at station X3H006 from 1959/60 to 1993/94

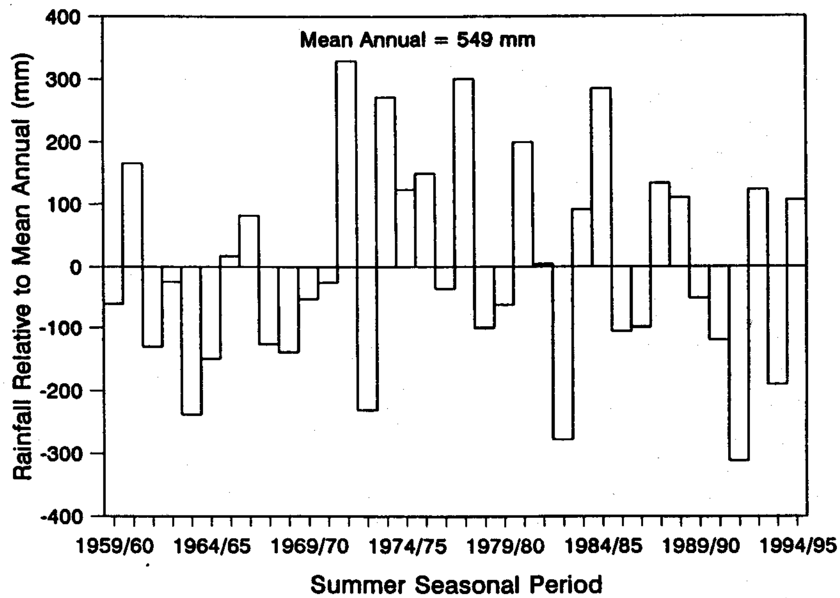

Figure 8: Rainfall recorded at Skukuza relative to the mean annual from 1959/60 to 1994/95

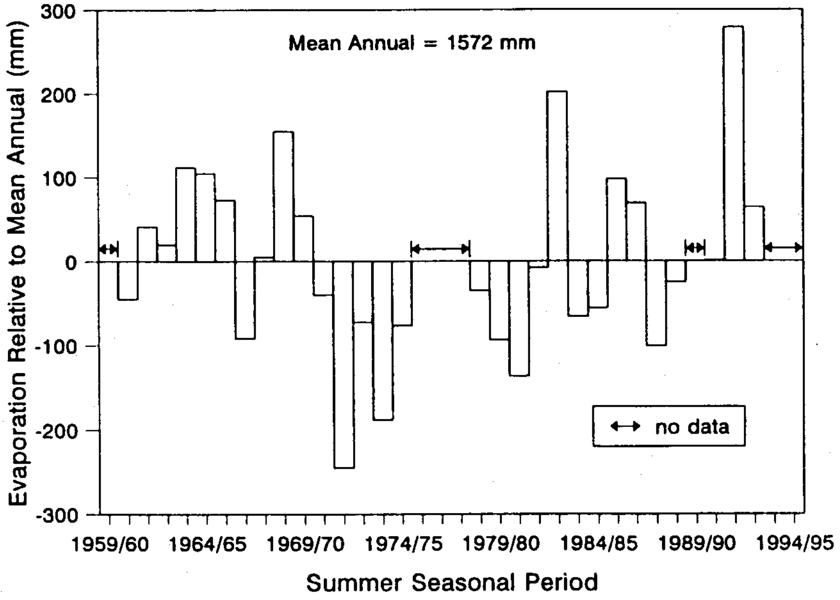

Figure 9: Pan evaporation recorded at Skukuza relative to the mean annual from 1960/61 to 1992/93

CONCLUSIONS

The two process determining water availability to riparian vegetation are dependant on the degree of bedrock influence and associated connectivity between surface and subsurface water. These processes include, (1) bank seepage in extensive alluvial deposits, which may be modelled deterministically; and (2) flow activation in cut-off seasonal distributaries by surface flooding. Application of a bank storage dynamics model shows that reservoir releases may be managed to maximise phreatic surface levels in the bank, but benefits are reduced with increased distance from the river. Riparian tree mortalities following a recent drought are strongly correlated with the degree of bedrock influence. Stress conditions noted in a particular species across a bedrock dominated morphological channel type show stress levels to coincide with a cut-off seasonal distributary. This illustrates the importance of periodic surface flooding which will be severely reduced by construction of the Injaka Dam. The widespread riparian tree mortalities observed along the Sabie River in 1992 are attributed to the timing and combination of low summer-flows and extreme climatic conditions.

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STRATIFICATION OF THE MIRAMICHI ESTUARY: IMPACT OF FRESHWATER DISCHARGE.

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ABSTRACT

The impact of freshwater discharge on stratification-destratification processes and the psositon of the salt wedge was examined in the Miramichi Estuary (New Brunswick, Canada). Variations in the longitudinal and vertical salinity gradients in the upper reaches of the estuary were described for 2 dates with relatively low freshwater discharge and 2 dates with relatively high freshwater discharge during the summer of 1992. The vertical salinity gradient in the upper reaches of the estuary, quantified using the Stratification Parameter, was similar to other salt wedge estuaries. The power required to mix the discharge was at least two orders of magnitude higher in the upper reaches of the estuary than in an estuary with similar shape and depth. An analysis of the isohaline curves showed that the tip of the salt wedge was consistently located further upstream than was suggested previously. The analysis also showed that the magnitude of freshwater discharge alone does not determine stratification, even when the assessment is limited to the upper reaches of the estuary.

The extensive intrusion of the salt wedge into the upper estuary indicates a larger nursery and rearing habitat for brackish water biota than was previously realized. Furthermore, the stability of vertical structure should result in a relatively constant and predictable environment for finfish and invertebrates. Expsosure of juvenile finfish to sediments containing high levels of metals and organic pollutants may be determined by vertical stratification.

KEY-WORDS: estuaries/ freshwater discharge /stratification /ecology /mixing power .

1. INTRODUCTION

Many abiotic factors interact in estuaries and affect estuarine ecology. Among them, the presence of a salt wedge and stratification-destratification processes have important effects on pelagic and benthic biota (Schroeder et al., 1990). Longitudinal salinity gradients and the location of the salt wedge influence the distribution of zooplankton (Bousfield, 1955) and both larval (El'sayed 1967; Laprise and Dodson 1989; Locke and Courtenay, 1995) and adult (Hanson and Courtenay, 1995) fish. Transport in surface or bottom currents, selected by vertical migrations between strata, is important to both invertebrate (Bousfield, 1955) and fishes (Laprise and Dodson, 1989).

The Miramichi Estuary (New Brunswick) is one of the largest estuaries in eastern Canada. It supports a number of important commercial fisheries as well as the biggest Atlantic salmon (*Salmo salar*) population in North America. Located in the southwest portion of the Gulf of St Lawrence, the drainage area of the Miramichi Estuary covers 14 000 km^2. The river portion of the estuary is typically less than 1 km wide and stretches approximately 50 km from the head of tide near Red Bank, to Sheldrake Island (Figure 1). The Inner Bay encompasses an area of 300 km^2, with depths typically varying between 2 and 5 m. A dredged channel, over 7 m deep, extends from outside the Barrier Islands to the town of Newcastle.

A number of studies have been carried out on the Miramichi Estuary over the past 50 years. Bousfield (1955) provided the first important report on the physical oceanography of the estuary. More recently, research effort has intensified and has led to a detailed analysis of the seasonal variations in the physical oceanography (Lafleur et al., 1995) as well as a general description of the stratification in the estuary (St-Hilaire et al., 1995). Much of this work relied on moored instruments and was concentrated in the lower part of the narrow portion of the estuary, between Newcastle and Cheval Point (Figure 1). Both of these studies showed that freshwater discharge greatly affects the stratification of the Miramichi. Lafleur et al. (1995) concluded that with increased freshwater runoff a thicker halocline develops and longitudinal salinity gradients between Chatham and the Inner Bay strengthen at the bottom but weaken at the surface. St-Hilaire et al. (1995) compared mixing powers of the wind, tides and the resistance to mixing related to freshwater discharge. For weak freshwater flows, the stratifying effects are always greater than the mixing power of winds and tidal currents. The study concluded that the estuary remains stratified during the ice-free season.The spring-neap migration of the salt wedge was estimated to be 8 km and the seasonal migration appeared to be 18 km (Lafleur et al., 1995). The question of the upstream limit of that migration could not be completely answered because of the lack of data in the upper part of the estuary.

Although these previous studies provide a good overview of the physical oceanographic conditions of the estuary, there is little information available on the extent of the upstream migration of the salt wedge and the influence of discharge on the stratification in the upper reaches of the estuary, upstream of Newcastle. Moreover, previous studies used different definitions of the salt wedge. Vilks and Krauel (1982) defined the salt wedge by water of salinity greater than 12 $^0/_{00}$. Lafleur et al. (1995) defined the limit of the salt wedge as the upstream limit of the salt-water intrusion at the bottom of a stratified estuary. Our definition is more quantitative and based on potential ecological applications and thus differs from that of Vilks and Krauel (1982). The limit of the salt wedge is the upstream location where bottom water has a measurable salinity (i.e greater than 0 ± 0.02 Practical Salinity Units, as measured by Conductivity, Temperature and Depth sensors).

The objective of this paper is to describe the variations in the longitudinal and vertical salinity gradients in the upper reaches of the estuary, and the influence of freshwater discharge. The characterization of the salinity gradients in the upper estuary was done through two types of analysis: (1) a descriptive study of isohaline curves and (2) a comparison of the power required to mix the discharge on days with similar

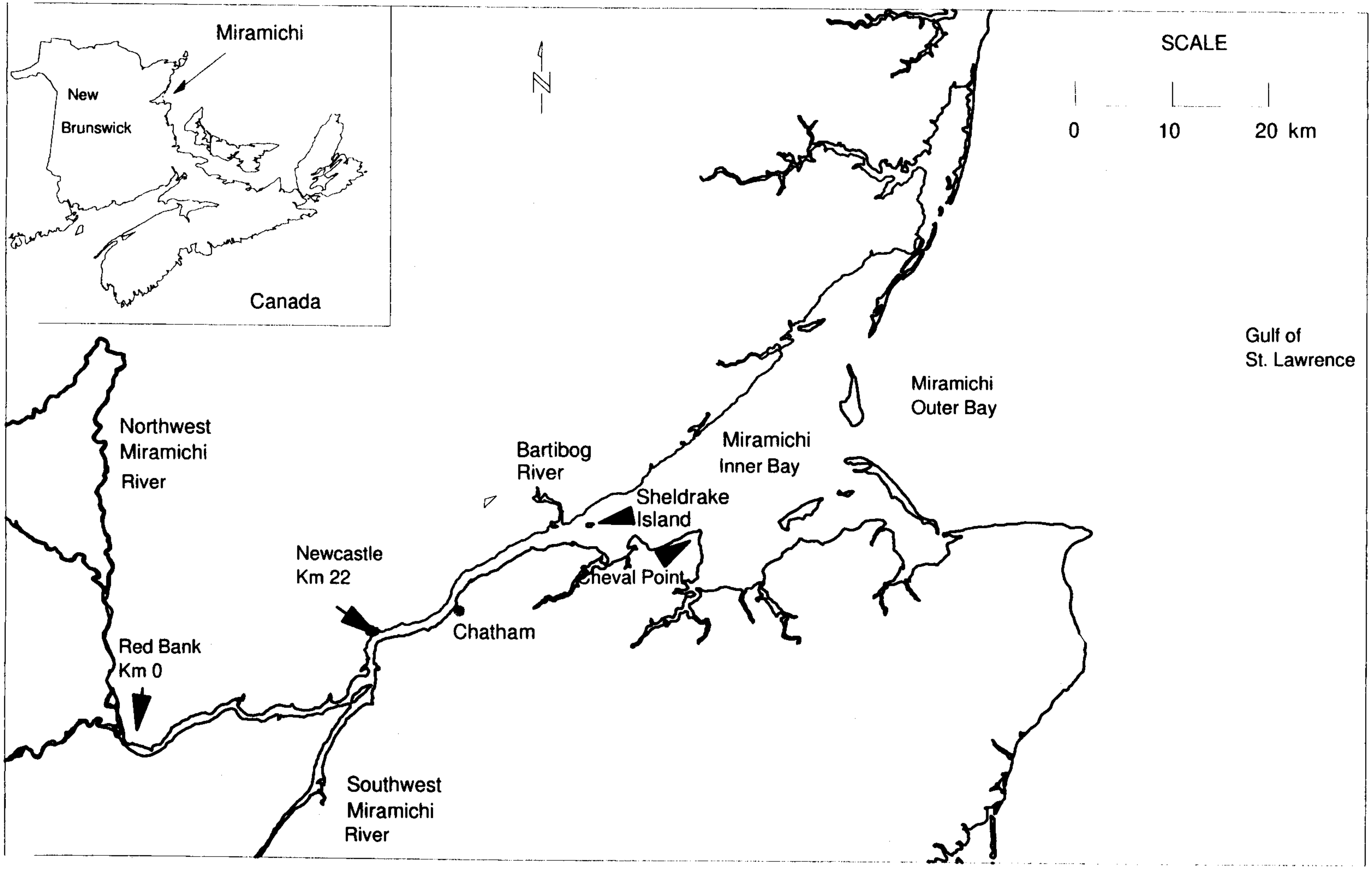

Figure 1. The Miramichi Estuary, showing places mentioned in the text. The study area was located between Red Bank (km 0) and Newcastle (km 22).

freshwater flows. Shroeder et al. (1990) used the same technique on a similar estuary, Mobile Bay (Alabama), which is also river-dominated. We were thus able to compare the two estuaries.

2. MATERIALS AND METHODS

Salinity data were obtained from a series of CTD (Conductivity,Temperature, Depth) casts taken on 4 dates (25 May, 10 June, 15 July and 25 August) in 1992, between Red Bank (km 0) and Newcastle (km 22). The instrument used to perform the CTD casts was a SEABIRD SBE-19 equipped with a conductivity cell, a thermistor and a hydrostatic pressure sensor. Table 1 gives the precision of the sensors. The SEABIRD was also equipped with a pump to ensure constant flushing of the conductivity cell. At each station, the CTD was held just below the surface for 45 seconds to allow the pump to be activated and to enable constant flushing of the conductivity cell. The CTD was then lowered at a constant speed of approximately 0.5 m s^{-1} from the surface to the bottom. Measurements were recorded every 0.5 seconds. The data stored in the CTD was downloaded later to a computer. Salinity was calculated from conductivity, temperature and pressure data using the Practical Salinity Scale (Fofonoff and Millard, 1983), which is virtually identical to the Parts Per Thousand (PPT) scale.

Table 1. Sensor Precision (SEABIRD CTD)

Sensors	Precision
Pressure (Depth)	0.25% of full range (25 cm)
Conductivity	0.01 mS cm^{-1}
Temperature	0.01 °C

Isohaline curves were constructed from CTD salinity data using a Contour Graph routine with a Distance Weighed Square (DWLS) smoothing function (Systat, Version 5.0). Freshwater discharge data were obtained from Environment Canada for station 01BO001 (Southwest Miramichi at Blackville), located upstream of the sampling area. This station has the largest gauged drainage area on the Miramichi drainage basin (5 050 km^2). The annual mean at the station is 146 $m^3 s^{-1}$ (Inland Waters Directorate, Environment Canada, 1991). The discharge values at the hydrometric station were used to estimate the discharge for the entire estuary, based on the ungauged:gauged area ratio (2.772:1).

2.1 Estimation of power required to mix the discharge.

Many simplified mixing models exist for estuaries. The main mixing factors are the wind and tidal currents. These factors are sufficient to mix the water column if their combined power is greater than the energy flux caused by freshwater flow, in the form of buoyancy variations. The method used here to estimate the power required to mix the freshwater flow is that of Shroeder et al. (1990) which is simpler than the method used by St-Hilaire et al. (1995) and allows for direct comparison between two estuaries: Mobile Bay (Alabama, U.S.), studied by Shroeder et al. (1990), and the Miramichi. Mobile Bay is also strongly influenced by discharge. It has a similar shape to the Miramichi and is relatively shallow (3 to 4 m). Equation 1 was applied to the discharge data for the sampling dates.

(1) $$W_r = \rho' D (g/\rho_w) (Q\rho_w/A)$$

where: W_r = Power required to mix a river discharge Q (W m^{-2})
ρ' = Surface-to-bottom density deficit (10 kg m^{-3})
D = Typical water depth in the study area (3 m)
g = Ggravitational acceleration
A = Area of the estuary (300 km^2)
ρ_w = 1020 kg m^{-3} typical density

W_r values were calculated for sampling days with equation 1 and estimated values of discharge. The surface to bottom density deficit (ρ') was estimated from our salinity data and temperature data from Figure 8 of Lafleur et al. (1995).

The stratification (S_p) parameter was also calculated from CTD casts. S_p is simply a ratio of the bottom to surface salinity difference to the depth-averaged salinity (Equation 2, from Kjerfve and Greer, 1978). It is often used in conjunction with other oceanographic parameters to quantify the stratification of estuaries (Hughes and Rattray, 1980).

(2) $S_p = \Delta s / s_o$

where: Δs = Bbottom to surface salinity difference
s_o = Depth-averaged salinity

3. RESULTS

3.1 Discharge and Power Required to Mix Discharge

Analysis focused on 4 dates during the summer of 1992 for which CTD data were available. The dates selected allow comparisons between two events with discharges near historic summer means (Table 2): 10 June (Q = 142 $m^3 s^{-1}$) and 25 August (Q = 166 $m^3 s^{-1}$) . The other two survey dates (25 May, Q = 318 $m^3 s^{-1}$ and 15 July, Q = 304 $m^3 s^{-1}$) had discharge higher than summer means but similar to the 1992 summer mean (Table 2). The main hydrological event is the spring flood associated with snowmelt (Figure 2). This is reflected in Table 2 by the higher monthly mean discharge for the month of May which in 1992 was lower than the historical mean. Sampling began during the falling limb of the main spring flood (25 May or day 145, Figure 2). During the sampling period, there were 3 important hydrological events with discharge exceeding 200 $m^3 s^{-1}$ at the hydrometric station. The maximum estimated discharge during the sampling period was 923 $m^3 s^{-1}$ on 6 August while the minimum was 89 $m^3 s^{-1}$ on 24 June. Generally, the months of July and August had higher than usual discharge (Table 2).

Table 2. Monthly mean discharge: 1992 and historical means (From Inland Waters Directorate, Environment Canada, 1991)

Months	Historical (1918-1990) monthly mean estimated for the estuary ($m^3 s^{-1}$)	Calculated Monthly mean for 1992 ($m^3 s^{-1}$)
May	887.0	676.4
June	307.7	169.6
July	166.3	342.3
August	155.2	320.2

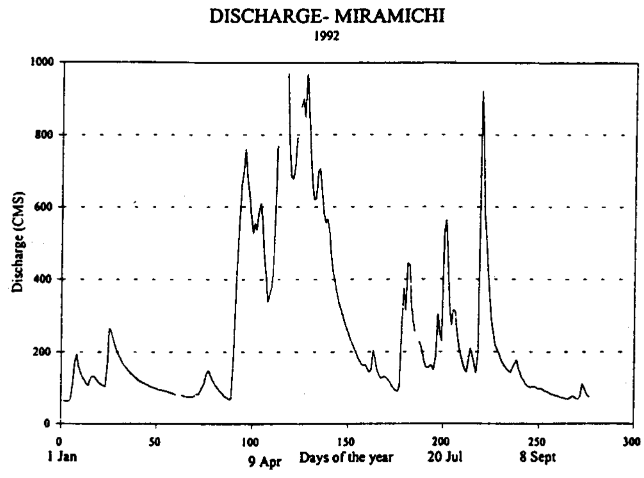

Figure 2. Daily discharge for the Miramichi Estuary, estimated from Hydrometric station 01BO001

The highest W_r value was 49.8 x 10^{-4} W m^{-2}, obtained on 25 May for a discharge of 318 $m^3 s^{-1}$ while the lowest value was 2.3 x 10^{-4} W m^{-2}, obtained on 10 June for a discharge of 142 $m^3 s^{-1}$ (Table 3). The W_r value for 15 July, which had a similar discharge to 10 June, was only 2.3 x 10^{-4} W m^{-2}

Table 3. Power required to mix the freshwater discharge in the upper reaches of the estuary

Dates	Surface-bottom density deficit ρ'(kg m^{-3})	Discharge ($m^3 s^{-1}$)	Power W_r (10^{-4} W m^{-2})
25 May	4	318	49.8
10 June	0.4	142	2.3
15 July	3	304	35.7
25 August	3	166	19.5

<u>3.2 Isohaline Curves</u>

Figures 3 to 6 show isohaline curves from 4 CTD surveys. Dense horizontal isohalines represent strong vertical stratification. On 25 May (Figure 3, Q = 318 $m^3 s^{-1}$), the entire water column upstream of km 17 had a salinity less than 2 PSU. The water column between km 10 and km 20 showed little vertical stratification (less than 0.4 PSU m^{-1}). Vertical stratification started to increase at km 20. At km 25, the salinity at the surface was 5 PSU and reached 14 PSU at a depth of 3 m. Figure 4 (10 June, Q = 136 $m^3 s^{-1}$) shows a similar position of the salt wedge, in spite of the fact that the discharge had decreased significantly from May 25. On 15 July (Figure 5, Q = 304 $m^3 s^{-1}$), the 2 PSU isohaline was pushed back downstream of km 20. A strong vertical gradient of 1 PSU m^{-1} occurred at km 25. Figure 6 (August

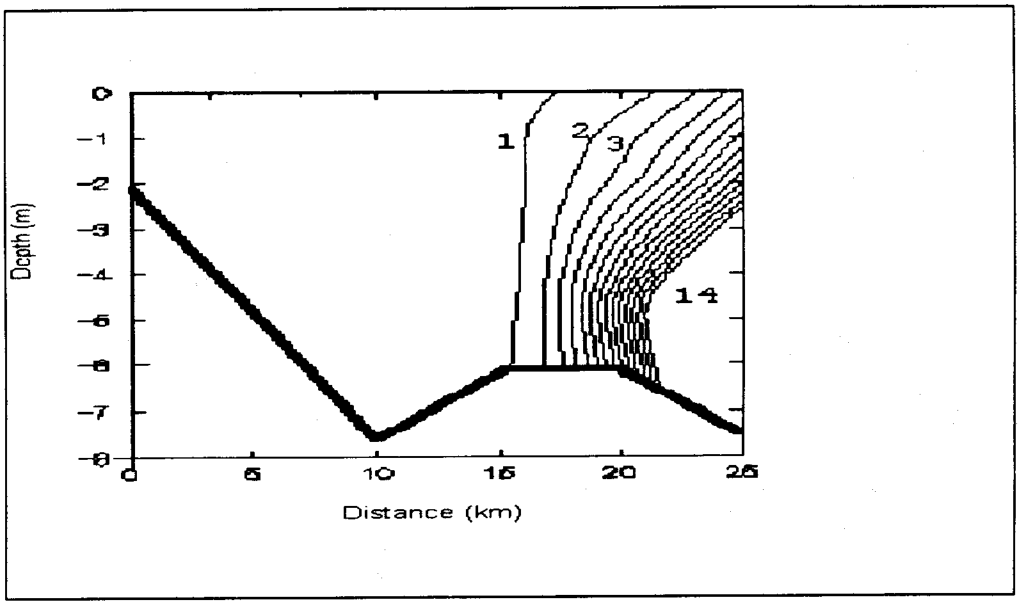

Figure 3. Isohaline curves from CTD data, Miramichi Estuary, 25 May (Q = 318 $m^3 s^{-1}$). The darker line represents the bottom. Distances on the x axis are taken from Red Bank and increase in the downstream direction.

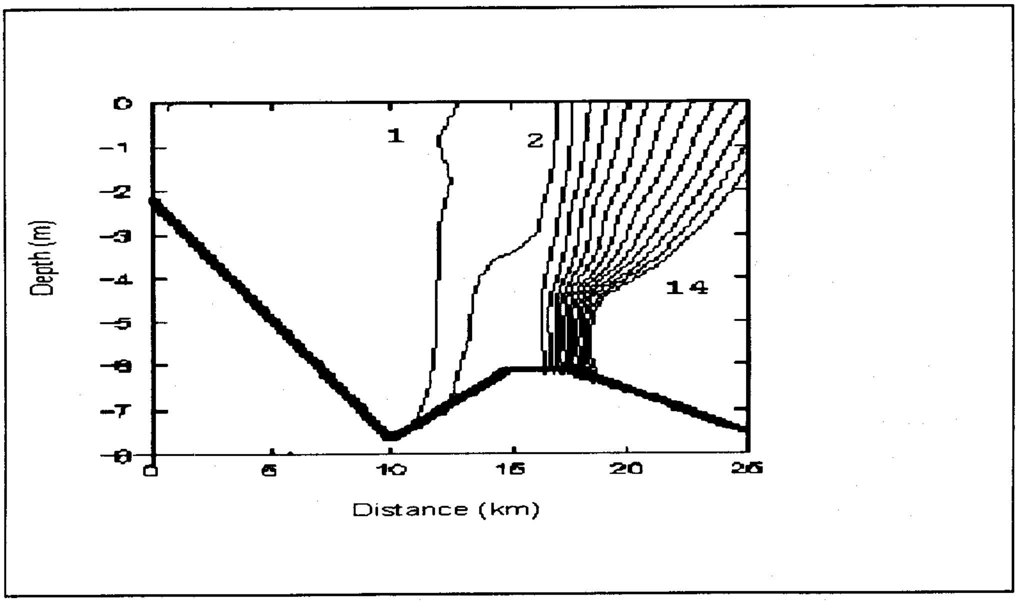

Figure 4. Isohaline curves from CTD data, Miramichi Estuary, 10 June (Q = 150 $m^3 s^{-1}$). The darker line represents the bottom. Distances on the x axis are taken from Red Bank and increase in the downstream direction.

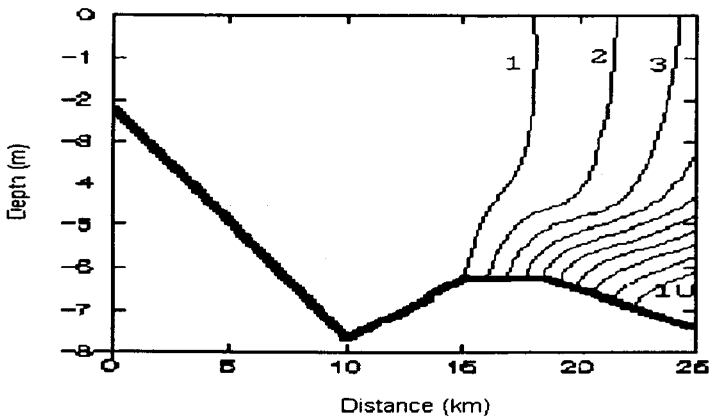

Figure 5. Isohaline curves from CTD data, Miramichi Estuary, 15 July (Q = 304 m^3 s^{-1}). The darker line represents the bottom. Distances on the x axis are taken from Red Bank and increase in the downstream direction.

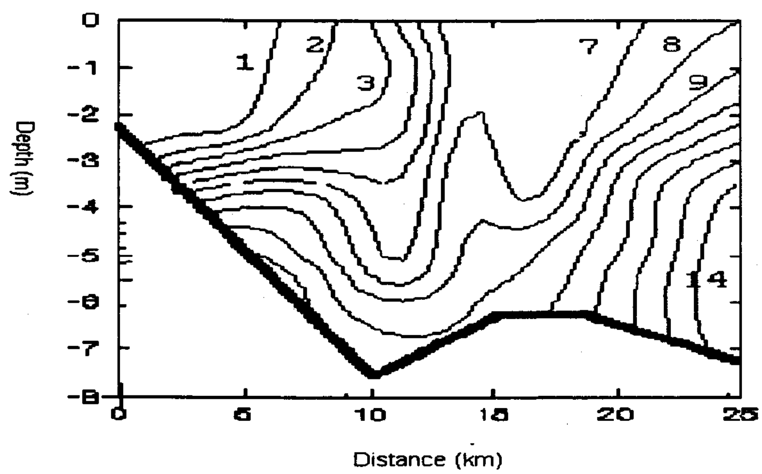

Figure 6. Isohaline curves from CTD data, Miramichi Estuary, 25 August (Q = 166 m^3 s^{-1}). The darker line represents the bottom. Distances on the x axis are taken from Red Bank and increase in the downstream direction.

25, Q = 166 $m^3 s^{-1}$) shows the 2 PSU isohaline extending from km 3 to km 7. Salinity of 15 PSU was measured at km 25 near the bottom. Table 4 gives the position at which the 2, 5, 10, 15 PSU isohalines are near the bottom on the 4 sampling dates:

Table 4. Position (river km) of isohalines on 4 dates and stratification parameter (S_p) at km 18.

Date	2 PSU	5 PSU	10 PSU	15 PSU	S_p at km 18
25 May	12	16	17	20	1.01
10 June	11	17	17.5	--	1.29
15 July	16	18	21	--	1.7
25 Aug	3	4	5	24	0.66

Table 4 also provides values of S_p, the stratification parameter, at km 18, a station included in all 4 surveys. The stratification parameter (S_p) varied between 0.66 and 1.7. The highest value of S_p was obtained on 15 July when the discharge was estimated to be 304 $m^3 s^{-1}$, on the rising limb of the second major summer discharge event (Figure 2). The lowest value of S_p (0.66) was obtained on 25 August (Q = 166 $m^3 s^{-1}$). For a similar flow on 10 June, S_p was nearly twice that value which means that the stratification was much greater.

4. Discussion and Conclusion

4.1 Stratification

Data from the 1992 survey confirmed that the upper Miramichi is of the salt wedge type, as stated in previous research (Vilks and Krauel, 1983; Willis, 1991). S_p values from Table 4 are similar to other salt wedge estuaries such as the Mississippi River with stratification parameter values between 1.2 and 1.9 (Kjerfve and Greer, 1978). The Columbia River, which is also characterized by a strong vertical salinity gradient has S_p values between 0.3 and 3 (Hughes and Rattray, 1980). In spite of varying discharge, the stratification parameter remained higher than that of partially mixed or well mixed coastal plain estuaries such as the Mersey River (U.K.) and the Santee River (U.S.A.; Kjerfve and Greer, 1978).

Power required to mix the discharge was at least two orders of magnitude higher in the upper portion of the Miramichi Estuary than in an estuary with similar shape and depths (Mobile Bay, Alabama) where Shroeder et al. (1990) calculated a W_r of 2 x 10^{-6} W m^{-2} for a river discharge of 500 $m^3 s^{-1}$. For a discharge of 318 $m^3 s^{-1}$, the power required to mix the discharge in the upper Miramichi Estuary was 49.8 10^{-4} W m^{-2}. St-Hilaire et al. (1995) used a two-layered model to calculate the resistance to mixing for the Miramichi Estuary and found values between 26 and 50 x10^{-4} W m^{-2} for the same period in 1991. The values in 1991 and 1992 were of the same order of magnitude, although mean discharge was greater in 1992.

Lafleur et al. (1995) stated that the salt wedge migrated further upstream than Newcastle (km 22) without quantifying this migration. Our data suggest that the tip of the salt wedge, when defined as water with measurable salt content, was consistently located at least 4 km upstream of Newcastle and as far as 17 km upstream of Newcastle between 25 May and 25 August (Figure 6). In fact, salinity of 2 PSU was detected near the bottom at Red Bank on August 13-14 1992 (A. Locke, unpub. data).

Hydrological conditions on the Miramichi showed important fluctuations in 1992. Three major hydrological events occurred between 23 June and 15 August. The first two surveys (25 May and 10 June) were conducted immediately after the spring flood. The 25 August survey was conducted after the largest flood of the summer period. It appears that freshwater discharge was insufficient to push back the salt wedge downstream. This may be explained by the short duration of that hydrological event. However, the energy budget is not complete without tidal and wind inputs. Moreover, the time lag between the discharge measurements between the hydrometric station and the upper reaches of the estuary is unknown but could be significant.

The average discharges for the months of July and August were more than twice the historical means. The baseflow conditions for the 1992 summer were therefore far from low, which implies that the salt wedge could migrate much further upstream in severe low flow conditions.

Both stratification and position of the salt wedge can differ significantly for events with similar discharge. Figures 4 and 6 are a good example. The stratification parameter is significantly smaller on August 25 than on 10 June, when isohalines were denser vertically. The location of the 2 PSU isohaline is 7 km further upstream on 25 August than on 10 June. The energy required to mix the discharge (W_r) was more than 8 times greater on 25 August. Factors such as tidal currents may explain such differences. Data from Lafleur et al. (1995) show that the tidal mixing power calculated at Chatham (km 25) was greater on August 25 than on 10 June. Again, this is an indication that a complete energy budget may be needed to fully understand the processes at work.

Comparison of similar hydrological events enabled us to examine the possible variation in the stratification of the estuary. The magnitude of freshwater discharge alone did not determine stratification, even when that assessment is limited to the upper section of the estuary. Duration of hydrological events, as well as tides and winds must also be included in an initial assessment of the oceanographic conditions.

4.2 Ecological implications

Longitudinal stratification in salinity has important implications for the distribution of fauna within the Miramichi Estuary. Spawning and nursery areas of a number of recreationally and commercially important fish species are defined by salinity. In early June 1992 for example, striped bass (*Morone saxatilis*) spawned immediately upstream of the edge of the salt wedge (river km 11-13) and resulting larvae were also most abundant near the edge of the salt wedge (Robichaud et al., 1996). Larvae of other anadromous fish species that spawn in tidal fresh water were also most abundant near the edge of the salt wedge, including the commercially important rainbow smelt (*Osmerus mordax*), gaspereau (*Alosa pseudoharengus* and *A. aestivalis*), and Atlantic tomcod (*Microgadus tomcod*). Distributions of larval fish may be related to the distributions of their zooplankton prey. Bousfield (1955) and Locke and Courtenay (1995) documented pronounced changes in species composition along the salinity gradient of the Miramichi Estuary. An important prey item found in the guts of larval tomcod, smelt and gaspereau, the calanoid copepod *Eurytemora affinis*, was most common in waters of surface salinities of 1-15 PSU (Locke and Courtenay, 1995).

Vertical stratification also has major implications for the distribution of estuarine fauna. Vertical migration between water strata moving in different directions or at different speeds serves as a mechanism of longitudinal transport for zooplankton organisms in the Miramichi. Species-specific differences in migration patterns of immature barnacles alter their ultimate destination in the estuary and serve to spatially separate congeneric adult populations (Bousfield, 1955). Vertical stratification also affects the distribution of phytoplankton and the nutrients on which they depend. A salt wedge and its associated counter current may act as a nutrient trap and prevent downstream flushing of phytoplankton and nutrients (Correll, 1978). Furthermore, over the short term, a stratified water column may enhance productivity by keeping phytoplankton in the photic zone.

Vertical stratification may also be important in determining the degree to which fauna are exposed to toxic contaminants. The upper Miramichi Estuary between McKay's Cove and Sheldrake Island serves as a nursery area for smelt, gaspereau and tomcod (Locke and Courtenay, 1995). Over the last 50 years, this area has received effluents from two pulp and paper mills, municipalities totalling approximately 50,000 people, commercial and recreational shipping, a wood preservation plant and an oil-burning electrical power station (Courtenay et al., 1995). MacKnight (1990) and Courtenay et al. (1995) document high levels of certain metals and organic contaminants in the sediments of this part of the river, an area where long-term trapping of sediments may occur (Kranck and Milligan, 1989). Vertical stratification of the water column may limit remobilization of sediment-trapped contaminants, whereas the break-down of stratification, or vertical migration of organisms into contaminated strata, may expose the sensitive early life stages of fish and invertebrates to toxic substances. Effects of pollutants on ichthyoplankton of Miramichi Estuary and their prey are unknown, but developmental abnormalities have been recorded in Atlantic fishes elsewhere (e.g., Longwell et al. 1992), and contaminants are present in adult benthic fish and invertebrates in the Miramichi system (MREAC, 1992).

The tendency of the upper Miramichi Estuary to remain stratified despite variations in freshwater discharge implies that flood events have a minor impact on conditions in this nursery habitat for larval fishes. Longitudinal transport in surface and bottom currents should remain relatively unaffected. Salt water intrusion as far upriver as Red Bank ensures that there is a large amount of habitat for brackish water species. In years of lower freshwater discharge, brackish water habitat would presumably extend even further upriver.

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CONTROL OF ECOLOGICAL SITUATION IN THE BASIN OF THE TIDAL POWER STATION (KISLAYA BAY, BARENTS SEA)

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ABSTRACT

In 1993-1995 there were made investigatigations aimed at developing control of ecological situation in the basin of the Kislogubskaya Tydal Power Station (Kislaya Bay, Barents Sea). The quantity influence of abiotic factors on the biota was assessed using the monitor species Littorina littorea, as an example. The following can be recognized as the major ecological factors limiting development of the hydrobionts in the water area studied: water temperature and transparence, concentration of biogenes and oxygen, range and rhythm of tidal oscilations. Principles of the formation of the natural-and-technical system on the basis of the Tidal Power Station (TPS) are proposed.

INTRODUCTION

The world community considers investigations of sea coastal zones as one of the priority problems for the near future. It is impossible to solve many problems of coastal zones without searching for quantity interrelations between biotic and abiotic characteristics of environment. Allowance for and prediction of mutual influences and changes in biota and abiotic factors will allow to minimize the possible negative consequences for the coastal zone. Solution of this problem is of vast practical importance (Zair-Beck I. A., Shilin M. B., 1993; Fedorov M. P., Shilin M. B., Ivashintsov D. A., 1995), and it is ultimately aimed at estimating the consequences of anthropogenic impact on the ecosystems of the sea coast zones. It should be pointed out that the assortment of that impact has extraordinarily widened by the end of the 20th century and includies various kinds of hydraulic construction, sea farming (macriculture), loads due to water transport, mining and transportation of mineral deposits etc. As for the methods of quantity account and measuring anthropogenic pressure, they are only being developed. Besides, while working on large and relatively homogeneous water bodies, a researcher is bound to collect a huge mass of facts to support statistical regularities. The cost of the experimental work of this kind as well as the ensuing laboratory work on the collected material proves to be extremely high. The financing problems significantly hinder development of this sphere of ecology.

Because of this, we consider as promising the idea of searching for quantity relationships based on much more limited material which could be obtained on small isolated water areas where the values of abiotic parameters would vary within a limited area. Fulfilling the requirements are semi-enclosed sea areas such as fjords, gulfs, lagoons of atolls,caldera of extinct volcanoes etc. Though such water bodies could have some specific features, in general they might be considered as natural models of coastal water ecosystems. Carrying out studies in similar model water areas with pronounced environmental gradients allows to receive the hydrobionts' responce to extremily different combinations of media variables. This is the way to prognostic modelling of relations between the biotic and abiotic components of ecosystems.

We choose the Kislaya Bay as a model water body experiensing anthropogenic impact from operation of the TPS and crib farms located there (Fig. 1,2).

DESCRIPTION OF THE STUDY AREA

The floatable construction methods was used in the Kislaya Bay in 1965-1968 for erecting the TPS. The floatable hull of the Station powerhouse measuring 36x18x15 m at 5.000 t displacement was erected in a dock at Kola-Bay, near Murmansk. The hull packed with equipment and set afloat was then tugget to the construction site and sunk with water and sand ballast on the previously prepared underwater foundation in the Kislaya Bay. That "Russian model" of TPS prove efficient for the realization of the tidal power, but ecologically not safe.

The Kislaya Bay, which is about 3.5 km long and less than 2 sq km in area, separated by the dam of the TPS from the Ura Bay of the Barents Sea, is under a complex impact of a number of anthropogenic factors. The quantity estimation of the biota's reaction to these factors allowed to determine the following limiting factors:

- weekened water exchange between the Bay and the Sea, which results in freshening of the TPS basin by the brooks entering it, and degradation of the sea species ;
- technogenic transformation of level oscillations (changes in range, disruption of rhythm);

- predisposition to asphyxiation phenomena due to the excess biogenes coming from aquaculture farms (sea cage farms for humpback salmon Oncorhynchas gorbuscha, trout Salmo trutta and cod Gadus morhua), combined with no vertical mixing at the bottom in summer.

CALENDER OF EXPEDITION AND METHODS USED

The investigations from 1993 till 1995 included studies of hydrological and hydrochemical conditions of hydrobionts' habitat in the coastal zone of the Polar seas: execution of in situ observations for ecoeconomic grounds for organizati on of farms to breed trout, humpback salmon and mussel, including exploitation of the current hydraulic structures (the complex of the TPS Kislaya Bay), selection of monitoring species and development of monitoring techniques for the areas under study; looking into industrial comercially valuable populations of hydrobionts in the coastal and central parts of the Barents Sea.

In the cource of the first stage of the work over the shallow parts of the Barents Sea coastal zone (Kislaya Bay, Ura Bay, Motovskiy Bay) there have been made 51 of observations (Fig. 2) for stations in the Kislaya Bay of determination of sea-water temperature and salinity, hydrochemical elements content (diluted oxygen,phosphates,alkalinity),and associated standart meteorological observations. Simultaneously with the hydrometeorological observatios in the tide periods there have been made 5 hydrobiological surveys with benthos sampling on the grounds including 9 stations (Fig.2., Grounds 1, 2, 4, 5) and 18 stations (Ground 3). From all the grounds samples were taken on three different horizons (upper, lower and middle littoral) (Fig . 4). Besides, 300 sampling frames 33sm x 33sm in size were investigated. In the head, middle and estuarine parts of Kislaya Bay there were made three diving stations with benthos sampling at the depths of up to 10 m (Fig. 2).

The dependence of the biota characteristics (quantity Nand biomass B) on the primary (limiting) factors was studied employing Multiple Regression Analysis (MRA) (Pesenko Yu. A., 1982). When constructing the regression equations we used the quantity of monitoring species for each station N, specemens/sq m, its biomass B, g/m^2, daily duration of the ebb period t, hrs/day, and mean weighted salinity S, %. The calculation of self organizing models of the monitoring species distribution was carriedout by the MGAA (Method of Group Argument Account) (Pogrebov V. B., 1990) with reducing the coefficients of the full description to zero. Unlike the classical regressive analysis which accurately describes the relation between variables in the studied interval, the MGAA is mainly directed to finding the most general and stable relationships. The latter determines its high cognitive and prognostic possibibilities.

As a supporting function we used a square polynomial with the argument covariation of the form

(1) $$A= A_0 + A_1 S + A_2 S^2 + A_3 t + A_4 S t,$$

where A is the index of the species abundance, quantity or biomass; S is salinity, %; t is duration of the ebb period, hrs per day.

The regression coefficients were estimated by the method of least squares.

RESULTS

The analysis of the results obtained testifies to presence in the basin of the Kislaya Bay of two near-surface water masses (north-eastern and south-western ones) separated by a well-defined frontal interface represented in all measured hydrological characteristics (temperature, salinity and density). The so-called south-western water mass mainly forms under the influence of the fresh-water runoff of the large brooks. The so-called north-eastern water mass mainly forms due to the water exchange through the dam of the TPS, and by all measured characteristics it can be attributed to the water of the ajacent Ura Bay (Fig. 3).

A scheme of the littoral benthos organismes distribution has been drawn (Fig. 4).

The composition of the coastal biological communities in the Bay is very depleted. Macrophyte algae form small colonies, but not a continuous border as in the neighbouring bays. Correspondingly, there are no species accompanying brown algae which usually find food and shelter in their thickets.

The data of studied values of studied values of the quantity of Littorina snails averaged for the whole Bay show that their quantity N(Fig. 5, a) and biomass (Fig. 5, b) almost equally change, depending on salinity changes and duration of ebb. The quantity of snails decreases as duration of the ebb period increases. The dependence on changes in salinity is of more complex character.The minimum quantity is recorded with salinity of 30.3-30.7% (Fig. 5, a, b). With the salinity increasing or decreasing the values of N and B start to increase. Evidently the obtained parameters under study differ from classical bellshaped curves of distribution of biological variables over environmental gradients.

It is not excluded that the significant deviations from the bellshaped distribution is caused by strong influence of unaccounted factors (turbulence, oxygen regime, etc.).

To control the ecological situation in the water body under study we think it necessary to do the following:

1. To increase water exchange in the TPS basin through non-stop station operation combined with no-load water passage through the upper and lower discharge sluices.
2. To pump air or aerated surface water into the oxygen-free depressions (artifical circulation) through perforated pipes.
3. To increase the number of species in the biota by creating of artifical biotopes (artifical kelp in the littoral zone) and introduce commercially valuable species (king crab Paralithodes camtschatica).
4. To develop poly-mariculture, in particular sea urchins Strongylocentrotus droebachiensis in cribs which eat algal film on nets.
5. To improve quality of pelletized feed.

Realization of the proposed decisions should facilitate formation of the natural-and-technical system on the basis of theTPS where the coastal zone management principles could be perfected.

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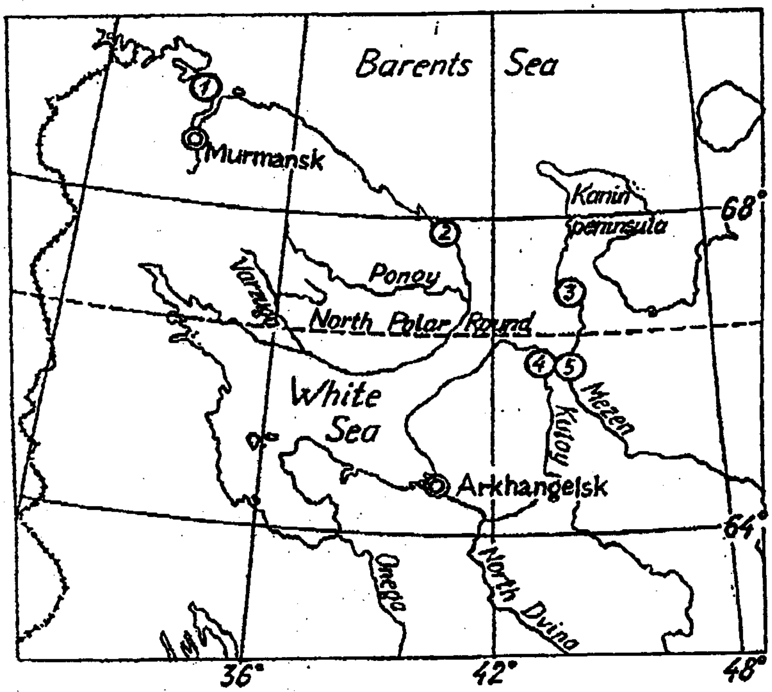

Figure 1: Scheme of location of the tidal power stations

① - TPS Kislogubskaya (functioning)
② - TPS Lumbovskaya (project)
③ - TPS Belomorskaya (project)
④ - TPS Kuloyskaya (project)
⑤ - TPS Mezenskaya (project)

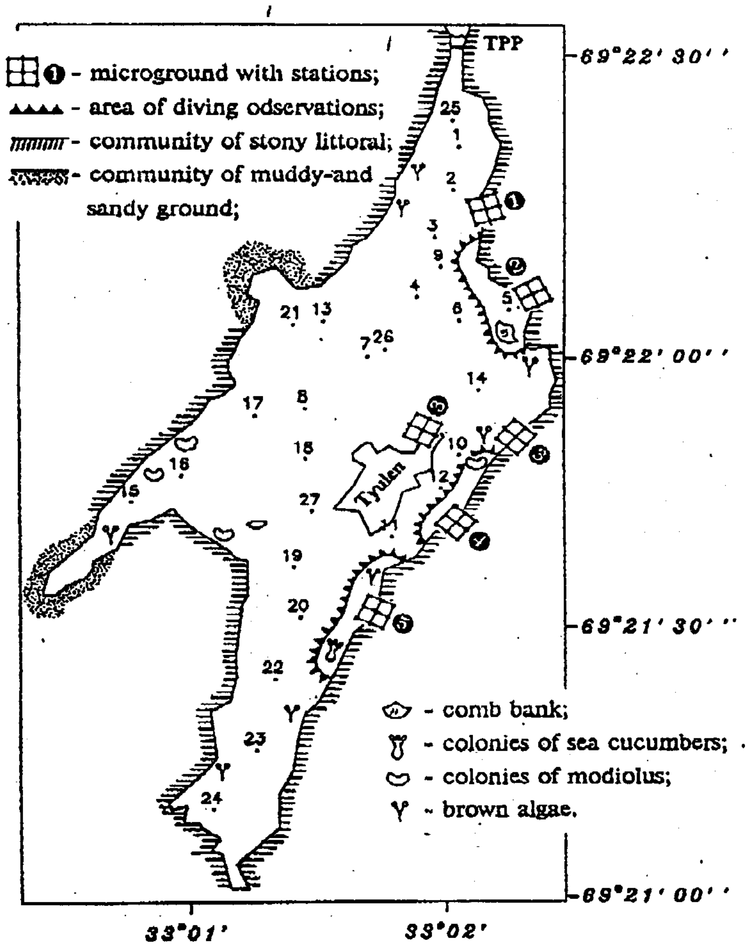

Figure 2: Scheme of location of hydrobiological stations in the Kislaya Bay
(1...27 - number of the stations).

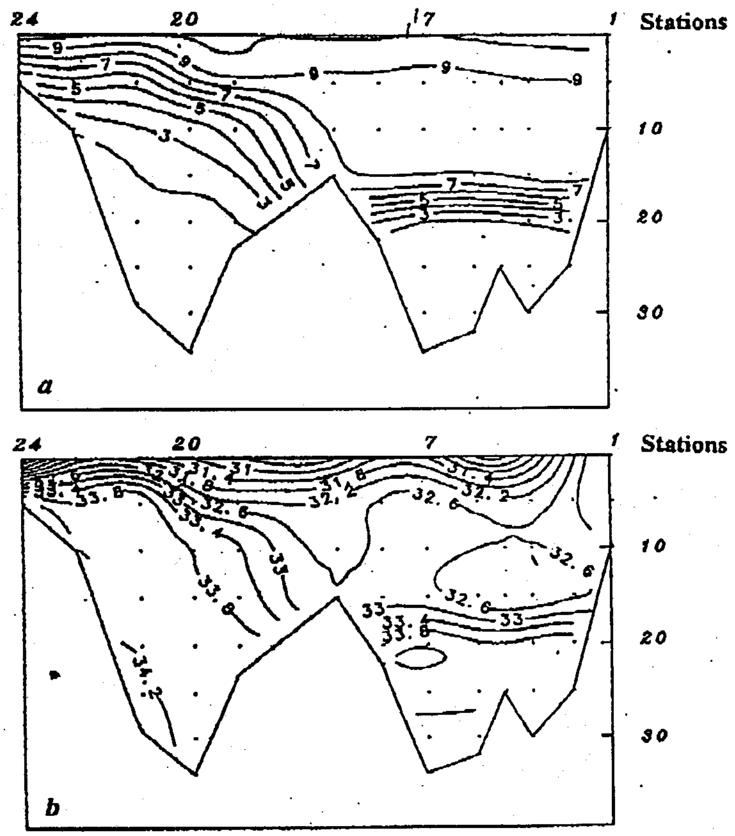

Figure 3: Temperature, °C, (*a*) and salinity, ‰, (*b*) across the section along the Kislaya Bay.

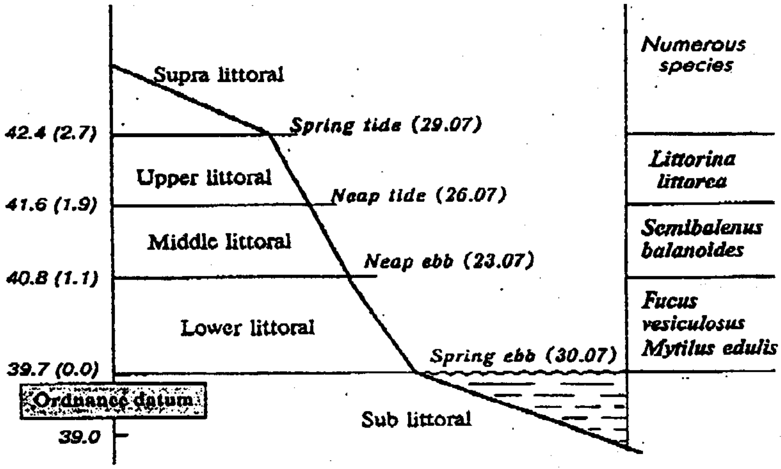

Figure 4: Littoral mapping in the Kislaya Bay.

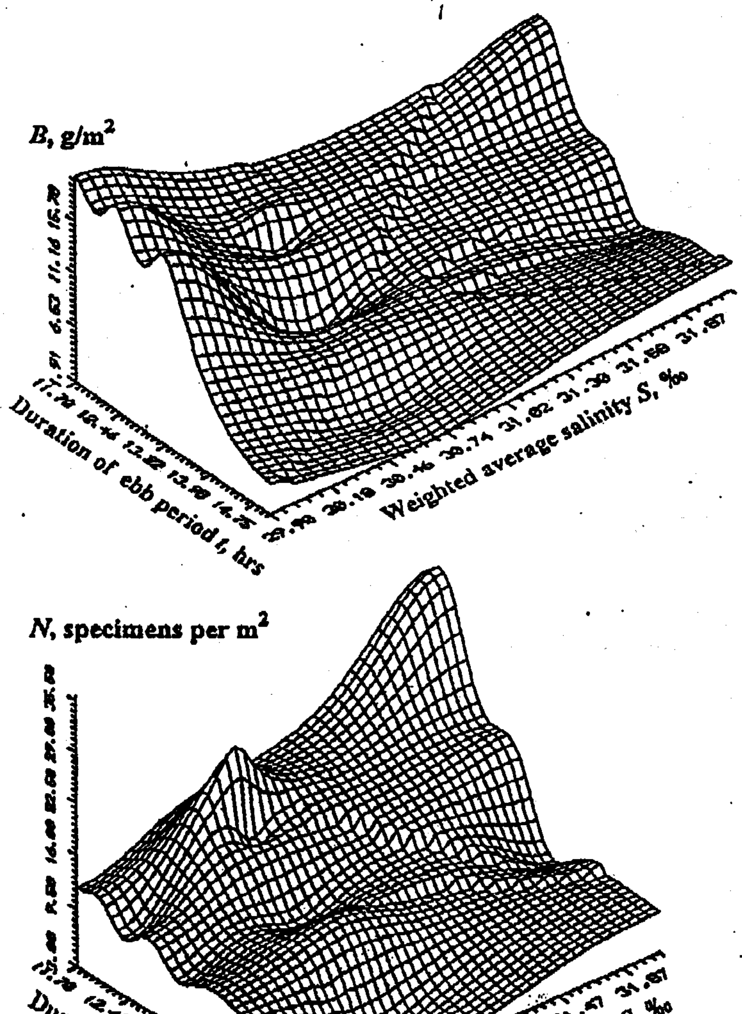

Figure 5: Biomass *B*, g/m², and quantity *N*, specimens per m², of Littorina depending on the weighted average salinity and duration of ebb period.

TWO LAYER FINITE ELEMENT NUMERICAL MODEL FOR SIMULATING SALMON RIVER ESTUARIES

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ABSTRACT

Many salmon river estuaries on the Saint-Lawrence maritime estuary are, during the upstream migration of the salmon, under low discharge conditions and consequently present strong stratification. Two hydrodynamic parameters have a significant importance for the use of the estuary by the salmon : the velocity distribution and the layout of fresh and salt water masses.

The use of a numerical model must allow the verification of hypothetical physical modifications of the estuary on the currents and the salt intrusion. The proposed model has been developed in a way to well represent the typical characteristic of the estuary to be studied, i.e., a strong stratification, weak vertical mixing and large tidal flats. The approach is based on the superposition of two two-dimensional hydrodynamic models, one for the fresh water and one for the salt water. For the tidal flats, the concept of negligible height of water has been used. This implies to take in account the flow regime in the friction term. The finite element method has been used to obtain the numerical formulation of the model.

KEY-WORDS: Estuary / Hydrodynamic Model / Salt Water / Density / Finite Element / Tide / Two Layer Model.

INTRODUCTION

In many small estuaries, conditions of strong stratification are often observed. In few of those cases, the transition between fresh and salt water is so small that the interface between the two masses of water can be seen by eyes because of the sharp change of refraction index. Some measurements done since two years in Matane river estuary show obviously a such condition.

In an other hand, it seems that the behaviour of migratory fish is influenced by those specific salt and clear water distribution in a sense that they could use the estuary as an area of transition and rest before going farther upstream for the spawn. As those conditions can vary during the tide cycle and depending the water discharge from upstream, it could be interesting to know in detail the hydrodynamic behaviour of the estuary. A numerical model can be considered for that purpose but two problems arise.

First is the need of a three dimensional model. Indeed, a vertically integrated shallow water equation based model could not represents correctly the hydrodynamic process because nor physical nor water quality stratification can be taken in account. In an other point of view, full three dimensional model are heavy to run, expensive in both human and computer time and their calibrations is not easy.

The second problem comes from the morphology of the estuary itself in a sense that low tide flats play an important role in the current pattern and must be considered if we intend to do a realistic simulation. If it is not so difficult to include tidal flats in a two dimensional model, the same is much more complicated in a three dimensional model.

In order to keep the advantage and the simplicity of a two dimensional model and to benefit the realism a three dimensional model, we decided to develop and test a vertically integrated two layer model, one for the clear water over the second one for the salt water, considering that a such model could only be applied in strongly stratified estuaries.

MODEL DEVELOPMENT

We describe here the essentials of model development with a special care on the difference of density between clear and salt water.

Governing equations

We start from the general equations written from the principle of momentum and mass conservation (Liggett, 1994):

Momentum:

(1) $$\frac{\partial u}{\partial t}+\frac{\partial uu}{\partial x}+\frac{\partial uv}{\partial y}+\frac{\partial uw}{\partial z}+\frac{1}{\rho}\frac{\partial p}{\partial x}=X$$

(2) $$\frac{\partial v}{\partial t}+\frac{\partial uv}{\partial x}+\frac{\partial vv}{\partial y}+\frac{\partial vw}{\partial z}+\frac{1}{\rho}\frac{\partial p}{\partial y}=Y$$

(3) $$\frac{\partial w}{\partial t}+\frac{\partial uw}{\partial x}+\frac{\partial vw}{\partial y}+\frac{\partial ww}{\partial z}+\frac{1}{\rho}\frac{\partial p}{\partial z}=Z$$

Mass:

(4) $$\frac{\partial u}{\partial x}+\frac{\partial v}{\partial y}+\frac{\partial w}{\partial z}=0$$

In which the assumption of a non viscous flow is taken and where:

(5) $X=fv+K_x, Y=-fu+K_y$ and $Z=-g$

with the Coriolis' coefficient $f=2\omega\sin\phi$, with ω, the angular speed of the rotation of the earth [rad/s], ϕ, the mean latitude [degree], K_x and K_y, tidal forces in x and y directions.

Then we add an assumption concerning the vertical distribution of pressure. Indeed, if slow vertical motion applies, the vertical acceleration of a mass of water can be considered small in respect of the gravitational acceleration. Then, the third momentum equation (eq. 3) reduces to:

(6) $$\frac{\partial p}{\partial z}+\rho g=0$$

Which defines the hydrostatic pressure assumption.

The above system of partial differential equations describe the instantaneous motion of water which includes the fluctuation of velocity and pressure. It is not practical to keep those instantaneous parameters because the need of a very small time scale. If we agree that some mean parameters could be used, the instantaneous parameters are replaced by the sum of a mean value and a fluctuation which is caused by turbulence. Then the mean value of equations themselves can be taken over a period of time. As the mean of a fluctuation over that period can be considered as zero, the major part of fluctuation terms vanishes from the equations except for those which content the product of two fluctuations. In that case, those products are not zero and we have a closure problem.

In order to close the system, we must adopt a turbulence model. Here, we will use the simplest one, the eddy viscosity concept by adding to the equations some viscous terms similar to the Newton's law of viscous shear stress. The two first momentum equations (1) and (2) are then rewritten according the assumption that now u, v, and p are mean values over a period of time:

(7) $$\frac{\partial u}{\partial t}+\frac{\partial uu}{\partial x}+\frac{\partial uv}{\partial y}+\frac{\partial uw}{\partial z}-fv+\frac{1}{\rho}\frac{\partial p}{\partial x}-\frac{1}{\rho}\left(\frac{\partial \tau_{xx}}{\partial x}+\frac{\partial \tau_{xy}}{\partial y}+\frac{\partial \tau_{xz}}{\partial z}\right)=0$$

(8) $$\frac{\partial v}{\partial t}+\frac{\partial uv}{\partial x}+\frac{\partial vv}{\partial y}+\frac{\partial vw}{\partial z}+fu+\frac{1}{\rho}\frac{\partial p}{\partial y}-\frac{1}{\rho}\left(\frac{\partial \tau_{yx}}{\partial x}+\frac{\partial \tau_{yy}}{\partial y}+\frac{\partial \tau_{yz}}{\partial z}\right)=0$$

These two equations with the simplified third momentum equation (6) and the continuity equation (3) constitute the base of the model to be develop.

Vertical integration

Principle

The formulation of the two-layer model is based on vertical integration of the system of equations in to steps, from -h to η_i and from η_i to η_s in order consider only the vertical mean values of physical parameters in each layer (fig. 1).

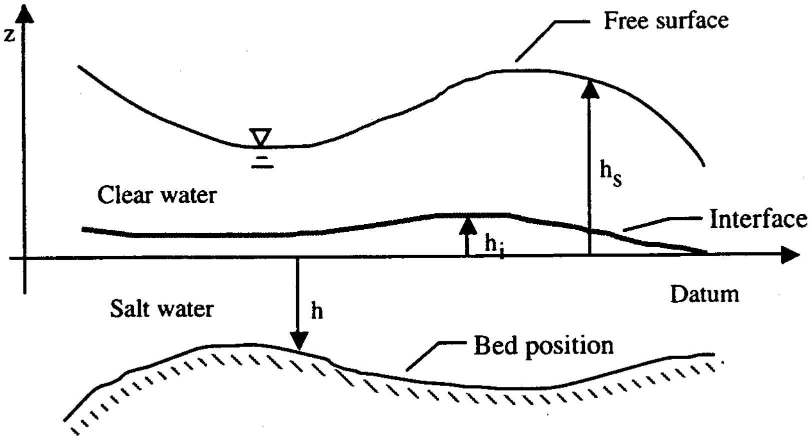

Fig 1 - Definition of boundaries in vertical direction.

The manipulation of equations is done with the Leibnitz rule:

(9) $$\frac{\partial \int_{-h}^{\eta} f(x,y,z,t)\,dz}{\partial t} = \int_{-h}^{\eta} \frac{\partial f(x,y,z,t)}{\partial t}\,dz + f(x,y,\eta,t)\frac{\partial \eta}{\partial t} - f(x,y,-h,t)\frac{\partial(-h)}{\partial t}$$

Once the vertical boundary conditions known, the definition of mean value parameters are introduced. By example, in the case of x component of water velocity, we have:

(10) $$U = \frac{1}{h}\int_{-h}^{\eta} u(z)\,dz$$

Vertical boundary conditions

As the free surface, the internal interface and the bottom surface can be assimilate to stream lines, we can state that a particle of water cannot cross them. This is rigorously true for the surface and the bottom but this is an assumption for simplicity in the case of the interface. Those surfaces are defined by:

(11) $$z_s = \eta_s(x,y,t),\ z_i = \eta_i(x,y,t) \text{ and } z_h = -h(x,y,t)$$

then the vertical velocity of any points of those surfaces can be obtained by total time derivative of respective z positions:

(12) $$\left.\frac{dz}{dt}\right|_{z=\eta_s} = w(\eta_s) = \frac{\partial \eta_s}{\partial t} + u(\eta_s)\frac{\partial \eta_s}{\partial x} + v(\eta_s)\frac{\partial \eta_s}{\partial y}$$

(13) $$\left.\frac{dz}{dt}\right|_{z=\eta_i} = w(\eta_i) = \frac{\partial \eta_i}{\partial t} + u(\eta_i)\frac{\partial \eta_i}{\partial x} + v(\eta_i)\frac{\partial \eta_i}{\partial y}$$

(14) $$\left.\frac{dz}{dt}\right|_{z=-h} = w(-h) = -u(-h)\frac{\partial h}{\partial x} - v(-h)\frac{\partial h}{\partial y}$$

Those vertical components of velocity will be used after application of Leibnitz's rule, having for effect to eliminate all explicit reference to vertical velocity in the resulting formulation.

The application of the Leibnitz's rule to the equation 4, 6, 7 and 8 and the introduction of the expression of vertical velocities at the boundaries deal to the following set of equation which will be used in the numerical model :

Pressure distribution :

(15) $$\begin{aligned} &\text{if : } \eta_i \le z \le \eta_s \\ &\quad p(z) = \rho_0 g(\eta_s - z) \\ &\text{if : } -h \le z \le \eta_i \\ &\quad p(z) = g(\rho_0 \eta_s + (\rho_s - \rho_0)\eta_i - \rho_s z) \end{aligned}$$

Momentum equations :

(16) $$\frac{\partial U_1}{\partial t} + U_1\frac{\partial U_1}{\partial x} + V_1\frac{\partial U_1}{\partial y} - f V_1 + g\frac{\partial \eta_s}{\partial x} - \frac{1}{\rho_0 h_1}\left[\tau_x^s - \tau_x^i\right] - \frac{\mu}{\rho_0}\nabla^2 U_1 = 0$$

(17) $$\begin{aligned} &\frac{\partial U_2}{\partial t} + U_2\frac{\partial U_2}{\partial x} + V_2\frac{\partial U_2}{\partial y} - f V_2 + g\left[\left(1 - \frac{\rho_0}{\rho_s}\right)\frac{\partial \eta_i}{\partial x} + \frac{\rho_0}{\rho_s}\frac{\partial \eta_s}{\partial x}\right] \\ &- \frac{1}{\rho_s h_2}\left[\tau_x^s - \tau_x^b\right] - \frac{\mu}{\rho_s}\nabla^2 U_2 = 0 \end{aligned}$$

(18) $$\frac{\partial V_1}{\partial t} + U_1\frac{\partial V_1}{\partial x} + V_1\frac{\partial V_1}{\partial y} + f U_1 + g\frac{\partial \eta_s}{\partial y} - \frac{1}{\rho_0 h_1}\left[\tau_y^s - \tau_y^i\right] - \frac{\mu}{\rho_0}\nabla^2 V_1 = 0$$

(19) $$\begin{aligned} &\frac{\partial V_2}{\partial t} + U_2\frac{\partial V_2}{\partial x} + V_2\frac{\partial V_2}{\partial y} + f U_2 + g\left[\left(1 - \frac{\rho_0}{\rho_s}\right)\frac{\partial \eta_i}{\partial y} + \frac{\rho_0}{\rho_s}\frac{\partial \eta_s}{\partial y}\right] \\ &- \frac{1}{\rho_s h_2}\left[\tau_y^s - \tau_y^b\right] - \frac{\mu}{\rho_s}\nabla^2 V_2 = 0 \end{aligned}$$

Continuity equations :

(20) $$\frac{\partial \eta_s}{\partial t}+\frac{\partial h_1 U_1}{\partial x}+\frac{\partial h_1 V_1}{\partial y}+\frac{\partial h_2 U_2}{\partial x}+\frac{\partial h_2 V_2}{\partial y}=0$$

(21) $$\frac{\partial \eta_i}{\partial t}+\frac{\partial h_2 U_2}{\partial x}+\frac{\partial h_2 V_2}{\partial y}=0$$

Friction terms

The stresses applied on the fluid at the surface and the bed are to be specified. The effect of bed roughness cant be taken in account in the salt layer using the Chézy formula :

(22) $$\tau_x^b=\rho g\frac{U_2\sqrt{U_2^{\,2}+V_2^{\,2}}}{C_c^2} \text{ et } \tau_y^b=\rho g\frac{V_2\sqrt{U_2^{\,2}+V_2^{\,2}}}{C_c^2}$$

C_c is the Chézy coefficient.

At the interface, from the Newton's law of viscosity and the Prandtl's mixing length concept, the following relationships are developed :

(23) $$\tau_x^i=\frac{\rho\lambda}{\Delta z^2}\Delta U\sqrt{\Delta U^2+\Delta V^2} \text{ and } \tau_y^i=\frac{\rho\lambda}{\Delta z^2}\Delta V\sqrt{\Delta U^2+\Delta V^2}$$

with $\Delta z=\frac{h_1+h_2}{2}$, $\Delta U=U_1-U_2$, $\Delta V=V_1-V_2$ and λ, a friction coefficient at the interface related to the square of the mixing length.

If necessary, at the free surface, the wind stresses can be evaluated by :

(24) $$\tau_x^s=C_x\rho_a W_x|W| \text{ and } \tau_y^s=C_x\rho_a W_y|W|$$

where C_x the drag coefficient at the surface, ρ_a, the density of air and W_x et W_y are velocity component of wind speed W.

Finite element formulation

The above set of equations is first introduced in an integral form according the weighted residual method. The weight function is chosen in the same functional space than the unknown functions which defines the Galerkin method. This formulation is then modified by integration by part of the second order terms to obtain a weak form allowing the use of lower degree interpolation polynomials and to make explicit some boundary terms to impose flux boundary conditions (Dhatt and Touzot, 1981).

The interpolation functions are chosen as quadratic two-dimensional polynomial function for the velocity components and linear one for the interpolation of the positions of free surface and internal interface.

These polynomial function are defined on a six node triangle (fig. 2) where surface positions appear only at vertex nodes.

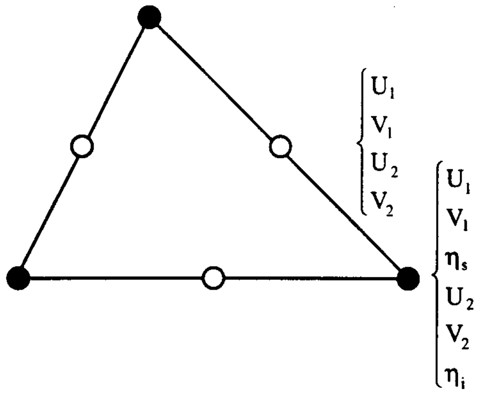

Fig. 2 - Element definition

The introduction of the polynomial interpolation function element by element in the elementary integral forms results in a discrete form as a set of non linear algebraic equations. This non linear time evolution problem is solved by the use of implicit Euler scheme to approximate the time derivative and by a Newton-Raphson procedure to process non linearity :

$$\text{(25)} \qquad \left[\mathbf{M} + \Delta t \mathbf{K}_T(\mathbf{U}_{t+\Delta t})\right]^{i-1}\{\Delta \mathbf{U}\}^i = \left\{[\mathbf{M}]\{\mathbf{U}_{\Delta t} - \mathbf{U}_{t+\Delta t}\} - \Delta t\left\{[\mathbf{K}(\mathbf{U}_{t+\Delta t})]\{\mathbf{U}_{t+\Delta t}\} + \{\mathbf{F}\}\right\}\right\}^{i-1}$$

where **M** is the mass matrix, $\mathbf{K_T}$, the tangent matrix and **F** the solicitation vector containing the effects of boundary conditions.

MODEL APPLICATIONS

Validation

The first problem with numerical modelling is to validate the formulation of the model and its translation in computer code. At the beginning, in order of increasing difficulties, some basics tests are performed. Those tests range from still basin, steady uniform flow to internal wave propagation. We will not describe all those tests here, we will just mention that they have been very useful to correct the coding of the model and that they give very satisfactory results. Among those tests, one is particularly interesting in a sense that it demonstrates the sensibility of one crucial but almost unknown parameter, the λ coefficient at the interface.

The test case is performed in a rectangular basin (fig 3.) where the initial position of the salt water/clear water interface has been set in an inclined position at 1,667%. Then the system starts in a free oscillation, which is more or less damped.

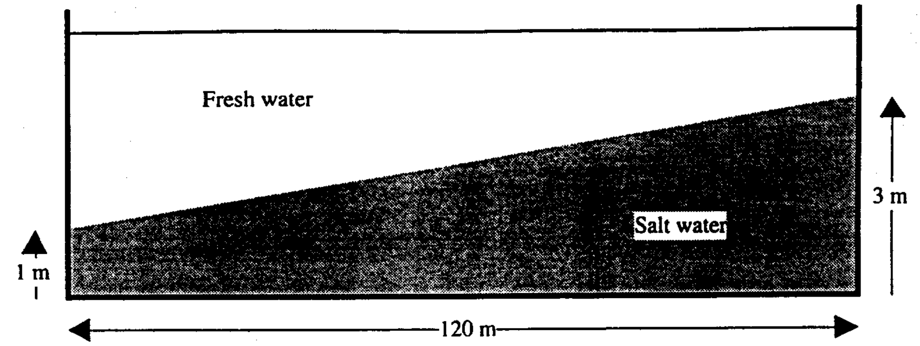

Fig. 3 - Test basin for damped free oscillation.

For this case, the bottom friction remains constant with Chézy coefficient at 1000 $m^{1/2}/s$ for a very low bed friction. The salt water density has been set at a constant 1035 kg/m^3. The interfacial coefficient has been tested for a range of values from 0,1 to 10 m^2. The greatest value is too strong, the interface takes a very long time to return to its equilibrium position and there is no perceptible oscillation.

The difference of behaviour between the choice of 0,1 and 1,0 for the interfacial coefficient is very significant. The time evolution position of the interface has been plotted for a point located at the left end of the basin (fig 4). The damping is much more important for the biggest coefficient and it is obvious that the propagation of the amplitude of an internal wave is strongly affected by the choice of this coefficient.

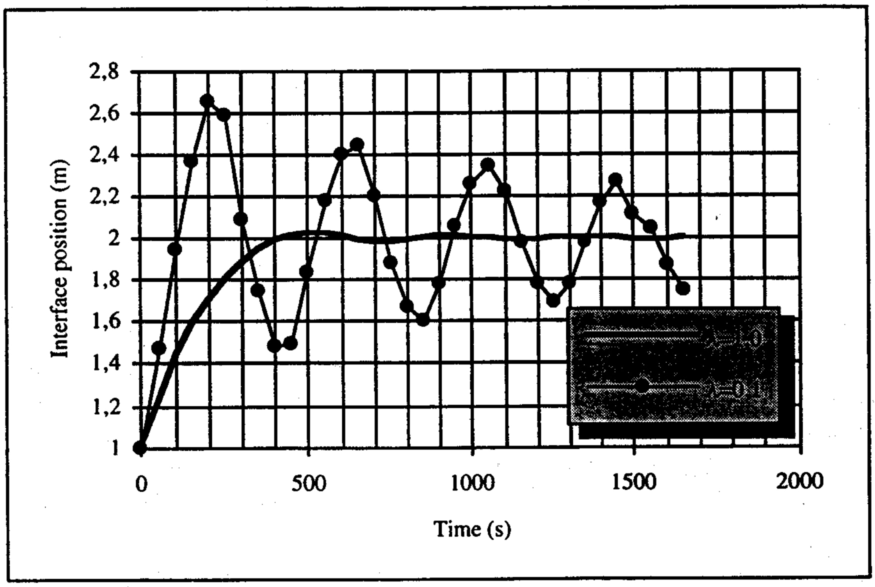

Fig 4 - Effect of interfacial friction factor of internal wave damping.

The problem is that the λ coefficient cannot be determined easily because it contains information which is related to turbulence near the interface. The approach based on the mixing length concept comes from physical considerations about the velocity profile in a steady uniform flow which is not the case here. The best way to calibrate this coefficient is to compare the behaviour of the model to measurement of the evolution of the interface in a real case to be studied.

Tidal flats

Depending the topography of an estuary, uncovered surfaces, at low tide, can become a significant part of the total surface of the estuary. In those cases, the boundary of the wetted part of the estuary must move during the tidal cycle. This implies that, ideally, the finite element grid should follows the boundary between water and bank and adapts its surface to the wetted part of the estuary at any time. This solution which is the best in term of mass and momentum conservation needs complex algorithms and a very large amount of computing time (Holz and Nitsche, 1980). In practical model, alternative procedure must be considered to obtained a more economical model.

If the computing grid is kept immobile, two approaches can be used to approximate ebbing and flowing on tidal flats. The first one consists to let the water level free to go under the bed and to tolerate negative water heights. The second one is based on the control of water height to keep a very thin sheet of water over uncovered flats.

The first approach is easy to implement and has the advantage to keep small hydraulic gradient at water line near the bank where the free surface is nearly horizontal. In an other hand, this approximation present a major disadvantage because it will not guarantee the mass conservation. More, in the case of a multi-layer model, if the salt layer vanishes in some part of the domain, we cannot consider negative salt layer thickness.

The second approach necessitate more algorithmic care. The control of water height is done by temporarily impose some water level or interface position at a very small height over the bed position. This can easily done without modifying the structure of the matrix of the system of equations by applying a huge coefficient at the diagonal term and correcting the corresponding solicitation term. The major drawback of this method is the apparition of steep hydraulic gradient over bank slopes. This can be controlled if those situations are considered as runoff flow (Zhang and Cundy, 1989). The thin sheet flow is generally laminar and the friction term must be adapted to the flow condition using a White-Colebrook formula.

CONCLUSION

Considering the special cases of strongly stratified estuaries, a two layer finite element model has been developed. This model combines the relative simplicity of two dimensional models and can approximate complex three dimensional behaviour within the limits defined by the assumption adopted for its formulation. It has been demonstrated that the choice of a turbulent model near the interface of the two layer is determinant to control the propagation of internal waves and that a calibration must be done to obtain realistic results. When the model is applied to inter tidal zones, the multi-layer concept forces the control of layer thickness up to a very small value.

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Land uses impacts

Impacts de l'utilisation du territoire

HYDRO-AGRICULTURAL MANAGEMENT IN THE RHONE RIVER DELTA, FRANCE : CONSEQUENCES ON DISSOLVED AND SOLID FLUXES, POTENTIAL IMPACT ON FISH POPULATIONS

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ABSTRACT

With the development of rice cultivation, the major part of the Isle of Camargue, the central zone of the Rhône river delta, was polderised. The central lagoon system of the Camargue (110 km^2, 64 km^2 for the Vaccarès lagoon itself) represent the main part of the " Réserve Nationale de Camargue ". Without human intervention, this wetland is hydraulically isolated from the Rhône river and the mediterranean sea. It receipt drainage water from its remaining catchment area, due to precipitations and flooded rice fields.

The hydrological study concerned the Fumemorte catchment (65 km^2, with 20 km^2 of rice fields) which was previously ungauged. Intensive hydrological studies were conducted during the years 1993 and 1994, coupled with a continuous monitoring at the outlet of the catchment, of nitrogen (N), phosphorus (P) and total and organic suspended materials (TSM and OSM) . The land use was obtained from satellite images processing with ground truth, and included in a georeferenced data base. The knowledge of spatial and temporal variability of irrigation volumes, combined with information on fertilisation practices, estimates of Rhône water quality and atmospheric inputs, was used to quantify the total loads entering the catchment. Mean annual fluxes of 402 tons of nitrogen (N) and 153 tons of phosphorus (P) entered the catchment. With a scenario of maximum transfer, only 6 % of introduced N and 4 % of P were exported from the Fumemorte catchment to the Vaccarès lagoon. Organic matter exported by the drainage network represents 5 to 6 times the quantity introduced with the Rhône water. Due to sedimentation in the rice fields and the drainage network, exportation of sediments is inferior to importation with irrigation water. Fish population dynamics in the Camargue has been studied in different compartments, including the drainage network of the Fumemorte drainage basin. On the base of previous and ongoing studies, potential impact of hydraulic management on fish populations is discussed.

KEY WORDS : Camargue / Rhône delta / Rice cultivation / Phosphorus / Nitrogen / Irrigation / Drainage / Wetland / Fish population / Fertiliser.

INTRODUCTION

A considerable increase of rice crop production was observed in the Rhône river delta, or Camargue, after World War II. From 11 km^2 in 1946, the area devoted to rice cultivation reached 160 km^2 in the 60's. Rice cultivation radically changed the hydrological regime of the Great Camargue, the central part of the delta, between the two branches of the Rhône. Great quantities of irrigation water from the Rhône river are introduced in the rice fields from may to september. The drainage volume could not be supported by the system of lagoons without risks of inundations for the surroundings area when heavy rains appeared in the autumn. As a consequence, on the base of the already existing drainage basins and networks, the major part of the agricultural catchment was equiped in the mid 50's with pumping stations for the drainage. So, since this date, 55 % of the potential agricultural catchment area of the lagoons is polderized (figure 1). The Fumemorte catchment (65 km^2) represents 55 % of the area actually drained to the lagoons, although some hydraulic transfers may occur from the normally polderized basins to the Vaccarès during the crop season (Heurteaux, 1994 ; Chauvelon, 1996). It was attempted to estimate the hydrological balance of the system (Heurteaux, 1994), with relatively sparse data concerning hydraulic management of the system. Research were conducted at the field scale to quantify processes affecting the nitrogen cycle in rice fields (Minzoni *et al.*, 1988; Golterman *et al.*, 1988).

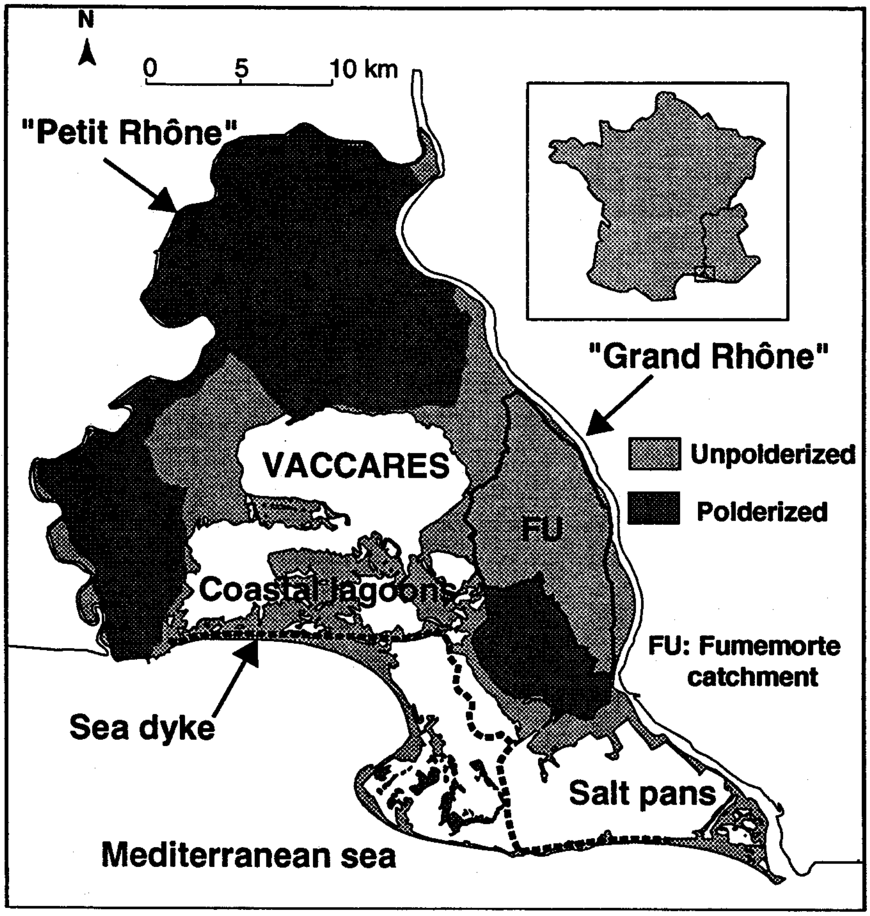

Figure 1: The Great Camargue : lagoon system and drainage basins (After Chauvelon, 1996).

Others works tackled quality of irrigation and drainage waters (Godin, 1990; De Groot, 1992). Nevertheless, due to discrete sampling and the absence of drainage discharge data, fluxes entering the Vaccarès lagoon system could not be estimated accurately.

The objectives of the work presented here were not to analyse precisely the bio-geochemical processes affecting the transfers of nutrients and sediments, but to estimate the major inputs and outputs of the catchment. Setting by this way, the bases for an estimation of fluxes entering the lagoons and their implications for primary production and eutrophication (Chauvelon, 1996).

It is conceptually sound to think that the hydraulic artificiality of the system has an impact on the status of the fish populations and their migrations within the hydrosystem. Studies were conducted, focusing on the fish of the Vaccarès (Bardin, 1994; Leroy, 1994), the freshwater irrigation and drainage system (Jeudy, 1995), and temporary marshes connected to it in the Fumemorte catchment (Crivelli, 1981; Pont *et al.*, 1991; Rosecchi and Crivelli, 1995). A review of major potential impacts due to hydro-agricultural practices and works on fish populations status and migrations will be discussed.

STUDY AREA

The Rhône river delta or Camargue, is located in the french mediterranean coast, at 80 km West of Marseille. It is characterized by a globally warm and dry climate. Winds are frequent and violent in all directions, but with a net predominancy of N-NW winds (called "mistral"). Potential evaporation and evapotranspiration are intense (1,300 mm yr^{-1}) due to high summer temperature, solar radiation and windy conditions. Precipitations are not abundant (590 mm yr^{-1} on average on a 50 year period of observations), and characterized by a rather important temporal variability (coefficient of variation of annual precipitation depth is estimated to 29 %).

In the Camargue, rice is cultivated on leveled fields of 1 to 3 ha, delimited by bunds. The fields are flooded during the crop period (from mid april or may to september) and drained dy ditches. Cut-offs are made in the bunds, in order to manage some rough weirs (usually with plastic bags filled up with ground). These weirs are used to control the water levels according to different crop stages and to allow the dewatering of the rice fields. The mean drainage density in the agricultural zone is 10 km km^{-2} (Chauvelon, 1996), the drainage is realized by gravity flow in a meshed network in the case of the unpolderized drainage basins. In 1994, rice fields covered 145 km^2 in the Great Camargue, from this total, 40 km^2 were on the actual drainage basin of the lagoons, the half being on the Fumemorte catchment.

Since the completion of its endykement, in 1869, the Great Camargue is hydraulically isolated from the Rhône river and the sea without human intervention. At the S-E of the Great Camargue, salines were also isolated from the lagoons system, in order to control water salinity (figure 1). The only gravity input of Rhône water in the system occured in October 1993 and January 1994, because of dyke breaching, causing severe flooding in the North of the delta. In general, inputs and outputs of water in the delta are, to a certain extent, controlled by man. This does not mean, however , that hydraulic data are easy to acquire, because water boards represent only half of the irrigation pumping capacity, the remaining being private. Furthermore, the gravity drained basins were ungauged, and the management of the sea dyke is not well known. The major problem lies in the fact that there is a lack of centralization of data concerning the hydraulic management of the system, mainly because historically, water management has always been a source of social tension in this area.

The central lagoons of the Camargue (Vaccarès and "Etangs inférieurs"or coastal lagoons) extend over a large area (about 110 km^2) but are shallow (1 m deep on average). They constitute a hydraulic unit in which water levels and salinities vary in space and time according to climate, the action of the sea and, above all, man's activities (Heurteaux, 1992). The Vaccarès is a permanently flooded lagoon of 1.5 m mean depth and 64 km^2 area, whose intermittent connection with the sea is controlled by means of sluices through the seadyke, at the Grau de la Fourcade (figure 2). There is a natural channel (Grau de Mornès) between the Vaccarès and the coastal lagoons, for a Vaccarès level inferior to -0.30 to -0.40 m (a.s.l), hydraulic connection between the two sub units is cut. Generally, salinity of the coastal lagoons is greater than in the Vaccarès, due to increase by inflow of sea water and salt-laden groundwater (Heurteaux, 1992; Chauvelon, 1996).

The Fumemorte canal is a drainage canal, 0.5 to 1.5 m deep, 14 km in length. A raisable barrier was installed near to the mouth of the canal in November 1987, to prevent incursions of saline water from the Vaccarès. The stream cross section is reduced, and downstream the weir, a drowned hydraulic jump appears, depending of the hydraulic gradient between the canal upstreams the weir and the Vaccarès. The agricultural area is not the only managed part of the catchment, artificial connections between the main drainage network and marshes exist. These intermittent connections and their management changed the ecological functionning of temporary marshes previously not colonized by fish (Crivelli, 1981; Pont *et al.*, 1991). Among these marshes, two were studied for fish populations dynamics purposes (Rosecchi and Crivelli, 1995; Poizat and Crivelli, in prep): the Relongues Nord and Relongues Sud, that are flooded seasonnally by autumn and spring rains. When the level of the Fumemorte is between -0.15 and -0.10 m (a.s.l) the connexion with the Relongues Nord is cut, and if it falls to -0.15 to -0.20 m (a.s.l) it no longer communicates with the Relongues Sud.

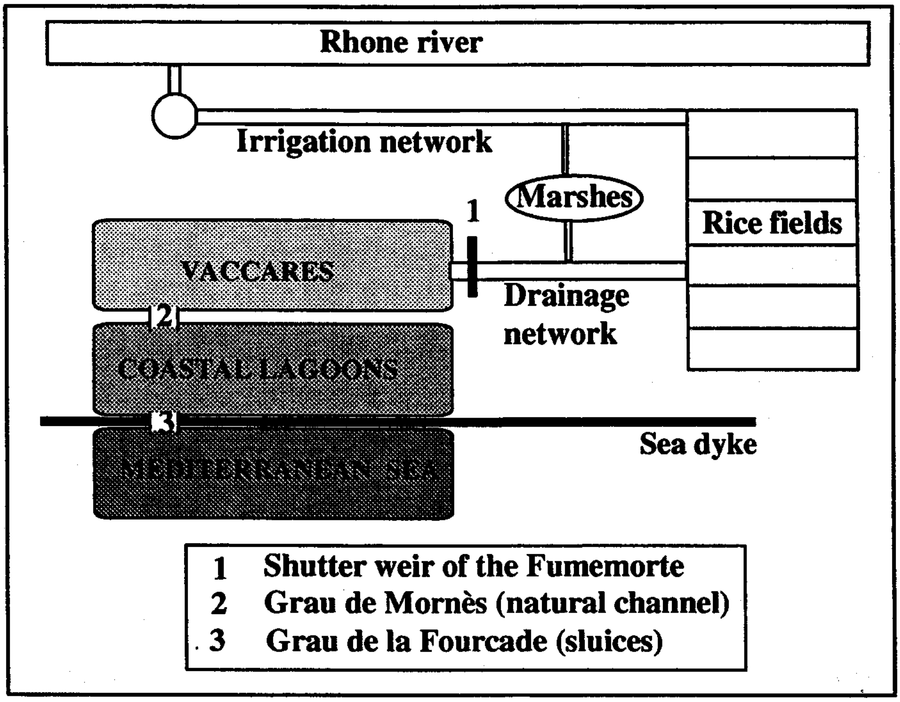

Figure 2: Diagrammatic map of study area.

MATERIAL AND METHODS

Land use and other geographical data

Digitized maps of agricultural and natural plots, hydraulic networks and wetlands were made from aerial photographs, and integrated in a georeferenced data base to permit actualization of land use. All parcels were gathered in larger agricultural units attributed to their own irrigation station (Chauvelon, 1996). Supervised classifications were realized on satellite images from Landsat Thematic Mapper-5, using a combination of Near Infra-Red (NIR) and Mid Infra-red (MIR) bands, well adapted to discriminate rice fields from their surroundings (Sandoz, 1996) after their filling.

Estimation Of Nutrients And Sediments Inputs

Importation with irrigation water

Concentrations of total suspended materials in the Rhône river were obtained on a daily basis for 1993 and 1994 (Pont, pers. comm.). Concentrations of nitrogen, phophorus and organic carbon were calculated using the relations presented by El Habr and Golterman (1987), between concentrations and discharge of the Rhône. The volume introduced for the irrigation of rice fields was obtained using the method described by Chauvelon (1996). It consisted of a distributed approach based on an experimental survey of pumping stations (duration of pumping, gauging tests, management practices) with the spatial and temporal sharing out of their irrigation volume between agricultural units.

Fertilisation Of Rice And Wheat

An enquiry about the nature and quantity of fertilisers used for rice and wheat crops on the Fumemorte catchment was conducted. Informations on the timing of fertilisation was also acquired. The sample covered 86 % and 70 % of the areas devoted to rice and wheat cropping respectively. Using the georeferenced data base it was possible to calculate fertiliser input on a discrete manner, taking into account the spatial variability of fertilisation practices. For the farms not sampled, fertilisation application recommended by the French Rice Center were used (Chauvelon, 1996). The French Rice Center recommend a total application of 150 kg N ha^{-1} for rice cropping. With a timing of application : 0 DAS, 30 DAS, 60 DAS (DAS : Days After Sowing), with 50 kg N ha^{-1} at each date. The used form of nitrogen is ammonium sulfate or prilled urea. Phosphorus (under the form of P_2O_5) is applied with the first N application (60 to 80 kg P ha^{-1}). For the fertilisation of wheat 120 kg N ha^{-1} and 80 kg P ha^{-1} are used.

Atmospheric Nitrogen and Phosphorus Inputs

In order to estimate direct atmospheric inputs of N and P with rain, data from the literature were used with our precipitation measurements. For N, we retained the value of 0.5 mg l^{-1}, measured in the west mediterranean coast (Loye-Pilot *et al.*, 1990). For concentrations of phosphorus we used the value of 0,04 mg l^{-1} P recorded by Auby *et al.* (1994).

Fluxes at the outlet of the catchment

Because it was not possible to derive a rating curve from level / discharge measurements, a ultrasonic open channel flow gauge was used to obtain the discharge at the outlet of the catchment. The device typically consisted of 3 to 4 pairs of transducers for velocity measurements and a piezocapacitive level sensor.
Between the gauging section and the shutter weir at the outlet, an automatic sampler was installed. In each flask of the sampler, 10 ml of a solution (20 g l^{-1}) of mercuric chloride were introduced before the sampling cycles. This was done in order to stop the biological activity in the samples, before their collection and refrigeration. Before each sampling, the suction pipe was automatically flushed out by air forcing. The chemical analysis were carried out according to the French norms for water analysis (AFNOR) by a specialized laboratory: ortho-phosphate (and total phosphorus after acid hydrolosis) using molybdate-antimony, ammonia using indophenol, total nitrogen using Kjeldhal method. Water samples were filtered on GF/C filters, dried at 110 °C, to obtain total suspended material, than passed to oven at 500 °C to derive the organic suspended material.

RESULTS

Inputs with irrigation water

During the crop season of 1993 and 1994, more than 50 10^6 m^3 of water from the Rhône river were introduced for irrigation in the agricultural area of the Fumemorte catchment (table 1). The associated sediment importations were mainly mineral, for both years, total loads of total suspended material (TSM) were superior to 2,000 tons (table 1). Inputs of nitrogen (N) and phosphorus (P) due to irrigation water were of the same order for the two studied years.

Table 1: Importation of irrigation water (10^6 m^3 yr^{-1}), sediments, nitrogen and phosphorus (10^3 kg yr^{-1}) from the Rhône to the Fumemorte catchment (from Chauvelon, 1996)

Year	IRRIG.	TSM	OSM	N	P
1993	56.2	2,300	69	69	9.5
1994	52.5	2,200	80	65	9

Fertilisation and atmospheric inputs

The major importations of nitrogen (about 300 tons each year) and phosphorus (about 120 tons) are due to the fertilisation practices, mainly for rice cultivation (table 2). It is worth noticing that estimates of atmospheric inputs with precipitations, in particular for nitrogen (N atm and P atm in table 2) are not negligible.

Table 2: Importations of N and P with fertilisers and precipitations (10^3 kg yr^{-1}) on the Fumemorte catchment (from Chauvelon, 1996)

Year	N rice	P rice	N wheat	P wheat	N atm	P atm
1993	302	122	40	26	16	1
1994	295	117	33	22	12	1

So we can estimate, on average for the two studied years, that annual loads of 402 tons of nitrogen (N) and 153 tons of phosphorus (P), all sources combined, entered the catchment of Fumemorte.

Dissolved And Solid Fluxes At The Outlet

Although inverted flow may occur, mean daily discharge varied between 0 and 10 $m^3 s^{-1}$ in 1993 and 1994. In the absence of precipitations, the drainage of the 20 km^2 of rice fields of the catchment, during the irrigation period, corresponded to a discharge of about 3 $m^3 s^{-1}$ at the outlet (Chauvelon, 1996). Increases of annual drainage volume and associated fluxes for the year 1994 (table 3) were due to a wetter autumn. Actually, the mesured precipitation depth for September 1994 (280 mm) was the highest of the last 30 years. Comparing inputs and outputs of TSM (table 2 and 3), we may conclude that a sedimentation occured in the rice fields and the drainage network. The fluxes of organic suspended material (OSM) were 5 to 6 times greater than inputs with irrigation water. Only 6 % of introduced N and 4 % of P were exported from the Fumemorte catchment to the Vaccarès lagoon (table 2 and 3).

Table 3: Annual drainage discharge (10^6 m^3 yr^{-1}) and sediments, nitrogen and phosphorus fluxes (10^3 kg yr^{-1}) from the Fumemorte catchment (from Chauvelon, 1996)

Year	DRAIN.	TSM	OSM	N	P
1993	45.4	1,500	390	21	6
1994	53.4	1,940	536	23	6

DISCUSSION

Dissolved and solid fluxes

The lowest detectable values for concentrations were 50 10^{-6} g l^{-1} for P-PO_4, 100 10^{-6} g l^{-1} for total-P, 500 10^{-6} g l^{-1} for total-N, and 50 10^{-6} g l^{-1} for N-NH_4. It is important to notice that a number of water analysis results were expressed as « inferior to these values ». In these cases we retained the limit values (Chauvelon, 1996) for the calculation of fluxes. As a consequence the calculated fluxes were made according to a scenario of maximum transfer from the catchment. Despite the fact that the fluxes are to considered as maximum values, we can conclude that transfers of nutrients from the catchment to the lagoon system are low. The intensity of nitrification-denitrification processes in rice fields was already demonstrated in experimental plots in the Camargue (Golterman *et al.*, 1988; Minzoni *et al.*, 1988); El Habr and Golterman, 1990). Boisserie (1990) reported that a quantity up to 10 % of applied N could be lost by volatilization in rice fields. Stutterheim (1994) showed that the efficiency of applied nitrogen for fertilisation of rice in the Camargue was about 20 %. Other authors emphasized the fact that precipitation under the form of apatite was the major process affecting phosphorus in the irrigation and drainage waters in the Camargue (Godin, 1990; De Groot, 1992). Extrapolating to the whole catchment, and comparing to other mediterranean catchment-lagoon systems, it seems (Chauvelon, 1996; Chauvelon, in prep.) that the Vaccarès system could be classified as « poorly feeded » by its catchment, as far as N and P are concerned.

Even for a rainy year, with an area of more than 2,000 ha of rice fields, importation of sediments with irrigation water globally compensate for the erosion of the catchment. The organic matter of the drainage water probably corresponded to exportation of biomass (green and blue-green algae) from the rice fields, and to phytoplankton production in the drainage network (De Groot, 1992; Chauvelon, 1996).

Eventually, despite the relatively great quantities of fertilisers used for rice cultivation in the Camargue, it does not seem that a problem of anormal eutrophication should occur in the lagoons of the « Réserve nationale de Camargue ». Nevertheless, a potentially important aspect of water quality in the Camargue concerns the fluxes of pesticides and heavy metals coming from the Rhône or introduced in the agricultural system. Further studies focusing on these problems ought to be done, because of their importance for the ecological functionning of this protected area.

Impact of rice cultivation and hydraulic management on fish populations

Introduction of irrigation water has two main effects. The first is that some fish are pumped with the water and introduced in the hydraulic network of the Camargue, contributing by this way to the recruitment (Crivelli, 1981). With irrigation pumping, recruitment concerns either species that could only enter the network by this way (juveniles of shad (*Alosa fallax*), barbel (*Barbus barbus*), or species that are also capable to come from the sea and the lagoons (mullet (*Mugil cephalus, Liza ramada*); *eel* (*Anguilla anguilla*)) (Crivelli, 1981). A migration may occur from the irrigation to the drainage network, but essentially by transit via marshes.

Secondly, a direct consequence of rice irrigation is the produced drainage volume of the agricultural areas. Although 15,000 $m^3 ha^{-1}$ should be sufficient for rice irrigation, values as important as 40,000 $m^3 ha^{-1}$ were observed (Chauvelon, 1996). By this way, an important freshwater volume is exported and stocked in the drainage network during the period of natural hydric deficit. Globally, this drainage volume offers a vital space for freshwater fish species such as: black bullhead (*Ictalurus melas*), goldfish (*Carassius auratus gibelis*), (Pumkinseed sunfish (*Lepomis gibbosus*), carp (*Cyprinus carpio*), pike (*Esox lucius*), pike perch (*Schizostedion lucioperca*), roach (*Rutilus rutilus*), rudd (*Scardinius erythrophtalmus*), tench (*Tinca tinca*). Because the salinity of the Vaccarès between 1982 and 1993 was superior to 15 g l^{-1}, (Heurteaux, 1992) the drainage network was the only refugee for these species. Due to rice fields drainage, the levels in the Fumemorte canal and its affluents prolongate the period of connection with temporary marshes and their use by fish for reproductive and trophic purposes (Rosecchi and Crivelli, 1995; Poizat and Crivelli, in prep.).

The drainage volume partly compensate for the evaporation of the lagoons, and maintain a level (except in 1989, a particularly dry year) which permit the hydraulic connection between the drainage network, the Vaccarès, the coastal lagoons and the sea. By this way, migration of fish within the hydrosystem is potentially always possible, whereas the lagoon system was partly dewatering during the summer before rice cultivation appeared in the Camargue (Heurteaux, 1992). Migrations between the sea and the coastal lagoons are totally conditionned by the management of the sluices of the sea dyke. Generally, the management of the sluices is done (total aperture) in order to permit the outflow of water from the lagoons, when N-NW winds are blowing in winter, allowing fish migrations. From February to June, at least one sluice is opened each night to allow recruitment (Bardin, 1994).

The raisable barrier on the Fumemorte canal may have a negative effect on fish migrations from the lagoons to the drainage system (Crivelli, unpubl. data), especially at the beginning of the irrigation period, when levels rise in the drainage network, or in winter if the Vaccarès level is low because of drought. Weak swimmers such as shrimps (*Crangon crangon*; *Palaemonetes spp*), small fish species (pipefish (*Syngnatus abaster*); three spined stickleback (*Gasterosteus aculeatus*); sand smelt (*Atherina boyeri*)), and fry (notably of mullet (*Mugil cephalus, Liza ramada*)) may particularly be affected by this obstacle.

Hydro-agricultural management in the Camargue has both negative and positive effects on fish populations status and migrations. Because of a management scheme not always adapted to the timing of fish migrations, in particular in the case of the shutter weir at the outlet of the Fumemorte, fish from the Vaccarès should have difficulties to enter the freshwater system. Nevertheless it seems that agricultural drainage contribute, to a certain extent, to the compensation for climatic extreme events and their effects on fish population status, in this highly artificial hydraulic system.

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RESPONSE OF A NATURAL RIVER VALLEY WETLAND TO SUPPLEMENTARY RUNOFF AND POLLUTANT LOAD FROM URBAN WASTEWATER DISCHARGE

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ABSTRACT

The Mädajärve wetland, the riverhead of R. Tänassilma, is a marshland 6 km long and about 0.5 km wide, formed in an ancient glacial river valley in Central Estonia. Its natural catchment area embraces 17.58 km^2 of mostly cultivated lands and belongs to the drainage area of Lake Peipsi. From the adjacent town of Viljandi, belonging to the drainage area of the Gulf of Riga, the wetland is separated by a water divide. The upper part of the wetland was drained at the beginning of this century and was used as a meadow. The hydrology and the vegetation of the wetland were thoroughly studied in 1921. Since 1948 Mädajärve wetland has received precipitation runoff and untreated urban wastewater from the town of Viljandi by a outlet which passes the water divide. In 1990-91 urban discharge amounted 2534 thousand $m^3 \cdot y^{-1}$ (80.4 $l \cdot sec^{-1}$) which is equivalent to runoff from a 10 km^2 area of natural basin. This supplementary runoff, making up more than a half of the natural one, has essentially changed the water regime of the wetland. As a result, former meadows have become swampy. Vegetation has undergone great changes and become poor due to increased nutrient and organic matter loading and changes in the water regime. Pollutant loads, calculated for different substances in population equivalents (PE), were nearly equal showing the predominance of domestic wastes in urban runoff. The mean value 15,130 PE is in a good accordance with the real number of inhabitants in the region. The nitrogen load originating from wastewater exceeds 5-fold this from an equivalent rural area, the phosphorus load 33-fold and the BOD_5 load by two orders of magnitude. The upper 2 km reach of the wetland, previously covered by a species-rich grassland vegetation, is overgrown with a mono-species cattail (*Typha latifolia* L.) community. Downstream, in places of former swamp vegetation, it is replaced by dense stands of reed (*Phragmites australis* (Cavan.) Trin. ex Steud.), and small groups of birches and willows. In spite of the minimized species diversity of the vegetation the survived plants are well adapted to the high pollutant load, and the wetland serves as buffer area which efficiently reduces the amount of several pollutants.. The mean mass balance of substances for 1975-91 showed an average reduction of more than 99 % of coliform bacteria, 96 % of total solids, 94 % of permanganate oxygen demand, 87 % of BOD_5, 51-71 % of different mineral nitrogen forms and up to 63 % of total nitrogen in the wetland. Still, only 7 % of the total phosphorus load was accumulated, while the amount of phosphates increased more than twofold due to mineralization.

KEY-WORDS: Wetland / Hydrological regime / Wastewater treatment / Macrophytes / Mass budget / BOD_5 / Phosphorus / Ammonium / Nitrification / Denitrification

INTRODUCTION

The present paper represents an environmental impact assessment of long-term urban wastewater discharge into a river valley wetland in Central Estonia and was ordered by the town council of Viljandi. The main objectives of the investigation were:

1. to characterize long-term changes of the wetland, caused by supplementary runoff and the pollutant load;
2. to find out basic processes controlling the reduction of pollutants, and to evaluate their efficiency;
3. to estimate ecological hazards and hazards to public health in downstream areas.

DESCRIPTION OF THE STUDY SITE

The 34 km long River Tänassilma in Central Estonia starts from a wetland located in an ancient glacial river valley, at a distance of 0.5 km from the town of Viljandi (23,000 inhabitants). In its upper course the river flows on the bottom of a 30 m deep and up to 900 m wide ancient valley with a 400 m wide flood plain. The 454 km^2 river basin has a mean specific discharge of 8.1 l·sec^{-1}·km^{-2} (Nõukogude..., 1978) which is collected by 21 tributaries. The two largest among them enter the main river bed only 2 and 2.5 km downstream of the river head, bringing water from nearly half of the basin. Upstream areas contribute only 6.2 % to total runoff (Fig. 1).

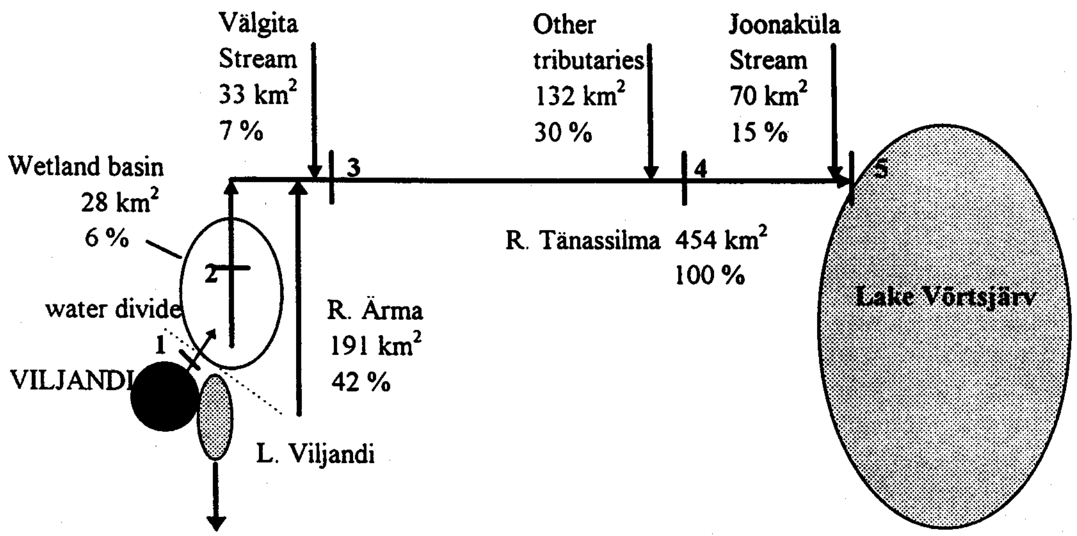

Figure 1: Scheme of the basin of the River Tänassilma. 1- 5 - measuring profiles

The flood plain wetland in the river head area is bounded by a water divide on the south and extends, as a 400-500 m wide and about 6 km long marsh, in NNE direction. Higher parts of the marsh were drained in the 1860s and repeatedly in the 1920s, and were used as haylands and peat mines. Since 1948, large amounts of untreated domestic sewage from the town were discharged to the wetland. At present the upper 2 km reach of the plain valley bottom, beginning from the wastewater inlet, is evenly covered with dense cattail (*Typha latifolia* L.) stands. Although old drainage ditches may still be partially functioning during flood, this 69 ha area has almost no surface overflow and is wholly functioning as a root system and peat filter. Intensive microbiological processes produce a lot of heat and prevent deep freezing of the soil even in severe winters. Two small lakes are located in the downstream part. Passing the first lakelet, the main part of the flow concentrates into the river channel, and the width of the wetland vegetation belt narrows down to 100 m. High water covers the slope areas of valley bottom only temporarily in April.

MATERIAL AND METHODS

The investigation was based on hydrochemical analyses of (1) wastewater for the period 1985-1991 (Fig. 1, profile 1), (2) effluent water from the wetland (profile 2), and (3) river water below the two larger tributaries (profile 3) for the period 1975-1991. In all years the four hydrological periods (winter low flow, spring flood, summer low flow, autumn high flow) were represented each by one to three analyses. Data on sewage amount (monthly discharge) were obtained from the local environmental agency. A highly significant correlation was found between daily flow measurements at the hydrological station which was operating on R. Tänassilma in 1955-1969 (profile 4) and at a hydrological station operating up to now on the adjacent R. Navesti ($r = 0.95$; $n = 5\ 480$). Runoff for profiles 1 and 2 was calculated from regression proportionally to the area of their basins. The transport of substances for measuring profiles was calculated by multiplying monthly runoff by ingredient concentration and expressed as $kg \cdot month^{-1}$. One-month gaps in concentration measurements were filled by linear interpolation. Longer gaps were first filled by extending the results pertaining to the same hydrological period to neighbouring months (e.g. results of July to June or August), until the gap narrowed to one month, which was thereafter filled by linear interpolation. Microbiological samples for determining the total count of bacteria, numbers of denitrifying, saprophytic and coliform bacteria were collected from profiles 2, 3 and 5 during winter low flow and flood peak periods in 1994. A 1:10,000 topographic map and aerophotos made at the flood peak of 1994 were used for describing flow conditions and for delimiting the areas of different plant cover involved in water purification. The investigation of the wetland area carried out by Rumma (1923) was used as the main reference for judging of changes in the hydrological regime and vegetation.

RESULTS

Hydraulic and pollution load

The hydraulic load of the Tänassilma wetland is formed by runoff from the natural basin and by the sewage of Viljandi. The natural catchment area embraces 17.58 km^2 of mostly cultivated lands and belongs to the drainage area of Lake Peipsi and the Narva River. Runoff water from about 60 %, i.e. 10.5 km^2 of the natural basin passes measuring profile 2. From the adjacent town of Viljandi, belonging to the drainage area of the Gulf of Riga, the wetland is separated by a water divide.

Sewage water is collected from approximately 2/3 of the town area and from two smaller settlements, and is discharged to the wetland by a pipe which crosses the water divide. The town is situated on the high western bank of the Tänassilma valley sloping steeply towards the wetland. A small stream which runs just along the eastern border of the town and enters Lake Viljandi, intersects the passage of runoff waters to the wetland. A combined sewerage is used in the town with precipitation water forming part of discharge. However, a big difference in the seasonal dynamics of wastewater discharge and runoff from natural areas (Fig. 2) allows to assume a rather small proportion of precipitation water in total discharge. In 1990-1991 sewage discharge fluctuated from 132 to 311 thousand $m^3 \cdot month^{-1}$ and was almost not affected by flood in April (159 thousand $m^3 \cdot month^{-1}$). The mean annual discharge in 1990-91 was 2.53 million m^3 or 80.4 $l \cdot sec^{-1}$, i.e. approximately equivalent to runoff from 10 km^2 of the natural basin. Consequently, total discharge to the wetland can be dealt with as discharge from an area of about 28 km^2, with two thirds of it having a natural dynamics and one third being rather constant.

The 6 km reach from the water divide to the mouth of the Ärma River with a total area of 92 ha forms a natural buffer zone before water enters a larger system. Effluent water from this area was controlled by measuring profile 2. The pollution load was derived as the sum of sewage load and runoff from the

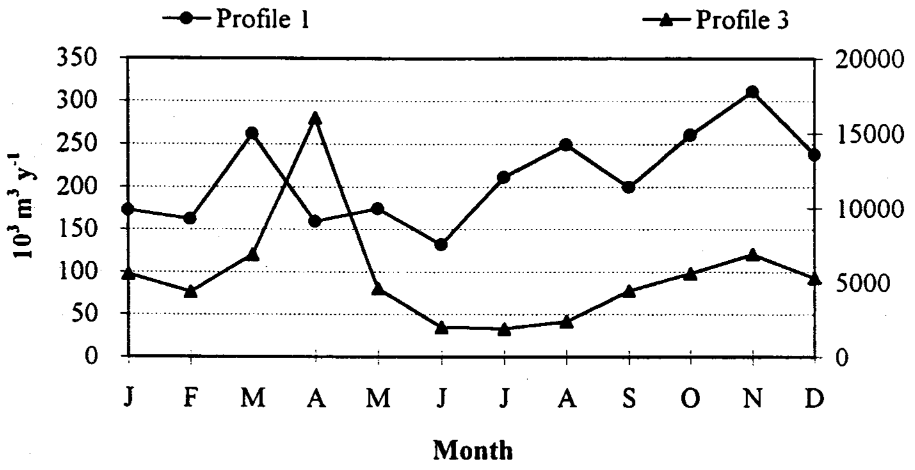

Figure 2. Seasonal dynamics of sewage discharge (profile 1) and runoff from the natural basin (profile 3)

Table 1: Total and specific pollution load to the wetland

Constituent	Load from the sewerage ($t \cdot y^{-1}$)	(PE)	Load from the natural basin ($t \cdot y^{-1}$)	Specific load ($kg \cdot ha^{-1} \cdot y^{-1}$)
Total suspended solids (TSS)	416		?	4 522
Biochemical oxygen demand (BOD_5)	240	12 177	6	2 674
Permanganate oxygen demand (COD_{Mn})	844		?	9 174
NH_4-N	59		-	641
NO_2-N + NO_3-N	1		21	239
Total nitrogen (N_{tot})	75	17 067	21	1 043
PO_4-P	3		0.3	36
Total phosphorus (P_{tot})	9	16 145	0.3	101
Mean:		15 130		

natural basin (Table 1). The latter was calculated according to Maastik (1984) as runoff from a predominantly (75 %) reclaimed basin. The following equations were applied to recalculate the load of different constituents into population equivalents (PE) (Maastik, 1984): 1 PE = 19.71 kg $BOD_5 \cdot y^{-1}$ = 0.53 kg $P_{tot} \cdot y^{-1}$ = 4.38 kg $N_{tot} \cdot y^{-1}$. The pollution load in PE, derived from different sewage constituents, gave rather similar estimates, which shows the predominantly domestic origin of wastewater. The mean value of 15,130 PE is in a good accordance with the real population in the urban area.

Mass budget of substances

The removal of substances was calculated as the external budget between measuring profiles 1 and 2 (Table 2). A comparison of profiles 2 and 3 enabled to evaluate quality differences between rural runoff and pre-treated urban discharge.

Table 3: External mass budget of the wetland and the role of urban sewage constituents in downstream water

Component	Input ($t \cdot y^{-1}$) (Profile 1)	Output ($t \cdot y^{-1}$) (Profile 2)	Removal (%)	Profile 3 ($t \cdot y^{-1}$)	Role of urban discharge (%)
Water		3 803 000		65 911 000	6
TSS	416	18	96	290	6
BOD_5	246	36	85	175	21
COD_{Mn}	844	48	94	783	6
NH_4-N	59	29	51	50	58
NO_2-N + NO_3-N	22	6.4	71	120	5
N_{tot}	96	35-96	≤64	539	6-17
PO_4-P	3.0	6.3	-110	11.4	55
P_{tot}	8.9	8.0	10	16.3	49

Total suspended solids (TSS)

The annual average of the removal of suspended solids in the wetland area was 96 %. Wastewater particles settle down or are removed by filtration through the rootzone. Output particulate matter consists mainly of detrital plant material but may contain also mineral particles during flood. The high proportionality of TSS transport with water flow confirms the mainly natural origin of output particles. In April the transport of TSS exceeded seven times the mean value of the rest of the year while retention dropped to 63 % (Fig. 3).

Biochemical oxygen demand (BOD_5)

A decrease in BOD_5 in the wetland, based on the heterotrophic degradation of organic matter, varied as a temperature function from 75 % in winter to 97 % in summer (Fig. 3). Only during high flow in April and November the purification efficiency was lower, probably due to the overflow of wastes. Contrary to the concentration of suspended solids, that of BOD_5 was inversely proportional to water flow. Both the high concentration of BOD_5 in the winter season and high flow in April resulted in an increase in BOD_5 runoff at profile 2. Six per cent of water carried 20 % of BOD_5 to profile 3, which shows a higher pollution potential of wetland runoff compared to waters from the rural basin.

Permanganate oxygen demand (COD_{Mn})

The reduction efficiency of COD_{Mn} was high (94 % on the average) and less temperature dependent than that of BOD_5. The transport of COD_{Mn} was proportional but its concentration inversely proportional to

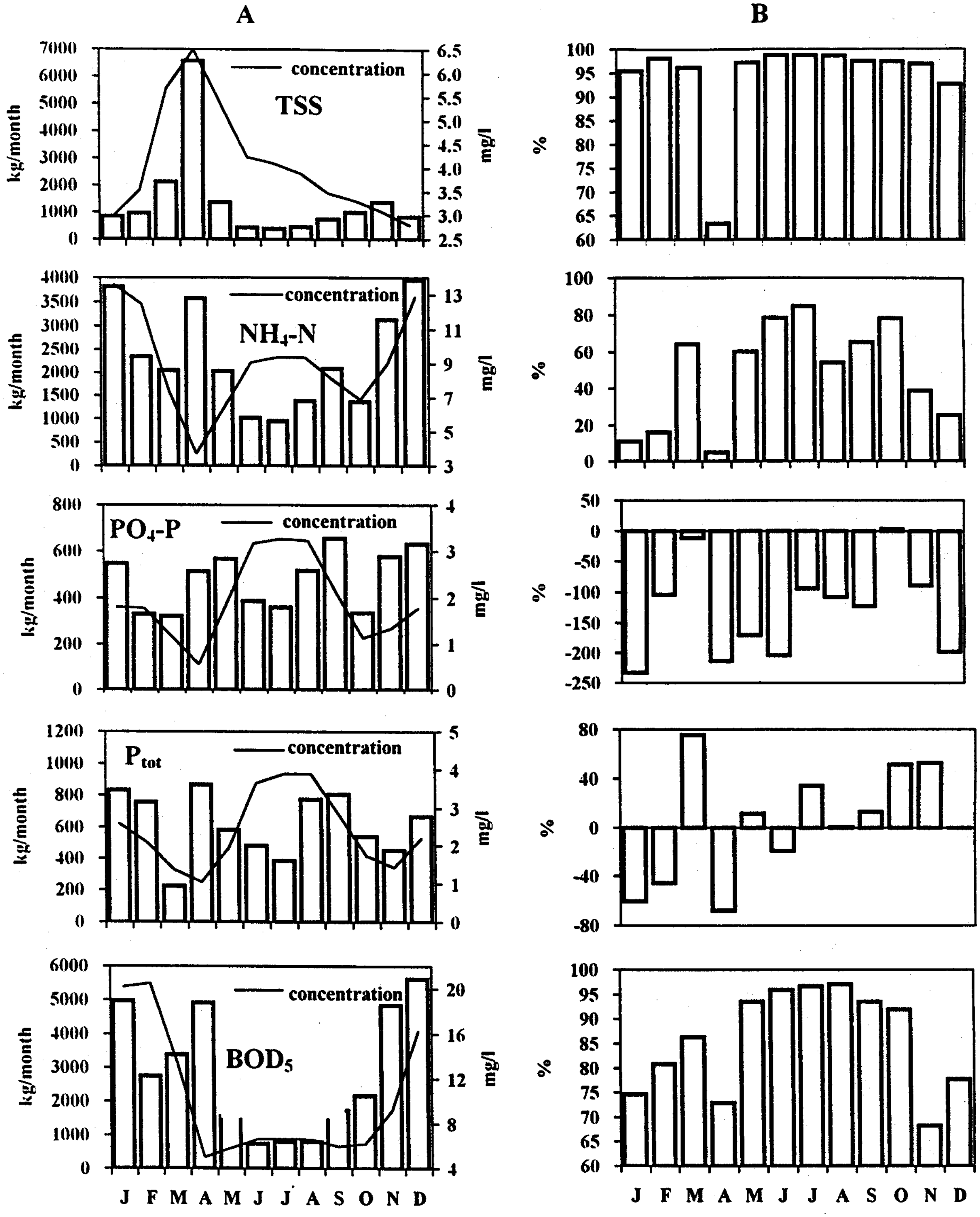

Figure 3: Seasonal dynamics of the concentration and transport of pollutants at measuring profile 2 (A) and their removal in the wetland in percenages from the input (B)

water flow at profile 2. Considering COD_{Mn} the quality of output water from the wetland did not differ from the quality of rural runoff.

NH_4-N

The reduction of ammonium nitrogen varied in a wide range from 5 to 85 % (mean 51 %) of the input in different seasons, clearly depending on temperature (Fig. 3). Approximately 80 % of nitrogen in sewage water was discharged as ammonium, and its concentrations at profile 2 remained extremely high (4 - 14 mg · l^{-1}). The seasonal dynamics of NH_4-N concentrations was formed mainly as a result of the dilution rate. Besides intensive leaching in winter, a high peak occurred also in April. Derivation of NH_4-N from urban wastes made up nearly 100 % of its total transport at profile 3 in summer, while rural runoff waters contained only trace concentrations of ammonium. On the average 60 % of NH_4-N was discharged from the area the hydraulic load of which formed only 6%.

NO_2-N and NO_3-N

The removal of oxydized forms of mineral nitrogen in the wetland was rather high reaching a mean value of 71%. These nitrogen forms were loaded mainly from the natural drainage basin (Table 1) but were also formed in the wetland as products of nitrification. As the seasonal dynamics of the load from the natural basin was unknown, the dynamics of nitrate removal could not be analysed. The patterns of concentration and transport at profile 2 were similar to those of suspended solids, both demonstrating a strong proportional relation to water runoff. The concentrations of nitrites and nitrates in runoff water from the wetland were equal or even lower than their concentrations in rural runoff, which demonstrates a high denitrification potential of the wetland.

Total nitrogen (N_{tot})

The shortage of data on total nitrogen for profile 2 hindered the building of a correct mass balance for the wetland, which can therefore be discussed only in terms of boundary conditions. Proceeding from the total load of nitrogen (96 t · y^{-1}) and supposing that all nitrogen in the wetland output was in the mineral form, the highest possible retention of N_{tot} will reach 63 %. Actually there is always a certain proportion of organic nitrogen in runoff water, the real nitrogen removal being smaller. The runoff of total nitrogen derived from the wetland forms 6 - 17 % of its runoff at profile 3: 6% in case all nitrogen is discharged in the mineral form and 17 % in case all 96 t · y^{-1} of nitrogen passes the wetland without any retention. The real value remains somewhere within these limits.

Total and phosphate phosphorus

The seasonal dynamics of P_{tot} and PO_4-P were similar to the dynamics of NH_4-N and COD_{Mn}, i.e. they were mainly shaped by the dilution effect of water (Fig. 3). The leaching of both forms with effluent water was rather stable and did not exhibit any distinct seasonality. As a result of mineralization the amount of output phosphate constantly exceeded the input, and the calculated retention obtained negative values, being on the average -110 %. In the case of P_{tot} periods of predominating accumulation alternated irregularly with periods when leaching exceeded the input. The annual retention of 600-900 kg P_{tot} yields a removal efficiency of 7-10 %. At profile 3 the load by effluent water from the wetland formed 55 % of PO_4-P load and 49 % of P_{tot} load, which exceeded 8-9-fold the proportion of the hydraulic load from this area.

Microbiological status

According to the saprobity classifications of Estonian waters by Maastik (1984), and Järvet and Saava (1990) (Table 3) the status of the R. Tänassilma during winter low flow was characterized as a-mesosaprobic to hypersaprobic at profile 2, b-mesosaprobic to polysaprobic at profile 3, and oligo- to a-mesosaprobic at the mouth of the river. The situation improved during flood, when the waters could be classified as oligo- to a-mesosaprobic. In general, the number of saprobic bacteria indicated lower saprobity than the total count and coli-index. All counts decreased from profile 2 to profile 5, showing a rather good self-purification capacity of the river. On the basis of the mean value of the coli-index in sewage, 10^9 cells l^{-1} as given by Maastik (1984), only a rough estimate of the disinfecting efficiency of the wetland could be made. A decrease in the coli-index was at least 99.8 %, however, the remining 0.2 % still exceeded the sanitary standard (5 000) nearly 500 times (Table 4).

Table 3: Saprobity and microbiological status of the R. Tänassilma. h - hypersaprobic, p - polysaprobic, α – α-mesosaprobic, β – β-mesosaprobic, o - oligosaprobic, TCB - total count of bacteria

Date	Profile	TCB 10^6 cells ml^{-1}		Saprobacteria cells ml^{-1}		Coli-index cells l^{-1}		Denitrifying bacteria cells ml^{-1}
27.02.94	2	11.7	h	36 000 α		>2 400 000	h	600
	3	5.4	p	2 400 β		>240 000	p	>250
	5	2.9	β	810 o		24 000	α	>250
12.04.94	2	5.0	α	7 800	β	62 000	α	250
	3	2.6	β	2 400	β	<5 000	β	250
	5	2.3	β	1 800	β	600	o	60

The influence of urban pollution on downstream areas

On the basis of a comparison of measured monthly mean concentrations of pollutants and maximum permissible concentrations (Saava, 1990; Järvet and Saava, 1990, Table 4) three components of environmental hazard are singled out: phosphorus, ammonium and coliform bacteria. Calculations showed that the additional nutrient load by downstream tributaries and self-purification during flow compensate

Table 4: Correspondence of effluent water quality from the wetland to sanitary and ecological standards

Parameter	Standard	Measured	Measured / standard
Coli-index, cells l^{-1}	5 000	62 000 - 2 400 000	**12 - 480**
TCB, 10^6 cells ml^{-1}	3	5.0 - 11.7	1.7 - 3.9
Saprobacteria, cells ml^{-1}	5 000	7 800 - 36 000	1.6 - 7.2
BOD_5, mg l^{-1}	3	5 - 20	1.7 - 6.7
COD_{Mn}, mg l^{-1}	15	10 - 17	0.7 - 1.1
NH_4-N, mg l^{-1}	0.4	4 - 14	**10 - 35**
NO_2-N, mg l^{-1}	0.02	0.01 - 0.05	0.5 - 2.5
NO_3-N, mg l^{-1}	1.2	0.2 - 1.9	0.2 - 1.6
PO_4-P, mg l^{-1}	0.03	0.6 - 3.3	**20 - 110**
P_{tot}, mg l^{-1}	0.10	1 - 3.8	**10 - 38**

exactly each other, and the amounts of nutrients registered at profile 3 (16.3 t P_{tot} and 539 t N_{tot} per year) reach L. Võrtsjärv. The mass budget of substances for L. Võrtsjärv, a 270 km^2 shallow eutrophic waterbody, revealed the second highest P_{tot} loading and the third highest N_{tot} loading to the lake by the R. Tänassilma (Nõges and Järvet, in press). High bacterial pollution turns the whole river into an insanitary state, especially during low flow periods.

Long-term changes

An increase in the hydraulic load caused by urban wastewater has remarkably changed the water regime of the area. Former drained grasslands which spread in the upper parts of the valley bottom in the 1920s (Rumma, 1923) have turned into an impassable swamp. The previous border of the swamp at a distance of 1.8 km from the water divide can still be recognized by a sharp front in vegetation where the dominating cattail (*Typha latifolia*) is replaced by dense reed (*Phragmites australis*) stands. The present hydrological situation resembles that of the beginning of this century, when the outflow of L. Viljandi was clogged, and one part of water flowed to the direction of the Tänassilma valley. The increased water flow has kept the two swamp lakes described by Rumma in a state of intensive overgrowing with sphagnum and emergent or floating-leaved vegetation (*Carex limosa* L., *C. lasiocarpa* Ehrh., *Phragmites australis*, *Typha latifolia*, *Ranunculus lingua* L., *Glyceria* spp., *Nuphar luteum* L., *Alisma plantago-aquatica* L., *Sagittaria sagittifolia* L., *Sparganium emersum* Rehm.). As a result of the increasing pollution load most of the species have vanished. Besides cattail and reed only *Glyceria plicata* Fr. and some carex species have tolerated the changes.

Two groups of substances with a different chemical behaviour could be distinguished by means of statistical analysis of the time series for 1975-1991. The first group consisted of natural macroconstituents of water such as Ca^{++}, Mg^{++}, SO_4^{--} and HCO_3^-, not discussed in this paper, whose runoff was closely related to water flow. Among the analysed polluting components COD_{Mn} showed a similar pattern. The second group was formed by three main pollutants, NH_4-N, PO_4-P and BOD_5, which were almost not affected by flow on a long-term scale, but had significant ($p < 0.01$) intercorrelations (NH_4-N - BOD_5 $r = 0.74$; NH_4-N - PO_4-P $r = 0.62$; BOD_5 - PO_4-P $r = 0.55$). The runoff of ammonium and BOD_5 are both inversely correlated with the capacity of oxydation processes in the wetland. An increase in these variables gives evidence of a temporal or spatial broadening of anaerobic conditions, which inhibit nitrification and aerobic destruction of organic matter. A similar behaviour of phosphate is probably caused by the different solubility of Fe^{2+} and Fe^{3+} phosphate.

DISCUSSION

Processes involved in the reduction of pollutant runoff

Filtration, sorption, and sedimentation

Suspended solids of wastewater as well as insoluble products of chemical reactions and complexation precipitate rapidly when flow rate decreases. In the Mädajärve wetland where the major part of water passes the area as subsurface flow, the filtration of solids by peat and the rootsystem represents one of the main retention processes. The low P removal efficiency points to the saturation of soils with phosphorus. Adsorption and precipitation of phosphate through the ligand exchange reactions with aluminium, iron calcium and clay minerals are the main removal mechanisms for phosphorus in most macrophyte systems (Gumbricht, 1993), however, the adsorption capacity of soils is finite and will be exhausted after some years depending on the initial adsorption capacity of soil and phosphorus loading rate (Jenssen *et al.*, 1991; Teal, 1991).

Bioproduction and burial of sediments

In the course of bioproduction the atomic ratio of C, N and P uptake by plants is equal to 106:16:1 (Redfield, 1958). In natural wetlands where plants cannot be harvested, most of the assimilated substances will be released by destruction, and the role of plants as a sink of nutrients is extremely small (Obarska-Pempkowiak, 1991; Teal, 1991). Still, in the Mädajärve wetland the accumulation of phosphorus in newely-formed peaty sediment is probably the only process controlling the retention of phosphorus, as older sediment layers are saturated with phosphorus and have lost the sorption potential. The seasonality of nutrient uptake by plants was clearly reflected in ammonium runoff which increased remarkably in the cold season. In general, the role of plants in supplying a surface for micro-organisms, in stabilizing accumulated sediments, in aerating and maintaining the permeability of soils (Hofmann, 1991) is much more significant than their direct accumulation capacity.

Destruction and mineralization

Higher than 95 % reduction of BOD_5 and COD_{Mn} levels, and an increase in the PO_4-P/P_{tot} ratio of effluent water point to intensive and sufficient mineralization in the Mädajärve wetland. Large loading of easily decomposable and oxygen consuming material lowers the redox potential of the soil and contributes to phosphorus mobilization (Holford and Patric, 1979). Ammonification as the main mineralization step of nitrogen is almost completed already in the sewerage.

Nitrification

In prevailing anaerobic conditions of the wetland, nitrification is inhibited, and large amounts of nitrogen leach in the form of ammonium. Apparent at a first glance contradiction between the efficient mineralization and inhibited nitrification, both based on oxydation processes, can be explained by physiological differences between bacteria involved. Decomposing heterotrophic bacteria (some of them being obligate anaerobes) are able to use nitrates, sulphates and other oxydized compounds as electron acceptors in anoxic conditions, unlike nitrifying bacteria which represent obligate aerobes and require a certain amount of free oxygen in their environment.

Denitrification

The lower concentration of nitrates in treated urban discharge compared to rural runoff water shows a high denitrification activity of the wetland. Anaerobic environment and a source of organic carbon, needed for denitrification, occur commonly in wetland systems; however, oxydizing conditions necessary for nitrate formation are often limited, hence the rate of nitrification restrains the rate of denitrification (Brix and Schierup, 1990) and nitrogen reduction as a whole.

The role of the water regime

Changes in hydraulic loading have probably played a minor role in the long-term changes of the area, compared with the role of pollution, but cannot nevertheless be neglected. First of all, they are responsible for the expanding of the wetland area southward to such an extent that this part forms the main buffer zone at present. As shown by van der Valk (1990), a rise of the water level in a wetland resulted in the elimination of wet-meadow species which could not survive permanent flooding and whose seeds did not germinate under water. In steady state conditions during the last decades, water level fluctuations affected the processes controlling the retention and/or leaching of substances. Freeman *et al.* (1993) demonstrated an increased release of many solutes including nitrate (1250 %), sulphate (116 %), dissolved organic carbon (37 %), sodium (66 %), chloride (65 %), iron (168 %) and magnesium (16 %) when the water table height was reduced in peat-soil cores from a riparian wetland. The authors conclude that wetlands may only represent a temporary solution to water quality problems, the beneficial effects of which can be reversed by climatic change.

CONCLUSIONS

In the conditions of supplementary runoff and pollution load from domestic wastewater a stable wetland community has formed which is dominated by broad-leaved cattail next to the sewerage outlet and by bur reed in the lower regions. The low retention of phosphorus, ammonium and coliform bacteria represents the main danger to the water quality of downstream systems. The leaching of phosphorus is intensive due to the exhausted adsorption capacity of the soil. Anaerobic conditions predominating in the wetland affect both phosphorus and nitrogen: the former by mobilizing iron, calcium and aluminium compounds which otherwise bind phosphorus, and the latter by the hindrance of nitrification carried out by obligatory aerobic bacteria. In spite of the high retention efficiency (over 99 %) of coliform bacteria, their number in effluent water exceeded sanitary requirements up to 500 times during the winter low flow period.

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Role of abiotic variables

Rôle des variables abiotiques

ANALYSIS OF FLOW VELOCITY FLUCTUATIONS IN DIFFERENT MACROPHYTE BANKS IN A NATURAL OPEN CHANNEL

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ABSTRACT

The aim of this study is to advance the knowledge and understanding of influence of different macrophyte stands on velocity distribution and, consequently, their influence on the flow resistance in the channel. With this aim in view, engineers and biologists have worked together to determine the influence of macrophyte stands on velocity distribution and on generation of turbulent kinetic energy and shear stress. Cross-sectional distribution of longitudinal velocity components and turbulent characteristics of flow velocity, as turbulent kinetic energy (k) and turbulent shear stress ($-\rho\overline{u'w'}$), were systematically analysed in this study, as this may be taken as a measure for the prevailing flow resistance.

The analysed macrophyte stands show different kinds of plant structures and of biomass distribution, like the pillar-type or canopy-type (Wychera *et al.*, 1993). Velocity was measured with different discharges at three different sites of the channel. Combined examinations show clear influences of water plants. A cross section zone with low velocity develops, exactly correlating with the area of the macrophyte stand. This causes a steeper gradient of velocity distribution in the cross section of the channel and an increase of the velocity maximum. Furthermore, turbulent kinetic energy and shear stress distribution show a peak at the boundary between free flow and macrophyte stand. These effects become more significant due to an increasing biomass concentration and also due to plant structures. The analyses have shown that the influence of water plants on velocity distribution and flow resistance depends on structure and biomass distribution of the water plants. The flow patterns, caused by the waterplants in the channel, makes it easier for submerged vegetation to spread from quieter water into swifter water.

KEY-WORDS: Velocity measurements / field measurements / open channel hydraulics / turbulence / flow resistance / macrophytes / flow pattern / habitat / shear stress / biomass analyses

1 INTRODUCTION

Rivers provide habitats for a rich variety of vegetation. As flow and substrate vary, their effects on plants vary as well, and so in one river site there may be many microhabitats each with specific flow and substrate conditions and thus at different vegetation. As soon as the flow pattern of a section of the river allows the development of plant colonies, these colonies alter the flow velocity as well as the substrate, so that they are then able to spread over the whole cross section of the river. Patches of quiet water tend to develop in them, while flow increases between the plants. Plants therefore alter the flow pattern of a stream as well as they are altered by it. Velocity is greater beside macrophyte stands and least downstream of them. This means that different habitats suitable for other plants occur around the larger weed beds. One species can allow another species to develop upstream of it in a site which would otherwise have too fast flow (Haslam, 1978). From an environmental point of view the conservation of vegetation is very important. From the hydraulic point of view, however, the attitude towards the colonisation of a river section by water plants is different, as problems, resulting from this colonisation, may arise in various respects:

- Reduction of cross-sectional flow velocity
- Increase of flow velocity in the cross-sectional area which has been reduced by the vegetation
- Decrease of flow velocity near the macrophyte stand
- Increased sedimentation in the cross-sectional areas with reduced flow velocity

For the recording and better understanding of these interactions between stream patterns and vegetation, the velocity was extensively measured by means of a three-dimensional electromagnetic current meter, together with detailed mappings and biomass investigations of the vegetation in a natural open channel. The present investigation is aimed at presenting and identifying the mean cross sectional velocity distribution (u), the turbulent kinetic energy (k) due to temporal fluctuations of velocity components and turbulent shear stress ($-\rho\overline{u'w'}$) for natural river flow.

2 EQUIPMENT AND DATA COLLECTION

Data were collected by the measuring of velocities, mapping and assay of the biomass of the submerged vegetation.

2.1 Equipment and data collection of velocity

Velocity data were collected using a 3-dimensional electromagnetic current meter (an ALEC ACM-300). The accuracy and basic capabilities for the current meter are presented in table 1.

Table 1: Current meter specifications

Current Meter	Manufacturer	Parameters	Range	Accuracy	Time Constant
ACM-300D	Alec Electronics Co.	u, v, w	0 ~ ± 250 cm/s	± 2 % or 1 cm/s	0.5 s

The sensor has a diameter of 20 mm and was mounted on a traversing mechanism (a catamaran) which carried the current meter and the data logger. The catamaran was secured to a rope bridge and could be positioned with an accuracy of ± 0.02 m. Two anchor sticks located on both side of the probe fixed the catamaran precisely within a cross section of the channel. The probe could be fixed at any point of the cross section, the density of measuring points being raised in transitional areas between quieter and swifter flow (transition of plant body - free flow, transition of bottom -free flow) up to a minimum distance of measuring points of 0.05 m in vertical, and 0.20 m in horizontal direction. Figure 1 shows the profiles with all measuring points. The minimum distance between probe and bottom was limited to 0.03 m.

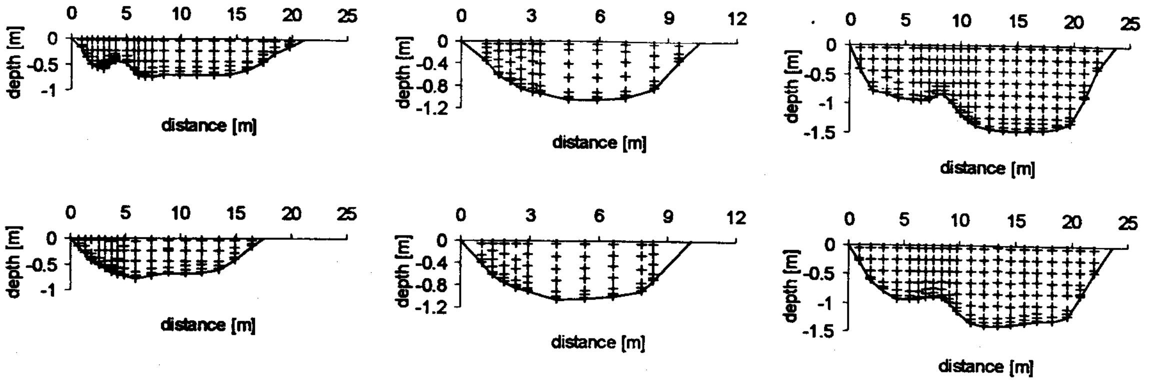

Figure 1: Profiles with all measuring points

For this study, velocity data were collected at the rate of two samples per second to study turbulent velocity fluctuations. The sampling time was 45 up to 180 seconds, depending on the situation in the river. In transition areas with a steep gradient of velocity, thus also with high turbulent kinetic energy rates, the measuring time was longer (180 s) than in cross-sectional areas showing a lower gradient of velocity and lower turbulent kinetic energy rates (45 s). The measuring time was chosen to be shorter in the zones of low momentum exchange, measuring being conducted at a greater number of measuring points in these zones.

2.2 Detailed mapping and assay of biomass

In order to be able to record the influence of the plants on stream patterns in mathematical terms and to compare these data, the structure of submerged macrophyte stands was described in the way of the vertical distribution of biomass. After having measured the flow, the macrophyte stands in the analysed area were measured exactly and drawn in an accurate scale map. After that, biomass was assayed by the stratified-clipping method (Myneni *et al.*, 1989; Wychera and Janauer, 1990), dividing the stand into 10 cm horizontal strata. Ten replicates were made at each site. All plants were cleaned and dried to determine dry matter per unit area, so that, consequently, the entire amount of biomass in the analysed section could be calculated.

3 DESCRIPTION OF THE FIELD SITE

The experiments were conducted in three different parts of the channel (Figure 2) with partly vegetated cross sections during summer (August - October 1995). These cross sections were selected with respect to different structures of plants and growth of the macrophytes and different morphological characteristics.

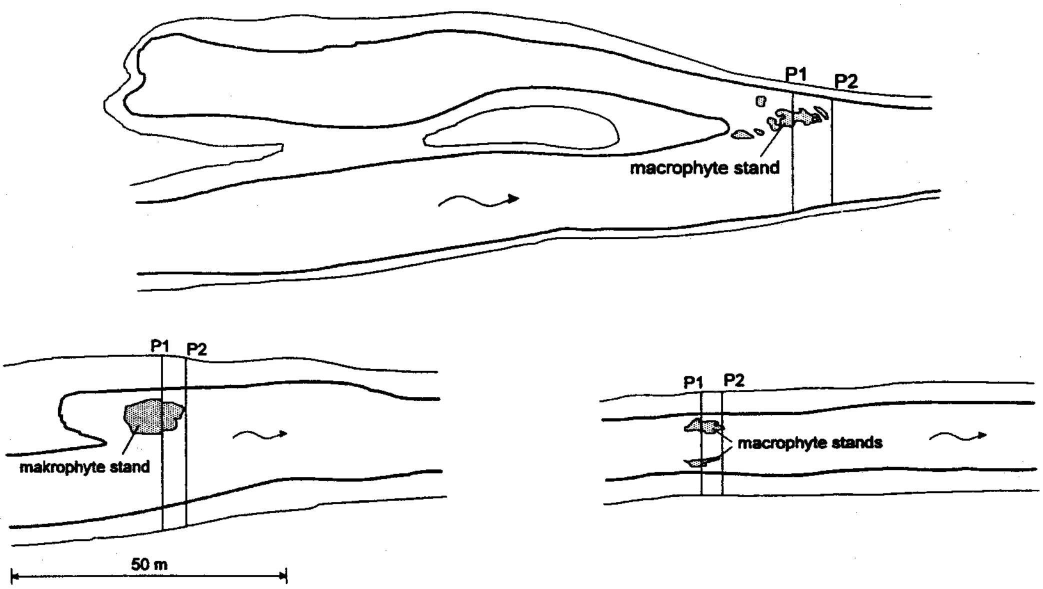

Figure 2: Field site of the analysed channel parts with macrophyte stands

The three analysed sections are:

- cross sections through a small baylet (back water region) with a macrophyte stand of *Myriophyllum spicatum* L.. This section lies in a straight part of the channel with variable widths and depths. Caused by the geometry of the channel, a flow reversal developed in the examined baylet. The macrophytes settled in the quiet water zone of the baylet. The distance between the analysed profiles are about 8.0 m. The first profile goes through the macrophyte stand and the second lies directly behind it.
- cross sections through a channel part with a shallow water region, also with a stand of submerged macrophytes (*Potamogeton lucens* L.). In this part of the channel, there is a sudden widening of the cross section, in the quiet water zone of which macrophytes settled. There are no bends in this part of the channel. The examined cross sections were located as in case 1, that is, in and behind the stand, respectively, at a distance of 4.0 m to each other.
- cross sections in a straight part of the channel, which shows almost constant widths and depths, with a macrophyte stand (*M. spicatum*) at the margin of the channel. The profiles are at a distance of some 1.7 m to each other. Profile 1, again, lies in the macrophyte stand and profile 2 directly behind it.

Table 2 summarises the morphological and hydraulic conditions for the field measurements for each of the analysed channel zones:

Table 2: Hydraulic conditions for the field measurements

Nr.	Channel sections	Profile number	Mean flow depth h [m]	Channel width B [m]	Mean bulk velocity U_m [m/s]	Reynolds number Re (ca. 18°C)	Discharge Q [m^3/s]	Measuring points in the cross section

1	**small baylet**	1	0.54	21.20	0.22	$1.12 \cdot 10^5$	2.47	115
		2	0.55	17.50	0.20	$1.04 \cdot 10^5$	1.96	83
		1 (1994)	0.49	18.70	0.19	$0.88 \cdot 10^5$	1.77	112
2	**shallow**	1	1.07	22.85	0.24	$2.42 \cdot 10^5$	5.83	166
	water region	2	1.02	22.80	0.25	$2.41 \cdot 10^5$	5.74	152
3	**straight**	1	0.76	10.40	0.24	$1.72 \cdot 10^5$	1.87	69
	channel	2	0.77	10.00	0.23	$1.67 \cdot 10^5$	1.78	62

4 THE STRUCTURE OF SUBMERGED MACROPHYTES

The structure of macrophyte stands and individual plants depends very much on environmental factors, like light, flow velocity, water depth and sediment. Therefore, in the examined part of the channel, quite different structures of one macrophyte species, *M. spicatum,* were found (see Figure 3). In zones of quiet water the watermilfoil showed rather equal biomass distribution along the more or less parallel vertical axis of the individual plants in the layers between 0.10 and 0.60 m waterdepth. In the first 0.10 m, however, 61 % of the total biomass were collected. *M. spicatum,* a species normally occurring in quiet water, was found in the analysed area directly in the flow. The structure of the plants, however, was clearly adapted to the altered environmental factors. At approximately 0.50 m water depth the mean length of the plants was 2.30 m. 41 % of it were in the uppermost 10 cm. *M. spicatum* showed longer internodes and less whorls to offer less resistance to flow

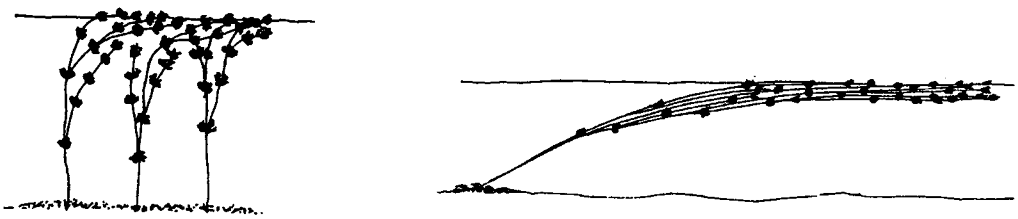

Figure 3: Habitus of *M. spicatum*-stands in the baylet and in the free flow

In addition, *P. lucens*, another submerged species, was analysed (see Figure 4). This species can be assigned to the so-called canopy-type, with a significant concentration of biomass in the top layers (Wychera *et al.*, 1993). 60 % of the total biomass were collected in the first 0.10 m. In the layers between 0.10 and 0.40 m depth, there were only 9 % of the total biomass. *P. lucens* has large submerged leaves and offers a high hydraulic resistance to flow. Therefore, it only grows in areas with very low current.

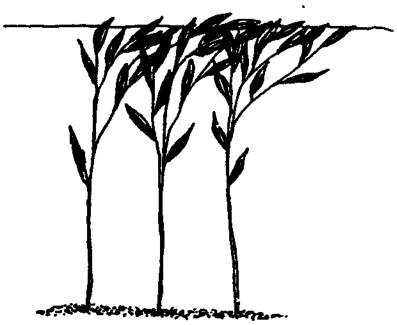

Figure 4: Habitus of *P. lucens*-stand

5 ANALYSED HYDRAULIC PARAMETERS

The data analyses of the velocity measurements were conducted by evaluating the cross-sectional velocity distribution u, turbulent kinetic energy (k) and turbulent shear stress ($-\rho\overline{u'w'}$). For this purpose, horizontal profiles were laid in the analysed part of the channel. Denoting the instantaneous flow velocity into two parts, a time-averaged component and a fluctuating component, the relationships for the three velocity components (u, v, w) can be written:

(1) $$u = \overline{u} + u' \qquad v = \overline{v} + v \qquad w = \overline{w} + w'$$

with $\overline{u}$, $\overline{v}$ and $\overline{w}$ as the time-averaged components and u', v' and w' as fluctuation components.

5.1 Turbulent energy

Turbulent energy

(2) $$k = \frac{1}{2}\left(\overline{u'^2} + \overline{v'^2} + \overline{w'^2}\right)$$

is computed from the fluctuating velocities due to the transfer of momentum (Bhowmik *et al.*, 1995) and, therefore, it may be regarded as a parameter for the hydraulic flow resistance taking effect on the current. The higher the turbulent kinetic energy of each measuring point gets, that is, the more the velocity components differ from the mean value of the velocity time-scale, the greater seems to be the frictional influence taking effect on the current. The theoretical concept (Kölling, 1994; Nezu and Nakagawa, 1984; Nezu, 1993; Nezu *et al.*, 1993; Nezu and Nakagawa, 1993) determining k as evaluative parameter is described as follows:

- In a circular, completely filled flow cross section, ideally, there is an isotropic turbulence.
- In any flow cross section with a geometrical shape differing from the ideal circular shape, the turbulent fluctuation of the velocity components normal to wall and open surface of the water is disturbed, which causes an anisotropy of turbulence.
- Channel bank and channel bed cause a significant alteration of the anisotropy of turbulent fluctuation. Water plants and riparian spinney produce the same effects.

- Thus a system of secondary currents develops, overlying the axial current.
- By turbulent exchange of momentum, the secondary currents for their part influence and alter the axial velocity distribution within the profile to a large extent.

Thus the cross-sectional and the vertical distribution of turbulent fluctuations, expressed by the parameter k, serves as parameter for the flow resistance in the channel.

5.2 Shear stress

For two-dimensional turbulent flow, turbulent shear stress or Reynolds stress is defined as:

$$\tau_{xz} = -\rho\overline{u'w'} \tag{3}$$

The Reynolds stress arises from the correlation of two components of the velocity fluctuation at the same point. A non-zero value of this correlation implies that the two components are not independent of one another (Tritton, 1988). Reynolds stress is mainly generated by momentum exchange of big turbulent elements. The force appearing at the surface normal to Z, that is, shear stress (Prandtl, 1993). Collected Data were used to compute the turbulent shear stress, $-\rho\overline{u'w'}$.

6 RESULTS

In the following two different macrophyte species (*P. lucens* and *M. spicatum*) are compared to one another, on the one hand with regard to their flow resistance in the stream. On the other hand it is shown that in completely different stream pattern one and the same species (*M. spicatum*) finds conditions which allow further spreading of the vegetation. This investigation is based upon the measuring of velocity in three different sections of the channel.

6.1 Comparison of a *P. lucens*-stand to a *M. spicatum*-stand

When comparing a *P. lucens*-stand to a *M. spicatum*-stand, especially the different plant structures and the different flow resistance are analysed.

6.1.1 P. lucens-stand in a shallow water region

As shown in table 2, the measuring of velocity was carried out with a flow of approximately 6.0 m³/s. The profiles were located about 15.0 m downstream of a sudden cross-sectional widening, allowing the development of a slight reverse flow at the left channel side. The *P. lucens*-stand came to a width of 6.5 m and a longitudinal extension of 12.0 m. Characteristic of the plant structure in this stand are the big leaves and firm, inflexible stalks (see figure 4). Thus the stand causes high resistance to flow.

Figure 5 shows the alteration of the velocity gradient, caused by the reduction of biomass in the longitudinal course of the channel. As the first profile goes through the stand, it shows a steeper gradient than the second profile. This steep gradient is due to an increased anisotropy of turbulence (impairment of the momentum exchange, caused by vegetation) in the transitional area between main stream and stream in the plant stand, as a consequence of the high biomass concentration (62.7 g/m² averaged over depth). At the same time this observation can be proved by a notation of shear stress and turbulent kinetic energy (see figure 5). The maxima of shear stress lie in the area of highest biomass concentration, yielding values around 2.0 N/m². The maximum of turbulent kinetic energy (0.02) referring to U_{max}^2 is also found in the area of highest biomass concentration. The notation of shear stress and turbulent kinetic energy for different depths of the channel was chosen because of the biomass variable in flow depth. The highest concentration of biomass is found in the upper 30 cm (117.8 g/m² in the upper 30 cm and 47.7 g/m² in the lower 120 cm). The maxima of the distribution of shear stress and turbulent kinetic energy also lie within the first 30 cm. The Y-axis (ordinate) presents the right margin of the weed bed (in flow direction).

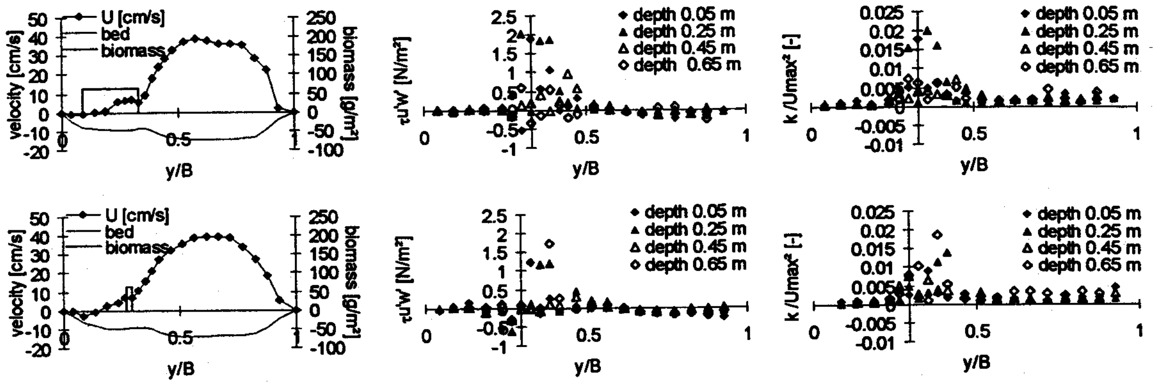

Figure 5: Stream velocity distribution with biomass concentration (both averaged over depth), distribution of turbulent kinetic energy and distribution of shear stress in four different depths of the channel (*P. lucens*)

6.1.2 M. spicatum-stand in the baylet

In this stand the measuring was conducted with a discharge of 2.0 m³/s. The first profile was situated at about 11.5 m downstream of the baylet opening, directly in the plant stand, and the second profile was 8.0 m downstream the first one, at the end of the plant stand. The *M. spicatum*-stand came to a length of 12.0 m and to a maximum of cross dimension of 4.5 m. The plant structure of this stand is characterised by bushy growth and flexible leaves (see figure 3). Therefore, in comparison with *P. lucens*-stands, there is less resistance to flow. Again, the preferred stream area is the zone of quieter flow (see figure 6).

Figure 6 shows the velocity distribution averaged over depth of the two analysed profile and the measured biomass concentration also averaged over depth. Looking at the velocity gradient, the distribution of shear stress and kinetic turbulent energy, there are essentially similar results as those described of *P. lucens*-stand: The gradient of velocity distribution is flatter in the cross section through the plant stand than in the cross section downstream of the stand. The distribution of shear stress and turbulent kinetic energy referring to U_{max}^2, too, show higher values in the upper 25 cm, that is in the areas of higher biomass concentration. The values come to 1.0 N/m² (distribution of shear stress) and 0.01 (turbulent kinetic energy referring to U_{max}^2). The highest concentration of biomass was found in the upper 20 cm (56.1 g/m²), whereas in the lower 50 cm of total flow depth 8.3 g/m² were collected. In contrast to the results concerning *P. lucens*-stands, the maxima of shear stress and turbulent kinetic energy are lower, despite a higher velocity maximum (43.7 cm/s, compared to 39.7 cm/s in the *P. lucens*-stand) and a greater difference in velocity between the zone of quieter flow and the main stream. This might be explained by the plant structure which causes less resistance to flow.

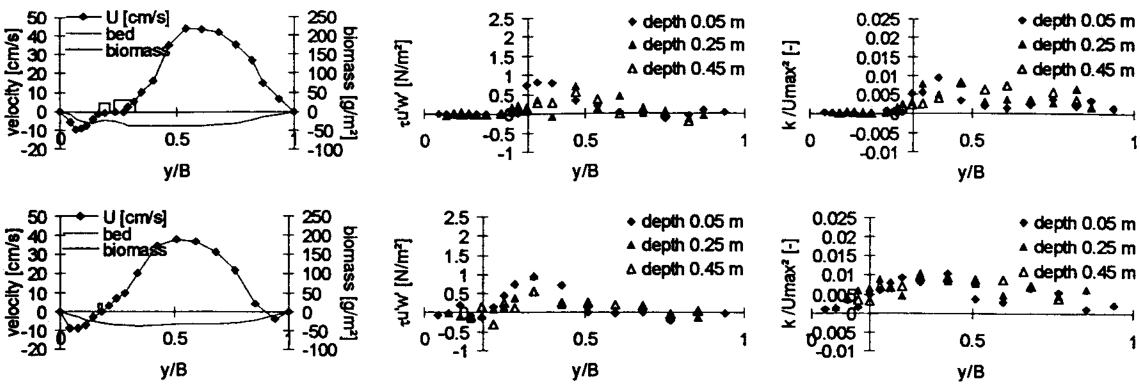

Figure 6: Stream velocity distribution with biomass concentration (both averaged over depth), distribution of turbulent kinetic energy and distribution of shear stress in three different depths of the channel (*M. spicatum*)

6.2 Comparison of two *M. spicatum*-stands

Comparing the *M. spicatum*-stands in the baylet and those in the straight part of the channel, the emphasis is put on the plant structure adapted to the current and the different types of habitats resulting from it.

6.2.1 M. spicatum-stand in the straight section of the channel

In this stand the measuring of velocity was conducted at a flow of approximately 2.0 m³/s. The analysed profiles were laid through and behind the stand at a distance of 1.7 m. The *M. spicatum*-stand reached a longitudinal dimension of 5.0 m and a width of approximately 2.5 m. Longer internodes and therefore less whorls (see figure 3) provide better adaptation of the plant body to the stream conditions.

Figure 7 shows the two analysed cross-sections through and behind the *M. spicatum*-stand, respectively, with the velocity distribution averaged over depth and the values of biomass concentration. Again, there is a visible alteration of the velocity gradient, caused by the reduction of biomass in the longitudinal course of the channel. Due to the vegetation, the profile is clearly divided into areas of swifter flow and quieter flow. The biomass concentration in the uppermost 30 cm of total flow depth (54.7 g/m²) divides the stream pattern horizontally as well as vertically. In the first 5 - 10 cm, flow velocity is reduced to almost 0 cm/s, and, downwards, it is gradually accelerated up to 10 - 15 cm/s (see figure 8). Thus, a first unstructured cross section yielded a markedly increased variety of biosphere. Despite a high biomass concentration in the surface area, the distribution of shear stress and turbulent kinetic energy have no marked maxima, but very low similar values across the channel. The Data indicate a rather low resistance to flow.

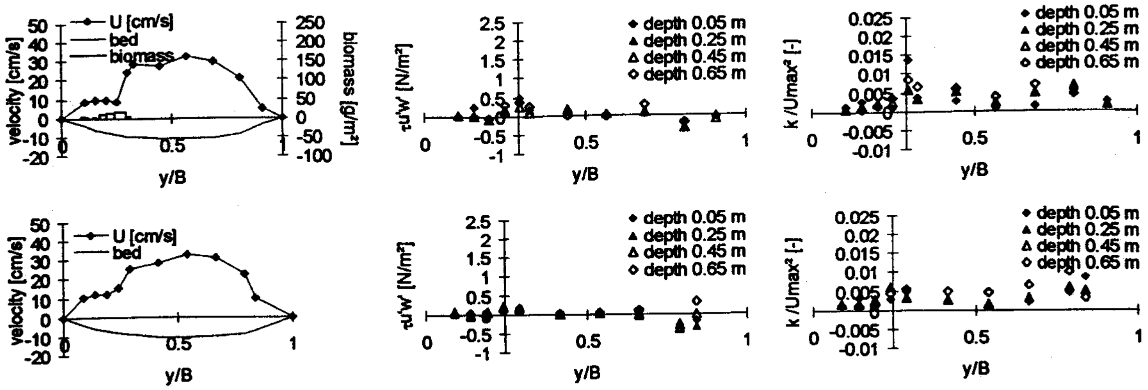

Figure 7: Stream velocity distribution with biomass concentration (both averaged over depth), distribution of turbulent kinetic energy and distribution of shear stress in four different depths of the channel (*M. spicatum*)

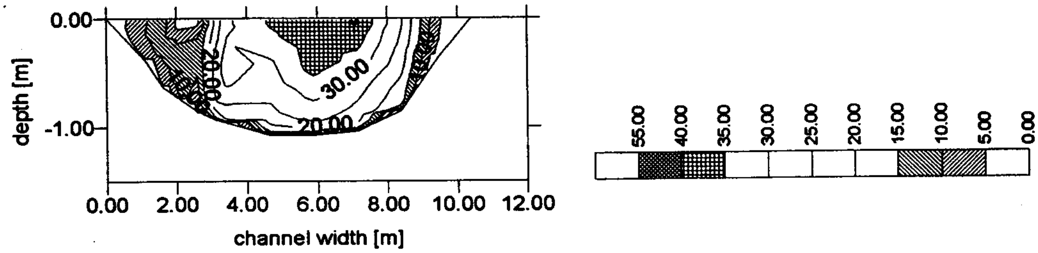

Figure 8: Cross section through a stand of *M. spicatum* in the straight channel part

6.2.2 M. spicatum-stand in the baylet

The growth pattern of this *M. spicatum*-stand is in many respects different from the stand in the open stream, described above: The main internodes are markedly shorter and the plants have lots of branches with short internodes. The whorls are concentrated at the top of the plants. The structure of the plant in this case rather resembles the species existing in quiet water. Figure 9 shows the velocity distribution of the same cross section with the measured flow values of two subsequent years, with and without vegetation. It clearly indicates that in this case the water plants settled exactly in the transitional area between reverse flow and main stream, that is in the cross-sectional area with the least velocity. The geometry of the channel morphology provided very good stream conditions for the habitat of the stagnophil species *M. spicatum*. The emerging water plants intensified these stream conditions and extended the quiet water zone. Thus, the quiet water zone in the cross section became larger, which again altered the habitat and allowed a greater variety of plants.

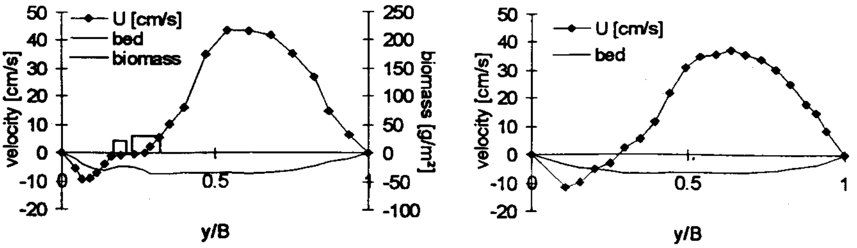

Figure 9: Comparison of the velocity distribution of cross section 1 through the *M. spicatum*-stand in the baylet and the same profile without the macrophyte stand (Measuring: September 1994)

7 CONCLUSIONS

By coordinated measuring of the velocities and detailed mapping of the macrophyte stands the interaction of stream pattern and aquatic vegetation became clear. On the one hand, it could be observed that by particular morphological conditions in the channel stream pattern arises as new habitats, which are preferably colonised by macrophyte stands. The examples 1 and 2 (*P. lucens*- and *M. spicatum*-stand) clearly show that the plants settled exactly in the area of slow flow between reverse flow and main stream, and that emerging vegetation intensified the previous stream pattern. By comparing the values measured before the colonisation by macrophyte stands and during the maximum extension of the stands, the measuring being conducted within and downstream of the stands, the following alterations could be observed: The gradient of the velocity distribution in the transitional area as well as the maximum of the velocity distribution rose, and the transitional area between negative and positive flow rates became wider and its dimensions were adapted to the size of the macrophyte stands. On the other hand, in channel parts with more unfavourable stream conditions (e.g. swifter flow in straight, narrow sections), as shown in example 3, the settlement of aquatic vegetation seems to be largely independent of the existing stream pattern. In this case submerged macrophytes settled at the margin of an unstructured profile without a zone of quieter flow. By measuring the flow velocity in the macrophyte stands and downstream of them, it became clear that there was a pattern of swifter flow and quieter flow, which divided the water in the channel into different habitats.

Furthermore, the analyses showed that the habitus of the plants (size of the leaves, branch structure) is adapted to the respective site, thus causing less resistance to flow in areas of swifter flow. Because of its big and stiff leaves a *P. lucens*-stand causes a higher resistance to flow, which is expressed by the distribution of shear stress and turbulent kinetic energy. Therefore, altered stream conditions have more influence on this stand than on a macrophyte stand with flexible, small leaves, such as the *M. spicatum*-stand described in example 3.

8 ACKNOWLEDGEMENTS

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9 NOTATIONS

u, v, w Velocity components

u', v', w'	Fluctuating velocity components
$\overline{u}$, $\overline{v}$, $\overline{w}$	Time-average velocity components
$-\rho\overline{u'w'}$	Turbulent shear stress
x	Longitudinal coordinate
y	Lateral coordinate
z	vertical coordinate
k	Turbulent kinetic energy
τ_{xz}	Turbulent shear stress

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LABORATORY INVESTIGATIONS OF NEAR-BED HYDRAULICS IN A RECONSTRUCTED COBBLE-BED STREAM SECTION

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ABSTRACT

Extensive measurements of near-bed hydraulics were made over a laboratory reconstruction of a section (1.5m x 0.5m) of a mountain cobble-bed stream using laser-Doppler velocimetry. Using these data four hydraulically different habitat types are identified in the above-bed benthic zone. The identified habitat types include: the exposed tops of roughness elements (TOPS), the sheltered lees of roughness elements (LEES), the exposed faces of roughness elements (FACE), and the partly sheltered areas mid-way between roughness elements (MIDS).

The four hydraulic habitat types are discussed with reference to the flow classification scheme of Davis and Barmuta (1989) (as clarified by Young (1992)), and with reference to the alternative classification of Barmuta (1994). The four categories of hydraulically rough flow recognised by the former system are: chaotic, wake interference, isolated roughness, and skimming flow. The two categories of flow recognised by the latter system (found between bed roughness elements) are cavity flow and diverse wake flow. It is demonstrated that the latter two flow types are not greatly different from the former classification. It is also shown that in skimming flow (and cavity flow), only LEES and TOPS will occur. In the other flow types, all four hydraulic habitat types will occur.

There is therefore a lower diversity of hydraulic habitats in skimming flow than in wake interference, isolated roughness or chaotic flow. Similarly, there is a lower diversity of hydraulic habitats in cavity flow compared to diverse wake flow. In addition, it is speculated that in skimming flow (and cavity flow) fluid exchange between the main flow and the flow between roughness elements is much less compared to other flow types. These differences are expected to affect both the density and diversity of benthic species occurring in these flow types. This expectation is qualitatively supported by the differences in macroinvertebrate species diversity and density in cavity flow and diverse wake flow reported by Barmuta (1994).

Analyses of the velocity data show significant differences between all four hydraulic habitat types. The average velocity in the most sheltered habitat type was 4% of the mean near-crest flow velocity, or just 1.7% of the mainstream mean velocity. In the partly sheltered habitat, the average flow velocity was 33% of the mean near-crest flow velocity, or 14% of the mean mainstream flow velocity. It is concluded that velocity measurements made near the crests of the bed roughness, do not characterise the flow conditions between roughness elements in cobble bed streams.

It is concluded that only one of the flow type thresholds has important ecological implications. This is the threshold which represents the change from skimming to wake interference flow, and the change from cavity to diverse wake flow. This threshold is relatively easily identified in the field, and should be considered when sampling zoobenthos.

KEY-WORDS: near-bed flow / stream hydraulics / benthic habitats / laboratory modelling

INTRODUCTION

Stream ecologists have for some time devoted considerable effort to researching the factors affecting the distributional patterns of benthic invertebrates (e.g. Percival and Whitehead, 1929; Edington, 1968; Rabeni and Minshall, 1977; Hearnden and Pearson, 1991). Both biotic and abiotic factors are known to influence the abundance and distribution of benthic fauna.

Because abiotic factors have been shown to influence the microdistribution of benthic invertebrates in so many studies, it is often assumed that it is these factors that are critical (Hart, 1983). In particular, many studies have reported the influence of flow velocity and bed substrate on microdistribution (e.g. Minshall and Minshall, 1977; Rabeni and Minshall, 1977; Hearnden and Pearson, 1991). Both Scott (1958) and Allen (1959) considered velocity as the most important parameter in determining distributional patterns of benthic invertebrates. Recent studies have begun to investigate whether or not biotic constraints prevent species from inhabiting unoccupied areas with tolerable physical conditions. For example, Pecarsky *et al.* (1990) present data which support the 'harsh-benign' hypothesis. This hypothesis suggests the degree of harshness to predators is a good predictor of predator impact on preferred prey species. In some cases therefore, abiotic factors appear to dominate over biotic factors in determining microdistribution.

The importance of flow velocity to benthic invertebrates relates to physical stress, feeding and respiration. The magnitude of the flow velocity, together with the body shape, determine the drag force which must be withstood. In addition, the rate of change of flow velocity, together with body size (volume) determine the acceleration reaction forces which must be withstood (Koehl, 1984). Increased velocities increase the exchange rate between the organism and the surrounding water, thereby promoting respiration and the acquisition of water-bourne food (Giger, 1973).

While ecologists have recognised the importance of flow velocity for some time (e.g. Percival and Whitehead, 1929; Edington, 1968), understanding was limited by a paucity of reliable velocity data describing benthic flow regimes (Davis, 1986). In many ecological studies mean flow velocities have been used to describe near-bed conditions, which as shown by Davis (1986) is inappropriate. There is however, an increasing awareness amongst ecologists of hydraulic theory and its relevance to habitat description, in particular the need to accurately describe near-bed conditions. For example Nowell and Jumars (1984) and Davis (1986) both give detailed descriptions of flow microenvironments with special emphasis on boundary layer concepts, and Davis and Barmuta (1989) explain the relevant parameters for describing and classifying both the mean flow and the near-bed flow.

Statzner *et al.* (1988) reviewed the techniques used to describe flow conditions in benthic stream ecology, and concluded that the simple hydraulic parameters of mean flow velocity and depth are not particularly useful in ecological studies, and that the more complex variables of Froude number, Reynolds' number, viscous sub-layer thickness and roughness Reynolds' number should be used. The first two of these variables provide a description of the mean flow, while the remaining two describe the near-bed flow; although as pointed out by Nowell and Jumars (1984) rough turbulent flow is usual in streams and hence there is no viscous sub-layer. The roughness Reynolds' number depends on the bed shear velocity, and is therefore the most relevant of these parameters for assessing the hydraulics of benthic habitats.

Assessing the bed shear velocity in the field at a spatial resolution appropriate to zoobenthos is difficult. Statzner and Muller (1989) describe a method for quantifying near-bed flow conditions using a standard set of uniform sized hemispheres of different densities. This is a useful technique for estimating the near-bed flow conditions at the crest of the bed roughness elements, and has lead to a rapid increase in field assessment of bed shear velocity (eg. Pecarsky *et al.* (1990), Dobson *et al.* (1992), Petts *et al.* (1993), Lancaster and Hildrew (1993), Gore *et al.* (1994), and Dittrich and Schmedtje (1995)). Some of these studies have focussed on the relationships between bed shear velocity and macroinvertebrate abundance.

However, estimating flow conditions between bed roughness elements is not practical in the field. In cobble bed streams especially, the areas between bed roughness elements are an important habitat for zoobenthos, and it would therefore be useful to be able to infer something about the flow conditions between bed roughness from a knowledge of the bed configuration and either the mainstream flow conditions or the near-crest flow conditions.

Davis and Barmuta (1989) identified four categories of hydraulically rough flow in streams namely: skimming flow, wake interference flow, isolated roughness flow and chaotic flow. The definitions of these flow types stem from a consideration of the energy dissipation characteristics of different bed configurations (Morris, 1955). They depend on the nature of the turbulence structure of the flow near the bed, and in particular how the wakes behind bed roughness elements interact (or not) with downstream elements. They therefore help describe the nature of the flow both at the crests of the roughness elements and in the wake zones between roughness elements.

Young (1992) clarified the threshold criteria defining these flow types, and provided descriptions of each flow type. Young (1992) also used the amended scheme to classify the flows in the experiments reported in this paper. It was concluded that the flow was predominantly wake interference flow, but with patches of the other three flow types also identifiable.

For this system of classification of flow regimes to be useful, it is necessary to demonstrate significant differences in the near-bed flow velocities within each flow type. In this paper near-bed velocity measurements taken in the laboratory flume are used to investigate the velocity differences between these different flow types.

METHODS

A brief description of the field and laboratory procedures is given below. For a more detailed account see Young (1992).

A 1.5 m x 0.5 m section (lengthwise downstream) of cobble stream bed was carefully measured in the field. The field site was a relatively straight section of the Taggerty River near Marysville, Victoria, in south-eastern Australia (37^{o} 30' S, $145^{o}47$' E). At the site the Taggerty River is a 4th order stream 420 m above sea level, with a catchment area of 64 km^2. The reach is 7.5 m wide and at the moderate flows present during sampling was 0.2-0.25 m deep. Average flow velocity in the field was 0.36 ms^{-1}, and mean bed slope for the reach was 0.006.

In the laboratory the section of stream bed was reconstructed in a glass-walled flume. Comparisons of roughness parameters of the bed in the laboratory with those of the original stream bed indicated that mean roughness spacing was correctly reproduced, but that there had been a significant reduction (40%) in the mean roughness height. However, the laboratory bed was still considered to be sufficiently realistic for investigating near-bed flow patterns. In the flume the bed slope was set to 0.006; the flow depth was 0.2 m and the average velocity was 0.35 ms^{-1}.

Detailed measurements of the near-bed velocities in steady flow conditions in the flume were made using a two-component laser-Doppler velocimeter. The laser probe was mounted vertically above the flume, with beams entering the water through the base of a small perspex view-box held level at the water surface. The probe supports were fixed to a traversing table which allowed manual positioning of the probe in all three coordinate directions. A horizontal measurement grid was defined, its limits being 0.05 m inside each flume wall, and 0.05 m in from each end of the reconstructed bed. The grid consisted of 15 points by 56 points, all at 25.2 mm spacings, thus giving a total of 840 horizontal measurement positions. At each grid location measurements were taken at 5 mm above the bed surface. Measurements were made of both the downstream and cross-stream components of the velocity, thus also allowing calculation of the magnitude and direction of the combined velocity vector. Between one and four thousand individual measurements of each component were made at each location within a one minute sampling interval. The measurements therefore describe the temporal variability as well as the spatial variability of near-bed velocities.

RESULTS AND DISCUSSION

Hydraulic Habitat Types

Using the flow areas as classified by Young (1992) for these experiments, an initial comparison was made of the near-bed velocities in the four different flow types. This revealed that a wide range of velocities was experienced in all flow types, and that the mean near-bed velocities were not greatly different between flow types. This was attributed to the fact that within each of these flow types there are a number of different flow zones which are of relevance at the scale of zoobenthos. For instance, the fast flows zones on the tops of roughness elements occur in all these flows types, and are very different to the sheltered wake zones directly behind roughness elements. It is the nature of the flows in the wake zones, that is between the roughness elements, that are expected to differ between flow types.

A consideration of a bed of block roughness elements, lead to the proposition that four hydraulically different habitats exist in the above-bed benthic zone (Figure 1). These are:

- the exposed tops to roughness elements ('TOPS')
- the sheltered lees in the wake of roughness elements ('LEES')
- the upstream faces of roughness elements which are exposed to the flow ('FACE')
- the areas mid-way between a sheltered and an exposed face ('MIDS')

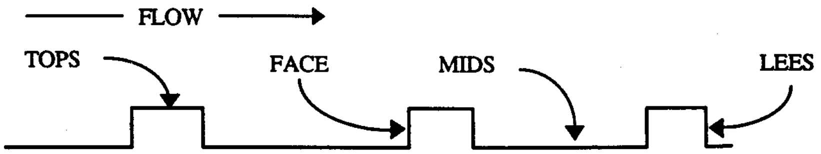

Figure 1: Conceptual definition of hydraulic habitats types

This scheme focuses at a smaller scale than the scheme of Davis and Barmuta (1989), and is therefore expected to be more directly relevant to the prediction of zoobenthos distribution. It is apparent that the near-bed flow velocities in each of these habitat types should be independent of the flow regime type, but that not all these habitat types will be present in each flow regime type. For instance, where roughness elements are close together, the upstream faces are not exposed to the flow, but are sheltered by the upstream element. The point at which an upstream face is first considered to be exposed, is when the main flow no longer skims over the crests, but rather the wake from the upstream element impinges on the downstream element. This corresponds to the definition of the threshold between skimming and wake interference flow (Young, 1992). That is, the FACE habitat type will only occur where the roughness elements are separated at least to the extent where the wake from the upstream element impinges on a downstream element. The actual roughness spacing represented by this threshold is not well defined, but for stream bed roughness is has been suggested that it is likely to occur at the point where the groove length is equal to the roughness height. The FACE habitat is also considered to occur in isolated roughness flow, where although the wake from the upstream element is dissipated before the next element is reached, the roughness elements are separated to the extent that no sheltering of the downstream element occurs.

The MIDS habitat is not considered to occur until the roughness spacing is significantly greater than the skimming flow-wake interference flow threshold. This threshold is the point where the FACE habitat first occurs, and until roughness elements are much further apart the grooves will only exhibit LEES and FACE type habitats. The MIDS habitat is therefore expected in isolated roughness flow, but also in wake interference flow (except perhaps near the lower limit of roughness spacing).

It is apparent than in this scheme all the hydraulic habitat types can be expected in both wake interference and isolated roughness flow. As postulated by Young (1992), the threshold between these two flow types is unlikely to have much biological relevance. Similarly, all the habitat types are expected to occur in chaotic flow, although under these flow conditions the flow structure is far more complex and unstable, and the usual notions of 'exposed upstream faces' and 'sheltered wake zones' are not very applicable. In chaotic flow, the entire flow is affected by the geometry of the bed, and the prediction of fast and slow water zones is extremely difficult.

In skimming flow, neither the FACE nor the MIDS habitat types are expected. Skimming flow is therefore characterised by just two habitat types: (i) the exposed tops of roughness elements, a fast water zone; and (ii) the sheltered lees behind roughness elements, a slow water zone. Skimming flow is thus characterised by a lower diversity of hydraulic habitat types than the other three flow types. The threshold which distinguishes between skimming and wake interference flow is therefore expected to be biologically significant.

Biological Significance

Barmuta (1994) reported a number of experiments, laboratory and field, investigating the effects of near-bed flow regimes on benthic colonisation and distribution. He focussed on the nature of the flow between the roughness elements by defining two types of flow in these regions namely: cavity flow and diverse wake flow. In essence this recognises that the exposed tops of roughness elements are a separate habitat type, but that different habitat types can be found between roughness elements due to the behaviour of the wakes. The description Barmuta (1994) gives of the threshold distinguishing these two flow types, is identical to the (albeit poorly defined) threshold between skimming and wake interference flow. As was concluded above for skimming flow, Barmuta (1994) concluded that cavity flows were characterised by a less diverse range of flow patterns.

The possible biological implications of the differences between skimming flow and the three other flow types (or between cavity and diverse wake flow) are several, although at this stage they are preliminary and speculative. The smaller range of flow types, and thus less velocity-related niches, suggests that a lower species diversity would be supported in skimming flows. This hypothesis is supported by the colonisation results of Barmuta (1994) who observed higher species diversity in diverse wake flow compared to cavity flow. In skimming flow, the level of exchange of fluid (and thus of dissolved and suspended materials) to and from the flow between roughness elements and the mainstream flow will be less than for the other flow types. This is likely to impact on the available food source for some zoobenthos, particularly filter feeders. Conversely, the low velocities between elements in skimming flow, should increase the deposition of suspended material that does enter this zone, to the advantage of detritivores. From these two hypotheses, it is difficult to predict whether the density of zoobenthos will be higher between roughness elements in skimming flow or in other flow types. Clearly, different feeding groups will be affected differently. Barmuta (1994) reported a higher density of zoobenthos in cavity flows than in diverse wake flows. He reported a strong response from grazers and browsers; however, filter feeders 'were too scarce too analyse properly'. It may be that the reduced drag and acceleration reaction forces experienced in the flows between roughness elements in skimming flow, are an important factor in determining total species density.

The identification of four different hydraulic habitats and their dependence on the nature of the bed roughness, together with the preliminary results of Barmuta (1994), suggest that the threshold between skimming and wake interference has ecological relevance. When sampling zoobenthos, classification of the near-bed flow regime is advised as this will provide additional information to help explain differences in species density and diversity between sites.

Velocity Measurements

To determine the differences in flow velocities between the four hydraulic habitat types descriptive statistics of the measurements from each habitat type were calculated. These are summarised in Figure 2 as conventional box-plots.

These box plots were produced using the *S Plus* statistical analysis software (Statistical Sciences, 1993). The central point of the box is the median, the box itself represents the inter-quartile range (IQR), and the 'fences' are at 1.5 x IQR (to the nearest data point) or to the most extreme data value, measured from the edge of the box.

The six data sets analyses and presented in Figure 2 are explained below:

(i) U_{mean}: this is the data set of sample means for each measurement location of the downstream component of the flow velocity (in ms^{-1}). Negative values imply local upstream flow. The spread of these data represents the differences between locations of the mean flow conditions in this component direction.
(ii) V_{mean}: as for Umean, but for the cross-stream velocity component.
(iii) UV_{mag}: this is the data set of the vector sums of the two velocity components at each location. By definition, these values are always positive.
(iv) UV_{angle}: this is the data set of the direction of the net velocity vector. Zero degrees represents directly downstream flow (with $V_{mean} = 0$), ± 180 degrees represents upstream flow (with $V_{mean} = 0$).
(v) U_{sd}: this is the data set of the sample standard deviations from each measurement location for the downstream component of the flow velocity. They provide an indication of temporal variability, or the turbulence level.
(vi) V_{sd}: as for U_{sd}, but for the cross-stream velocity component.

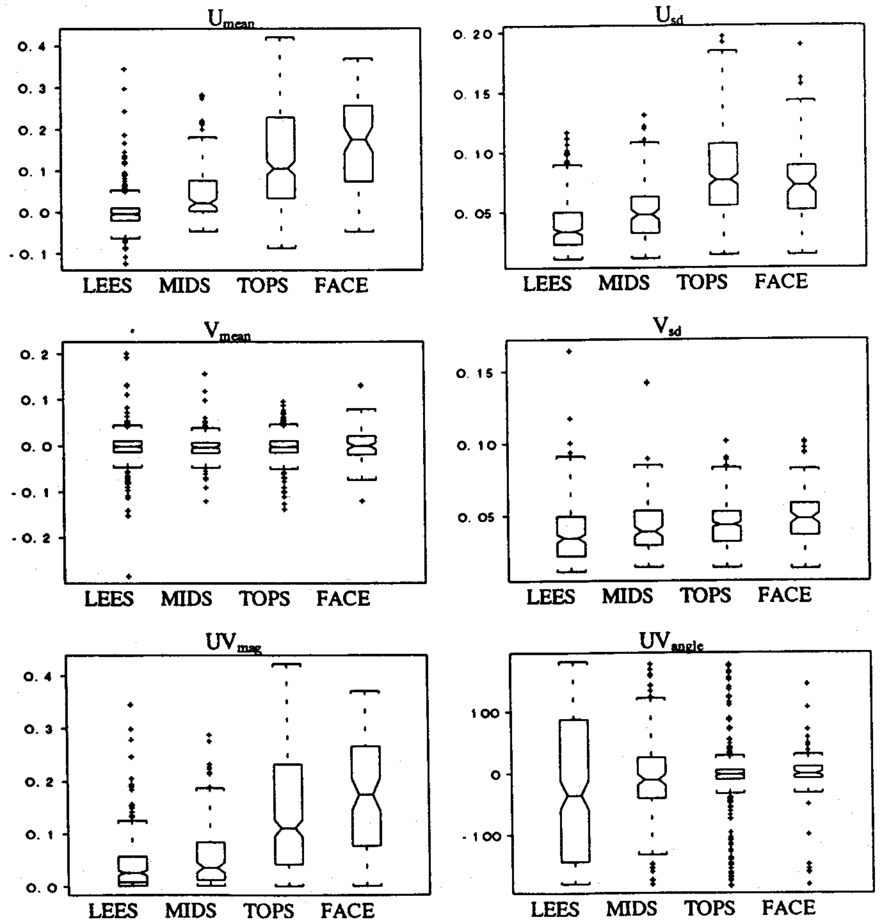

Figure 2: Box plots of velocity measurements by hydraulic habitat type

Given the accuracy of the measuring equipment and the large individual sample sizes (1000-4000 measurements), every value in the data sets can be considered as 'real'. This allows strong inferences to be made about near-bed flows in each habitat type, since every statistical 'outlier' has a physical meaning. Furthermore, the size of the data sets (91-367) allows their means and variances to be accepted as having been essentially determined 'without error'.

The box plots in Figure 2 reveal several differences between flow conditions in these four hydraulic habitat types. As expected, LEES and MIDS generally exhibit significantly lower downstream (U_{mean}) and combined vector (UV_{mag}) velocities than FACE and TOPS. However, positive and negative extreme values are to be found in all habitat types. Note that these extremes are still average conditions at these particular locations. The cross-stream velocities (V_{mean}) generally do not appear to be significantly different between habitat types. However, the highest extreme values (means) appear to be found in LEES.

The temporal variability in the downstream direction (U_{sd}) is also generally lower in LEES and MIDS compared to FACE and TOPS. In the cross-stream direction, the temporal variability (V_{sd}) does not appear to be significantly different between habitat types, although it is apparent that the highest variabilities occur in the sheltered habitats (LEES and MIDS), and that in these habitats the magnitude of cross-stream variability is at least as high if not higher than the variability of the downstream component.

Although the LEES U_{mean} data are more tightly clustered around the median than the MIDS U_{mean} data, higher extremes (both negative and positive) are experienced in LEES. Again, note that these 'extremes' are still average conditions for these particular locations. Although all V_{mean} data are tightly clustered around zero, there are locations where the average condition has a high cross-stream velocity. The largest cross-stream velocities are found in LEES.

Both the exposed habitat types show the UV_{angle} data tightly clustered around zero. In the sheltered habitat types the average vector direction is more variable, especially in LEES. Extreme values (means) can be found in all habitat types.

The box plots mainly demonstrate the differences between the sheltered and exposed habitat types. These differences are generally as expected. Of greater interest is whether differences exist between LEES and MIDS, and between FACE and TOPS; that is, can four different hydraulic habitats types be identified, or are there only two. To determine whether any such differences exist, the *t*-test has been used to test pairs of means from the six data sets.

Table 1 presents the *t* statistic for paired means, together with sample sizes, means and standard errors.

Table 1: means, standard errors and *t* statistic for paired means

Type	Statistic	U_{mean}	U_{sd}	V_{mean}	V_{sd}	UV_{mag}	UV_{angle}
LEES	mean	0.0059	0.0393	-0.0028	0.0381	0.0448	-26.7
	S.E.	0.0035	0.0014	0.0027	0.0014	0.0034	7.6
MIDS	mean	0.0494	0.0517	-0.0028	0.0426	0.0616	-7.0
	S.E.	0.0061	0.0016	0.0022	0.0014	0.0044	6.3
	t statistic	*6.19*	*4.61*	0.00	1.93	*2.46*	*2.00*
FACE	mean	0.1636	0.0750	0.0027	0.0502	0.1713	-4.7
	S.E.	0.0115	0.0020	0.0026	0.0013	0.0066	4.9
TOPS	mean	0.1318	0.0825	-0.0019	0.0431	0.1411	-5.8
	S.E.	0.0063	0.0025	0.0019	0.0010	0.0071	2.8
	t statistic	*2.44*	1.92	1.03	*3.09*	*2.42*	0.21

At the 95% confidence level means can be said to be different when $t \geq \pm 1.96$. The significant *t* values in Table 1 have been italicised. This test indicates that between LEES and MIDS there are significant differences between the mean values for all data sets except for V_{mean} and V_{sd}. Between FACE and TOPS there are significant differences between the means of the U_{mean}, V_{sd} and UV_{mag} data sets. This analysis allows a qualitative description of the differences between the LEES and MIDS habitat types, and between the FACE and TOPS habitat types.

Typically, LEES are characterised by lower velocities in the downstream direction (U_{mean}) than MIDS. Not only is mean value of the LEES U_{mean} data significantly lower that that of MIDS, but there is very little overlap of the inter-quartile ranges. As noted earlier however, LEES exhibits higher extreme U_{mean} values (positive and negative) than MIDS. The mean temporal variability in the downstream direction is higher in MIDS than in LEES, although the overall spread of the distributions are similar. In both LEES and MIDS the mean of cross-stream velocity (V_{mean}) data sets are not significantly different to zero. The spread of these distributions are similar. The *t* statistic indicates that the mean temporal variability in this cross-stream direction is not significantly different between MIDS and LEES. As suggested by the higher U_{mean} values, the mean UV_{mag} is higher in MIDS than in LEES. The mean UV_{angle} is significantly different between LEES and MIDS.

The mean downstream velocity (U_{mean}) in TOPS is significantly lower than in FACE. Although the spread of the two (U_{mean}) distributions is similar, higher extreme values (positive and negative) are found in TOPS. In this downstream direction the mean temporal variability is not significantly different between FACE and TOPS. Once again, the average V_{mean} values are not significantly different from zero in either data set, although the distribution spread is less in TOPS than FACE. In this cross-stream direction the *t* statistic shows that temporal variability is higher on average in FACE than TOPS, although Figure 2 shows a large overlap of the inter-quartile ranges and a similar spread overall. As suggested by the higher U_{mean} values, the mean UV_{mag} is higher in FACE than in TOPS. The mean UV_{angle} is not significantly different between FACE and TOPS.

<u>Interpretation of Hydraulic Habitat Types</u>

The analyses of velocity measurements have identified that both sheltered and exposed hydraulic habitats exist in the benthic zone. Only velocities above or near the crests of roughness elements can be simply measured in the field. Dittrich and Schmedtje (1995) recognise three regions of a vertical velocity profile, including two 'roughness sub-layers'. They assert that the calibrated hemispheres of Statzner and Muller (1989) will characterise velocities in the upper sub-layer well. They also assert that 'because the mean velocities only vary slightly with water depth' in the sub-layer regions, measurements made here will characterise the near-bottom velocities well. However, the data presented here indicate that at least for cobble-bedded streams, the velocities between roughness elements are very different from those at or near the crests. Furthermore, these data show that velocities are not uniformly low in these sheltered regions, but vary considerably both spatially and temporally, and depending on the bed configuration, different sheltered habitat types (LEES and MIDS) can be identified.

Table 3 shows the downstream velocities and temporal variabilities (U_{sd}) are a percent of the mean mainstream flow velocity. This shows that in the near-bed zone, average velocities vary between habitat types from less than 2%, to nearly 50% of the mainstream flow velocity.

Table 3: Downstream velocities as a percent of mean mainstream flow

		LEES	MIDS	FACE	TOPS
U_{mean}	mean	1.7	14.1	46.7	37.7
	s.d.	15.8	20.1	31.2	34.2
U_{sd}	mean	11.2	14.7	21.4	23.6
	s.d.	6.3	7.4	9.1	11.2

The data analysis supports the differentiation of four different hydraulic habitat types, and certainly confirms that once the skimming flow-wake interference flow threshold is passed, a greater diversity of flow conditions can be found between roughness elements. This threshold clearly has biological significance, as different inferences can be made about the between element flow regime above and below this threshold.

The nature of the four hydraulic habitat types can be summarised as follows:

LEES: a generally low velocity habitat but with high mean velocities at some locations. Velocities are quite variable in time, so that benthos must withstand velocities several times as high as the mean.

MIDS: a generally low velocity habitat, but on average significantly faster flows than in LEES. Both spatial and temporal variability are significantly greater than in LEES. Once again temporal variability are such that benthos must withstand velocities several times as high as the mean.

TOPS: a generally high velocity habitat but with low mean velocities in some locations. Temporal variability is highest in this habitat, so again, benthos must withstand velocities far in excess of the mean at most locations.

FACE: the highest velocity habitat but again both high and low mean values at some locations. Significant variability, but less than in TOPS.

One general observation to emphasise, is that flows in the near-bed zone are highly variable in both time and space. As organisms must be able to withstand extreme conditions not just average conditions, an assessment of this variability is very important. In all habitat types, locations were found where the mean downstream velocity was negative (ie. upstream).

The temporal variability discussed here is essentially random and short-term in nature (due to turbulence) and occurs under constant discharge conditions. The longer term variations caused by changing discharge add another level of complexity to the picture. However, many zoobenthos are able to move to more favourable locations in response to these longer term velocity variations. The experimental data do not allow investigation of these aspects of near-bed hydraulics.

The data here do not show the sequencing of individual velocity measurements, so it is not possible to assess the magnitude of likely acceleration reaction forces in these flows, but they are likely to be significant component of the total force acting on zoobenthos.

The relative flow magnitudes in these habitats (shown in Table 3) may be transferable to other situations, however, more data is need to assess their dependence on relative roughness height. The flow velocities between roughness elements can also be considered relative to the near-crest flows. The mean velocity in LEES is just 4% of the near-crest flows, and the mean velocity in MIDS is 33% of the near-crest flows. Theses figures highlight the differences between the sheltered and exposed habitats.

CONCLUSIONS

From an analysis of near-bed flow velocities over a reconstruction of a cobble stream bed four hydraulic habitat types have been defined; two being exposed and predominantly high velocity habitats, and two being sheltered and predominantly low velocity habitats. Identification of these habitat types is based upon the threshold between skimming and wake interference flows, and so their existence can be explained in terms of how the bed roughness influences the near-bed flow. In skimming flow only two habitat types are observed, and so the variety of flow conditions and the diversity of available niches are less. In wake interference (and other high energy dissipation flows), all habitat types are observed, so under these flow types a greater range of ecological niches will be found. The threshold between skimming and wake interference flow is therefore concluded to have ecological relevance, and the colonisation experiments of Barmuta (1994) support this. The thresholds between other near-bed flow types (isolated roughness and chaotic flow) do not appear to be ecologically important. It is recommended that classification of the near-bed flow be undertaken in conjunction with zoobenthos sampling.

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CHARACTERISING THE ACOUSTICAL ENVIRONMENT OF STREAMS

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ABSTRACT

Although fish have a well developed sound perception, characterisation of aquatic habitat rarely considers the acoustical environment of streams. Fish may use flow noises to navigate while migrating, or when seeking favourable conditions for feeding and avoiding predators. At this stage there is no accepted method for characterising the acoustical environment of streams. This paper describes an approach to this problem and presents the results of its application to a mountain stream and to a fish ramp in a lowland stream. In pool-riffle streams, much of the underwater sound is generated at hydraulic jumps. The acoustics of hydraulic jumps were examined using laboratory modelling. A relation was found between underwater sound intensity and power loss in hydraulic jumps. This suggests a relation between the distribution of power loss intensity and the distribution of sound intensity within a stream reach. Consequently, underwater sound measurements may be useful in characterising the spatial pattern of energy dissipation or power loss intensity in mountain and pool-riffle streams. The effects of bed material and depth on the propagation of sound signals may complicate this relation. A relative sound intensity is proposed for comparing sound intensity characteristics in different streams, although its effectiveness is yet to be tested.

KEY-WORDS: Fish Habitat / Fish Migration / Hydraulics / Hydrodynamics / Acoustics / Underwater Acoustics / Fish Hearing / Turbulence / Hydrodynamic Sound

INTRODUCTION

Existing methods of characterising the physical habitat of fish have borrowed many descriptors from traditional hydraulics such as current velocity averaged over time, depth and bed particle size measurements. These measures have only been partially successful in explaining habitat selection by fish. Underwater acoustical signals generated by streams are 'finger prints' of the physical environment in which they were generated and propagated. It is possible that lotic fish species are processing flow sounds to 'hear' important aspects of the stream environment. There is some potential for improving the understanding of fish habitat by examining the relationship between fish behaviour and the acoustical environment of streams. As a first step towards this objective, this paper summarises some basic aspects of underwater acoustics and fish hearing, and presents an approach to characterising the acoustical environment of streams.

A laboratory experiment was conducted to investigate the acoustics of hydraulic jumps, a major source of sound in pool-riffle and mountain streams. Two case studies are presented to demonstrate how the acoustical technique can be applied. The first case study, in Hince Creek in northern Victoria, demonstrates an approach to characterising mountain streams using the acoustical technique. The second case study was conducted in the Yarra River, a lowland river which flows through Melbourne, Victoria. This study demonstrates application of the acoustical technique to compare sounds propagated in a ramp built on a weir to provide for fish passage, with those in a natural riffle 200 m downstream of the weir. The effectiveness of the fish ramp in allowing the upstream passage of fish is discussed in light of the results of the study. The background for the flume experiment, and the two case studies, are included separately from the general background to underwater sound and fish hearing. A summary discussion of the laboratory experiment and two case studies is included at the end of the paper.

BACKGROUND

Underwater Flow Sound

Two fundamental mechanisms are responsible for sound generation; the vibration of solid bodies, and flow-induced noise resulting from pressure fluctuations induced by turbulence and unsteady flows, generally referred to as aerodynamic or hydrodynamic sound (Blake, 1986; and Landahl and Mollo-Christensen 1992). Sound propagates as a pressure wave through an elastic medium such as water at some characteristic speed c (ms^{-1}) depending on the bulk modulus of elasticity K (Nm^{-2}) and density ρ (kgm^{-3}) of the medium (Equation 1) (Streeter and Wylie, 1983). At standard conditions the speed of sound is 330 ms^{-1} in air and 1490 ms^{-1} in water.

(1)
$$c = \sqrt{\frac{K}{\rho}}$$

A sound source transfers acoustical energy to the surrounding medium at a rate called the sound power. The power passing through a unit surface area perpendicular to the direction of sound propagation is called sound intensity, i (Wm^{-2}). Sound pressure is the alternating pressure variation that results from a sound about the mean or ambient pressure. Hydrodynamic sound is characterised by a random signal and must be analysed using statistical parameters such as the root mean square (RMS) pressure p (Nm^{-2}). For a particular medium there is a relation given by Equation 2 which relates the RMS sound pressure to sound intensity. ρc is sometimes called the characteristic impedance of the medium.

(2) $$I = \frac{p^2}{\rho c}.$$

There has been almost no research on the flow-induced sounds of streams. Hawkins (1975) showed a relationship between the average acoustical radiation above the stream surface and the energy dissipated by the stream per unit stream length. Although related, ambient sound intensity above the stream surface is orders of magnitude less than underwater sounds and has very different spatial characteristics. At this stage, modelling of hydrodynamic sound generation and propagation in streams has received very little attention.

A splash made by the entry of an object into water, such as a water droplet in a waterfall, radiates not only surface, but also sound waves in both air and water (Bleckmann, 1988). Hawkins (1975) reported that surface noise in mountain streams is associated almost entirely with "white water" at overflows and hydraulic jumps, and not with conventional roughness elements.

The air-water interface is an excellent reflector at all frequencies, reflecting almost all sound back into the water (Rogers and Cox, 1988). In shallow water, sound can only propagate over distances greater than the water depth by repeatedly interacting with the surface and bed. Reflectance at the bed depends on the frequency of the sound signal and its angle of incidence with the bed surface. However, the reflectance at the bed is generally poor. In shallow water no sound below a certain cut-off frequency will propagate much further than the depth of the stream because the reflectance of these frequencies is negligible. This cut-off frequency f_c (s^{-1}) is dependent on the depth of water y (m) and the speed of sound in both water c_w, and the bed sediment c_s (ms^{-1}) as shown in Equation 4 (Rogers and Cox, 1988). The speed of sound is generally greater in bed sediments than in water.

(4) $$f_c = \frac{c_w / 4y}{\sqrt{1 - c_w^2/c_s^2}}.$$

As depth increases, this low cut-off frequency decreases. In 0.1 m of water, almost no sound that is in the hearing range of fish will propagate further than the depth of the water. Water depths of 1 m will propagate frequencies of 500 s^{-1} or higher in clay bed streams, but the cut-off frequency in rock-bed streams is much higher. For this reason, fish that live in shallow water have developed extended high-frequency capabilities (Rogers and Cox, 1988).

Underwater Sound Recording and Spectrum Analysis

Underwater acoustical pressure fluctuations can be measured using a hydrophone which produces a fluctuating voltage proportional to the amplitude of the incoming sound wave. Calibration is required to compare sound amplitudes observed with different equipment or to interpret sound in energy units (Charif et al., 1995).

A sound signal can be digitised by measuring or sampling the voltage amplitude of the analogue signal at a particular sampling rate. The highest frequency that can be represented in a digitised signal is half this sampling rate (Charif et al., 1995). Once digitised, the signal can be transformed from the time-domain to the frequency-domain using a variety of computer software packages available for Fourier transform analysis. This analysis produces a spectrum in which the amplitude of the signal is expressed as a function of frequency. Hydrodynamic sounds generally have broad-band signals, which means a sharp peak in the spectra is not observed. Rather, sound amplitude is distributed over a range of frequencies. The peak intensity ($Wm^{-2}s$) over this range and frequency (s^{-1}) at which this peak occurs are characteristics of the signal.

Fish Hearing

It is apparent that all fish must be capable of receiving acoustical stimuli (Tavolga, 1971). Many fish can detect a sound intensity of 0.01 μWm^{-2} at 500 s^{-1} (Rogers and Cox, 1988). Fish have an impressive complement of hydrodynamic and acoustical sensors, commonly referred to as the lateral-line and inner-ear organs (Tavolga, 1971; Kalmin, 1988; Popper and Platt, 1995). The lateral-line organs are arranged in linear arrays on the head and along the trunk of the fishes. The lateral line-system detects accelerations in the flow field relative to the fish location. So far, the precise functions of the inner-ear organs have escaped identification. Behavioural experiments have shown that many fish species can perform frequency and intensity discrimination for pure tone sounds, and identify the direction of a sound source, as well as detect its presence when masked by background noise (Popper et al, 1988).

Denton and Gray (1988) discussed the mechanics of the lateral line in detail. The most likely natural stimuli to lateral line sense organs are: local pressure gradients produced by the fish's own swimming movements; mechanical disturbances caused by distortion in fish tissue arising from its movements; and local pressure gradients caused by external sources such as neighbouring animals, disturbances at the surface, or by flow of water around rocks in streams. Different parts of the lateral line system measure different aspects of the mechanical disturbances to which the fish is exposed. It seems that for most fish, the lateral lines are concerned with low frequencies (<200 s^{-1}) and with disturbances that originate either close to the fish or from its own movements (Denton and Gray, 1988).

Popper et al. (1988) provided a model of the inner-ear system which takes into consideration available behavioural, physiological and morphological data. It is likely that sound reaches the ear of fish by two pathways. Many fish have a gas-filled chamber in the abdominal cavity called a swim-bladder. Sound pressure probably sets the wall of the swim-bladder into motion, because of the relatively high compressibility of the gas it contains. The vibration of the swim bladder wall radiates sound energy in the form of particle displacement which stimulates the sensory hair cells. This path for sound to the ear is probably found in most fish with a swim bladder. A second method for stimulation of the ear enables detection of the particle motion of an acoustical disturbance relative to an inertial frame of reference. Since the density of fish is nearly the same as that of water, the fish moves in the sound field as if it were part of the water mass. The otolith (organs of the fish ear) have a density three times greater than the density of water and the fish's body. Consequently the otolith moves with a smaller amplitude and a different phase from the rest of the body. It is likely that all fish have sensory organs to detect displacement of the otolith relative to the rest of the fish body. These two pathways are complementary since the pathway via the swim-bladder is more effective above perhaps 200 s^{-1}, while the second pathway, which detects motion relative to an inertial frame, is most effective below 200 s^{-1} (Popper et al., 1988).

Studies have revealed that fish can detect the direction of a sound source. For example Hawkins and Sand (1977) report that cod discriminate between sound sources 16° apart in the median vertical plane and 20° in the horizontal plane. Schuijf and Buwalda (1988) explain that it is unlikely that fish could use the same binaural differences for directional cues, as does a human, because the close proximity of fish ears and the high speed of underwater sound propagation produce differences in stimulus timing that arise from unequal path lengths orders of magnitude smaller than in a human. An alternative mechanism for locating the direction of sound sources is proposed by Popper et al, (1988). The acceleration effects of acoustic field result in motion of the otolith relative to the fish body. This relative motion differs for each sound direction. Sensory hair cells in the inner ear are oriented in different directions. The acoustical particle acceleration is decomposed into components along the directional polarised axes of the otolith organs. Consequently there is a unique ratio of responses from the perpendicularly oriented hair cell groups for any sound source direction (Popper et al. 1988).

METHOD (Underwater Sound Recording)

A calibrated Brüel and Kjær hydrophone (type 8103) and a Brüel and Kjær charge amplifier (type 2635) were used for most sound recordings discussed in this paper. It was mounted in the end of a 500 mm length of steel tubing of the same diameter as the recording device. All recordings were taken with the rod oriented longitudinally to the flow and the hydrophone oriented in the upstream direction to minimise noise generated by flow over the recording apparatus.

Sound was recorded digitally using a Macintosh Powerbook 165C that had an external mono-sound recording port with a 4 mV voltage ceiling. The 11,127 s^{-1}, 8-bit digital recordings, provided sufficient signal resolution for the purposes of this study. If stored in a binary data file, a 10 s recording uses a little over 100 kBytes. Spectral analysis and calculation of average sound intensity were performed using Canary™, a dedicated bio-acoustics analysis software package developed at the Cornell Laboratory of Ornithology (Charif et al., 1995).

FLUME EXPERIMENT TO EXAMINE ACOUSTICAL CHARACTERISTICS OF HYDRAULIC JUMPS

Background

To date, there appears to be no investigation of flow-related sound generation and propagation in streams. From observation it would seem that in pool-riffle streams, hydraulic jumps are the most significant sound source. For this reason, exploratory experiments were conducted to investigate the acoustics of hydraulic jumps in a laboratory flume.

The relation between depth, y_1 (m) and velocity, V_1 (ms^{-1}) upstream, and the depth y_2 (m) downstream of an hydraulic jump in a horizontal rectangular channel, derived using the principles of continuity and conservation of momentum is given in Equation 5 (Streeter and Wylie, 1983). The slight modification of this Equation, required to account for channel slope (Chow, 1961) was considered unnecessary for this study. Hydraulic jumps are effective energy dissipaters. From energy considerations, Power P_h (W) loss in the hydraulic jump can be given by Equation 6.

(5)
$$y_2 = -\left(\frac{y_1}{2}\right) + \sqrt{\left(\frac{y_1}{2}\right)^2 + \frac{2V_1^2 y_1}{g}}.$$

(6)
$$P_h = \rho g Q \frac{(y_2 - y_1)^3}{4 y_1 y_2},$$

where Q is discharge (m^3s^{-1}).

Method

A laboratory flume 0.3 m wide and 6.4 m long with a bed slope of 0.035 was used to examine the acoustics of hydraulic jumps. Supercritical (shooting) flow entered at the upstream end of the flume. A drop-gate at the downstream end of the flume was used to establish a section of subcritical (tranquil) flow upstream of the gate (Figure 1). A hydraulic jump was established part way along the flume. Gauged flows of up to 0.035 m^3s^{-1} could be supplied to the flume.

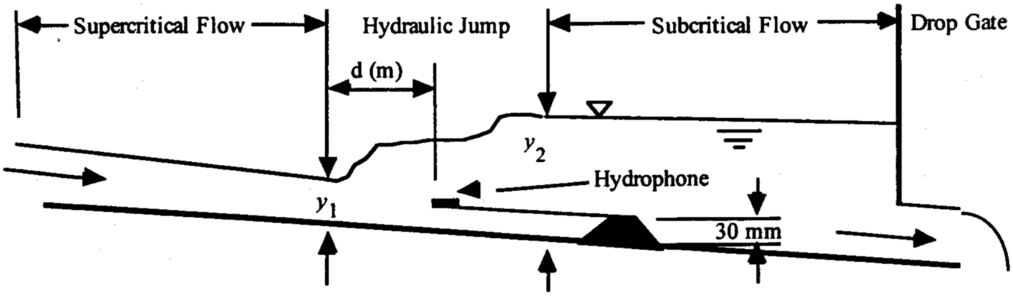

Figure 1: Side view diagram of laboratory flume used to examine the acoustics of hydraulic jumps (not to scale).

Experiments were conducted at night to minimise irregular noises; however, some noise was produced by the laboratory pump, and water flowing into the sump at the end of the flume. A recording was made in still water to establish the contribution of the pump noise to recordings. The average sound intensity of the pump noise was low compared with flow sounds in the flume.

Sound recordings were taken for six different hydraulic jumps at 0.1 m intervals along the flume starting at the upstream edge of the jump and moving into the highly turbulent flow downstream. The tip of the hydrophone was located by its distance from the upstream end of the hydraulic jump (d in Figure 1).

Results

Sound intensity was found to vary along the hydraulic jump (Figure 2). The strongest sound signal detected was located under the leading edge of the jump for the four lowest flows. At the two highest flows, the highest level was recorded downstream of this point.

The maximum sound intensity was selected from the five values measured for each jump. This maximum sound intensity was strongly related to the power dissipated by the jumps (Figure 3). The regression equation fitted to these data suggests a linear relation.

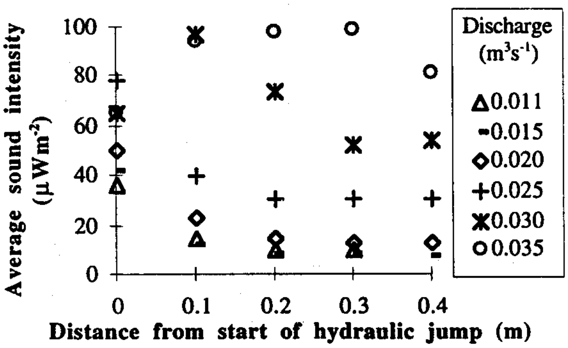

Figure 2: Underwater sound intensity along a hydraulic jumps of varying power in a rectangular laboratory flume.

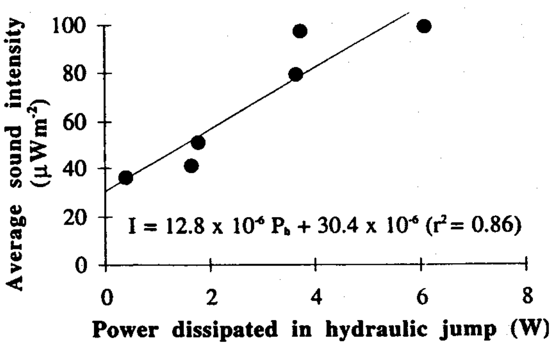

Figure 3: Underwater sound intensity and power loss in hydraulic jumps in a rectangular laboratory flume.

ACOUSTICAL CHARACTERISATION OF THE HYDRAULIC ENVIRONMENT IN A MOUNTAIN STREAM

Background

Characterising the hydraulics of mountain streams is a difficult task because of the high degree of irregularity in the channel morphology and large roughness elements which cause turbulence, three dimensional flow patterns, steps in the water surface, and fluctuations in elevation of the water surface over periods of a few seconds. Mountain streams are generally steeper than lowland streams. Waterfalls, cascades and hydraulic jumps account for much of the power loss in mountain streams. The generation of underwater sound is associated with the rate of energy dissipation. Consideration of energy dissipation as part of conventional methods of characterising the hydraulics of mountain streams is limited to measurements of water surface slope over distances of several metres or more.

The laboratory experiment discussed above demonstrated that sound levels in a hydraulic jump were related to power loss. For this reason the distribution of sound intensity throughout a reach is likely to be related to the spatial distribution of power loss intensity (Wm^{-2}) dissipated by the flow over small local plan areas of the stream bed. Depth of water and bed material may affect this relationship by influencing the distance over which underwater sound travels. Power loss intensities will generally increase with stream slope and discharge. For the purpose of comparing behaviour in different streams it may be useful to introduce a non-dimensional, or relative, underwater sound intensity i', defined as the ratio of recorded sound intensity at a point i, to the average sound intensity recorded in the stream $\bar{i}$ (Equation 7). It is proposed that the distribution of this relative underwater sound intensity is a suitable way of characterising the micro-scale distribution of energy dissipation within a stream reach.

(7) $$i' = \frac{i}{\bar{i}}$$

Method

The acoustical approach to characterising mountain streams was applied to a site on Hince Creek, an upland tributary of the Murray River in south-eastern Australia. The creek had a slope of 0.025 and was lined with cobbles and boulders that were up to 1 m in diameter. There was a cascade located half-way along the sampled reach at a bedrock outcrop. Recordings were made in the stream when the flow was relatively low.

Forty-three randomly located sound recordings were made along the reach, each with a duration of 10 s. These recordings were analysed to determine the average sound intensity at each recording point. The sampled intensities were divided by the average sampled intensity for the stream to give the relative sound intensity defined in Equation 7.

Results

Much of the energy dissipated in the reach appeared to be lost at a few large steps in the water surface caused by lines of boulders across the stream. Hydraulic jumps were observed downstream of these boulder-lines that seemed more vigorous than elsewhere in the reach. Smaller hydraulic jumps occurred below large cobbles distributed throughout the shallower sections of the stream. There were a few sections of pooled water in which energy dissipation was unlikely to be a significant component of the energy dissipated over the reach as a whole.

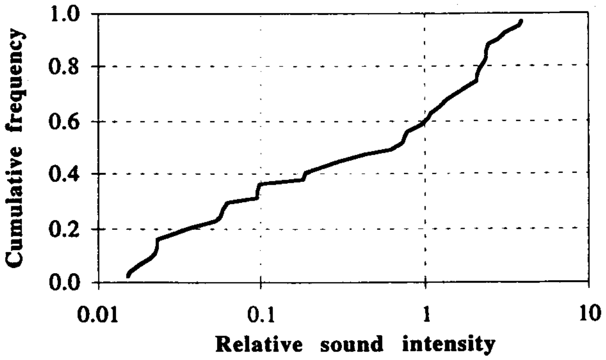

Figure 4: Cumulative distribution of sound intensity in Hince Creek, south-eastern Australia.

The relative sound intensity distribution in Hince Creek was close to linear when plotted as a cumulative distribution on log-linear axis (Figure 4). This indicates a that few areas with high sound intensities exist in the stream. However, 60% of the sound recordings had intensities below the average value for the reach. Approximately 35% of the recordings had intensities less than one tenth of the average value.

APPLICATION OF THE ACOUSTICAL TECHNIQUE TO IMPROVE THE DESIGN OF A FISHWAY

Background

It is commonly thought that fish locate the upstream direction during upstream migrations using the stream current, although swimming in the direction of the on-coming current is likely to be a costly exercise in terms of energy consumption. A more energy efficient approach would be to map the upstream course in advance, with the aim of minimising contact with high velocity zones. Fish could select from the range of courses with an upstream component, the directions from which the least sound intensity is propagating. This energy saving behaviour would seem particularly appropriate for fish with low or moderate swimming capabilities. Selecting a low energy course may be less important for fish with stronger swimming capabilities, but they could identify suitable rest areas below overflow and hydraulic jump formations using acoustical signals. Acoustical signals are probably more important in turbid streams where visual evidence of flow condition is not available.

Dight's Falls is a weir located at a natural cascade on the Yarra River in Melbourne, Australia. Since construction of the weir last century, it has obstructed the passage of most migratory native fish species to the Yarra River and its tributaries upstream of the weir. Recently, a ramp was constructed along the face of the weir to provide passage for fish past the weir. The fishway was constructed of boulders that form a number of steps and pools at moderate and low flows. A greater variety of migratory species have been observed above the weir since construction of the fishway, and netting surveys on the ramp itself have shown that it has been at least partially successful in providing fish passage for some species (T. O'Brien, Arthur Rylah Institute, pers. comm.). There is a concern that because the entrance to the ramp is located to the side and slightly downstream of the weir, many fish moving upstream may not locate the ramp and continue past it to the vertical weir. Modifications to the ramp to improve its efficiency are currently being negotiated with the authority responsible for management of the Yarra River at Dight's Falls.

If flow sounds are used for upstream navigation by migratory fish, it follows that structures designed to provide fish passage should propagate sounds similar to those typically encountered in the stream. A study was conducted to compare the underwater sounds generated by flows down the ramp with sounds generated in a riffle 200 m downstream of the weir. The riffle appears to be a natural feature, so it was assumed that the riffle generates acoustical conditions that are typical of those within the stream.

Method

Because flow conditions in the ramp varied substantially in all three dimensions, linear transect sampling techniques were inappropriate. Instead, 14 recording points were selected randomly from a single step-pool sequence. The hydrophone was oriented into the flow and held in place using a lead weight.

Random sampling of sounds in the riffle was not practical as the current was too strong to allow safe wading to randomly located recording points. Instead, sounds propagated in the riffle were recorded at evenly spaced points along a transect located across the stream in the centre of the riffle. The hydrophone was suspended from a rope stretched across the stream at a point about 200 mm below the water surface. It was weighted down and oriented into the flow using a streamlined lead weight. The hydrophone projected well in front of the lead weight to minimise recording of noise generated by flow around the weight and ropes. A 10 s sound recording was taken from each of 18 points across the transect.

Results

Sound intensity recorded in the riffle varied from 2.2 to 14 μWm^{-2}. Lower sound intensities were recorded close to the banks (Figure 5). The highest intensity was observed 15 m across the stream, about 2 m below a large boulder in the river. It was observed that the boulder caused a short stretch of shooting flow and a relatively large hydraulic jump just upstream of this recording point. There was considerably more variation in the sound intensities recorded in the ramp, ranging from 0.6 to 340 μWm^{-2}. The cumulative frequency distributions of sound intensity at both sites (Figure 6) shows that sound intensities in the ramp vary over a range two orders of magnitude greater than in the riffle.

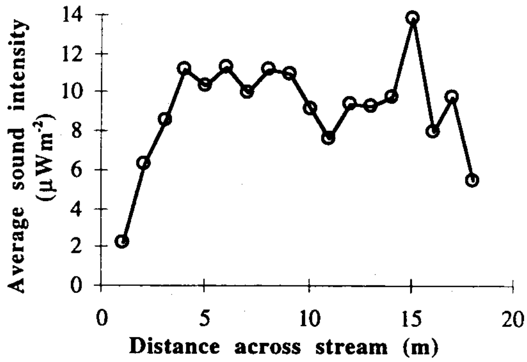

Figure 5: Sound intensity across a riffle in the Yarra River below Dight's Falls.

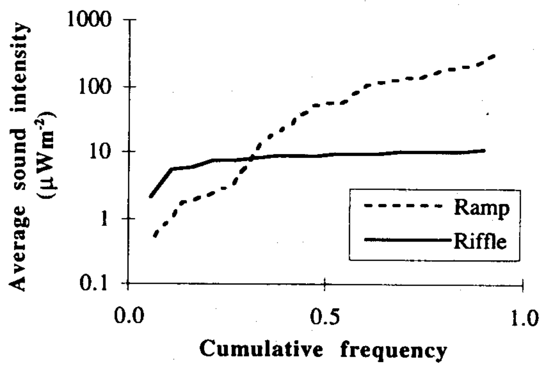

Figure 6: Cumulative frequency distribution of sound intensity sampled in a riffle and fishway ramp below Dight's Falls in the Yarra River.

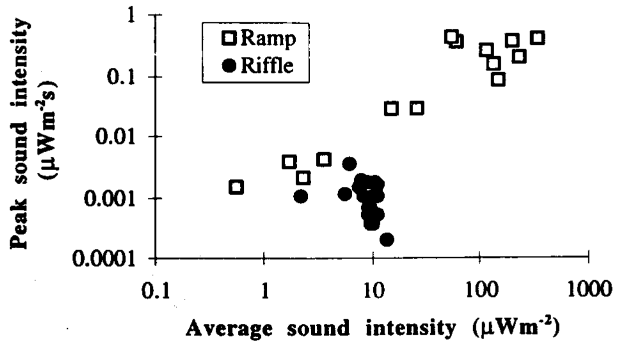

Figure 7: Peak and average sound intensity recorded in a riffle and fishway ramp below Dight's Falls in the Yarra River.

One method of characterising the shape of a frequency spectrum is using the ratio of peak intensity in the frequency spectrum to average intensity. This ratio was consistently greater in the fishway ramp than in the riffle (Figure 7).

DISCUSSION

Experiments in a flume using a hydraulic jump suggested that acoustical intensity is related to power loss in hydraulic jumps. Further experimentation is required to define the nature of this relation, and to identify other factors that influence sound levels in streams. Laboratory modelling of underwater stream acoustics should be conducted in such a way as to appropriately scale the lower cut-off frequency (Equation 4).

Given a relation between power loss in streams and sound intensity, sound intensity measurements may be a useful way of characterising the spatial distribution of power loss intensity over a stream reach. The distribution of sound intensity recorded in a reach of Hince Creek illustrates this approach. The distribution of recorded sound intensities was consistent with the distribution that might be expected from the observed spatial pattern of energy losses in the stream reach. The few higher intensity recordings were associated with large hydraulic jumps that were located below boulder-lines. Moderate sound intensities were associated with the many small hydraulic jumps that were located downstream of large cobbles in the shallower sections of the reach. The low intensity recordings were associated with deeper, slower flowing areas, including a few pools. The ability of this acoustical approach to discriminate between streams with different hydraulic characteristics is yet to be tested.

Sound intensities in the riffle below Dight's Falls in the Yarra River were less than the those recorded in the laboratory flume. Although not measured, the power loss intensity in the riffle was unlikely to be as high as that dissipated by the hydraulic jump in the laboratory flume. This probably accounts for the lower sound intensities recorded in the riffle. The fishway (ramp) at Dight's Falls had a significantly greater slope than the riffle, and hydraulic jumps of the vigour generated in the laboratory flume were located below boulder overflows. The power loss intensity in the ramp was likely to be greater than in the riffle; this probably explains its higher sound intensity levels.

The variability and discontinuity of the water surface slope in the ramp indicates significant variability in the rate of stream energy dissipation across and along the ramp. Water surface slope was more regular in the riffle, indicating a more even distribution of stream energy dissipation. The greater variability in sound intensities recorded in the ramp than in the riffle was at least partially the result of greater variability in the spatial pattern power loss intensity.

Frequency spectra in the ramp were more peaked than in the riffle. Sound recorded at a point in the riffle integrated signals generated over a larger volume than in the ramp because sound propagates further in the deeper waters of the riffle (Equation 4). Integration of a number of sound sources with different peak frequencies would create a broader band signal than the sound from each source individually, this is a possible explanation for the riffle's broader band signal.

The possibility that fish are mapping their upstream path during migration using acoustical signals suggests that modification of the fishway at Dight's Falls should consider the sounds produced by the ramp. The ramp currently generates sound of a greater intensity and variability than would be expected from a riffle. Such a modification to the natural stream acoustical environment may disorient fish if they are navigating upstream using hydrodynamic sounds. One approach to incorporating acoustics into the design would be to construct a short riffle with a slope and bed material similar to the natural riffle further downstream at the base of the ramp. This might produce the type of acoustical signals that attract fish to navigate upstream. Higher, less attractive, sound intensities could be introduced into the stream at either side of the ramp using a row of large boulders. These could act as acoustical baffles that help direct fish up the ramp. Comparing the performance of the modified ramp with and without these acoustical baffles would be a rough test of the hypothesis presented in this paper of how fish use acoustical signals to negotiate riffles during upstream migration.

CONCLUSION

Many fish may have the ability to discriminate between sounds of different intensity, frequency and source direction. Interpretation of sensory performance requires some knowledge of fish behaviour, the quality and amplitude of stimuli to which they respond under different conditions, the stimuli they may ignore, and the effects of motivation and habitation (Blaxter, 1988). Further research may define the aspects of acoustical signals that are important to lotic fish species. It seems reasonable to assume that fish use sounds generated by water flow to both locate rest areas and navigate through fast water.

Fish are capable of detecting sounds at intensities commonly encountered in streams. From a review of the literature it appears that underwater acoustical signals generated in streams do not propagate over distances much greater than the depth of the stream, particularly in rock bed streams and for lower frequencies ($< 500\ s^{-1}$). The implication of this for fish hearing is that any hydrodynamic acoustical signals generated by flow that are important to fish behaviour are generated close to the fish and are not masked by noise generated at greater distances.

A method was presented for measuring and analysing stream sounds. A relation was found between underwater sound intensity and power loss in hydraulic jumps. This suggests a relation between the distribution of power loss intensity and the distribution of sound intensity within a stream reach. Consequently, underwater sound measurements may be useful in characterising the spatial pattern of energy dissipation or power loss intensity in mountain and pool-riffle streams. The effects of bed material and depth on the propagation of sound signals may complicate this relation. A relative sound intensity is proposed for comparing sound intensity characteristics in different streams, although its effectiveness is yet to be tested. Future research could examine the effect of water depth, bed material, slope and discharge on underwater sound intensity and relative intensity distributions.

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Measurement of Burst Swimming Performance in wild Atlantic Salmon (*Salmo salar*) using Coded Radio-transmitted signals.

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Abstract

The swimming performance of wild Atlantic salmon (*Salmo salar*) in an experimental flume was investigated using coded radio signals. To calculate swimming speed, distance moved and time elapsed were measured with a digital spectrum processor using real-time spectrum analysis. This device was designed to be used in a co-processing arrangement with a receiver, thereby providing pulse position code discrimination, verification and continuous data storage. Radio-tagged adults (48.30-54.80 cm long) voluntarily swam against water velocities, ranging from 1.32 to 2.85 $m.s^{-1}$, in an 18 m long flume at a mean water temperature of 10°C. At water velocities of 1.32 to 1.55 $m.s^{-1}$, individuals successfully ascended the flume at swimming speeds of 3.30 to 4.79 body lengths per second ($L.s^{-1}$), respectively. At high water velocities ranging from 1.92 to 2.85 $m.s^{-1}$, individual swimming speeds increased from 4.94 to 7.27 $L.s^{-1}$, respectively. However, above a threshold value of 1.92 $m.s^{-1}$, individuals traversed shorter distances and were unable to ascend the flume. The highest swimming speed observed was 4.13 $m.s^{-1}$ or 8.35 $L.s^{-1}$. The results of this study indicate that in addition to its applicability in the determination of burst swimming speeds, radio telemetry could prove a useful tool in the design and evaluation of future fishways and culvert installations.

KEY-WORDS: Atlantic salmon / burst swimming / acceleration / digital telemetry

INTRODUCTION

Dams have been widely constructed on waterways to provide a stable water supply, to control floods and to provide hydroelectric power. Some of these dams have constructed fishways intended to provide safe passage for fishes. High flow rates are often associated with fishway entrances, and for a fish to progress through a fishway it must be able to swim at velocities greater than the opposing water currents. Generally, successful passage requires a high swimming speed. However, few studies have assessed burst swimming capabilities in salmonids in relation to fishway design.

The capacity for high speed swimming for short periods of time is important for the survival of many fish species. Acceleration involved in fast-starts from rest, high speed manoeuvres, or speed changes from one steady speed to another is an integral part of swimming. The former two propulsive patterns involve high rates of acceleration and are important in prey capture and predator avoidance (Hunter, 1972), and in the successful negotiation of such obstacles as waterfalls and fishways (Katopodis, 1990; 1994; 1995). A few species like salmon use high speed bursts to swim up or leap otherwise impassible water falls, allowing fish to migrate upstream to reach spawning grounds. As well, Weihs (1974) has suggested, by alternating periods of fast swimming and gliding fish can greatly reduce energy expenditure. Thus, fast bursts of swimming are vital components of a fish's locomotory repertoire.

Swimming of fish can be classified into three major categories: sustained, prolonged and burst swimming speeds. Sustained swimming represents speeds which can be maintained for longer than 200 mins (Beamish, 1978) and energy is derived exclusively from aerobic processes. Prolonged swimming covers a spectrum of speeds between burst and sustained and is often categorized by steady swimming interspersed with periods of vigorous efforts. The highest speeds of which fish are capable, are classified as burst swimming. In fish, as in all vertebrates, the highest levels of exercise performance are achieved anaerobically (Jones, 1982). These high speeds can be maintained only for brief periods (less than 20 secs), and are terminated by the exhausation of extracellular energy supplies or by accumulation of waste products. Burst activity may be described by three kinematic stages described by Weihs (1973): preparatory, propulsive and variable.

There are several biological and environmental constraints on fish swimming which deserve consideration. In a laboratory study on burst swimming, Bainbridge (1958) supported the view that fish up to 1 m in length should be able to swim up to 10 times their own body length for a short time of about 1 sec ($L.s^{-1}$), beyond which swimming speed would decrease exponentially. Recent studies suggest variability in burst swimming among species and for some, the relative performance maximum of 10 $L.s^{-1}$ is a conservative measure (Webb and Corolla, 1981; Wardle and He, 1988). Furthermore, the maximum speed which fish can achieve is influenced by fish size (Webb, 1975) and body form (Webb 1978; Taylor and McPhail, 1985). In addition, anaerobic metabolism at burst activity levels is relatively independent of water temperatures (Webb, 1978). However, recent studies indicate that temperature acclimation effects burst speeds at 5°C and 15°C (Beddow *et al.*, 1995). Therefore, biological factors such as size, nutritional state and species, as well as environmental factors may affect burst swimming performance.

The objective of this study was to assess high speed swimming performance in anadromous Atlantic salmon, *Salmo salar*, in an experimental passage structure. In particular this study focussed on the influence of flume velocity on the burst swimming speeds of this species during its spawning migration.

MATERIALS AND METHODS

Experimental Flume

The study was conducted in an experimental stream flume constructed at the Noel Paul Brook Research Facility in central Newfoundland, Canada (49°N, 57°W). The flume consisted of an upstream head pond (upper pool, area 6 m^2) created in the existing sluiceway of the dam and a downstream pond (lower pool, area 17 m^2) constructed in Noel Paul Brook. Between these two pools was a wooden flume which measured 17.73 x 0.50 x 0.61 m and was filled to varying depths greater than 0.30 m. The discharge into the upper pool was controlled by varying the number of stoplogs in the first sluiceway of the dam. In addition, two gates in the stoplog section were regulated to adjust and stabilize the water level in the head pond. The fish exit (into the upper pool) was constructed to develop as laminar a flow as possible, reducing turbulent and transient water flows. This was accomplished by constructing a straight section on the top and two sides of the flume entrance. The lower pool contained a trapezoidal (or Cipolleti) weir located in a "still" area of the pool. Test fish were held in the lower pool while experiments were being conducted. The flume had a slope of 0.2% and where it entered the upper pool its bottom was flush with the floor of the upper pool.

Radio Telemetry Equipment

Radiotelemetry equipment manufactured by LOTEK Eng. Ltd, Newmarket, Ontario was used to monitor the movement of fish in the flume. Radio transmitters used in this study were cylindrical with rounded ends (1 cm in diameter x 5 cm long) and weighed 8.6 grams or less than 1% of the weight of the experimental animals. An insulated flexible wire antenna 24 cm long trails from one end of the tag. Transmitters were encapsulated in waterproof epoxy resin and activated by a magnetic reed switch. Transmitters were uniquely coded and assigned a corresponding channel and frequency. The transmitters were designed to broadcast at frequency intervals of 20 kHz within an operating band of 149.540-149.740 MHz with a battery life of 15 days.

Signals from the transmitters were received by 7 stationary, underwater antennae connected to a telemetry receiver and digital spectrum processor (SRX_400 and DSP_500). The digital receiver was equipped with multiple antennae switching capability which determines the location of a transmitter relative to the 7 antennae (Figure 1). This is achieved via pulse position code discrimination, in which each radio transmission is assigned a unique coded time signature. Antennae were simultaneously scanned every 7.5 msec. The fixed antennae array was placed at equally spaced intervals along the bottom of the flume (0, 2.53, 5.06, 7.59, 10.12, 12.65, 15.18 m). Calibration of signal strength permitted determination of distance between transmitter and antennae, thereby eliminating non-quantitative visual monitoring. This procedure involved mapping individual antenna reception cells and areas of cell connection zones down the length of the flume prior to experimentation. The digital spectrum processor records events in real time, provides measurements of transmitter position, thus monitoring fish passage through the flume.

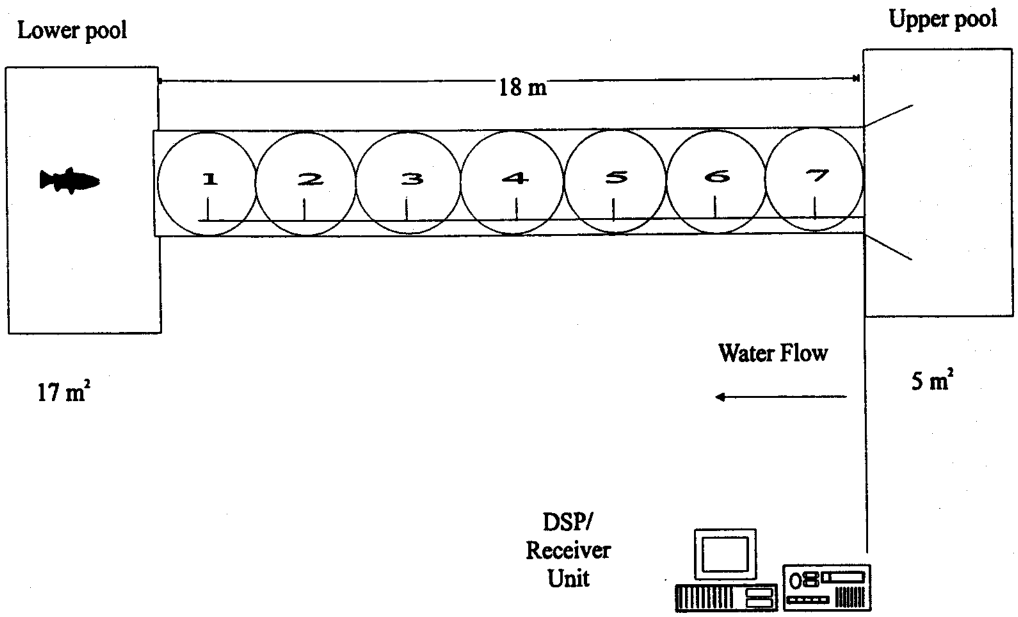

Figure 1: A schematic diagram showing the DSP system and an array of 7 antennae.

<u>Fish</u>

Wild adult Atlantic salmon (N=15, mass=1.16±0.18 kg, fork-length=51.17±2.31cm, girth=23.07±1.38cm: mean±sd) were collected from fishways on the Exploits River during September 1995. The Exploits River is 267 km long and has a drainage area of 11, 272 km^2. All salmon were transported to an upper tributary stream at Noel Paul Brook Research Facility where they were retained in covered outdoor enclosures which measured 3 x 3 x 1 m. All animals were allowed to recover from transport for 7 days prior to swim speed trials in the flume. Fish were removed from their holding pen 24 h prior to surgery and placed in a separate pen to recover from the stress of capture. Prior to transmitter implantation, fish were quickly anaesthetized in 60 $mg.L^{-1}$ MS222 buffered with 10 $mg.L^{-1}$ sodium bicarbonate following the procedure outlined by Richardson (1985). Once anaesthetized, fish were transferred to a V-shaped surgical table. Oxygenated water was directed across the gills through a tube inserted in the mouth and the individual's head and trunk was kept moist by a cover of pre-soaked towelling.

Internal reproductive implantation involved gently inserting and pushing the transmitter anteriorly, through the urogenital opening into the oviduct. The transmitter's antenna wire (approximately 22 cm) was left hanging externally. All fish survived the surgical procedures. Fish were given 24 hr in freshwater to recover from surgery at which point they were introduced to the lower pool of the experimental flume. They were allowed a further 24 hr to acclimate, at which point the head elevation in the upper pool was randomly raised to obtain one of 3 velocities. Water velocities tested were 1.55, 1.92 and 2.55±0.05 $m.s^{-1}$. The number of fish, lengths and velocities tested depended on the volitional swimming of individuals as there was no attempt to force fish to move. Water velocity, depth and temperature were measured as described below, and the monitoring of fish movements started. After 48 hours, tracking data from the digital spectrum processor was downloaded to a MSDOS compatible computer and individuals were removed from the pool. Fork-length, girth and wet-weights were recorded and transmitters were removed, sterilized and reused.

Hydraulics

Water velocities and depths were measured at sixteen transects equally spaced along the flume's length and at the centre and sides of each transect. Velocity measurements were taken with an electromagnetic current meter (Marsh-McBirney Model 2000) positioned 3 cm above the flume floor, 3 cm below the water surface and at mid depth. Mid depth refers to a point at a vertical distance of 0.6 times water depth below the surface. Mean mid depth measurements at the centre transects were taken to represent the average water velocity through the flume. Depth measurements were taken with a metre stick (to the nearest cm) at every transect in order to determine flow rates and develop surface profiles in the flume.

Statistical analyses

The effects of varying water velocities on kinematic variables were analysed in a model-1, randomized design, one-way analysis of variance (ANOVA) with individuals as a random factor and three test water velocities. ANOVA tests incorporated contrasts for polynomial trends, specifically linear and non-linear trends. The tests were considered significant at an alpha level of 0.05.

RESULTS

The water temperature in the flume during the study period was 10.13°C±1.63 (mean±sd). Water surface profiles for flows during fish tests are shown in Figure 2. Small standing waves are evident as flows were near critical (Froude number was estimated at 0.8-1.2). Several fish were observed swimming up the flume with their bodies submerged in the water column.

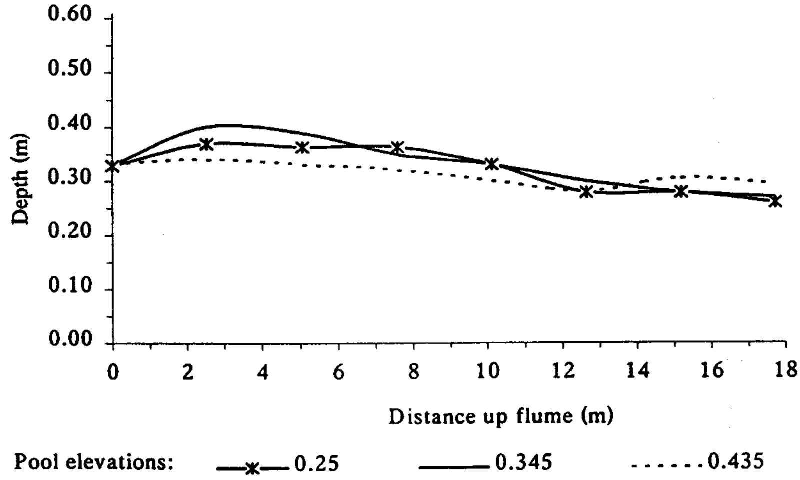

Figure 2: Depth profiles of the flume with corresponding upper pool elevations. Elevations of 0.25, 0.345 and 0.435 m represent mean velocities of 1.55, 1.92 and 2.55 m.s^{-1}, respectively.

Many fish attempted to ascend the flume during late evening and early morning periods (46.67%). At the low water velocity (1.55 m.s^{-1}, N=5) all fish tested successfully ascended the flume and reached the upper pool (Figure 3a,b). At the high water velocity (2.55 m.s^{-1}, N=4) no fish were capable of successfully ascending (Figure 3c,d). At the moderate velocity (1.92 m.s^{-1}, N=6) the success rate was 33.33% (Figure 4). As well, the number of unsuccessful passes in front of the flume entrance varied with water velocity. At high water velocities the number of unsuccessful attempts (1.20 attempts/min) were significantly lower as compared to the low and moderate water velocities (2.38 and 2.15 attempts/min, respectively).

Marked differences were noted between the maximum distances attained for each of the three water velocities (Table 1). Distances travelled up the flume significantly declined as water velocity increased ($P<0.05$, Figure 5). In addition, mean fish speed (Figure 6) and acceleration rates significantly increased as water velocity increased ($P<0.01$). However there was no apparent trend in the time spent swimming at each of the three test velocities. No significant differences were found between the size and condition factor (0.91±0.027: mean±sd) of tested fish versus the kinematic variables listed in Table 1.

Table 1: Swimming characteristics of Atlantic salmon (*Salmo salar*) in the flume.

	Mean water velocity ($m.s^{-1}$)		
	low 1.55	moderate 1.92	high 2.55
Maximum Distance (m)	18.706	13.579	10.770*
Mean Speed ($m.s^{-1}$)	2.019	2.287	3.168**
Maximum Speed ($m.s^{-1}$)	3.239	3.437	4.060
Mean Speed (Ls^{-1})	3.831	4.593	6.190**
Maximum Speed (Ls^{-1})	6.165	6.916	7.906
Mean Acceleration ($m.s^{-2}$)	1.588	1.965	2.471**
Maximum Acceleration ($m.s^{-2}$)	2.186	2.793	2.977

Values represent means±1S.E.M.
Significant differences between values for low, moderate and high-water velocity fish are indicated:*P<0.05 and **P<0.01.
L, body length.

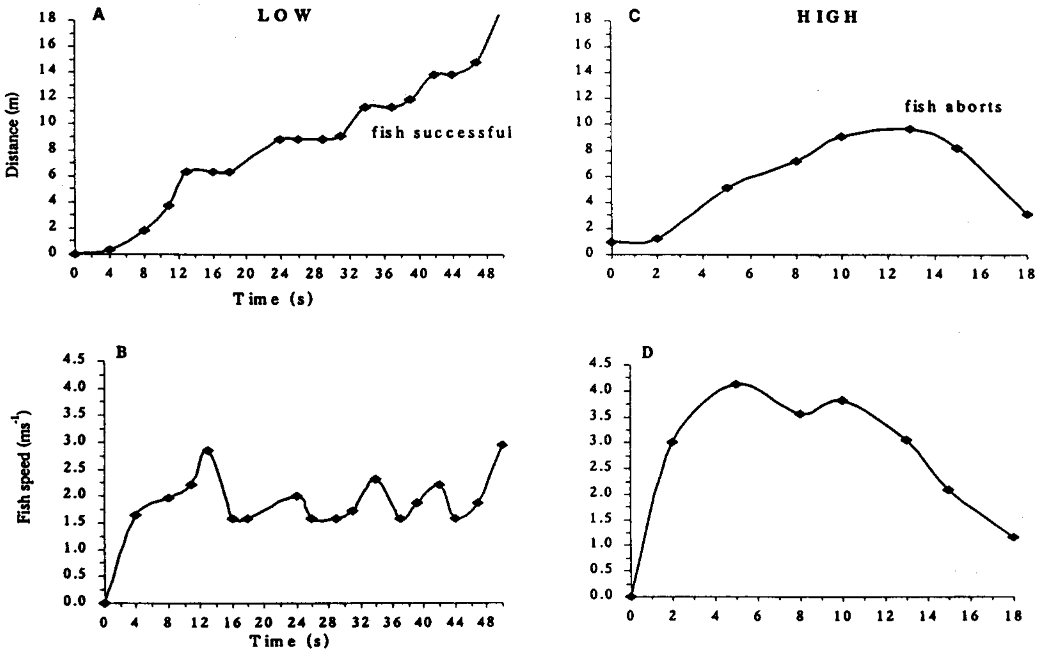

Figure 3: Movement of individual fish through the flume at a (a,b) low water velocity and at a high water velocity (c,d). Traces correspond to two separate fish.

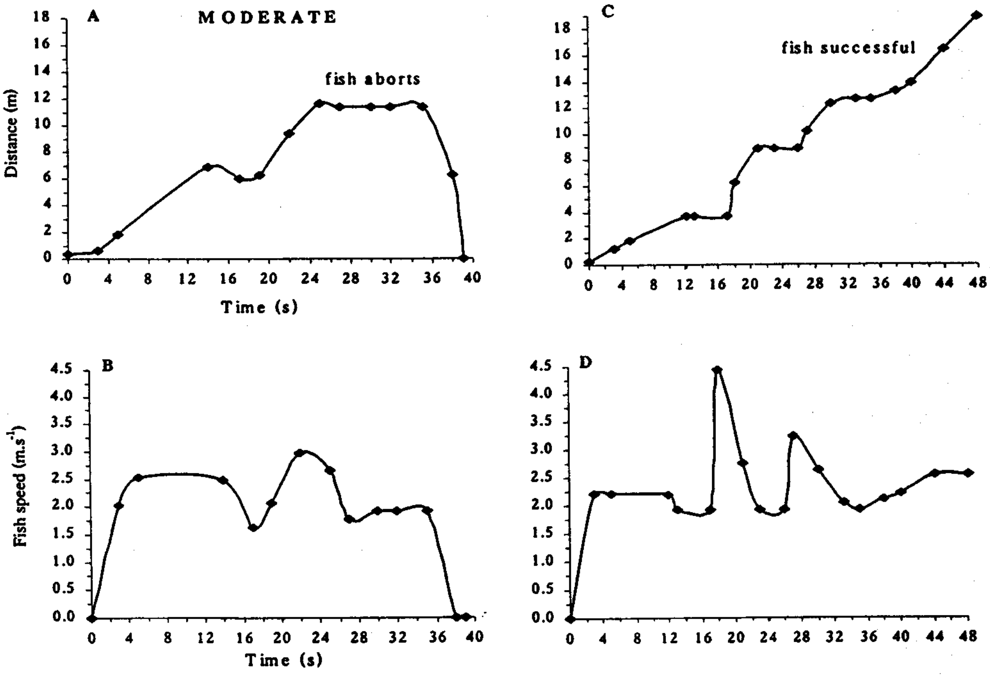

Figure 4 : One fish's unsuccessful attempt to ascend the flume at a moderate water velocity (a), and its corresponding high average speed (b). The same fish decreases its average speed (d) and successfully ascends (c).

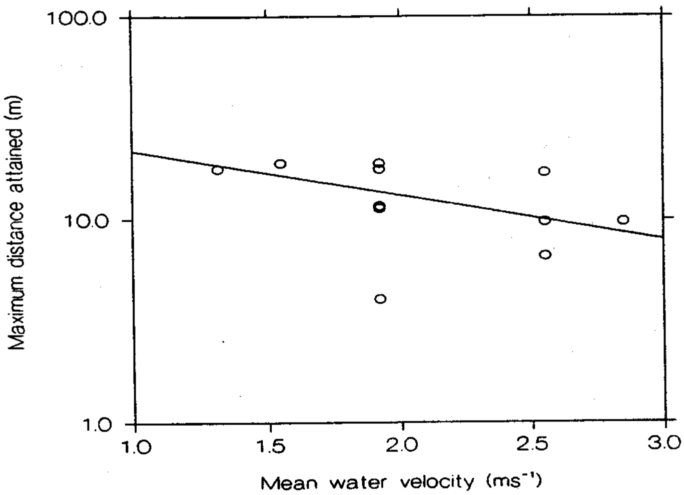

Figure 5 : Maximum distances attained in the flume versus increasing water velocity.

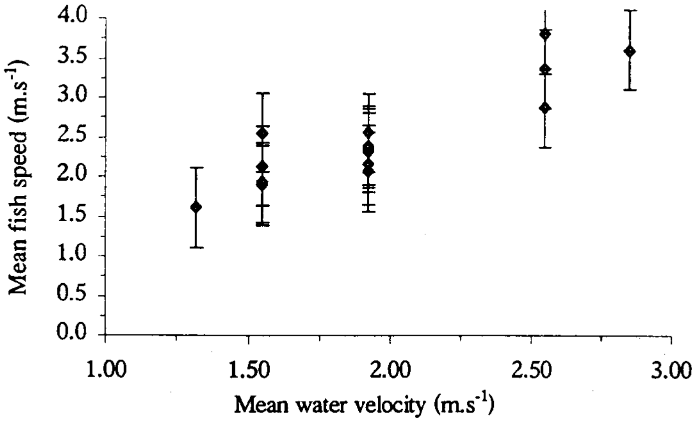

Figure 6: Average fish speed increasing with water velocity.

DISCUSSION

Early studies on high-speed swimming qualitatively investigated kinematics (Gero, 1952; Hertel, 1966). Few studies have employed procedures other than laboratory film analysis or accelerometry to study high speed swimming. High-speed cinematography has been employed to evaluate specific aspects of fast-starts in relation to predator-prey interactions, and the effects of temperature and body form on escape performance (Webb 1976; 1977). In addition, many of these fish were stimulated or shocked to induce movement so performance was likely sub-maximal. In this study, all fish movements were evaluated under natural conditions and the fish swam voluntarily. Furthermore, since many of the fish tested were active during low visibility hours (46.67%) the use of film was precluded for the analysis.

Webb (1975) has suggested that, to evaluate fast-start performance properly, data for the duration of the event, mean and maximum accelerations, mean and maximum velocities and the distance covered must all be reported. Maximum distances attained by fish declined as water velocity increased (Figure 5), which suggests a definite distance-velocity barrier resulting in fish fatigue. Beyond water velocities of about 1.92 $m.s^{-1}$, success rates fell significantly. At this water velocity some fish varied their average speed by taking periodic pauses then bursting to maximum speed to complete passage of the flume (Figure 4). These fish may be displaying a strategy to reduce energy expenditure during high speed swimming. In addition, individuals made several passes in front of the flume entrance before returning to resting pools in the lower pool. For some, this behaviour continued for several hours and even days before an individual attempted to pass through the flume. Delays in spawning migration such as this can tax energy reserves (Osborne, 1961) and reduce reproductive success (Geen, 1975).

The physiological and biochemcial mechanisms associated with burst-type exercise in fish have been extensively studied (Dobson and Hochachka 1987; Wood 1991) and there is considerable variability in the physiological responses to exhaustive exercise between individual fish. Several factors may have contributed to the variability of test fish during this study. It has been shown that hormonal conditions are closely related to the deterioration of salmon swimming muscle during spawning migration (Ando *et al.*, 1995). Extensive gonadal development was evident in several test females, likely contributing to the variance noted. Temperature acclimation may also led to changes in fish muscle recruitment (Rome *et al.*, 1984) and white muscle contractile properties (Johnson and Johnston, 1991). These studies suggest that temperature may have a significant influence on the physiological response to exhaustive exercise. Lastly, critical swimming speeds of salmon are influenced by water temperatures at 12°C and 18°C (Booth *et al.*, 1996, in press).

Ideally, fishways should be able to successfully pass all migratory species inhabiting the river in which they are constructed. Water velocities within the fishway should be less than the maximum attainable speed of all upstream migrating species. The quantification of burst swimming abilities may aid in optimizing and evaluating the design of future fishways and culvert installations. Digital biotelemetry has provided data on kinematics and strategies employed by salmon during high speed swimming which were previously not attainable using conventional mark-and-recapture procedures.

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The Effect of Velocity and Turbulence on the Growth of Periphyton in a Cobble-Bed Stream: The 5-Stone Experiment

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ABSTRACT

An experiment was conducted in a cobble-lined flume (d_{50} = 38 mm) to investigate how the growth of periphyton varies in the transition between a riffle and a run. Five stones were selected in a 3 m reach where the nearbed velocities and turbulence decreased by a factor of five. The detailed velocity field around each stone was measured and samples were taken of the periphyton growing on each stone. Biomass increased with decreasing velocity and turbulence, reaching a maximum, then decreasing as the velocity and turbulence decreased further. The community was dominated by the diatom, *Synedra rumpens*. Its density paralleled the pattern for biomass, but other species, such as *Fragilaria vaucheriae,* were highest at the minimum velocity. *Acnanthidium linearis* had approximately the same density on all stones, but in terms of relative abundance, its importance increased with increasing velocity reaching 10% of the community for the maximum velocity. The experiment showed that the mid-column velocity often used in ecological experiments is not a good indicator of the nearbed velocities. The measurements also revealed some unusual physical features of our cobble-bed stream, namely that the turbulent energy spectra decayed at a much faster rate than observed in other streams and the shear velocity was almost constant over the reach. The experiment has highlighted further work which needs to be done to obtain a better understanding of both the physics and the biology of cobble-bed streams.

KEY-WORDS: periphyton / diatom / biomass / turbulence / mean-velocity / cobble-bed-stream /riffle-run / nearbed-hydraulics / habitat-hydraulics /

INTRODUCTION

Periphyton are the main primary producers of stream ecosystems and fill an extremely important role as the source of energy for many animals (Biggs, 1996a). However, as with other biota in streams we still have little knowledge of the nearbed hydraulic fields under which they grow in natural situations and, thus, of their hydraulic habitat requirements (Biggs, 1996b; Stevenson, 1996).

Previous research on the hydraulic habitats of periphyton in gravel-cobble bed streams have generally used conventional flow meter technology (e.g., Biggs and Hickey, 1994; Biggs and Sokseth, 1996). As such, it has not been carried out at scales commensurate with that of the organisms' size, nor habitat dimensions. For example, velocities are normally measured at 0.4 of the depth from the bed (often 20 - 200 millimetres above the substrata), whereas periphyton cells are often only protruding tens of microns above the surface of the stones. This results in the point of measurement being thousands of times the organism's length away from where it actually resides and the hydraulic conditions that the organisms are "cueing" into may be quite different to what is being recorded in the freestream by the current meter.

Improvements to the technology of measuring velocities in streams have allowed us to examine the interaction between the flow field near the bed and the periphyton growing on cobbles in more detail. This paper describes a preliminary experiment which exploits these improvements in technology and advances our understanding of the important parameters in determining the growth of periphyton in different flow fields.

MATERIALS AND METHODS

Overview

The experiment was carried out in a concrete flume at NIWA's outdoor experimental facility at Silverstream, north of Christchurch, New Zealand. The flume was 10 m long by 1.6 m wide and 0.6 m deep. Its bottom was lined with cobbles with median size d_{50} = 38 mm. Water was diverted from a natural stream into the flume through a vertical sluice gate which controlled the flow. The water level in the flume was controlled by a free overfall weir at the downstream end. The sluice gate was only half the width of the flume, so flow entered the flume as a highly turbulent jet concentrated on the central half of the width. However, within 3 m of the upstream end, the flow had become tranquil and the distribution of velocity across the channel was uniform. Such a transition from swift, highly turbulent flow to slow, tranquil flow occurs in nature in the transition from riffles to pools and the opportunity that presented itself as a result of the inlet conditions was used to examine how periphyton might respond to such a transition in nature.

Experimental Design

Flow in the flume was started in late winter (1 August 1994) and was maintained at a nearly constant rate by monitoring the level using a pressure transducer at the downstream weir. Only very minor adjustments to the control gate were ever necessary. The flow was maintained for a period of 3 months (late winter and early spring) which enabled the development of a diatom dominated periphyton mat. On 27 October, 1994, five cobbles were selected along the centreline of the flume at various locations from the highly turbulent upstream region to the tranquil downstream region, as shown in Figure 1. Samples of periphyton were collected from the top of each of the stones by carefully retrieving the stone from the flume (see below for sampling methods). The stone was restored to the exact location at the completion of each sampling. Over the following week a set of velocity distributions were measured above each stone at four locations, in front, on the leading edge, on top and in the lee. For each velocity distribution, up to 20 measurements were taken in the vertical at spacings which varied from 2 mm near the cobble to 20 mm near the water surface, as shown in Figure 2.

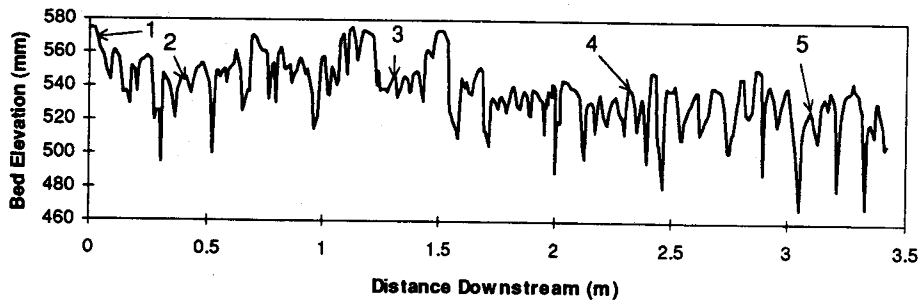

Figure 1. Profile of the bed along the centreline of the row of stones (numbered).

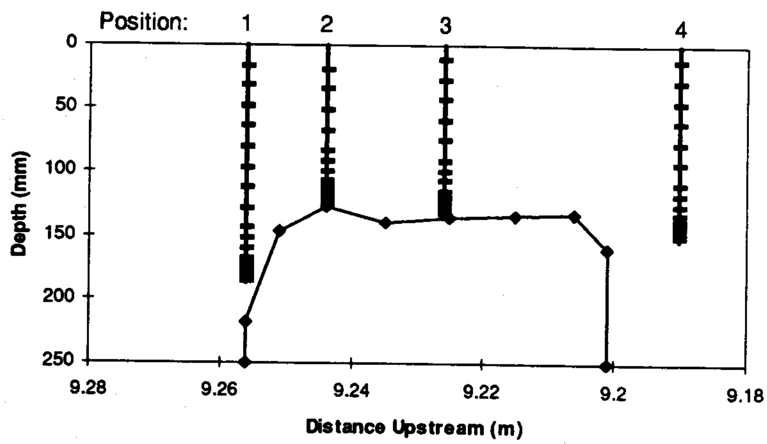

Figure 2. Profile of Stone 2 showing the velocity measurement positions.

Physical Measurements

Rails were fitted along the walls of the flume and these were carefully leveled to ensure they were horizontal. A trolley ran on the rails and spanned the width of the flume. A beam fitted to the trolley allowed precise positioning of instruments with better than 1 mm accuracy in any of the three dimensions.

The bed profile shown in Figure 1 was measured using an automatic profiler consisting of a surface follower capable of sensing both the water surface and the bed. The surface follower took measurements along the length of the flume at equispaced intervals of 9.3 mm by means of a stepping motor. Thirty such profiles were taken at 10 mm intervals across the flume on either side of the centreline.

Velocities were measured with a hot film anemometer (HFA) connected to an analog-to-digital converter which sampled at 10 Hz. Data were logged directly into a PC. Velocities were sampled for periods of at least 50 s duration, giving 500 samples for each velocity measurement.

The velocities were assumed to comprise two parts:

(1) $$u = \overline{u} + u'$$

where u is the measured velocity, $\overline{u}$ is the temporal mean over the 50 s sampling period and u' is the turbulent fluctuation in velocity. In the description that follows, the term "turbulence" will be used to refer to the standard deviation of u, which corresponds to the root mean square of u'.

Shear velocity u_* was calculated from the vertical distribution of mean velocity by fitting the data to a logarithmic profile (see, for example, Daily and Harleman, 1966):

(2) $$\overline{u} = \frac{u_*}{\kappa} \ln \frac{z}{z_0} + C$$

where κ is von Karman's constant (assumed to be 0.41), z is the distance above the surface of the cobble, z_0 is a representative cobble parameter and C is a constant.

Both the bed profile and the velocity data were analysed using routine Fourier analysis methods to compute the power spectral density (see, for example, Otnes and Enochson, 1978). The data were first transformed using the fast Fourier transform (FFT) of Singleton (1966), then windowed using the Goodman-Enochson-Otnes (GEO) window. Velocities were frequency-averaged using 2 side-lobes to smooth the spectrum. For the longitudinal bed profiles, the quadratic trend was removed from each profile, the data were transformed to the frequency domain, a GEO window was applied and the 30 transformed profiles were ensemble-averaged to yield an average power spectral density.

Biological Measurements

The top of each of the 5 stones along the hydraulic transition was sampled for periphyton. Initially it was intended to take at least three replicates from each stone (front, middle, aft) to correspond to the location of the hydraulic measurements. However, this was not feasible because the communities had developed into a thick, fragile, mat on three of the stones. Instead, a circle of known diameter (usually 25 mm) that encompassed the whole upper surface of most of the stones was scribed and all the periphyton from within the area was removed using a scalpel and pipette. The collected periphyton was then returned to the laboratory on ice and stored in the dark at -18 °C for later analysis of ash-free dry mass (AFDM), chlorophyll *a*, and species densities.

In the laboratory, the samples were homogenised and subsampled (Biggs, 1987) for the analyses. AFDM was determined by filtering 3 aliquots of the homogenase through one Whatman GFC filter to concentrate the periphyton. This was repeated 3 times to give analytical replication. The filters and associated organic matter were then dried for

24 h at 105 °C, weighed, ashed for 4 h at 440 °C, and reweighed. The samples were cooled in a desiccator before each weighing. Chlorophyll *a* was determined by filtering as for AFDM, with 3 analytical replicates also. Chlorophyll *a* was extracted from the periphyton by submersing the filter and periphyton in boiling 90% ethanol for 5 min and steeping in the dark for 24 h (Sartory and Grobbelaar, 1984). Absorbance was measured at 665 and 750 nm on a spectrophotometer. Acidification was used to correct for phaeopigments, and a chlorophyll *a* coefficient of 28.66 was used. Percentage chlorophyll *a* was calculated as chlorophyll *a* divided by AFDM (both in units of mg m^{-2}). Chlorophyll *a* is used in the text as a measure of total autotrophic biomass in the mat.

The density of taxa was determined by counting cells in subsamples using an inverted microscope (maximum magnification, 800 x). At least 300 cells were enumerated for each stone.

RESULTS

Characterisation of the periphyton's hydraulic environment

Description of the hydraulic environment has been split into three sections: coarse scale, fine scale and a comparison of the two. The coarse scale parameters describe the overall flow field and are the parameters which can be obtained by traditional current meter measurements of velocity. The fine scale parameters define the flow near the bed where the periphyton are living. Identification of these parameters requires more sophisticated velocity measuring equipment, such as the hot film anemometer.

Coarse scale

The longitudinal bed profile in the study reach (Figure 1) was very rough, with a mean height of 38 mm and a relative roughness of $z/h = 0.25$. The overall slope of the bed was $S = 0.0099$. Mean column velocities (measured at 0.4 of the depth from the bottom) decreased from 0.47 m s^{-1} at Stone 1 to 0.13 m s^{-1} at Stone 5, corresponding to the spreading of the inlet jet. The turbulence at the mid-column points was significantly correlated with the mean velocities ($r = 0.909$, $P = 0.03$). However, depths over the reach did not vary markedly (0.125 to 0.151 mm, Table 1).

Fine Scale

Figure 3 shows how the mean velocity and turbulence varied with depth from the water surface. The profiles for the 4 measurement positions across the stone are essentially the same, both for mean and standard deviation of velocity, indicating that the velocity profile did not change across the stone. Just above the stone, the mean velocity was at a minimum and increased to a maximum at ~60 mm, or half depth, thereafter remaining essentially constant. The standard deviation of velocity (i.e., turbulence) was maximum just above the stone and decreased towards the surface. The two points numbered 1 and 2 in Figure 3(b) from the profile of Position 2 on the leading edge of the stone are particularly interesting because the standard deviation of 2 is twice that of 1, even though the measurement points were only 2 mm apart. Figure 4 compares the velocity time records and spectra for these two data points. These figures are typical of all velocity measurements taken. The time records display apparently chaotic and continually changing velocities about the mean, with no trend over the 50 second duration of the sample. The spectra show a good fit of the Kolmogorov -5/3 Law from 0.5 to 2 Hz, but the energy decays at a much higher rate for frequencies greater than 2 Hz.

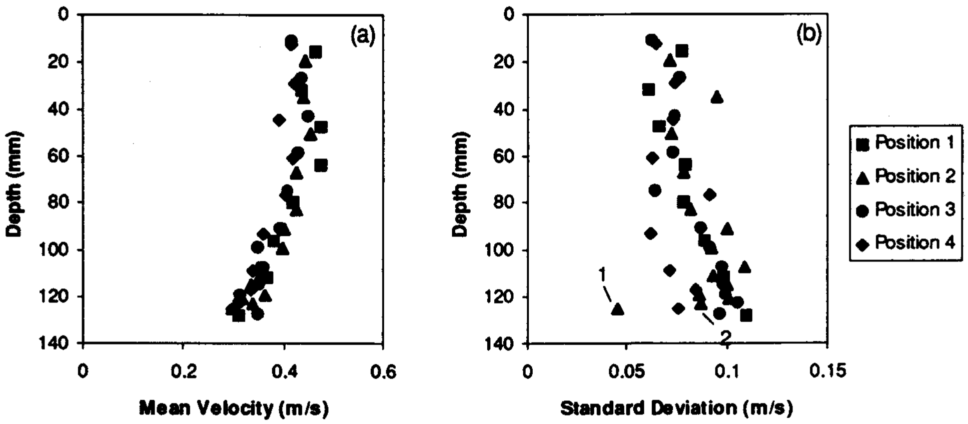

Figure 3. Distribution of (a) mean velocity, $\overline{u}$, and (b) standard deviation of velocity with depth from the water surface for Stone 2.

Table 1 summarises the flow parameters for each stone, with the measurements for all positions combined into one dataset. The shear velocity for Stone 1 was not available because of extreme scatter of the mean velocities in the vertical profile. The reasons for this scatter are discussed in the next section.

Coarse vs fine scale measurements

Traditionally, velocity is measured by stream ecologists at the mid-point in the velocity profile (0.4 of the depth from the bed). The following comparison of nearbed_{2mm} and mid-column values has been carried out to place our results in a context with others and draw attention to some biologically important differences.

First, as could be expected from the shape of the velocity profiles (Figure 3), nearbed_{2mm} velocities were much lower than mid-stream velocities (except for Stone 1). However, there was no significant correlation between mid-column and nearbed_{2mm} velocities ($r = 0.747$, $p = 0.147$) with the magnitude of the differences between locations (i.e., mid-column/nearbed) varying greatly among the stones (-15% - 98%). This indicated that the mean velocity was not a good indicator of nearbed_{2mm} velocities in this section of simulated stream. Second, conditions in the mid-column region were far more uniform than was recorded near the bed (Table 1). Thus, over our hydraulic transition zone there was only a 2.5 fold change in mean column velocity vs. a 5.2 fold change in nearbed_{2mm} velocity, and only a 3.0 fold change in mid-column turbulence vs. a 5.5 change in nearbed_{2mm} turbulence. The high bed roughness greatly increased the complexity of the nearbed flows resulting in high variations in velocity and turbulence over small distances.

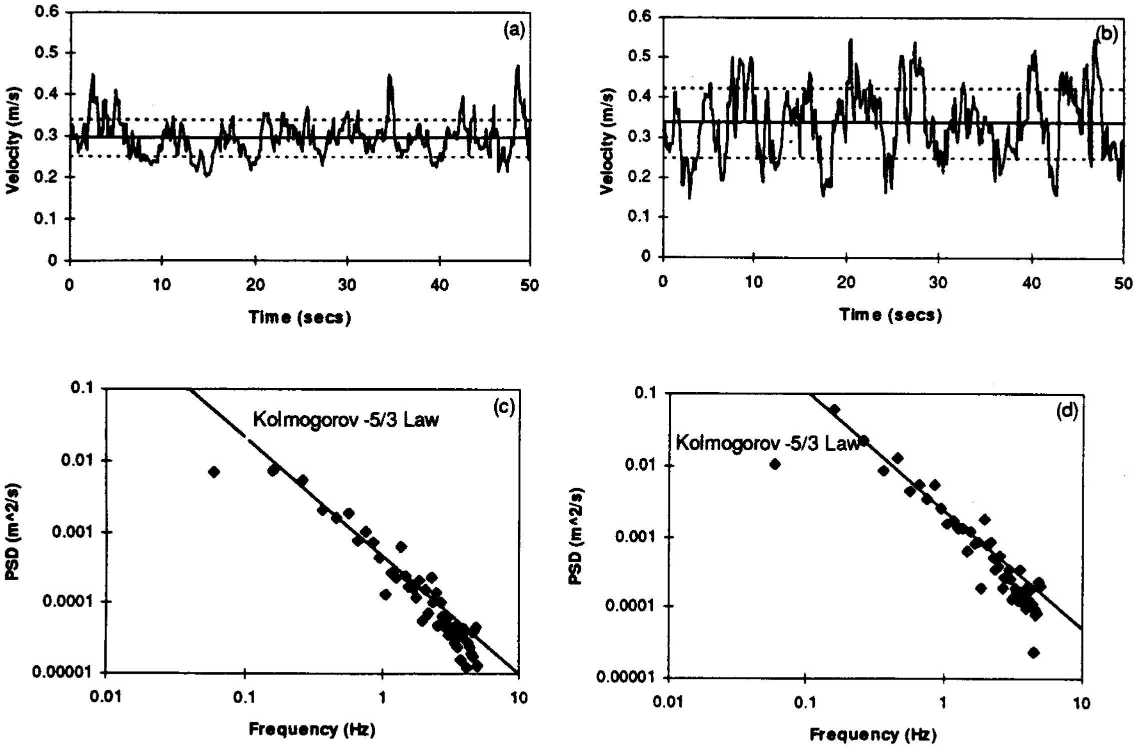

Figure 4. Comparison of time series (a) and (b) and corresponding spectra (c) and (d) for points at Position 2 of Stone 2 at 2 mm above the stone (a) and (c) and 4 mm above the stone (b) and (d). Horizontal lines in (a) and (b) represent mean (solid line) and ± standard deviation (dashed line). Lines in (c) and (d) represent the Kolmogorov -5/3 Law.

The shear velocities calculated from the slope of the velocity distributions near the bed (i.e., from Equation (2)) were remarkably constant (except for Stone 3), considering the vastly different velocity regimes at each stone and the different colonisation of periphyton. We had expected to find a decrease in shear velocity (indicating a decrease in shear stress) as the flow became more tranquil. The reason why this did not occur is not understood and further work is required on this.

Periphyton Responses

The nearbed$_{2mm}$ data presented in Table 1 were used to represent the average hydraulic environment for the periphyton and the responses were analysed as a function of these data.

Biomass

A clear gradient was observed in the amount of periphyton along the hydraulic transition zone in the flume. Only a fine green film, with the occasional mucilaginous balls of *Nostoc* sp., occurred on the stones at the head of the zone

Table 1. Summary of flow parameters for each stone in which the mean and standard deviation of velocity at both mid-column (0.4 of the depth from the bottom) and nearbed$_{2mm}$ (i.e., 2 mm above the top of the stone) are the values averaged over the 4 measurement positions. Shear velocity was calculated using Equation (2) with $z_0 = d_{90} = 55$ mm.

Stone No	Dist d/s (m)	Depth (m)	Velocities (m s^{-1}) Mid-Column Mean	Mid-Column Std Dev	Nearbed$_{2mm}$ Mean	Nearbed$_{2mm}$ Std Dev	Shear
1	0	0.147	0.362	0.082	0.417	0.109	n/a
2	0.38	0.135	0.399	0.083	0.323	0.093	0.022
3	1.32	0.125	0.388	0.061	0.199	0.050	0.030
4	2.33	0.150	0.225	0.047	0.184	0.036	0.020
5	3.06	0.151	0.159	0.027	0.080	0.020	0.022

where velocities were relatively high. Moving downstream, a marked increase in growth occurred over the whole bed as the water became more tranquil and the communities became thick, fluffy, brown mats, dominated by diatoms, with the occasional tuft of green algal filaments.

The total biomass (AFDM), amount of live algae (chlorophyll *a*), and the percentage of the total biomass as live algae (% chlorophyll a) all displayed a similar response to nearbed$_{2mm}$ velocity (Fig. 5a). Values were intermediate at the low velocity of 0.08 m s^{-1} (Stone 5) and rose to a maxima at 0.184 m s^{-1} (Stone 4). A very small, 8%, increase in nearbed$_{2mm}$ velocity to 0.199 m s^{-1} (Stone 3) then resulted in a 40% reduction in AFDM, but > 70% reduction in chlorophyll *a*. Both AFDM and chlorophyll *a* were low for the higher velocities (> 0.25 m s^{-1}) on Stones 1 and 2 at the head of the transition zone. The low % chlorophyll *a* on these stones indicated that the organic matter that was present at these velocities had a low content of live algae.

Using our hot-film anemometry system we were, for the first time, able to get a measure of turbulence around the periphyton mats (Fig. 5b). It is clear that the AFDM, chlorophyll *a* and % chlorophyll *a* responses were very similar to that for velocity. Indeed, as noted earlier, we found that mean nearbed$_{2mm}$ velocity and turbulence above the stones were highly correlated. Because of this correlation we present an analysis of only the mean velocity component in the following section on species performances.

Because shear velocity was reasonably uniform through the transition zone (Table 1) none of the biomass parameters displayed any relationship with this parameter.

Species densities

Thirty taxa occurred in the mats at densities > 100 cells mm^{-2}. The unicellular diatom *Synedra rumpens* dominated the assemblages on three of the five stones. Five main patterns in density occurred for the common taxa as a function of velocity. In the first pattern, five taxa had their peak densities at the lowest nearbed$_{2mm}$ velocity (0.080 m s^{-1}) and then progressively declined in density as velocity increased (Figure 6). *Fragilaria vaucheriae* dominated the communities in this low velocity range. The most common pattern was for peak densities at nearbed$_{2mm}$ velocities of 0.184 m s^{-1} (11 taxa) and parallels the overall pattern for biomass (Figure 5). *Synedra rumpens* was by far the most abundant at this nearbed$_{2mm}$ velocity with densities peaking at > 10,000 mm^{-2}. The third group had peak densities at 0.199 m s^{-1}. At this velocity densities were becoming quite low and there were only two taxa with this pattern. The remaining taxa had little pattern in their densities (Group 4) or only occurred at one of the five velocities (usually 0.184 m s^{-1}) (Group 5).

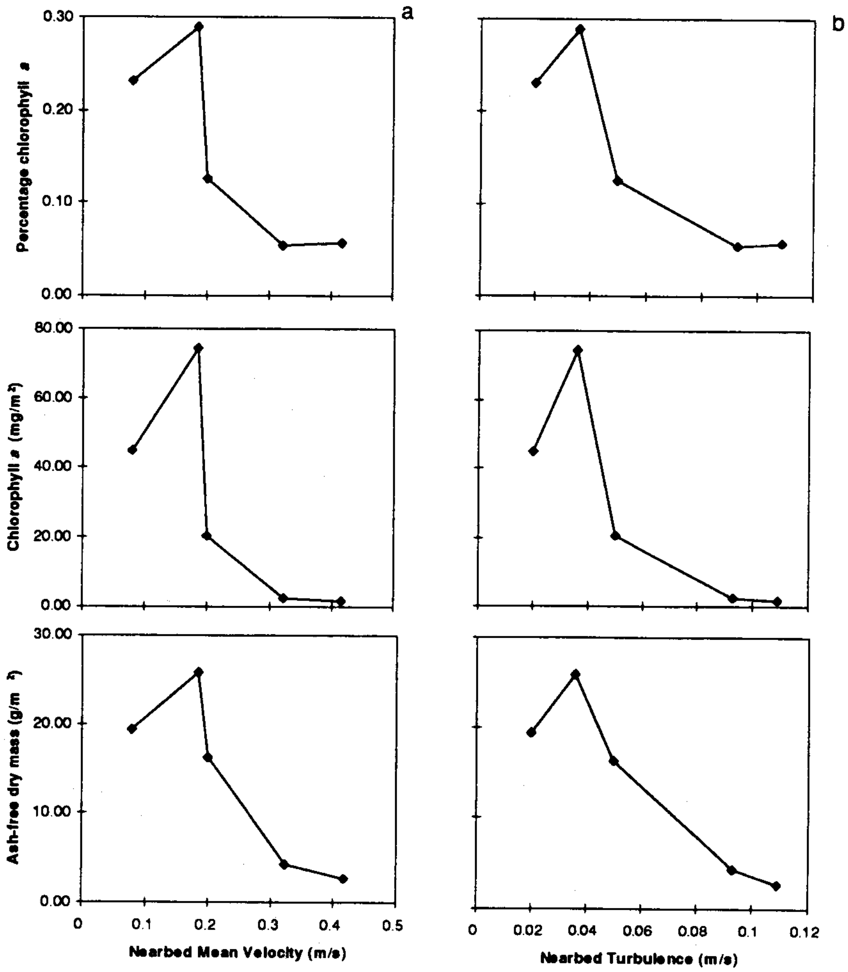

Figure 5. Variation of biomass parameters with nearbed$_{2mm}$ (a) mean velocity and (b) turbulence.

The shape of the distribution of densities as a function of nearbed$_{2mm}$ velocity varies when analysed as relative abundance (Figure 6). Of particular note is the broadening of the peak for *Synedra rumpens*, indicating that even though densities decreased from > 10,000 mm^{-2} at 0.184 m s^{-1} to < 500 mm^{-2} at 0.323 m s^{-1}, this taxa retained a relative abundance of 30 - 35%. Thus, this taxon was the strongest competitor over quite a wide range in velocities. However, *Achnanthidium linearis* did not change greatly in density as the velocities increased, but in relative terms it did increase in importance at the highest nearbed velocity (0.417 m s^{-1}) and here contributed over 10% of the community. This indicated that it was much more resistant to high velocities. Lastly, under low velocities *Fragilaria* *vaucheriae* out-perfomed *Synedra rumpens*. These results clearly demonstrate habitat preferences and associated niche separation for stream periphyton along a gradient of hydraulic stress.

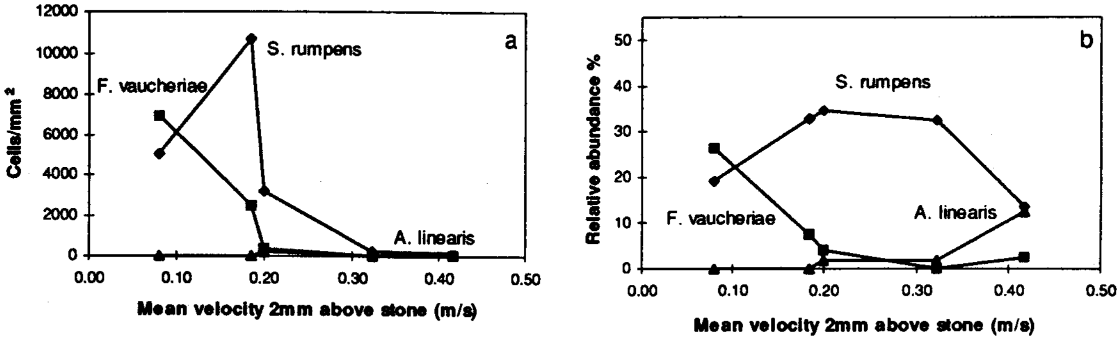

Figure 6. Variation with mean velocity of (a) abundance and (b) relative abundance of the major periphyton species on each stone.

DISCUSSION AND CONCLUSIONS

General

This study of a riffle - run hydraulic transition provides the first accurate record of near-bed flow conditions to which periphyton communities can be subjected in streams and documents their responses. The bed sediment comprised cobble-sized particles with a very high relative roughness and this strongly affected velocity and turbulence fields. We found that the average water column velocities (and turbulence) were not always good surrogates for the nearbed hydraulic conditions in streams. A unimodal distribution of periphyton biomass as a function of nearbed velocities with a peak at a nearbed$_{20mm}$ velocity of 0.18 m s^{-1} occurred. The shape of this distribution agreed with theoretical predictions based on a trade-off between stimulation of production through enhanced nutrient mass transfer and retardation of accrual as drag begins to exceed attachment and tensile strength of the community. Only 33% of the 11 common taxa had densities which reflected changes in biomass as a function of velocity. 50% of the taxa had no clear velocity preference and 17% were more abundant in the lowest velocities. These results demonstrated fairly specific hydraulic habitat preferences for some taxa, but also considerable overlap for others

Velocity Measurements

Using an HFA to measure velocity in highly turbulent flow is difficult because bubbles and coarse particulates in the water attach themselves to the probe and cause drift in the measurements. In a laboratory setting the bubbles may be visible through the side walls of the flume, but in our field application, they were almost impossible to detect even using a periscope to pierce the water surface. We could infer their presence only by appearance of drift in the data. Unfortunately, detection of drift is not always easy in the field, although it becomes obvious once detailed numerical analysis begins. This was a particular problem for the Stone 1 and resulted in our inability to obtain sufficient measurements in the depth to calculate the shear velocity (see Table 1). The problem with bubbles was not as severe near the bed as it was in the freestream; however, if the periphyton mats had been thicker and protruded greater than 2 mm above the stone (the distance from the tip of the HFA to the hot-film itself), we expect that there would have been measurement problems with these data as well. These problems seem insurmountable, so in future experiments an alternative means of measuring velocity will be employed.

Decay of Turbulent Energy

The decay of turbulent energy with frequency shown in Figures 4(c) and (d) matches the Kolmogorov -5/3 Law for the range from 0.2 to 2 Hz, but the energy decays at a faster rate for frequencies higher than this. This result is contrary to findings of other researchers (e.g., Isawa and Asano,1980; Grinvald and Nikora, 1988) who have shown that the Kolmogorov Law is applicable in a wide range of stream-types from slow, deep, sand-bed streams to swift, shallow, gravel-bed streams. Velocities near the bed in the 5-stone experiment varied from 0.417 to 0.080 m/s (Table 1). Thus, for a frequency of 2 Hz, corresponding to the point where the energy begins to decay faster than the theory, the length scales of the flow are from 200 to 40 mm. These represent the maximum length scales. For higher frequencies, the lengths reduce. There are two such length scales in the 5-stone experiment, the depth (~ 150 mm) and the size of the cobbles (~38 mm). (Another factor is the periphyton themselves, but for this experiment the length of the periphyton was only of order 5 mm) Further experiments are planned to address the question of why the turbulent energy in our flume at Silverstream decays faster than the theory predicts and whether it is because of the depth or the size of the cobbles.

Effect of Periphyton on Flow

Nikora *et al* (submitted) found that the growth of periphyton on a concrete bottom had a substantial effect upon turbulence and roughness. A question that arises is whether periphyton growth on cobbles could affect turbulence? For the 5-stone experiment the length scales of the flow were one or two orders magnitude greater than the length of the periphyton. Therefore, it is unlikely that periphyton had any effect on the turbulence at the frequencies being measured. Indeed, periphyton with length scales of order 1 mm in velocities of order 0.3 m/s would be expected to affect the flow at frequencies of order 300 Hz or greater, well beyond our present measuring capability. Further experiments in a similar flow field are underway in which the effect on turbulence of filamentous green algae with lengths of order 50 mm will be examined.

Effect of flow on periphyton

Figures 5 and 6 show that flow has a major effect on periphyton, with Stones 3, 4 and 5 in the tranquil region having more than an order of magnitude more biomass than Stones 1 and 2 in the high velocity region. However, the substantial increase in chlorophyll *a* between Stones 3 and 4 with a minor increase in mean and standard deviation of velocity means that there must be other factors affecting the periphyton as well. One explanation is that insufficient periphyton samples were taken and that the measurements reflect natural spatial variation in periphyton. However, it is unlikely that this explains all of the enormous difference. Other factors (from Table 1) are that Stone 3 has the minimum depth and the maximum shear velocity. Perhaps the flow regime in between Stones 3 and 4 is critical in terms of the habitat for periphyton and the small physical difference between the stones represents a major habitat change.

Future Work

This experiment has raised a lot more questions than it answered and has generated a three year research programme into nearbed hydraulics and periphyton in cobble-bed steams. Important physical questions to be addressed include: why the turbulent energy spectra decay so rapidly and why the shear velocity is constant throughout the reach. Acquisition of an acoustic Doppler velocimeter which remotely samples velocity in three dimensions at up to 25 Hz will allow us to address these matters. Of course, the experiment barely touched upon the interaction between flow and periphyton and there are a multitude of questions which arise such as: the preference of individual species for different velocity regimes; the importance of nearbed vs freestream velocity and turbulence; how periphyton adapt to a changing flow regime, to name just a few.

ACKNOWLEDGMENTS

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FRICTION FACTORS FOR VEGETATION

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ABSTRACT

Increase of roughness with depth of flow as a result of increasing momentum absorbing area "MAA" for non-submerged tall vegetation, and decrease of roughness as a result of streamlining were evaluated by testing individual pine and cedar saplings in water flume. In order to model the effect of vegetation flexibility for short submerged grass roughness, simulated plastic strips were embedded in a flume floor. For both submerged and non-submerged conditions, a dimensional analysis, supported by experimental results was applied to model the resistance to flow in flood plains and vegetative zones of natural waterways.

KEY-WORDS: Friction Factor / Roughness / River / Vegetation / Momentum Absorbing Area / Flexibility / Stiffness.

EXPERIMENTAL RESULTS AND ANALYSIS

A. Non-Submerged Tall Vegetation

Vegetation on flood plains and rivers banks are commonly assumed to behave as rigid roughness (Chow, 1954; Barnes, 1967; Arcement and Schneider, 1984). This can lead to a large error in the relationships between the velocity and the momentum absorbed by vegetation and a clear violation of the fundamental fluid mechanics. Experiments were performed to show the effects of non-rigidity and inundation depth of vegetative roughness for the non-submergence condition in flood plains and vegetated zones of rivers. Individual pine and cedar tree saplings were used to model the resistance to flow in a water flume, and to determine the amount that streamlining decreases the drag coefficient and reduces the "Momentum Absorbing Area (MAA)". According to the results, the Darcy-Weisbach friction factor varies greatly with the velocity due to bending, and with depth due to a direct increase of the MAA .

A dimensional analysis, supported by experimental results, is developed in order to obtain a relationship between roughness conditions (i.e., density and flexural rigidity) and flow conditions (i.e., velocity and depth). If it is assumed that resistance to flow in vegetation zone is a function of vegetation density and rigidity, as well as velocity and depth of flow, Then the final form of the dimensionless parameters could be (Fathi and Kouwen, 1995):

(1)
$$C_d\left(\frac{A}{a}\right) = f_4\left(\frac{\rho V^2 {y_n}^4}{EI}\right)$$

where A = "momentum absorbing area, MAA" which is a function of one side area of leaves and stems, a = horizontal area of channel bed covered by the vegetation, C_d = drag coefficient, ρ = density of water, V = average channel velocity, y_n = normal depth of flow, and EI = flexural rigidity.

Using a force equilibrium for uniform flow in a vegetated reach (Petryk and Bosmajian, 1975) , the average boundary shear stress (τ_o) can be expressed as:

(2)
$$\tau_\circ = \frac{F_D}{a} = \frac{1}{2} C_d\left(\frac{A}{a}\right)\rho V^2$$

where F_D is momentum absorbed by vegetation.

The Darcy-Weisbach friction factor (*f*) is defined as (White, 1974):

(3)
$$\frac{f}{8} = \frac{V_*^2}{V^2}$$

where $V_\bullet$ is shear velocity and V is channel stream velocity. Since by definition $\tau_\circ = \rho V_*^2$, using equation (2), the friction factor (f) will be:

(4)
$$f = 4C_d\left(\frac{A}{a}\right)$$

The plots in figure 1 show the variation of the friction factor (f) with submergence depth (Y_n/h , where h = 300 mm, the height of models) and velocity for cedar and pine models. The curves show up to a 40 fold increase in the friction factor from shallow, low velocity stem flow to the nearly submerged foliage flow. Similarly, there is up to a 4 fold decrease in the friction factor when the velocity is increased from 0.1 to 0.8 m/s. In practice, the MAA for each depth is predicted by sampling in a vegetated canopy and it can be determined as a function of flow depth (y_n) for each specimen. The variation of MAA with depth of submergence has also been shown in figure 1 with the same vertical scale as the friction factor (f). In figure 1, stiffness (EI), MAA, and velocity appear to be the dominant factors for all cases.

Non-submerged Experimental Procedure

A system of load cells and a knife edge table were designed and installed beneath of the flume to instantaneously measure the drag force and the center of efforts. The recorded data were averaged for the same 30 seconds duration as the velocity measurements. The tree models in channel bed were attached to the top of the table through a sealed hole. For each of the pine or cedar models, experiments were run for five depths of 60, 120, 180, 240, 300 mm. At the 60 mm depth, the flow was only stem flow and at the 300 mm depth, the model was nearly submerged. The 300 mm depth experiment was repeated seven times with different models of both pine and cedar in order to estimate the effect of experimental repetition as well as the variation of model samplings (Fathi and Kouwen, 1995).

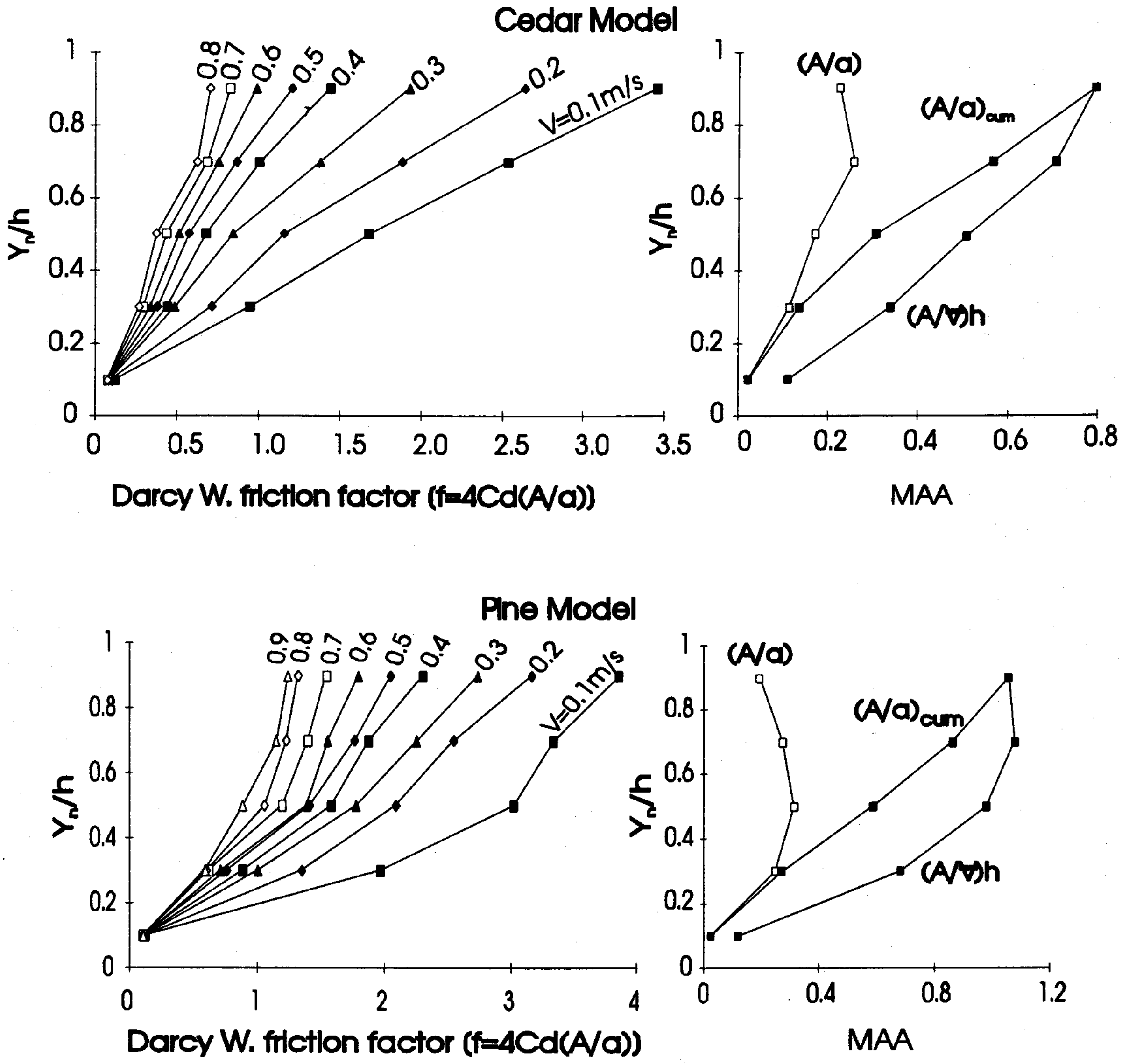

Figure 1: Variation of friction factor (*f*) with velocity, depth, and MAA (cedar and pine models)

Photographs of one side of leaf and stem area of models were scanned, digitized by using Easi/Pace software ("Easi User's Guide", 1994) in a UNIX workstation, and recorded as cumulative MAA per unit horizontal area, $[(A/a)_{\text{cum}}]$ for each flow depth (Figures 1). The $[(A/a)_{\text{cum}}]$ is normalized with relative depth of submergence (Y_n/h) and shown as $[(A/\nabla)h]$ in Figures 1, where $\nabla = aY_n$.

B. Submerged Short Vegetation

The resistance to flow in vegetated channels has been modeled by relative roughness approach in a manner very similar to widely accepted resistance relationships for rigid roughness in pipes and channels. Vegetation bends when subjected to shear, adding the additional requirement of predicting the effective roughness height as a function of the flow. The governing relationships are described in detail in Kouwen (1992). According to above model, the friction factor (f) is calculated based on the relative roughness through the logarithmic formula:

(5) $$\frac{1}{\sqrt{f}} = a + b\log_{10}\left(\frac{y_n}{k}\right)$$

where k = the deflected height of roughness, a and b = coefficients dependent upon the extent to which the vegetation is bent. According to the measurements of flow over flexible plastic roughness, the roughness height (k) is a function of the amount of drag force that flow exerts on vegetation, and is shown to be (Kouwen, 1992):

(6) $$k = 0.14h\left(\frac{(MEI/\tau_{\circ})^{0.25}}{h}\right)^{1.59}$$

where h = stem length, MEI = stiffness, (M is number of stems in unit horizontal area). For equation (6), the boundary shear stress ($\tau_{\circ}$) is computed by,

(7) $$\tau_{\circ} = \gamma\, y_n S$$

where γ = water specific weight, and S = slope of energy line. The a and b coefficients are listed in Kouwen (1992). The amount of a and b were computed based on ratio of shear stress to a critical shear velocity V_{*crit} for each type of vegetation. In general

(8) $$a,b = fn\left(\frac{V_*}{V_{*_{crit}}}\right)$$

where the critical shear velocity is given by

(9) $$V_{*_{crit}} = \min\ of\ (0.028 + 6.33MEI^2, 0.23MEI^{0.106})$$

The critical shear stress is the total stress required to change the aggregate boundary roughness from a rough (erect and waving) to smooth (flattened or shingled) wavy boundary.

Submerged Experimental Procedure

Equation (5) states that for a constant y_n, a smaller k (higher deflection) results in a lower f, implying a smoother boundary. To verify Equation (5), different sets of experiments were conducted at the University of Waterloo laboratory in a 9m flume with a width of 620 mm. Simulated plastic strips of 100 mm and 150 mm heights and 5 mm width were subjected to a series of flow rates at different slopes. The experiments were repeated at different depths and the results are shown in Figure 2. As a result of an increase in channel slope and velocity, the plastic elements were deflected and imposed a smoother surface roughness against the flow. Similarly, according to the definition of relative roughness, the friction factor (f) is decreased as the depth of flow increases.

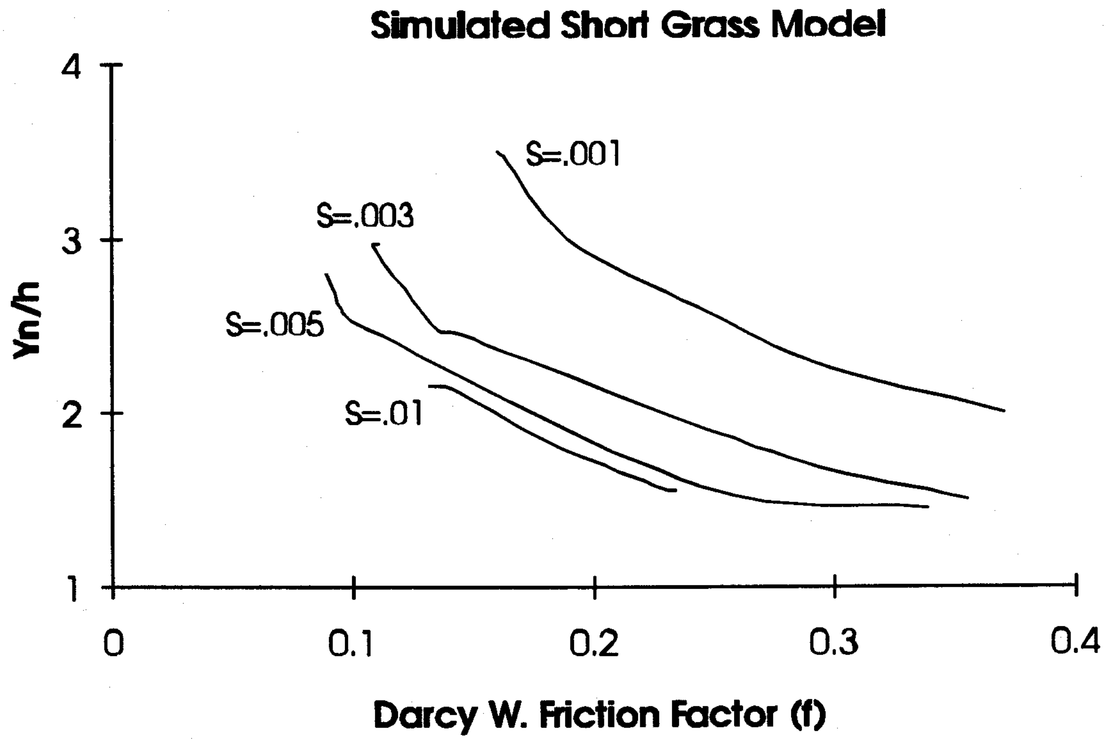

Figure 2: Variation of friction factor (f) with channel slope and flow depth (simulated short grass model)

CONCLUSION

The isolated pine and cedar tree saplings, and simulated submerged grass were imposed to the flow of water in a water flume to model roughness for non-submerged and submerged conditions respectively. Experimental measurements confirm the strong effect of vegetation flexibility as the velocity increases. The results of repeated experiments with different models were similar enough to allow a mathematical model to be developed based on dimensional analysis for each condition. Evaluation of the curves reveal the following:

(a) The friction factor (f) is directly proportional to the depth of flow and inversely proportional to the velocity for non-submerged conditions. As soon as the vegetation becomes submerged, a decrease in the friction factor is obtained due to a decrease of relative roughness.

(b) Regardless of tree species, or foliage shape and distribution, depth variation of the friction factor (f) is directly proportional to the depth variation of MAA, so density of vegetation is always a dominant parameter for the non-submerged condition.

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Geomorphology and habitats

La géomorphologie et les habitats

FLUVIAL GEOMORPHOLOGY AND FISH HABITAT: IMPLICATIONS FOR RIVER RESTORATION

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ABSTRACT

Fluvial morphology provides the physical framework for all fish habitat, but it is often some relatively minor or transient aspect of the overall morphology (over-hanging banks, small backchannels, early successional vegetation) that provide the critical or most limiting fish habitat. Only by preserving the fluvial processes responsible for any given morphology can these small features be maintained and renewed over the long term. The primary factors determining the channel morphology and fluvial processes along a river reach are the water and sediment supply from upstream and the local geology. Local climate and floodplain vegetation are the most important secondary factors. Engineering projects, floodplain land use, upstream land use, and global climatic changes all potentially affect the primary and secondary factors and can thereby cause widespread morphological changes. Recent studies of the effects of deforestation on flood magnitude indicate larger and more widespread effects than was previously believed. Global climatic change is now seen as having pronounced effects on flood magnitude. Floodplain clearing is also recognized as having significant morphological effects. The scientific basis for predicting the resulting morphological changes is poor. The direction of change is sometimes obvious, but in most instances little can be said about the rate of change or the eventual magnitude of the change. Predictions about fish habitat changes are even more tenuous.

There have been extensive efforts to restore fish habitat in channel reaches that have been degraded by morphological changes. The success rate for fisheries habitat restoration is quite low if measured in terms of the physical survival of the works. If measured on the basis of increased fish productivity, it could be lower yet. Re-naturalizing river reaches by providing space for natural processes to re-establish channel morphologies and associated aquatic habitat is proving more successful. However, this strategy can only deal with locally-degraded river reaches. If the major factors determining downstream morphology have been altered, mitigation or restoration is not generally feasible.

INTRODUCTION

Fluvial geomorphology is the branch of earth science devoted to classifying and understanding the land forms associated with or created by rivers. It therefore provides the physical framework for the ecology and all habitats associated with rivers, both fluvial and terrestrial. In general it is the fluvial geomorphology, combined with other factors, such as the regional climate, local geology, etc. that determines the resulting ecology and habitat. There are however also feedback mechanisms. Bank and floodplain vegetation is an important aspect of the terrestrial ecology, associated with a river, which can affect geomorphological parameters, such as channel width and channel pattern. Beavers are also well known for their ability to create their own channel morphology in small streams.

Many fluvial landforms are easily and often unwittingly changed by man. The first indication of a change is often far removed from the fluvial morphology of a river; it may be the decline or disappearance of a particular fish species. Only a very detailed analysis might later reveal the true cause. It may be that some engineering project or a land use change is interfering with a hydrologic factor that, in turn, affects the local fluvial morphology and thereby eliminates a critical habitat of that fish species.

Some of the most ecologically important fluvial land forms have a very short life expectancy on a geological time scale. Examples are overhanging banks, which eventually collapse, debris jams that rot away, become buried in the floodplain, or float away in a large flood, side channels that become filled in with sediment and become part of the floodplain or the floodplain itself, which will eventually turn into a terrace unless it is renewed by being eroded away and thereby re-built. The long term survival of all such landforms requires the preservation of the processes that create them and provision of space in which the processes can take place. This may appear obvious and trivial, yet it is almost universally ignored.

The study of fluvial geomorphology has recently become quite popular in hydrotechnical circles, because of the realization that much valuable management and design information can be extracted from the careful study of fluvial features. Floodplains often contain detailed records of past bank erosion, channel shifting, rates of sediment deposition, flood level and ice jam levels. Some of the most reliable sediment transport data have been extracted from delta progradation rates. In this context one cannot over-emphasize the usefulness of new and old air photos and maps, an often neglected or overlooked resource.

The objectives of the present paper are to point out the potential for extremely widespread, inadvertent man-made, morphological changes and to demonstrate the key role of morphological considerations in the management of fluvial habitats. World-wide, most major rivers and many small ones are now subject to some degree of man-made morphological changes and, if they are not, chances are that they will be soon. These changes invariably affect habitat and the vast majority of effects will be negative. Mitigation will be difficult, particularly if it is not based on the best possible understanding of the underlying processes.

DEFINING MORPHOLOGY

There is an extensive literature on how to define and measure the morphology of a river reach. Our approach is summarized in *Kellerhals et al., 1976*. The procedures and classifications described in *Newbury and Gaboury, 1993* and *Ministry of Natural Resources, Ontario, 1994* are also widely used in some parts of Canada. Besides the obvious quantitative measurements such as width, depth, drainage area, floodplain slope, channel slope, mean flow, various flood flows, bed and bank materials, sediment transport rates, etc., there are other parameters, such as channel plan form, degree of channel splitting, cross-sectional shape, etc., that require classification systems which are invariably somewhat subjective in their application. The system developed by the first author is described in *Kellerhals and Church, 1989*. *Figure 1*, out of that reference, illustrates the basic elements. A classification system developed by *Rosgen (1994)* is widely used in American fisheries circles.

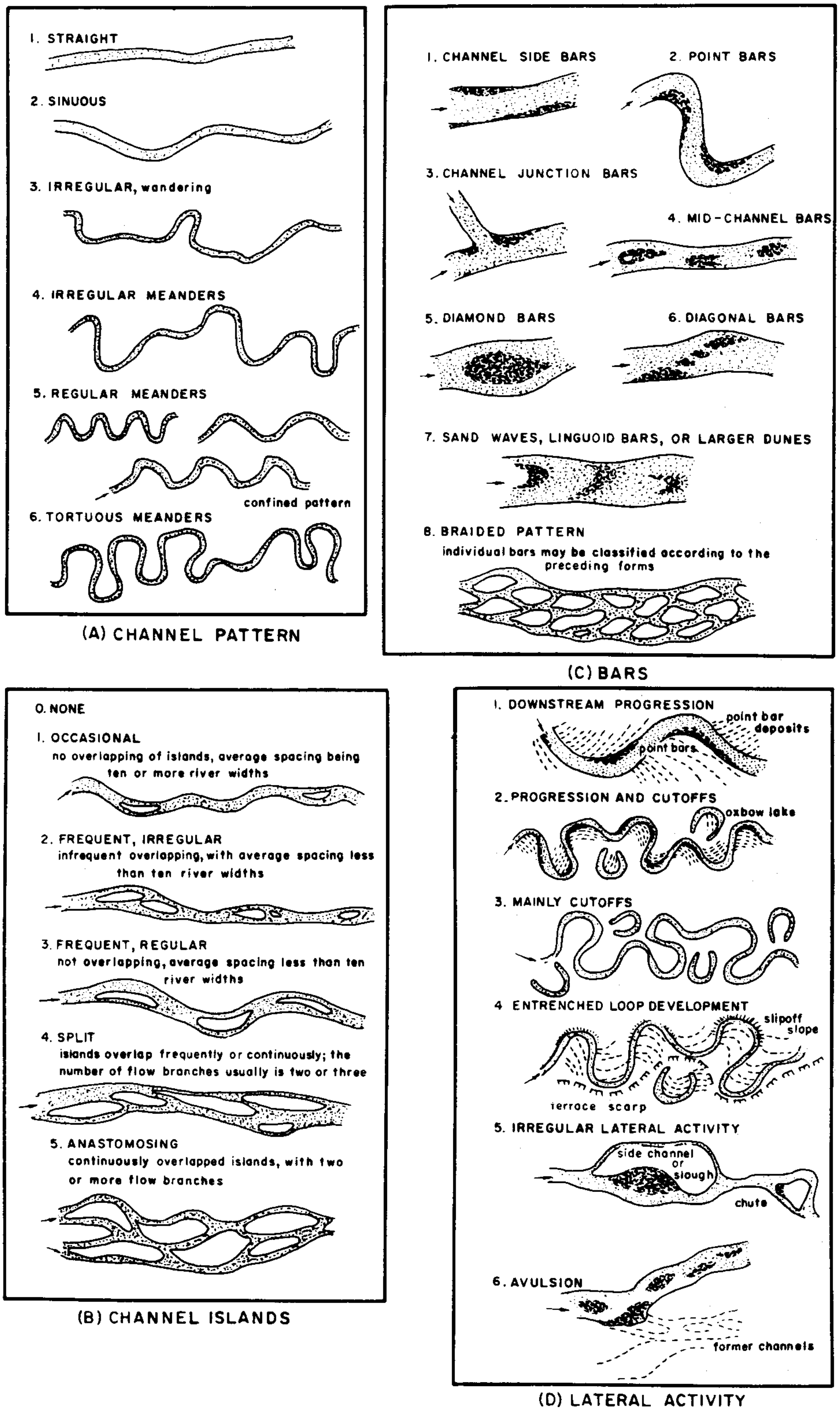


Figure 1: Classification of planform features or river channels *(from Kellerhals and Church, 1989).*

FACTORS DETERMINING MORPHOLOGY

A source of water carrying some sediment into a lake basin will eventually replace the lake with a river and its associated floodplain (as illustrated schematically on *Figure 2*). Nothing else is needed to create a basic equilibrium or "regime" river and the "experiment" is repeatable in a statistical sense. For a given hydrological and sedimentological regime of water and sediment flowing into a lake basin, the fluvial morphology and fluvial processes that will eventually develop on the infilled lake will generally always be the same. Many details such as the exact position of the channel or channels at any one time will be subject to random variations and other factors, such as the floodplain vegetation, will introduce systematic differences.

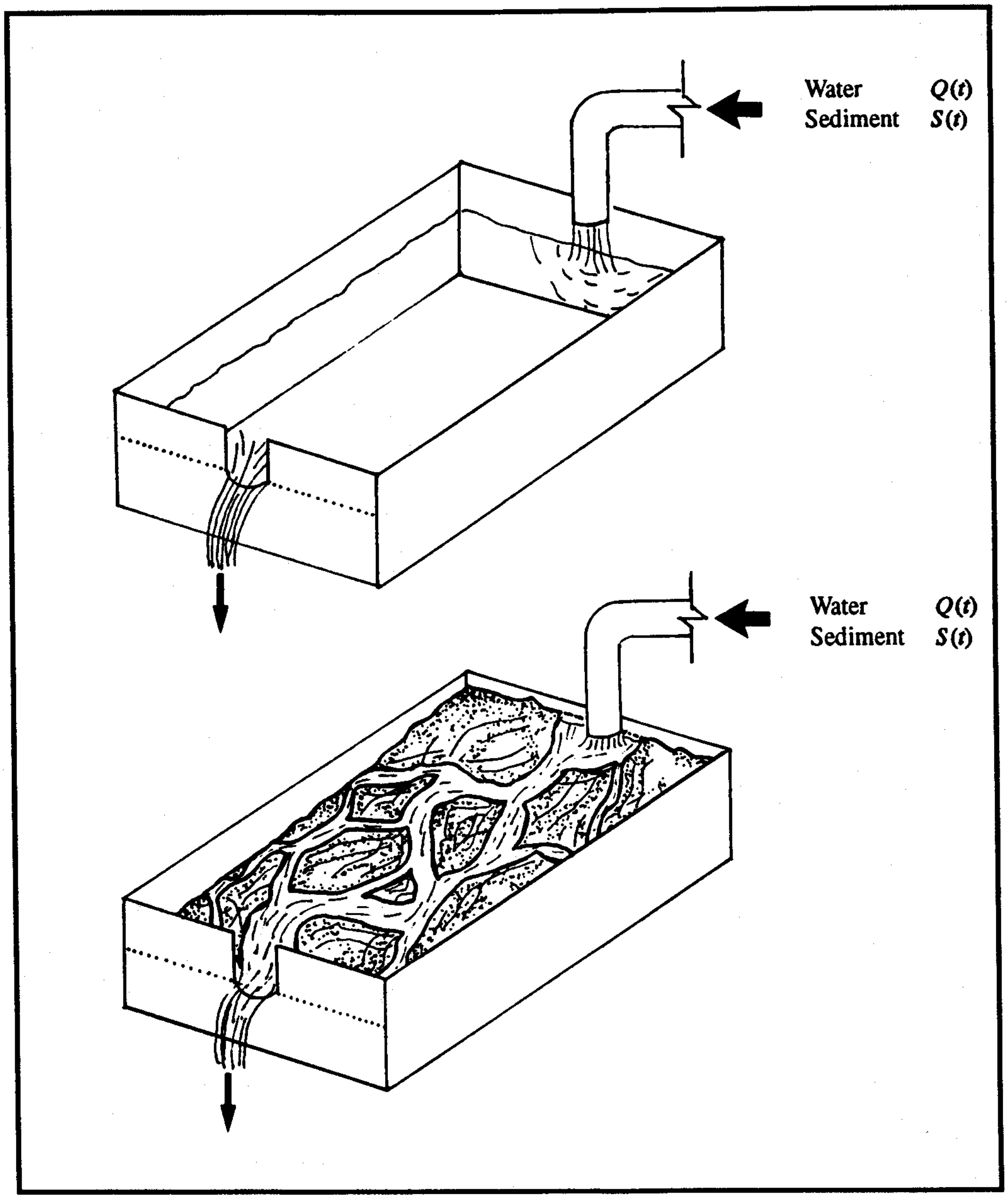


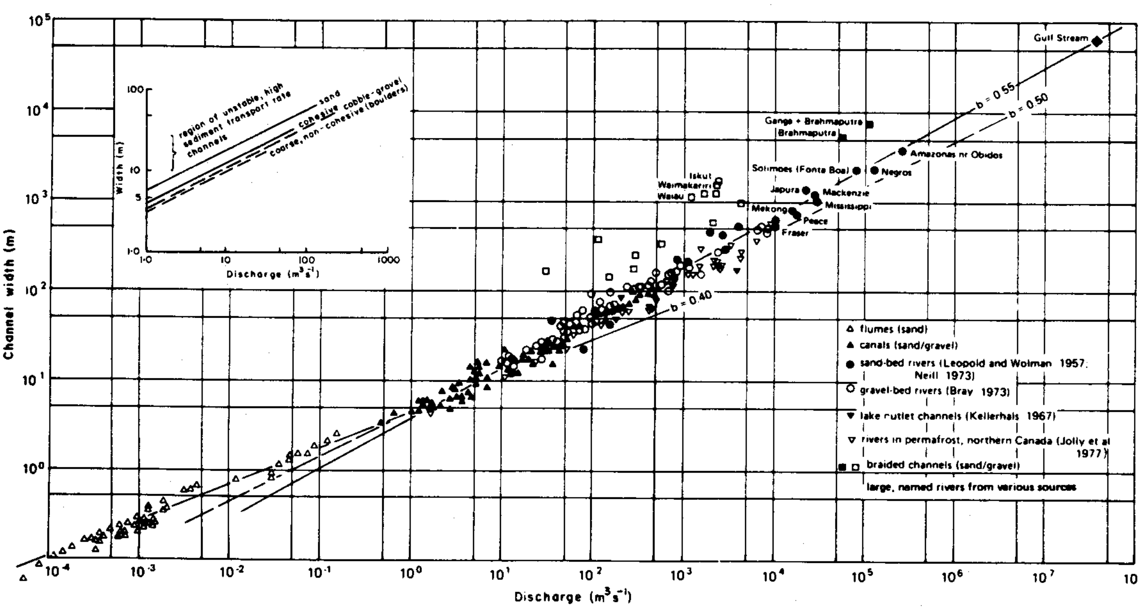
Figure 2: The equilibrium river.

Canada and other northern countries have many infilled lake basins, due to the relatively recent Pleistocene glaciation, and they often contain almost perfect "equilibrium" rivers. However, such rivers are quite rare because of the very long time they take to develop. Other processes, such as plate tectonics, isostatic uplift and climatic changes, interfere and consequently the local geology becomes the third primary factor, besides the hydrological regime and sediment supply from upstream, that determines the local fluvial morphology.

There are also many secondary factors affecting morphology, the most important ones being the local climate, in-so-far as it affects the floodplain vegetation and the ice regime. Actions of man are naturally important and will be discussed later. Scale is another important parameter as it relates to bank vegetation, bed materials and ice. Trees have very different effects, depending on whether a stream is large enough to move them after they topple into the channel. Similarly it matters greatly whether, and how readily, a stream can move the materials over which it flows. Some northern streams are too small to form normal winter ice covers. They freeze to the bed and subsequent flows accumulate in the valley as icings. The result can be a rather peculiar morphology, characterized by unusually deep and narrow channels.

As the above determining factors change along the course of a river, different morphologies result. Often transitions are quite abrupt and are easily identified by features such as tributary confluences, lake inlets or outlets, the apex of a fan at the mouth of a valley, bedrock sills, a canyon entrance, etc. Other morphologic transitions are more gradual and may only become apparent as a study progresses. When investigating the fluvial morphology or morphological processes of a river it is important to address one morphologically homogeneous reach at a time, based on an *a-priori* sub-division that can later be revised or refined.

While the basic factors that determine the morphology of a river reach are well known in a general sense, transforming this into practical, site-specific and quantitative relationships remains largely elusive. The scale relation for channel width as a function of discharge, illustrated in *Figure 3*, remains the best quantitative global relationship



NOTE: Small straight channels have $w \propto Q^{0.4}$ indicating that dynamical similarity is maintained. However, large river channels have $w \propto Q^{0.55}$, approximately, indicating that they become distorted to relatively greater width as they grow larger. Inset: variation in channel width at a given discharge due to material, properties based on *Simons and Albertson (1960)*.

Figure 3: Scale relation for channel width vs. flow *(from Kellerhals and Church, 1989)*.

available, yet it exhibits very wide scatter. More reliable results are achievable if the analysis is restricted to particular geologic and hydrologic settings, such as glacial outwash streams, or streams developed on the bed of a particular, large Pleistocene lake. The attempt by *Mollard (1973)* to correlate channel morphology in plan (essentially the parameters illustrated on *Figure 1)* to empirical factors such as the sediment supply and the hydrological regime remains unique *(Figure 4)*. The geological, hydrological and sedimentological data needed to quantify his qualitative relationships is however presently not available.

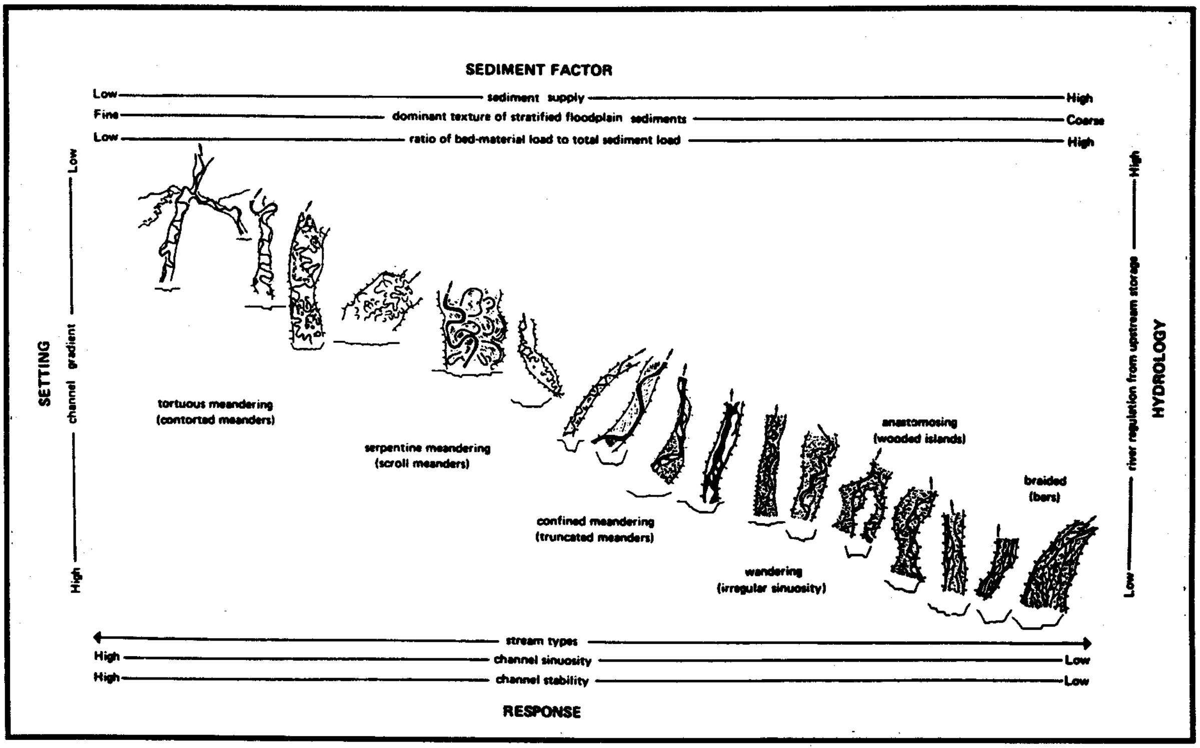


Figure 4: Continuum of stream channel and floodplain types (modified after *Mollard, 1973*).

MORPHOLOGICAL CHANGE

Actions of man can affect the fluvial morphology of a river reach in two different ways:

(i) by direct, local interference with the existing channel morphology and/or the locally active fluvial processes by means such as bank armouring, diking, channel re-alignment, channel confinement at bridge openings, etc.; or

(ii) by altering the external factors that determine the fluvial morphology of a river reach, such as changing the flow regime with flow regulation or interbasin diversions, decreasing the bed load supply through gravel mining upstream of the reach, etc.

This presentation addresses morphological changes of the second type, those brought about indirectly. However, it is important to keep in mind that actions of the first type, local interference with channel morphology, often initiates changes of the second type downstream. Examples include channel straightening and confining that can affect downstream sediment loads and flood peaks, gravel mining which can decrease downstream bed material load and extensive diking which can eliminate floodplain storage and increase flood peaks downstream.

Because most rivers are not truly in equilibrium, man-made morphological changes take place in an environment that is also naturally changing at a wide rate of time scales. Often, these natural changes can be ignored because they act at time scales of thousands to ten thousands of years. Natural changes at more human time scales occur on active deltas and fans, in pro-glacial environments and in some other very fragile situations.

Man-induced morphological changes also take place over a wide range of time scales. On the very long term, all morphological parameters of a reach, including channel slope and plan form depend on the imposed factors, but in practical terms, some parameters change so slowly that they may have to be accepted as constant.

Channel and floodplain slope are particularly difficult parameters to assess. In true equilibrium systems, channel slope is related to discharge by an inverse power relationship with an exponent of around 0.4, and to sediment quantity and sediment size by direct power relationships with exponents which can range to well over four. Since changes in slope usually involve the movement or the deposition of very large quantities of material, they generally occur at time scales of thousands (or more) years. These time scales are beyond those of immediate planning interest. However, the process of slope adjustment involves important morphological changes that can develop over tens to hundreds of years and take on all the practical attributes of an equilibrium state. Increasing channel slope, as a result of decreased (or regulated) flows, increased sediment loads or coarser loads (or some combinations of the above) will manifest itself initially as channel aggradation, possibly accompanied by lateral instability, finer bed materials, different channel bars, different plan form, etc. All of these morphologic changes are highly relevant from the perspective of stream ecology and associated habitat. Decreasing channel slope, as a result of increased flows, decreased loads or finer sediments will lead to channel degradation, incision and more stable channels. An initial transient phase consisting of a wider, more unstable channel has also been observed. This period may persist until a flood occurs which is large enough to mobilize the bed material and initiate downcutting.

The practical complications arising out of these simple concepts are illustrated with two well-documented examples of *Church, 1995*:

(i) The completion of the W.A.C. Bennett Dam on the Peace River in British Columbia in 1968 reduced the morphologically active flood flows downstream of the dam by a factor of two to three. At a geologic time scale, this will result in a smaller channel flowing at a steeper slope. At a shorter time scale, vegetation encroachment and suspended sediment deposition are converting parts of the formerly active channel zone into a new, lower floodplain. As a consequence, many side channels are filling in and disappearing. At an intermediate time scale, the river is now unable to move the sediment loads of its major tributaries and this is gradually leading to a more irregular longitudinal profile with ever longer backwater reaches above the confluences and steepening slopes immediately below the confluences; and

(ii) A large interbasin diversion into the lower Kemano River in British Columbia, a very active, steep, gravel bed river, will result in a reduced stream slope and therefore incision of the river into its natural floodplain below the diversion outlet. This did not become evident until some 20 to 30 years after the diversion took place. The initial effect was a rapid widening of the active channel zone. With the onset of incision, active channel width is decreasing again.

Following an idea of *Schumm (1977)*, *Kellerhals and Church (1989)* compiled a listing of the directions of change for a range of typical man-made interferences based on their experiences and cases reported in the literature. *Table 1* is a modified version of their results. For most entries, neither the rate nor the magnitude of the change is quantifiable on a general basis. Quantitative results are sometimes achievable in specific situations, if an applicable precedent can be found. This could be a natural channel reach that closely mimics the man-induced situation or a project which has similar effects on the parameters which form channel morphology.

Table 1: Qualitative changes in major morphologic parameters for selected actions of man

Action	Determining Factors				Probable direction of resulting downstream changes			
	Q	q_{bm}	q_w		W	S	D_{50}	Lateral Stability
Storage dam, flow regulation	−	−	−	near dam	− *nc*	− *nc*	+	+
	−	−	− +	far downstream	1) −	−	−	+
					2) −	+	+ −	−
Diversion into channel	+	−	−	near start	−/ + −	*nc* / −	+ *nc*	−/ + +
	+	+	−	far downstream	1) +	+	+	−
					2) +	−	*nc* −	−
Upstream deforestation	+	+	+		+	3) − *nc*	−	−
Upstream gravel mining	*nc*	−	+ *nc*		−	− *nc*	+	+
Floodplain clearing	*nc*	+ *nc*	*nc*		+	*nc*	−	−

LEGEND:
- Q — Channel-forming discharge, approximately 2-10 yr flood
- q_{bm} — Relative bed material load, Q_{bm}/Q
- q_w — Relative wash load, Q_w/Q
- W — channel width
- S — channel slope
- *nc* — no change

NOTE: All parameters are associated with discharge Q
If initial changes are thought to be different from long-term changes they are separated by /.
If change can occur in either direction, it is shown as ±
Imposed changes are assumed to be relatively large but not large enough to change the order of magnitude of the affected parameter

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1) upstream of major tributary
2) downstream of major tributary
3) assuming this is a major river channel, not a gully

From an ecological point of view, the morphological changes that should be of particular concern, because they are most widespread and most insidious, are those resulting from the following three primary causes:

<u>Floodplain Clearing</u>

Riparian clearing leads to wider and more unstable channels along alluvial reaches. Within gravel bed rivers, channel width (W) is typically related to bankfull discharge (Q) by the following equation:

(1) $$W = KQ^{0.5}$$

Hey and Thorne (1986) found the constant K varied in value as follows:

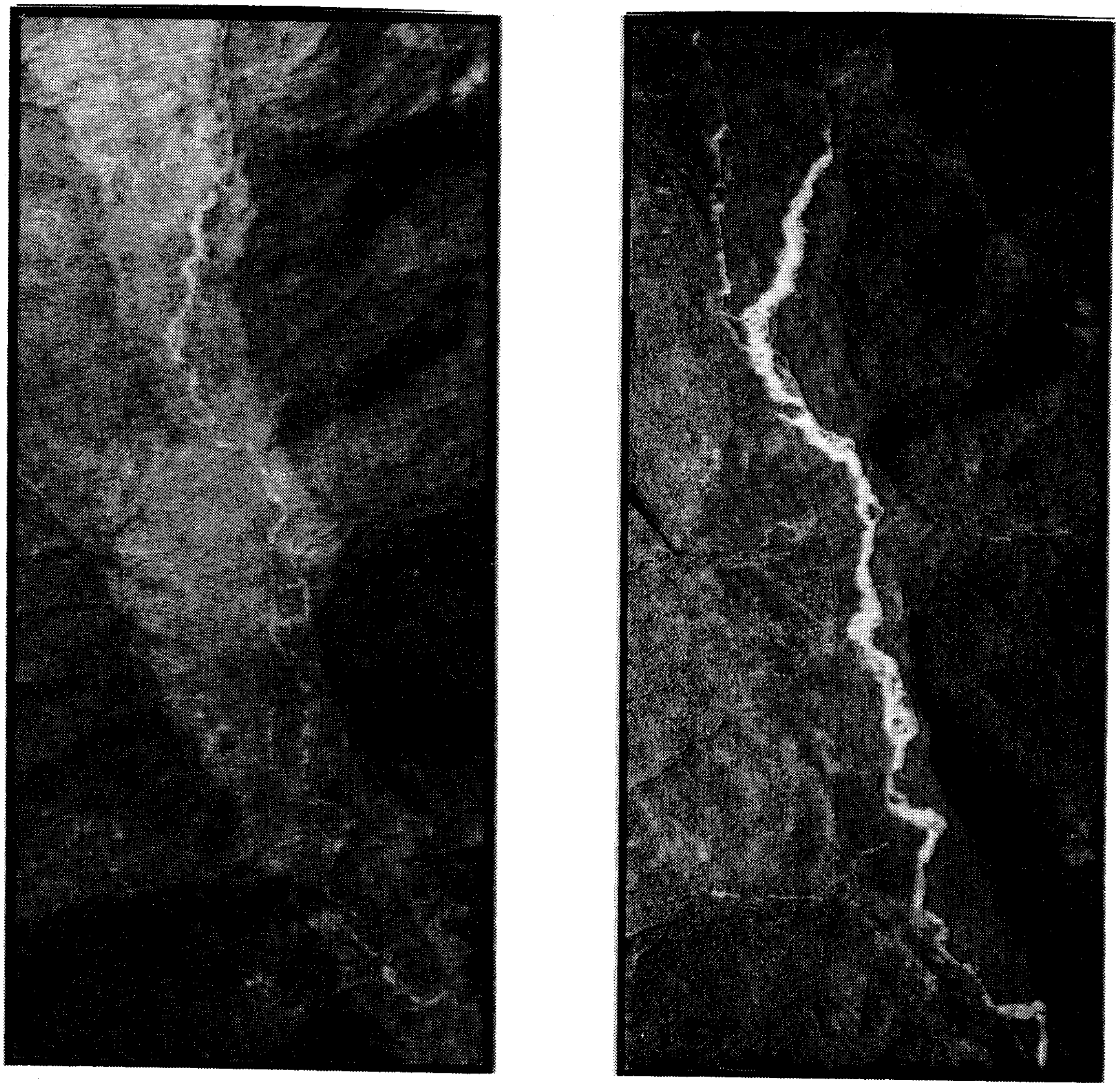
no bank side trees or shrubs	$K = 4.33$
1 - 5% trees and shrubs	$K = 3.33$
5 - 50% trees and shrubs	$K = 2.73$
>50% trees and shrubs	$K = 2.34$

This analysis indicates that channel reaches without bank vegetation are approximately twice as wide as those which have a vegetated riparian cover. The amount of channel widening and the style of resulting instability will of course vary with channel morphology. *Figure 5* illustrates how valley bottom logging has affected the Tahsis River on western Vancouver Island. The earliest available air photograph illustrates conditions immediately following valley bottom clearcut logging in 1956. The river channel is generally single thread or anastomosing, laterally stable and narrow. The second air photo was taken in 1980. The river has adopted a multi-thread channel pattern, extensive sections of river bank are eroding, the unvegetated channel width is substantially wider and there are numerous unvegetated gravel bars, indicating that the rate of sediment transport has been dramatically increased.

Agricultural clearing and grazing activities can also cause significant changes in channel stability. For example, *Figure 6* illustrates recent changes in channel morphology on Deadman River in central BC. The upper photo shows conditions in 1948, prior to the removal of the riparian vegetation. The channel has a tortuous pattern, indicating little coarse textured sediment transport is occurring. The single thread channel has a narrow, deep cross-section and the stable banks have a nearly continuous cover of overhanging vegetation. The lower photo shows conditions in 1992 following extensive riparian clearing and grazing. The channel has adopted an irregularly meandering plan form, has dramatically increased in width and most river bends are actively eroding. Large quantities of sediment are also stored in unvegetated gravel bars. This system is lake-regulated and there are no indications of sediment input from the valley walls. The increase in sediment supply appears to be entirely derived from bank erosion. These changes have resulted in a wider and generally shallower river.

The second author has recently completed an assessment of channel stability on four rivers in central BC which have been subject to floodplain clearing for ranching and farming activities *(M. Miles and Associates Ltd., 1995a & b; 1996a & b)*. The distribution of eroding banks as a function of bank type is summarized on *Table 2*. These data indicate that banks without woody riparian vegetation are much more likely to erode. Banks with only a riparian fringe (wider than approximately half a channel width) or more extensive riparian vegetation can be seen to be substantially more stable. Over the long term, banks with only a riparian fringe frequently destabilize. This occurs when the riparian fringe is outflanked by channel shifting or in cases when riparian trees have died from old age and grazing activities have prevented the establishment of younger trees. It should be noted that, in each of these four rivers, downstream sediment movement from eroding unvegetated banks were destabilizing sections of river with riparian vegetation.

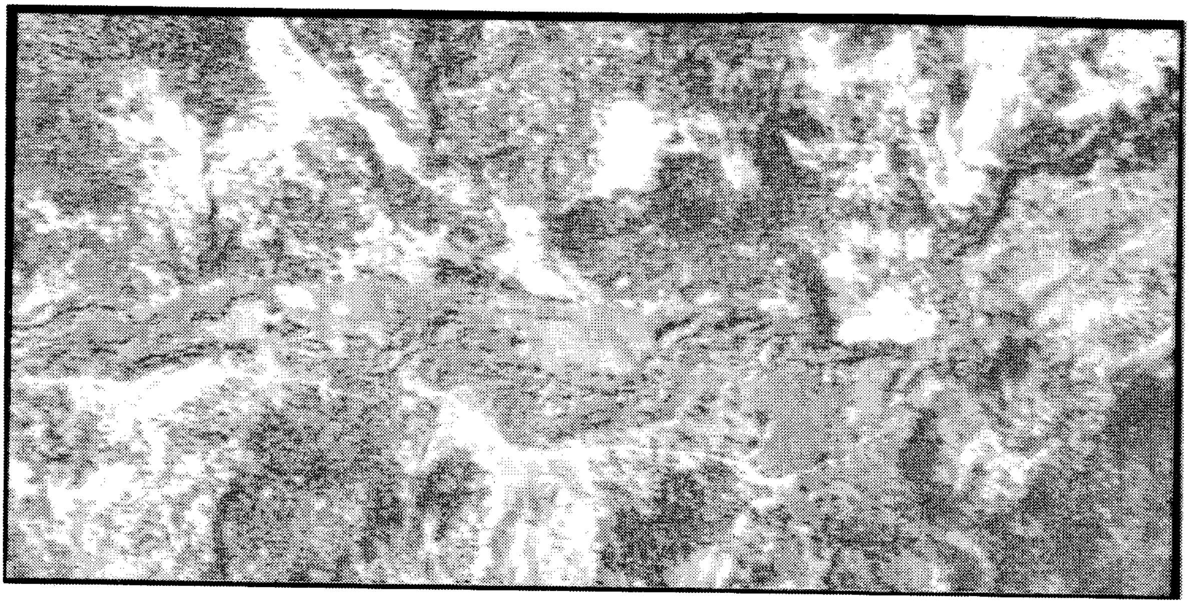
The changes in channel morphology which result from riparian vegetation loss can dramatically affect fisheries habitat. Clearing results in an immediate loss of streamside cover and long term reduction in "cover complexity" in the channel, as woody debris is removed from the system and is not replaced. Bank erosion results in an increased availability of fine-textured materials that elevates rates of suspended sediment and bedload transport and decreases the bed material size. This results in lower quality salmonid egg incubation habitat *(e.g. Scrivener, 1988)*. The wider, shallower channel is also likely to have a larger percentage of flow travelling sub-surface within the river bed during the low flow period. This reduces the total area of habitat available during low flow conditions. Warmer water temperatures (due to both a lack of streamside shade and the changed river cross-section) will reduce summer low flow water viscosities and could result in smaller stream discharges in channels that are losing water to sub-surface materials *(see Constantz et al., 1994 and Constantz & Zellweger, 1995)*.



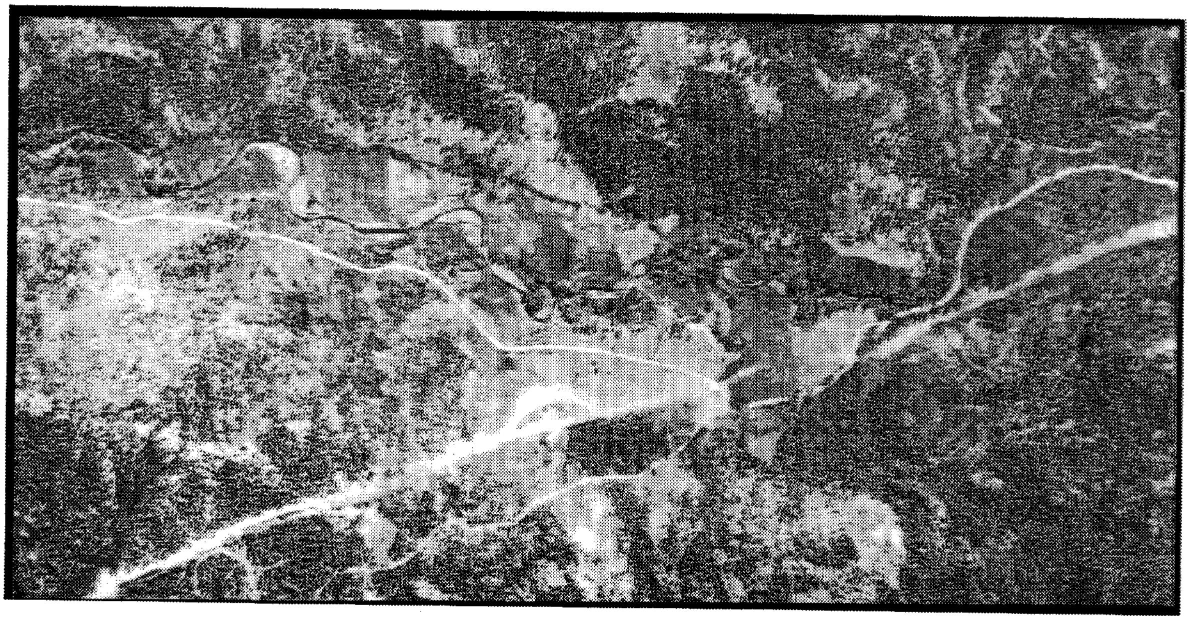
Approximate scale 1:31,000

October 7, 1965 BC 2309 #107 August 10, 1980 BC 80095 #189

Figure 5: Post-logging changes in channel morphology on Tahsis River
(from Kellerhals Engineering Services Ltd., 1993).



September 13, 1948 BC 646 #22 Approximate scale: 1:19,000



June 23, 1992 BCC 92015 #223 Approximate scale: 1:19,000

Figure 6: Changes in channel morphology on Deadman River resulting from agricultural clearing and grazing. ***(modified from Miles, 1995b)***

Table 2: Length of eroding river banks as a function of bank vegetation type

River	Total Bank Length	Length of Actively Eroding Bank with: No Woody Vegetation			Woody Veg. <0.5 Channel Width Wide			Selective Logging			Woody Veg. >0.5 Channel Width Wide			Total Length of Eroding Banks		Total Length of Eroding Banks with Reduced Vegetation Cover	
	km	km	% 1	% 2	km	% 1	% 2	km	% 1	% 2	km	% 1	% 2	km	% 1	km	% 3
Louis	112.9	14.3	13	39	2.3	2	14				12.0	11	20	28.6	25	16.6	58
Deadman	63.6	16.2	25	58	2.2	3	15				1.6	3	8	20.0	31	18.4	92
Salmon	153.8	13.0	8	31	1.1	1	3				5.8	4	7	19.9	13	14.1	71
Hiuihill	56.2	3.0	5	79	2.5	4	45	2.9	5	38	2.3	4	6	10.7	19	8.4	79

1 as a percentage of total river bank length
2 as a percentage of the length of that bank type
3 as a percentage of the total length of all eroding banks

Upstream Land Use Changes

Many comparative land use studies (on the scales of a few hectares to a few square kilometres) have demonstrated that deforestation and urbanization have long term hydrologic effects, such as increased runoff and larger flood peaks. However, these effects did not appear to occur, or were supposedly less pronounced, in the long-term discharge records of rivers with drainage areas of the order of 100 km^2 or larger. A recent study by *Jones and Grant (in press a & b)* of six 60 to 600 km^2 drainages in Oregon that have experienced varying degrees of clear-cutting and road building is challenging this view on the basis of very detailed statistical work. They demonstrate average peak flow increases of 20 to 30% due to 5% differences in cumulative forest area cut. On smaller watersheds (<1 km^2) they report that 25% clearcutting with roads increased the size of all flood peaks by 50% over the first five years. Twenty-five years after logging, flood peaks were still increased by 25%. These results may not necessarily be extrapolated directly to other regions, but they leave no doubt that morphologically significant peak flow increases have occurred in Oregon and have probably occurred in many other places. Since the analysis is based on paired comparisons of drainages with different rates of forest removal, other effects such as global or regional climatic changes do not affect the results.

Global Climate Change

Within the last year or so the evidence for anthropogenic global warming has become close to irrefutable and it is clearly not an isolated effect. Other aspects of climate are changing too, although the exact nature of these changes is the subject of much on-going study and debate. *Karl et al. (1995)*, in a review of the United States climate during the twentieth century, found an "increase in precipitation, especially during the cold season" and "a greater proportion of warm season precipitation derived from heavy convective rainfall compared with gentler, longer-lasting rainfall", among several other trends. *Knox (1993)* shows how relatively modest changes in climate can lead to large effects on flood magnitude. A recent report compiled by a Canadian government board *(Canadian Climate Program Board, 1995)* lists seven established changes indicating that human activities affect climate, one of which is: "a trend towards higher frequency of extreme rainfall in recent decades was evident in Japan, the USA, former Soviet Union and China." Looking into the future, the report states: "Many models suggest an increase in the probability of intense precipitation with increased greenhouse gases."

Because the runoff process is highly non-linear, it almost always amplifies variability in precipitation, sometimes by an order of magnitude. This, combined with the well-established fact that channel morphology depends far more on the regime of floods than on mean flows or low flows, suggests a potential for global changes in river morphology. With few of the world's larger rivers not affected by land use changes, it may be difficult to separate climate effects from land use effects. The important point is that, in so far as morphology is concerned, they often act in the same direction.

Not all recent changes and trends in climatic or hydrologic data can unequivocally be attributed to actions of man. *Church (in press)* has undertaken a very detailed study of the longest climatic and hydrologic records of Cordilleran Canada and finds a distinct and widespread shift towards a wetter climate starting around 1950. Mean annual precipitation increased by 10 to 30 per cent, but mean annual floods increased by up to 50 per cent, both in rivers affected by land use changes and in essentially pristine drainages. The morphologic changes resulting from these imposed hydrologic changes illustrate the widely differing abilities or time scales with which rivers respond. The channel width of the Bella Coola River, a relatively unstable gravel-cobble bed river, is seen to respond to the prevailing flood regime within a very few years, more or less as indicated by the width-discharge relationship of *Figure 3*. In contrast, the Skeena River, a more stable gravel-cobble river, decreased in width even though the post-1947 mean annual floods were 50% larger than for the 15 years before 1947.

HABITAT CHANGES

The relationships between channel morphology and fish habitat are exceedingly complex because they generally involve various life stages of many species. The important habitats are often very minor components of the dominant morphology, such as small backchannels on a floodplain, overhanging bank vegetation, irregularities in the bank alignment, minor bed forms, etc. The composition and vegetation of the channel banks may be far more critical than the exact channel width. Some of the most important habitats are often relatively short-lived by-products of a morphological process. For example, channel shifting or meander migration on a floodplain produce transient features such as oxbow lakes, backchannels and wetlands on the floodplain, undercut banks, overhanging vegetation and toppled trees on the outside of bends. Overhanging bank vegetation can have major effects on water temperature and on the availability of food.

The wider, shallower channels which result from riparian vegetation removal commonly lack deep water refuge habitat. Increased water temperatures can have direct effects on the stream biota. For example, juvenile salmonids are less tolerant of increased summer water temperature than are warm water species. Bed material size distribution is an important parameter that determines channel morphology and it may be an even more important aspect of fish habitat. For example, small changes in the percentage of fines in a bed material gravel may have no detectable effects on major morphologic parameters, but can adversely affect salmonid egg to fry survival *(eg. Hartman and Scrivener, 1990)*. Very small changes to the hydrological or sediment regime, that may or may not result in large scale morphologic changes, can therefore have significant effects on aquatic habitat.

MODELLING CHANGES

Most engineering projects affecting rivers are now subject to some kind of environmental review. As a result, there is a large demand for models capable of predicting morphologic changes and their effects on habitat. The recognition of land use changes and climate changes as further driving forces towards morphologic changes, and the desire to try and restore degraded habitats, all add to the need for predictive models. Unfortunately the state-of-the-art is primitive and there is no indication that this will change in the near future. *Figure 3* illustrates what can be said about channel width, with the time scale for width changes being largely unknown. The corresponding functions for channel depth and slope are somewhat similar, but not nearly as well defined. *Figure 4*, although published some 23 years ago, is still essentially the state-of-the-art for the relationships between channel morphology in plan and the imposed driving factors of discharge, sediment supply and valley slope.

The main reason for this unsatisfactory situation is our poor understanding of sediment transport processes in a three-dimensional environment. For decades the engineering community has been absorbed with two-dimensional sediment transport work in laboratory flumes and in some artificially trained and straightened European rivers. When the results of this work are misapplied to the more normal 3-dimensional river situations (where channel width and alignment are more or less free to adjust) they often lead to wrong answers. In some cases, even the predicted direction of change is wrong *(e.g. Kellerhals and Miles, 1990)*.

The matter is complicated further by the nature of the relationship between sediment transport and discharge at any given river site. The general form of the relationship is illustrated on *Figure 7*. It is characterised by a threshold discharge for the initiation of transport and an initial, very rapid rise of the transport rate *(see Church and Walcott, 1991)*. This is where most natural gravel-bed channels operate, yet many of the sediment transport formulas apply to the upper, more gradual part of the function, because that part is easier to determine with laboratory experiments. The steep lower part of the range suggests that small changes to the hydrologic regime can have significant morphologic effects.

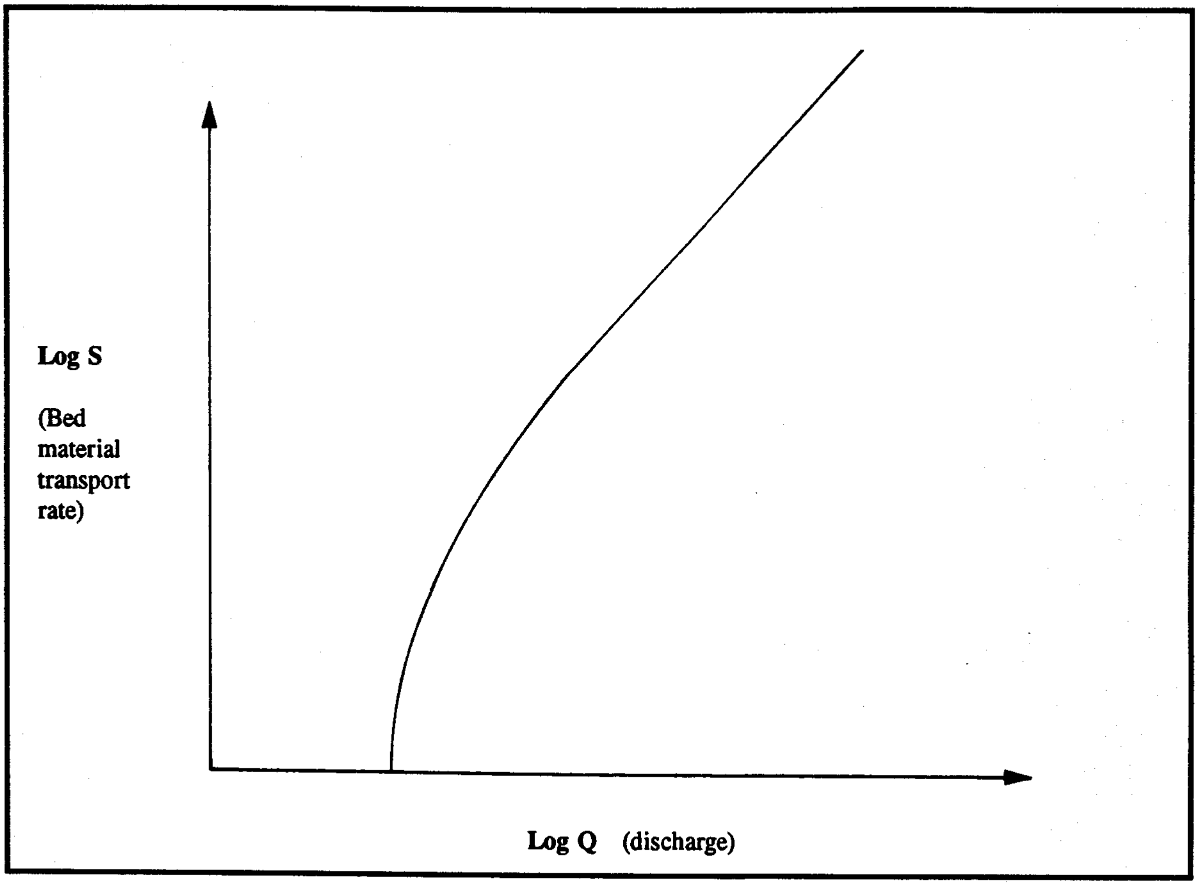


Figure 7: Bed material transport rate as a function of channel discharge.

The demand for predictive models has been particularly strong in fisheries circles because of the pressures that project approval processes place on fisheries managers. These pressures have been so great that they seem to have overcome the only possible rational conclusion, which is that predictive habitat models for situations with a possibility for morphological change are beyond the present state-of-the-art. Instead of admitting that morphologic change cannot be quantified, it is simply being ignored. For example, the commonly used Instream Flow Incremental

Methodology *(Bovee, 1982)* assumes that river channels have fixed dimensions under conditions of altered discharge and sediment transport regimes. An analysis of open channel hydraulics is being substituted for an understanding of channel morphologic response, to the long term detriment of the resource. The biological part of these models is also subject to considerable uncertainty as, for example, the assumptions between predicted "weighted usable area" and biomass of fish (used in the IFIM model) have not been verified (see *Mathur et al., 1985; Shirvell, 1986*).

RESTORATION AND MITIGATION

Koski (1992) indicates that 5.2 x 10^6 km of US streams need restoration. Similar statistics are not available for Canada, but the amount of riparian disturbance in southern Canada is huge. For example, the four studies in central BC discussed previously found that 49% of the 390 km of mapped channel banks had either no woody riparian vegetation or only a narrow fringe. Fisheries and forestry regulations in BC *(eg BC Ministry of Forests and Environment, 1995 and The Scientific Panel for Sustainable Forest Practices in Clayoquot Sound, 1995)* now require a riparian leave strip on Crown Land. Recent watershed restoration programs in BC *(Watershed Restoration Program, 1994)* will attempt to replant previously-damaged riparian areas. There are however no comparable regulations or program for private lands. Similar problems occur throughout other areas of Canada and a massive education program is required to change current riparian land use practices.

River restoration or enhancement projects have principally been undertaken for biological reasons and have focused on building replacement fisheries habitat. These structural repairs typically include increasing the channel roughness by rock placement, providing local areas of deeper water by constricting the channel or by constructing weirs which will form plunge pools and providing cover by placing logs or other structures along or within the channel.

A recent study in BC *(Hartman & Miles, 1995)* evaluated the success of 99 projects constructed by the BC Ministry of Environment, Lands and Parks. Only 55% of the projects were found to be physically successful. Biological success was estimated as 50%, despite there being a general lack of any detailed post-construction biological evaluation. More disturbingly, the average post-construction assessment period was only two years, which indicates that the longer term success statistics will likely be much lower.

A recent review *(Miles, in press)* indicates that the BC experience is not unusual. Habitat restoration projects in Alaska have a 47% success rate *(Parry et al., 1993; Parry and Seaman, 1994)*. A review of projects in Oregon and Washington found a 40% success rate *(Frissell & Nawa, 1992)*. Results of a review of projects in SW Alberta *(RL&L et al., 1994; Fitch et al., 1994)* reported a 64% success rate. However field studies following a sizeable flood in June, 1995 indicate that approximately 77% of structures have now been destroyed or severely damaged *(Fitch et al., 1996)*.

A large number of instream habitat improvement projects have been undertaken in the Willamette, Mt. Hood and Snoqualmie National Forests in the Pacific North West. Studies by *(Uthank, 1994; Higgins and Forsgren, 1987 and Doyle, 1991)* have reported over 80% success rates. These unusually high survival rates appear to result from:

(i) use of log structures which were designed "to function with the natural tendencies of the stream flow and hydrologic functions" *(Uthank, 1994)*;

(ii) design improvements resulting from extensive experience in a geographically limited area; and

(iii) an annual inspection and maintenance program.

The above statistics indicate that structural measures which attempt to physically recreate lost stream habitat frequently don't perform well. Analyses in Alberta indicate that success rates are likely to be poorer in higher energy environments or on streams that are either laterally unstable or are carrying high sediment loads *(RL&L*

et al., 1994; Fitch et al., 1994). Unfortunately, the land use changes described in the section on Habitat Changes increase flood flows, accelerate channel instability and elevate rates of sediment transport, all of which decrease the likelihood of successful structural performance.

The above discussion is not intended to indicate that structural measures to enhance or restore fisheries habitat should not be undertaken. These measures are however frequently ephemeral and are therefore best suited as temporary measures to enhance fisheries productivity over the short term. The longer term solution is to re-establish or maintain the fluvial processes which are responsible for habitat formation.

Biological studies, such as those by *Hunt (1976)* or *Ward and Slaney (1981; 1993)* indicate that in some circumstances instream structural enhancement can increase fisheries production. However, there is increasing evidence that structural measures alone do not necessarily produce an incremental increase in fish numbers. For example, on-going studies on the Oldman River Dam project in Alberta *(O'Neil & Pattenden, 1994)* has not demonstrated that habitat enhancement structures have significantly increased trout production. Similar results have been observed in an intensive study on Fish Creek in Oregon *(Reeves et al., 1990)*. These and other studies suggest that habitat availability is not necessarily the only factor which limits biological production. There are unfortunately few long term studies that evaluate natural biological productivity *(Lawson, 1993)* or the response to enhancement projects over the longer term *(see Beschta, 1994; Hartman et al, in press)*. Consequently many projects are being undertaken without an understanding of either the biological limiting factors or a sound basis for predicting the results of proposed habitat manipulations.

CONCLUSIONS

Man is changing river morphology and associated aquatic habitats through direct (or local) interference and by altering the formative external factors, such as streamflow, sediment loading, bank characteristics, etc. These changes can result from global climatic changes, from mega-projects, such as a river regulation or hydroelectric project, or by the impact of numerous small land use changes. The cumulative impact of the small projects can equal or exceed that of a mega-project.

Over the past thousands of years, salmonids have evolved continuously and adapted to the available fluvial, geomorphological and habitat conditions. However, these mechanisms of change operate slowly and cannot keep up with the rate of man-made changes. Most, if not all, man-made morphologic changes are therefore likely to be detrimental to the natural fish stocks.

Evidence continues to accumulate indicating that the majority of river restoration projects are unsuccessful. The failure rate is particularly high for projects that rely on some form of habitat construction or artificially-maintained channel morphology. Better design practice may improve performance, however experience indicates that it is very difficult to restore a river once it has been severely altered. For example in Europe, river restoration projects in Denmark *(Madsen, 1995)* and the Netherlands *(Klaassen et al. in press)* involve removing all training works, providing a riparian corridor and assisting the river to re-establish a "regime", laterally unconstrained, river pattern.

Re-naturalizing a river by establishing riparian corridors, within which fluvial processes are allowed to re-establish natural morphologies, is frequently the best restoration strategy, but does not provide quick results. In addition, it does not address the fundamental problem if the formative external factors that determine fluvial morphology are altered significantly through climatic changes, cumulative land use impacts, or mega-projects. In these circumstances there will be relatively-unpredictable downstream morphologic changes, even along perfectly "natural" river reaches, and the effects on natural fish stocks will almost certainly be negative. However, providing a natural stream corridor is still beneficial.

Efforts to quantitatively model habitat alterations, resulting from changing the formative external factors, have no scientific basis and the mitigation and compensation arrangements negotiated on the basis of such modelling results are, consequently, rarely successful. As hydro-technical specialists it is our duty to tell the decision makers in the political arena what general types of morphological changes are brought about by proposed land use changes or mega-projects and just how uncertain the details of the changes and the effects on fish really are. In most instances the only reasonable course of action is to make decisions on the basis of conservative worst-case scenarios and hope for pleasant surprises.

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TRANSLATING CHANNEL MORPHOLOGY INTO HYDRAULIC HABITAT: APPLICATION OF THE HYDRAULIC BIOTOPE CONCEPT TO AN ASSESSMENT OF DISCHARGE RELATED HABITAT CHANGES.

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ABSTRACT

Modification of the flow regime, for example through impoundment or interbasin transfer schemes, impacts on the available habitat in the downstream channel. Assessments of Instream Flow Requirements (IFRs) are based on our ability to predict the available habitat in relation to discharge. The available habitat at any point along the channel is dependent on the channel morphology which in turn determines flow width, depth, velocity and related hydraulic variables such as Froude number, shear velocity and roughness Reynolds number. Variations in width, depth and velocity with discharge at any given channel section can be described by the at-a-station hydraulic geometry. Separate relationships for different morphological units such as pools or riffles, or for different channel types such as straight, meandering or braided can be used to predict changes in available habitat at a cross section. A major limitation of this method is that these relationships are based on average figures whereas in fact there is a considerable variation in the hydraulic characteristics of the flow within any one morphological unit. Moreover, width and depth are described independently whereas it is the combination of the various flow characteristics at a point which determines habitat suitability for a given species.

An alternative method for characterising habitat is to classify point flows into hydraulic biotopes. Research in a variety of South African rivers has demonstrated that the surface flow conditions, or flow type, together with substrate class, can be used as a simple technique to classify flows into spatially distinct instream flow environments with characteristic hydraulic attributes described in terms of Froude number, shear velocity and roughness Reynolds number. Surface flow characteristics can therefore be used to distinguish habitat classes which are relatively distinctive in terms of both near-bed and mean flow conditions. Eight hydraulic biotopes have been identified for South African rivers: backwater pools, pools, runs, glides, riffles, rapids, chutes, cascades. Classifying flows across transects according to hydraulic biotopes allows an assessment of habitat diversity at any discharge.

Research in the Buffalo River, Eastern Cape, South Africa, has demonstrated the variation in hydraulic biotope distribution and associated flow characteristics for different morphological units at a range of channel scales and a range of discharges. The sampling framework included the following morphological units: step, plunge pool, alluvial pool, bedrock pool, plane bed, riffle, rapid and bedrock pavement. The results have been used to develop a hydraulic biotope diversity index which has potential application to the assessment of flow related changes in habitat diversity.

KEY-WORDS: Geomorphology / hydraulic habitat / hydraulic biotope / instream flow requirements / South Africa

INTRODUCTION

Concern about the impact of dam construction and flow regulation on the ecology of regulated streams has prompted efforts by ecologists to quantify and preserve the flow patterns required for the survival of aquatic species (Gore & King, 1989; King & Tharme, 1994). These environmental flows are referred to as the Instream Flow Requirements. Standard methods exist to predict these flows such as the Instream Flow Incremental Methodology (IFIM) (Bovee, 1982) and the Physical Habitat Simulation Model (PHABSIM) (Milhouse *et al.*, 1989). These systems have been tested in South Africa (King & Tharme, 1994) with limited success, largely because of the complexity involved, the data and manpower intensity and the resultant expense. The aim of many instream flow studies is to assess the flow requirements of target species and to develop a recommendation for flows needed to assure maintenance of the population. In South Africa, significant gaps in the ecological understanding of river ecosystem process and the absence of target species has led to a focus on habitat maintenance rather than species management (O'Keeffe, 1996).

Available aquatic habitat can be described in terms of the wetted perimeter (the total availability of habitat), the flow depth, flow velocity and related hydraulic variables such as Froude number, shear velocity and roughness Reynolds number; together these determine the environment both on the bed and within the water column which is experienced by aquatic organisms. As flow conditions are directly dependent on discharge it follows that aquatic habitat will also vary with discharge. The relationship between discharge, flow characteristics and habitat availability provides the basis of most IFR assessments.

Variations in width, depth and velocity with discharge at any given channel section can be described by the at-a-station hydraulic geometry. Separate relationships for different morphological units such as pools or riffles, or for different channel types such as straight, meandering or braided can be used to predict changes in available habitat at a cross section (Kellerhals and Church, 1989; Hogan and Church, 1989). A major limitation of this method is that these relationships are based on average figures whereas in fact there is a considerable variation in the hydraulic characteristics of the flow across any one channel section. (Hogan and Church, 1989). Moreover, width and depth are described independently whereas it is the combination of the various flow characteristics at a point which determines habitat suitability for a given species.

An alternative method for characterising habitat is to classify point flows into hydraulic biotopes (Wadeson 1996). Research in a variety of South African rivers has demonstrated that the surface flow conditions, or flow type, together with substrate class, can be used as a simple technique to classify flows in to spatially distinct instream flow environments with characteristic hydraulic attributes described in terms of Froude number, shear velocity and roughness Reynolds number (Wadeson 1966). Thus surface flow characteristics can be used to distinguish habitat classes which are relatively distinctive in terms of both near-bed and mean flow conditions. Eight hydraulic biotopes have been identified for South African rivers: backwater pools, pools, runs, glides, riffles, rapids, chutes and cascades (Table 1). Classifying flows across transects according to hydraulic biotopes allows the diversity of habitat at a given discharge to be assessed. Although research has not yet been carried out to validate the ecological significance of hydraulic biotope classes, collective ecological experience points to their likely acceptance.

It is necessary at this point to make a distinction between discharge invariant morphological units and discharge dependent hydraulic biotopes so as to clear up possible confusion in semantics (Wadeson, 1994). The morphological units are the basic structures or building blocks recognised by fluvial geomorphologists as comprising the channel morphology and may be either erosional or depositional features. Although in the long term their characteristics are dependent on the imposed flow regime which determines erosion and sediment transport processes, in the short term they can be considered to be constant features. Morphological units can be subdivided into two groups: pools and

Table 1: Definitions of hydraulic biotopes recognised in the Buffalo River (after Rowntree, in press)

Hydraulic biotope	General description	Flow type[1]
Backwater	The definition as used in this study includes true backwaters: a morphologically defined area along-side but physically separated from the channel, connected to it at its downstream end or as slack water or dead zone: an area of no perceptible flow which is hydraulically detached from the main flow but is within the main channel Backwaters may occur over any substrate.	Barely perceptible flow
Pool	Has direct hydraulic contact with upstream and downstream water.	Barely perceptible flow
Glide	Occur over any substrate as long as the depth is sufficient to minimise relative roughness. Glides exhibit uniform flow with no significant convergence or divergence.	Smooth boundary turbulent flow: clearly perceptible flow without any surface disturbance.
Chute	Typically occur in boulder or bedrock channels where flow is being funnelled between macro bed elements. Chutes are generally short and exhibit both upstream convergence and downstream divergence.	Smooth boundary turbulent flow at higher flow velocities than glides
Run	Occur over any substrate apart from silt; relative roughness low. They often occur in the transition zone between riffles and the downstream pool;.	Rippled flow
Riffle	Occur over coarse alluvial substrates from gravel to cobble; relative bed roughness high.	Undular standing waves or breaking standing waves
Rapid	Rapids occur over a fixed substrate such as boulder or bedrock.	Undular standing waves or breaking standing waves
Cascade	Occurs over a substrate of boulder or bedrock. Small cascades may occur in cobble where the bed has a stepped structure due to cobble accumulations.	Free-falling flow

[1]Definition of flow types used in Table 1

No flow.	no water movement
Barely perceptible flow	smooth surface, flow only perceptible through the movement of floating objects.
Smooth boundary turbulent	the water surface remains smooth; streaming flow takes place throughout the water profile; turbulence can be seen as the upward movement of fine suspended particles
Rippled surface	the water surface has regular disturbances which form low transverse ripples across the direction of flow; the degree of disturbance may vary from faint ripples to strong ripples.
Undular standing waves	standing waves form at the surface but there is no broken water
Broken standing waves	standing waves present which break at the crest (white water)
Free falling	water falls vertically without obstruction

hydraulic controls. Pools are scour or erosional features with relatively high depths relative to width and within which the macro-scale flow hydraulics are controlled by a downstream hydraulic control. The hydraulic controls are usually aggradational or erosionally resistant features with relatively low depth relative to width and within which the macro-scale flow hydraulics are not controlled by downstream hydraulic features. Shallow flows and large bed material calibre leads to micro-scale hydraulic controls at low flows so that these features tend to be hydraulically complex. A description of the morphological units considered in this paper are given in Table 2.

Table 2: Morphological units within the Buffalo River adapted from Wadeson (1996)

Morphological Unit	Description
Plunge Pool	Erosional feature below a resistant strata (waterfall).
Bedrock Pool	Topographic low point formed behind resistant strata lying across the channel.
Alluvial Pool	Topographic low point formed by scour within an alluvial bed, often associated with fines in the bed, especially at low flows.
Step	Occur within headwater streams, characterised by large clasts organised into discrete channel spanning accumulations.
Plane Bed	Consist of large clasts relative to flow depth and lack well defined bedforms
Riffle	Topographic high points in an undulating bed long profile composed of coarser sediments.

Hydraulic biotopes can often be related directly to morphological features and are therefore commonly given the same terminology, but being flow units rather than sedimentological units, they vary with discharge. Thus riffle hydraulic biotopes are associated with riffle morphological units, but, as will be demonstrated in this paper, a riffle morphological unit contains an assemblage of hydraulic biotopes which changes as flow discharge changes.

As noted above, the at-a-station hydraulic geometry varies with different channel morphologies. Discharge related variations in hydraulic biotopes should also be dependent on channel morphology. It is postulated that if a relationship can be found between channel morphology and discharge related changes in the distribution of hydraulic biotopes, this would lay the basis of more cost effective methods for assessing IFRs. This relationship is explored in this paper using a case study from the Buffalo River in the Eastern Cape Province of South Africa.

THE RESEARCH AREA AND STUDY SITES

The Buffalo River is a relatively short, steep coastal river system, fairly typical of those draining the eastern escarpment of South Africa (Figure 1). From its headwaters in the Amatola Mountain range at an altitude of 1300 metres (amsl) it flows in a south-easterly direction for 125 km before discharging into the Indian Ocean at the river port of East London. The location of research sites used to develop the hydraulic biotope classification are shown in Figure 1. Results will be presented in this paper for the three sites in the upper reaches. Here a relatively steep mountain or foothills type boulder or cobble bed stream flows through indigenous yellow wood forest. It is unimpacted by impoundments. Flows are perennial. The mean annual precipitation over these upper catchment areas is between 1500 mm to 2000 mm, against a mean annual potential evapotranspiration of 1360 mm.

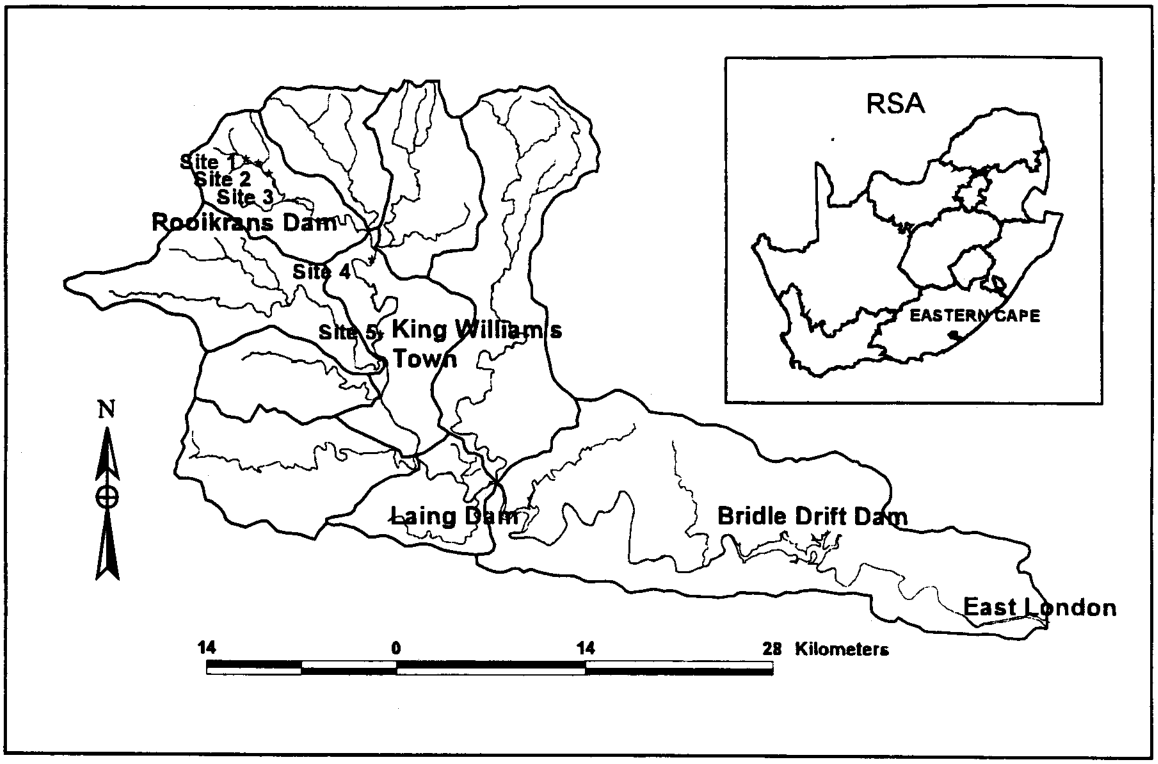

Figure 1: Location of study sites within the Buffalo River

The three study sites represented a range of channel types and channel morphologies. Site 1 had a step-pool morphology, characterised by large clasts (boulders and cobbles) organised into discrete channel spanning accumulations that form a series of steps separating pools containing finer material in the size range of sand, gravel and small cobble. Channel gradient within this reach is relatively steep (0.17). Morphological units associated with this site included plunge pools and steps.

Site 2 was described as having a plane bed (Montgomery and Buffington, 1993), lacking well defined bedforms and characterised by long stretches of relatively planar channel bed that is punctuated by occasional channel spanning bedrock rapids. Water flows around cobble particles that are large relative to the flow depth. According to Montgomery and Buffington (1993), these reaches may occur at gradients and relative roughness intermediate between pool-riffle and step-pool reaches, this is clearly so in the upper reaches of the Buffalo River (gradient 0.11). Specific bed morphology found within this reach included plane beds and bedrock pools. Thalweg substratum was dominated by large cobbles and boulders which were interspersed with finer material in the lee areas. Particle shape was disk like and the larger material tended to be relatively well packed giving rise to a stable bed.

Site 3, with a gradient of 0.009, had a pool-riffle morphology. Riffles were composed of cobble and boulder interspersed with gravel. Pools had beds dominated by similar size material which has been covered by a thin layer of fine material (silt and mud). Substratum shape was disk like and well packed to create a stable bed.

METHODS

Data collection

The aim of this study was to assess the discharge related changes in hydraulic habitat characteristics and diversity for a range of morphological units in the Buffalo River. The three sites described above provided a range of morphological units: step and two plunge pools at site 1, plane bed and two bedrock pool sat site 2 and two riffles and two alluvial pools at site 3. At each site, survey transects were set out across the separate morphological units, the number of transects being determined by the potential hydraulic variability of the site. A total of 25 transects were laid out and surveyed; 9 at site 1, 8 each at site 2 and site 3. Along each transect approximately 12 sampling points were selected. Measurements of flow characteristics and identification of the hydraulic biotope class using the concepts and ideas which are formalised in Table 1 was carried out at four different discharges as indicated in Table 3 . Flow exceedence values were calculated from a daily flow duration curve based on 48 years of record from a gauging site immediately downstream of site 3. Care was taken to ensure that the same sampling points were selected at each discharge. Flow characteristics included velocity (0.6 depth from surface), depth, bed roughness profile and water temperature; these measurements were used to develop and test the hydraulic biotope classification as discussed fully elsewhere (Wadeson, 1996).

Table 3: Flow discharges measured at the research sites using the velocity-area method.

Flow exceedence (%)	Discharge ($m^3 sec^{-1}$)		
	Site 1	Site 2	Site 3
92	.015	.015	.015
73	.037	.04	.04
50	.045	.075	.084
3	.93	.97	1.87

Data analysis

For each site, pie-charts were prepared showing the percentage distribution of hydraulic biotopes at each discharge (Figure 2 and 3). The pie-charts give a good visual indication of both the diversity of hydraulic biotopes and the dominant hydraulic biotope at each discharge. Habitat diversity was further described by means of Hydraulic Biotope Diversity Curves from which a Hydraulic Biotope Diversity Index was derived. The Hydraulic Biotope Diversity Curves and Index were based on the principle of Lorenz curves as described by Hammond and McCullagh (1978).

For each morphological unit-discharge combination the hydraulic biotopes were ranked by frequency from high to low and the percent frequencies cumulated. The resulting curves (Figure 4) were compared to a regional curve derived from all sampling points at all measured discharges for the original five sites in the Buffalo River. This curve gives an indication of the total habitat diversity for the river and provides a reference point for comparison. When interpreting the curves, the more convex the curve the lower the diversity, the straighter the curve the higher the diversity.

The cumulative frequency data was used to derive a Hydraulic Biotope Diversity Index (HBDI) using a modification of a formula given by Hammond and McCullagh (1978).

(1) $$HBDI = 1 - (A-R)/(M-R)$$

where A is the cumulative percentage total for the specified data set, R is the cumulative percentage total for the river as a whole and M is the maximum cumulative percentage total assuming 100 percent of the frequencies are in Rank 1. In this case, with eight possible hydraulic biotopes, M is 800. R was calculated as 647. The index varies from 0 for low diversity, with all hydraulic biotopes falling into one class, to 1 for a high diversity in which the distribution resembles that for the river as a whole.

RESULTS

Hydraulic biotopes changes for the different morphological units are shown in Figures 2, 3, and 4. Changes in the HBDI and the dominant hydraulic biotope are summarised in Table 4. For the purpose of discussion, morphological units will be subdivided into two groups: pools and hydraulic controls.

Table 4: Hydraulic biotope diversity indices for different morphological units in the Buffalo River

Discharge (% exceedence)	**92**	**73**	**50**	**3**
Alluvial pool :HBDI	0.1	0.33	0.31	0.25
Dominant biotope	backwater pool	backwater pool	pool	run
Plunge pool: HBDI	0.16	0.26	0.35	0.27
Dominant biotope	backwater pool	pool	pool	run
Bedrock pool: HBDI	0.10	0.11	0.10	0.10
Dominant biotope	pool	pool	pool	run
Step: HBDI	0.58	0.84	0.77	0.89
Dominant biotope	run	run	run	run
Plane bed: HBDI	0.67	0.53	0.56	0.35
Dominant biotope	pool	run	run	run
Riffle: HBDI	0.41	0.61	0.65	0.57
Dominant biotope	pool	pool	riffle	run

All pools showed a similar response, with backwater pool and pool dominating at the three lowest discharges and a significant increase in run biotopes at the highest discharge. The inclusion of some riffle flow at high discharges may have been due to the lateral extension of the water into shallow margins with high relative roughness. Hydraulic biotope diversity was low for all pools at all discharges, with hydraulic biotopes concentrated in one or two classes. The bedrock pool at the causeway site is particularly consistent. Maximum values of the HBDI occurred at discharges with flow exceedences between 73 percent and 50 percent. Some diversity was lost at spate discharges as runs came to dominate the pools.

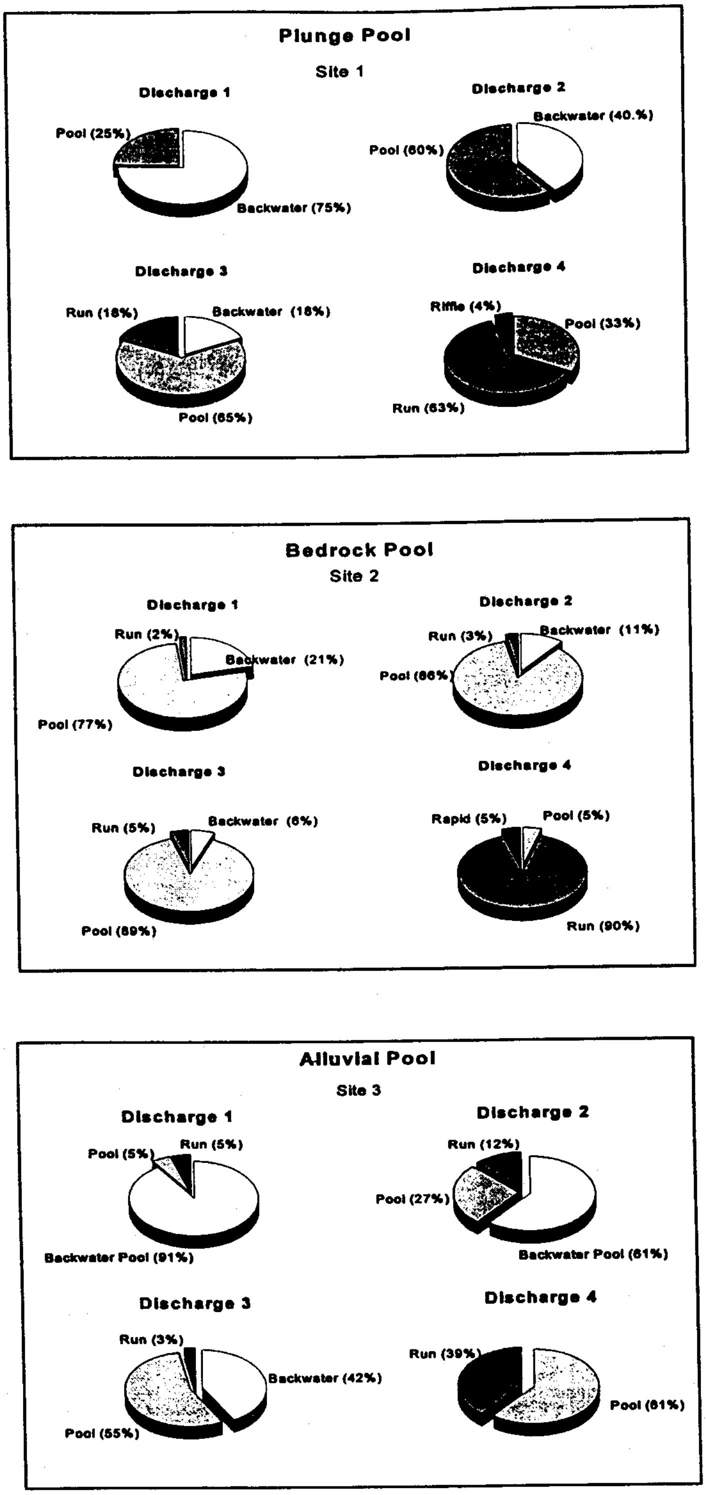

Figure 2: Variation in the distribution of hydraulic biotopes with discharge for pool morphological units

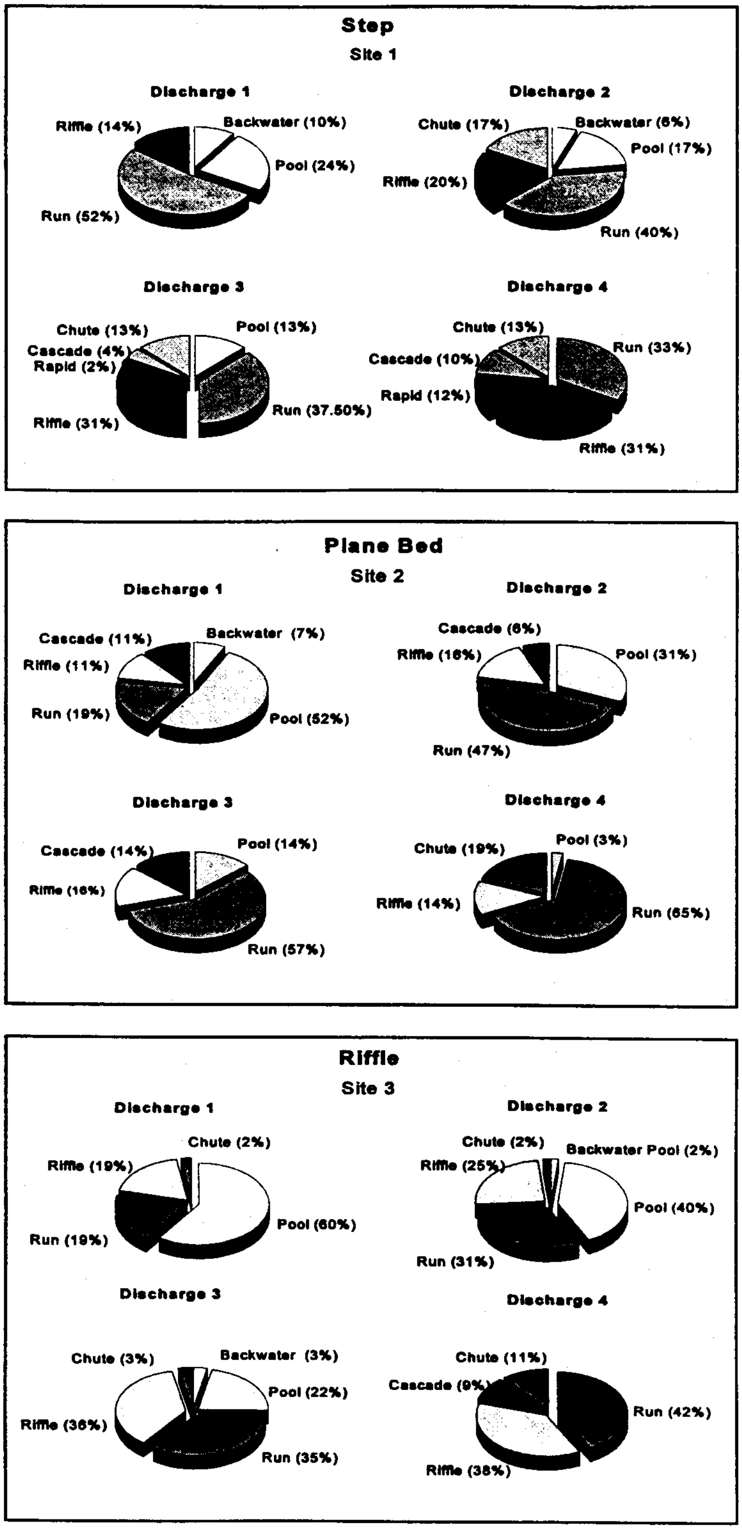

Figure 3: Variation in the distribution of hydraulic biotopes with discharge for hydraulic control morphological units

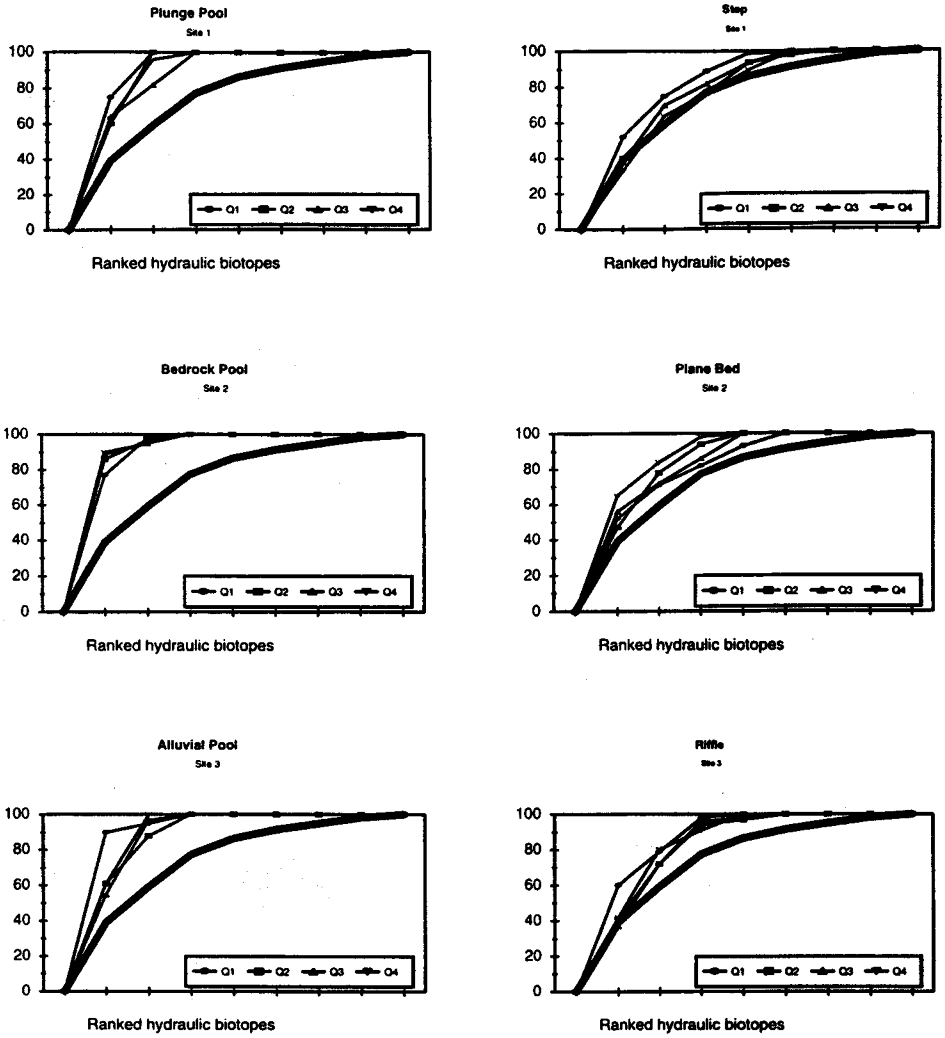

Figure 4: Hydraulic biotope diversity curves

It is clear that at discharges with flow exceedences between 92 percent to 50 percent there was little change in the type of hydraulic habitat available whereas a major change took place when the river was in spate. Unfortunately no intermediate discharges were sampled so that it is not possible from the available data to pinpoint the discharge exceedence at which major changes started to take place.

In the case of hydraulic controls - the step, plane bed and riffle - greater differences can be observed between the three morphological units. It is clear that at all discharges there was a far greater diversity of hydraulic biotopes. The step morphological unit showed a general increase in diversity from discharge 1 to discharge 4, with both a greater number of hydraulic biotopes present and a more uniform distribution amongst the different biotopes. Low energy biotopes such as backwater pool and pool gave way to high energy biotopes such as chutes, cascades and rapids. Run and riffle biotopes were maintained at all discharges. It should be noted that at the highest discharge loss of pools may have lead to an effective reduction in biotope diversity which is not brought out by the high HBDI of 0.89 in Table 4.

At low discharges the plane bed had a relatively high diversity, dominated by pool and backwater pool. Runs, riffles and cascades were also present. As discharge increased backwater pools were lost and pools were largely replaced by runs. Riffles and cascades were maintained. This pattern continued through the higher discharges, with increases in runs at the expense of pool and replacement of cascades by chutes at the highest discharge. Diversity was lowest at the highest discharge.

Perhaps surprisingly, riffle biotopes only came to dominate the riffle morphological units at discharge 3 when maximum diversity was observed. At low discharges the riffle was dominated by pool biotopes, with run and riffle being more or less evenly represented. At the highest discharge, increasing flow depth over the coarse cobble substrate caused riffle biotopes to give way to runs; pool biotopes disappeared and chutes and cascades also became significant as water began to flow over the largest cobbles.

It can be seen from Table 4 that, with the exception of the plane bed, all morphological features showed a significant increase in hydraulic biotope diversity as discharge increased from 92 percent flow exceedence to 73 percent exceedence. Little change in overall diversity occurred as flow increased to the 50 percent exceedence level, but diversity tended to fall significantly at the highest discharge. It would therefore seem that a discharge lying between the 70 percent and 50 percent exceedence level would be the most favourable in maintaining optimum habitat conditions.

CONCLUSIONS

For selected morphological units in the upper reaches of the Buffalo river it has been shown that available habitat described in terms of hydraulic biotope classes varies both between morphological units and with discharge. Not surprisingly, the pool morphological units showed the least diversity of hydraulic biotope classes whilst hydraulic controls, due to their high relative roughness and shallow depth, provide far greater habitat diversity at all measured discharges. The next challenge is to extend the research to a wider range of morphological units and river environments to see if general relationships can be found.

Hydraulic biotope diversity was measured using the Hydraulic Habitat Diversity Index (HBDI). In nearly all cases this was highest for intermediate discharges, indicating that in terms of instream flow requirements, it is these intermediate discharges that would provide the most favourable habitat conditions. It is suggested that the HBDI has the potential to provide a tool for assessing instream flow requirements. It is based on a simple classification of flow characteristics and requires limited field measurements of hydraulic modelling. A limitation of the hydraulic biotope classification as presented here is that it does not give a measure of actual velocity or depth, both important habitat variables in themselves. It is possible that the HBDI could be combined with at-a-station hydraulic geometry to give a more complete picture of how habitat changes with discharge at a particular site. There is obvious scope for further research along these lines.

ACKNOWLEDGEMENTS

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FISH HABITAT AND MICRO STRUCTURE OF FLOW IN GRAVEL BED STREAM

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ABSTRACT

This study treats the fish habitat in gravel-bed stream. At first, we have to know the interrelating system composed of flow, sediment transport and river morphology, in order to evaluate the physical properties of the fish habitat. According to IFIM, coupling of preference curves for some species of fish and the evaluation of local velocity and depth would provide us a fish-habitat evaluation. In the first part of this paper, spatial variety of hydraulic parameters are discussed by introducing the survey of the river morphology and employment of numerical analysis of flow (depth-averaged two-dimensional model with k-ε turbulence model), where the effect of vegetation is also discussed.

Even in such a methodology, the micro structure of flow is not well taken into account yet. In gravel-bed streams, more microscopic structures would appear, and they affect the fish habitat somehow. In this study, the author has focused the following fact: Fine sand is transported even in a gravel-bed stream, and we often observe sand ribbons which are constituted by longitudinal stripes of sand. And, in such conditions, we always find a group of fish staying facing upstream there. When the bed roughness changes laterally, cellular motion appears and the upward motion appears in the region above the fine sand stripe. In other words, the preference properties of fish or the fish habitat may be related to more microscopic structure than depth-averaged structure. Hence, the second part of this paper, the micro structure related to cellular motion is discussed, and an experiment using a species of fish in a laboratory flume for the preference of micro flow structure is reported. In oredr to investigate the fish preference to the micro structure of flow, the cellular motion control is a useful technique and numerical analysis supports it.

KEY-WORDS : Fishway habitat / Fluvial hydrauilcs / Flow with vegetation / Depth-averaged flow analysis / Cellular motion / k-ε model / Strip roughness / Preference properties

INTRODUCTION

In rivers, flow, sediment transport and river morphology are interacted. Furthermore, one has to include vegetation into the interrelating system in rivers. Such an interrelating system should be well understood when the safety against flood, water resources utilization and river environment are discussed. From the viewpoint of habitat hydraulics, these four elements provide the habitat of many organisms respectively, and furthermore, they are closely affected one another. This interrelating system is affected by artificial impacts such as river regulation, discharge regulation, construction of dams and reservoirs, and so on. In addition to direct change of rivers, the frequency of flood, daily change of discharge and sediment budget have been changed and its affect the above-mentioned system in rivers.

Particularly, in a reach of rivers in a fluvial fan, gravel-bed rivers, the following facts affect the fish habitat very actively: (i) the daily change of water discharge is in wide range; (ii) sediment transport during flood is very active and morphological change is appreciable; (iii) spatial variation of hydraulic parameters (depth, velocity and so on) is comparatively wide; (iv) the interrelating system responds sensitively and rapidly to various impacts (dam construction and so on); (v) species of vegetation has less variety (willow (*Salix gilgianna*) and Tsuruyoshi (reed, *Phragmites japonica*) are representative), but it grows rapidly in low-flow stage while it easily destroyed during flood. Furthermore, the species of fish in this reach is relatively limited (compared with more downstream reach).

This study treats the fish habitat in gravel-bed stream. At first, we have to know the interrelating system composed of flow, sediment transport and river morphology, in order to evaluate the physical properties of the fish habitat. According to IFIM (Instream Flow Increment Method; Platts *et al.*, 1983), coupling of preference curves for some species of fish and the evaluation of local velocity and depth would provide us a fish-habitat evaluation. At the first part of this paper, spatial variety of hydraulic parameters are discussed by introducing the survey of the river morphology and employment of numerical analysis of flow (depth-averaged two-dimensional model with k-ε turbulence model), where the effect of vegetation is also discussed.

Even in such a methodology, the micro structure of flow is not well taken into account yet. In gravel-bed streams, more microscopic structures would appear, and they affect the fish habitat somehow. In this study, the author has focused the following fact: Fine sand is transported even in a gravel-bed stream, and we often observe sand ribbons which are constituted by longitudinal stripes of sand (laterally alternate sorting). And, in such conditions, we always find a group of fish staying facing upstream there. When the bed roughness changes laterally, cellular motion (longitudinal vortices) appears and the upward motion appears in the region above the fine sand stripe. In other words, the preference properties of fish or the fish habitat may be related to more microscopic structure than depth-averaged structure. Hence, the second part of this paper, the micro structure related to cellular motion is discussed, and an experiment using a species of fish in a laboratory flume for the preference of micro flow structure is reported.

MORPHOLOGY OF GRAVEL BED AND SPATIAL VARIATION OF FLOW

Field Survey of River Morphology

Thought there are several scales depending on the viwepoints. Particularly, the following two scales are important from the view point of river hydraulics as well as fish habitat : (i) *Segment* : a reach with characteristic slope, bed materials, and bed configurations. A reach of fluvial fan is one of segment, which is characterized as a gravel-bed river with a slope around 1/100. (ii) *Unit of river morphology* : such as a wave length of alternate bars. For the scale (i), one dimensional hydraulic analysis may be suitable where the cross sectionally averaged flow is treated; while for the scale of (ii), two-dimensional model would be employed where the depth-averaged flow is described associated to the river-bed morphology. The effect of vegetaion is also significant in this level. However, the cellular motion cannot be described without three-dimensional analysis.

In order to understand the scale of unit of river morphology, river morphology and vegetation on the flood plain should be surveyed simultaneously. The accuracy and the method of survey were discussed and carried out in the river Tedori, one of the typical fluvial fan rivers in Japan (Tsujimoto, 1994). The coordinates of locations to characterize the morphology, bed materials and vegetation (species, density and dimensions) are surveyed by using a theodolite and a staff. The contour pitch is 0.5m and the data-discretizing grid is 5m x 5m. The method by using a theodolite makes possible to get such an accuracy. The data of elevation of the ground, vegetation density and its dimension, bed materials are stored in grids of 5m x 5m. Reversely from such discretized data, one can draw the spatial distrubutions of species, density and dimensions for each vegetation overwrapped on the contour of the ground elevation of the flood plain. The morphology of the submerged region is possible to surveyed in the same manner to the above during the lowest water stage. In general, the depth of water in dry season is very small for a fluvial-fan river.

In summer season of 1995, we conducted a morphological survey of one unit within straightened low-water revetment lines of the Sai river (Ishikawa Prefecture). Because the crest of the revetment was artificially constructed and its elevation was recorded, we could know the elevation from the sea level. As easily seen from the contour or the perspective shown in Fig.1, a bar (a part of alternate bars) exists on the right hand side. In the figure, x=longitudinal coordinate (flow direction), and y=transverse coordinate measured from the right hand side low water revetment line.

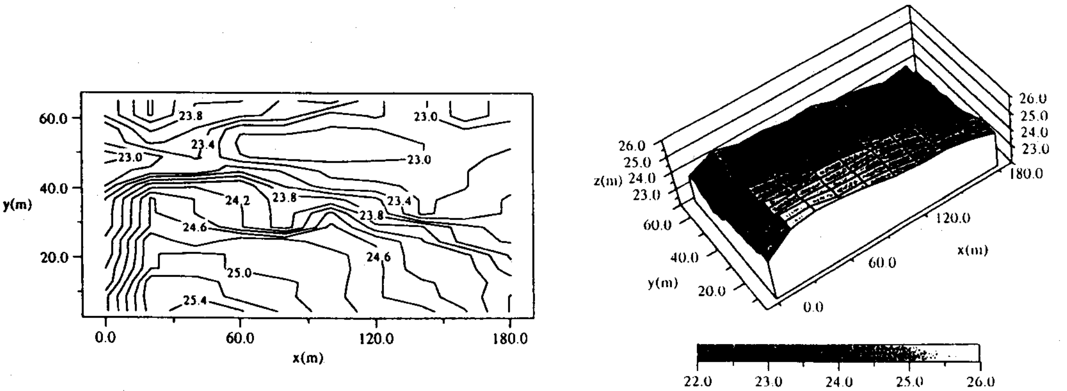

Fig.1: River morphology represented by contours of elevation and perspective (The Sai River, Ishikawa Prefecture, October 1995)

Figure 2 shows the daily discharge during a year at the surveyed reach of the Sai river, where the series of 10 years are analyzed statistically, and the average (solid curve) and the standard deviation (dotted curve) are shown. It is characterized by floods due to melting snow in April, due to "tsyu" front in early July, and due to Typhoon. Among them, the snow melt flood in this area is rather regular as recognized from a small standard deviation of discharge. Such characteristics of discharge variation must be inspected in consideration of the habitat conditions in rivers.

Vegetation

In summer season, the dried area was covered by Tsuruyoshi (*Phragmites japonica*) though its density and height were not appreciable in the area with higher "relative height". The author (Tsujimoto, 1994) analyzed the correlation of the vegetation and morphological and hydrologic parameters for the data of the River Tedori. According to his results, the area where the reed (*Phragmites japonica*) grow actively is limited due to the relative height from the watre level of the annual mean discharge and the transverse bed slope. In this study, the relative height is defined as the ground level from the water surface for the annual mean discharge. If the basic discharge is defined in another way, the relative height changes. In the previous studies, the definition of the relative height was ambiguous. In order to know the relative height, one has to know the water surface level for the given discharge, and hydraulic analysis would help it.

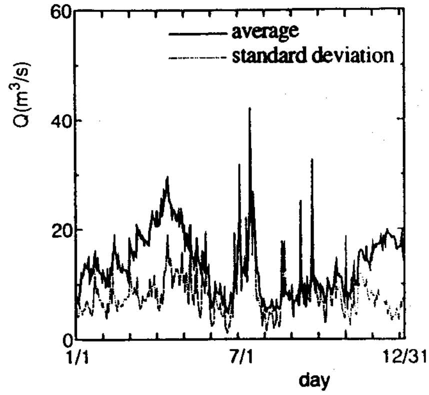

Fig.2: Daily change of discharge (The Sai River)

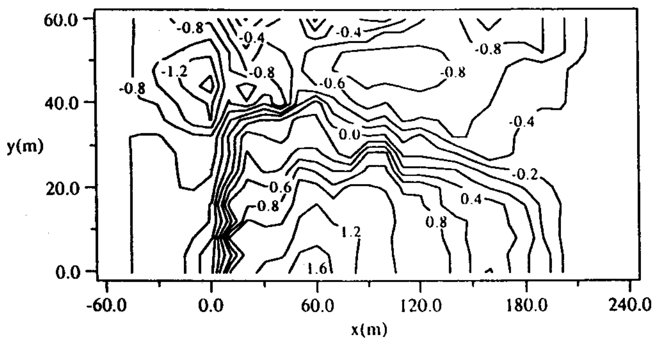

Fig.3: Spatial distribution of relative height in the Sai River

By the two-dimensional analysis of flow for the annual mean discharge (the method of analysis will be explained in the following section), the relative height of the reach of which morphology is depicted by elevation in Fig.1 is shown in Fig.3. The zero relative height implies the water edge, and the negative values implies the submerged area to contribute the flow. According to the results of the field survey in the Tedori river, the preference zone of Tsuruyoshi (reed) has the range of the relative height -1.0m~1.0m. But it cannot be applied directly to the Sai river. One has to take the difference of the scale of rivers into account. The preference relative height might be normalized by the range of the water surface due to the annual change of discharge, for example. Here, the scale is represented by the width, and it is assumed that the preference zone of reed in the Sai river is assumed to have the relative height -0.2m~0.2m.

Depth-Averaged Flow Analysis

When the microscale fish habitat is discussed in the scale of the unit of river morphology such as a bar, depth-averaged flow analysis has a suitable accuracy, which would provide us spatial distributions of depth and velocity in the reach, and would support the methodology such as IFIM (Platts *et al.*, 1983) to work. In general, the flow with complicated morphology and/or vegetation is treated and thus the change of the turbulent structure might be expected. Hence, the k-ε turbulence model is here employed. The governing equations are written as follows:

(1)
$$\frac{\partial}{\partial x}(hU)+\frac{\partial}{\partial y}(hV)=0$$

(2)
$$\frac{\partial}{\partial x}\left(\rho hU^2-\rho h\nu_T\frac{\partial U}{\partial x}\right)+\frac{\partial}{\partial y}\left(\rho hVU-\rho h\nu_T\frac{\partial U}{\partial y}\right)$$
$$=-\rho gh\frac{\partial(h+z_b)}{\partial x}+\frac{\partial}{\partial x}\left(\rho h\nu_T\frac{\partial U}{\partial x}\right)+\frac{\partial}{\partial y}\left(\rho h\nu_T\frac{\partial V}{\partial x}\right)-\rho\left(C_f+\frac{1}{2}C_D\lambda h\right)U\sqrt{U^2+V^2}$$

(3)
$$\frac{\partial}{\partial x}\left(\rho hVU-\rho h\nu_T\frac{\partial V}{\partial x}\right)+\frac{\partial}{\partial y}\left(\rho hV^2-\rho h\nu_T\frac{\partial V}{\partial y}\right)$$
$$=-\rho gh\frac{\partial(h+z_b)}{\partial y}+\frac{\partial}{\partial x}\left(\rho h\nu_T\frac{\partial U}{\partial y}\right)+\frac{\partial}{\partial y}\left(\rho h\nu_T\frac{\partial V}{\partial y}\right)-\rho\left(C_f+\frac{1}{2}C_D\lambda h\right)V\sqrt{U^2+V^2}$$

in which x, y=longitudinal and lateral directions; U, V=longitudinal and lateral components of depth-averaged velocity; h=flow depth; ρ=mass density of water; z_b=bed elevation; C_f=friction factor of the bed; C_D=drag coefficient of individual plants; λ=vegetation density (projected area of plants to the flow per unit volume of water); and ν_T=depth-averaged kinematic eddy viscosity. The effect of vegetation has been taken into account as spatially averaged drag. When a k-ε turbulence model is employed, the equations for the turbulent energy k and its dissipation rate ε are added, into which the additional turbulent energy production is taken into account corresponding to the work done by the drag.

(4)
$$\frac{\partial}{\partial x}\left(\rho hUk-\rho h\frac{\nu_T}{\sigma_k}\frac{\partial k}{\partial x}\right)+\frac{\partial}{\partial y}\left(\rho hVk-\rho h\frac{\nu_T}{\sigma_k}\frac{\partial V}{\partial y}\right)$$
$$=P_k+P_{kb}+P_{kv}-\rho hC_\varepsilon\varepsilon$$

(5)
$$\frac{\partial}{\partial x}\left(\rho hU\varepsilon-\rho h\frac{\nu_T}{\sigma_\varepsilon}\frac{\partial \kappa}{\partial x}\right)+\frac{\partial}{\partial y}\left(\rho hV\varepsilon-\rho h\frac{\nu_T}{\sigma_\varepsilon}\frac{\partial V}{\partial y}\right)$$
$$=\frac{\varepsilon}{k}\left\{\left[P_k+P_{kb}+P_{kv}\right]-\rho hC_\varepsilon\varepsilon\right\}$$

in which P_k=turbulence energy production rate per unit area due to the outlined velocity profile; P_{kb}=turbulence energy production rate per unit area due to bottom friction; P_{kv}=turbulence energy production rate per unit area due to drag by vegetation; ε=depth-averaged dissipation rate of turbulent energy per unit volume of water; and σ_k, σ_ε, C_ε=model parameters (standard values are employed here: $\sigma_k=\sigma_\varepsilon=C_\varepsilon=1.0$). If the vertical variation would be anyhow taken into account, P_{kb} must be related to the vertical gradient of velocity. If the local flow behavior around respective plants is described, P_{kv} can be related to the local velocity gradients, but here it corrsponds to the workdone by locally averaged drag by plants. These are written as follows:

(6)
$$P_k=\rho h\nu_T\left[2\left(\frac{\partial U}{\partial x}\right)^2+2\left(\frac{\partial V}{\partial y}\right)^2+\left(\frac{\partial U}{\partial y}+\frac{\partial V}{\partial x}\right)\right]$$

(7) $$P_{kb} = \rho C_k u_*^2 \; ; \qquad C_k = \frac{1}{\sqrt{C_f}}$$

(8) $$P_{kv} = hC_{fk}(F_x U + F_z W)$$

in which $u_* \equiv \sqrt{ghI_e}$=shear velocity; I_e=energy gradient; C_{fk}=model parameter; and F_x, F_z=longitudinal and lateral components of depth averaged drag per unit volume of water. F_x, F_z are written as follows:

(9) $$F_x = \frac{1}{2}\rho C_D \lambda U\sqrt{U^2+W^2} \; ; \qquad F_y = \frac{1}{2}\rho C_D \lambda W\sqrt{U^2+W^2}$$

As for P_{kb}, the model by Rastogi & Rodi (1979) has been employed; while as for P_{kv}, the model by Tsujimoto, Kitamura & Shimizu (1991) has been employed where C_{fk}=1.0 and $C_{f\varepsilon}$=1.3. C_μ=0.09; $C_{\varepsilon 1}$=1.44; $C_{\varepsilon 2}$=1.92; σ_k=0.9; and σ_ε=1.3. The parameters C_{fk} and $C_{f\varepsilon}$ are determined by consulting the study on flow over vegetation layer (Shimizu *et al.*, 1994). The kinematic eddy viscosity is related to the turbulent energy and its dissipation rate as follows:

(10) $$\nu_T = \frac{C_\mu k^2}{\varepsilon}$$

in which C_μ=0.09.

Figures 4 and 5 shows the depth-averaged flow structure for flood which will happen once a year. Fig.4 is the results calculated without vegetation, while Fig.5 with vegetation. The shaded area of Fig.5 is postulated as a vegetated zone with the vegetation density λ=0.8m^{-1} (100 stems of reed with the diameter 8mm are assumed per 1m^2). In the figure, the isovels, vector expression of velocity and the spatial distribution of the depth are depicted, and when they are couples with the preference curves of hydraulic parameters, IFIM will provide a reasonable evaluation of fish habitat better than the previous evaluation.

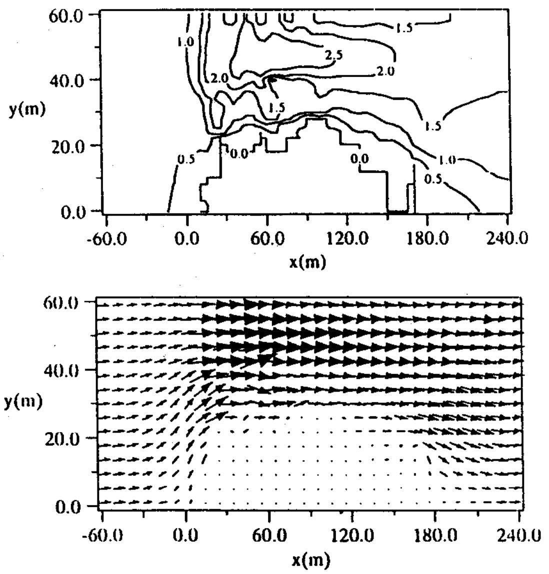

Fig.4a: Isovels and velocity vector of flow (without vegetation, Q=75m^3/s)

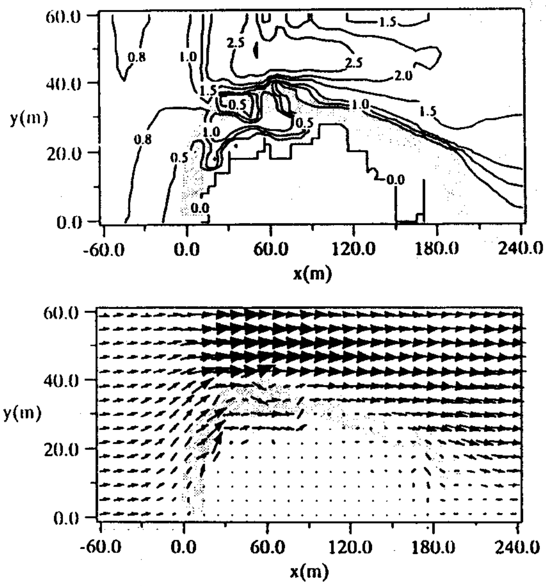

Fig.5a: Isovels and velocity vector of flow (with vegetation, Q=75m^3/s)

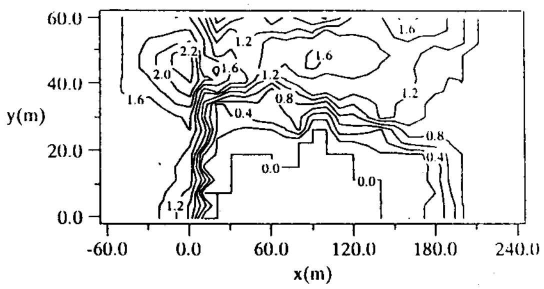

Fig.4b: Isovels and velocity vector of flow (without vegetation, $Q=75m^3/s$)

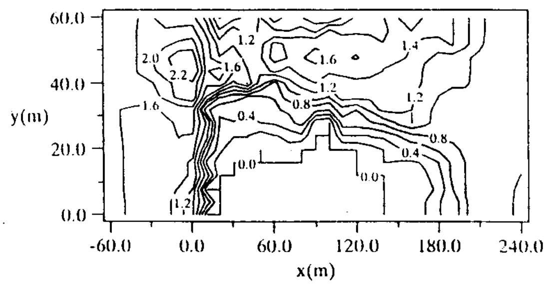

Fig.5b: Isovels and velocity vector of flow (with vegetation, $Q=75m^3/s$)

CELLULAR MOTION AND MICRO HABITAT FOR FISH

As mentioned in the introduction, the fish habitat is often subjected to more micro structure of flow, and cellular motion is a key of fish habitat. The cellular motion is often caused by sand ribbon as alternate sorting sorting and it might be related to the fish habitat. However, it is not suitable to control artificial. On the other hand, the author proposed a method to control cellular motion by oblique arrangement of strip roughness (1992, 1995), and this device is employed here to investigate the relation of micro structure of flow and fish preference properties.

Three-Dimensional Analysis of Flow

More microscopic or detailed flow structure than that evaluated by depth-averaged two-dimensional analysis might be often characterized by secondary flow and it is analyzed by three-dimensional analysis. Generalized three-dimensional analysis is not so simple, but sometimes the flow which includes secondary motion but hardly changes longitudinally represents the micro feature of habitat. Cellular motion appearing in a stream accompanied with laterally alternate sorting is a typical example (Tsujimoto and Kitamura, 1996), where the algebraic stress turbulence model was employed. Moreover, Tsujimoto (1995) studied the flow with cellular motion induced by obliquely arrangement of strip roughness, where k-ε turbulence model was employed, and such an analysis was tried to describe the flow structure of stream-type fishways (Tsujimoto, 1996). The cellular motion appearing at the flow with alternate lateral sorting is induced by the heterogeneity of turbulence, while the cellular motion induced by obliquely arranged strip roughness is driven by the non-longitudinal components of the drag. Thus, the former should be analyzed by the algebraic stress model, while the latter is analyzed by employing k-ε model.

The governing equation is abbreviated here because these are written in another paper of the author's on stream-type fishways (1996), where the drag terms written as follows are considered.

(13) $$F_x = F\sin\theta\sin\varphi\,;\quad F_y = -F\sin\theta\cos\varphi\,;\quad F_z = -F\cos\theta\,\sin\varphi$$

(14) $$F = \frac{1}{2}C_D(U\sin\theta\sin\varphi - V\sin\theta\cos\varphi - W\cos\theta\sin\varphi)\cdot\sqrt{U^2+V^2+W^2}\,\lambda$$

in which F=amplitude of drag force per unit mass of water; λ=projected area of obstacles per unit volume of water; C_D=drag coefficient of a fin, K=height of fin; and ρ=mass density of water.

Flume Experiment of Cellular Motion Related to Fish Preference Property

Cellular Motion by M- or W-type Arrangement of Strip Roughness

In order to induce cellular motion in a laboratory flume, the strip roughness is arranged in M- and W-type patterns as shown in Fig.6 with reference of the study on V- and Λ-type patterns of strip roughness (Tsujimoto, 1992, 1995). The strip was a wooden timber of which cross section was 1cm x 1cm. The interval of the neighboring strip roughness elements was set 10cm (λ=0.1cm^{-1} in the roughness layer), and the bottom was covered by gravel layer of which diameter was 5mm. The angle of skew arrangement was set $\pi/4$. The width of the flume was divided into three region: (A) zone of upward motion; (B) zone of downward motion; and (X) zone affected by side walls. Downward motion appears near the side wall in case of M-type arrangement, while upward motion appears near the side wall in case of W-type arrangement. However, in the zone near the wall, the wall effect must be more effective than the secondary flow effect, and these are treated as the X zone.

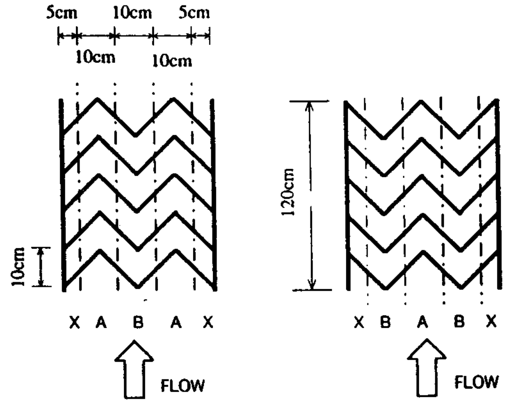

Fig.6: M and W-type arrangements of strip roughness

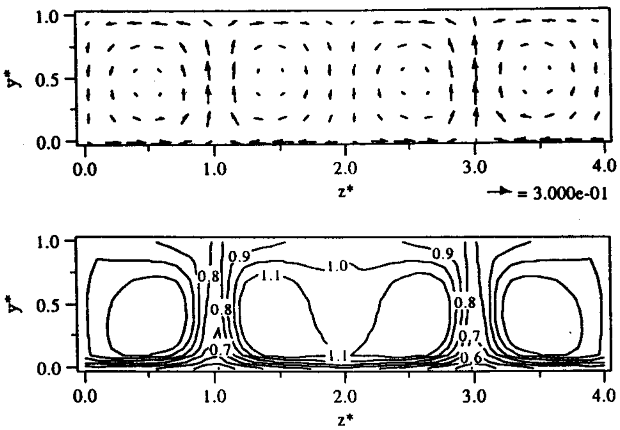

Fig.7: Results of velocity measurements of flow over M arrangements of strip roughness

By using an electromagnetic current meter, the three components of the velocity were measured. The probe of current meter was a cylinder with 8mmϕ and two components could be measured simultaneously. By changing the orientation of the probe, (U,V) and (V,W) were measured respectively. Fig.7 shows the results of flow measurement, and upward motion appears in (A) while downward motion appears in (B). In the figure, the velocity is expressed by cm/s. The primary velocity is decreased in the zones with upward motion, while the flow is accelerated in the zone with downward motion.

Figure 8 shows the calculated results by the method explained above, and the micro structure of the flow can be comparatively well described by the calculation. However, no reliable data could be obtained near the boundary (water surface , side walls and bottom) because of the size of the probe.

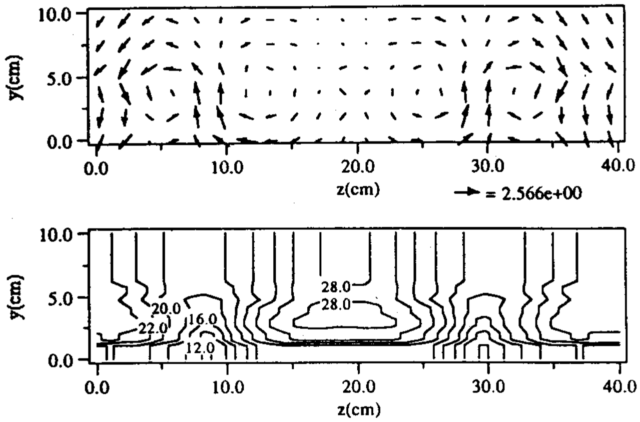

Fig.8: Results of numerical analysis of flow over M arrangements of strip roughness

Observation of Fish Behavior in Flow with Cellular Motion

In order to inspect the fish preference tendency and the secondary flow structure, the behavior of fish was observed in the flume. For observation, pale chub (*Zacco platypus*) is employed. Samples (around a few cm long) were collected at the Jintsu River (Toyama Prefecture). 8 samples were released in the flume. The test reach was 2.0m and the nets were prepared at its downstream and upstream ends to prevent the fish from swimming out of the reach. The observation of fish behavir was conducted by using a CCD video camera from the ceiling of the laboratory. The flow depth was about 10cm, and the fish stayed at the height about 2~3cm above the bed facing upstream. Most of the samples gathered near the upstream end of the test reach.

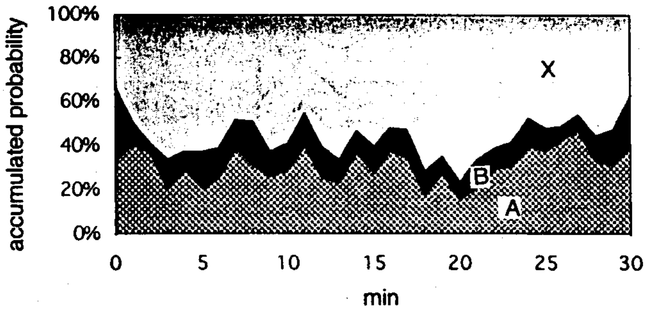

Fig.9: Probability for fish to stay in A, B and X zones

From the video tape, the location of respective sample fish was recorded per each 2s, and the probability for the fish stay in the respective zones was investigated. The results are shown in Fig.9 where the result of flumes with M and W-type arrangements of strip roughness were added and averaged to avoid the difference of the width of zones. The zone near the side walls (Z zone) shows the highest probability. Comparing a and B zones, the probability for fish to stay in the A zone is obviously higher than B zone. In this study, A and B zones are characterized by vertical secondary motion, but the primary velocity in A zone is lower than that in B zone and it is difficult to conclude the preference of secondary motion directly.

Experiment by Using a Special Arrangement of Strip Roughness

To avoid the effect of difference of the primary velocity for comparison of the secondary flow effect, the following special arrangement of strip roughness was devised. At first, Λ-type arrangement roughness was set in the half of the width to induce the up ward motion at the center of the half width; while strip roughness was set perpendicular to the longitudinal axis in the other half of the width not to induce the secondary flow. The flow is retarded by the upward motion. In order to get the similar condition for the primary velocity near the bottom (1/5 depth from the bottom where sample fish usually stays), the non-skew strip roughness is densely arranged. The interval of the strip roughness are determined by trial and error by using a numerical analysis. Finally, the strip roughness was arranged as shown in Fig.10. Then, the flow structure as shown in Fig.11 is expected.

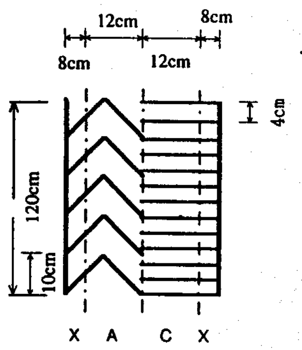

Fig.10: Special arrangement of strip roughness

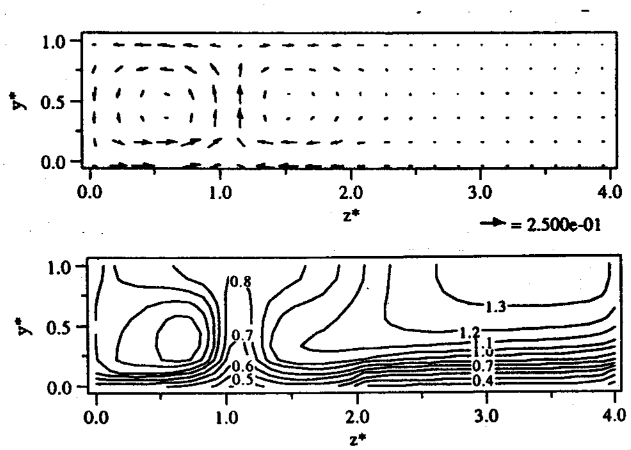

Fig.11: Micro structure of flow over special arrangement of strip roughness

The observation of fish behavior in the flow with the above-mentioned special arrangement strip roughness was conducted in the same manner to the preceding section. The result is shown in Fig.12. After the sample fish was released to the test reach of the flume, fish stays in almost same probability, but increasingly the probability to stay in the zone of upward motion becomes high.

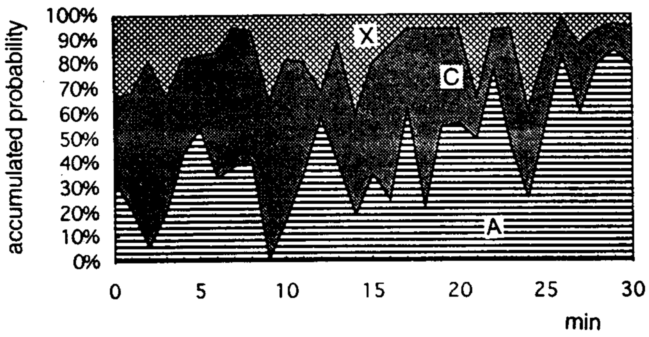

Fig.12: Probability for fish to stay in A, C and X zones

CONCLUSION

In this study, hydraulic approach to the fish habitat is discussed by developing the numerical analysis and experimental techniques. In the first part of this paper, the habitat of the scale as large as the unit of river morphology such as alternate bars. In this scale, the river morphology and vegetation affect the flow structure appreciably, and depth-averaged two dimensional flow analysis can clarified the spatial distribution of hydraulic parameters. If a method like IFIM is employed, the results would be coupled with the habitat observation data. Meanwhile, more micro structure of flow such as cellular secondary motion often becomes important factor for fish habitat. The numerical analysis of such a flow and cellular motion control for investigation of fish preference properties by obliquely arrangement of strip roughness are developed.

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BETWEEN-REACH CONTRASTS IN CHANNEL FORM, HYDRAULICS, LATERAL STABILITY AND RIFFLE SAND CONTENT IN AN UNSTABLE GRAVEL BED RIVER: IMPLICATIONS FOR ATLANTIC SALMON HABITAT

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ABSTRACT

As part of ongoing research into the fluvial geomorphology of salmonid habitat, a study was conducted on the Nouvelle River (Gaspé Peninsula, Québec) into the effects that strong between-reach contrasts in channel form and lateral stability have on Atlantic salmon habitat. The lower Nouvelle River is a "wandering-type" gravel-cobble system that has been impacted this century by riparian forest clearing and channel rectification.

The geomorphic controls affecting two key physical dimensions of salmon habitat, ie. the spatial distribution of hydraulic habitat suitable for juvenile growth and the quality of spawning substrate, were investigated. Five reaches of contrasting geomorphic character but formed under the same discharge regime were studied. The statistical distribution of "hydraulic habitat" (channel areas with various combinations of depth, velocity and substrate) suitable for fry and parr growth was estimated at two summer discharges and contrasts in weighted usable areas were related to the dominant morphological processes in each of the study reaches. Juvenile habitat at low summer stages, expressed as average usable width in each reach, varied from 4 to 21 m for fry and 4 to 12 m for parr between the study reaches, reflecting characteristic morphological contrasts. Two reaches, rectified in the 1960s and presently displaying very high local curvature and a minor backchannel, offered the least juvenile habitat because of the narrow and deep character of the main channel. Single thread, moderately sinuous reaches offered intermediate amounts of habitat, while an overwidened braid-like reach dominated by a massive riffle offered the most hydraulic habitat. These contrasts can be explained in terms of variation in channel width/depth ratio, in pool/riffle amplitude as it relates to curvature and in the availability of mobile slugs of riffle-forming bed materials.

The study of spawning substrate integrity focused on the potential for detrimental sand enrichment in spawning beds downstream from sandy-gravel cutbanks which, locally, undergo extremely high rates of shifting (up to 10 m/yr). Eight cutbank zones were selected for study. Each zone incorporated a retreating bank, for which sediment production and sand content were determined, and the two nearest riffles (ie. immediately upstream and downstream of the eroding bank). Six to eight bulk samples (averaging 40 kg) were extracted from potential spawning zones at these riffles. Paired *t*-tests revealed no statistically significant sand enrichment effect downstream of highly eroding cutbanks. This suggests that in systems with relatively high levels of hydraulic power, like the Nouvelle River (70-100 W/m^2), the geomorphic mechanisms working to prevent sand buildup in riffle zones are capable of handling the substantial sand input from cutbanks with 30-60% sand content retreating up to 10 m per year.

KEY-WORDS: Atlantic salmon / river geomorphology / parr and fry habitat / spawning substrate / bank erosion/ braiding / meandering.

INTRODUCTION

As part of ongoing research into the fluvial geomorphology of salmonid habitat, this study investigates the effects that strong between-reach contrasts in channel form and lateral stability can have on key physical parameters of Atlantic salmon habitat. Many riverine fish species require a specific range of depths and current velocities for optimal growth and survival. Hogan and Church (1989) termed this range the "hydraulic habitat" of a species. The geomorphic variables which control channel hydraulic geometry, such as discharge, valley gradient and bed materials, heavily influence the amount of hydraulic habitat in rivers (Frissell *et al.*, 1986; Binns and Eiserman, 1979; Platts, 1979). However, spatial variation in the supply of bed calibre sediment to a given reach can also introduce local variability to the planimetric form, even if variables at the watershed scale remain constant (Church, 1983; Carson, 1984a,b). The effect of planform on fish habitat is widely recognized (Rabeni and Jacobson, 1993; Beschta and Platts, 1986; Heede and Rinne, 1990), but to date, few studies have systematically examined the extent to which variations in this parameter along a length of river can affect the distribution of salmonid rearing habitat.

The variable content of fine sediment in gravel-bed river substrate is also a key issue for managers wishing to preserve salmonid habitat (Adams and Beschta, 1980; Lisle, 1989). For salmonid species, excess fines can lower survival by inhibiting the supply of oxygen to, and the removal of metabolites from, salmon embryos developing in the interstices of the gravel (McNeil and Ahnell, 1964). Fines also act to block the emergence of salmon fry from the gravel bed (Phillips *et al.*, 1975; Hausle and Coble, 1976). In response to large-scale bank erosion from road building and forest clearing activities, gravel beds can become enriched by fines to the point where spawning gravels are damaged (Platts and Megahan, 1975; Everest *et al.*, 1985).

In this study we investigate spatial variations in the distribution of spawning and juvenile rearing habitat for Atlantic salmon (*Salmo salar*) in geomorphologically contrasting reaches of a gravel-bed river of "wandering" pattern (Church, 1983), i.e. a pattern transitional between meandering and braiding. First, we report on the effects of planform type on the relative distribution of hydraulic habitat for juvenile Atlantic salmon (fry and parr). Secondly, we investigate the effects that rapidly eroding, sand-rich banks have on the grain-size composition of potential spawning sites downstream from these banks.

STUDY SITE

The Nouvelle River in eastern Quebec is one of several gravel-cobble rivers flowing through the Southern Gaspé peninsula of eastern Québec. Managers are currently conducting an Atlantic salmon stocking program on this river; unlike neighbouring rivers, there is no historical evidence that the river has supported a significant salmon population in the past, although there is a resident population of sea-run brook trout (*Salvelinus fontinalis*). The Nouvelle River originates at an elevation of 600 m in the Chic-Choc mountains and drains an area of 1200 km^2 into the Baie des Chaleurs. Stream flow is unregulated and subject to large seasonal fluctuations. The study focuses on the lower 18 km of the river where mean annual discharge is 26 m^3/s. Peak daily flow with two-year recurrence is 260 m^3/s and occurs during the spring snowmelt freshet. A typical late-summer low flow with 24 hour duration and 2 year recurrence is 6 m^3/s. Channel slopes range from 0.0015 to 0.0032 producing a total hydraulic stream power of some 6200 W per metre of channel length at bankfull stages. The specific flow power exerted on the channel perimeter varies from 70 to 100 W/m^2 in the study zone. These hydraulic measures place the Nouvelle River in the category of moderately high energy rivers according to the classification of Ferguson (1993). As is typical of rivers with wandering-type patterns, strong inter-reach contrasts in river dynamics exist. Reaches of high stability are relatively straight, reaches of moderate instability display gentle meanders, and highly unstable reaches are characterised by split channels and, locally, a braid-like pattern. Because of this geomorphic variability, the Nouvelle River is well suited to study the effects of geomorphic pattern on key physical parameters of salmonid habitat.

While in many areas bank erosion is minimal, lateral channel instability along a number of cutbanks is extremely high, with shifting rates between 5 and 10 m/y. High rates of bank retreat are related in part to the progressive reestablishment of an equilibrium pattern of curvature following channel rectification projects undertaken in the 1960's. In the study reaches, the river is bordered by a mixture of riparian forest and cleared floodplain. The top stratum of the cutbanks is generally 0.2 to 1.5 m thick and composed mainly of sand. Underlying this is a layer of mixed sand and gravel averaging 1 m thickness, which itself overlies a gravel and cobble stratum extending down to thalweg level. While sand content in the low flow channel substrate averages under 10 %, aggregate sand content in active cutbanks is substantially greater, ranging between 30% and 60%. Thus rapidly retreating bank sections (up to 10 m/yr) release substantial amounts of sand to the active channel and, potentially, to the substrate. In such contexts, resource managers are occasionally tempted to prescribe bank stabilisation, in order to mitigate possible deterioration of spawning habitat. In this study we will test whether, at the local scale, salmon spawning beds can be deteriorated by the strong attack of a moderately high energy river on its floodplain or low terrace cutbanks.

METHODS

Hydraulic habitat for juveniles

From aerial photographs and field visits, five river reaches of contrasting planform were chosen for quantification of hydraulic habitat (Fig. 1). Table 1 describes summary morphological parameters, such as shifting rates, average bankfull widths, radius of curvature, and degree of channel splitting for each reach. Geomorphic controls responsible for these contrasting morphological patterns will be discussed under Results.

Standard field methods were used to quantify hydraulic habitat (Bovee, 1982). Measurements of depth and mean velocity were taken along cross-sections placed approximately one to two channel widths apart throughout the study reaches. Cross-sectional data points were spaced so that each reading represented the depth and velocity conditions of a hydraulically uniform area of river, or habitat cell. Density of cross-sections and readings increased where

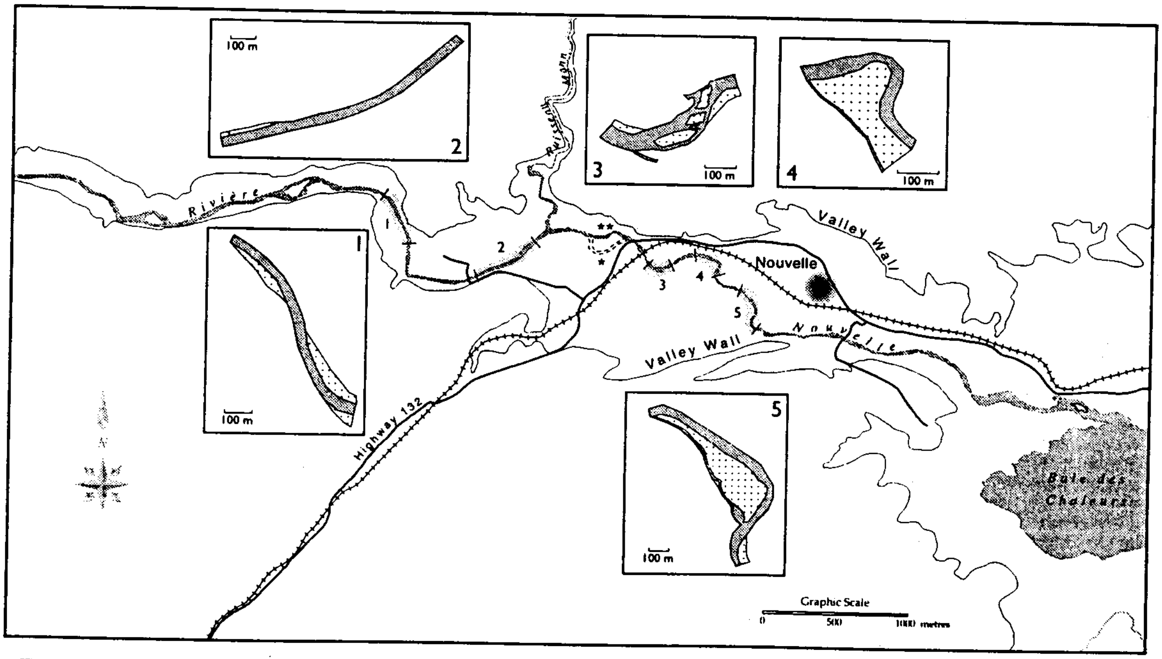

Figure 1: The Nouvelle River in the Baie des Chaleurs region of Eastern Québec, Canada. Insets show plan morphology of study reaches 1 to 5. Asterisks (near the highway) refer to the recent avulsion discussed in the text. (current river path; * river path pre-1981).**

Table 1: Planform characteristics of study areas. SR_{max} is the shifting rate at the point of maximum bank attack in a reach. B is bankfull width. Numbers in parentheses are back-channel widths. $W:D_{Q1}$ and $W:D_{Q2}$ are the ratios of average wetted width to average flow depth at Q1 (24 m^3/s) and Q2 (8 m^3/s), respectively. Curvature is measured by r_c/B, where r_c is the centreline radius of curvature. T_{max} and T_{mean} are the mean and maximum numbers of distinct thalwegs counted at channel cross-sections.

Reach	SR_{max} (m/yr)	B (m)	$W:D_{Q1}$	$W:D_{Q2}$	r_c/B	T_{mean}	T_{max}	Channel pattern description
1	4	44	56	65	9.3	1.0	1.0	single channel, moderate curvature, moderate shifting
2	0	43	72	91	34.9	1.0	1.0	single channel, low curvature, laterally stable
3	9	103	130	108	3.5	2.1	4.0	wide bend, dominant dissected riffle, rapid shifting
4	10	72 (21)	41*		1.2**	2.0	2.0	twin channel, extreme curvature, rapid shifting
5	5	61 (20)	50	35	2.9**	1.7	2.0	twin channel, strong curvature, moderate shifting
Avg.	5.6	65	77***	75***	10.4	1.6	2.0	

* Reach 4 was surveyed at an intermediate discharge of 18 m^3/s.
** main channel only. *** average excludes reach 4.

hydraulic complexity was high. Visual estimates of the size composition of gravels on the bed surface (calibrated by Wolman samples; Wolman, 1954) were conducted at all survey points since, in addition to depth and velocity, the size of bed surface particles is also a determinant of habitat useability by salmonids (Gibson *et al.*, 1990; Gibson, 1993). Juvenile Atlantic salmon habitat preference curves established by Morantz *et al.* (1987) for substrate, depth and average water-column velocities were used to compute weighted usable areas (WUAs) for fry and parr in each reach. The Morantz *et al.* suitability curves are based on detailed observations of juvenile Atlantic salmon distributions in six eastern Canadian salmon streams of small to intermediate sizes.

<u>Effects of severe bank erosion on local spawning substrate</u>

Sequential aerial photos were used to identify bank regions with high rates of lateral migration during the 1981 to 1992 period. Within the lower 10 km of the river, eight eroding cutbanks and the riffles located immediately upstream and downstream from them were identified. The general substrate sampling design in the neighbourhood of each eroding bank is schematised in Fig. 2. To compare fines content, substrate samples were extracted from the riffles both upstream and downstream of the cutbank, along cross-sections a short distance upstream of a riffle crest in the zone most commonly used for salmon spawning. To control for the strong tendency for lateral variability in substrate composition, four generic cross-sectional locations were generally sampled. Two of these samples, one sub-aerial and the other sub-aqueous, were extracted from the "barhead" side of the channel, i.e. the side submitted to strong flow velocities during formative flows (Fig. 2, sample positions 3 and 4). Similarly, a pair of sub-aerial and sub-aqueous samples were taken from the bartail or low velocity side of the channel (sample positions 1 and 2). Sub-

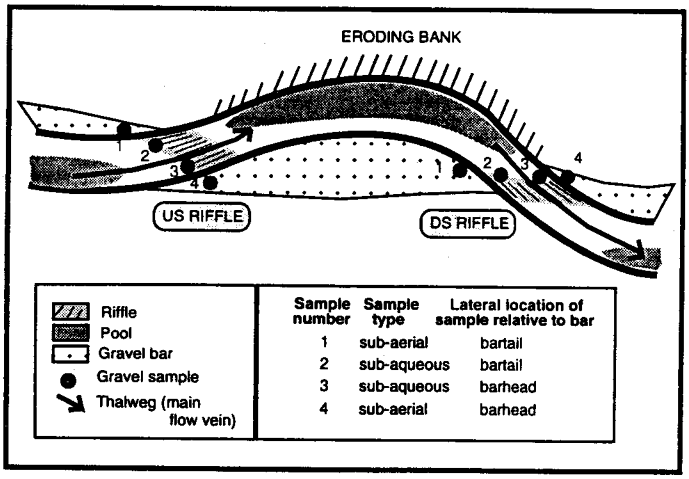

Figure 2: Schematic diagram of cutbank, riffles and generic sampling locations. Differences in sand content between upstream and downstream samples alike in location were used to evaluate the effect of eroding banks on gravel composition

aqueous samples were located such as to evenly flank the thalweg (the deepest point on the riffle cross-section). In total, 41 substrate samples were extracted this way in the neighbourhood of the eight cutbank sections.

All substrate sampling was conducted at low flow between August 13 and September 1, 1994. Discharges ranged from 7 to 15 m^3/s, which is less than the mean discharge for October (23 m^3/s), the month during which Atlantic salmon spawning commonly occurs in Gaspé rivers. The sub-areal samples (1 and 4, in Fig. 2) were extracted from the lowest elevations on exposed bars, as close as possible to the late summer water's edge: these sites would thus typically be underwater and potentially accessible for spawning during the October spawning season.

Substrate sampling technique

Volumetric samples in sub-aerial sites were extracted by shovel according to conventional bulk sampling techniques (Lisle and Madej, 1992; Church *et al.*, 1987). Sample holes were 25-30 cm deep, similar to the published depth of spawning redds (Gibson, 1993; Crisp and Carling, 1989), and the average mass of substrate extracted was 50 kg. In sub-aqueous sites (2 and 3 on Fig. 2), samples were excavated within the confines of a flow isolation cell, 60 cm in diameter and 75 cm in height, which was driven several inches into the gravel bed. Affixed to this cylindrical cell was a water-permeable panel facing upstream and a conical net facing downstream, both constructed of 100 micron, Nytex mesh. Water currents passing through the front mesh panel carried any fines thrown into suspension during digging into a cup at the end of the downstream mesh net. After substrate extraction cup contents were collected and added to the sample. Escapement of fines during underwater sampling of gravel is a common problem (Rood and Church, 1994; Crisp and Carling, 1989) which is eliminated by our bulk sampling technique. The average mass of sub-aqueous samples was 38 kg. Both sub-aerial and sub-aqueous samples were truncated at 64 mm to eliminate bias for larger size classes that are often poorly represented using other common substrate sampling techniques (Rood and Church, 1994; Church *et al.*, 1987). Coarse fractions (above 32 mm) were sieved in the field, and splits of the finer fractions were brought back to the laboratory for sieving at half phi intervals.

RESULTS

Hydraulic habitat for juveniles

In all Nouvelle River study reaches, values of D_{50}, the median particle size at the bed surface, lie between 4 and 5 cm. D_{90}, the surface particle diameter below which 90 % of particles fall, lies between 9 and 13 cm. These results indicate that the size of bed surface particles in the Nouvelle River is within the broad range of gravel-cobble substrate preferred by juvenile Atlantic salmon (Morantz *et al.*, 1987; Gibson *et al.*, 1990). For simplicity, substrate size will be excluded from subsequent comparisons of habitat among reaches.

Natural fluctuations in discharge can confound inter-reach comparisons of stage-dependent hydraulic habitat. Fortunately, in the summer of 1994, climatic conditions produced two extended periods of relatively uniform flow in the Nouvelle River, so that several study reaches could be surveyed under uniform conditions of discharge. Depth and velocity surveys were conducted in reaches 1,2,3, and 5 during the first period of uniform flow, which occurred in early summer at an average discharge (called Q1 henceforth) of 24 m^3/s. The same four reaches were surveyed again during the late summer at an average discharge (called Q2) of 8 m^3/s. This later discharge approaches the mean, two-year, summer low flow of 24 hour duration (6 m^3/s). In reach 4, time constraints allowed only one survey at a mid-summer discharge of 18 m^3/s.

Bivariate distributions of depth and velocity

Hogan and Church (1989) showed that bivariate distributions of river area by depth and velocity class are an effective way of illustrating the spatial distribution of hydraulic habitat in a channel. Bivariate distributions were generated for this study by applying a Kernel smoothing function to sampled cell data (Silverman, 1986). The relative abundance of river bed areas with various conditions of depth and velocity is depicted by the solid contour lines of Fig. 3. Early summer (Q1) conditions are displayed on the left, later summer (Q2), at the right. Each contour corresponds to increments of 2.5 % of the total wetted surface area, and peaks in the distribution represent the most widespread combinations of depth and velocity.

The differences in hydraulic habitat between the reaches are precisely described by the bivariate distributions. Given an ecologically appropriate choice of preference curves, the implications for fish densities can be assessed in a separate step by overlaying selected habitat criteria onto each distribution. To illustrate the salmon habitat availability in each reach in Fig. 3, the fry and parr habitat preference curves for depth and average water-column velocities generated by Morantz *et al.* (1987) were converted into probability-of-use (POU) factors. These factors range from 1 to 0 for optimal and unusable habitat, respectively. Dashed contours superimposed on Fig. 3 show depth-velocity combinations with POU values of 0.8 (inner contour) and 0.2 (outer contour) for two juvenile life-stages, fry (Fig. 3a, 3c, 3e, 3g) and parr (Fig. 3b, 3d, 3f, 3h). Fry and parr contours apply at either discharge, but for clarity, they are shown on separate distributions.

Reach 1

Reach 1 has a regular, moderately curved planform with well expressed outer bank erosion (Table 1 and Fig. 1). Pools and riffles are therefore well-developed (Rabeni and Jacobson, 1993; Newbury and Gaboury, 1993; Platts, 1979). The lobes in the upper left of both the Q1 and Q2 distributions of reach 1 (Fig. 3a, 3b) correspond to fast, shallow riffles; the lobes in the middle of the panels correspond to pools; and the parts of the distribution falling near the origin represent the shallow edge areas found in greatest abundance at the inner sides of bends (Rabeni and Jacobson, 1993). Because of the excessive velocities over riffles and the excessive depths of pools in this reach, edge zones are the major source of habitat for juveniles at both Q1 and Q2.

Reach 2

Reach 2 is the straightest of all study reaches. It also has slightly coarser substrate and small vertical amplitude of pool/riffle oscillation. This is reflected in Figs. 3c and 3d by a low range of depths and a poor definition of pool and riffle lobes. Much of the reach displays the moderate depths and velocities characteristic of a "run", the flat, transitional zone between a pool and riffle.

In reach 2, the lobe on the bivariate distribution corresponding to edge area is larger than in reach 1. This difference in the distribution arises because, while widths in the two reaches are similar (Table 1), the pools of reach 2 are shallower. Cross-sections in reach 2 have a higher width:depth ratio and a wider zone of edge habitat (Table 1). The wide edges of reach 2 are suitable for fry, who prefer mainly shallow water. Given the low bar and bend resistance

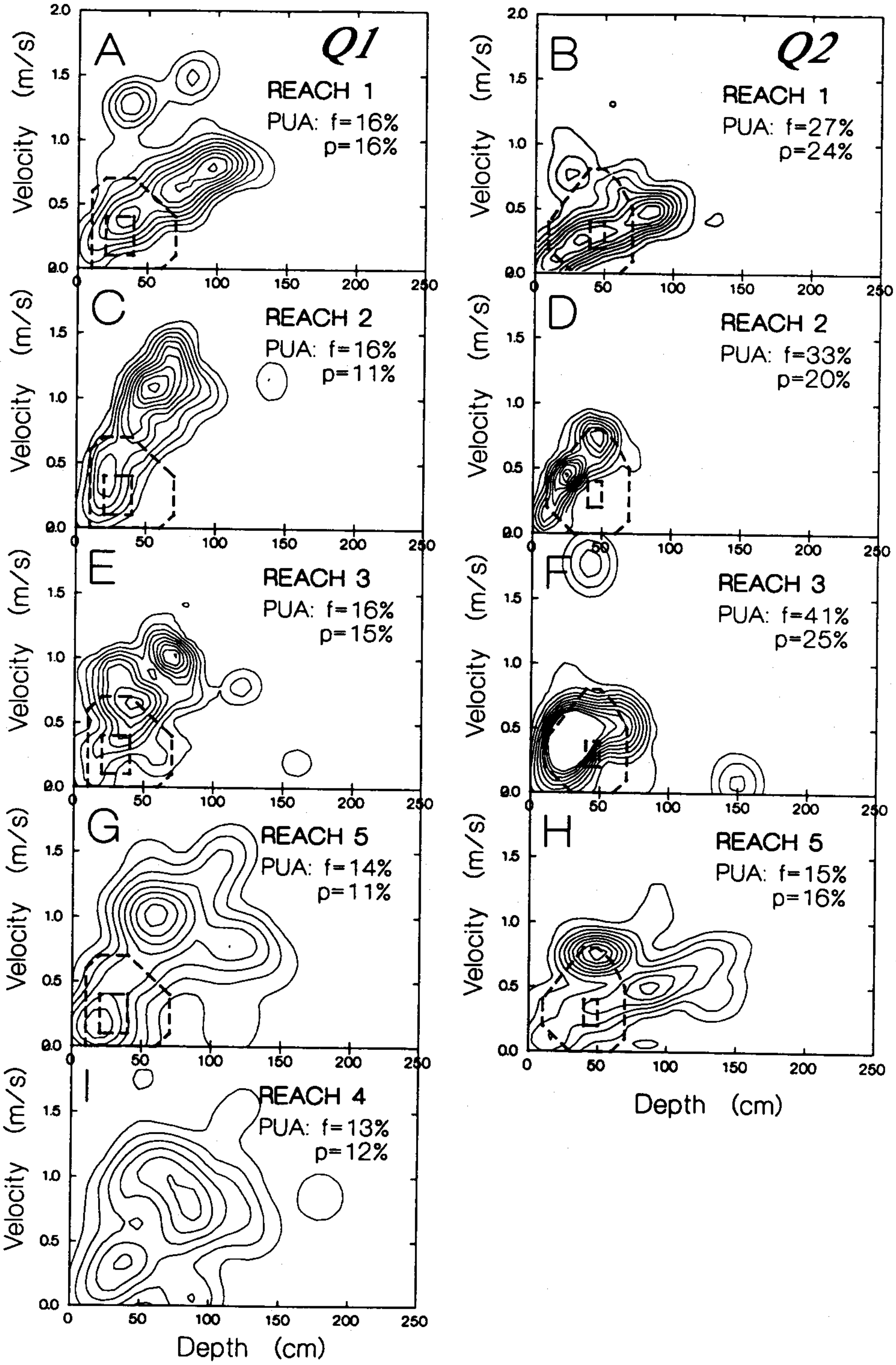

Figure 3: Bivariate distributions of depth and velocity in the study reaches. Left panels are for Q1 (24 m^3/s) conditions; right panels are for Q2 conditions (8 m^3/s). Reach 4 was sampled at an intermediate discharge of 18 m^3/s. Dashed contours correspond to probability of use values of 0.8 (inner line) and 0.2 (outer line) for fry (Q1 panels) and parr (Q2 panels) (Morantz *et al.*, 1987). Percent useable area (PUA) is shown on all plots for both fry (f) and parr (p).

in this reach, velocities over zones of moderate depth are higher than in other reaches (Fig. 3c, 3d). As a result, in the moderate depths preferred by parr, habitat tends to be limited by excessive velocities.

Reach 3

In wandering rivers, braiding behaviour often develops downstream from large sources of bed calibre sediment (Carson, 1984a,b; Church, 1983; Needham and Hey, 1992). Such local inputs can produce large-scale "waves" or "slugs" of sediment that propagate slowly downstream, triggering channel widening and severe bank erosion. Historical air photographs show that the massive riffle presently found in reach 3 (Fig. 1) resulted from such a mechanism. An avulsion, or sudden channel shift, occurred just upstream of reach 3 in the early 1980's (Fig. 1; see *, **, near highway). This avulsion involved the abandonment of a meander bend and the scouring out of an entirely new channel on a logged section of the floodplain, thus injecting a very large volume of mobile gravel into the reach immediately downstream.

Reach 3 is constituted of a wide run with moderate velocities, on the stoss or upstream side of a massive, dissected riffle structure. A narrow pool is squeezed against the right channel bank downstream of the riffle face. The generally shallow depths and complex pattern of gravel deposition lead to an irregular bivariate distribution with few areas extending beyond 1 m in depth (Fig. 3e, 3f). This particular reach morphology has two major effects on habitat availability. First, decreased depths lead to increased habitat availability for juveniles because, as was seen in reaches 1 and 2 (Fig. 3a,b,c,d), in a moderately high slope, high-energy system such as the Nouvelle River, preferred habitat falls into the range of low depths and velocities. Second, the massive gravel accumulation has increased channel widths and, therefore, increased the total potential area of habitat available compared to other reaches (Table 1). Note that between Q1 and Q2, the bivariate distribution of reach 3 undergoes a greater change in overall shape than the distributions of reaches 1 and 2. The weir-like hydraulic control exerted by the large riffle is greater at the lowest flow stages (Q2). Except for a few, fast flowing spillways down the face of the riffle (appearing as lobes in the upper left of Fig. 3f) and an almost stagnant pool along the right, riffle margin (lobe in the lower right in 3f), at Q2 the reach is dominated by fairly uniform shallow and moderately slow-flowing conditions in the wide run upstream of the riffle crest. The abundance of areas with low depths and velocities in this backwater zone is particularly suited to fry at Q2.

Reach 5

A prominent characteristic of the hydraulic habitat distribution of reach 5 is the peak appearing near the origin of Fig. 3g. This peak corresponds to the shallow, ephemeral back channel in this reach (Fig. 1) that provides the majority of habitat for juvenile salmon at Q1. By Q2, this channel dries up, leaving a sparsity of slow, shallow areas for use by fry and parr (Fig. 3h). Carrying only part of the flood flow, the highly curved main channel of reach 5 has a lower width:depth ratio than the single channel in reaches 1 or 2 (Table 1). The proportion of areas with depths greater than 75 cm is high relative to other reaches. This is in keeping with the tendency, observed between reaches 1 and 2 of this study, as well as in other rivers (Thorne, 1992) for pools to grow deeper with increasing bend curvature. Overall, the deep pool and narrow summer width in the main channel of reach 5 lowers the width: depth ratio of cross-sections and thus the availability of shallow edge habitats for juvenile salmon.

Reach 4

Reach 4, surveyed at a flow between Q1 and Q2, falls into the same general planform class as reach 5; both reaches have highly curved, narrow main channels and a minor chute channel which dries up by mid summer. The bivariate distributions of reach 4 (Fig. 3i) and reach 5 (Fig. 3g, 3h) reflect this similarity. Both distributions have a sparsity of depths below 50 cm (excluding the back channel of reach 5), an abundance of depths over 75 cm, and relatively poorly expressed pool and riffle hydraulic lobes.

Summary habitat statistics

Summary, between-reach comparisons of juvenile habitat are presented in Table 2, separately for fry and parr and for flow stages Q1 (early summer) and Q2 (late summer). Reach 4 was surveyed at a flow intermediate between Q1 and Q2. To normalise habitat availability comparisons, weighted useable areas were corrected for between-reach variations in channel length and total wetted area. In Table 2, average useable width (AUW), is the weighted useable area per unit channel length, and percent useable area (PUA), is the weighted useable area per unit of total water surface area at the given flow stage in each reach.

Table 2: Results of weighted useable area calculations for fry and parr habitat based on Morantz *et al.* (1987) criteria. PUA = percent useable area, or weighted useable area divided by the total wetted reach area; AUW = average useable width, or weighted useable area divided by the reach length.

Reach	PUA_{fry} (%)		AUW_{fry} (m)		PUA_{parr} (%)		AUW_{parr} (m)	
	Q1	Q2	Q1	Q2	Q1	Q2	Q1	Q2
1	16	27	6	9	16	24	6	8
2	16	33	6	10	11	20	4	6
3	16	41	11	21	15	25	11	12
4	13		4		12		3	
5	14	15	4	4	11	16	3	4
Avg.*	16	29	7	11	13	21	6	8

* average excludes reach 4

There are notable between-reach differences in weighted useable area. These differences are greatest at Q2, reflecting the tendency for hydraulic variations induced by river form to increase with falling stage. For instance, habitat availability in reach 3 at Q2 reflects the beneficial effects on habitat availability in this system of having a braid-like, massive riffle. Usable width for fry is 21 m, twice the average for all reaches and five times that found in reach 5 (where AUW_{fry} = 4 m). Low width:depth ratios and narrow channel edge zones place reaches 4 and 5 at the bottom of habitat rankings. In neither reach does AUW exceed 4 m. Finally, in reaches 1 and 2, the availability of shallow habitats on the inner side of bends adjacent to bars is balanced by unusable mid-stream areas. As a result, these reaches lie in the middle habitat rankings (Table 2).

<u>Effect of eroding banks on spawning habitat</u>

The left-most columns of Table 3 present characteristics of the eight cutbank sections selected for study. The total input of fines to the channel (column 4) is the product of the percent fines content of bank materials (which varies from 30-60%), the average bank migration rate, the average bank height from the thalweg and the total length of the eroding cutbank. The percent fines (< 2 mm) in the eroding bank section was estimated by a weighted average of the fines content of each bank stratum, as surveyed in the field. A sediment bulk density of 1.6 tonnes/m^3 was assumed. Table 3 shows that cutbanks erode at rates from 3.3 to 10.5 m/yr. One 260 m long cutbank section (D.2) is estimated to yield 1310 tonnes of sands per year to the channel.

On the Nouvelle River there is, as expected, a tendency for somewhat higher fines content in subareal samples than in sub-aqueous samples. Similarly, fines content tends to be greater in samples from the bartail side (where flood velocities are lower) than in samples from the barhead side (high velocity side) of the channel (see Fig. 2). Given such systematic effects of sample location, we conducted pair-wise comparisons on samples taken from similar generic locations to test for differences in the fines content of riffles upstream and downstream of eroding cutbanks. The right of Table 3 shows differences in the under 2 mm fraction for each sample pair spanning an eroding bank section. Only 5 of 15 riffle pairs show an actual downstream increase in the content of fines. Figure 4 plots fines enrichment (or depletion) across eroding banks against the estimate of yearly sand input in each case (Table 3). Scatter due to the inherent spatial variability in substrates sand content in gravel bed rivers is high. Although the slope of the best fit lines suggest an overall enrichment effect for coarse sands and fine gravel, regression analysis

Table 3: Differences in sand content in substrate samples spanning bank erosion sites. Samples paired according to lateral location on riffles, and cutbank site. Positive values in column 7 (Difference in fines content) indicate enrichment by fines downstream from eroding banks.

		Bank Material Inputs (tonnes/yr)		fines content (% < 2 mm)			
Bank	**Peak Rate of Migration (m/yr)**	**Total Input**	**Fines (<2mm) Input**	**Upstream**	**Downstream**	**Difference**	**Lateral sample location**
A	5.4	3180	960	12.7	17.3	4.6	4
A				11.3	8.4	-2.9	2
B.1	3.6	500	220	7.6	6.8	-0.8	2
B.1				8.2	3.3	-4.9	3
B.2	3.6	1400	560	22.8	18.6	-4.2	1
B.2				6.8	9.9	3.1	2
C	4.5	1220	430	15.0	24.2	9.2	1
C				12.3	7.6	-4.7	2
C				6.5	3.2	-3.3	3
C				9.7	13.0	3.3	4
D.1	3.3	630	320	8.5	5.3	-3.2	3
D.2	10.5	2900	1310	4.8	10.2	5.4	3
E.1	4.5	1220	730	11.0	6.8	-4.2	2
E.2	4.5	540	270	6.8	5.3	-1.5	2
E.2				17.5	8.0	-9.5	4

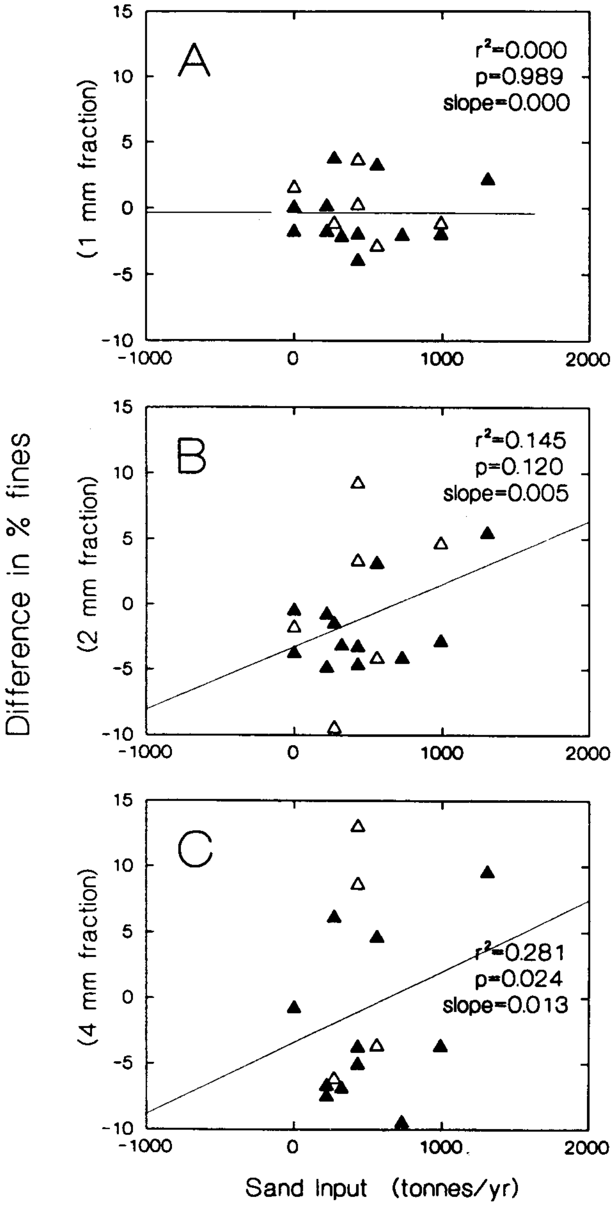

Figure 4: Upstream to downstream differences in the fines content of riffle substrate against cutbank sand input. Fractions < 1 mm (A), < 2 mm (B) and < 4 mm (C). Solid triangles represent differences within sub-aqueous sample pairs; open triangles represent differences within sub-aerial pairs. Positive ordinates denote enrichment by fines.

indicates that the positive trends are not statistically significant and are mostly due to isolated data points. Overall, paired t-tests of grain-size composition at comparable locations reveal no statistically significant downstream change in fines content due to the intervening presence of a rapidly eroding bank (Table 4, Trial A, P> 0.49 for all fine fractions). We also compared pooled upstream versus pooled downstream samples, across every eroding bank section. Again, no significant sign of enrichment was detectable (Table 4, Trial B).

CONCLUSION

The importance of planform for fish hydraulic habitat has been recognized by several authors. Mosley (1982), in his study of the braided Ohau river in New Zealand, showed that in multiple channels of variable width and depth, which characterize braided rivers, suitable hydraulic conditions for fish are maintained over a wider range of discharges than in a single-thread channel. Beschta and Platts (1986) discuss the difference between straight and meandering reaches in the availability of hydraulic habitat for fish. Stream restoration projects often call for the introduction of meanders into previously straight rivers to foster pool-riffle development and to increase hydraulic diversity, thereby improving the quality of habitat in the river (Newbury and Gaboury, 1993).

The importance of geomorphic patterns to stream habitat availability for fish is confirmed by our data. Variation in planform between reaches leads to as much as a five-fold difference in mean useable widths for fry, and three-fold differences for parr (Table 2). Conversely, in reaches with a high degree of planform similarity (4 and 5), habitat statistics and bivariate distributions are remarkably similar (Fig. 3 and Table 2). Note, however, that although reaches 3 and 4-5 all display channel splitting tendencies (multiple thalwegs, Table 1), their hydraulic habitat statistics are radically different. Clearly, the availability of hydraulic habitat depends strongly on the size of mid-channel bars and their degree of incision, rather than simply on the number of thalwegs. Note, finally, that the differences observed between reaches are highly discharge-dependent. In an extremely dry summer, reach 3 could, for example, become too shallow for parr or fry, such that the deeper reaches 1 and 2 might rank highest in suitable habitat.

Table 4: Paired *t*-test results for the effect of eroding cutbanks on the fines content of riffles. Paired upstream and downstream samples from the same lateral sample location are compared in Trial A. In Trial B, pooled upstream and downstream samples are compared. P = probability of zero enrichment effect. For the 1 mm to 16 mm fraction, positive values in column 2 (mean difference) are associated with downstream enrichment by fines. For D_{50}, enrichment is indicated by a negative value.

Trial	Size parameter	Mean difference (%)	N	SD	P
A	% < 1 mm	0.4	15	2.5	0.510
	% < 2 mm	0.9	15	5.0	0.492
	% < 4 mm	-0.2	15	8.3	0.904
	% < 16 mm	3.0	15	2.2	0.356
	D_{50}	-1.3	15	7.4	0.516
B	% < 1 mm	0.2	8	2.6	0.809
	% < 2 mm	-1.3	8	3.6	0.348
	% < 4 mm	-0.8	8	6.8	0.761
	% < 16 mm	0.0	8	13.7	0.996
	D_{50}	3.7	8	7.0	0.178

This study also tested whether, at the local scale, potential salmon spawning riffles on the Nouvelle River can be deteriorated by the river's aggressive attack on its own cutbanks. A significant overall trend of sand enrichment in riffles along laterally unstable reaches could not be detected. It may be, of coarse, that any enrichment effect on the Nouvelle River is relatively weak and hard to detect above the natural "background noise" of variation in substrate sand content in gravel river beds. However, if it is present, enrichment on the Nouvelle River is, for the most part, insufficient to deteriorate riffle substrate in terms of fines content. Overall, spawning substrate quality is good, even in the highly unstable reaches. Only 4 out of 41 samples, all located in the subareal margins, had truly deleterious composition (ie. Fredle indices < 4.5, Lotspeich and Everest, 1981; or sand index > 1.5, Peterson and Metcalfe, 1981). Despite the high lateral instability of this river, mean sand content for the riffle samples extracted near the channel thalweg was only 9%. The findings indicate that for this moderately high-energy, gravel-bed river, eroding into either floodplain or low-elevation terraces, the normal fluvial mechanisms that evacuate fines from the low flow riffle zone are sufficient to keep the grain-size composition of riffles in the optimal range for salmon spawning.

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MODELING OF INSTREAM FLOW NEEDS: THE LINK BETWEEN SEDIMENT AND AQUATIC HABITAT

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ABSTRACT

Numerical techniques have been developed to calculate instream flow needs when sedimentation management is one of the goals of an environmental flow. The techniques were used to calculate instream flow needs in the Gunnison River in western Colorado that may improve the habitat for Colorado Squawfish. The Gunnison River is a reasonably large river with a mean discharge of 73 cubic meters per second (cms) located in Southwestern Colorado, USA. The lower reach of the river is habitat for Colorado Squawfish, an endangered species. The flows of both sediment and water in the river have been modified by the construction of reservoirs and by major diversions for irrigation. The reservoirs have reduced the capacity of the river to cleanse the bed material of fines and sand with the result that there has probably been an increase in fines and sand on, and within, the stream bed.

There are three types of bed material in the lower Gunnison River: 1) material forming the protective surface of the river bed (armour), 2) material making up most of the stream bed (substrate), and 3) material on the surface of the armour at low flows (ephemeral bed material). Numerical models which consider the three types are used to obtain the flushing flow needs. The flows needed to maintain the spawning habitat for the Colorado Squawfish by removing fines and sand from the bars (riffles) are 430 cms; the flows needed during the spawning period to keep fines from settling on the cobbles in the spawning areas is 25 cms; and the flows needed to remove sand and fines from the pools used by the adult fish is 226 cms. These flushing flows are probably not required each year but they are required periodically; the maintenance flow is needed every year. An index to the capacity of the river to transport sediment (Sediment Transport Capacity Index, STCI) has been developed based on average daily streamflows. The frequency of flows available to remove sediment from the pools has been reduced from 82% of the years up to 1936, to 46% of the years between 1966 and 1995. The frequency of the flows with the capacity to clean the potential spawning bars was 57% for the years up to 1936, in comparison, the frequency was 14% in the 1968-1993 period. The average annual STCI for pool cleaning has been reduced from 122 to 28, an 82% reduction; and the STCI for bar cleaning reduced from 31.1 to 3.3, a 90% reduction.

KEY WORDS: Sedimentation / Habitat Modeling / Sediment Transport / Hydraulics / Instream Flow Needs / Gunnison River

INTRODUCTION

An improved technique for the calculation of streamflows needed to manage sediment in rivers has been developed. The improved technique has three major components; 1) a biological component that determines the objectives of sediment management in the stream, 2) a hydraulic component that determines the hydraulic conditions needed to accomplish the biological goals, and 3) a selection component linking the hydraulic and biological components to determine instream flow needs for the management of sediment.

An application of the improved technique is presented. The task was to determine the instream flows needed in the lower Gunnison River in western Colorado (USA) to improve the sediment related aspects of the habitat of Colorado Squawfish.

Biological Component

The objective of the biological component is to determine the biological needs related to the sedimentation processes in a stream. This requires knowledge of the habitat needs of the aquatic animals as related to sedimentation. The task is not a small task. Four examples of sediment related habitat considerations are below:

1) Some species of fish spawn by broadcasting their eggs which settle to the stream bottom and stick to gravel or larger particles where they stay until they hatch; if the stream bed is covered with sand or fines before spawning, or the eggs are covered after spawning, spawning will be unsuccessfully.

2) Other species of fish (for example salmon and trout) dig redds (nests) into the stream bed, deposit eggs, and cover the eggs with gravel. If the bed materials are too large, the redd can not be dug; and if fines are intruded into the stream bed, or cover the stream bed after spawning, incubation may not be successful.

3) Some species of amphibians live in voids within a cobble/boulder stream bed, if these voids are filled with sediment the species can not use the stream as habitat.

4) Mussel adapted to gravel bars may not survive if the bars are covered with sand or fines. The impact on the mussels of the sand and fines can be directly on the mussels, or on the fish important in mussel reproduction, or on both.

The objective of the biological component is to develop a 'model' of the relationship of sediment to the species of interest. In many cases the 'model' should relate to the total aquatic ecosystem, or an assemblage of aquatic animals, and not a specific species of aquatic animal. Most often the 'model' will specify the sizes of sediment that must not be deposited in, or on the stream bed, or that must be removed by flushing. The channel morphology aspects of the model will often be a need to maintain side channels that can be used as a refugia for small fish.

Hydraulic Component

The objective of the hydraulic component is to determine the hydraulics needed to remove and transport sediment through the stream channel and to maintain the channel morphology as required to meet the needs of the biological 'model'.

If the biological model specifies the size of sediment that must be removed form the stream bed, the maximum size of the sediment to be removed becomes the maximum size of the wash, suspended, or bed load depending on other aspects of the biological model. Equations are available to calculate the maximum size of the sediment load. specifics about the development of the equations are given in Milhous (1996).

The equations used to calculate the maximum size of the wash (dmaxw), suspended (dmaxs), and bed load (dmaxbl) are:

(1) $$dmaxw = R\ S_e/(0.56\ (G_s-1))$$
(2) $$dmaxs = R\ S_e/(0.28\ (G_s-1)$$
(3) $$dmaxbl = ((R\ S_e/(0.018\ (G_s-1))^{2.85})\ (d50_a{}^{-1.85})$$

where R is the hydraulic radius, S_e is the energy slope, G_s is the specific gravity of the particles, and $d50_a$ is the median size of the stream bed armour. The equation for the maximum size of the bed load should only be used when the median size of the bed load is less than the median size of the armour and the objective is to keep material moving through the stream when the armour is relatively stable. The equations used in the calculation of the median size of the bed load (d50bl) is:

(4) $$d50bl = (R\ S_e/(0.046*(G_s-1))^{2.85} * (d50_a{}^{-1.85})$$

when the median size of the bed load is less than the median size of the armour. The bed load equations are for the calculation of the sizes of the bed load during flushing of the bed material and not for general movement at higher streamflows.

In previous work, Milhous and Bradley (1986), a stream substrate movement parameter, ß, was used to determine the flushing flow needs in a stream. The equation for the substrate movement parameter is:

(5) $$ß = R\ S_e/(G_s - 1)\ D50_a$$

where R is the hydraulic radius, S_e is the energy slope, G_s is the specific gravity of the bed material (substrate), and $D50_a$ is the median size of the bed armour material.

The selection of the values of the substrate movement parameter needed to determine a flushing flow need was done using data obtained during bed load transport research in Oak Creek, Oregon. The Oak Creek data indicated the value of ß required for removal of fines and sand from the surface of a gravel bed river (surface flushing) is 0.021; and for removal of material from with in the substrate (depth flushing) 0.035. An important assumption is that the Oak Creek results could be generalized to other rivers.

The use of a surface flushing ß of 0.021 should be used if the 'biological model' does not give the size of sediment to be removed. If the sizes of sediment to be removed are known the equations for ß are:

(6) $$ß = 0.56\ (dt_w/d50_a)$$
(7) $$ß = 0.28\ (dt_s/d50_a)$$
(8) $$ß = 0.018\ (dt_{bl}/d50_a)^{0.35}$$

where dt is the target size of sizes of sediment to move specified in the biological model. The subscript w is for wash load, s for suspended load, and bl is for bed load. Note that the substrate movement parameter is <u>always</u> the dimensionless shear stress calculated using the median (d50) size of the stream bed armour.

The undesirable sediment may be moved either as wash, suspended, or bed load. The selection of the appropriate type of load to use in the analysis is based on the 'biological model'. Sometimes the biological model will indicate certain sizes should be moved in one state (i.e. wash load) and other sizes moved in another state (i.e. suspended or bed load).

Selection Component

The selection component combines the results from the maximum size analysis and the biological analysis to define the instream flows needed to manage the sediment flows through the river so as not to have a detrimental impact on the aquatic system. The design of the selection process depends on the nature of the aquatic ecosystem. In many cases there will be a critical period during which fine sediment or sand should not be deposited on the stream bed and a need to remove sand and fines deposited on the stream bed. For salmon and trout there is a needed to flush sand and fines from within the stream bed. For walleye perch and paddlefish there is need to keep fines from settling on the eggs deposited on the stream bed during incubation.

The development of criteria for the selection of instream flow needs for flushing is the critical component of the process. If the aquatic ecosystem is not understood, or the goals of aquatic ecosystem management not know, the selection component can not be designed. If the selection component can not be designed, an instream flow need can not be determined.

APPLICATION TO THE GUNNISON RIVER

The Gunnison River, a Southwestern Colorado river, is habitat for Colorado River Squawfish (*Ptychocheilus lucius*), an endangered species. The Colorado Squawfish are found in the lower reach of the river. The techniques presented in the previous section are used in this section to determine the instream flow needs for the management of sediment to benefit the Colorado Squawfish. Little is know about the habitat needs of the Colorado Squawfish from the sediment viewpoint; consequently, the estimates of the instream flow needs given in this paper will change as the habitat needs of the fish are better understood.

The Gunnison River

The Gunnison River has a mean discharge of 73 cubic meters per second (cms) and a 75 meter width just upstream (at Whitewater) of its junction with the Colorado River. The upper (eastern) part of the Gunnison basin is in the Rocky Mountain physiographic province and the rest in the Colorado Plateau province. The drainage area of the basin above Whitewater is 20,534 km^2. Downstream of Whitewater the river enters the Grand Valley before joining the Colorado River in Grand Junction. The flows of sediment and water in the river have been modified by the construction of reservoirs and by major diversions for irrigation and for municipal and industrial purposes. About 94,300 hectares are irrigated in the basin above Whitewater.

The water storage capacity in the Gunnison River basin has increased significantly between 1917 and 1993. The water storage history of the basin can be divided into three periods. The period from the beginning of discharge records (in 1896 but with missing years before 1917) until the completion of Taylor Park Reservoir (137 hm^3) in 1937, the period following construction of Taylor Park Reservoir until completion of Blue Mesa Reservoir

(1020 hm^3) in 1965 (in 1966 there was 1187 hm^3 of major reservoir capacity), and the period from 1965 until 1995 (reservoir capacity increased to 1475 hm^3).

The reservoirs in the upper Gunnison basin and the use of water for irrigation have had a significant impact on the daily streamflows in the lower river but less impact on the annual flows. The average daily discharge for two periods are given in Figure 1. The reservoirs have reduced the daily discharges during the spring runoff period and increased significantly the discharges during the summer, fall, and winter. The various uses of water have reduced annual flows by roughly 20 cms at the present time. The early 1980's were wet compared with previous periods. The bankfull discharge is about 500 cms.

The lower Gunnison River can be characterized as a cobble and gravel river, with considerable sand and fine sediment on the surface of and among the cobbles and gravel. The substrate of the lower river is gravel and cobble, with significant quantities of sand and fines both on the surface of the cobbles and boulders, and within the bed material. The median (d50) size of the armour is 63 mm, of the substrate below the armour 29 mm, and of a fine material on the surface of the armour in some locations 0.025 mm.

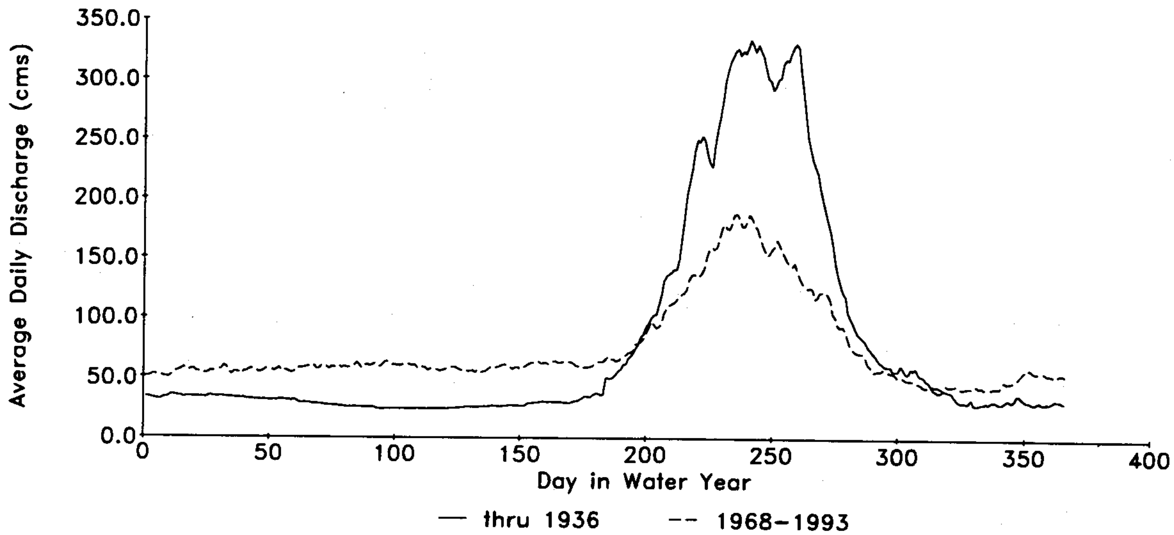

Figure 1. Average daily discharge in the Gunnison River. The period up to 1936 is prior to major reservoir construction in the basin. There was significant water use for irrigation during this period. In 1965 the major storage capacity in the basin could hold about sixty percent of the mean annual discharge of the river.

A reach in the lower Gunnison River about 61 km upstream of the junction with the Colorado River, and about 38 km upstream of the Whitewater gage, (the Dominguez Flats reach) has been studied in detail.

Sediment and Colorado Squawfish Habitat

The Colorado Squawfish is an endangered fish native to the warmwater reaches of the Colorado River. The Colorado Squawfish is a predator, the adults feed on other fish. Young fish feed on small invertebrate animals, but as they grow they become increasingly dependent on fish. (Behnke and Benson, 1980).

Colorado Squawfish spawning behavior has two phases (Tyus, Jones, and Trinca, 1987): 1) a resting phase in pools or large shoreline eddies where the fish rest and feed between spawning forays or where males gather around females until they are ready to deposit eggs; and 2) a deposition-fertilization phase on cobble bars (riffles) where the females deposit adhesive eggs on and among the cobbles and males fertilize them. Spawning occurs between late June and mid August. The small fish move to side channels and backwaters where they grow to a size that can handle the much higher velocities and, sometimes, lower temperatures in the main channel.

Based on the comments above, the sedimentation analysis must consider 1) the critical discharge required to clean the stream bed of fines and sand from the riffles used as spawning sites, 2) the discharge needed to cleanse fines and sand from the pools, 3) discharges to maintain the substrate free of the finer material during the spawning season, and 4) the discharge required to maintain a channel morphology that has side channels and backwaters. This paper considers only the first three items.

The objective of instream flows for sediment management are to remove all sand and pea gravel from the riffles, remove sand from the pools, and maintain the quality of the riffle areas by preventing fine sediment from being deposited in the riffles during the spawning periods. The sizes selected as target sizes for removal, or to prevented from being deposited, are given in Table 1.

Table 1. The sizes of sediment needed to be removed from the substrate of the Gunnison to maintain Colorado Squawfish Habitat.

Colorado Squawfish Activity	Target Sizes (mm)
Flushing of spawning sites (riffles)	4.74
Flushing of adult habitat (pools)	2
Maintenance of spawning sites	0.5

The streamflows adequate to remove the pea gravels and smaller sediments from the riffles, and to remove the sand from the pools, should transport the removed sediment as suspended sediment in order to prevent the sediment from being re-deposited on the bed surface. In order to maintain the riffle areas during spawning, the streamflows should be adequate to transport the fines and fine sand as wash load.

Bed Sediment in the Dominguez Flats Reach

The bed material is quite variable with considerable quantities of sand on, and among, the gravel and cobbles on the surface of the stream. The variation in the sizes across one diagonal bar are shown in Table 2. The large sizes on the left are in an area of fast water at all discharges. The samples on the right are from an area of low velocity, or are dry, at low discharges but with reasonably fast water during the spring runoff period.

Table 2. Median sizes (d50) of the bed material of the Gunnison River along one bar in the Dominguez Flats Reach (diagonal bar between cross section 6 on the left and cross section 7 on the right). Left side is looking upstream

	Armour (mm)	Sub-surface (mm)	Surface (mm)
Left Side	90.4	33.7	
Middle Left	70.1	31.6	
Middle Right	29.8	23.2	0.019
Right Side	29.7	15.1	0.032

The maximum typical median size of the armour in the Dominguez Flats Reach is 94 mm, this value is used in the analysis described in this paper; the diagonal bar between the cross sections tends to be smaller than other bars or riffles.

Removal of Fines and Sand From Bars (Riffles)

One of the two flushing flow objectives considered in this paper is the removal of fines, sand, and pea gravel from the parts of the river that may be used as spawning sites by the Colorado Squawfish.

Criteria for flushing of the surface of the riffles described above was to remove the material finer than pea gravel as suspended load. The equation for the substrate movement parameter (ß) is:

$$ß = \frac{R * S_e}{d50_a * (G_s - 1)} = 0.28 \frac{dt_s}{d50_a} \tag{9}$$

where R is the hydraulic radius, S_e is the energy slope, $d50_a$ is the median size of the armour on the bed surface, G_s is the specific gravity of the armour material, dt_s is the target size of the material to be removed. For the case of riffle spawning sites, dt_s is 4.74 mm which means ß is 0.015 when $d50_a$ is 94 mm. An alternative criteria is to use the value of ß required for surface flushing (ß=0.021).

The relationship between the sediment transport parameter and discharge is shown in Figure 2 for the riffle cross sections. If the surface flushing criteria is used the median relation should be used because the logic used to obtain the criteria was closer to the median than the minimum. The required discharge, based on the surface flushing flow criteria, is 480 cms. The functions in Figure 2 are based on a one-dimensional model of the river with a constant median size of the armour material of 94 mm (the maximum size on the riffles) used to calculate the movement parameter. The WSP program in the Physical Habitat Simulation System (Milhous et al, 1989) was used to simulate water surface elevations. The WSP model was calibrated to 6200 cfs (175.6 cms).

The discharge required to remove the fines, sand, and pea gravel should have a substrate movement parameter of at least 0.015 at all cross sections. The discharge that gives a ß of 0.015 at the minimum cross section is 430 cms. This value is used as the required riffle maintenance flow in this paper.

The variation in the substrate movement index along the river at a discharge of 430 cms are shown in Figure 3. Four of the cross sections have a movement parameter of 0.021. At a discharge of 430 cms all of the riffle cross sections meet the criteria establish for flushing pea gravel and smaller sediment.

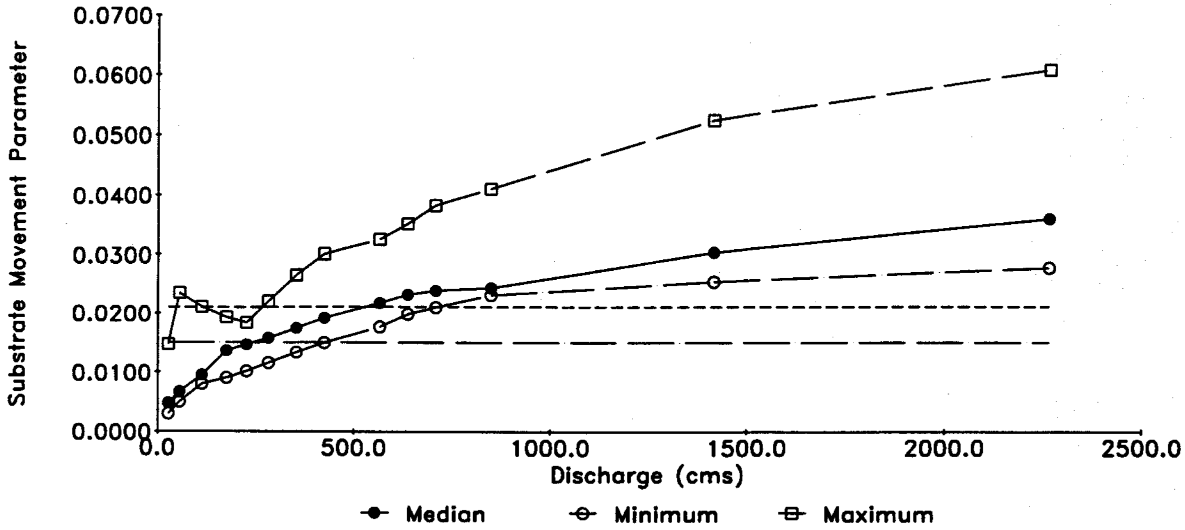

Figure 2. The substrate movement parameter as a function of discharge in the Dominguez Flats reach of the Gunnison River.

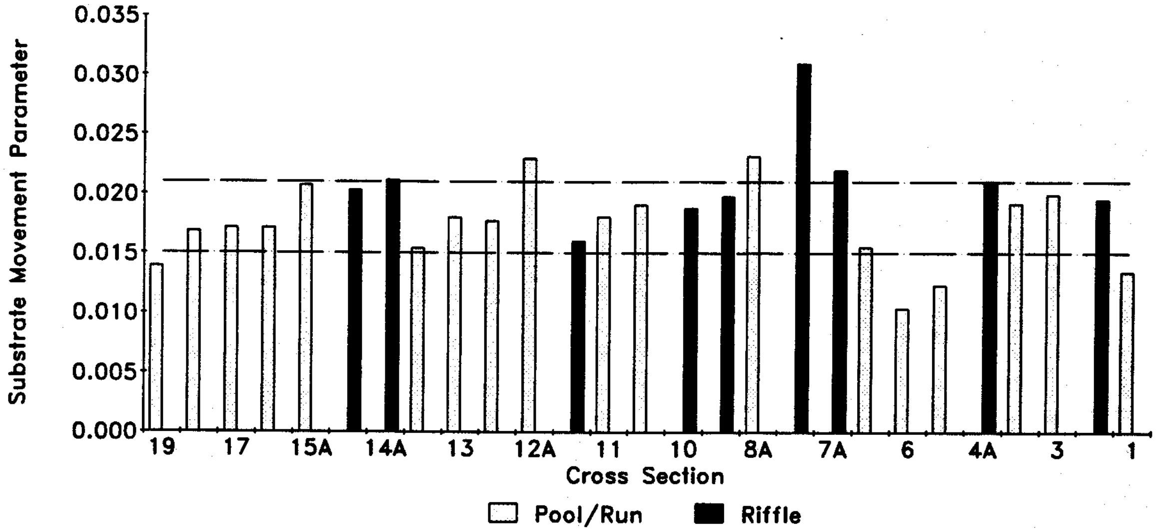

Figure 3. Variation of the substrate movement parameter at a discharge of 430 cms along the Dominguez Flats reach of the Gunnison River. The specific gravity was 2.65, the median size of the bed surface (armour) material was 94 mm.

The use of a sediment transport capacity index (Milhous, 1995) is helpful in investigating impacts of water projects and water management on the ability of a river to move sediment. The sediment transport capacity index (STCI) is calculated using the equation:

$$\text{STCI} = \Sigma(Q * (Q\text{-}QCRT)^{b})/(QREF^{b+1}) \qquad (10)$$

where Q is the daily discharge, QCRT is a critical discharge, QREF is a reference discharge, and b is the power term in the sediment concentration versus discharge equation (1.0 for Gunnison River). The summation is over a period of interest; in this paper the summation period is a water year.

The ability of the river to cleanse riffles is related to the capacity of the river to transport sediment when the discharge exceeds the critical discharge needed to clean the riffles. The critical discharge for the removal of fines and sand from the Gunnison River is 430 cms. The variation of the annual sediment capacity index over the period from water year 1917 to 1995 is shown in Figure 4.

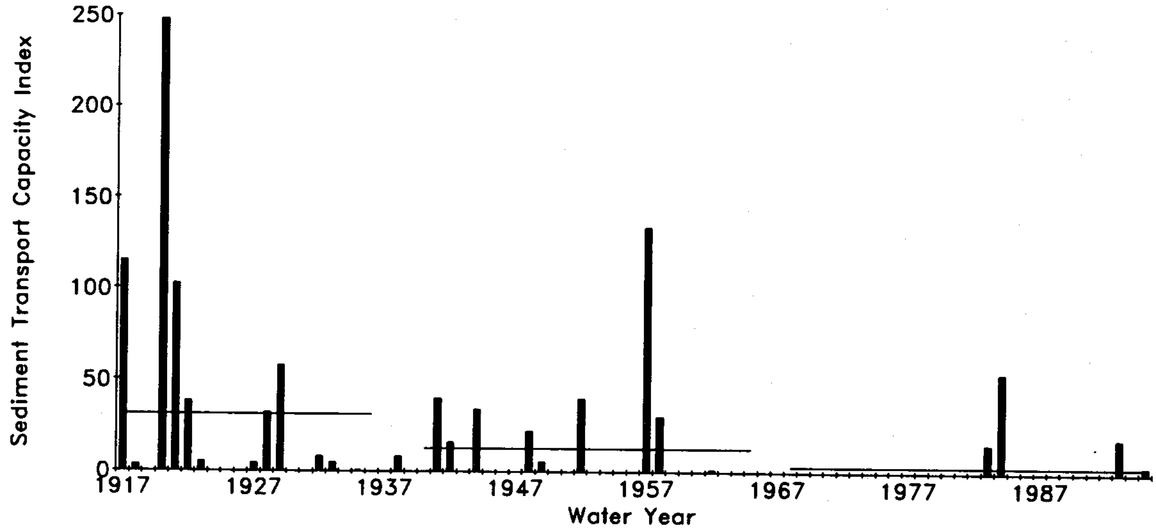

Figure 4. The annual sediment transport capacity index for removal of fines and sand from the bars (riffles) of the Dominguez Flats reach of the Gunnison River. The critical discharge was 430 cms and the reference discharge 150 cms.

Changes in the water resources system and in the management of the system have had a significant impact on the ability of the river to clean the cobble and gravel bars of sand and fine sediment. These impacts are summarized in Table 3.

Table 3. The frequency of years with sediment transport capacity (STCI) adequate to flush bar (riffle) areas needed by Colorado Squawfish for spawning. The 1897-1936 period has 28 years. Minimum discharge for flushing of bars is 430 cms.

Period	Percent of Years With Flushing Flows	Average STCI	Percent of 1897-1936 STCI
1897-99,1902-06,1917-36	57	31.1	100
1940-1965	35	12.4	40
1968-1995	14	3.3	10

Determination of Maintenance Flows

The maintenance streamflow needed is a flow adequate to keep sediment 0.5 mm or smaller moving in the river as wash load at the locations (in this case riffles) which can be used as spawning. The discharge needed to transport 0.5 mm sediment as wash load at the nine riffle sections is shown in Figure 5. The required maintenance flow during the spawning season is 25 cms. The maintenance streamflow selected is the second highest discharge because field observations suggest cross section 2 is poor spawning habitat.

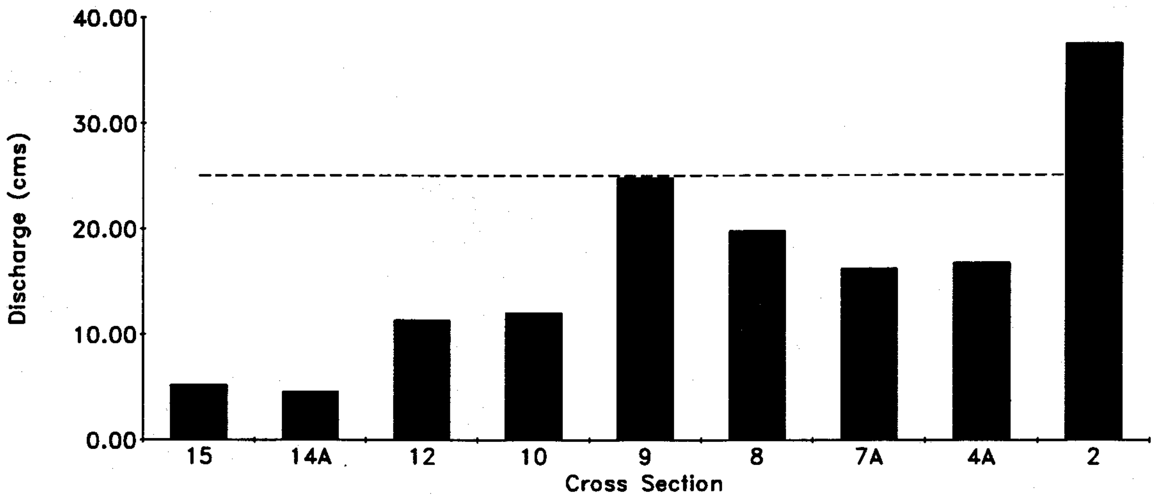

Figure 5. The discharge needed to keep sediment 0.5 mm or smaller moving as wash load at the riffle cross sections.

Removal of Fines and Sand From Pools

The calculation of a critical discharge for removal of sand and fines used the maximum wash load size analysis presented in Milhous (1996). The critical discharge required to remove sizes up to 2 mm was found to be 226 cms. The sediment transport capacity index was calculated using a critical discharge of 226 cms. The change in the frequency and the magnitude of the sediment capacity to clean pools is shown in Table 4.

Table 4. The frequency of years with sediment transport capacity (STCI) adequate to flush pool areas needed by Colorado Squawfish. The 1897-1936 period is the 28 years of the record. Minimum discharge for flushing of pools is 226 cms.

Period	Percent of Years With Flushing Flows	Average STCI	Percent of 1897-1936 STCI
1897-99,1902-06,1917-36	82	122	100
1940-1965	69	63	56
1968-1995	46	28	25

DISCUSSION

The conclusions of this paper are that the discharge needed to maintain pools in the condition needed by Colorado Squawfish is 226 cms, to maintain bars in the state needed by the Colorado Squawfish for spawning is 430 cms, and to maintain the spawning habitat during the spawning period is 25 cms. The frequency of the flows with the capacity to clean the potential spawning bars was 57% for the years up to 1936, in comparison to 14% in the 1968-1995 period. The frequency of flows available to remove sediment from the pools has been reduced from 82% of the years up to 1936 to 46% of the years between 1966 and 1995. The sediment transport capacity index for pool cleaning has been reduced from 122 to 28, an 82% reduction; and the STCI for bar cleaning reduced from 31.1 to 3.3, a 90% reduction.

In 1993 the sediment transport capacity was adequate to remove fines and sand from both pools and riffles. (The STCI for bar flushing was 18, and for pool flushing 115) This runoff event did cause flushing of the pools as is shown in Figure 6.

There was also a change in at least one bar. Measured bulk density in February 1993 was 2.61 kg/m^3 which means few voids existed. The measured bulk density after the 1993 runoff event was 2.08 kg/m^3. The sediment transport capacity needed to clean the bars was zero the eight years previous to 1993.

The observed scour of the thalweg and the change in bulk density suggest the conclusions of the paper are at least reasonable. There are two problems that should temper these conclusions. The first is that the spawning needs of the Colorado Squawfish are poorly understood. The second is that a one-dimensional analysis was used. The spawning needs of the fish should be better understood and a two-dimensional analysis technique used that relates the specific locations in the stream that have the bed conditions needed for spawning with the two-dimensional velocity structure before definitive conclusions can be made. The conclusions from this paper can be used on an interim basis.

ACKNOWLEDGMENTS

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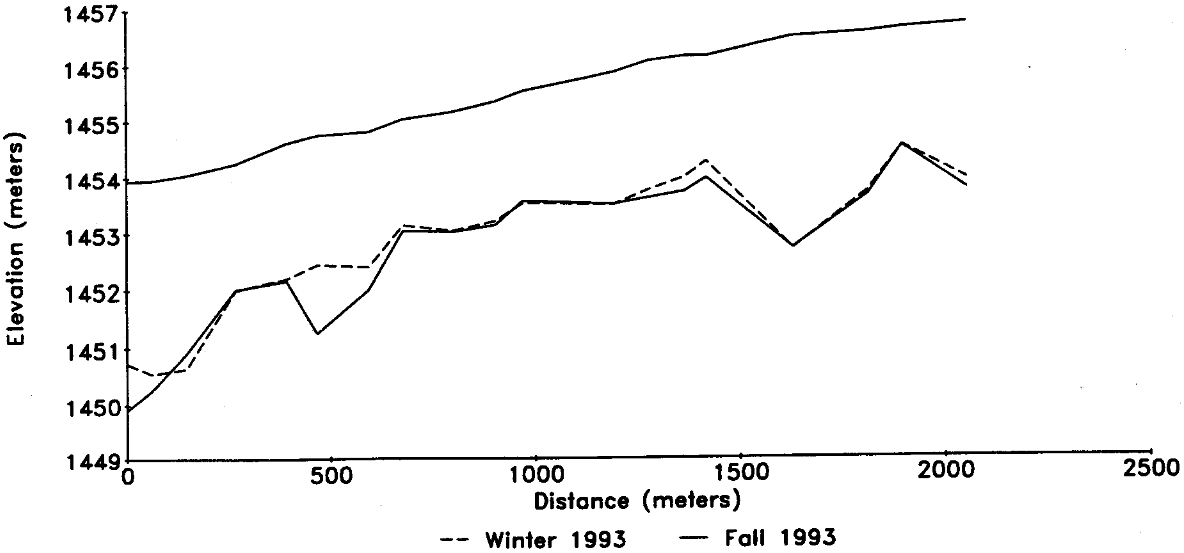

Figure 6. Change in the thalweg of the Dominguez Flats reach of the Gunnison River caused by the 1993 spring runoff. The water surface (line at the top) is for a discharge of 171 cms.

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MODELLING THE RESPONSE OF RIVER GEOMORPHOLOGY AND RIVERINE VEGETATION TO WATER RESOURCES DEVELOPMENT IN THE SABIE RIVER, SOUTH AFRICA

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ABSTRACT

The major rivers of the Kruger National Park all rise beyond the park's boundaries and their ecosystems are vulnerable to developments in the upper catchments. The Sabie River in particular is threatened by new dam construction. Estimation of ecological flow requirements and prediction of ecological response to changes in the flow and sediment regimes require prior knowledge of the morphological characteristics of the river, which also change with changing water and sediment input. Predictions of river response to development therefore require process-based morphological modelling. The Sabie River has a complex mixed alluvial-bedrock morphology which cannot be described in terms of conventional classification systems, and a new hiearchical system has been developed to enable description for modelling purposes. A number of approaches have been followed for predicting morphological changes, the most promising being a rule-based model which is quantitative and time-dependent and can incorporate processes not easily described by differential equations, as well as information from expert opinion. Studies of instream vegetation indicate significant interaction between reed growth and sediment dynamics which can easily be included in a rule-based modelling structure. A rule-based model has been developed for the Sabie River. The model predicts sediment movement and reed growth using discrete-state variables for monthly flows and reed condition. Sediment routing under supply-limited conditions, reed growth and sediment trapping by reeds are modelled using simple rules. The model has been applied to three flow scenarios to illustrate the impact of catchment development, dam construction and vegetation-sediment interaction on morphological change, and also the potential of this kind of model..

KEY-WORDS: Fluvial Geomorphology / Modelling / Vegetation / Sabie River / Sediment / Dam Impact

INTRODUCTION

The Kruger National Park (KNP), located in the Mpumalanga province on the north-eastern border of the Republic of South Africa with Mozambique (Fig. 1), is an important nature conservation area. The seven major rivers flowing through the Park all rise beyond its boundary to the west, and the extensive areas of their catchments outside the Park have been developed to different degrees for agriculture, industry, forestry, subsistence farming and urban settlement. The associated increase in water demand has reduced the flow in the rivers and is threatening the ecological viability of the riverine habitats within the Park.

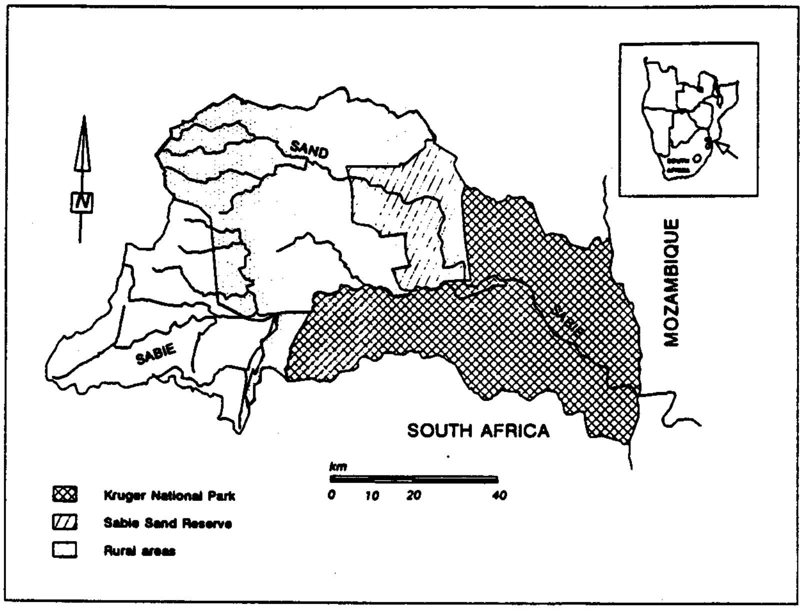

Figure 1: The Sabie River in the Kruger National Park

The Kruger National Park Rivers Research Programme was initiated in 1988 to develop ways of predicting the impact on KNP river systems of changing flow regimes and water quality, and to develop water management expertise to minimize this impact. Attention has been focused primarily on the Sabie River in view of its relatively pristine condition and the immediacy of threats associated with new dam construction.

Estimates of ecological flow requirements are based on the habitat requirements of riverine biota. The habitat is defined by the nature of the substrate and local hydraulic conditions, which are determined by the physical form (morphology) of the river channel and the occurrence of water within it. The anticipated development upstream of the Park will result in significant changes in the flow and sediment supply regimes, with consequent adjustments to the river morphology. The riverine vegetation, which also provides important habitat components for a variety of animals, responds to changes in morphology. Predictions of habitat changes can therefore not be made without prior predictions of morphological change. Instream plants also influence the morphological adjustment of the river by enhancing sediment deposition and stabilizing deposits, so the morphological and vegetation responses must be considered together to account for their mutual feedback interaction.

Various approaches are being followed to develop the capability of predicting the morphological and associated vegetation responses to natural and anthropogenic changes in water and sediment supply to the Sabie River. This capability will enable ecological requirements to be included effectively in future planning and management of upstream water resources. The nature of the river and the resolution of both the predictions required and the data available make conventional modelling techniques unsuitable, and a novel rule-based approach is being developed.

THE SABIE RIVER

The Sabie River rises on the Great Escarpment 104 km upstream from its entry into KNP and flows a further 110 km through the Park to the Mozambique border, at which point it commands a catchment area of 6250 km^2 (Fig. 1). Mean annual rainfall in the catchment varies from about 2000 mm in the mountainous upstream portion to about 400 mm near the Mozambique border. The flow in the river is seasonal and highly variable, with low winter base flows of 1 to 2 m^3/s and occasional high summer flows greater than 100 m^3/s. During normal years several summer mid-range flows of about 20 to 40 m^3/s occur. During drought periods summer and winter base flows are lower and summer high and intermediate flows are less frequent (Moon *et al.*, in press).

From its source to the Mozambique border, the Sabie River occupies four distinct geomorphological zones, differentiated by major breaks in slope associated with variations in the underlying geology (van Niekerk *et al.*, 1995). In the Lowveld Zone (including the KNP) the river is incised as a macro-channel containing sediment deposits, which form one or more active channels, as well as the instream and riparian vegetation communities.

The river has developed on a template of highly variable geology, producing numerous bedrock controls and local changes in gradient. Sediment deposition upstream of bedrock outcrops has led to some extensive alluvisl reaches. Elsewhere, the river channel is formed predominantly in bedrock. The sediment in the river is supplied mainly from sand-bed tributaries during relatively infrequent events, and is reworked by more frequent (e.g. seasonal) high flows. Under favourable conditions, sediment deposits become colonised by vegetation (e.g. *Phragmites mauritianus*) and become stable, relatively permanent features. One tree species (*Breonadia salicina*) becomes established in bedrock fissures and subsequently promotes sediment deposition. The Sabie River therefore displays characteristics of both bedrock and alluvial channels, both influenced strongly by vegetation, and the morphology varies considerably on different spatial and temporal scales.

The complex mixed alluvial-bedrock nature of the Sabie River in the Lowveld Zone cannot be adequately described in terms of existing classification systems, and a new hierarchical classification has been developed (van Niekerk *et al.*, 1995). To account for a range of scales, the river is described in terms of geomorphological characteristics with different magnitudes, using a bottom-up approach (Fig.2). At the smallest scale are *geomorphological units* (measured in tens to hundreds of metres), each with distinctive physical characteristics, including features such as bars, riffles and cataracts. The geomorphological units combine to form a variety of *channel types*, characterising the channel at the scale of tens of metres to kilometres. Channel *reaches* are classified on the basis of functional relationships between the channel types and are characterized by planform over hundreds of metres to tens of kilometres; 22 reaches have been identified in the Lowveld Zone of the Sabie River. A *macro-reach* consists of one or more reaches grouped on the basis of common geology, hydrology, macro-channel gradient and width, and vegetation; 9 macro-reaches have been identified in the Lowveld Zone. *Zones* are defined in terms of relief and geology; the Sabie River occupies the Lowveld and three other zones.

PREDICTION OF CHANGE

Morphological Change Models

Aerial photographs taken at intervals over the last 50 years show significant changes in some reaches of the Sabie River, largely due to the redistribution of sediment. During some periods the volume of sediment in the river increased, and during others the volume in certain reaches decreased. Such morphological changes would affect habitat characteristics.

The observed changes reflect response to natural climatic variations as well as to development in the upstream catchment. Future development is expected to reduce the river flow further and to increase the sediment supply through erosion of degraded land; construction of a new dam on one of the Sabie River's major tributaries has recently begun. Prediction of the response of the river to future developments requires the development of models

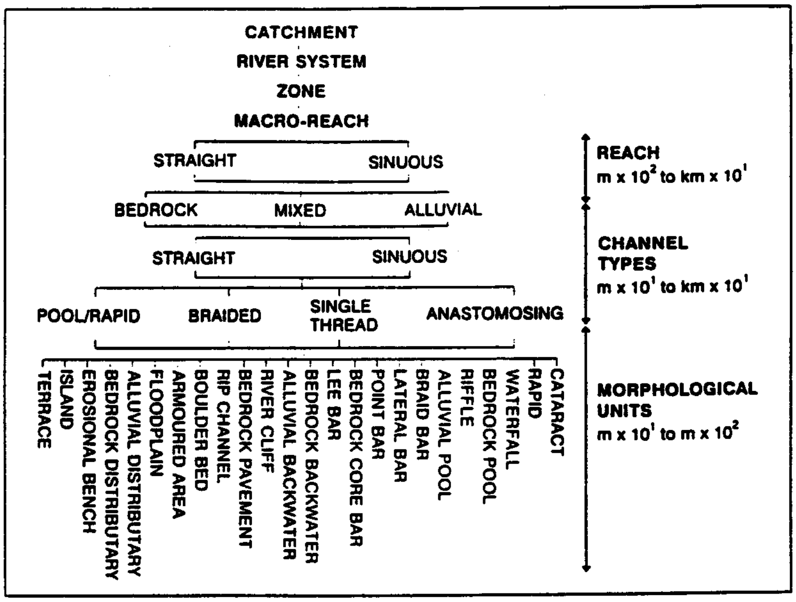

Figure 2: Hierarchical classification system for mixed bedrock alluvial rivers

based on understanding of the channel-forming processes. Several modelling approaches have been developed and applied to the river within the KNP boundaries.

A conceptual model of the progression between channel types, defined in accordance with the classification system shown in Fig. 2, was developed by Heritage and van Niekerk (1994) and is presented in Fig. 3. The factors influencing change were identified as the flow magnitude (QMAG) and variability (QVAR), the supply of sediment (SEDI), and the channel competence (COMP). The relative magnitudes of these variables determine the nature and direction of change between channel types. This model is entirely qualitative and cannot predict the amount of change required in any one of the factors to cause a shift from one channel type to another, or the time required for such a shift to take place.

Locations likely to experience change can be identified by computing the variation of sediment transport capacity along the river: where transport capacity is relatively high there will be a tendency for erosion and where it is relatively low there will be a tendency for sediment accumulation. Sediment transport capacities were computed along the Sabie River within the KNP for different discharges using steady gradually-varied flow analyses and the total load equation of Ackers and White (1973). The predicted variation of transport capacity correlates reasonably well with measured long-term sediment accumulations. This model is static, however, and cannot provide important information about time-dependent responses.

A sediment mass balance model was developed to account for changes over time. It is based on total load calculations (according to the Ackers and White (1973) equation) using daily flow data and representative cross sections for the different channel types (Moon *et al.*, in press). Predicted locations of erosion and deposition compare reasonably well with aerial photographic evidence if sediment injections from tributaries are accounted for. The major shortcomings of this approach are its assumptions that a single cross section representats all reaches within a given channel type, and that sediment transport is always at the capacity rate. Improving these assumptions adequately within this model structure would require impractically extensive data input.

Conventional sediment routing using the Saint-Venant equations together with a sediment continuity equation is not appropriate for the Sabie River considering the complexity and variability of the cross-sections and the heterogeneity of the channel types. The high resolution of data and the high grid resolution required in the numerical solution would be unrealistic and incompatible with the resolution required of the results. The structure of this type of model requires availability of sediment for transport at all sections, which is not always the case in bedrock-controlled channels where sediment supply rather than transport capacity may determine the actual transport rate. The interaction between sediment distribution and vegetation growth is also not easy to incorporate in partial differential equation based models.

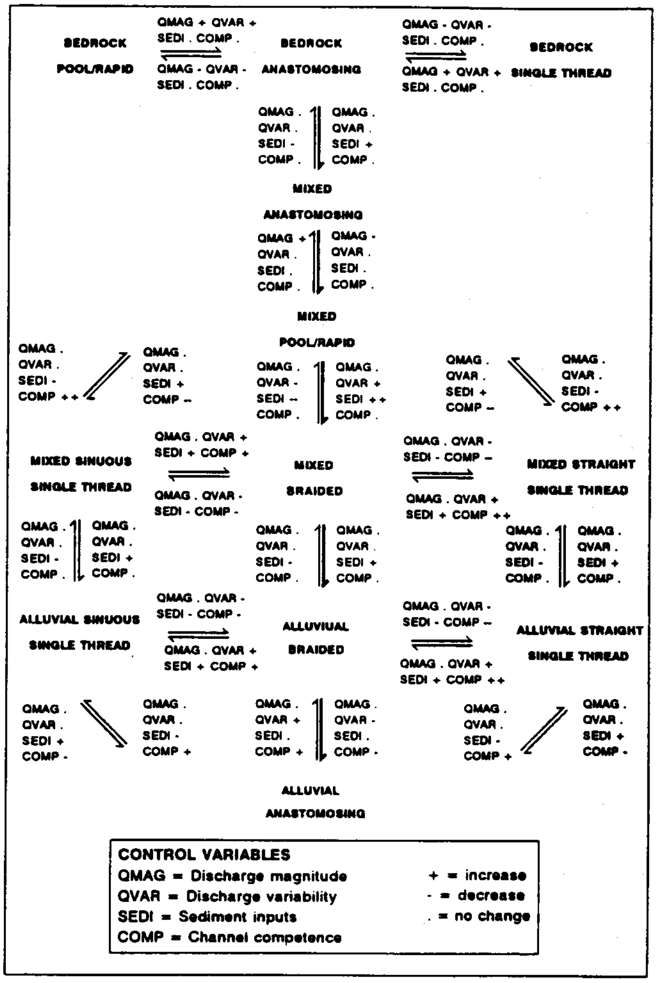

Figure 3: Conceptual model of channel change for the Sabie River

Vegetation Change Models

Carter and Rogers (1989) have described the occurrence and growth dynamics of *Phragmites mauritianus* reeds in the Sabie River. They define the six river-bed states of water, sand, rock, reeds, other herbacious plants and woody riparian vegetation, and use a Markov model to describe the state replacement sequences. They show that the sequence of water-sand-reeds-woody vegetation is becoming increasingly probable and prominent. The Markov model is purely descriptive and based on historical observations, and cannot account for the different types and rates of catchment development in the future. It also does not account for the feedback interaction involved in the establishment of reeds on sand and the stabilizing and trapping effect of sediment by the reeds.

Van Coller (1993) showed that the occurrence and distribution of riparian tree species in the Sabie River are closely related to physical characteristics of the local environment. In particular, the abundance levels of different species were shown to peak at different elevations above the river bed level, corresponding to different flow stages. Van Coller *et al.* (1995) examined the distribution of riparian tree species in relation to the geomorphic units defined by the classification system described above and the occurrence of flooding. They concluded that fluvial geomorphology and hydrology play an interactive role in influencing species distribution patterns. Prediction of the response of riparian vegetation to upstream water resources management therefore requires prior prediction of the geomorphic response.

Rule-based morphological and vegetation modelling

A model for predicting changes in the Sabie River at an ecologically relevant resolution must be quantitative, time-dependent and realistic, but also implementable in terms of data input. It must be able to route sediment under supply- limited conditions, account for vegetation interaction and incorporate other expert knowledge. The vegetation change models described above suggest that reed growth is linked to sediment deposition and erosion and is therefore a part of the channel-forming dynamics, while woody riparian vegetation reponds to morphological characteristics without influencing their development. The modelling approach should therefore be able to account for the feedback interaction with reeds in a coupled way, while the woody vegetation response can be treated as uncoupled and inferred from long term morphological predictions.

A rule-based modelling approach (Starfield, 1990; Nicolson and James, 1995) has been developed to meet these requirements. The movement of sediment and the response of reed growth is simulated in a series of time steps during which the flow is defined in discrete categories. The sediment and reed responses to a series of flows are described by a set of rules, which can be derived from rigorous analysis, empirical results, or expert opinion.

Application of these concepts requires representation of the actual river system (positions and lengths of the different channel types), quantification of the inputs (the flow and sediment input regimes), and quantification of the processes involved (sediment transport capacity and velocity, reed growth and sediment interaction).

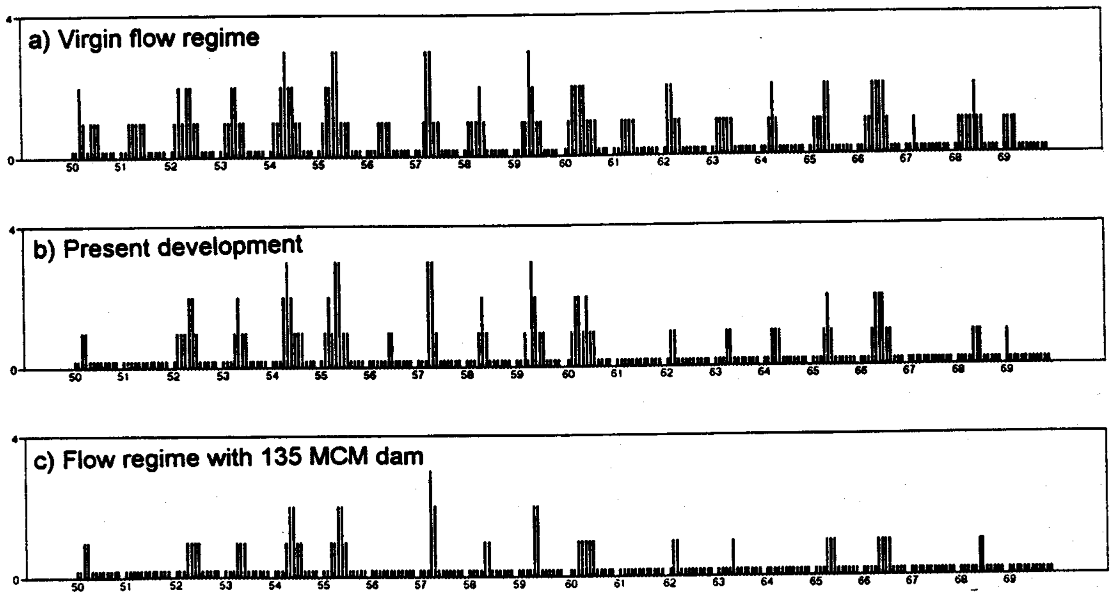

Figure 4: Flow scenarios for Sabie River modelling

A model has been constructed for a 27 km reach of the Sabie River. This reach includes several different channel types (bedrock anastomosing, alluvial braided and pool-rapid) and several small tributaries, and is bounded by two major tributaries (the Saringwa at the upstream end and the Nwaswitshaka at the downstream end).

The flow regime is represented by 20-year simulated records of monthly flow volumes representing three scenarios: virgin conditions in the catchment, present-day development and present-day development with a 135 million cubic metre dam upstream of the study reach. Flows are specified in the discrete categories of low, average, high and flood. The simulated records are presented in Fig. 4.

Sediment is supplied to the Sabie River by several ephemeral tributaries and it is likely that these feed most of the sediment carried by the main channel in major pulses when they flow. The sediment inputs from the tributaries have been specified as shown in Table 1, based on an esitmate of the sediment production rate for the region of 300 tons/km^2/year (van Niekerk and Heritage, 1994) and assumptions regarding the relative contributions of different tributaries and their temporal distributions. These values are assumed to be repeated identically every year, as there is insufficient information available for a more realistic representation.

Table 1: Sediment inputs from tributaries

Month	Saringwa (major tributary)	All other minor tributaries
November	140	0
December	140	20
January	140	0
February	0	0
March	0	0
All other months	0	0

Tributary sediment input varies between drought, average and wet years (Heritage and van Niekerk, 1995), and even within two years of the same type there may be differences in the timing of inputs. The first flow of a year could flush most available sediment into the main channel, leaving little for mobilization by subsequent flows. These variations could be described by simple rules,. such as the following.

If (trib_flow=high) and (month=jan) and (flow=first) then Saringwa_Trib_Input = 100 tons

At present there are insufficient data to quantify rules of this type and the simulations have been carried out with sediment inputs made according to Table 1.

Sediment transport capacity depends on discharge and channel type, and has been specified as shown in Table 2.

Table 2: Transport capacity related to channel type and discharge (tons per time step)

Discharge	Bedrock Anastomosing	Braided	Pool-Rapid
Low	15	4	4
Average	160	80	80
High	160	80	80
Flood	200	150	150

The rule for sediment routing is very simple: the river is divided into a series of connected cells and at each time step sediment is transferred from each cell to the cell(s) downstream, subject to the local transport capacity being able to move the available sediment. Longitudinal dispersion is simulated by allocating different proportions of the sediment leaving cell i to cells i+1, i+2, i+..., etc.. The transport velocity and the rate of diffusion can be controlled by adjusting the destination proportions. For example, if sediment is to be transferred from one cell to the three contiguous cells immediately downstream, allocating the proportions for cells i+1, i+2 and i+3 as 25%, 65% and 10% respectively would move a pulse of sediment faster than if the proportions were 65%, 25% and 10%. This formulation structure can easily allow for different transport velocities in different reaches, and at different discharges. The following pseudo-code summarizes the in-channel routing process:

```
Do for each time step
        Do for each cell, i
                Determine the amount of sediment available for transport (avail)
                Adjust the amount of sediment in storage (depending on avail)
                Remove (avail) from reach i
                Add (0.6 x avail) to reach i+1
                Add (0.3 x avail) to reach i+2
                Add (0.1 x avail) to reach i+3
        Next cell
Next time step
```

The sediment velocity can also be controlled by adjusting the model time step. For example, if sediment is transferred in the specified proportions twice a month the movement of a sediment pulse will be twice as fast as if only one transfer per month is specified. In this model the effect of channel type is accounted for by varying the proportions of sediment allocated to downstream cells (Table 3) and the effect of discharge is accounted for by varying the number of model time steps within each month (Table 4). There are insufficient data at present to estimate the parameter values accurately and those selected are based largely on judgement.

Table 3: Sediment velocity related to channel type

Channel Type	% moved to (i+1)	% moved to (i+2)	% moved to (i+3)
Bedrock anastomosing	10	25	65
Braided	80	20	0
Pool-rapid	80	20	0

Table 4: Sediment velocity related to discharge

Monthly Flow Level	Monthly Flow Volume (Million cubic metres)	Model Time Steps Per Month
Low	0-25	1
Average	26-110	2
High	111-300	4
Flood	>301	6

The sediment available for transport is determined by dividing the total amount of sediment in each cell into amounts that are mobile and in storage. If all stored sediment in potentially mobile, three possibilities exist:

1. Transport capacity exceeds (stored + moving) sediment.
2. Transport capacity exceeds moving sediment but is less than (stored + moving) sediment.
3. Moving sediment exceeds the transport capacity.

In the first case all of the (stored + moving) sediment is available to be moved downstream. In the second and third cases the available sediment is equal to the transport capacity and the difference between (stored + moving) and (capacity) is allocated to storage.

The vegetation model of Carter and Rogers (1989) suggests that reed growth is an integral process in the channel-forming dynamics and should therefore be incorporated in the sediment model. This requires modelling firstly the growth and decay of the reed beds, and secondly the interaction between reed density and the sediment dynamics.

To simulate reed growth and destruction, the condition of the reed beds is described in terms of three qualitative states: seedlings, low-density young reeds, and high-density mature reeds. Each state is characterized by a reed index with values in the range 0-1.0 for seedlings, 1.0-2.0 for low-density young reeds and 2.0-4.0 for high-density mature reeds. Depending on the flow conditions the reeds may increase in density, die back as a result of waterlogging, or be ripped out by large floods. This behaviour can be expressed by rules such as the following.

```
If (reeds=seedlings) and (flow=high) then Reed_Index = 0
If (reeds=low density) and (flow=average) then increase Reed_Index by 0.1
```

The rule details used for these simulations are specified in Table 5. The numbers in the table are reed index increments, depending on the flow category.

Table 5: Reed-index adjustment for discharge

Reed State	Low Flow	Average Flow	High Flow	Flood Flow
Seedlings	+0.02	+0.1	Reset to 0	Reset to 0
Young	+0.02	+0.1	Reset to 0.5	Reset to 0
Mature	+0.02	+0.1	+0.1 or -0.3*	Reset to 0

* If the flow is *high* for two consecutive months the index is decreased by 0.3, but if the previous month was *low* or *average* the index is increased by 0.1

The interaction between reeds and sediment dynamics can be expressed in terms of trapping and stabilizing effects. Under average and high flow conditions the reeds trap sediment moving through the reach; this is represented by allowing a small proportion of the sediment moving through a reach to be trapped and added to a store of stabilized sediment. Once sediment is stabilized by reeds it cannot be remobilized until the reeds are ripped out by a flood. The rules describing this behaviour have the following structure.

```
If (Reed_Index > 1.0) and (flow=average) then reedstore=reedstore+0.02 x movingsed
If (Reed_Index > 1.0) and (flow=high) then reedstore=reedstore+0.05 x movingsed
```

The movement of sediment through the whole reach was simulated for the three flow scenarios shown in Fig. 4. The simulations were performed with and without the influence of vegetation, to illustrate the effect that vegetation might have. The variation of sediment occurring within each major defined sub-reach through each flow scenario is shown without the influence of vegetation in Fig. 5. The total amount of sediment in each reach is not significantly different for the virgin and present-day development scenarios, although the variation with time has changed,. The dam on the river results in major changes in the pool-rapid and braided alluvial reaches, but not in the bedrock anastomosing reach, with significantly greater volumes of sediment in the affected reaches.

The response and influence of vegetation is shown for the alluvial braided reach in Fig. 6. Figure 6b shows the growth and destruction of the reed beds for the virgin condition scenario, and Fig. 6c shows the influence this has on the sediment accumulation.

The structure of riparian woody vegetation communities is not described in this model, although it depends

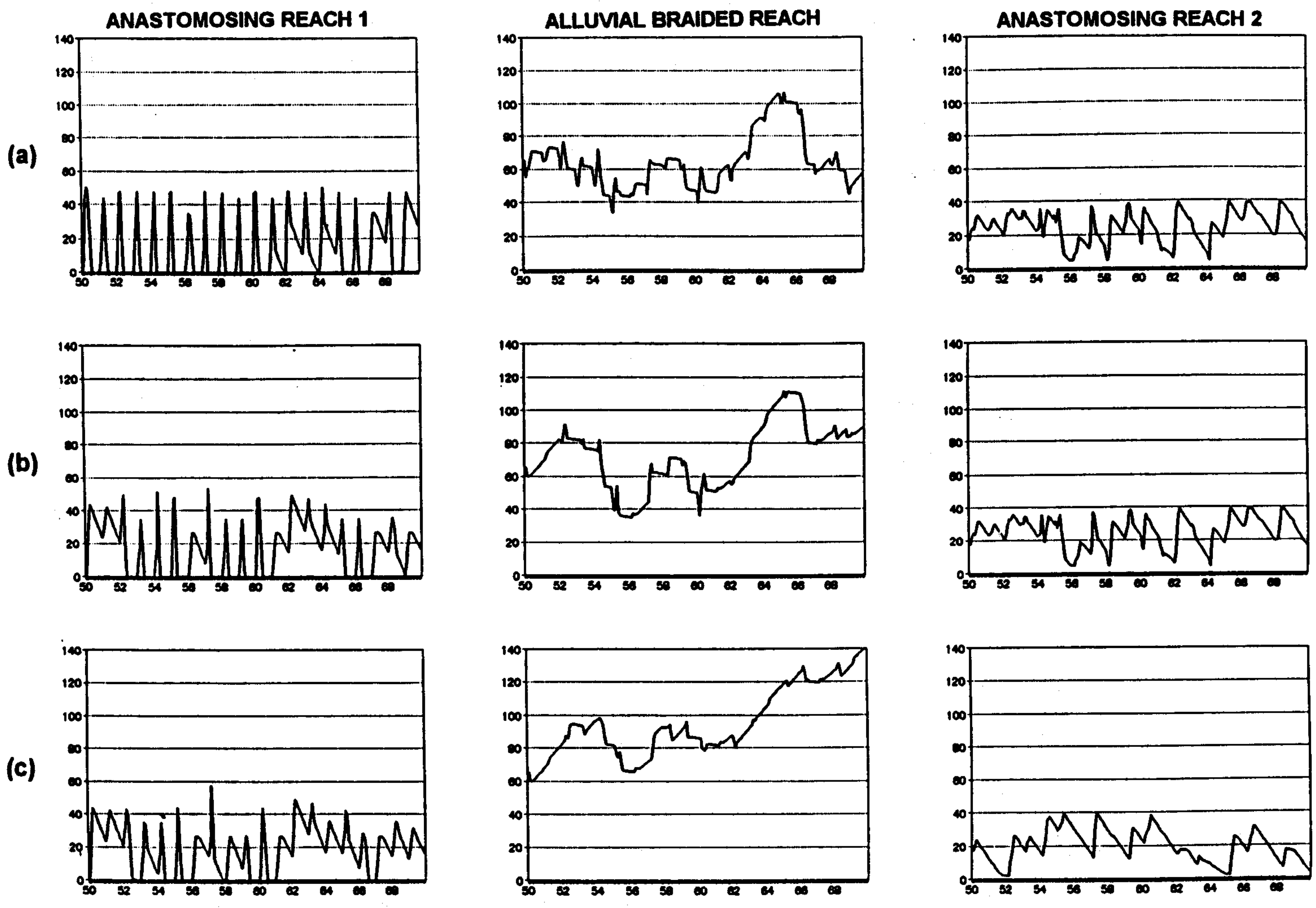

Figure 5: Sedimentation of three consecutive reaches for the period 1950 to 1969
(a) Virgin flow conditions (b) Present development (c) 135 MCM dam

significantly on the morphology and should respond to morphological adjustments. It is excluded because there is no significant feedback interaction with the sediment dynamics over the time scale considered. Adjustments may therefore be considered independently and inferred from the morphological predictions made by the model.

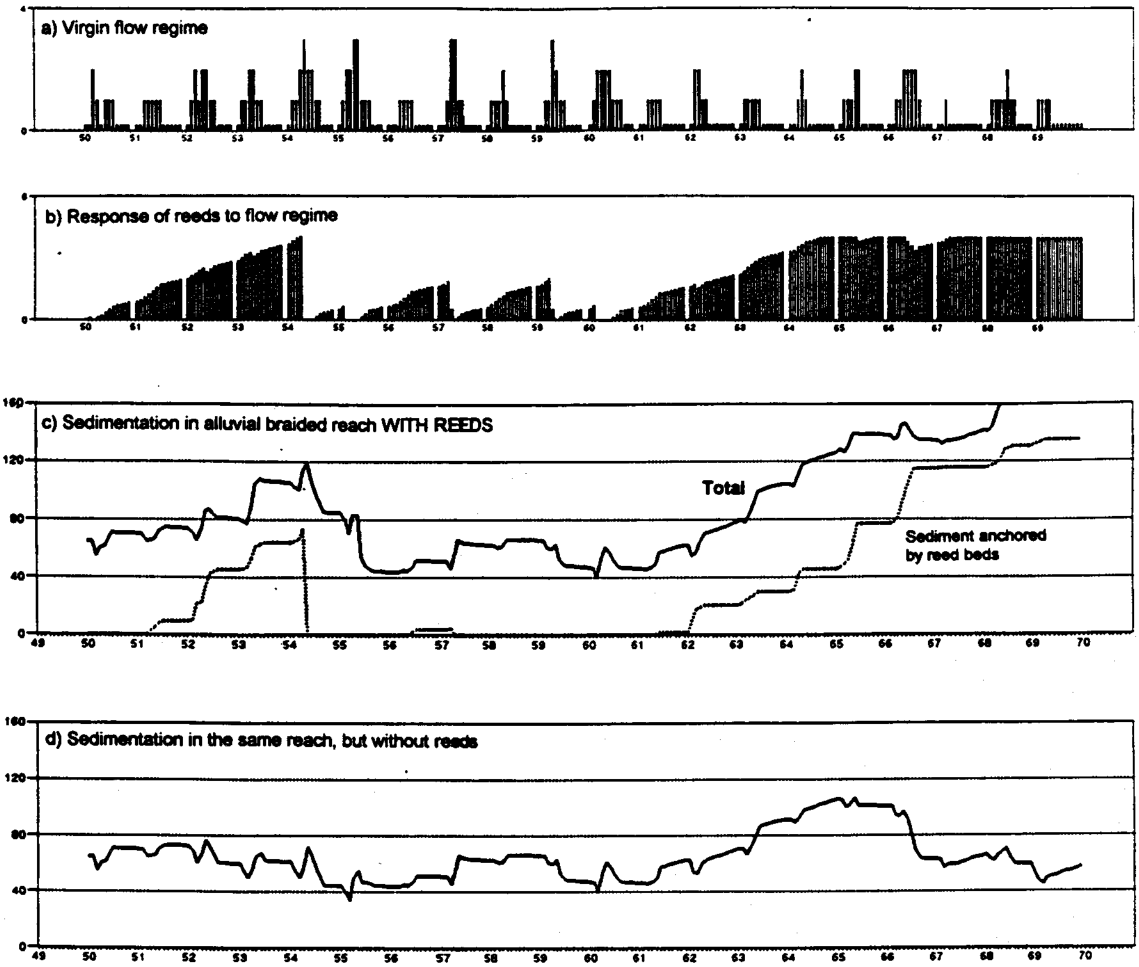

Figure 6: The effect of reeds on sedimentation in an alluvial braided reach

DISCUSSION

The simulated results presented are not intended to be accurate predictions of the behaviour of the Sabie River - the relevant processes are not yet sufficiently well understood for that. The model does, however, assist in developing an understanding of the processes underlying the system behaviour. The results suggest that the impact of dam construction will be significant in some of the geomorphic types occurring in the river, but not in others. It is also apparent that the effects of the interaction between vegetation and sediment dynamics are significant for both the vegetation growth and the channel morphology, and requires further elucidation. The complexity of the river response as governed by the very simple rules formulated challenges the reliability of inferring process dynamics from limited field information in natural systems.

The model provides a structure that will be able to accommodate better information in the future. The rule-based approach has certain advantages over conventional partial differential equation type models. It enables the interaction between vegetation growth, river flow and sediment dynamics to be included, and does this by incorporating expert opinion, rather than requiring additional differential equations. It allows lumping of parameters and simulation time steps at resolutions which are ecologically meaningful and consistent with the data available. The rule structure enables easy expansion to include other ecological responses, such as animal populations to habitat characteristics, and management actions.

ACKNOWLEDGEMENTS

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MECHANICS OF SEDIMENT TRANSPORT FOR ESTIMATING HABITABLE CONDITION OF BIVALVES IN BEACH

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ABSTRACT

Thousands of dead bivalves washed up beaches are sometimes observed after a storm, in Japan. Bivalves are important not only as the marine products but also as an index of the quality of the coastal environment. Hence, the field observations of the distribution of bivalves in a coastal zone and the laboratory experiments on the characteristics of the behavior of bivalves have been performed.

Yamashita and Matsuoka (1994) found that, in a laboratory experiment on the burrowing process of bivalves under eroding condition due to the oscillatory flow, even when the descending velocity of sand surface is sufficiently less than the burrowing velocity of bivalves, some bivalves cannot stay in a sand layer and are picked-up by oscillatory flow.

In this study, the physical background of this results are considered from a viewpoint of sediment transport mechanics. Two important aspects are investigated: the stochastic aspect of the burrowing process of bivalves and the reverse grading phenomenon observed in a motion of mixed-size grains.

KEY-WORDS: Sediment transport / Bivalve / Stochastic model / Sheetflow sediment transport / Granular material model / Distinct element method / Reverse grading

INTRODUCTION

In the coastal region around Japan, thousands of dead bivalves are sometimes washed up beach after a storm. Many field observations and laboratory experiments have been performed (Watanabe, 1982; Higano and Yasunaga, 1988; Yamashita and Matsuoka, 1994; and Kuwahara and Higano, 1994), because bivalves are important as marine products. The importance of bivalves is not limited in a fishery. Bivalves or other creatures living in sandy beach are also important as the index of coastal environment.

Moving process of bivalves has a probabilistic aspect because of the irregular factors; fluctuation of bottom velocity, descending velocity of sand surface due to scoring, size of bivalves, activity, or moving ability, of bivalves and so on. Therefore the moving process of bivalves should be treated as a stochastic process.

The other important aspect is mode of sediment motion around bivalves. In general, sediment is transported in various modes; bed load, suspended load above sand ripples and sheetflow. Especially, during storm, or under the action of very high tractive force, sheetflow should be a dominant sediment transport mode. In the sediment transport in sheetflow regime, the particle/particle interaction, or the momentum transport due to interparticle collision is a dominant mechanism governing the structure of the flow. To understand the behavior of bivalves in a sediment layer moving in sheetflow is an important subject to clarify the mechanism of bivalves picked-out of a sand layer.

Numerical calculations are performed in this paper from the two points of view mentioned above: the stochastic model for simulating the probabilistic characteristics of environment around bivalves and the granular material model for simulation the sediment/sediment and sediment/bivalves interactions.

YAMASHITA & MATSUOKA'S EXPERIMENT

Very interesting characteristics of bivalves are found out in the laboratory experiment conducted by Yamashita and Matsuoka (1994). They performed the experiment on the burrowing process of infant bivalves, or *Spisula sachalinensis*, maximum length of which is more than 5 mm and less than 20 mm, under the oscillatory flow generated by U-tube type oscillating water tunnel. They measured the burrowing velocity of bivalves to collect an information of statistical characteristics of bivalves. Furthermore, they investigated the probability of the occurrence of the bivalves picked-out-of sand layer under the various values of the ratio of the descending velocity of sand surface, or v_e, to the burrowing velocity of bivalves, or v_s. Figure 1 shows the one of the results of their study on the relation between the probability of the occurrence of the bivalves picked-out and the velocity ratio v_e/v_s.

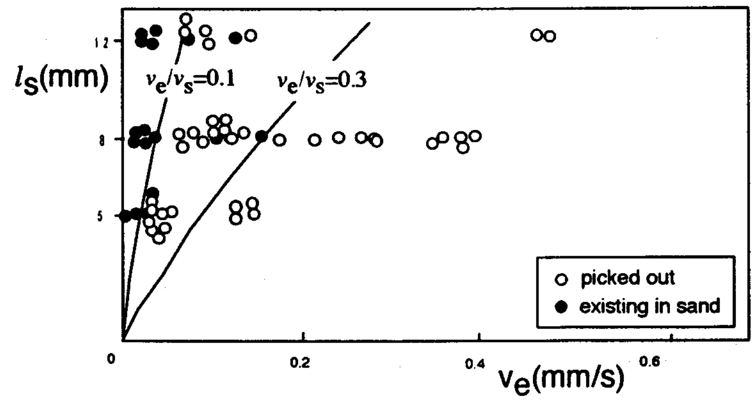


Figure 1: Occurrence of the bivalves picked-out (After Yamashita and Matsuoka, 1994)

According to this figure, in the region where the ratio $v_e/v_s \geq 0.3$, in other wards, the decending velocity of sand surface due to the erosion is larger than the one-thirds of the burrowing velocity of bivalves, whole of the bivalves in a ssand layer are picked-out-of sand. In their experiment, the burrowing velocity of bivalves are measured in a still sand layer. To understand the physics of this phenomena, the behavior of bivalves in a streamwise-moving sand layer should be investigated.

Another interesting characteristics is the existence of the region where some bivalves are picked-out and others are existing in a sand, detected in the region $0.1 \leq v_e/v_s \leq 0.3$. This means the importance of stochastic model of the motion of bivalves to express the probabilistic aspects of the moving process of bivalves.

STOCHASTIC CALCULATION FOR BURROWING PROCESS OF BIVALVES

Procedures of Stochastic Calculation

To perform the stochastic calculation of the motion of bivalves, the following assumptions, some of which are based on the previous experiments conducted by Higano, Kimoto and Yasunaga (1993) and Yamashita and Matsuoka (1994) are introduced.

(1) The shell length of bivalve l_s=8 mm, as schematically shown in Fig. 2. (2) The bivalve is burrowing in a sand layer keeping shell-length axis parallel to the vertical direction. (3) Initially, the top edge of bivalve coincide with the surface of sand layer, as schematically shown in Fig. 2.

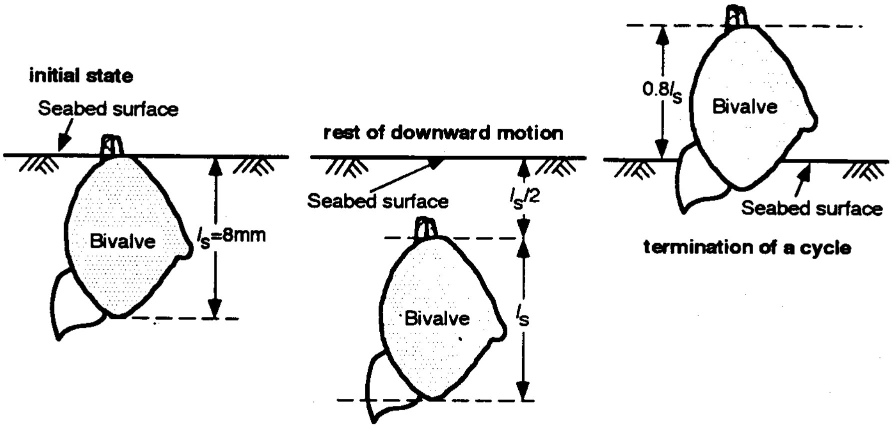


Figure 2: Definition of parameters in bivalves

(4) Bivalve begins to burrow in a sand layer when its top is exposed due to the descent of the surface of sand layer. The ratio of exposed length to the shell length follows the distribution experimentally investigated by Higano et al. shown in Fig. 3. (5) Bivalve rests in sand layer at the position where the clearance between its top edge and the surface of a sand layer is equal to the half of shell length as shown in Fig. 2. Hence burrowing and resting is iterated alternately. (6) Bivalve is defined to be picked out of sand layer when the 80% of the shell length are exposed in water as shown in Fig. 2. The satisfaction of this condition means the termination of the calculation for tracing the motion of individual bivalve.

(7) If the condition (6) has not been satisfied for 2200 s since the beginning of one cycle calculation, calculation is stopped. This situation is defined as the survival of bivalves. (8) As it is mentioned above, bivalve is repeating the burrowing process and resting process, the average of the repeating period of which is equal to 2.2 s, according to Yamashita and Matsuoka (1994). They also investigated the distribution of burrowing velocity of bivalves. In this calculation, the burrowing velocity of bivalves changes at every 2.2 s based on the Monte Calro Method, namely by generating the random numbers following the distribution of the burrowing velocity experimentally investigated. (9) The descending velocity of the surface of sand layer is treated as a probabilistic variables following the normal distribution, the standard deviation of which is equal to one-third of averaged descending velocity, or $\sigma=v_e/3$.

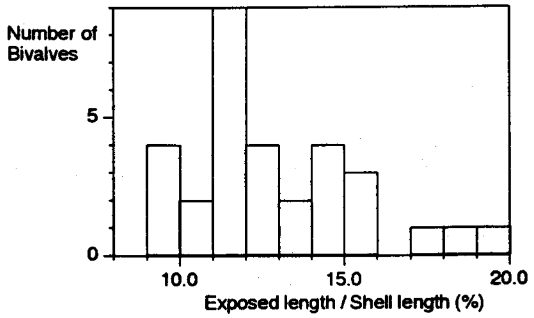


Figure 3: Distribution of exposed shell length (After Higano et al., 1993)

Results of the stochastic calculation

By continuing the calculation cycle for m-times with assessing the occurrence of the bivalves picked-out-of a sand layer, the probability of the survival of bivalves, or Pm, is calculated.

Figure 4 shows one of the calculated results of the one-cycle of the burrowing process of bivalves under the condition of v_e/v_s=1.0. In this figure, the time series of the existing height of bivalves, elevation of the surface of sand layer and the thickness of the sand layer above the top of bivalves, or δ, are shown. In this case, the thickness of the sand layer above the top of bivalves is decreasing rapidly during $0.0 \le tv_s/l_s \le 1.0$, and on the verge of the picking-out of bivalves, the thickness of the sand layer increase after tv_s/l_s=2.0. In this case, the bivalve survives through one cycle of calculation. If the decreasing velocity of the thickness of sand layer during $0.0 \le tv_s/l_s \le 1.0$ is a little more rapidly, this bivalve is picked-out-of sand layer. Whether a bivalve is picked-out of sand layer or not strongly depends on the accidental-drastic change of the surface elevation of sand layer.

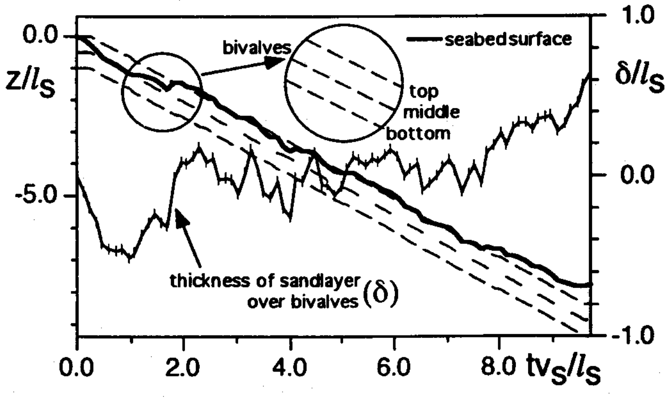


Figure 4: Burrowing process of bivalve

Figure 5 shows the existing probability of bivalves after 100 cycle of calculation, or P_{100}, against the ratio of the descending velocity of sand-bed surface to the burrowing velocity of bivalve, $v_e/v_s(=\alpha)$. Two cases of the calculation are shown in this figure. The one is the calculation in which the probabilistic characteristics of bivalves are only considered; and the other is the calculation in which both of the probabilistic characteristics of bivalves and that of sand-bed surface are considered. The transition range in which the probability of the survival of bivalves changes from 1.0 to 0.0, experimentally investigated by Yamashita and Matsuoka is also shown in this figure.

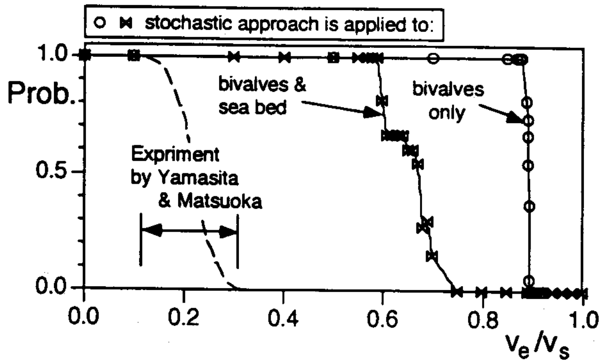


Figure 5: Survival probability of bivalve

The transition range of the experiment is $0.1 \leq \alpha \leq 0.3$, while the calculation, in which the probabilistic characteristics of bivalves is only considered, shows very drastic transition around α=0.9. The calculation, in which both of the behavior of bivalves and the motion of sand-bed surface are treated as the probabilistic variables, predicts the transition range in $0.6 \leq \alpha \leq 0.75$. By treating the change of the sand-bed-surface elevation as the probabilistic factor, the width of the transition range becomes large and the transition range also shifts to the direction to improve the agreement between the calculation and experiment.

But even if the probabilistic characteristics of sand-bed-surface elevation is considered, the threshold of the emergence of bivalves picked-out-of sand layer are overestimated. The stochastic calculation performed above reproduces the existence of the transition region in the probability of the existence of bivalves picked-out-of sand layer against the change of velocity ratio v_e/v_s. But the threshold of the existence of bivalves picked-out-of sand layer is not reproduced well at least quantitatively. This fact implies the existence of other mechanism, which promotes the picking-out-of bivalves.

BEHAVIOR OF BIVALVE IN SHEETFLOW LAYER

Reverse grading as the mechanism of upward vertical motion of bivalves

In debris flow, which occurs in the steep-gradient channel in the mountain area, the boulders are frequently observed to be concentrated at the near-surface region of the front part. This phenomena, namely the existence of the boulders above other grains and gravel, is called reverse grading.

The situation of the sediment transport in sheetflow regime has same characteristics as the debris flow, namely, both of the mechanism governing the momentum transport in debris flow and sheetflow are the interparticle collision. Because the concentration of the sediment is sufficiently high to keep the momentum transport due to the frequent interpartisle collisions. Therefore the same kind of phenomena shown in debris flow can be seen in sheetflow, in other words, the reverse grading can be thought to be a mechanism to promote the bivalves to be picked-out-of sand layer.

In this study, the behavior of bivalves are traced numerically in the moving sediment particles in sheetflow regime by the distinct element method (=DEM).

<u>Distinct element method</u>

Sakai and Gotoh (1995) performed the numerical simulation of the motion of sediment particles, on the surface-layer of which shear stress is acting, based on the distinct element method. In their simulation, moving particles in a uniform diameter were treated. In this study, this simulation is applied to the particle-flow with distributing particle size, by regarding large particle as a bivalve.

Governing Equations of Sediment Particles

In this simulation, the sediment particles are modeled by the rigid cylinders with uniform diameter *d*; and the bivalve is modeled by the rigid cylinder with diameter *D*. Among each cylinder, the spring and dashpot systems are introduced to express the particle/particle and particle/bivalve interaction, and the equations of motion of cylinder are solved by explicit method.

Equations of motion of the *i*-th particle or bivalve in the vertically two-dimensional coordinate are as follows:

(1) $$\frac{\pi\sigma_i d_i^2}{4}\left[\ddot{x}_i\right] = \sum_j \left\{-\left[f_n\right]_t \cos\alpha_{ij} + \left[f_s\right]_t \sin\alpha_{ij}\right\}_j + F_{0i}$$

(2) $$\frac{\pi\sigma_i d_i^2}{4}\left[\ddot{y}_i\right]_t = \sum_j \left\{-\left[f_n\right]_t \sin\alpha_{ij} + \left[f_s\right] \cos\alpha_{ij}\right\}_j - \frac{\pi(\sigma_i - \rho) d_i^2 g}{4}$$

(3) $$\frac{\pi\sigma_i d_i^3}{16}\left[\ddot{\phi}_i\right]_t = \sum_j \left\{\left[f_s\right]\right\}_j$$

in which f_n, f_s=normal and tangential components of the force acting between the *i*-th and *j*-th particles on the local coordinate system *n-s*; α_{ij}=contacting angle between the *i*-th and *j*-th particles; F_{0i}=shear force acting on the *i*-th particle; σ_i=density of particle; and d_i=diameter of particle; and *g*=graviational acceleration. $[f]_t$ means the *f* at the time *t*; and dot "·" means the time derivative.

Assecemnet of Interparticle Contact

In this simulation, all of the particles have same diameter, hence the accessment of the contacting particle is simply formulated as follows:

(4) $$R_{ij} \le 2r \quad ; \quad R_{ij} = \sqrt{\left(x_i - x_j\right)^2 + \left(y_i - y_j\right)^2}$$

in which (x_i, y_i), (x_j, y_j)=coodinate of the centroid of the *i*-th and *j*-th particles; R_{ij}=distance between the coodinates (x_i, y_i) and (x_j, y_j).

Calculation of Interparticle-Acting Force

Figure 6 shows the schematics of the interaction between two contacting particles. Between two particles, springs and dashpots are assumed to exist in both of the normal and tangential direction, to describe the dynamic interparticle relation. The acting force between the *i*-th and *j*-th particles in normal and tangential direction, f_n and f_s, can be written as follows:

(5) $$[f_n]_t = [e_n]_t + [d_n]_t \quad ; \quad [f_s]_t = [e_s]_t + [d_s]_t$$

(6) $$[e_n]_t = \min\{[e_n]_{t-\Delta t} + k_n \cdot \Delta\xi_n,\ e_{nmax}\} \quad ; \quad [d_n]_t = \eta_n \cdot \Delta\dot{\xi}_n$$

(7) $$[e_s]_t = \min\{[e_s]_{t-\Delta t} + k_s \cdot \Delta\xi_s,\ e_{smax}\} \quad ; \quad [d_s]_t = \eta_s \cdot \Delta\dot{\xi}_s$$

in which en, e_s=forces working on springs in the normal and tangential direction; d_n, d_s=forces working on dashpots in the normal and tangential direction; $\Delta\xi_n$, $\Delta\xi_s$=displacement of particle in the normal and tangential direction during the time Δt (Δt=time step of the calculation); k_n, k_s=spring constants in the normal and tangential direction; and η_n, η_s=damping coefficients in the normal and tangential direction. In the natural environment, sheetflow is three-dimensional phenomena; while in this simulation, the motion of particles in the vertical-two-dimensional plane is treated. To compensate this gap, the effect of the aggregation of particles is considered by introducing the upper limit of the compression force acting on the spring, e_{nmax}, e_{smax}.

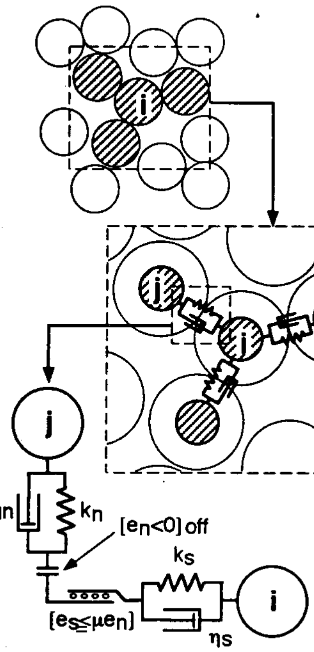


Figure 6: Schematics of the interaction between two contacting particles

In this simulation, particles are non-cohesive, hence the tensile force does not act between two contacting particles. To describe this characteristics, the joint, which no resistance to the tensile force, is assumed to exist in the normal direction. While, in the tangential direction, the friction force works. To describe this characteristics, the joint, which slips at the limit of the shear stress, is assumed to exist in the tangential direction. These joints can be formulated as follows:

(8) $$[f_n]_t = [f_s]_t = 0 \quad when \quad [e_n]_t < 0$$

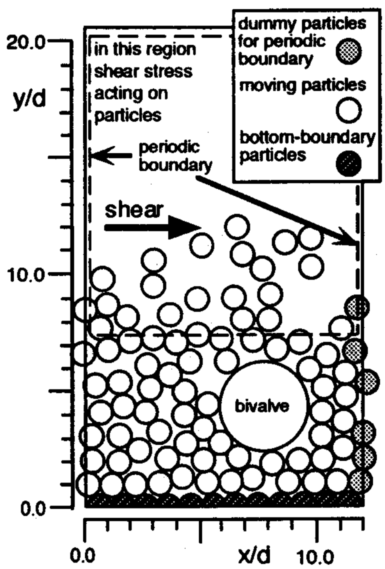
(9) $$[f_s]_t = \mu \cdot \mathrm{SIGN}([e_n]_t, [e_s]_t) \quad when \quad |[e_s]_t| > \mu \cdot [e_n]_t$$

(10) $$\mathrm{SIGN}(a,b) = |a| \quad when \quad b \geq 0 \quad ; \quad = -|a| \quad when \quad b < 0$$

in which μ=coefficient of friction.

Initial Conditions and Boundary Conditions

Figure 7 shows the schematic expression of the calculating domain. In this simulation, streamwise uniform condition is treated, therefore, the both sides of the calculating domain are the periodic boundaries to save the time of the calculation. The bottom boundary is the fixed rough bed constituted by the random-arranged particles with the same diameter as the moving particles. Before the beginning of the main calculation, the packing to determine the initial location of the particles are executed. At the initial state of the packing, particles are arranged with leaving 0.004 cm gap between each particles. During the packing process, the velocity of particles are monitored to assess the convergence of the packing calculation. The time for the packing is 0.1 s.



. **Figure 7: Schematics of calculating domain**

After the packing, the main calculation is performed to trace the motion of the sediment particles under the action of the shear force at the surface of the sand layer. The shear stress is distributed to the particles in the neighborhood of the surface of sand layer in a following procedure. First step is to set the threshold y_{th} (see Fig. 8); second one is to calculate the area of the particles above the threshold, or the shaded area in Fig. 8, for the particles exist on and above the threshold; and final one is to distributed the shear force to the particles existing on and above the threshold as the area of particles above the threshold $S(\theta_i)$ as follows:

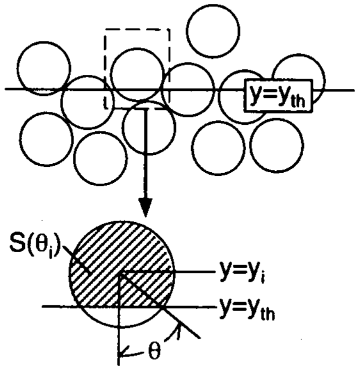


Figure 8: Shear stress distributed to sediment particles

(11)
$$S(\theta_i)=\begin{cases}\dfrac{r^2}{4}\left(\pi-\theta_i+\dfrac{1}{2}\sin 2\theta_i\right) & ;\ y_i \geq y_{th}\\ \dfrac{r^2}{4}\left(\pi-\theta_i+\dfrac{1}{2}\sin 2\theta_i\right) & ;\ y_{th}-\dfrac{r}{2}<y_i<y_{th}\end{cases}$$

(12)
$$\theta_i=\cos^{-1}\left(\frac{y_i-y_{th}}{r}\right)$$

Distributed shear force to the i-th particle is written as

(13)
$$F_{0i}=Lw_i\tau_0 \quad ; \quad w_i=S(\theta_i)\Big/\sum_{j=1}^{N}S(\theta_i)$$

in which L=length of the calculating domain in horizontal direction; τ_0=bottom shear stress par unit area. The test particle is 0.5 cm in diameter and 2.65 in specific gravity; and the model of bivalve is 2.0 cm in diameter and 1.30 in specific gravity. The specific gravity of bivalve is determined based on the measurement by Higano et al. (1993). In the calculating domain, 91 particles and 1 bivalve are traced. The model constants are shown in Table 1.

Table 1: Model constants

k_n	9.45×10^6 N/m	k_s	2.36×10^6 N/m
η_n	40.0 Ns/m	η_s	20.0 Ns/m
μ	0.577	Δt	2.0×10^{-5} s
e_{nmax}	2.5 N	e_{smax}	0.025 N

Initially the bivalve is contacting to the bottom constituting particles. The time step of the calculation is 2.0×10^{-5} s and the totally 80,000 cycles of calculation is executed, hence the motion of sediment particles and bivalve for 1.6 s are traced.

Results of simulation

Huge amount of numerical data, or data of the position and the velocity of sediment and bivalve at every 2.0×10-5 s are given as the results of the simulation. To understand easily the behavior of sediment and bivalve, the visualization of the numerical data is essential. Figure 9 shows the snapshots of the sediment-and-bivalve motion with 0.2 s intervals under the action of the shear stress $\tau_*(= u_*^2 / \sqrt{(\sigma/\rho-1)gd}\,)=1.5$ on the surface of sediment layer.

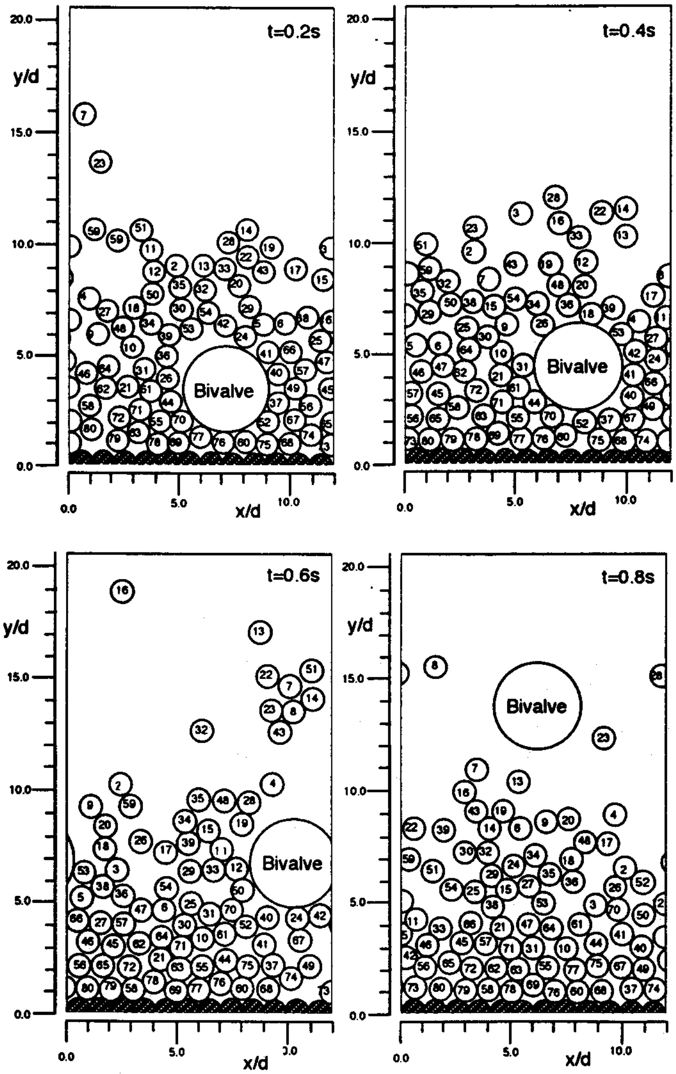


Figure 9: Snapshots of the sediment-and-bivalve motion

The bivalve, which is initially contacted to the bottom constituting particles, moves upward gradually, and it is picked-out-of a sand layer 0.8 s after the initiation of the shear action. Some of the sediment particles in the surface region are flowing down into the empty space due to the upward shift of bivalve. Although, this motion promotes the momentum transport from the upper region to the lower region to some extent, much of the particles in lower layer has laminar motion, hence the momentum exchange in lower layer is inactive. While, in the upper layer the concentration of sediment particles is less than that in the lower layer, the vertical motion of the sediment particles is less frequently obstructed by the particles in the upper layer than in the lower layer.

To understand the physics of the upward motion of bivalve, a following hypothesis is proposed. The bivalve exists in the shear layer, in which the sediment particles has a vertical velocity distribution as schematically shown in Fig.10. Therefore the particle colliding at the upper part of bivalve is faster than that colliding at the lower part of bivalve. This velocity difference of the colliding sediment particles makes the bivalve to rotate in clockwise. Supposing the existence of the contacting sediment particle just the lower part of the bivalve in the downstream section (hatched particle in Fig.10) and the hypothetical plane at the contacting point between bivalve and the particle, a simple situation, such as the clockwise rotating cylinder on the slope, can be considered. In this situation, the cylinder rolling upward on the slope is no doubt.

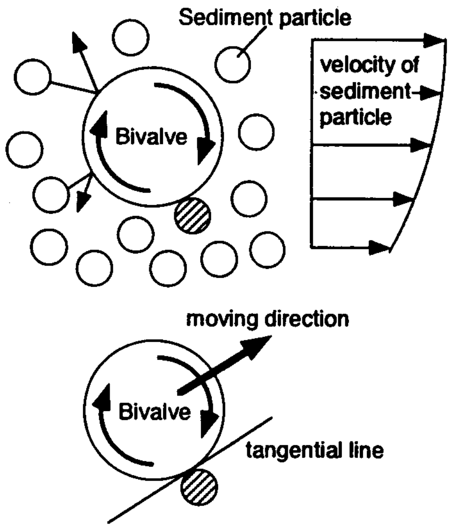


Figure 10: Upward motion of bivalve

Figure 11 shows the time series of the rotating angle, the sign of which is positive in counterclockwise, and the elevation of the centroid of bivalve. Bivalve begins to move upward gradually accelerating the rotational motion. Around 0.6 s from the beginning of the calculation, bivalve moves in upward direction drastically, and at the same time the rotational motion is accelerated also drastically. This fact means the rotational motion of the bivalve is the driving mechanism of the upward motion of bivalve.

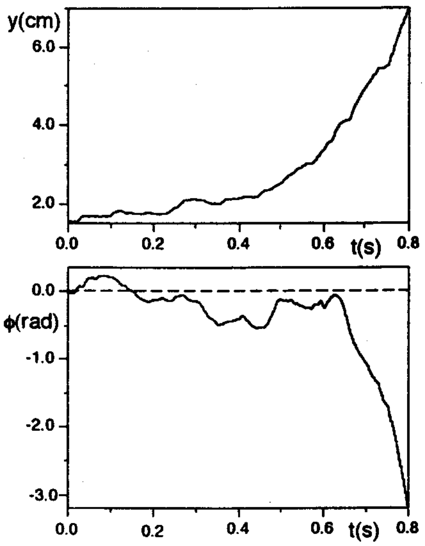


Figure 11: Rotating angle of bivalve

CONCLUSION

The physical background of the behavior of bivalve in sand layer during storm are considered from a viewpoint of sediment transport mechanics. Two important aspects are investigated: the stochastic aspect of the burrowing process of bivalves and the reverse grading phenomena observed in a motion of mixed-size grains.

The stochastic model simulates the probabilistic characteristics of environment around bivalves. And the granular material model, or distinct element method, simulates the motion of bivalve in upward direction based on the sediment/ sediment and sediment/bivalves interactions.

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A MODEL OF FISH ASSEMBLAGE RESPONSE TO HYDRAULIC AND FLUVIAL GEOMORPHOLOGICAL CHANGE IN A POOL-RAPID CHANNEL TYPE

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ABSTRACT

Fish assemblage dynamics have been shown to be influenced by changing annual flow regime and local hydraulic conditions. Additionally, changing abiotic factors act to alter the available habitat in the system. A dynamic model is proposed that relates the relative influence of hydraulic and geomorphological change on the fish species compositions of a pool/rapid channel type on the Sabie River in the Kruger National Park, South Africa. The model uses an example of increased sedimentation and decreased flows to predict geomorphological and associated fish assemblage change. Fish micro-habitat preferences have been investigated through an intensive field sampling programme conducted on the Sabie River, and abiotic species preference indices have been generated. Links have also been established between these indices and small scale geomorphological units, enabling prediction of fish assemblage change in response to an alteration of the local geomorphological structure of the river. Short term fish response appears to be a function of the hydraulic regime with changes in the hydrology (e.g. drought) changing local hydraulic conditions which modifies reproduction and predation rates. Recovery appears possible, given a return to the previous flow regime, if other abiotic factors have been unaffected. Where geomorphological change is also predicted as a response to changing flow regime, local micro-habitat availability is altered and a new fish assemblage develops in response to this.

KEY-WORDS: Channel change/Fluvial geomorphology/Instream habitat/Sabie River/Biotic response/Pool-rapid/Geomorphological change/Flow regime

INTRODUCTION

Attempts to predict the environmental requirements of riverine biota have concentrated on establishing the discharge regime which will maintain or enhance the habitat for riverine flora and fauna. Flow regimes were established using historical data to set flow minima (Tennant 1976) or periods of increased flows to correspond with fish migration and spawning (Hoppe 1975). Transect methods were developed which used cross-sections and the flow record to simulate values of ecologically important variables such as flow depth and velocity across the discharge range (Cochnauer 1976, White 1976). A further refinement of this approach was achieved by the technique of Instream Flow Incremental Methodology (IFIM) which linked the changing physical conditions to specific habitat preferences of the species present in the river (see Bovee 1982).

The IFIM approach has gained widespread acceptance and has been used in many river systems across the globe with computer packages being developed such as PHABSIM (Milhous *et al.* 1981), RHYHABSIM (Jowett 1989) and RIMOS (Vaskinn 1985). There have been criticisms of the method, however, including the interpretation of the weighted usable area data derived for each species (see Gore and Nestler 1988). Biological interactions and other biotic factors are also ignored. The approaches to simulation of the physical hydraulic data have been shown to break down under unsteady flow conditions, with calibration of backwater rating curves difficult, particularly under low flow conditions (Osborne *et al.* 1988).

Fundamentally, the models make the assumption that the channel does not respond to altered flow conditions by altering its dimensions and thus affecting the distribution of physical habitat in the river and the weighted usable area for resident species. Given the extent of documented river channel change following an alteration to one or more of the controlling catchment variables (see Gregory 1977) it would appear necessary to predict changes in habitat availability given changes to the fluvial geomorphology.

This paper outlines a conceptual model which provides the link between the geomorphological units present in a river and the percentage habitat available to resident fish species. The model is demonstrated using recent documented geomorphological changes to a pool-rapid reach of the Sabie River in the Kruger National Park, South Africa. The model assumes that the fish present will initially respond following different hydrological events (e.g. droughts, floods, freshets) causing a short term change in the species composition. It is, however, the resulting geomorphology that influences the substrate and cover available and this can be used to predict potential long term species composition for the reach.

MODELLING SHORT AND LONG TERM HABITAT AVAILABILITY

Short term changes in the biotic composition of a particular river reach following an alteration to the flow regime may be estimated with reference to the historical effects of discharge variation on the study river. Historical data for the Sabie River in the Kruger National Park indicates that under extreme drought flows many lotic fish species suffered increased mortalities and failed to breed. Lentic fish survived in placid reaches and migratory fish failed to migrate (See Pollard *et al.* 1996, Weeks *et al.* 1996).

Longer term channel change can be usefully measured at the scale of geomorphological unit (pools, rapids, bar types etc.) and new geomorphological associations following channel change can be mapped for an entire reach quickly and easily using aerial photographs (Fig. 1a). Each of these geomorphological units can be split into potential utilisable habitat sub-categories using data compiled by an experienced riparian ecologist who has studied the habitat structure of a representative number of examples of each geomorphological feature in the field. It is then possible to convert the geomorphological map of the study reach into a habitat availability map (Fig. 1b) and to calculate the percentage availability of each habitat category available to the biota present. Finally this can be related to the habitat preference curves generated for each of the species present in the river to estimate the likely long term fish assemblage.

Linking the short and longer term models of fish response to changing flow regime and fluvial geomorphology provides a conceptual framework for the prediction of likely fish assemblages given an alteration in the physical state of a river system following an alteration to its catchment control variables such as flow regime (Fig. 2). This model will be illustrated using data from a pool-rapid channel type in the Sabie River in the Kruger National Park.

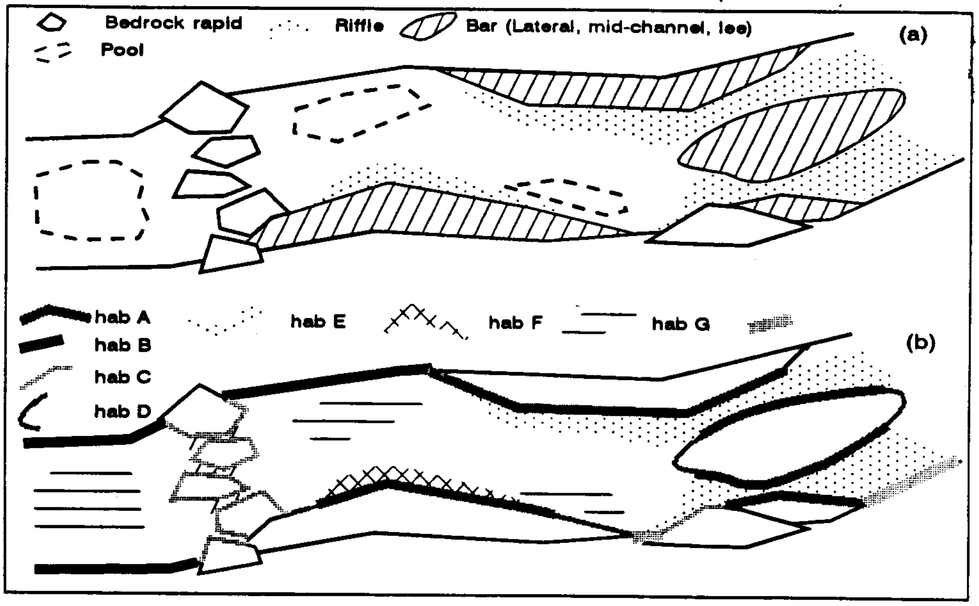

Figure 1: Geomorphological associations (a) translated into related fish habitat (b)

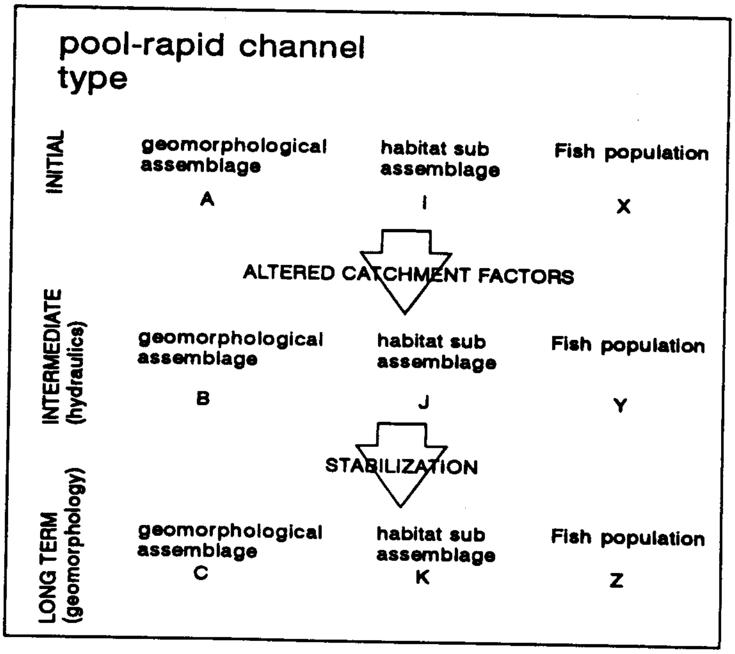

Figure 2: Fish assemblage response to geomorphological change as a result of catchment variable change

CHARACTERISTICS OF THE STUDY RIVER.

The Sabie River drains a catchment area of 7096km^2 in the Mpumalanga Province of South Africa. It has a perennial flow regime that is subject to seasonal extremes of flow with winter flows as low as 0.5 m^3/s and summer high flows of 25 to 40 m^3/s, extreme floods events in excess of 1000 m^3/s have occasionally been recorded. The river is incised into the host rock in the Lowveld region creating a narrow deep valley termed a macro-channel. Within this macro-channel a variety of channel types have been recorded including bedrock and mixed anastomosing channels, alluvial single thread, braided and anastomosing channels and pool-rapid channels (van Niekerk *et al.* 1995), these are a reflection of the degree of bedrock and alluvial influence on the river locally. This paper concentrates on the pool rapid channel type of van Niekerk *et al.* (1995) which has a characteristic assemblage of geomorphological sub-units (Table 1)

Table 1 Morphological units associated with the pool-rapid channel types on the Sabie River. (after van Niekerk *et al.* 1995).

Description	Channel Type	Morphological Units
System of shallow faster flowing steeper bedrock dominated rapids and associated upstream backwater pools.	Pool/Rapid	Bedrock/mixed pool (1), Rapid (1), Bedrock core bar (2), Lee bar (2), Cataract (2), Boulder bed (2), Armoured area (2), Floodplain (2), Alluvial distributary (3), Rock distributary (2), Terrace (2), Rock backwater (2).

(1) = Definite occurrence, (2) = Probable occurrence, (3) = Rare occurrence

SHORT TERM EFFECTS OF ALTERED FLOW REGIME ON FISH HABITAT FOR THE SABIE RIVER

Generalised seasonal fish response to differing flow scenarios have been investigated for the Sabie River in the Lowveld (Table 2) (Weeks *et al.* 1996 and Pollard *et al.* 1996 report).

The results indicate that fish response varies depending upon the season and whether they are lotic, lentic or migratory. Generally the Cichlids are most resilient to persistent extreme low flow conditions with Cyprinids and Silurids severely reduced in numbers. The pattern changes for moderate and elevated flows with the Cyprinids and Silurids remaining unaffected or increasing and the Cichlids initially increasing and then decreasing as its breeding sites are scoured by higher energy flows.

Although fish numbers and species abundance appear to be severely affected by extreme alterations to the flow regime populations, can begin to recover due to recolonisation from offstream and tributary refugia. The success of this recolonisation process will depend on the degree of influx of individuals from the refugia and the distribution of habitat suitable for reach individual species. This habitat distribution can be assessed through an examination of the new fluvial geomorphology of the reach.

Table 2. The response of fish to different duration's of discharge in the dry and the wet season (after Pollard *et al*. 1996 and Weeks *et al*. 1996)

Flow Scenario	Dry Season	Wet Season
extreme drought flows (0-1 m^3s^{-1})	▼ **Lotic Fish:** Increased mortalities. Fish in poor condition. No breeding. ▼ **Lentic Fish:** Good survival in placid reaches. Most fish remain in good condition. Breeding continues except in more sensitive species as water quality deteriorates. ▼ **Migratory Fish:** No migration. ▼ **Fish Assemblage Changes:** Cichlids: Most abundant population. Cyprinids: Reduced to remnant population. Silurids: Reduced to remnant population.	▼ **Lotic Fish:** High mortalities with surviving fish in poor condition. No breeding. ▼ **Lentic Fish:** Increased mortalities with fish stressed and in poor condition. Limited breeding. ▼ **Migratory Fish:** No migration. ▼ **Fish Assemblage Changes:** Cichlids: Still the most abundant species following previous dry season. Cyprinids: Further reduction of dry season numbers. Silurids: Further reductions of dry season low numbers.
drought flows (1-3 m^3s^{-1})	▼ **Lotic Fish:** Increased mortalities and poor condition of many species. No breeding. ▼ **Lentic Fish:** Good survival and condition. Increased breeding potential at higher temperatures. ▼ **Migratory Fish:** No migration. ▼ **Fish Assemblage Changes:** Cichlids: Increase greatly. Cyprinids: Reduced. Silurids: Reduced.	▼ **Lotic Fish:** Increased mortalities. Most species stressed and in poor condition. No breeding. ▼ **Lentic Fish:** Good survival rates with fish mostly in good condition and breeding. ▼ **Migratory Fish:** Little to no movement expected. ▼ **Fish Assemblage Changes:** Cichlids: Most abundant and increasing in number, following the dry season. Cyprinids: Decrease further from dry season numbers. Silurids: Decrease in numbers.
low-flows (3-6 m^3s^{-1})	▼ **Lotic Fish:** Good survival and condition. No fish in breeding condition and no breeding. ▼ **Lentic Fish:** Excellent survival of fish in good condition. Ideal breeding conditions at higher temperatures results in prolific breeding. ▼ **Migratory Fish:** No migration expected. ▼ **Fish Assemblage Changes:** Cichlids: Increased. Cyprinids: No change. Silurids: No change.	▼ **Lotic Fish:** Good survival with adequate habitat available. Most species in good condition, most in breeding condition. Little breeding at these low-flows unless preceded by a freshet. ▼ **Lentic Fish:** Excellent survival potential with fish in good condition in slower flows. All cichlid species breeding. ▼ **Migratory Fish:** Little migration expected but some seasonal/local movements possible. ▼ **Fish Assemblage Changes:** Cichlids: Marked increase. Cyprinids: No change in numbers. Silurids: Slight reduction in numbers.
low-flows to freshets (6-20 m^3s^{-1})	▼ **Lotic Fish:** Good survival and improved condition of all species. Early summer month freshets result in species coming into breeding condition early. No breeding if these aseasonal flows occur in the cooler months (prior to September). ▼ **Lentic Fish:** Good survival and condition. Breeding starts when the waters are warmer, in the dry season summer months (September-November). ▼ **Migratory Fish:** Little migration although early summer flushes may result in some recolonization migrations. ▼ **Fish Assemblage Changes:** Cichlids: Increase in the latter phase of the dry season. Cyprinids: No change in abundance. Silurids: No change in abundance.	▼ **Lotic Fish:** Excellent survival potential. All species in good condition. All species in breeding condition with some serial spawning. ▼ **Lentic Fish:** Good survival potential with fish equally in good condition. Some breeding. ▼ **Migratory Fish:** Some migration and local movement takes place. Some recolonization of reaches impacted during the dry season. ▼ **Fish Assemblage Changes:** Cichlids: Persist at levels attained so far. Cyprinids: Increase slightly. Silurids: Increase slowly.

Freshets to floods ($>20\ m^3s^{-1}$)	▼ Lotic Fish: Good survival potential. Species maintain condition although not in breeding condition. No breeding unless very close to the onset of the summer season. ▼ Lentic Fish: Increased mortalities as disrupted early summer breeding and the flushing of young fish from runs. Fishes surviving in backwaters in good condition. If the waters are warm prior to the floods, breeding is disrupted. ▼ Migratory Fish: No migration of fish is expected in winter. If flood flows occurred during the summer dry season months, early fish movements are expected. ▼ Fish Assemblage Changes: Cichlids: Reduction due to scouring of channel habitats. Cyprinids: Remain relatively abundant due to reduced dry season mortalities. Silurids: No change.	▼ Lotic Fish: Excellent survival potential. All lotic species in good condition, with all species in breeding condition and breeding. Some flood dependent spawners spawn massively. ▼ Lentic Fish: Reduced survival potential in flowing reaches. Those that find refuge from flow remain in good condition. Breeding is disrupted. ▼ Migratory Fish: Potamadromous or migratory fish move extensively within the catchment while the catadromous species migrate from fresh to salt water to breed. Migrations serve both to recolonize (juveniles and adults) and to provide suitable spawning and nursery sites (adults). ▼ Fish Assemblage Changes: Cichlids: Decrease. Cyprinids: Increase dramatically. Silurids: Increase.

LONG TERM EFFECTS OF ALTERED FLUVIAL GEOMORPHOLOGY ON FISH HABITAT FOR THE SABIE RIVER

Links Between The Fluvial Geomorphology And Available Fish Habitat.

Analysis of the habitat structure associated with the geomorphological units present in a pool-rapid channel type on the Sabie River was quantified by investigation of several examples of each unit from aerial photographs and field validation. Each unit was sub-divided according to cover and substrate character (Table 3).

Table 3. Habitat cover and substrate categories

Cover category	Substrate category
None	Sand
Offstream overhead (e.g. overhanging vegetation)	Gravel
Instream object (velocity shelter only)	Cobble
Instream overhead (visual shelter - e.g. aquatic vegetation, undercut banks, loose substrate in placid water)	Boulder
Combined cover	Bedrock

The averaged results of the habitat subdivision are given in table 4.

Links Between Available Habitat And Fish Preferences On The Sabie River.

An intensive study has been conducted into the habitat preferences of the fish species in the Sabie River (Weeks *et al.* 1996 and Pollard *et al.* 1996). From this several species may be highlighted as preferring pool-rapid channel types and their habitat preference curves are presented in figure 3.

Application Of The Fish Habitat - Geomorphology Links To Changes In A pool-Rapid Reach.

Initial State

A reach in a pool-rapid channel type was selected on the Sabie River in the Kruger National Park and the geomorphological units were mapped and quantified using 1:3000 scale aerial photographs (Fig. 4 and Table 5).

Table 4. Geomorphological units and habitat categories characteristic of pool-rapid channel types in the Sabie River.

Geomorphological unit	Biotope (%)	Habitat category	Percentage of habitat
Active channel braid bar		no cover, sand	98
		offstream overhead, sand	2
Active channel mixed pool	Pool (70)	no cover, sand	24.5
		no cover, boulder	1
		instream overhead, sand	45
	Run (20)	no cover, sand	14.5
		no cover, gravel	3
		no cover, bedrock	.5
		velocity shelter, gravel	.2
		velocity shelter, cobble	.2
		velocity shelter boulder	.5
		instream overhead, sand	.5
	Backwater (10)	no cover, sand	5.5
		offstream overhead, sand	.3
		instream overhead, sand	3
		instream overhead, boulder	1
Active channel bedrock rapid	Rapids (85)	combined cover, bedrock	76.5
		combined cover, boulders	8.5
	Runs (10)	no cover, bedrock	6
		velocity shelter, cobble	.1
		velocity shelter, boulder	.5
		instream overhead, bedrock	3
		combined cover, gravel	.1
		combined cover, cobble	.1
		combined cover, boulder	.2
	Backwater (5)	no cover, sand	2
		no cover, bedrock	1.5
		velocity shelter, sand	.05
		velocity shelter, bedrock	.05
		instream overhead, cobbles	.125
		instream overhead, boulders	.125
Macro-channel lateral bar		no cover, sand	5
		offstream overhead, sand	20
		velocity shelter, sand	2
		instream overhead, sand	63
		combined cover, sand	10
Macro-channel bank		offstream overhead, sand	75
		instream overhead, sand	20
		combined cover, sand	5
Active channel lateral bar		no cover, sand	25
		offstream overhead, sand	15
		instream overhead, sand	60
Lee bar		no cover, sand	95
		offstream overhead, sand	5

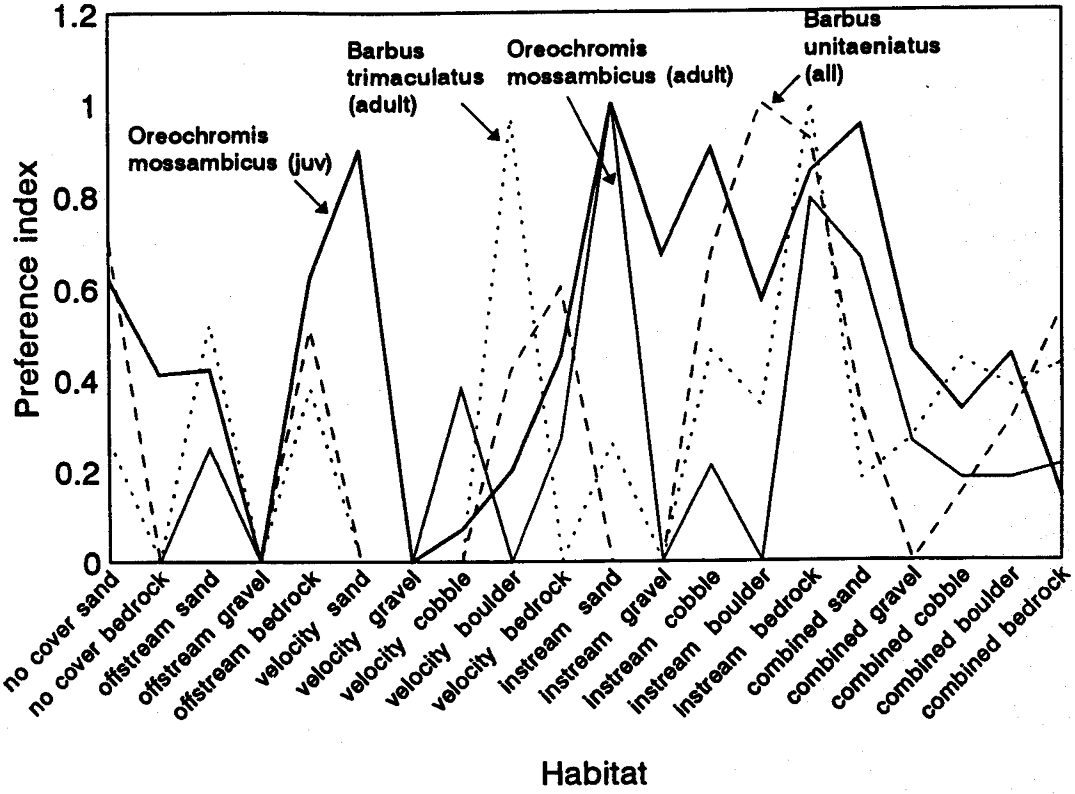

Figure 3: Normalised habitat preferences for three Sabie River fish species

Table 5. Quantification of geomorphological unit dimensions in contact with the flow at the study site.

Geomorphological unit	Linear extent (m)		Aerial extent (m2)	
	Original state	New state	Original state	New state
Macro-channel lateral bar	142	102	-	-
Active channel rapid	-	-	324	228
Active channel lateral bar	114	156	-	-
Active channel lee bar	14	22	-	-
Active channel braid bar	409	271	-	-
Macro-channel bank	110	59	-	-
Active channel mixed pool	-	-	1784	1152
TOTAL	789	620	2108	1380

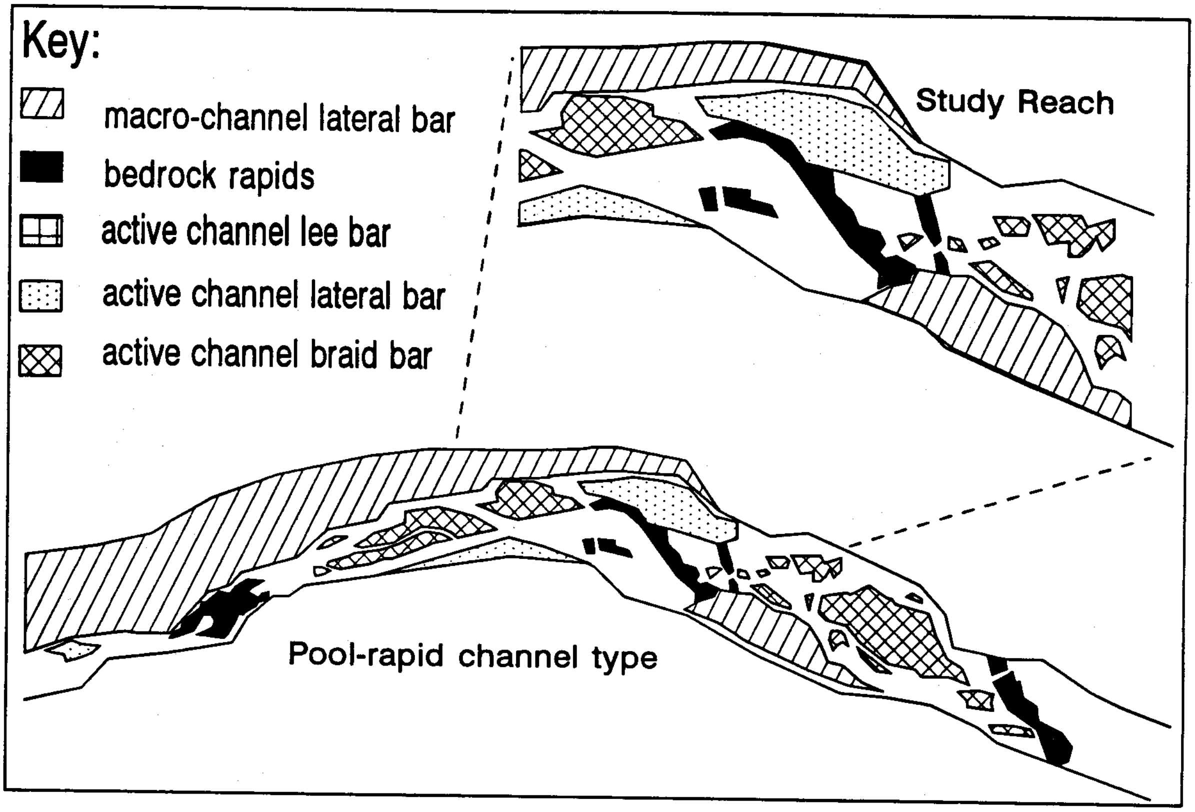

Figure 4: Geomorphological units of the pool-rapid study area

The reach consists of the following geomorphological features impacting on the flowing channel: Macro-channel bank, macro-channel lateral bar, active channel lee bar, active channel lateral bar and active channel braid bar. The high percentage of habitat associated with the fringes of braid bars will encourage fish species that prefer mobile shallow sandy substrates and a high percentage of rapid dwelling species will find suitable habitat in this reach where bedrock and localised loose substrate occurs and visual and velocity cover are offered.

Transitional State

The transitional fish assemblage following a flow reduction can be estimated from historical data recorded during the drought conditions that affected the Sabie River in the early 1990s (Table 2). During that period conditions approached those described under extreme drought flows (0-1m^3/s), lotic fish suffered mortalities, lentic fish survived in placid reaches and fish failed to migrate. In general Cichlids remained abundant while Cyprinids declined and Silurids were reduced to remnant populations.

Long Term State

Given an increase in sediment coming into the river and a decrease in flow duration and magnitude Heritage *et al.* (1996) have demonstrated using historical aerial photographs that braid bars tend to coalesce and lateral and lee bars extend their area in pool-rapid channel types. These extensions are often at the expense of bedrock geomorphological features such as rapids. This change in geomorphological makeup within such a reach is illustrated in figure 5 and quantified in table 6. Although there is a general increase in the aerial extent of braid bars the perimeter impacting on the water is reduced thereby causing a lowering in the available habitat for those species requiring mobile shallow sandy conditions, similarly lateral bars extend increasing their wetted perimeter in contact with the flow and providing increased offstream overhead and instream vegetative cover. The area of bedrock rapids and loose substrate is reduced leading to a decline in velocity cover and bedrock substrate.

The product of the quantified geomorphological parameters for the pool-rapid reach (Table 5) and the sub-habitat percentages (Table 3) produces an estimate of the amount of each habitat type available under initial conditions and following the change in geomorphological configuration (Table 6). The sandy no cover areas and sandy areas with instream vegetative cover displayed a significant reduction in aerial extent and sandy areas with offstream vegetative cover almost disappeared. The loss of rapids to sedimentation resulted in a decrease in the bedrock areas offering all types of cover.

Table 6. Available habitat for fish for the initial and altered geomorphological state of the pool-rapid study reach.

Habitat		Macro-channel lateral bar	Active channel rapid	Active channel lateral bar	Active channel lee bar	Active channel braid bar	Macro-channel bank	Active channel mixed pool	Total
No cover	sand	7.1/5.1	6.5/4.5	28.5/39	13.3/20.9	401/265.5		794/485	1250.5/820
	gravel							53.5/34.5	53.5/34.5
	cobble								
	boulder							18/11.5	18/11.5
	bedrock		19.9/17.1					9/5.7	28.9/22.8
Offstream	sand	28.4/20.4		17.1/23.4	.7/1.1	8.2/5.4	82.5/44.2	5.3/3.5	142.2/98.3
	gravel								
	cobble								
	boulder								
	bedrock								
Velocity	sand	2.8/2	.1/.07						2.9/2.07
	gravel							3.6/2.3	3.6/2.3
	cobble		.3/.2					3.6/2.3	3.9/2.5
	boulder		1.7/1.1					9/5.7	10.7/6.8
	bedrock		.1/.07						.1/.07
Instream	sand	89.5/64		68.5/93.6			22/11.8	865/558	1045/727.4
	gravel								
	cobble		.4/.3						.4/.3
	boulder		.4/.3					18/11.5	18.4/11.8
	bedrock		9.7/6.8						
All	sand	14/10					5.5/2.95		19.5/12.95
	gravel		.3/.2						.3/.2
	cobble		.3/.2						.3/.2
	boulder		28/19.8						28/19.8
	bedrock		248/174.4						248/174.4

Figures refer to original geomorphological state / new geomorphological state

Given the habitat preference curves for several species characteristic of pool-rapids on the Sabie River (Fig 3) it can be deduced that *Barbus unitaeniatus* and *B. trimaculatus* are likely to show a long term decline while

Oreochromis mossambicus will be favoured by the new geomorphological conditions.

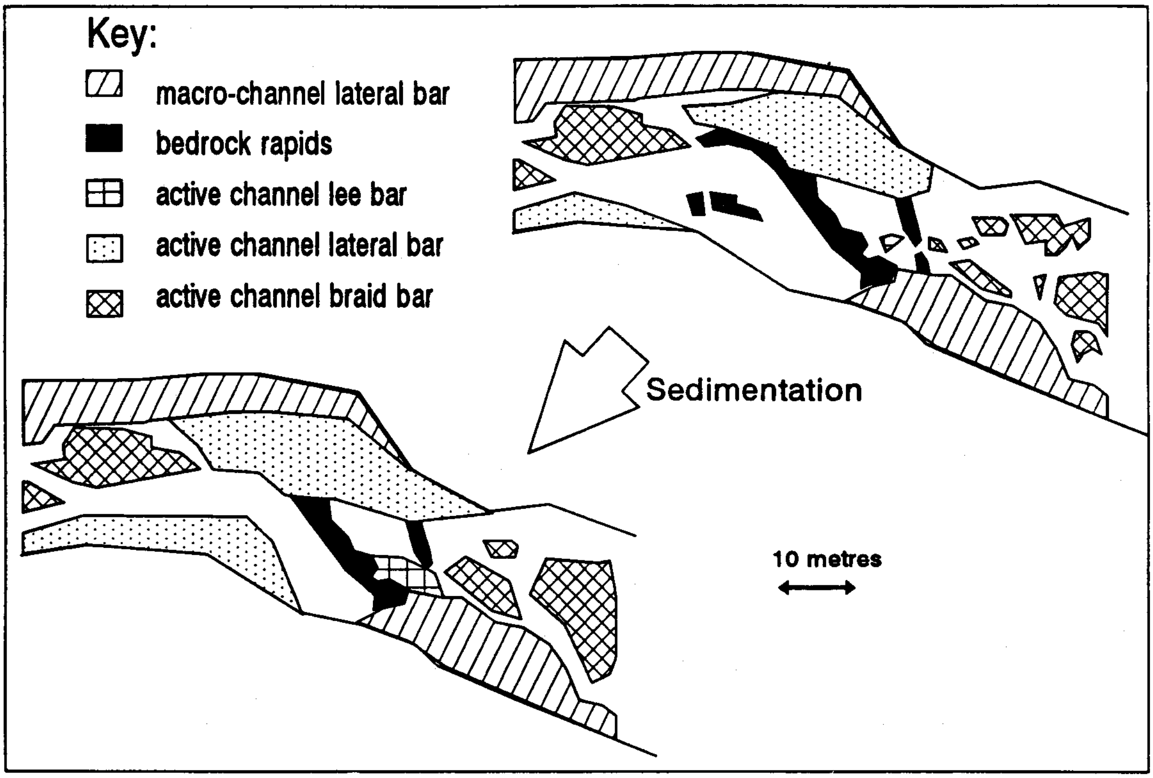

Figure 5: Change in the geomorphology of the pool-rapid study reach as a result of increased sedimentation

CONCLUSIONS

A model is presented that predicts long term fish assemblages as a function of the initial effects of a changed flow regime and the long term changes in fluvial geomorphology.

Direct links are drawn between habitat categories and geomorphological units allowing the mapping of the aerial extent of habitats from a geomorphological template.

The habitat availability is linked to potential fish assemblage through comparison of the habitat availability and habitat preference curves.

An example is presented for a reach in a pool-rapid channel type on the Sabie River in South Africa, this illustrates the geomorphological assemblage associated with the initial state of the pool-rapid and following geomorphological change after a change in the flow and sediment regime of the river. Fish assemblages characteristic of the initial, transitional and changed state of the pool rapid were identified and key affected species identified.

Efforts are continuing to validate the model using data for known geomorphological change and flow conditions and sampled fish assemblages, it is also envisaged that the model can be linked to the channel change model developed by Heritage *et al.* (1996) in order to predict the consequences of further catchment degradation and flow modification for the riparian biota.

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IMPORTANCE OF GRAVEL BARS AS SPAWNING GROUNDS AND NURSERIES FOR EUROPEAN RUNNING WATER CYPRINIDS

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ABSTRACT

Gravel bars are common riverine habitat features most frequently regarded as useless obstacles to water flow and thus as potential hazards for flooding in urbanised areas. Dredging policies are thus most common and frequently applied since dredged bars are progressively reconstituted during spates. However there is a dearth of knowledge on the actual impact of dredging and removal of gravel bars on fish communities, essentially because few investigations have been dedicated to the ecological importance of these structures as fish habitats. These considerations mainly apply to running water cyprinids which are essential components of the ichthyofauna of European rivers. This study was conducted in 1993-1994 in a 10 km stretch of the barbel zone in the River Ourthe (width: 50 m; mean annual discharge: 43 $m^3 s^{-1}$; River Meuse Basin, Belgium). It aimed to assess the importance of gravel bars as potential spawning grounds and nurseries for running water cyprinids and to model the impact of their removal in terms of loss of carrying capacity for 0^+ juvenile fish.

The common barbel *Barbus barbus* was selected as a target species for the study of spawning grounds, based on detailed knowledge of spawning behaviour and on the representativity of their spawning grounds for other species of the community. Only six sites, covering an overall 886 m^2 area (1.75 % of the stretch surface) were identified as spawning grounds of which the two largest (450 and 350 m^2) corresponded to large gravel bars targeted by hydraulic works. Fish sampling with DC powered electric fishing frames revealed that few 0^+ juveniles (< 20 g m^{-2}) were encountered in lotic or lentic habitats independent of the gravel bars during summer and autumn. In contrast large mixed 0^+ juvenile cyprinid shoals were captured along the edges of the bars (up to 240 g m^{-2}, mean: 84 g m^{-2}), substantiating their functionality as nursery habitats. Based on models of habitat utilisation by 0^+ fish and mapping of habitat availability, the overall carrying capacity of the two large gravel bars for 0^+ rheophilous cyprinids was estimated at 35,000 and 18,500 fish in late summer. Dredging these gravel bars as a preventive measure against the flooding of urbanised countryside would reduce the carrying capacity of these sites for 0^+ juveniles by an 80 % margin. Additional and more severe hydraulic works including river bed profile and bank rectification would imply a further decrease of the capacity down to 0.7-1.6 % of its present value. These results highlight the ecological importance of gravel bars and the potential impact of regular dredging practices on the fish community of the River Ourthe, especially in stretches where man-made obstacles may restrict the free circulation of spawners. Perspectives for a more ecological management of rivers and gravel bars are discussed, focusing on the possibility of integrating the current knowledge on fish biology and local population dynamics into the planning of hydraulic works.

KEY-WORDS: Hydraulic works / Dredging / Gravel bars / Spawning / Nursery / Habitats / Running waters / Cyprinids / *Barbus barbus* / River Meuse Basin

INTRODUCTION

The use of habitat by fish species has intrigued naturalists and fish biologists since the early days of these professions (Léger, 1909 in Souchon *et al.*, 1995). As early as 1949, Huet provided a comprehensive basis of fish distribution in European rivers depending on stream slope, although no real quantitative assessment was undertaken before the early 1970's (Waters, 1976; Bovee and Cochnauer, 1977; Stalnaker, 1979; Bovee, 1986). Numerous studies investigated habitat utilisation by fish species focusing on the relevance of habitat features (Bovee, 1978), the ecological plasticity of species in an assemblage (Grandmottet, 1983), the relationships between habitat availability and fish abundance (Binns and Eiserman, 1979; Fausch *et al.*, 1988) or between ecological diversity and taxonomic diversity (e.g. Winnemiller, 1991). Attempts to model the carrying capacity of rivers and streams with respect to existing habitat features or to their modification by human activity have been undertaken at different levels, starting from the local microhabitat up to the river basin. The relative success and relevance of these approaches is still debated since any approach that would look for accurate data would fail in assessing the spatial and temporal complexity of ecosystems whereas large scale investigations that would integrate this inherent complexity would compromise the minimum accuracy that is necessary for the understanding of biological phenomena. Understanding the relationships between habitat and fish communities is complex for several reasons: i) habitat is not rigid and changes at daily and seasonal scales, as well as between consecutive years; ii) fish are mobile organisms that rarely use a single set of habitat conditions: habitat preferences or utilisation patterns vary with their age or size, and depend on their activity (resting, feeding, spawning) as well as on the qualitative or quantitative composition of the assemblage; iii) the role of habitat features can be restricted to one life stage or activity, or influence several of them; their influence on the abundance or diversity of fish communities could be higher at life stages earlier than the ones considered in the study (notion of functional describers, Copp *et al.*, 1991); and iv) our understanding closely relies on our own way of investigating the problem (sampling strategies), what could be regarded in ethology as an influence of the observer on the investigated behaviour. Quoting M.B. Bain (1995), "the analysis of microhabitat use (by fish) has almost always been impossible because data have not been collected in non-used or unoccupied microhabitats". This consideration dramatically applies to temporary habitat features such as gravel bars which could not be regarded as fish habitat at the time when the survey is conducted, simply because they are emerged.

Gravel bars are common riverine habitat features in rivers and streams where they occur in natural meanders as well as in the vicinity of man made obstacles such as dams and bridges. They are most frequently regarded as useless obstacles to water flow and thus as potential hazards for flooding in urbanised areas. Dredging policies are thus most common in densely populated countries and frequently applied (every 3 or four years in Belgium) since dredged bars are progressively reconstituted during spates. However there is a dearth of knowledge on the actual impact of dredging and removal of gravel bars on fish communities, essentially because few investigations have been dedicated to the ecological importance of these structures as fish habitats. These considerations mainly apply to running water cyprinids, which are essential components of the ichthyofauna of European rivers. Although adult fish rarely use shallow habitats, cyprinids have been observed to spawn in the vicinity of gravel bars (Hancock *et al.*, 1976; Baras, 1992). Their larvae and juveniles have been found to select shallow depths and to use the edges of gravel bars as habitats during summer and autumn (Copp, 1992; Rincon *et al.*, 1992; Copp and Jurajda, 1993; Baras, 1995; Baras *et al.*, 1995). Since early life history stages are among the most critical ones and presumably condition the survival at the end of the first winter and thus the population abundance gravel bars could have a higher importance than initially suspected. This study aimed to assess the actual use of gravel bar structures as spawning grounds and nurseries for running water cyprinids and to give a preliminary insight on their importance in terms of carrying capacity for 0^+ juvenile fish.

MATERIAL AND METHODS

Study Area

The study was conducted in the River Ourthe which is the main tributary of the River Meuse in Belgium: it flows over 135 km, from the Belgian Ardennes down to the confluence with the R. Meuse in Liège. The study area is delimited by the mobile weir of Méry (12.7 km upstream of confluence) and by the weir of Esneux, 10 km upstream. The mean width, slope and discharge of the river in this stretch are 50 m, 1.3 ‰ and 43.3 $m^3 s^{-1}$, respectively. The water temperature averages 19.2°C in July (thermograph records 1989-1995).

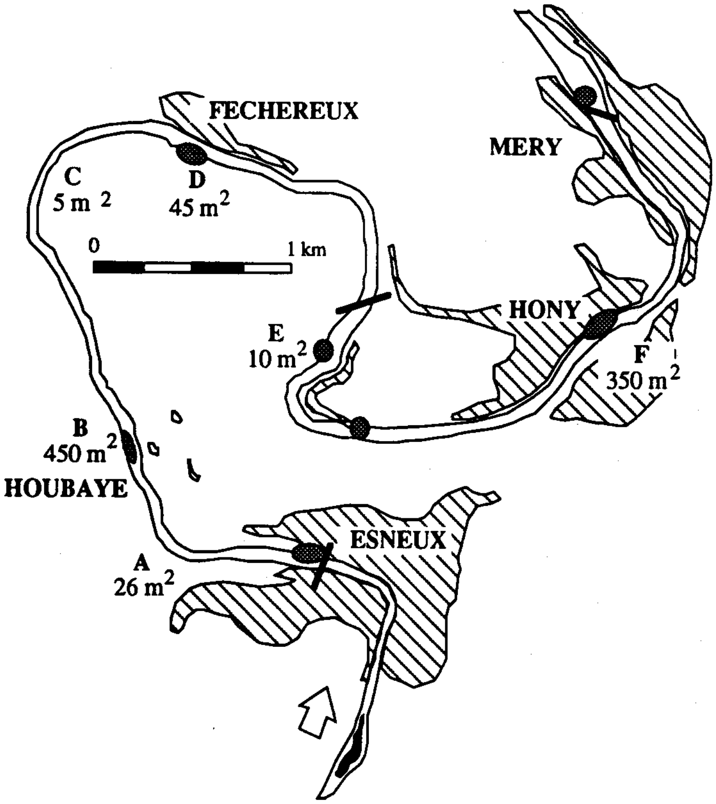

Figure 1: Map of the study area on the River Ourthe, between the weirs of Méry and Esneux. Oblique bold bars are water regulation weirs. Dashed polygons and grey ellipses represent urbanised areas and gravel bars, respectively. A-F: spawning grounds and surface.

All indigenous fish species recensed in the River Ourthe (*Anguillidae*, *Cobitidae*, *Cottidae*, *Cyprinidae*, *Esocidae*, *Gasterosteidae*; *Percidae*, *Salmonidae*, *Thymallidae*; Philippart & Vranken, 1983) are present in the study area, of which the assemblage is dominated by three categories of cyprinids: rheophilous species (essentially *Barbus barbus*, *Chondrostoma nasus*, *Leuciscus cephalus* and *L. leuciscus*), rheophobic species (essentially *Alburnus alburnus* and *Rutilus rutilus*) and accompanying species (*Gobio gobio* and *Phoxinus phoxinus*). Originally, this stretch of the river was typical of the lower barbel zone (Huet, 1949). Nowadays, it consists in about 85 % of mid to deep lentic habitats since the longitudinal and transversal profiles of the river bed have been modified by the building of dams and weirs for flow control, in order to maintain minimum levels for tourism during summer (Esneux) and to buffer the impact of spates during winter. The urbanisation pattern is also typical of the river Ourthe, with ancient villages on foot hills and more recent constructions (residencies, camping) in the floodplain, irrespective of the presence of rectified banks. Lateral or central gravel bars are frequent habitat features, both in natural meanders and downstream of man-made constructions (weirs and bridges). The largest bars are illustrated in figure 1.

Identification And Characterisation of Spawning Grounds

The common barbel *Barbus barbus* was selected as a target species for the study of spawning grounds, based on its representativity of the study area and fish assemblage (Philippart, 1987), as well as on its relatively low ecological plasticity with respect to the selection of spawning habitat (Grandmottet, 1983; Baras, 1992). Its selection also relied on detailed knowledge about the spawning ecology of this species. Spawning of *B. barbus* is most spectacular and boisterous when it involves hundreds of breeders gathering in shallow riffles and can be seen from distance by day or night without any interference on fish behaviour (Baras, 1994, 1995a). There was thus a minimum risk that operators would miss a major spawning ground of this species. The risk was further minimised as the periodicity of spawning behaviours in *B. barbus* was found to be dependent on precise temperatures (Baras, 1992, 1993, 1995a) that enabled the operators not to search in vain and to concentrate their efforts during a much more restricted period of the year. In addition, the spawning grounds used by *B. barbus* are frequently used by several other species in the assemblage (*Thymallus thymallus* in March, *Alburnoides bipunctatus*, *L. cephalus* and *P. phoxinus* in May and June).

Accordingly with previous findings, the search for spawning grounds of *B. barbus* in the study area started as soon as daily minimum water temperature in the river Ourthe (i.e. early morning) reached 13.5°C (late April 1993). Sites were watched by two operators on the banks until a spawning episode was detected. Each spawning place was located as the intersection of compass headings from marks on the banks. For each spawning ground, the observation went on until no spawning episode took place outside of previously observed limits. The operation was repeated during the spawning period on the whole river stretch. In order to give a complementary insight that no major spawning ground had been missed during this survey, a female barbel breeder (42 cm FL, 700 g) was captured in late April 1993 by electric fishing (nearby the village of Hony), equipped with a radio transmitter and followed by radio tracking from its release site. In each part of the river where spawning episodes had been detected, habitat features (water column height, water velocity, substratum) were measured at points 1 m apart on transects at 5 m intervals. All habitat surveys were carried out at the end of the spawning period since the water level had remained almost constant (1 cm variation) during this period of the year. The accuracy of spawning detection was empirically checked by the search for barbel eggs on the edges on the spawning grounds. The areas identified as spawning grounds in spring were later compared to emerged gravel bars in summer (proportion of overlap).

Sampling Of Juvenile Stages

The sampling methodology relies on the use of DC electric fishing in prepositioned frames that combine the advantages of direct current and of electric barriers and enable the quantitative sampling of microhabitats (Baras, 1995b). Each frame encompassed a 2 m^2 habitat whose perimeter was delimited by a closed steel cathode, with a 0.04 m^2 steel anode placed its centre. Following a recolonisation delay of 15 min, the frame was powered by a DC generator (EPMC, 2.5 KVA) and remained energetised until all fish had been captured. Just after fish capture, each 2 m^2 sample site was characterised by five points where we measured depth (nearest cm), water velocity (nearest cm s^{-1}) and type of substratum (Wentworth index). All sites were sampled at times of the day when 0+ fish density showed little variation between consecutive hours (10:00-17:00; justification and further details on methodology given in Baras, 1995b). Fish were sampled during summer (24 June - 17 August 1994) in the vicinity of gravel bars and in other microhabitats independent from gravel bars (riffle, run, pool, glide, various bank types,...). All captured fish were identified, counted and measured in each site. When hundreds of specimens of the same species were collected in a single site, length measurements were made on 50 randomly sampled fish. Biomass estimates were calculated from weight length relationships ($r^2 \geq 0.98$ for each species, based on samples $\geq$ 100 fish) and applied to habitat availability to extrapolate the carrying capacity of different areas or river profiles.

RESULTS

Use Of Spawning Grounds

As expected, barbel breeders started spawning on the first day when water daily minimum temperature reached 13.5°C in the early morning (28 April 1993) and maintained reproductive activity as long as the thermal regime of the river was above this threshold (2 May). Spawning was then suspended by colder temperatures and resumed for a few days later on (10-12 May). Barbel breeders were observed spawning in no more than six sites in the study area (A-F, figure 1). Four sites were poorly frequented and only few spawning episodes were observed despite intensive coverage: 7, 3, 10 and 4 episodes in sites A, C, D and E, respectively. In contrast, 110 and 83 episodes were recensed in site B and F, respectively, with up to 9 female breeders present simultaneously on a single ground (site B, deduced from almost simultaneous episodes). The radio tracked female barbel started migrating upstream soon after its release, tried to spawn on site E on April 28 but abandoned the site after a few minutes. It resumed its upstream movement but failed to clear the dam a few hundred metres upstream and finally moved downstream to site F, where it was observed spawning on 28 and 29 April. After spawning, it settled 150-200 m downstream of the ground and was consistently detected in this area until the end of the spawning period. As suggested by the movements and habitat use by this radio tracked probe fish and by complementary information from local anglers, it was unlikely that our coverage of the study area would have missed any of the major spawning grounds used by barbel breeders.

Barbel eggs were found in spawning pits 5-8 cm below the substratum in each of the six spawning grounds. Habitat features of the spawning grounds were (90 % range of distributions): depth from 10 to 26 cm, water velocity from 26 to 67 cm s^{-1} and gravel substratum (2-5 cm). The surface of spawning grounds ranged from 5 to 450 m^2 (figure 1) and amounted to 886 m^2, thus about 1.75 ‰ of the river stretch (about 50 ha). Except for site C, all spawning grounds were located on gravel bars emerged during summer or in their immediate periphery, especially the largest grounds for which the overlap with bars targeted by hydraulic works was 100 % (figure 2).

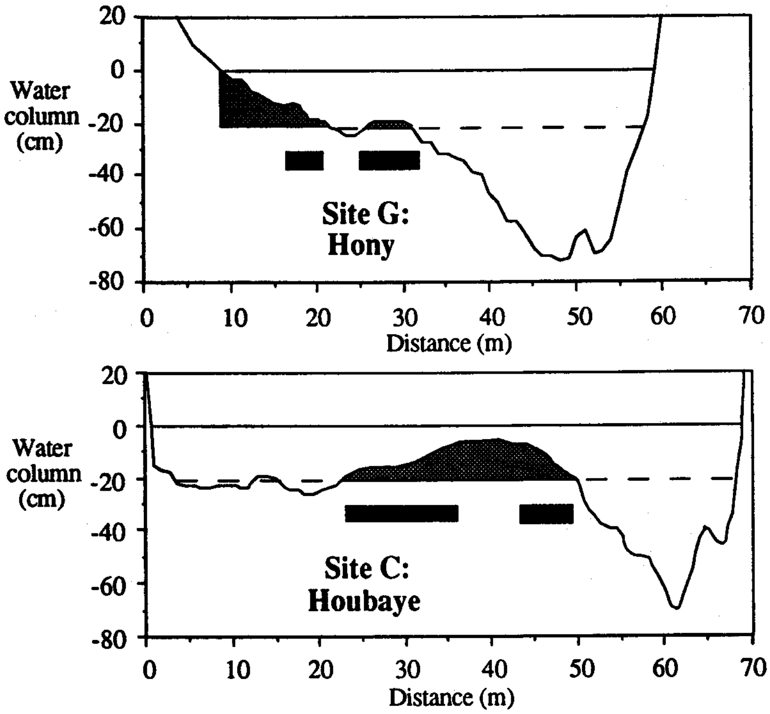

Figure 2: Transversal depth profiles of the River Ourthe in sites where barbel breeders were observed in spring 1993. Black rectangles: spawning areas. Solid and dotted lines: water levels in early May and in late summer, respectively. Grey areas: gravel bars as targeted by hydraulic works.

Habitat Use By Young-Of-The-Year Juveniles

During summer 1994, we sampled 60 sites belonging to 12 different habitat types of which five were intimately dependent on the existence of gravel bars, five others were open river habitat independent of gravel bars and two others which corresponded to modified banks as they stand downstream of Esneux (table 1). We captured 14,705 young-of-the-year (0^+) juveniles belonging to 14 fish species. Rheophilous cyprinids (*B. barbus*, *C. nasus*, *L. cephalus*, *L. leuciscus* and to a lesser extent *A. bipunctatus*) were the most frequently encountered species in our samples. They amounted 83.04 % of captures vs 6.80 % for accompanying cyprinids (*G. gobio* and *P. phoxinus*), 8.60 % for rheophobic cyprinids (*A. alburnus* and *R. rutilus*) and 1.56 % for other species (*C. gobio*, *G. aculeatus*, *N. barbatulus*, *S. trutta*, *T. thymallus*). Fish density and biomass ranged from 0 to 1,552 fish m^{-2} and 0 to 253.9 g m^{-2}, respectively, with fish length ranging from 11 to 39 mm in late June and from 33 to 69 mm in late August. Whilst fish density decreased throughout summer, fish biomass within similar habitat types showed less variation, the loss of density being compensated by the growth of fish (± 0.1 g and 0.95 g for mean fish lengths of 20 and 45 mm, in late June and late August, respectively). The distribution of fish biomass depending on habitat type and groups of species is illustrated in table 1. Except for stickleback, no-cyprinid species were mainly found in open lotic habitats with coarse substratum. Rheophobic cyprinids were encountered in all lentic habitats with fine substratum but they were proportionally more abundant in places with aquatic vegetation (*Ceratophyllum* sp.) and in the vicinity of gravel bars. Rheophilous and accompanying cyprinids also predominantly used lentic habitats but mainly in shallow and open environments as those on the edges of gravel bars, where the biomass was substantially higher than along natural or consolidated banks devoid of bars or along rectified banks (84 g m^{-2} on average for gravel bars vs 9.8 and 4.8 g m^{-2} for the two latter habitats).

Table 1: Relationships between habitat type in the River Ourthe and biomass (g m^{-2}) of young-of-the-year juveniles captured in summer by DC electric fishing in 2m^2 prepositioned frames. Values stand as mean (range) of five samples in each habitat.

Habitat type	Depth (cm)	Velocity (cm s^{-1})	Substr. (cm)	Aquat. Veget.	Rheophilous and accompan. cyprinids		Rheophobic cyprinids		Others (cottids, salmonids,...)	
1. Shallow riffle on front edge of gravel bar)	< 10	10-25	2-10	-	14.7	(8.3-22.6)	0.2	(0.0-0.9)	0.3	(0.0-1.2)
2. Shallow side edges of gravel bar	< 10	5-10	2-5	-	68.5	(42.7-83.2)	5.6	(0.0-15.8)	0.0	
3. Shallow rear edges of gravel bar or backwater	< 10	< 5	silt	-	146.8	(79.5-243.9)	7.5	(2.6-20.2)	0.0	
4. Open calms down of gravel bar or backwater	5-25	< 5	< 0.05	-	83.8	(67.6-128.9)	15.3	(3.9-31.6)	0.0	
5. Weeds down of gravel bars	10-40	< 5	< 0.5	+++	34.1	(6.2-52.4)	61.3	(25.6-83.7)	0.0	
6. Shallow riffle	< 10	10-25	2-10	-	2.6	(0.0-4.2)	0.0		3.4	(1.4-7.3)
7. Deep riffle	10-25	20-50	5-10	-	0.2	(0.0-1.8)	0.0		4.8	(0.8-20.7)
8. Rapids and runs	≥ 25	40-100	≥ 10	-	0.0		0.0		2.3	(0.0-6.9)
9. Open pool	≥ 50	5-10	variable	-	0.2	(0.0-0.8)	2.8	(0.0-9.3)	0.0	
10. Pool with vegetation	≥ 50	< 5	variable	+++	1.6	(0.0-4.3)	15.9	(2.8-25.7)	0.0	
11. Natural or consolidated bank (boulders)	5-25	< 5	variable	-	9.8	(2.9-26.3)	6.5	(3.1-9.7)	0.0	
12. Rectified bank	5-25	< 5	variable	-	4.8	(0.0-10.6)	3.5	(0.0-8.3)	0.0	
- Rootwads	not representative of summer habitat (emerged)									

In order to estimate the carrying capacity of the two largest gravel bars in the study area (sites B, central bar and F, lateral bar, figure 1) for rheophilous and accompanying cyprinids, we measured the availability of the different habitat types (1-5) and extrapolated the biomass from the samples collected by electric fishing, assuming that the structure of the assemblage would be homogenous over these areas. The estimates were 31.3 and 16.2 kg for bars B and F, corresponding to populations of about 33,000 and 17,000 young-of-the-year juveniles of rheophilous and accompanying cyprinids averaging 45 mm in late August. These bars would additionally carry about 2,000 and 1,500 juvenile rheophobic cyprinids. With respect to the overall distribution of habitat types in the study area, these two large gravel-bars offer a combined carrying capacity for juvenile rheophilous and accompanying cyprinids which would be similar to those of stretches of about 1,800 m and 2,500 m, with natural and rectified banks, respectively.

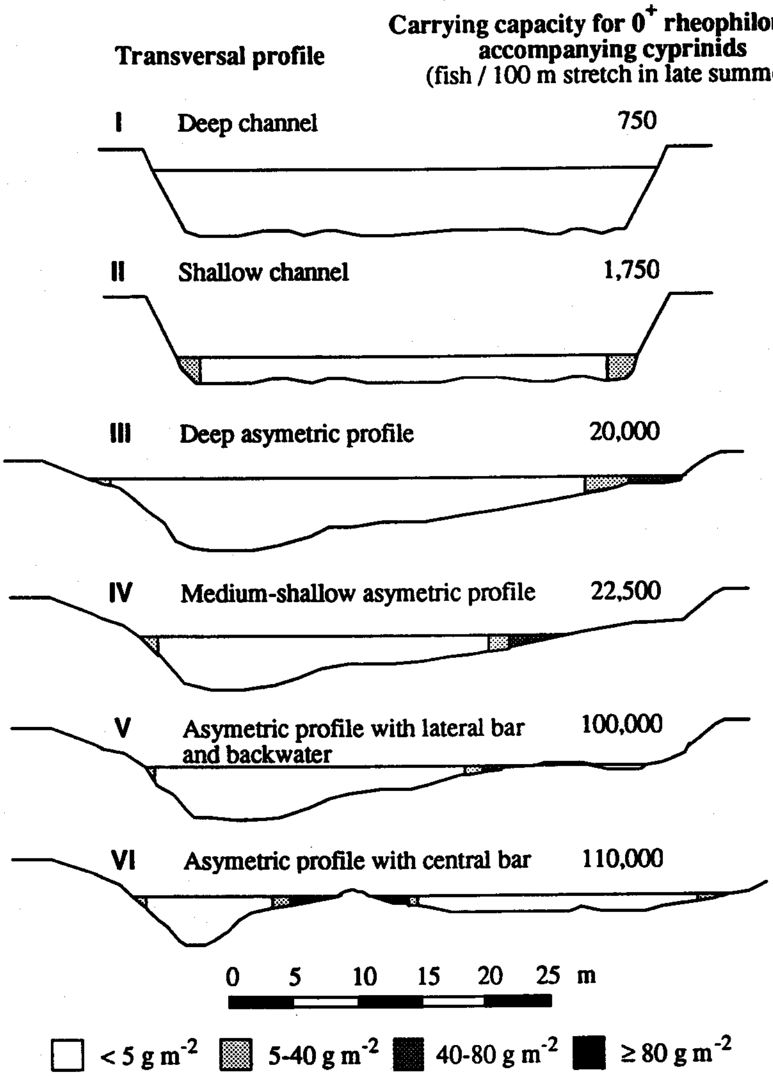

Figure 3: Estimates of carrying capacity of various profiles of the River Ourthe for young-of-the-year juveniles of rheophilous and accompanying cyprinids in late summer (fish averaging 45 mm FL). Profiles IV, V and VI are natural profiles. Profiles III, II and I correspond to increasing riverine habitat modifications by hydraulic works. Estimates based on the relationships between fish biomass and habitat type in table 1.

Figure 3 provides comprehensive estimates of what could be the carrying capacity of river stretches with different transversal profiles corresponding to increasing modifications of riverine habitat by hydraulic works. A modification exclusively consisting in the removal of surface gravel in excess, to prevent the settlement of terrestrial vegetation (profile III), would cut down the carrying capacity by a severe but temporary margin as these bars would be progressively reconstituted during winter spates. Such a possibility would not be granted by more severe hydraulic works including bank rectification and deepening (profiles II and I).

DISCUSSION

This study clearly highlighted the importance of gravel bars as key habitats for cyprinid fish in the River Ourthe, especially as spawning grounds since all sites where barbel breeders were observed spawning in the study area corresponded to emerged gravel bars in summer. We can not systematically exclude the possibility of underestimating the number of sites where breeders did actually spawn, although the intensity of the survey as well as the concordance between observation and radio tracking data makes it most unlikely, at least for large grounds used by numerous spawners. As suggested by the high frequency of spawning episodes, the two largest grounds (B and F, figure 1) probably were densely occupied by barbel spawners. Their removal by dredging practices would cut down the carrying capacity of the stretch by about 90 % and presumably cause barbel breeders to move to and spawn in other sites or river stretches. However, it is uncertain that breeders could clear that easily man-made obstacles at that time of the year, as suggested by the history of the radio-tracked female barbel. This further restriction could result either i) in spawning delays of which the consequences on the survival of the progeny are unknown (see Baras *et al.*, 1994 for parallel in large canalised rivers and use of fish passes), ii) in the selection of sites with lesser adequacy or iii) in the overcrowding smaller grounds with lesser adequacy or carrying capacity. With respect to the findings by Hancock *et al.* (1976) and Baras (1994) on spawners strategies, the latter alternative (overcrowding) could cause females to abandon most spawning episodes or to lay their eggs in less adequate habitats.

These perspectives for *B. barbus* apply to a certain extent to other running water cyprinids of which lithophilous breeders (Balon, 1975) are less exigent than barbel when choosing their spawning habitat (see Grandmottet, 1983). Breeders of *Alburnoides bipunctatus*, *Leuciscus cephalus* and *Phoxinus phoxinus* were indeed observed in the vicinity of the spawning grounds used by *B. barbus*. Similarly, the high density of young-of-the-year cyprinids captured on the edges of gravel bars suggests that these habitats had been used as spawning grounds by all lithophilous species. Larvae and juveniles of riverine fishes are known to drift (e.g. Penáz *et al.*, 1992) and could have colonised these gravel bars from upstream spawning sites, although the low density of juveniles observed in places other than gravel bars makes this hypothesis unlikely.

Gravel bars were described in this study as key nursery habitats during summer for young-of-the-year juveniles, essentially for rheophilous and accompanying cyprinids. Young-of-the-year juveniles probably need habitats other than gravel bars to successfully complete their ontogeny and growth until early winter but the simple fact of observing very large populations of juveniles gathering in these habitats indicates that they play an essential role in their ecology. Comparisons with studies in other riverine and lacustrine environments suggests that these shallow habitats would offer young-of-the-year juveniles size-limiting refuges from predation by larger fish (Bohl, 1980; Schiemer and Spindler, 1989; Copp, 1992; Sanders, 1992; Copp and Jurajda, 1993; Baras, 1995b). The estimates of carrying capacity produced in this study were high (up to 250 g m^{-2}) and presumably reflected abundant recruitment resulting from favourable weather conditions in spring. Despite this context, it is most likely that we underestimated the actual carrying capacity of gravel bar habitats due to exclusive daytime sampling. This is probably not the case for *Alburnus* or *Leuciscus* sp. which mainly move to near-shore waters during daytime (Winfield and Townsend, 1988; Copp and Jurajda, 1993; Baras, 1995b) but could apply to *Rutilus rutilus* which is known to migrate to weekly-sloped habitats during night-time (Copp and Jurajda, 1993) and possibly to other species whose ecological traits are less documented.

Whether rivers and streams should return to pristine conditions or dredging activity be maintained in river management is not the purpose of the paper. Man and fish species nowadays both are users of water resources with conflictual interests, implying that a compromise must be found that would satisfy both parties. From the calculations presented here, it is obvious that bed deepening and longitudinal profile rectification would cut down the carrying capacity of the river by a dramatic margin, making it impossible for the fish assemblage to recover. Such examples were provided in the past in the neighbouring River Meuse (Philippart *et al.*, 1988). The removal of the superficial layers of a gravel bar by hydraulic works would cause juveniles to loose important refuges from predation, shelters against displacement by sudden increases of river flow and would cut down recruitment by limiting the availability of spawning grounds. In rivers and streams with no or little flow control, the loss of carrying capacity resulting from the removal of superficial gravel would be most temporary as bars would naturally reconstitute during winter spates (within 3 to 4 years in the upper River Ourthe). It would still enable species with short life span (i.e. *A. bipunctatus*, *P. phoxinus*, *G. gobio* and to a lesser extent *L. leuciscus*) to complete their life cycle in appropriate conditions. In contrast, flow control measures, that limit the normal erosion process and drift of coarse sediments, could induce longer delays and compromise the ecology of these species, in a way similar to that of more drastic hydraulic works (e.g. profile rectification) for species with longer life span, such as *B. barbus* or *L. cephalus*. The impact of dredging bars will also depend on the ecological context, with a proportionally more severe effect on population abundance in the long run if dredging would take place following years with low recruitment success.

In order to minimise the damage caused to fish populations, dredging policies should further integrate updated biological data on fish assemblage, both at local and regional scale. These data would make it possible to decide more accurately which site should or should not be dredged, in order to prevent the destruction of all crucial bars in the same region or river stretch. In the area studied in the River Ourthe, it is most obvious that the gravel bars in Houbaye and Hony (sites B and F, figure 1) should not be dredged on the same year since they presumably are the two key habitats for rheophilous cyprinids in this region. Similarly, as the free circulation of breeders between the downstream and upstream reaches of this part of the river could be compromised by the dam in Fêchereux, a minimum of one large or several small bars should be spared by the dredging process. Ideally the key gravel bars of Hony or Houbaye should not be dredged before the year when fish population surveys indicate that the recruitment is higher than normal, what could be assessed as soon as June or July with DC electric fishing frames. Based on such information, dredging could thus be efficiently planned in late September or early October. This time of the year still grants most acceptable water level conditions for hydraulic works. It would also enable 0^+ juvenile fish to achieve most of their annual growth and reach a age and size at which they have higher swimming capacities and better chances to escape the hazards of predation and displacement by high flows.

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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)

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ABSTRACT

The data reported in this paper present clear effects of the channel morphology and flow dynamics, controlled both by the tidal regime and the ice cover, upon the migratory behaviour of the Atlantic tomcod population of the Sainte-Anne river (Sainte-Anne-de-La-Pérade, Québec). Measurements of channel morphology and flow velocity at low water indicates that the formation of the ice cover in winter, combined to the presence of sand bars near the confluence, causes a downstream decrease in the cross-sectional area of the channel. This reduction in area is compensated by an increase of flow velocity at the mouth of the river where velocities larger than 30 cm/s were measured at low water. Underwater video observations of tomcod movements in the Sainte-Anne river indicate that such flow velocities limit access of upstream migrating fish to the spawning site. The data demonstrate that upstream fish migrants avoid the downstream flow velocities occuring during falling tide and favor the short period of flow reversal associated to large rising tides to penetrate upstream.

KEY-WORDS: Atlantic tomcod / fish migration / channel morphology / flow velocity / tidal cycle

INTRODUCTION

Every winter, the majority of the Atlantic tomcod (*Microgadus tomcod*) population of the Saint-Lawrence middle estuary migrates to the tidal freshwaters of the Sainte-Anne river near Sainte-Anne-de-La-Pérade (Québec) to reproduce. While this tomcod population used to be abundant and stable, it has been declining dramatically since the mid-80's, thereby causing important economic losses to tomcod recreational and commercial fisheries (Mailhot *et al.*, 1988). In 1979, prior to the decline in the fish population, the economic activity associated to tomcod fisheries in Sainte-Anne-de-La-Pérade was evaluated at $2,5 millions. Since this time, the economic activity has been decreasing steadily, reaching a minimum low evaluated at $300 000 in 1988.

Mailhot *et al.* (1988) suggested that the presence of sand bars at the mouth of the Sainte-Anne river could be one of the factors explaining the decline in the fish population. This hypothesis is based on data showing a significant positive relationship between tomcod year-class strength and mean monthly discharge in the Sainte-Anne and Saint-Lawrence rivers during the spawning migration (Mailhot *et al.*, 1988; Fortin *et al.*, 1990). This relation suggests that for low water levels, the sand bars located at the mouth of the river block the upstream migration of tomcods. However, the authors did not provide any field data nor mechanism to support this hypothesis.

Because the available information concerning the migratory behaviour of Atlantic tomcod is scarce and controversial, it is difficult to assess the effect of the sand bars on migrating tomcods. Belzile and Leclerc (1992) documented the migration of tomcods in the Saint-Lawrence river near Grondines, a town located approximately 18 kilometers downstream from the mouth of the Sainte-Anne river. Their data indicated that upstream migration occurs mainly at the beginning of the falling tide. They suggested that this migration pattern could be explained by the fact that tomcods avoid the highly unstable flow conditions (high flow velocities and turbulence) associated to the rising tide. In contrast, results obtained by Cloutier and Couture (1985) in the Sainte-Anne river showed that tomcods tend to migrate upstream during the rising tide. While Fortin *et al.* (1990) also found a similar pattern in the Saint-Anne, they noted that an important proportion of tomcods was migrating against the falling tide.

The primary objective of this study is to investigate the migratory behaviour of Atlantic tomcod in the Sainte-Anne river in order to elucidate the effects of channel morphology and flow dynamics on this behaviour.

STUDY AREA AND METHODS

Study Area

The study was conducted on a 2 km-long reach of the Sainte-Anne river located immediately upstream from its confluence with the Saint-Lawrence river (Figure 1). This area is characterized by the presence of numerous sand bars that were deposited following the Saint-Alban landslide which occured in 1894, approximately 20 km upstream from Sainte-Anne-de-La-Pérade. The landslide injected more than 17 000 000 m^3 of sand and clays into the Sainte-Anne and modified the morphology of the river dramatically (Laflamme, 1900). Most of the material was transported rapidly along the steep upper portion of the stream but was then partly deposited in the vicinity of Sainte-Anne-de-La-Pérade where the stream gradient is gentler. Salient modifications of channel morphology occured near the confluence with the Saint-Lawrence (Laverdière, 1938). Prior to the landslide, the stream channel was deep and flow velocities were reduced in this section. Following the massive sand deposition, the channel became multi-threaded and shallow with higher flow velocities, a situation that is still prevailing today (Figure 2).

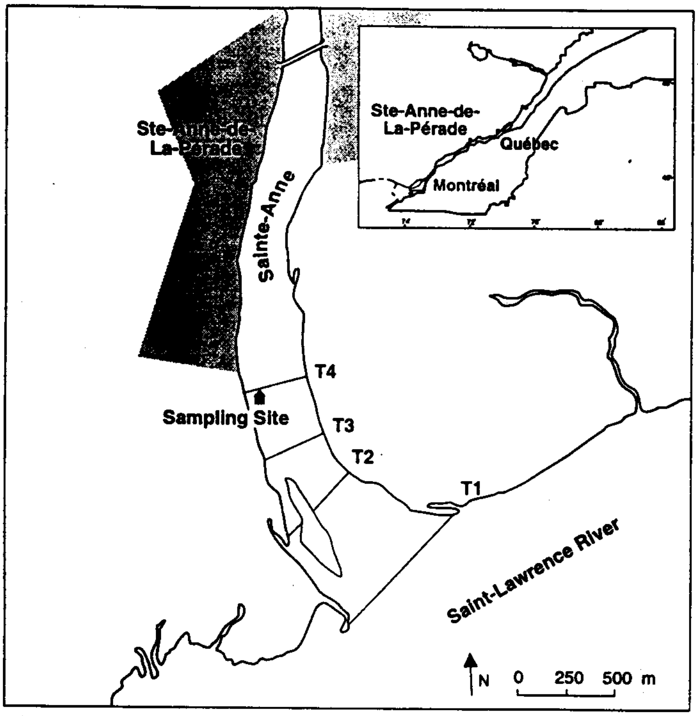

Figure 1: Location of study site.

Figure 2: The Sainte-Anne river near its confluence with the Saint-Lawrence river (July 1988).

Within the study reach, flow velocity and orientation is strongly influenced by semi-diurnal tidal fluctuations. For large tides, flood flows are sufficient to induce a flow reversal in the Sainte-Anne. For smaller tides, the velocity of the water outflowing the Sainte-Anne is only slightly reduced. Figure 3 shows an example of the temporal variation of flow velocity and water level over a complete tidal cycle at neap tide and two tidal cycles at spring tide. On this figure, flow velocities are positive when the discharge is outflowing while they are negative when it is inflowing.

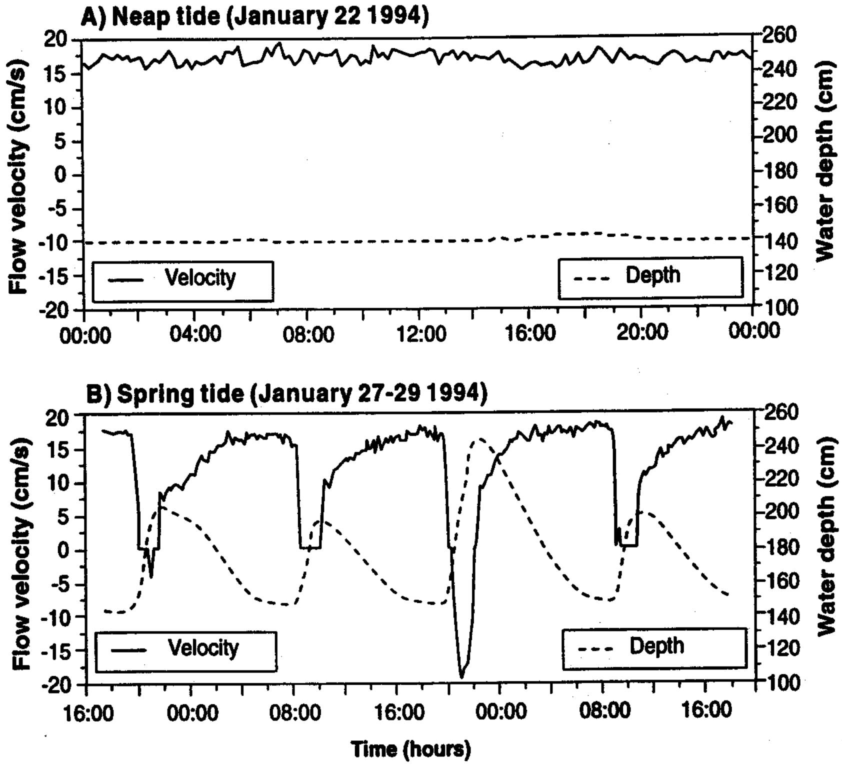

Figure 3: Variation of water depth and flow velocity at A) Neap tide and B) Spring tide.

<u>Flow and Channel Morphology Data</u>

In March 1994, the longitudinal variation of channel morphology and flow velocity was measured at four different transects in order to determine the effect of the presence of sand bars on the characteristics of the flow (Figure 1). The measurements were performed at low water so as to evaluate the strongest opposing flows that a fish would need to overcom to reach the spawning site. Along each transect, holes were drilled into the ice cover at regular intervals of 20 m. At each hole, the depth of flow and the thickness of the ice cover was recorded. The near-bed flow velocity was measured using a Price-type current meter positioned at 10 cm above the bed. This measure of flow velocity is believed to be the one providing the best description of the flow encountered by upstream migrating tomcods, as preliminary observations have shown that this species usually occupies the lowest 20 cm of the water column.

Biological Data

In January 1994, the migratory behavior of the Atlantic tomcod population of the Sainte-Anne river was documented in relation to the different phases of the tidal cycle The observations were made 1,3 km upstream from the river mouth, near Transect 4 (Figure 1). At this location, the channel is relatively deep (up to 1,6 m) and narrow (110 m), which ensures that all migrating tomcods are concentrated into a small area, thus facilitating the observation of their movements. A heated cabin, normally used for recreational fishing, was installed on the ice in order to facilitate instrumentation and data collection. An opening (0,4m x 3m) in the floor of the cabin allowed the instruments to be lowered and manipulated from inside the cabin. An underwater video camera was installed at the bed surface in order to monitor fish movements. The camera axis was oriented normal to the flow direction. Lighting was provided by two 200 watts spots suspended 1 m above the bed. The resulting camera field of view was 1,5 m long, 2 m wide and 1 m high. A Price-type current meter was installed 1,25 m in front of the camera and the probe positionned 20 cm above the bed. During video recording periods, the orientation and mean velocity of the flow was measured at 10-min time intervals over periods ranging from 45 to 80 sec. Between January 22 and January 29 1994, four series of data were collected in the field for a total of 124,5 hours of video observations. Each period of observation lasted approximately 25 hours in order to cover two complete tidal cycles. On January 27-29, the observation period lasted nearly 50 hours and thus covered four complete tidal cycles. Sampling dates were selected in order to cover a range of tidal heights and hydraulic conditions.

The video tapes were analyzed in order to determine the temporal pattern of migration of tomcods. The recordings were divided into 10 minute-periods, the center of each period corresponding to the time where the flow velocity was measured. Within each 10 minute-period, the number of tomcods and the direction (upstream or downstream) of their movements were determined.

RESULTS

Channel Morphology and Flow Dynamics

Analysis of morphological data indicates a decrease in the cross-sectional area of the channel in the downstream direction at low water (Figure 4). This situation is due to the combined effects of the ice cover and of the sand bars. In 1994, the very cold winter resulted in the formation of a thick (0,8 to 1 m) ice cover. Near the mouth of the river, the developement of the ice cover at the onset of winter gradually closed several of the shallowest sections of the multi-threaded channel, thereby reducing the cross-sectional area of the stream. In order to maintain continuity, the downstream decrease in channel area was compensated by an increase of flow velocity. Indeed, Figure 4 shows that the maximum velocity recorded at each transect increases from 16,7 cm/s upstream (Transect T4) to 35,5 cm/s near the mouth (Transect T1).

An investigation of the swimming capacity of Atlantic tomcod showed that flow velocities of the order of 30 cm/s have a negative impact on the movements of tomcods (East and Magnan, 1988). Flow velocities measured near the entrance of the Sainte-Anne at low water are therefore sufficiently high to restrain the upstream migration of tomcods toward the spawning site.

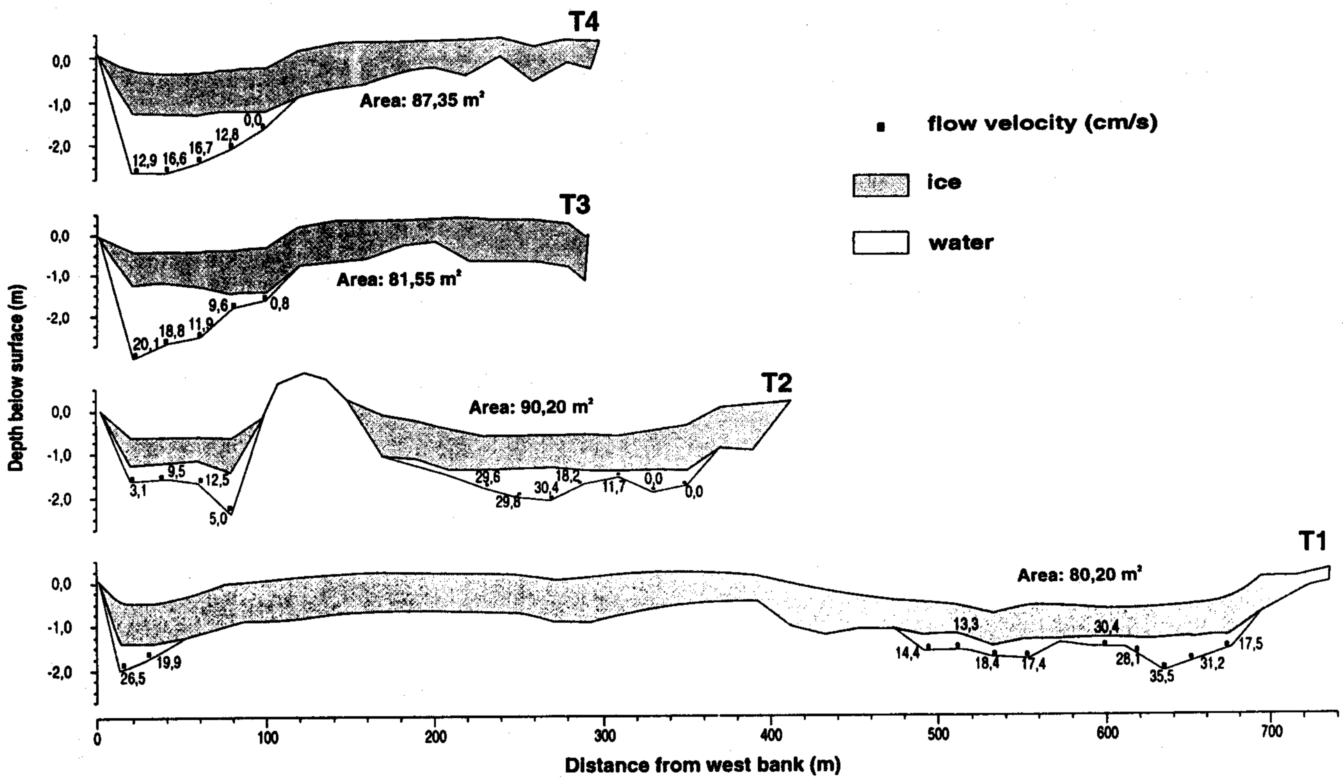

Figure 4: Variation of cross-sectional area and flow velocity at the four documented transects.

Migratory Behaviour of Atlantic Tomcod

Figure 5 shows examples of the temporal variation of upstream and downstream tomcod migration for two different tidal conditions. Figure 5a illustrates the pattern of migration observed on January 22 at neap tide while Figure 5b depicts the pattern observed at spring tide between January 27-29 1994.

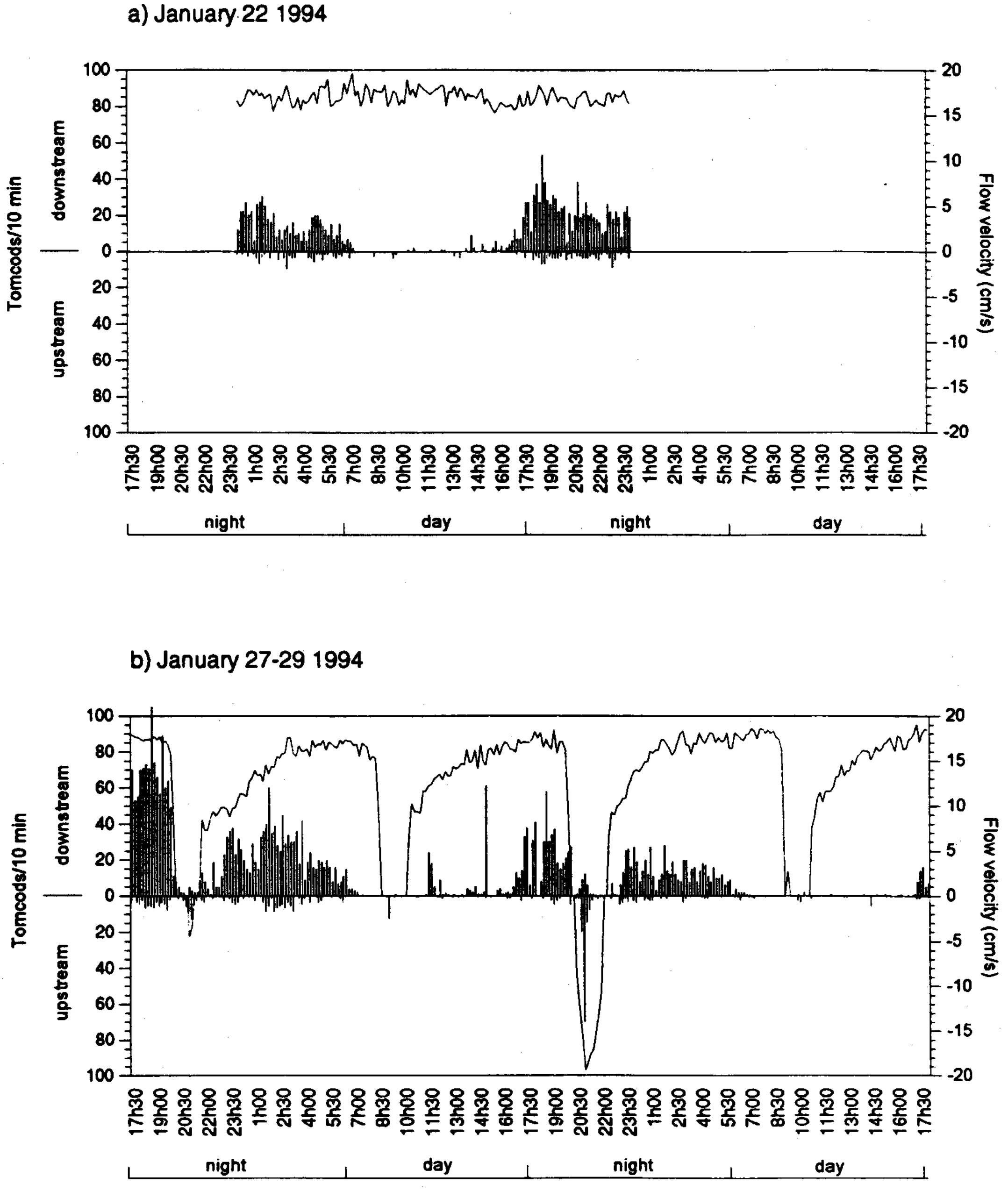

Figure 5: Upstream and downstream movements of tomcods in relation to the tidal cycle.

Diurnal Pattern of Migration

Figure 5 suggests an important influence of the circadian cycle on tomcod movements. For example, Figure 5a clearly shows that overall fish movements are predominantly nocturnal. In order to test this hypothesis, fish movements were divided in two groups: those observed during the day and those observed at night. The results indicate that an average number of 18,5 tomcods/10 min. were observed at night compared to only 1,85 tomcods/10 min. during the day (Table 1). They also show a larger variance of observations at night than during the day. Assuming unequal variances, day and night observations were compared using a z test. Results of the statistical analysis show that the difference between the two mean values is significant at the 0,01 level of confidence.

Table 1: Tomcods movements in relation to the circadian cycle

	Day	Night
mean*	1,85	18,5
standard deviation	4,99	17,6
n (periods)	317	417
z value	18,34**	

* mean of tomcods/10 minutes periods
** significant at the 0,01 level of significance

Tidal Pattern of Migration

Figure 5 also suggests that tomcod movements within the tidal section of the Sainte-Anne river are significantly related to the direction of the flow as it varies throughout the tidal cycle. Upstream movements of tomcods occur predominantly during rising tide while downstream movements occur during falling tide. Moreover, it demonstrates that upstream migration is almost non-existent during neap tide (Figure 5a) and during the falling tide period at spring tide (Figure 5b). In fact, upstream migration events seem to be confined to the short period of flow reversal associated with the rising tide. The data set collected between January 27 and 29 1994 was used to test this suggestion. This data set was selected because flow variations related to the tidal cycle are important. In order to remove the effect of the diurnal pattern of migration from the data, only nocturnal observations were used to perform the statistical analysis. Nocturnal observations of fish movements were thus divided in two groups, those for which flow velocity was larger than zero and those where it was smaller than or equal to zero. The results show that an average number of 7,25 tomcods/10 min. migrated upstream during the period of flow reversal (velocity smaller than or equal to zero) while only 2,32 tomcods/10 min. moved upstream during the positive flow velocity values associated to falling tide (Table 2). The results of a t test shows that the difference between the means is significant at the 0,01 level of confidence. The significantly larger variance of upstream movements during flow reversal is related to the apparent correlation between the intensity of the flow reversal and the number of upstream migrants per 10 minute-periods. Indeed, Figure 6 suggests that a strong correlation exists between the maximum negative flow velocity measured during rising tide and the number of upstream migrating tomcods. However, the available data set is not large enough to permit a statistical analysis of this relation.

Table 2: Tomcods movements in relation to the direction of the flow

	Upstream migration		Downstream migration	
	velocity > 0	velocity <= 0	velocity > 0	velocity <= 0
mean*	2,32	7,25	23,85	4,12
standard deviation	2,20	14,32	19,72	5,97
n (periods)	136	24	136	24
t value	3,82**		4,85**	

* mean of tomcods/10 minutes periods
** significant at the 0,01 level of significance

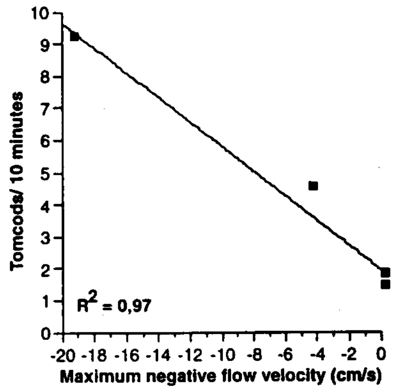

Figure 6: Upstream migration of tomcods in relation to maximum negative flow velocity at rising tide.

DISCUSSION AND CONCLUSION

The data reported in this paper present clear effects of the channel morphology and flow dynamics, controlled both by the tidal regime and the ice cover, upon the migratory behaviour of the Atlantic tomcod population of the Sainte-Anne river. Measurements of channel morphology and flow velocity at low water indicates that the formation of the ice cover in winter, combined to the presence of sand bars near the confluence, causes a downstream decrease in the cross-sectional area of the channel. This reduction in area is compensated by an increase of flow velocity at the mouth of the river where velocities larger than 30 cm/s were measured at low water. Our observations of tomcod movements in the Sainte-Anne river indicate that such flow velocities limit access of upstream migrating fish to the spawning site. The data demonstrate that upstream migrants avoid the periods of downstream flow velocities occuring during falling tide and favor the short period of flow reversal associated to large rising tides. Although this flow reversal is marked by important turbulence intensity (Martin *et al.*, this volume), it is the prefered upstream migration period of tomcods. This result is in contradiction with the suggestion of Belzile and Leclerc (1992) that migrating tomcods avoid the unstable flow conditions associated to the rising tide.

It is possible that the migrating pattern documented in this study is not entirely caused by the presence of sand bars near the confluence. Indeed, this migratory behaviour may also reflect the general necessity for tomcod to adopt an energetically efficient migrating strategy. However, it is impossible to test this hypothesis because of the lack of comparable data for other rivers. For this reason, it is suggested that future research should document the migratory behaviour of tomcod in a river unaffected by the presence of sand bars.

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RIVER-BED DEGRADATION INFLUENCED BY GROWTH OF VEGETATION ALONG A STREAM

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ABSTRACT

River-bed degradation is a current topic in river engineering because most sediments are stopped at many reservoirs and check dams constructed along the river. It causes not only severe local scours around hydraulic structures but it also affects the habitat of many organisms along the river. In addition, few floods after discharge control by dams and reservoirs cause enormous growth of vegetation. During base-flow stage, the river has the region where no water flows in cross section, therefore vegetation grows and expands its territory to the water edge of the river. Then, the flow has to accompany the luxurious vegetation during flood. The flow is much retarded by vegetation zone along the river, and water discharge concentrates in the narrow main channel. Therefore, river-bed degradation proceeds more severely in the narrowed main channel. The degradation concentrated in a limited part of the cross section promotes the growth of vegetation in base-flow stage. The growth of vegetation and the river-bed degradation are affected by each other.

In this study, the interaction between the growth of vegetation and the river-bed degradationdue to the stop of sediment supply from the upstream region and the regulation of discharge is discussed. The simplified situation is postulated and the rotational degradation is theoretically studied. The results suggest that combination of flood and low-flow stage discharges determines the growth of vegetation related to degradation. The proper control of the discharge is a key to keep favorable combination of flood control and habitat control in rivers under degradation.

KEY-WORDS: river management / rotational degradation / bed-load transport / vegetated channel / discharge fluctuation

INTRODUCTION

River metamorphosis easily occurs when the discharge and the sediment supply from the upstream changes. For example, Nadler and Schumm (1981) reported that the Arkansas and South Platte Rivers in eastern Colorado had been changed dramatically during the past 150 years due to the change of the use of the water. At first, these rivers were relatively straight, wide channel. However, these rivers became narrower and more sinuous with growth of vegetation due to regulated stream flow, increase of sediment by irrigation and a decrease in discharge during drought. A change of river, especially growth of vegetation like the above example, brings a change of habitat.

Now, many rivers have an artificial levee to keep their water course and to prevent an overflowing, and they are constrained inside the levees. Even in this situation, the change of the discharge and the sediment supply causes a change of river, though we may not call it metamorphosis. Tsujimoto (1994), and Shimatani and Kayaba (1995) reported that vegetation (wood) cover increased in flood plain after dam construction located at upstream. The dam construction brought the decrease of sediment supply in flood stage and discharge in base-flow stage. During flood, the decrease of sediment supply degenerates the balance of sediment transport and then river-bed degradation occurres. During base-flow stage, the decrease of discharge and the rive-bed deformation promote the concentration of the flow in a thalweg and it makes easy for vegetation to invade the dried area. As mentioned above, the growth of vegetation and the bed deformation are affected by each other, or they form an interaction system.

Increase of vegetation cover in flood plain brings about not only a change of habitat of many organisms but also some problems on flood control, for example, increase of hydraulic resistance. Thus, we must control the vegetation in flood plain from both view points of river environment and flood control. For this purpose, we have to understand the mechanism of a river system with growth of vegetation. In this study, an interaction between the river-bed degradation and the growth of vegetation is discussed, which is often observed after dam construction that causes a stop of sediment supply and flow regulation.

The aim of this study is to understand of the interaction mechanism of fluvial process in vegetated rivers. Thus, in this study, the following simplified situation is postulated: The rotational degradation (Gessler, 1971; Nakagawa and Tsujimoto, 1986) influenced by the growth of vegetation is treated under the assumption of quasi-uniform flow with the downstream direction. The key of the phenomenon focussed in this study is a property of the temporal change of discharge that is regulated by dam construction.

DISCHARGE FLUCTUATION

The fluctuation of discharge is characterized by repetitions of flood and base-flow or low flow stage. In this study, we simplify the discharge fluctuation as follows: The floods that have the constant duration time T_f and discharge Q_f repeat under the constant intervals T_b, and the discharge of the base-flow stage is Q_b (see Fig.1). Thus, the propertyof the discharge fluctuation is represented by 4 parameters T_f, Q_f, T_b

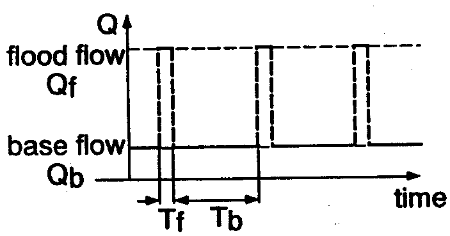

Fig.1: Model for discharge fluctuation

and Q_b. The flood stage is chracterized by a deformation of rive-bed, while the base-flow stage is chracterized by the period of growth of vegetation. We assume the flood flow to be the mean annual maximum discharge. Thus, the order of interval of flood is assumed to be about 1~2 years. Thus, the meaning of base flow is the averaged state of the interval of flood. However, dam construction tends to promote the discharge regulation in base-low stage and it makes the fluctuation to be small. Some kinds of willow (*Salix gilgianna*) or reed (*Phragmites japonica*) that we often observe in a river can grow into 1~2m tall within 1~2 years without flood damage. Here, we assume growth of such vegetation in base-flow stage.

ROTATIONAL DEGRADATION AND INVADING OF VEGETATION

We assume that the channel has a reach L, with the slope i_{b0} and the width $2B_0$. When no sediment is supplied from the upstream end of the channel and the bed-elevation at the downstream end is fixed, the slope decreases rotationally under the assumption of quasi-uniform flow with the downstream direction, which is termed rotational degradation, as shown in Fig.2. In this study, we discuss the interaction between degradation and growth of vegetation under the situation of the rotational degradation.

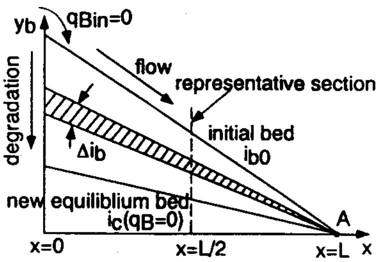

Fig.2: Rotational Degradation

As an initial condition, a vegetation zone of constant width b_0 is assumed along the side wall of the channel (Fig.3(a)). When flood discharge is not so large, there is still a zone without bed-load transport in the main channel, because the flow near the vegetation is retarded even in the non-vegetated zone (Fig.3(b)). As a result, parts of the cross section is never in degradation (Fig.3(c)). During the base-flow stage after a flood , if the water stage is so low that the region where no water flows appears in the cross section, the vegetation can invade there (Fig.3(d)). Then, the stream is to wait the next flood with wider vegetation zone. When the next flood comes, the water discharge concentrates more and more in the narrow main channel, and thus the degradation proceeds more severely in the narrow main channel (Fig.3(e)-(g)). The interesting topics are as follows: What will happedn by repetitions of the above-mentioned process? How wide will the vegetation zone develop?

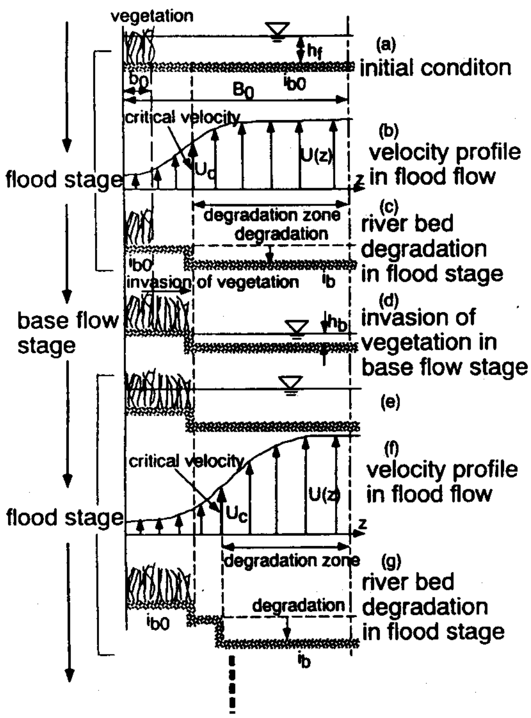

Fig.3: Rotational degradation with vegetation growth

MODEL FOR ROTATIONAL DEGRADATION IN FLOOD FLOW

MODEL FOR ROTATIONAL DEGRADATION IN FLOOD FLOW

The degradation zone in cross section is estimated by the analysis of flow with vegetyation zone. The development process of degradation by one flood is estimated by the simplified method by Gessler (1971), Nakagawa and Tsujimoto (1986) for rotational degradation.

The variation of the slope in the degradation zone is analyzed here under the following assumptions: (a) The total volume of degradation of the reach during Δt is equal to the total bed-load transport rate in the degradation zone during Δt. (b) the bed-load transport rate is expressed by the discharge per unit width q and the slope i_b. Then, the slope in degradation zone after n'th flood $i_b(n)$ is derived as follows;

(1)
$$i_b(n) = i_c(n) + [i_b(n-1)-i_c(n)]\exp[-\frac{3T_f}{T_{deg}(n)}]$$

(2)
$$T_{deg}(n) \equiv \frac{3L}{q(n)}$$

where $q(n)$=discharge per unit wodth on the degradation zone at nth flood; $i_c(n)$=equilibrium slope for critical tractive force at n'th flood. When the dimensionless critical tractive force τ_{*c} is 0.05 (Re_*>70; $Re_* \equiv u_* d/\nu$; ν=kinematic viscosity; d=sand diameter), $i_c(n)$ is expressed by the sand diameter d and the friction coeffcient C_f as follows;

(3)
$$i_c(n) = \frac{0.024 g^{0.5} d^{1.5}}{C_f(n)^{0.5} q(n)}$$

where g=gravitational acceleration. When we assume the resistance low to follow the Keulegan's equation, the friction coefficient of bed C_f is given as follows;

(4)
$$C_f(n) = \left\{ \frac{1}{\kappa} \ln\left[\frac{h(n)}{d}\right] + 6.0 \right\}^{-2}$$

where κ=kármán constant; and $h(n)$=water depth at the nth flood.

MODEL FOR INVASION OF VEGETATION IN BASE FLOW

In this study, for simplicity, we treat the situation that the time scale of base-flow stage T_b is much larger than the time scale for growth of vegetation T_v. In other words, we assume that the vegetation can grow sufficiently to cover the non-flow region during base-flow stage.

We assume that the zone of vegetation growth in the cross section is determined by the water edge of the representative section, which is located at $x=L/2$, in base flow stage and the vegetation zone expands into the region where no water flows at the representative section in the same width to the downstream direction. Under such assumptions, the water level during base-flow stage is easily estimated by the water depth of the uniform flow.

PROCEDURE OF ANALYSIS

The solusion of the flow analysis for the fully-developed and quasi-uniform flow provides the water depth and the lateral distribution of the primary velocityas shown in Fig.4 (Tsujimoto *et al.*, 1996; Tsujimoto and Shimizu, 1995). These determine the degradation zone and the discharge per unit width, q, there. The progress of the degradation in the degradation zone during the duration of flood T_f is estimated by eqs.(7)~(9), and then the zone of vegetation growth during the base-flow stage is estimated. After the vegetation zone is replaced to the enlarged new situation, the velocity profile of the flow with vegetation zone is calculated. Then, such a process is repeated in simulation.

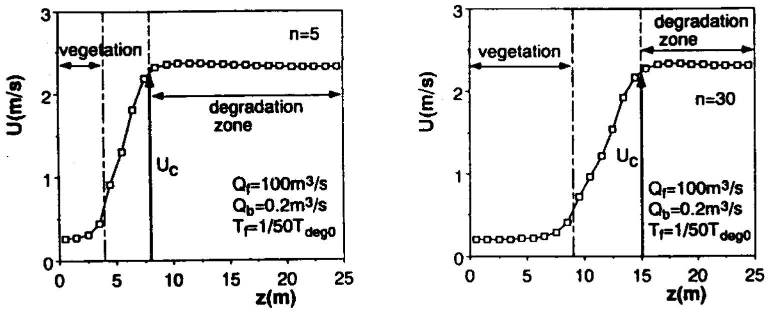

Fig.4: Examples of calculated lateral distribution of primary velocity

RESULTS AND DISCUSSION

Numerical simulation based on the above-explained scenerio is conducted under the condition as follows: L=1000m; $2B_0$=50m; i_{b0}=0.02; d=1.0cm; $C_D\lambda$=0.5cm^{-1}; l=2m; and b_0=2.0m, where $C_D\lambda$ is the parameter that represents the vegetation density and l is the length of vegetation; C_D=drag coefficient of the vegetation element; and λ=vegetation density defined as the projected area by vegetation element per unit mass of water.

Figure 5 shows the changes of the width of vegetation zone b_v and the bed slope i_b with the number of flood experiences. The time is made dimensionless by T_{deg0} which is the time scale of degradation without growth of vegetation and is given as follows: $T_{deg0} \equiv 3L^2/[Q_f/(2B_0)]$. 3 figures are obtained for different scale of the flood time T_f and compared with one another, where the initial channel geometry and the dischrage are constant. The case with the shorter T_f results the wider vegetation zone and the milder slope. This result implies that the time scale of flood is important to the phenomenon.

For the case T_f=1/10T_{deg0}, the equilibrium values ($n \rightarrow \infty$) of the width of vegetation zone and the bed slope are calculated and plotted against the flood discharge Q_f in Fig.6. Three curves in the figures are obtained for different base-flow discharge Q_b. The cases of $Q_f \approx$ 300m^3/s bring the widest vegetated zone. Too small discharge Q_f cannot transport the sediment in the whole cross section, and then the degradation occurs in narrow zone. Too large discharge Q_f results wide zone of degradation, and then the vegetation zone cannot expand so widely. In the case of $Q_f \approx$ 1000m^3/s, the degradation zone occurs in the whole cross section, and thus the vegetation zone no longer expand. These results imply that the optimum condition for the vegetation to expand or not to expand may exist depending on the combination of flood and base-low stage.

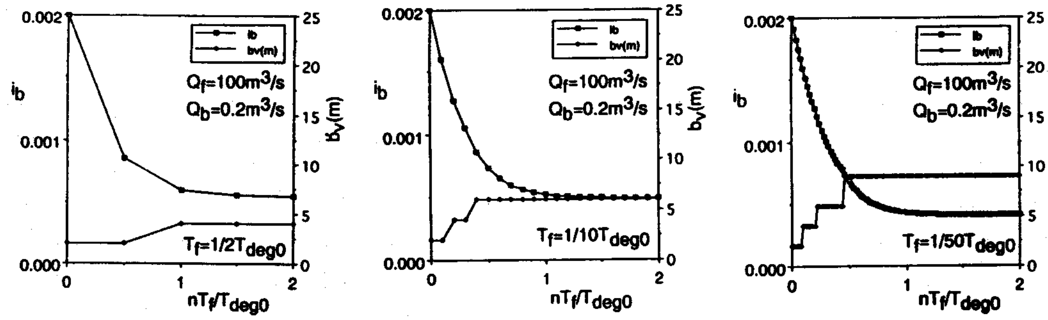

Fig.5: Change of bed slope and width of vegetated zone by experience of floods

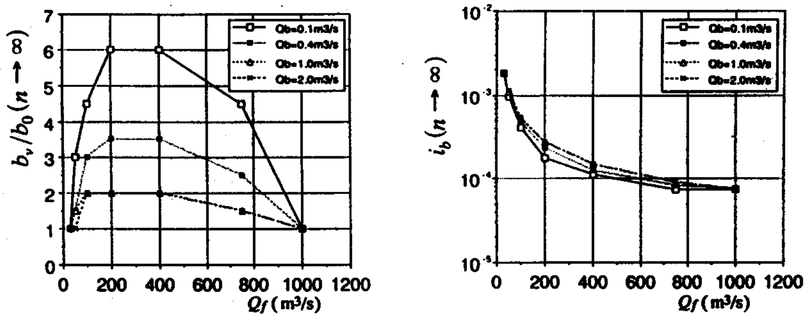

Fig.6: Equilibrium values of width of vegetated zone and bed slope

CONCLUSION

In this study, the river-bed degradation affected by the growth of vegetation, is discusses theoretically. Existence of vegetation zone retards the flow even in the main channel, and the zone of degradation is limited in the central part oc the main channel. During base-flow stage, the flow discharge concentrates the central zone and newly dried zone is invaded by vegetation. Hence, the repetition of flood and base flow stage promotes the narrow zone with severe degradation.

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BASIC MORPHOLOGICAL PROCESS DUE TO DEPOSITION OF SUSPENDED SEDIMENT AFFECTED BY VEGETATION

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ABSTRACT

In this study, the interaction between growth of vegetation and morphological change in a river is studied to develop hydraulic approaches to describe and predict the fluvial process affected by vegetation. Particularly, the repetition of flood and low-stage flow, and the morphological change due to deposition of suspended sediment affected by vegetation is treated.

When there is a vegetated zone in a river, suspended sediment deposits around the vegetated area during flood. After the flood passes and the water stage returns to the lower level, a newly dried zone appears. The vegetation expands its territory to such a new zone, and the growth of vegetation affects also the flow during the next flood. It is investigated how he vegetation zone changes through the repetition of flood.

A laboratory experiment is conducted. Vegetated zone is idealized as the permeable body made by the entwined plastic fibers and the flume bed is assumed to be a rough flat bed. Fine sand is supplied from the upstream end of the flume during a simulated flood. After a flood, the water returns to a low level and vegetation zone are expanded to the zone on which the fine sand deposited. The next simulated flood is introduced to the flume with fine sand supplied from the upstream, and the process is repeated. As a result, the speed of the expansion of the vegetation zone becomes slower with the increase of the vegetation zone. It is related the flow with vegetation zone.

The above-mentioned process is also simulated by using a numerical analysis, where a depth-averaged k-ε model is employed, and a depth-averaged diffusion equation for suspended sediment is solved for the solved flow field. The numerical simulation can explain the flume experiments well, and then the simulation can provide prediction of fluvial process with vegetation growth under various postulated conditions.

KEY-WORDS: river management / fluvial hydraulics / flow with vegetation / sand deposition / suspended sediment / growth vegetation / repetition of flood / fluvial-fan river

INTRODUCTION

Recently, river improvement and management should be proceeded from both view points of environment and flood control. The control of growth of vegetation in a river is a key of the river management. Because vegetation is a kind of inhabitant in some sence and it also provided a habitat for other organisms together with river morphology. A change of vegetation cover in a river causes a change of the habitat. Furthermore, an increase of vegetation cover in a river causes a decrease of hydraulic capacity against a flood, reversely too much vegetation cover in a river is undesirable from the view point of flood control. Hence, an appropriate control of the vegetation should be found.

The authors believe that the deeper understand of a river as a fluvial system in streams with vegetation leads the more appropriate control of the vegetation. The vegetation is related to a river morphology and hydrology. Figure 1 shows the river system with vegetation postulated in this study. The elements of the system are vegetation, river morphology, flow, sediment transport and subsurface flow. A combination of vegetation and morphology decides the flow during a flood. The flow affects the sediment transport, and the morphology is determined by the sediment transport. The flow affects also the vegetation. For example, the vegetation is sometimes destructed by the flowing water. After a flood, the zone on which the vegetation grows might be influenced by the surface and subsurface flow and the deformed morphology of the river. These are an interaction system. While, the discharge of water and sediment supply acts as an environment for this system.

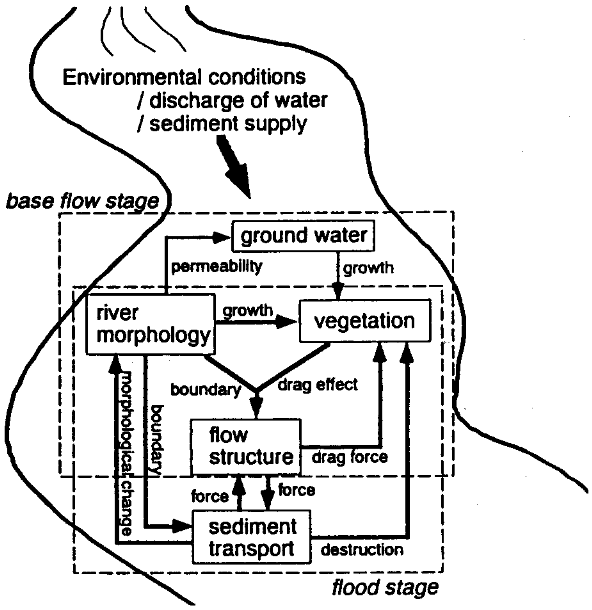

Figure 1: Interrelating system in fluvial process of vegetated river

By the hydraulic method, the authors have been studied the flow and sediment transport in a stream with vegetation as subsystems of the above mentioned system (For example, Tsujimoto and Kitamura, 1994; Tsujimoto and Shimizu, 1994; Murakami *et al.*, 1994). However, these studies hardly treated the feed back mechanism between the morphological change during flood and the growth of vegetation in base flow stage. In this study, the feed back mechanism is focused and studied from the hydraulic view point.

Photo 1 shows the deposition of fine sand around an isolated bush of willow (*Salix gilgianna*) after the flood occurred at 1995 in the River Tedori, Japan. The River Tedori is a typical fluvial-fan river, and then the bed slope and the representative diameter of the bed materials around there are about 1/150 and 10cm respectively. The flood flow contained much wash load as suspended sediment, it hardly deposited in the main channel, but it deposited around the willow bushes, because the willow bushes decreased the flow velocity around it. After the flood, some kinds of vegetation like willow (*Salix gilgianna*) and reed (*Phragmites japonica*) invaded to the zone where the fine sand was deposited. Without damages by flood, these vegetation can grow up to be 1-2m high in summer season and it is big enough to influence the flow in during flood. In the River Tedori, the Tedori-dam was constructed for flood control, water resources utilization and power plant at 1980 in the upstream of the fluvial-fan reach, and the sediment was stopped at the reservoir of this dam. It caused the river bed degradation. In addition, the dam construction decreased the frequency of floods large enough to destruct vegetation, and the vegetation cover became to increase in the river (Tsujimoto, 1994).

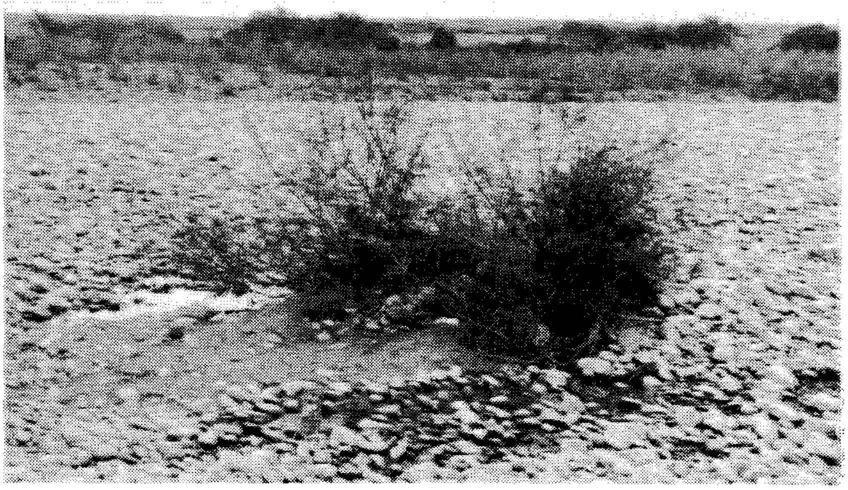

Photo 1: Typical sand deposition around isolated willow bush (The River Tedori, Japan, 4 November, 1995)

In this study, the following simplified situation is discussed. In a gravel-bed river where the bed-load transport hardly occurs to change the primary morphology, but deposition of fine suspended sediment occurs locally and the vegetation invades into the deposited area to increase the vegetated area. After the vegetated area is enlarged, the successive flood is postulated. Such a succession of flood and enlargement of vegetation area is studied through flume experiment and unmerical simulation in order to study how such a phneoema contributes the features of fluvial-fan rivers.

EXPERIMENT

Experimental Set-Up

The experiment was conducted in a straight rectangular-channel, 12m long and 0.5m wide. The gravel of which diameter was 0.22cm was glued on the flat bed, which determined the bed roughness. The permeable material made by the entwined plastic fibers was used as a model of the vegetation. The permeability test clarified the vegetation density, which is defined as a projected area of vegetation elements per unit mass of water λ, as $1.2cm^{-1}$. Such a permeable material was cut in a cube each side of which was 5cm, and it was set at 5.5m upstream from the downstream end of the flume and at the center of the cross section. Figure 2 shows a schematic figure of the flume.

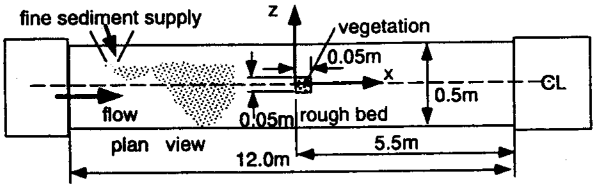

Figure 2: Experimantal set-up

A model flood flow was introduced to the flume, and the water depth was adjusted as uniform flow depth by a wier at the downstream end of the flume. The experimental condition was as follows: The bed slope i_b was 1/1000. The flood discharge Q_f was $5{,}200cm^3/s$ and the water depth h_0 was 4.2cm. The bed shear coefficient C_f defined by $(u_{*0}/U_0)^2$ was 0.0065 in which $u_{*0}\equiv(gh_0i_b)^{1/2}$; g = gravitational acceleration; and U_0 = cross sectional averaged velocity.

The very fine particles made by a vinyl chloride was employed for the suspended sediment. The properties of the materials are as follows: The diameter of the particles d_s is 0.0075cm; the specific weight is 1.3; and the terminal velocity v_0 is 0.09cm/s. These were supplied from the upstream end of the flume. The discharge of the particles Q_s was $1.34cm^3/s$, and the cross-sectionally averaged concentration of the suspended sediment at the upstream end C_0 was $2.6x10^{-4}$. In this experimental condition, the suspended sediment can hardly be deposited except around the vegetation.

Figure 3 shows the shape of the deposition of the sediment after 2400s. The deposition occurred behind the vegetation zone but not in the vegetation zone, and its maximum thickness was 0.35cm.

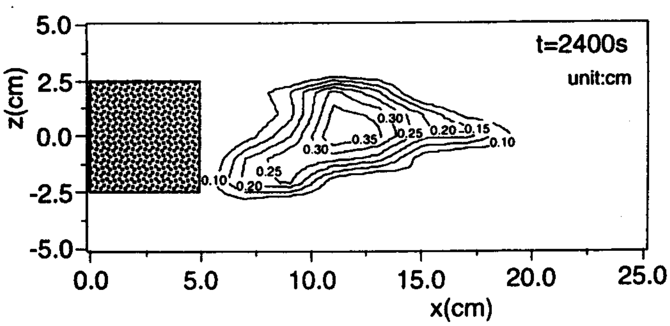

Figure 3: Shape of the fine sand deposition after 2400s

<u>Expansion of Vegetation Zone by Repetition of Flood</u>

We have assumed that the vegetation invades to the deposited area after the flood. For simplicity, we added the permeable material to cover the deposited area. Then, we introduced the flood flow to the flume with the suspended particles again. The water and sediment supply were stopped 1200s after the restart of the experiment. Such procedures were repeated.

Photo 2 shows the expansion process of the vegetation zone after respective floods. The length of the deposition behind the vegetation area becomes shorter with the increase of the length of the vegetation zone as shown in Fig.4.

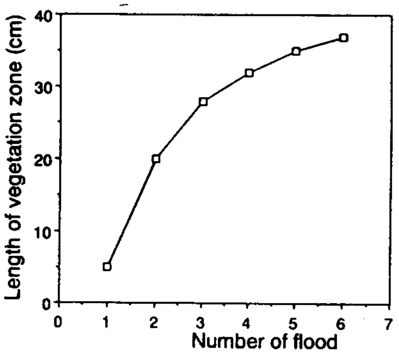

Figure 4: Expansion of vegetation zone with repetition of floods

<u>Velocity around Vegetation Zone</u>

The velocity around the vegetation zone was measured by a propeller current meter (the diameter of the

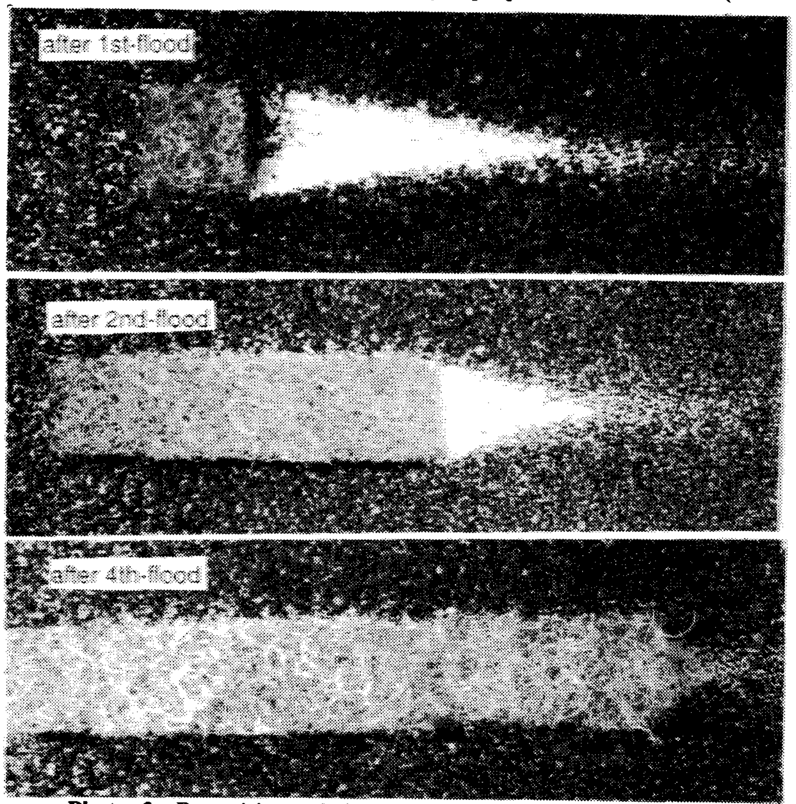

Photo 2: Deposition od fine sediment behind vegetated area

propeller was 3mm). The velocity at the 40% height of the depth and along the center line of the cross section is shown in Fig.5. During the first flood when the length of the vegetation zone is 5cm, the minimum velocity appears behind the vegetation zone, because the length of the vegetated area is too short for the flow velocity to decrease. Hence, the fine materials deposited behind the vegetated area. On the other hand, during the 6th flood, of which the vegetation zone is 37cm in length, the velocity accelerates behind the vegetated area, and thus the fine materials hardly deposit behind there.

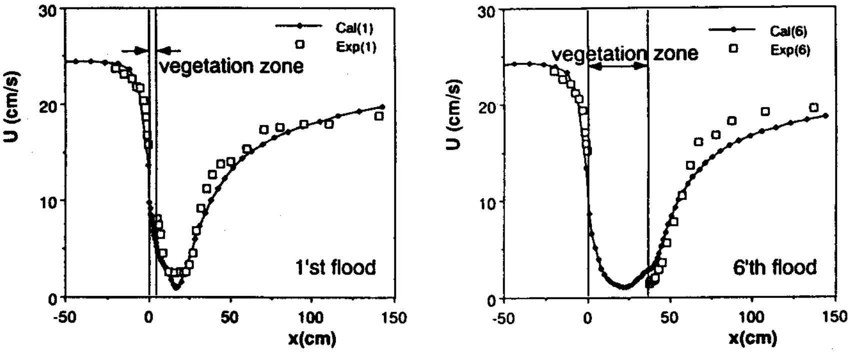

Figure 5: Velocity along the center line around vegetated area

TWO-DIMENSIONAL ANALYSIS OF FLOW WITH VEGETATION ZONE

Governing Equations and Procedure of Calculation

The three-dimensional structure of a flow is dominant around a non-permeable body. However, around the permeable body like vegetation zone, the flow is rather two-dimensional, because the water can flow through the vegetation zone. Therefore, a depth-averaged model of flow is here assumed to analyze the flow with vegetation zone. In this study, we employ the depth-averaged turbulence model based on the Rastogi & Rodi's k-ε model (Rastogi and Rodi, 1978).

The existence of vegetation elements produces not only the drag force in momentum equations but also the additional turbulence energy and its dissipation in k and ε equations. These effects were discussed by Shimizu *et al.* (1991, 1992), Tsujimoto and Shimizu (1994), Shimizu and Tsujimoto (1995). Shimizu *et al.* (1992, 1994) proposed the k-ε model with considering the effect of vegetation for the vertical two-dimensional flow over vegetation-covered bed, and its result is good agreement to the experimental data. In this study, vegetation is assumed to be rigid, and the effect of flexibility is not considered. By the combination of the idea of Shimizu *et al.* (1991, 1992) and Rastogi and Rodi's model (Rastogi and Rodi, 1978), the depth-averaged model for flow with vegetation zone is derived as follows;

(1)
$$\frac{\partial}{\partial x}(hU) + \frac{\partial}{\partial z}(hW) = 0$$

(2)
$$\frac{\partial}{\partial x}\left(hU^2 - h\nu_T\frac{\partial U}{\partial x}\right) + \frac{\partial}{\partial z}\left(hWU - h\nu_T\frac{\partial U}{\partial z}\right) = -gh\frac{\partial(h+y_b)}{\partial x} + \frac{\partial}{\partial x}\left(h\nu_T\frac{\partial U}{\partial x}\right) + \frac{\partial}{\partial z}\left(h\nu_T\frac{\partial W}{\partial x}\right) - F_x h - \tau_{bx} h$$

(3)
$$\frac{\partial}{\partial x}\left(hWU - h\nu_T\frac{\partial W}{\partial x}\right) + \frac{\partial}{\partial z}\left(hW^2 - h\nu_T\frac{\partial W}{\partial z}\right) = -gh\frac{\partial(h+y_b)}{\partial z} + \frac{\partial}{\partial x}\left(h\nu_T\frac{\partial U}{\partial z}\right) + \frac{\partial}{\partial z}\left(h\nu_T\frac{\partial W}{\partial z}\right) - F_z h - \tau_{bz} h$$

(4) $$\frac{\partial}{\partial x}\left(hUk - h\frac{\nu_T}{\sigma_k}\frac{\partial k}{\partial x}\right) + \frac{\partial}{\partial z}\left(hWk - h\frac{\nu_T}{\sigma_k}\frac{\partial k}{\partial z}\right) = (P_k + P_{kv} + P_{kb} - \varepsilon)h$$

(5) $$\frac{\partial}{\partial x}\left(hU\varepsilon - h\frac{\nu_T}{\sigma_\varepsilon}\frac{\partial \varepsilon}{\partial x}\right) + \frac{\partial}{\partial z}\left(hW\varepsilon - h\frac{\nu_T}{\sigma_\varepsilon}\frac{\partial \varepsilon}{\partial z}\right) = C_1 h\frac{\varepsilon}{k}(P_k + C_{v\varepsilon}P_{kv} + C_{b\varepsilon}P_{kb}) - C_2 h\frac{\varepsilon^2}{k}$$

(6) $$\nu_T = C_\mu \frac{k^2}{\varepsilon}$$

(7) $$F_x \equiv \frac{1}{2} C_D \lambda U\sqrt{U^2+W^2} \quad (h<l)\ ; \qquad F_x \equiv \frac{1}{2} C_D \frac{l}{h}\lambda U\sqrt{U^2+W^2} \quad (h>l)$$

(8) $$F_z \equiv \frac{1}{2} C_D \lambda W\sqrt{U^2+W^2} \quad (h<l)\ ; \qquad F_z \equiv \frac{1}{2} C_D \frac{l}{h}\lambda W\sqrt{U^2+W^2} \quad (h>l)$$

(9) $$\tau_{bx} \equiv C_f \frac{1}{h} U\sqrt{U^2+W^2}\ ; \quad \tau_{bz} \equiv C_f \frac{1}{h} W\sqrt{U^2+W^2}$$

(10) $$P_k \equiv \nu_T\left[\, 2\left(\frac{\partial U}{\partial x}\right)^2 + 2\left(\frac{\partial W}{\partial z}\right)^2 + \left(\frac{\partial U}{\partial z} + \frac{\partial W}{\partial x}\right)^2 \right]$$

(11) $$P_{kv} \equiv F_x U + F_z W\ ; \qquad P_{kb} \equiv \tau_{bx} U + \tau_{bz} W$$

where (x, z)=longitudinal and transverse axes; (U, W)=x-,z-components of depth-averaged velocity; h=water depth; y_b=elevation of bed; g=gravitational acceleration; ν_T=kinematic depth-averaged eddy viscosity; k= depth-averaged turbulence energy; ε=depth-averaged dissipation rate of turbulence energy; F_x, F_z=x-, z-components of depth-averaged drag force due to vegetation element per unit mass of water; τ_{bx}, τ_{bz}=x-, z-components of shear stress due to bed per unit mass of water; P_k=depth-averaged energy production due to shear in x-z plane; P_{kv}=depth-averaged energy production due to vegetation element; P_{kb}=energy production due to bed shear; C_D=drag coefficient of vegetation element; λ=vegetation density defined as the projected area of vegetation element to the flow per unit volume of water; C_f=coefficient of bed shear; and $C_\mu, \sigma_k, \sigma_\varepsilon, C_1, C_2, C_{v\varepsilon}, C_{b\varepsilon}$=empirical parameters.

The empirical parameters involved in the governing equations were determined by adopting the recommended values except $C_{v\varepsilon}, C_{b\varepsilon}$ as follows; C_μ=0.09; σ_k=1.0; σ_ε=1.3; C_1=1.44; and C_2=1.92. $C_{b\varepsilon}$ is given by Rastogi and Rodi' study (1978) as follows;

(12) $$C_{b\varepsilon} = 3.6\frac{C_2}{C_1}\frac{k}{\varepsilon}C_\mu^{1/2}C_f^{1/4}\frac{\sqrt{U^2+W^2}}{h}$$

$C_{v\varepsilon}$ that is introduced for taking account of the drag effect of vegetation is determined by comparing the flume data of the statistical properties of turbulence of open-channel flow over vegetated bed with the results of numerical analysis based on k-ε model (Shimizu *et al.*, 1991, 1992), as follows: $C_{v\varepsilon}$=1.3.

The governing equations are discretized by the staggered grid, and the SIMPLE algorithm by Patanker (1980) is employed. The boundary condition at rigid boundaries is given by the wall function. At the upstream boundary of the calculation, the discharge is given At the downstream boundary, the water depth is given.

Calculated Results

Figure 6 shows the calculated contour of depth-averaged velocity U. The velocity behind the vegetation zone is retarded but the velocity never becomes zero or negative behind the vegetation zone.

The change of the velocity along the center line is shown in Fig.5 with the experimental data, and it shows a good agreement with the experimental data. The calculation can describe the following properties of the flow with isolated vegetation zone : 1) At the first flood, of which the vegetation zone is too short with downstream direction for the velocity to decrease, the minimum velocity never appears in the vegetation zone and the velocity continues to decrease with downstream direction even behind the vegetation zone. 2) At the 6th flood, of which the vegetation zone is too long for the velocity to decrease, the minimum velocity appears in the vegetation zone, and the velocity accelerates behind the vegetation zone.

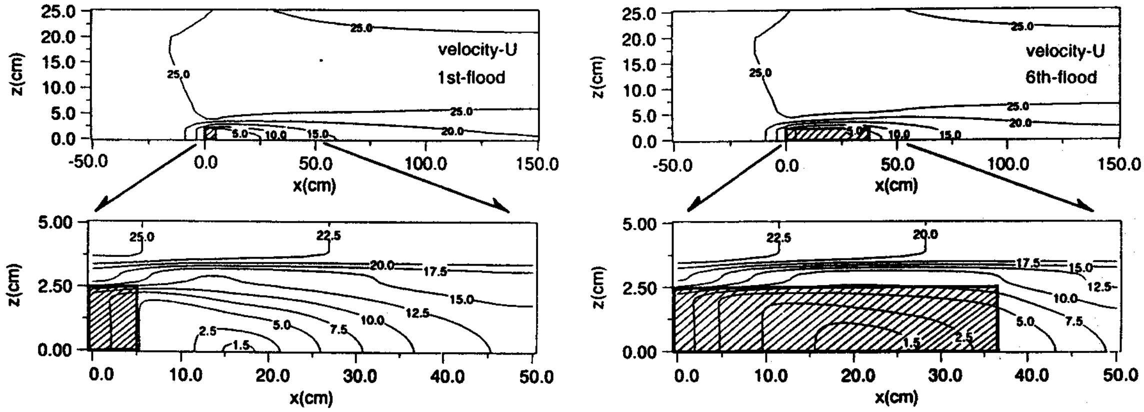

Figure 6: Calculated flow patterns

TWO-DIMENSIONAL ANALYSIS OF SUSPENDED SEDIMENT TRANSPORT

Governing Equations and Procedure of Calculation

The governing equation of suspended sediment transport is as follows;

(13) $$\frac{\partial}{\partial x}\left(hUC - h\frac{\nu_T}{\sigma_C}\frac{\partial C}{\partial x}\right) + \frac{\partial}{\partial z}\left(hWC - h\frac{\nu_T}{\sigma_C}\frac{\partial C}{\partial z}\right) = S_C$$

where C=depth-averaged concentration of suspended sediment; $1/\sigma_C$=turbulent Schmit number; and S_C= production term of suspended sediment. When river-bed is fixed like the experiment, S_C is given as follows;

(14) $$S_C = \nu_0 C_{ae} - \gamma\nu_0 C \quad (\nu_0 C_{ae} < \gamma\nu_0 C); \qquad S_C = 0 \quad (\nu_0 C_{ae} > \gamma\nu_0 C)$$

where ν_0=terminal velocity of sand; C_{ae} = bottom concentration under equilibrium condition; and γ= coefficient for the vertical distribution of concentration of suspended sediment. The terminal velocity is given by Rubey's formula (1933) as follows;

(15)
$$\frac{v_0}{\sqrt{(\sigma/\rho-1)gd_s}} = \sqrt{\frac{2}{3}+\frac{36}{d_s^*}} + \sqrt{\frac{36}{d_s^*}}$$

where σ, ρ =mass densities of sand and water; d_s=sand diameter; $d_s^* \equiv (\sigma/\rho-1)gd_s^3/\nu^2$; and ν=kinematic viscosity. The equilibrium bottom concentration C_{ae} is related to the shear velocity u_*. After the investigation of the previous data, the following empirical formula (Tsujimoto, 1992b) is employed here.

(16)
$$C_{ae} = 0.002\left(\frac{u_*}{v_0}\right)^2\left(1-\frac{\tau_{*c}}{\tau_*}\right)^{1.5}$$

where $\tau_* \equiv u_*^2/[(\sigma/\rho-1)gd_s]$; and τ_{*c} =dimensionless critical tractive force. The shear velocity is given by the calculated result of flow as follows;

(17)
$$u_* = \sqrt{C_f(U^2+W^2)}$$

When the vertical distribution of concentration is assumed to be the Lane-Kalinske's equation (1941), γ is expressed as follows;

(18)
$$\gamma \approx \frac{15v_0}{u_*}$$

The production term S_c in the conservation equation of concentration of suspended sediment represents also net deposition rate of sediment to the bed. Deformation rate of bed elevation y_b is expressed as follows;

(19)
$$\frac{\partial y_b}{\partial t} = -\frac{S_c}{(1-\rho_0)}$$

The governing equation is discretized in the same control volume to the analysis of flow. The numerical analysis of flow is conducted in advance, and then its results for flow velocity (U,W) and the kinematic eddy viscosity ν_T are employed in calculation of suspended sediment behavior. The boundary condition at the rigid boundary is that the flux is zero. At the upstream end, the concentration of suspended sediment C_0 is given for the boundary condition. For simplicity, it is assumed that $1/\sigma_c$ is constant and equal to 1.0.

<u>Calculated Result</u>

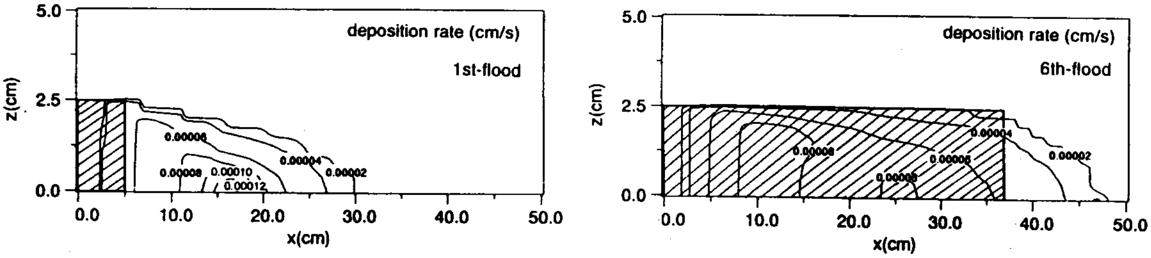

Figure 7: Calculated net deposition rate

The calculated results of the net deposition rate for the first flood and the 6th flood are shown in Fig.7. The calculated zones of deposition show good agreements to the experimental results. The deposited area recognized in the flume experiment corresponds to the region where the net deposition rate is larger than about 6.0×10^{-5}cm/s in the present calculation.

SIMULATION OF EXPANSION OF VEGETATION ZONE

In the present experiment, we gave arbitrary time scale of flood and we reset the vegetation zone on the all deposition zone after flood without consideration on the time scale for the vegetation growth. However, in real rivers, various combination of the time scales are expected. Meanwhile, such time scales are even artificially controlled by dam and reservoirs some how. Furthremore, the base-flow condition might influence the vegetation growth appreciably. From such a view point, a simulation based on the present numerical scheme has been conducted.

The examples of the results of the simulation are shown in Fig.8. Most of the conditions are same to the experiment, and the time scale of the flood is 2400s in the simulation. But the base-flow depth has been changed. The case with lower base-flow stage (h_b=0.1cm) has higher speed of expansion of the vegetation zone than that with higher base-flow stage (h_b=0.3cm).

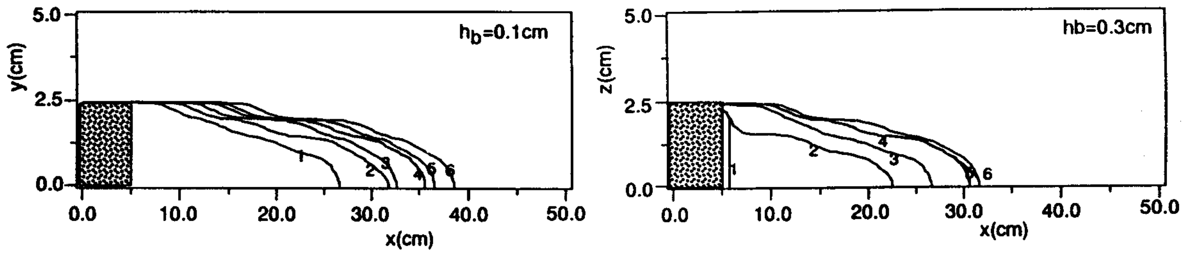

Figure 8: Simulation of expansion of vegetation zone

CONCLUSION

By flume experiment and numerical simulation, the expansion process of the isolated vegetation zone through the repetition of flood has been discussed. The experimental results have demonstrated the following features: Sand deposits behind an isolated bush and the vegetated area can expand behind itself. However, it cannot expand so much. The reason why the length of such bushes are finite is as follows: when the length of vegetation zone is too long for the velocity to decrease within the vegetation zone, the velocity accelerates behind the vegetation zone, and it causes that sand hardly deposit behind the vegetated area. These properties are explained by numerical simulation. The results of the numerical simulation show good agreements with the experimental results, and the simulation is available to know the basis of the interaction between growth of vegetation and morphological change due to flood flow.

The situations treated in this study are very simple, though a real river is more complicated. Especially morphology in a river has several patterns, for example, a dune, a bar, a meander and so on. The authors will develop the study about the interaction between these complicated morphology and growth of vegetation, on the basis of the method in this study, with field survey.

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Ecohydrology

Écohydrologie

IMPORTANCE OF HYDROLOGICAL MODELS IN HABITAT ISSUES : COMPARISON BETWEEN A STOCHASTIC AND A DETERMINISTIC AT CATAMARAN BROOK (NEW-BRUNSWICK)

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ABSTRACT

Many physical variables affect fish habitat in streams. Among them, stream discharge has been known to be an important parameter in many abiotic and biotic processes. Hydrological models are often used either independently, or as components of water quality models, for modelling physical processes which affect habitat quality or quantity. Two models were use to simulate stream discharge at Catamaran Brook (New-Brunswick), a third order stream on an experimental drainage basin. The first model was a deterministic distributed hydrological model called CEQUEAU. The second model was a stochastic model using a Neuronal Network Theory approach.

Model simulations of stream discharge were compared to measured values using two numerical criteria, the correlation coefficient and the Nash coefficient. Simulations performed on two years of data from Catamaran Brook showed that the CEQUEAU model out performed the Neuronal Network. The model outputs were analyzed in terms of potential ecological implications. Spring floods, as well as summer and winter baseflow events were better simulated by the CEQUEAU model.

KEY-WORDS: Hydrological models /deterministic/neuronal network/CEQUEAU/discharge/stream/ habitat/ fish/ Catamaran Brook.

1. INTRODUCTION

Stream discharge has been known to be one of the most important parameter in many abiotic and biotic processes of river systems. For instance, stream discharge is of major influence in habitat modelling (Bovee 1982). Stream discharge determines important habitat attributes such as habitat area and volume, water depth and velocity, substrate composition, channel geomorphology and others. Hydrological modelling is often required in the simulation of many chemical parameters (e.g. pH, conductivity) as well as modelling many physical processes (e.g. sediment transport) affecting fish population. One of the main concern in many ecological studies, is a lack of good stream discharge data as most study rivers are ungauged. In view of this, the major challenge still facing aquatic habitat scientists is to improve our ability to predict and manage anthropogenic effects as well as understanding natural processes. In order to achieve this goal, hydrological models have become important tools in the management and understanding of fish habitat. These models are primarily used by managers and scientists for two reasons: First, they permit the simulation of the natural variability of the hydrological conditions for a given ecosystem. Secondly, by varying input variables or model parameters, anthropogenic effects can be simulated.

Two major categories of hydrological model exist, namely the deterministic and the stochastic models. Deterministic models attempt to represent the mechanisms of the physical phenomenon studied. By contrast, stochastic models relate the estimated variable (discharge in this case) to physical parameters considered significant with statistical relationships (Morin and Couillard, 1990). Typically, the choice between a stochastic or deterministic model has often been dictated by the available data. Marceau et al. (1986) state that the deterministic models often require a greater number of input variables while stochastic models require long time series of fewer variables.

In the case of experimental drainage basins such as Catamaran Brook, there are sufficient meteorological data and time series of several months are available (Cunjak et al., 1993). In such cases, researchers and habitat managers have the luxury of being able to compare various models and select one which is best suited to answer the scientific question being asked and in particular pertaining to habitat studies.

In New-Brunswick, timber harvest has a major influence on many lotic and lentic ecosystems. The Catamaran Brook Habitat Research Project is a long term project which aims at investigating and quantifying fish habitat and production and their variations, both natural and man-made. This 15 year programme proposes to quantify habitat changes before, during and after timber harvest at Catamaran Brook, a third order stream with a drainage area of 50 km^2 (Cunjak et al. 1990).

A number of studies (e.g. Campbell and Doeg, 1989) have suggested that timber harvest may have significant impact on stream discharge. These changes depend on many variables, including the size of the logging effort, the methods used for timber harvest, and the stream-side treatment (Hartman and Scrivener, 1990). Campbell and Doeg (1989) state that the influence of catchment vegetation on discharge is related to many variables which are often site specific (slope, soil type, etc.). However, some trends can be asserted from the many studies carried out in the past. In general, the main impact of removing forest vegetation is to decrease evapotranspiration, thereby increasing stream runoff. Over the years, it is often found that this effect slowly diminishes as regrowth takes place.

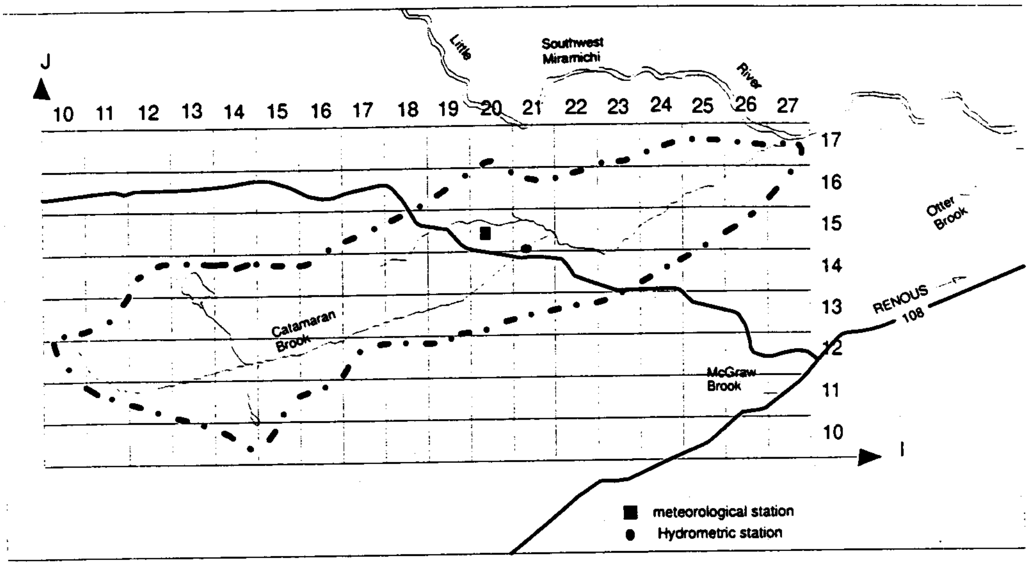

Figure 1. Catamaran Brook Drainage Basin with Whole Squares

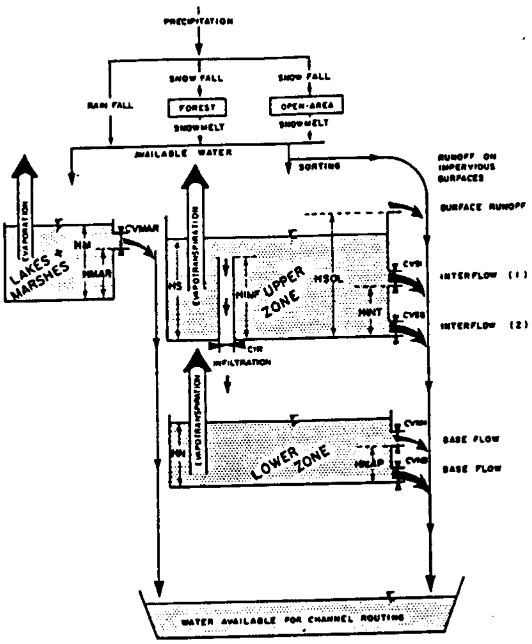

Figure 2: Production Function schematic for the CEQUEAU Model

The objective of this study is to compare two hydrological models used at Catamaran Brook to simulate daily discharge during the first two years of the program. Ecological implications of streamflow variability will also be discussed pertaining to the Atlantic salmon (*Salmo salar*) populations.

2. MATERIAL AND METHODS

2.1 Description of Models

2.1.1 Deterministic model

The deterministic model used is the CEQUEAU hydrological model (Morin et al., 1995; Morin and Couillard, 1990). It is a distributed model which takes into account the physical characteristics of the basin. The basin is subdivided into surface elements called "whole squares" by superimposing a square grid on a map of the drainage basin area (Figure 1). Each whole square is further divided into so-called "partial squares", in accordance with the topography (sub-basin divides). Routing of water can then be calculated from one square to the next, thereby enabling the user to follow the evolution of streamflow in time and space. Since CEQUEAU is a distributed model, discharge can be calculated at any point of the drainage network.

The input data required for the CEQUEAU model are precipitations (solid and liquid) as well as air temperatures (daily maximum and minimum). The meteorological data are interpolated for each whole square from the available meteorological stations on the basin.

Water movement on the basin is modelled using two functions: the production function and the transfer function. The production function simulates the vertical movement of the water, from when it falls on the ground to the moment it is routed in the river. Processes included in the production function include formation and melt of the snowpack, evaporation and evapotranspiration, water volumes in lakes and marshes and water in the various soil layers. The CEQUEAU model represents the ground as connected reservoirs (Figure 2). Hydrological balance is calculated on a daily basis using a series of mathematical relations to simulated the interconnections between reservoirs. Equation (1) represents the water-balance equation calculated on each whole square :

$$(1) \qquad Q_j = P_j - ET_j + (HS_j - HS_{j-1}) + (HN_j - HN_{j-1})$$

where Q_j = Flow (mm) for day j
P_j = liquid precipitation or snowmelt (mm) for day j
ET_j = evapotranspiration (mm) for day j
HS_j = water accumulated in the upper reservoir (mm) for day j
HN_j = water accumulated in the lower reservoir (mm) for day j

The second function, the so-called "transfer function" deals with routing the flow from upstream to downstream. A routing coefficient is calculated for each square, related to the hydraulic characteristics of the flow within its boundaries and to the storage capacity of the drainage network. The routing coefficient of each partial square is determined by Equation (2) (see Morin and Couillard, 1990 for details).

(2) $XKT_i = 1 - \exp((EXXKT \times SA_j / SL_i) \times (100/CEKM2))$

where: XKT = daily routing coefficient of the partial square i
EXXKT= fitting parameter whose value is determined by trial and error
SA_j = area of the basin upstream of the partial square i (km^2)
SL = area of surficial waters on the partial square i (km^2)
CEKM2 = area of the whole squares (km^2)

Model constants (i.e. values are determined by the physical characteristics of the drainage basin) and parameters (i.e. values derived from the physical characteristics of a phenomenon or by trial and error) are described in details in Morin and Couillard (1990). Adjustment and calibration is performed in two steps. First, parameters are adjusted by trial and error, while ensuring that all parameters are adjusted within a realistic range of values. Finally, an optimization routine which seeks to minimised the absolute difference between observed and calculated discharge, given a realistic range for each parameter.

2.2.2 Stochastic Model

The stochastic model applied on Catamaran brook is a relatively new technique derived from electrical engineering and artificial intelligence theory called neuronal networks (NeuralWare inc. 1991). It was applied by Bastarache (1994). More specifically, the neural computing technique applied here is called back-propagation and is based on the same principles as the electrical engineers' feedback loop (Rumelhart et al. 1986).

A back-propagating neuronal network is composed of the input signals, the input layer, a number of "hidden layers" and an output layer. Each layer consists of a series of nodes. The input signals are first transmitted to the nodes of the input layer where they are normalized . The input nodes then transmit the normalized input to the nodes of the hidden layer. A weight is applied to each signal received by a hidden node and the sum of all processed signals is calculated. A transfer function is applied to the signal at each node and the transformed output is passed to the next layer. The output layer produces the last summation and applies the last transfer function. The output is then compared to the input signal and adjustments are made to each weights of each node in order to minimize the error between input and output, up to a given criterion. This phase, which corresponds to the model calibration, is called the learning or training phase for neuronal networks (Marchand, 1990).

Bastarache (1994) gives a detailed description of the algorithm used for this application. Equations (3) and (4) summarize the mathematical procedures of neuronal networks. For each node j, the transfer function used to calculate the output at that node is a non-linear function:

(3) $$O_j = f(S_j + \theta_j) = \frac{1}{1 + e^{-(S_j + \theta_j)}}$$

Where θ_i = threshold at node i
f = transfer function
S_j = sum of inputs at node j

The threshold ensures that an output other than 1 will exists for each node, even in the case where equation (3) produces a value of 0. For each pattern (series of inputs), a final error is calculated for all output nodes. The error is calculated as the sum of the square of the difference between the real value and the output value. Weights w_{ij} are therefore modified to minimize the error. The correction Δw_{ij} applied to the weight w_{ij} is proportional to the change in error (E_p) with respect to w_{ij}. (equation 3).

(4) $$\Delta w_{ij} = -\eta \frac{\delta Ep}{\delta w_{ij}}$$

The partial derivative are developed for both the input layer and the hidden layer (see Bastarache, 1994 for details).

2.2.3 Model Inputs

For the neuronal network , the number of input variables is defined by the user. Thus two different patterns were attempted. In both cases, the inputs used in the CEQUEAU hydrological model (maximum and minimum air temperatures, solid and liquid precipitations) were also used in the neuronal approach. In an attempt to improve the results of the neuronal network, other input variables were added. They are summarized in table 1.

Table 1: Input variables for two patterns used in the neuronal network application
(from Bastarache, 1994)

Pattern number	**Input variables**
1	Maximum and minimum daily air temperatures, solid and liquid precipitations Maximum, minimum and mean temperature for 3 day periods Maximum, minimum and mean liquid precipitation for 3 day periods Maximum , minimum and mean solid precipitation for 3 day periods
2	Maximum and minimum daily air temperatures, solid and liquid precipitations Total liquid precipitation for 2 day , 3 day, 4 day and 5 day periods

One meteorological station exists on the Catamaran Brook drainage basin. It is located in the middle of the basin (Figure 1). The time series of daily air temperatures (maximum and minimum) and precipitations for 1990 and 1991 were used as model inputs for both approaches. Daily discharge was also required by both models for calibration. A hydrometric station (01BP002), operated by Environment Canada, is located at mid-basin. The gauge consists of a stilling well connected the stream via a pipe. Water level is measured with a pressure transducer. Hourly discharge is then obtained using a pre-calibrated rating curve (Cunjak et al. 1993). Daily means are calculated and the time series for 1990 and 1991 are used to calibrate both models.

3.0 RESULTS

3.1 Model Performance

In order to enable the user to judge the results objectively and compare the performance of both models, two numerical criteria were used. The correlation coefficient and the Nash coefficient, given in equation (5) were calculated for the results of 1990 and 1991. The correlation coefficient measures the linear dependence between calculated and observed discharge. A value of 1 indicates perfect agreement. The Nash coefficient represents the ratio of residual variance to the variance calculated from observed discharge. N has a value of 1 when observed and calculated values are identical. It decreases with an increase in the difference between observed and calculated values (Morin et al., 1995).

$$N = 1 - \frac{\sum (Q_{ci} - Q_{oi})^2}{\sum (Q_{ci} - Q_{mo})^2} \qquad (5)$$

where Qci and Qoi = Calculated and observed daily discharge
Qmo = mean observed daily discharge

Table 2 gives the Nash and correlation coefficients for results produced with CEQUEAU and both patterns used in the Neuronal Networks. Bastarache (1994) had attempted 7 different patterns. The two patterns showed here are the ones with the highest Nash and Correlation coefficients. Average annual calculated discharges are also given in table 2 for comparison with measured annual means.

Table 2. Average Discharge, Nash and Correlation Coefficients for both models.

	1990			1991		
Model	Nash	Correlation	Q_{mc} (m^3s^{-1})	Nash	Correlation	Q_{mc}(m^3s^{-1})
CEQUEAU	0.73	0.80	0.86	0.87	0.91	0.80
Pattern 1	0.39	0.73	0.54	0.25	0.66	0.45
Pattern 2	0.66	0.81	0.78	0.66	0.85	0.88
Q_{mo}(m^3s^{-1})			0.82			0.83

3.2 Deterministic Model

Figures 3 and 4 show the time series for both observed and calculated daily discharge for the period of 1990-1991. In general, observed and calculated values follow the same annual trend. In the summer of 1990, measured discharge peaked twice. On July 25 (day 205, Figure 3) and on 12 August (day 223, Figure 3). Observed and calculated values were similar in the first case (Qo = 2.07 m^3s^{-1} and Qc = 2.23 m^3s^{-1}) . In the second case, there was an important discrepancy (Qo = 3.64 m^3s^{-1} and Qc = 0.84 m^3s^{-1}). A third important rainfall occurred on October 24 (day 297, figure 3). The observed discharge peaked at 5.32 m^3s^{-1} while the simulated value was

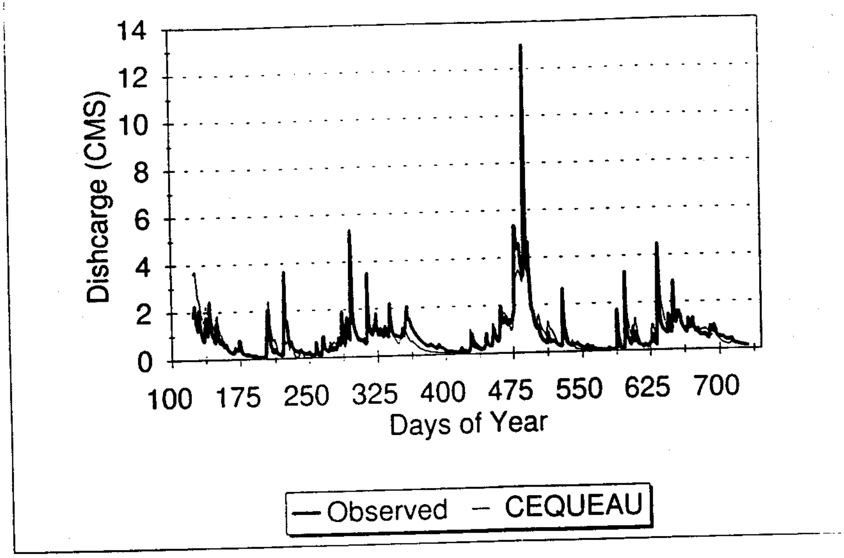

Figure 3. Comparison between measured discharge and discharge simulated by CEQUEAU, 1990-1991.

2.89 m^3s^{-1}. In 1991, the most important hydrological event shown is the spring snowmelt. Discharge values reached a maximum of 13 m^3s^{-1} on May 3 (day 488, Figure 3) while the simulated maximum on the same date was 10.86 m^3s^{-1}. The mean annual discharge was overestimated by 4.8 % in 1990 and underestimated by 3.6% in 1991.

Of equal ecological importance are the base flow value and duration during the winter and summer low discharges. Generally, low discharge values were well simulated by the CEQUEAU model. There were 3 exceptions in September of 1990 where simulated values overshoot the measured base flow (days 244 to 265, Figure 3). The base flow was well simulated in 1991 for winter low flows (day 416, Figure 3). Similarly in the summer of the same year, the base flow was also close to observed values (e.g. day 600, Figure 3).

3.3 Stochastic Model

3.3.1 Pattern1

Figures 4 gives results for the Neuronal Network as applied by Bastarache (1994) with the input data shown in Table 1 for pattern 1. In general, the model tends to overestimate minor events. However, one major event in October of 1990 (day 297 on Figure 4) was well simulated. The observed discharge, Q_o, was 5.32 m^3s^{-1} while the calculated discharge Q_c with pattern 1 was 5.96 m^3s^{-1}. Figure 4 show that the model with pattern 1 tends to underestimate winter base flows. In 1991, relatively poor agreement was observed between calculated and observed flow during the winter and spring. However, simulations improve after day 566 (Figure 4). Overall, the mean annual discharge was underestimated by more than 38% in 1990 and 1991.

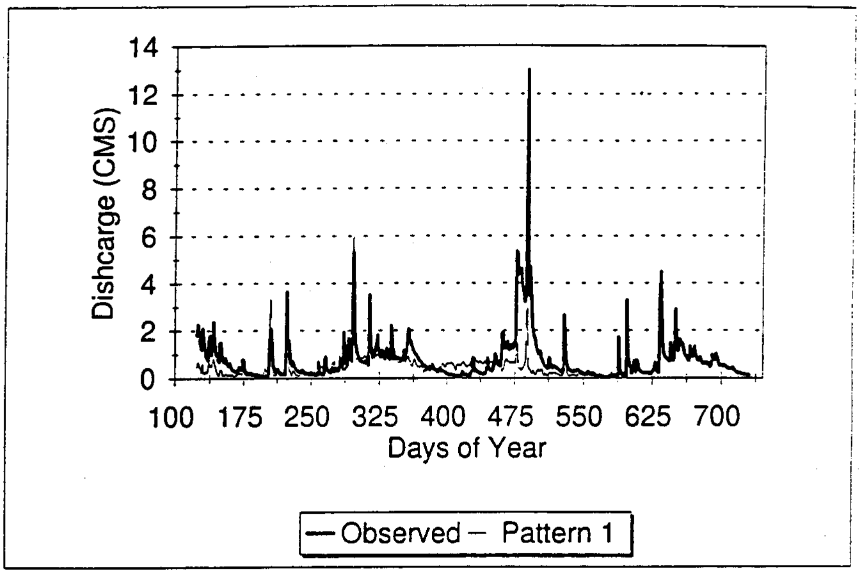

Figure 4. Comparison between measured discharge
and discharge simulated by Neuronal Network, pattern1, 1990-9

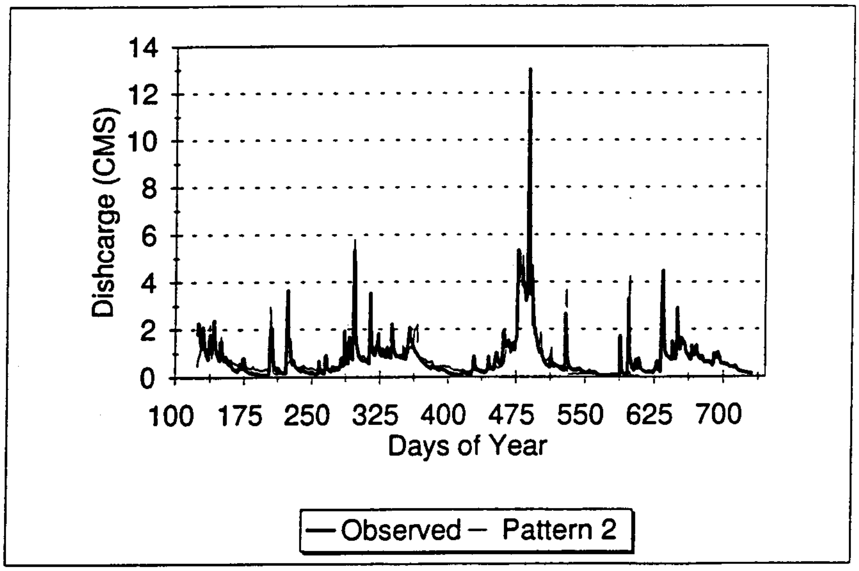

Figure 5. Comparison between measured discharge
and discharge simulated by Neuronal Network, pattern2, 1990-9

3.3.2 Pattern 2

Figure 5 gives results for the Neuronal Network applied with the input data shown in Table 1 for pattern 2. Table 2 shows that this pattern gave better results than pattern 1 (Nash = 0.66 and R > 0.8 for 1990 and 1991). The mean annual discharge was underestimated by 4.8% in 1990 and overestimated by 6% in 1991. In general, the simulated base flow was overestimated but the major events (spring flood and major fall events) were well reproduced by the model. One event in 1990 is significantly underestimated by the Network (day 224, Figure 5; $Q_o = 3.64\ m^3s^{-1}$, $Q_c = 0.54\ m^3s^{-1}$).

4. DISCUSSION AND CONCLUSION

4.1 Model Comparison

Based on the numerical criteria shown in Table 2, the CEQUEAU model had the best overall performance. The deterministic model was best able to simulated the most significant flood of the time series (spring snowmelt of 1991). The simulation of other important events varied in quality, but overall, the deterministic model was superior.

As expected, the variation of inputs for the Neuronal Network has a very significant impact on its learning ability. The difference in Nash and correlation coefficients merely indicate that the variance in discharge is better explained by total precipitation than by maximum and mean precipitations. It should be pointed out, however, that there is a human factor involved with the implementation of the Neuronal Network. It can be argued that a relatively objective choice in model inputs can be done, based on known hydrological characteristics of the basin. However, the number of nodes and hidden layers, as well as the type of transfer function chosen are all subjective and are function of the experience of the user. Thus, the results shown here could be improved with more trials (Bastarache, Université de Moncton, per. comm.).

4.2 Ecological implications

Ecological implications of streamflow variability on aquatic resources is of importance as hydrological events such as floods have been identified as having important impacts on fish (Elwood and Waters 1969). The present study showed that both high and low flow events can be simulated with a good degree of agreement to observed flows. Such hydrological modelling approach is very important in ecological studies to assess anthropogenic impact and also in studies (e.g. biological, habitat) where discharges are needed but not available from a stream gauge station.

Low and high flow conditions were well simulated by the hydrological models (more so with the CEQUEAU model). Low flows have shown to affect fish movement and stream water temperature (Cunjak et al. 1993; Edward et al. 1979). Such low flow and high flow conditions seem to have a greater impact on fish resources than do average conditions as reported in the literature. For instance, there is evidence that winter floods kill young-of-the-year fish as noted by corpses and significantly reduced summer electrofishing success for that age class (Erman et al., 1988). Erman et al. (1988) found that the dominant factor in fish mortality during floods was the large-scale bed load movement. At Catamaran Brook, the flood of 1991 of over 13 m^3s^{-1} (figure 3) showed no clear evidence of greater reduction in fish population than other years (Caissie 1995).

Summer low flows can affect fish populations as the low water conditions can be associated with high stream water temperatures and lower concentration of dissolved oxygen (DO). Another potential impact of low water condition is a reduction of fish habitat. Also, migration of spawners can be affected during autumn by obstacles (beaver dams, culverts, natural falls) that may prevent upstream fish movement at low flow. For instance, many spawners at Catamaran Brook were limited to the first km of a 20 km stream during some years as a result of a beaver dam and low water conditions on the main stem of the brook (R.A. Cunjak, per. comm.).

In general, fish habitat (defined in terms of Weighed Usable Area) availability at Catamaran brook increase from a very low value at low flows to a maximum value near mean annual flow and decrease at higher flow (Caissie, 1995). It is therefore critical that the model used to simulate discharge behaves at its best during extreme events. All models seem to simulate most extreme events in 1990 appropriately. The spring high flow of 1991, however, is significantly underestimated by both patterns used in the Neuronal Network model. This could be related to a lock of consideration of snow melt processes. In contrast, CEQUEAU performed well in spring snowmelt periods. In ecosystem studies, it is important that the flow variability be well simulated (i.e rising and falling limbs of important events), as disturbance and flow variability interact to provide a physical template that influences not only selected species, but community patterns of stream organisms (Poff and Ward 1989). The degree at which models simulate this variability was quantified by the Nash coefficient. For both 1990 and 1991, the CEQUEAU hydrological model had significantly higher Nash coefficients than the Neuronal Network.

Data at Catamaran brook on fish densities and stream discharge suggested that winter low flow conditions could be as important as summer and fall low flows. Low flow conditions in combination with cold winter have presumably affected salmonid populations, as reported by Chadwick (1982). Egg mortality could have resulted from freezing of redds and low water may limit movement of fish. It can thus be seen that proper modelling of low flow is as important as flood simulations. In modelling winter and summer low flows, the CEQUEAU model out performed the Neuronal Network (figures 3 to 5).

For hydrological model applications, It should be stressed that the alteration in stream discharge caused by timber harvest is associated with important changes in the hydrological pathways (Hartman and Scrivener, 1990). With such human intervention, increased suspended sediment concentrations can be a factor affecting fish during floods and breakups. In temperate forests, overland flow is rare as most of the precipitation percolates through the soil and reaches the stream either as interflow or groundwater. The soil compaction associated with the use of heavy machinery prevents the natural percolation of precipitation and surface runoff becomes more important (Walker, 1986).

In conclusion, it is the collection of long-term physical, chemical and biological data in multidisciplinary ecosystem studies such as Catamaran Brook that we will allow to better understand the different process affecting fish habitat. Studies of change in fish population in relation to environmental factors such as the hydrological regime will help to identify key components that can be considered as limiting factors and therefore enable us to better estimate the carrying capacity of streams. In light of the potential changes that may be caused by timber harvest on the hydrological regime, it becomes important to attempt to quantify the natural variability of that regime. The CEQUEAU hydrological model appears best suited to achieve this goal.

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LIKELIHOOD EXPRESSION OF HYDROLOGICAL STRUCTURING EVENTS FOR BROWN TROUT POPULATION DYNAMICS : METHODOLOGICAL APPROACH

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ABSTRACT

The Continous Over(Under) Threshold procedure was developed by Capra *et al.* (1995a) to analyse in a practical way cumulative temporal effects of unhabitual habitat or flow conditions on fish populations. This procedure is sustained by the basic feeling that aquatic organisms loss or gain energy in time depending on local aquatic habitat conditions they are exposed to. It is an adaptation of the flow-duration-frequency curves procedure for regional floods analysis (Oberlin *et al.*, 1989). Analysis of durations and frequencies of the aquatic habitat conditions would provide some temporal stream behavior characteristics to explain fish population dynamics. Fish species are presumed to be more exposed to 'temporal flow bottlenecks' than benthic species wich are ***a priori*** more dependant of local characteristics like subtrate composition wich could also vary in time.

Capra *et al.* (1995a) observed that aquatic habitat temporal variability was correlated to brown trout population dynamics. A main result concluded that continous durations for underthresold dayly discharges during adult brown trout life stage were significantly related to low yearly cohort densities. The ten years observation period (with seven yearly biological sample for adult brown trout) led to interpret identified hydrological events as short term effect structuring conditions. It means with no dramatic effect on trout population. The question rised that such events could regulate population dynamics on a long term.

The objective of this communication is to verify a such hypothesis. We try to quantify limiting hydrological events likelihood to compare them with brown trout life cycle, especially for adult life stage in the three studied streams.

Depending on natural hydrological regimes, likelihood could vary for a set of trout streams and then could be more or less relevant to explain a long term control. For that reason we selected a total of fourteen disharge stations to delineate a likelihood boundary for hydrological events of interest.

KEY-WORDS: Continous Under Threshold Discharge Duration (CUTDD) / limiting flow conditions / hydrological event likelihood / coupled (biological-hydrological) data / discharge station /

INTRODUCTION

Main ecological theories like 'habitat templet concept' (Southwood, 1977), 'intermediate disturbance hypothesis' (Ward and Stanford, 1983) include temporal variability of environmental conditions as a major structuring factor for aquatic species. Such theories become from large scale biological observations (*i.e.* global balance) but are quite difficult to verify at local scale due to a general poor biological data availibility in time (Statzner and Resh, 1993). Regional hydrology is based on large scale hydrometric networks and provides some operational stockastic models to detemine flow characteristics anywhere along rivers. However, these characteristics are mainly devoted to human activities and some developments are required for ecological objectives (Breil and Malafosse,1993), (Breil and Malafosse, 1994). Both regional and local ecological scales are irelevant for hydraulics works management. An intermediate scale must be described, combining regional framework (as reference situation) and local functional relationships between flow fluctuations and stream ecological response. This is a multi-disciplinary work because stream ecology, hydraulics, hydrology and fluvial morphology are concerned (Hérouin *et al.*, 1995). At a local scale, physical habitat modeling methods are based on instream hydrodynamic characteristics wich are fairly dependant on flow fluctuations into running waters. Thus, we could use local scale physical-biological data to define some relevant temporal flow characteristics for an ecological river management objective.

On a practical aspect, it needs a set of coupled biological-discharge data in time. Time duration and biological data sample frequency depend on the 'biological question' to analyse in term of population dynamics (discharge recording is assumed to be continous). For example, bentic aquatic species needs a 'hight frequency' sampling effort in comparison to fish life cycle duration. A biological question defined, likelihood identification of potential hydrological controlling effects seems possible.

MATERIALS

Backround results

Continous Under Threshold Discharge Duration (CUTDD) variables are defined as durations, along a discharge time - series, where observed discharges are continuously under a predefined discharge threshold. Using this procedure, Capra (1995a) studied several discharge thresholds. At a discharge threshold corresponding to the average dayly discharge plus one standard deviation a significant linear relationship was pointed out between relative adult brown trout (*Salmo trutta*) cohort densities and relative calculated CUTDD. Figure 1 shows this result. Cohort densities and CUTDD were scaled dividing sampled values by their average to give relative values. Maxima CUTDD for sampled years are mainly linked to low cohort densities. All CUTDD belonging to low right quarter of the figure are identified as limiting hydrological events linked to continous 'low discharges'. We could note that low densities could also occur for low CUTDD. In that case, it means that uncontrolled factors could also influence trout population dynamics.

As a basic hypothesis in this study, we refer to the functional observed relationships between CUTDD and trout adult cohort density established for three trout streams, sampled from 1985 to 1995. These streams belonged to the set of fourteen studied streams of this paper and they were numbered 1 to 3.

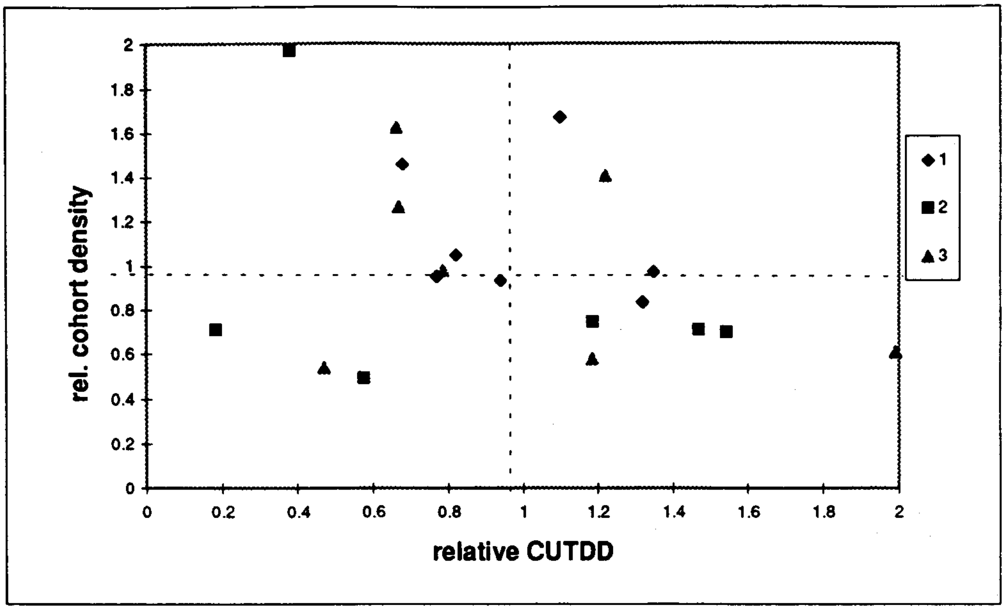


Figure 1: Oberved relationships between relative brown trout cohort density and relative CUTDD for three trout streams (after Capra *et al.*, 1995a).

Hydrological Data

Table 1: Specifications for selected set of dayly discharge recording stations.

stream	Discharge station	station number	Watershed area (sq. km)	Time series duration (years)	Discharge (meam+sd) threshold (m3/s)	country
Semine	Coz (6.6)	1	45	1959-1995	3.48	France
Boralde de St Chely	Castelnau de Mandailles	2	53	1961-1995	3.35	France
Boralde de Bonneval	Bonneval	3	85	1961-1995	5.38	France
La Souloise	St Etienne en Dévoluy	4	42	1977-1995	0.97	France
La Dordogne	St Sauves d'Auvergne	5	87	1929-1995	7.72	France
Le Gand	Neaux	6	85	1973-1995	1.45	France
Le Jabron	Souspierre	7	85	1967-1983	1.15	France
Le Meaudret	Meaudre	8	44	1970-1995	1.58	France
Le Risse	St Jeoire	9	58	1974-1990	4.38	France
La Roizonne	La Rochette	10	72	1920-1995	5.13	France
L'Isère	Val d'Isère	11	46	1948-1988	3.92	France
La Vence	Pt de l'Oule	12	64	1963-1993	3.04	France
Hoeggaas bru	124.2.0.1001.1	13	491	1912-1993	47.1	Norway
Tangfoss (regulated)	124.3.0.1001.1	14	591	1933-1991	45.58	Norway

A station number was attribute to each one, from one to fourteen. We will refer to this numbering scheme in the rest of the study. All stations have a minimum of ten years dayly discharge observation with possible breaks. As defined above, discharge threshold indicates value of average dayly discharge plus one standard deviation. Streams 1,2 and 3, used by Capra, are in shaded areas. Dayly discharge time series for station 1 was reduced by a rate of 6.6 corresponding to watershed areas rate between Coz discharge station and Capra's biological station. We could note that watershed areas are quite similar and around 70 squared kilometers, excepted for norvegian stations.They are all natural streams excepted last one, wich is regulated, and are classified into three major hydrological regimes as shown in figure 2. These annual monthly discharge distributions were calculated grouping similary stations as indicated in legend of figure 2 (Norvegian monthly discharge values were diveded by ten to be scale in this figure). Dashed line shape corresponds to a rainy hydrological cycle with hight waters on february-march and low waters on july-august period. Full line shape indicates combination between rain and, for a little part, of snow melt with almost same periods for hight and low flows. Snow melt contibutes to sustain low flows. Dotted line shape is quite different in hight and low flow distribution along a year. There is a major snow melt effect following by ice melt over period from march to august. Combination of water temperature and hydrodynamics seasonal characteristics (flow velocity, water depth,...) influence life strategy for aquatic species. Each hydrological regime could then have an effect on life trout cycle.

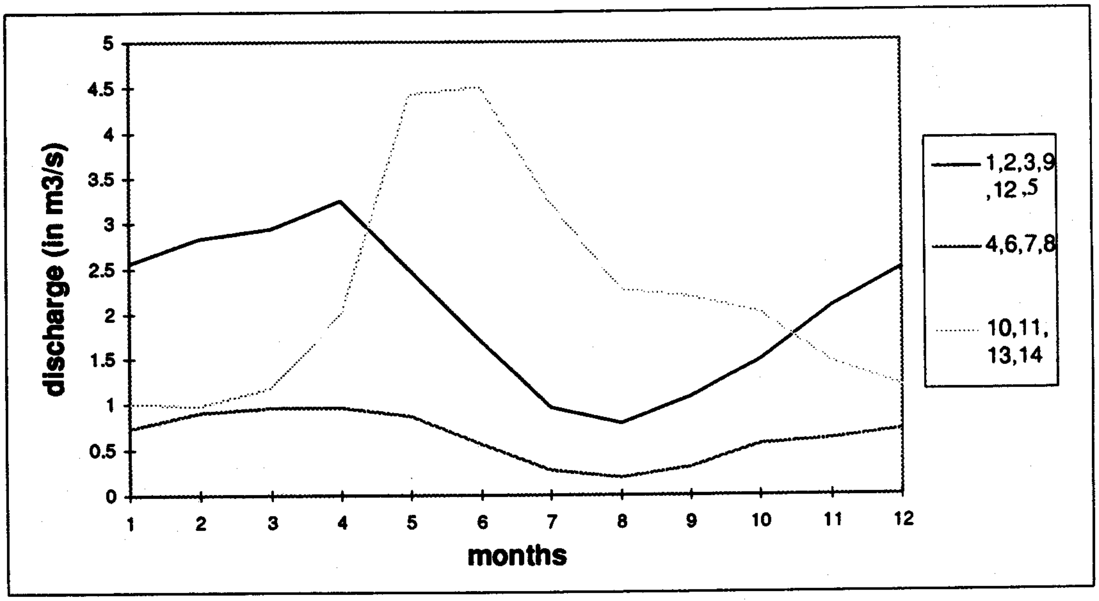


Figure 2: Main shapes of annual monthly discharge distributions in the selected river set

METHOD

CUTDDx Definition

A discharge threshold of average value plus one standard deviation cannot be considered representative for low flow conditions but obviously integrates all small 'flash floods' during these periods. Such floods are often resulting from local rainfall contributions to a natural stream netwnork. A reduced reach length could be concerned and no sensible biological impact would then be expected. Furthermore, under these presumable limiting conditions, local events will probably not provided a substantial increase in trophic ressource, *i.e.* energy gain. Maxima CUTDDs could then be interpreted as a measure of cumulative stress for brown trout. In the future, it seems of interest to resume maxima CUTDDs in terms of managing discharge-characteristics rather than into durations, however we before have to assess relevancy of maxima CUTDDs on long term brown trout population dynamics.

A long term survey means that hydrological events of interest are quite frequent, regarding to a life trout cycle duration, wich is 3 to 4 years long in studied streams. Accordingly, they have no dramatic effects on trout population. In that sense we prefer to speak of 'limiting hydrological' events rather than 'hydrological disturbance'. In Capra's study maximum CUTDD were defined to be greater than their average value calculated from small sample sizes (seven values by station). A more precise and unbiased definition is required for larger sample s. CUTDDx are then defined as maxima CUTDD in a frequency domain of one to a few years. A one year threshold was retain on the assumption that frequent hydrological events could not have a long term limiting effect on fish population dynamics.

CUTDDx Average Return Period (ARP) Calculation

As hydrological cycles are yearly cycles and trout life cycle is a minimum of 3 years duration, a compromise is to express CUTDDx likelihoods in terms of average return periods (ARP) in year unit. After samples composition by station, each sample is sorted in a decreasing order and then ARP values are associated to CUTDD values. Calculation of ARP is simply computed by dividing time series duration TSD (expressed in year unit) by its CUTDD value rank order (RO) (1).

(1) $$ARP = TSD / RO$$

ARP is an average value. Due to rank orders and time series durations, ARP value could be fractionnal. A one an half ARP value doesn't mean more than the associated CUTDD value was reach or overpassed two times into three years but maybe in the two first years, or maybe years one and three, or maybe in the same year. As defined, only CUTDD with a ARP value greater than one will be analyse. A formal mathematical expression leads to :

(2) CUTDDx(ARP) with ARP greater than one year

Sampling Methods

As seen above in the hydrological regimes description, CUTDDx temporal location will sensibly differ depending on stream hydrological type. These seasonal effects may interfer with trout cycle, by the way of trophic ressource availibility, life stages and so on. For adult brown trout and rainy flow regimes, season of interest will range from end of february to end october. For snow melt flow regimes, season will range between the end of June to the end of March. However, yearly conditions must be consider for other purposes, like young life stage wich contributes, one or two years after, to adult cohort density. For this reason, we also sampled one maximum CUTDD by year. However, a maximum could overpass a year duration. At least, CUTDDx could not exist during wetted years. This consideration leads to a third sample method, without delimited season to sample.

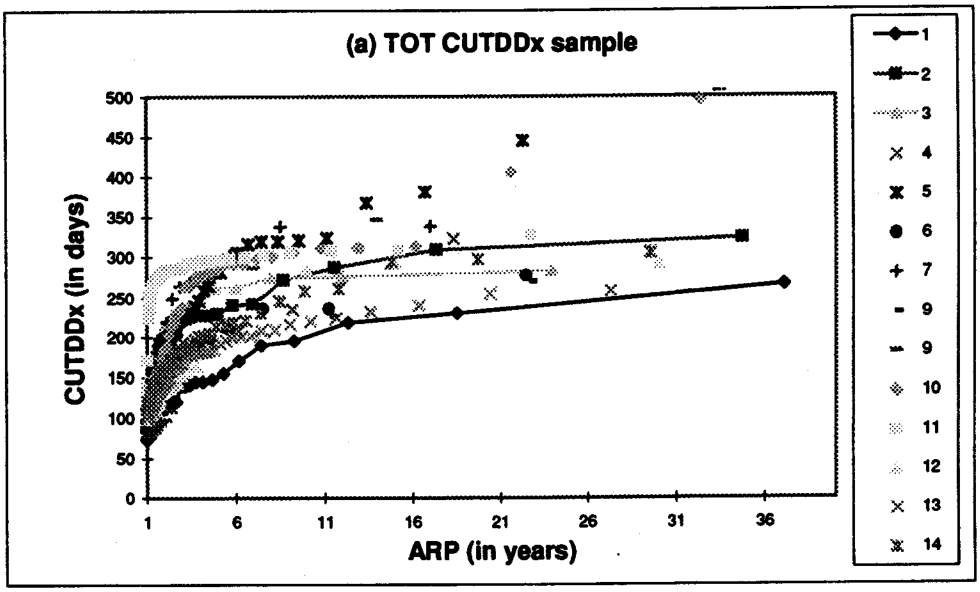
Hence, three sampling methods for CUTDDx were used : First one is called TOT (TOTal sampling method) where all CUTDD are sampled from a time series. Second one is called ANU (ANnUal sampling method) where each yearly maximum CUTDD is sampled each year of the survey. Third one is called SEA (SEAsonal sampling method) where each yearly seasonal maximum CUTDD is sampled. In the aim to compare CUTDDx likelihood from station 1,2 and 3 with Capra's results, we took, as defined above, same season limits from end of february to end of october.

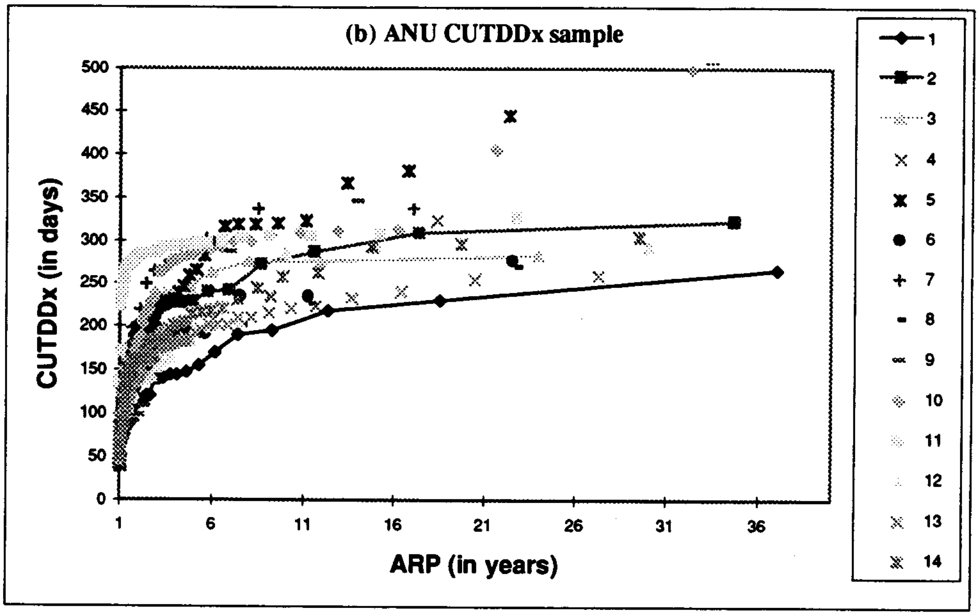
The TOT method leads to sample maxima observed CUTDDx. The ANU sample contains every lived years by fishes. The SEA sample contains every lived season of interest by fishes. Depending on the season duration, maxima values will be quite similar into the three samples for a giving discharge time series but, as said above, could differ in the frequency domain of interest for population dynamics.

RESULTS AND DISCUSSION

CUTDDx Likelihood Representative Domain

Figure 3 (a,b,c) shows CUTDDx versus ARP values. There is one graphic by sampling method. A maximum ARP value of 37 years was retain to keep ARP domain of interest in the graphics. Results for stations 1,2 and 3 are pointed out using full lines.





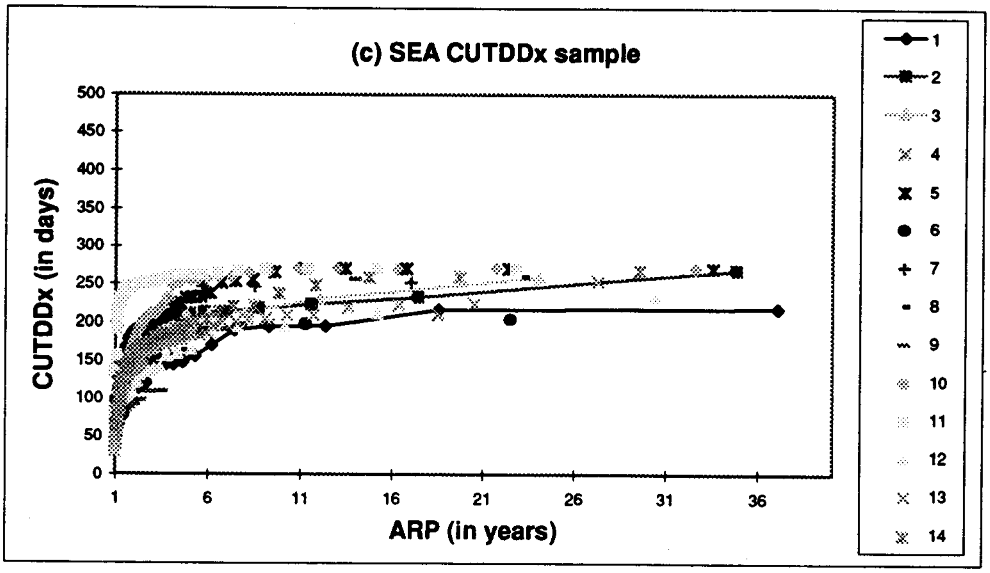


Figure 3: Average Return Period (ARP) values with three CUTDDx sampling methods (a, b, c) applied to fourteen representative trout streams. (stations 1, 2 and 3 in full lines)

Three sampling methods used indicate that CUTDDx increased mainly between ARP values from 1 to 6 years. Over this limit, we could note asymptotic limits depending on hydrological regime characteristics by stations. In SEA method, CUTDDx are limited by season duration wich is 270 days long. Asymptotic trends in natural processes are well understand. However, in some cases, another processes could take place, like for stations 5 and 10. For trout population dynamics, it seems obvious that unfrequent but not rare events could play a role on a long term. A

CUTDDx occurrence of one to six years seems in accordance to trout life cycle duration. Consequently, we will focus our analysis on ARP domain from one to six years.

<u>On Discharge Threshold Relevancy</u>

Relevancy of experimental threshold from Capra's study could be assessed as follow : (1) on one hand, a lower discharge threshold of average dayly discharge (for example) would lead to a sample of short durations with a reduce increment in CUTDDx values and then a limited range. It means, that differences of one or two days in CUTDDx would have sensible effects on trout population dynamics. A such hypothesis seems irrelevant for fish; (2) On the other hand, a higher discharge threshold of one average plus two standard deviations (for example) would lead to a sample of long durations in the case of TOT and ANU sampling methods (with SEA method, season duration would be often sampled without interest for likelihood attribution). Sample size would then be limited by event realisations. In other words, long limiting durations are too rare to have a long term struturing effect on trout population dynamics. As shown in figure 3a, b, and c, norvegian stations are inner the whole range of sampled CUTDDx. Watershed areas for these stations are about 500 squared kilometres and 8 times greater than other watershed areas. Thus, a threshold of average dayly discharge plus one standard deviation is a 'stable' information for our application.

<u>Sampling Methods And a Biological Purpose</u>

Sampled CUTDDx by SEA method (figure 3c) are limited to 270 days long. CUTDDx could also be cutted by season limits. This sampling method is used by Capra and must be compared to the two others in term of sample temporal representativity for the biological question of interest. CUTDDx values from TOT and ANU (figure 3a and 3b) methods are similar. They only differ by their lower CUTDDx values because TOT method only keep maximum values over the whole time series rather than one maximum value each year in ANU method. However, ANU method seems more adapted for biological purposes because each year is "lived" (for flood risk calculation, TOT method is used to fit a probabilistic law), and cumulative effects could occur over few years.

Comparing sampled values between three methods, we could note that SEA method reduced, as expected, larger CUTDDx values. However (and consequently), season duration has a limited effect in ARP range from 1 to 6 years. Question is now to know if trout activities required more or a minimum of energy (i.e. food, temperature,...) during a particular season and in accordance with annual flow regime distributions in streams of interest.

<u>CUTDDx Domains Of Variation</u>

Figure 4 is a plot of CUTDDx (6 years) minus CUTDDx (1 year) versus CUTDDx (1 year). It shows how range of limiting durations, for all stations and three sampling methods used, broadly decreases with CUTDDx (1) increase. Limiting durations could vary roundly from 100 to 250 days with a one year ARP value from 30 to 130 days. CUTDDx domains for each method is explain as follow. TOT sampling method leads to lower range of CUTDDx variations because only years of maximum are sampled. In that case, CUTDDx(1), for each station, is highter than in the two other sampling methods. In SEA method, sampled CUTDDx can be cutted by season limits and are then shorter than CUTDDx from ANU method.

Always in figure 4, dashed lines between points give positions for stations 1 to 3. It shows that stations 2 and 3 are in the boudary limits of selected set of stations. Station 1 is in the lower part of the domain. Such a result could be explained by a 'rougth' reduction of dayly dicharges by a rate of 6.6 as explained in hydrological data presentation. However, discharge fluctuations globaly reduce dowstream. In that case, 'transfered CUTDDx' must be enlarged !. However it must be verify by a temporary discharge monitoring in the future.

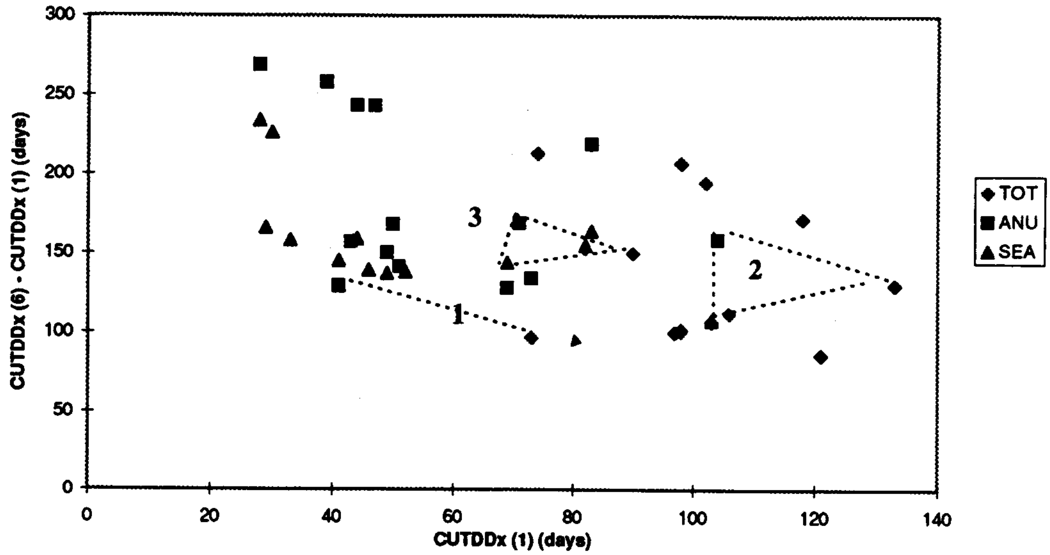


Figure 4: One to six years CUTDDx domain versus one year CUTDDx values, using TOT, ANU and SEA sampling methods

CUTDDx Likelihood for stations number 1 to 3

Likelihood for observed CUTDDx on stations number 1, 2 and 3 are reported in table 2. These CUTDDx are limited to fish sample dates, it means to lived limiting conditions in year of observation. Mean values are presented to resume influence of sampling method on likelihood determination.

Table 2: ARP and mean values for observed CUTDDx on streams 1 to 3

	station number 1				station number 2				station number 3			
	CUTDDx days	TOT years	ANU years	SEA years	CUTDDx days	TOT years	ANU years	SEA years	CUTDDx days	TOT years	ANU years	SEA years
observed	86	1.28	1.43	1.48	211	2.48	2.67	4.98	211	1.97	2.15	6
observed	75	1.03	1.24	1.32	179	1.93	2.04	2.05	182	1.48	1.58	1.69
observed	67		1.1	1.17	53				164	1.35	1.44	1.51
mean val.	**76.00**	**1.16**	**1.26**	**1.32**	**147.67**	**2.21**	**2.36**	**3.52**	**185.67**	**1.60**	**1.72**	**3.07**
CUTDDx(1)		**days 73**	**days 41**	**days 41**		**days 90**	**days 71**	**days 69**		**days 133**	**days 104**	**days 103**

Mean durations for observed CUTDDx increase from station 1 to station 3. This is not due to a sampling fluctuation effect but rather to consistent likelihood distributions as shown in figure 3. Note that CUTDDx likelihood attribution is not link to duration characteristics. CUTDDx(1 year) are indicated to show why some ARP values were not calculated for some observed CUTDDx wich were under CUTDDx(1). We could also note that mean likelihood values are under 1.4 years for station 1, and over 1.6 for stations 2 and 3. As discussed above, station one is a particular site with 'tranfered CUTDDx'. It could explain that CUTDDx ARP seems too frequent to be consider as sensible structuring events for trout population dynamics. Now, considering stations 2 and 3 we could say that all sampling methods give reasonable ARP results. They must be discussed in the ligth of long biological time series if available on other sites.

<u>About Spatial CUTDDx Representativity</u>

For regulated streams, where trouts are under severe hydropeaking conditions, Valentin *et al.* (1995) concluded that channel morphology (wide and deep profiles) and refuges could play a major role during peak flows. Such morphological characteristics are obviously not described by dischrage data. Long term fish population dynamics also results from refuge availibility under natural unsteady flow conditions. However, under CUTDDx conditions, limiting factors are not high flow gradient and velocity but 'low flow' durations. In that case, a discharge station could bring information on trout population dynamics upper and downstream of its position.

<u>About Temporal CUTDDx Representativity</u>

Norvegian stations 13 and 14 are sub-watersheds of Stjoerdal river system wich is about 2110 squared kilometers. Its regulated area coresponds to station 14 and station 13 is in the natural part of the whole watershed. We could note on figure 3 that CUTDDx for station 14 are always upper than CUTDDx for station 13. Is this a result of water management ? What are implications on trout population dynamics? This is a concrete study case wich could bring relevant informations in the future, using contrasted situations as made by Capra *et al* (1995b).

CONCLUSION

Main result of this work concems methodological approach to include a continuous temporal and functional relationship to interpret fish population dynamics. Use of discharge time series is quite simple, however a real field of research exist in statistical sampling method wich need to be adapted to research of limiting events for biological systems. This is a fundamental condition to generalize statistical relationships using probabilistic laws and then to characterize stream behaviours. Future studies will have to focus on these aspects, but it requires coupled data linking precise biological questions to discharge time series.

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Fish overwintering

Hivernage des poissons

STRUCTURE OF TURBULENT FLOW IN AN ICE-COVERED TIDAL ENVIRONMENT

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ABSTRACT

Simultaneous measurements of instantaneous velocities (longitudinal and vertical) were performed during a complete tidal cycle (from low to low tide) in the estuary of an ice-covered tidal river: Sainte-Anne river. This river is the world most important spawning site of the Atlantic tomcod (*Microgadus tomcod*). Spawning period is concentrated during the month of January. An array of four ECM's was positionned near the bed and under the ice respectively to permit the description and comparison of the turbulent characteristics of both boundary layers. The relative thickness of each boundary layer and the influence of bed roughness on the flow turbulent structure are both related to the variations in water level induced by tides. During flood tide, the presence of the ice cover enhances turbulence intensity in both layers and causes an increase in vertical exchanges. The addition of another turbulence generating boundary mixed with an increase of longitudinal and vertical velocities during flood flow entail important changes in the turbulent characteristics of the whole flow. Despite the important changes in turbulent flow structure linked to the addition of the ice cover, the migration behaviour of tomcods is mostly a function of the longitudinal velocity direction. Tomcods migration to their spawning site is concentrated during flood flow, when the flow reversal is important and generates rapid upstream velocities.

KEY-WORDS: river / turbulent boundary layer / turbulence intensity / ice cover / tidal regime / estuary / fish habitat

INTRODUCTION

Average flow characteristics under an ice cover have been studied by several researchers in the natural environment (Carey, 1966 and 1967; Larsen, 1969; Ashton, 1971; Tsang and Szucs, 1972; Engmann and Kellerhals, 1974; Lawson et al., 1986), in the laboratory (Gogus and Tatinclaux, 1980; Zufelt and Zhao-Chu, 1988), and through the development of numerical models (Carstens, 1968; Shen and Harden, 1978; Lau and Krishnappan, 1981; Chee and Haggag, 1982). In the presence of an ice cover, the flow is separated in two upside down boundary layers. The turbulent structure of one layer is controlled by the roughness of the bed while the other is controlled by the roughness of the ice. Transverse mixing and vertical exchanges are limited by the presence of an ice cover because velocity and shear stress distributions are modified by the addition of a boundary at the surface (Engmann and Kellerhals, 1974; Shen and Harden, 1978). However, no studies have directly investigated the structure of turbulent flow in an ice-covered river.

Turbulent flow characteristics modification induced by tides in estuarine environments have also been studied (West and Shiono, 1985 and 1988; West and Oduyemi, 1989; Kawanisi and Yokosi, 1993). During spring tides, longitudinal velocities (u) are higher at flood flow than during ebb flow. Root mean squared longitudinal velocities (u') are up to 9 times higher during flood flow than during ebb flow. Values of u' decrease over the flow depth while v' values increase with distance from the bed (Kawanisi and Yokosi, 1993). Vertical exchanges are then enhanced away from the bed. Turbulence intensities are higher during flood flow for both velocity components. However, turbulence intensities for the longitudinal velocity ($u'/\bar{u}$) is larger than that of the vertical component ($v'/\bar{u}$) (West and Shiono, 1988; West and Oduyemi, 1989). The turbulence intensity ratio (v'/u') ranges from 0.3 to 0.8 (West and Shiono, 1988; Kawanisi and Yokosi, 1993). The originality of the present paper lies in the addition of the ice boundary to the tidal conditions.

The primary objective of this paper is to describe the turbulent structure of the flow at one site in a tidal ice-covered river. The second objective is to infer from the turbulent flow structure potential effects on the migration behaviour of the Atlantic tomcod (*Microgadus tomcod*). Tomcods migrate through the estuary to reach their spawning site located approximately 10 km upstream from the river mouth. In fact, Sainte-Anne river is the world most important spawning site of the tomcod.

METHODS

Turbulence Data

The Sainte-Anne river is a north shore tributary of the Saint Lawrence river located 80 km west of Québec City (figure 1). The measurement site is located in the estuarine part of the river, 1.4 km upsteam of the river mouth. This portion of the channel is subject to semi-diurnal tides. In this area, the bed material of the Sainte-Anne is mainly composed of medium sand and the bed is fairly uniform with occasionnal ripples or organic debris. At the time of data collection (March 1994, spring tides period), the ice cover was 0.8 m thick and presented an hydraulically smooth surface to the flow. The water level varied from 1 m at low water to 2.25 m at high water during the study period.

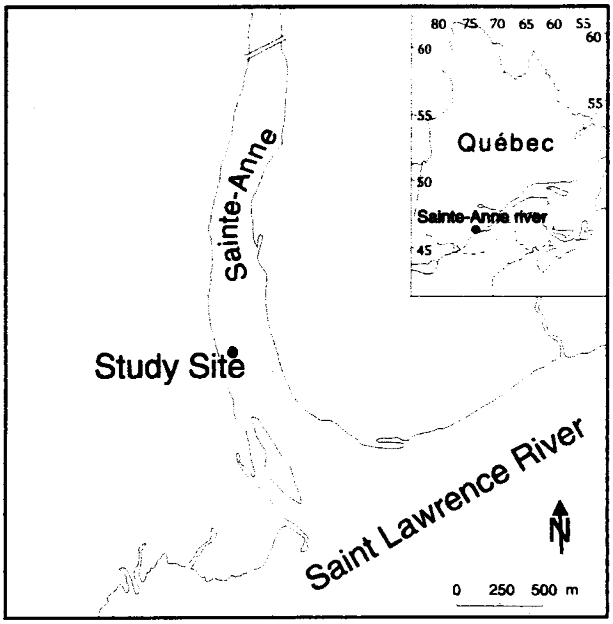

Figure 1: Location of the study site

An array of four Marsh McBirney bidirectional ECM's including three small sensors (1.28 cm in diameter, model number 523) and a large one (3.8 cm in diameter, model number 512) was used to collect the velocity data. The electronics of the four ECM's are linked through a master/slave configuration which allows in phase measurements of the velocity. Longitudinal (u) and vertical (v) velocity components were measured simultaneously. The ECM's were mounted at a 0.25 m interval on an ABS tube that was moved and tightened on a vertical wooden rod of 2.4 m long (figure 2). Two positions were monitored. At the bottom position, we recorded data concerning the boundary layer associated with the bed. In this case, the lowest sensor was positionned 5 cm away from the bed. The top position was located inside the ice boundary layer. Because the sensors are fragile, the top one was located at 10 cm under the ice cover to avoid damaging it. This distance between the sensor and the ice was conditionned by the very rapid downward changes in water level during the ebb tide.

Sixty-four series of measurements were collected at both positions, covering 14 hours over one tidal cycle from low to low tide. Data sets were taken at 15 minute intervals during flood tide and at 30 minute intervals during ebb tide. The choice of these two intervals is dependent upon the rate of change in water level due to tidal phases. The sampling frequency was 20 Hz and the record length 110 s. Because the instruments have a time constant of 0.05 s, the half-power frequency is 3.18 Hz. Thus, sampling at 20 Hz did not entail aliasing in the signal (Roy et al., in review).

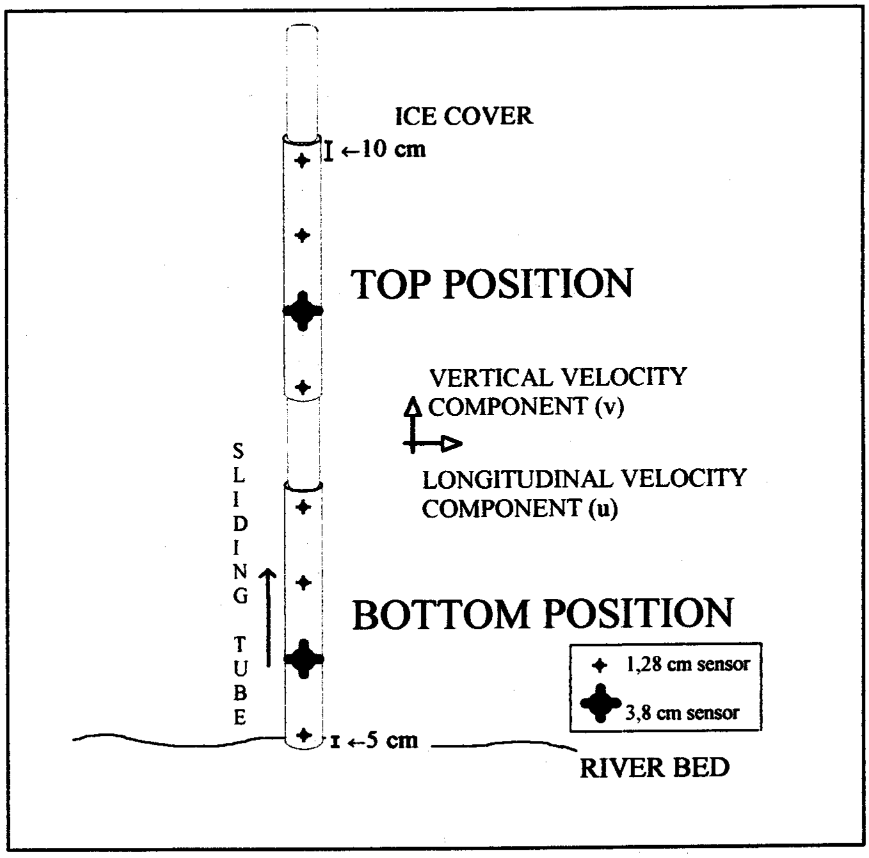

Figure 2: The experimental set-up for the velocity measurements

Fish Data

Data on the migration behaviour of tomcods were obtained at the same site during the month of January preceding the measurement of turbulence data. The count of downstream and upstream migrating tomcods was derived from video records taken with a subaquatic camera. The camera volume of view was 1.5 m in depth, 2 m in width and 1 m in height. The tidal conditions were similar to the ones that occured during the turbulence data measurement period (spring tides) and the video record length covered 48.5 continuous hours. The number of tomcods moving in both directions (upstream and downsteam) was estimated for each 10 minute period to allow representation and comparison of the data (Bergeron et al., 1994). The relationship between the turbulence data and the fish migration will not be established directly but will be inferred on the basis of the similarity in flow conditions.

RESULTS

Spatio-Temporal Variations

Longitudinal Velocities

Figure 3 presents temporal and vertical changes in longitudinal velocities over the complete tidal cycle.

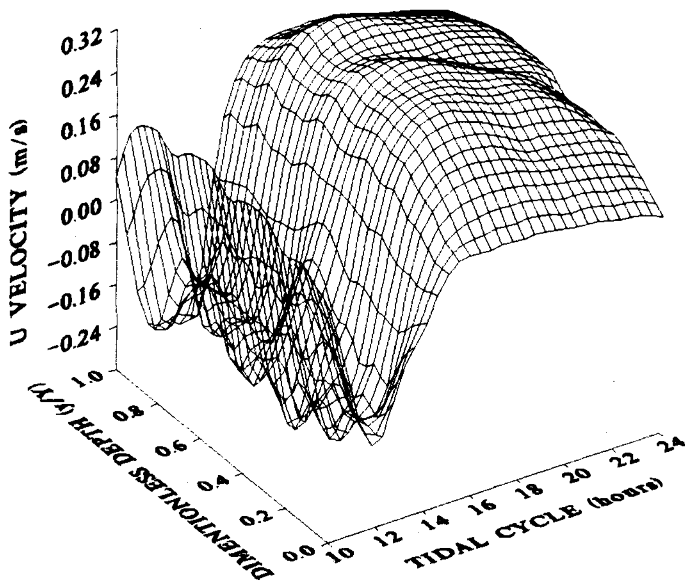

Figure 3: Vertical variations of longitudinal velocity component (u) over the tidal cycle

In the period between 10 to 14 hours, a complete flow reversal linked to the penetration of flood flow is observed. During this phase, velocities are up to two times higher than during the ebb tide. Velocities are lower near the bed than they are near the ice boundary over the whole cycle. A gradual decrease in velocities from the end of flood tide to the end of ebb tide is also visible over the whole flow depth. There is an important difference between the bottom and top positions. Velocities recorded by the four probes are grouped with more homogeneity at the top position, despite an identical spacing of the sensors. The velocity gradient is thus more important near the bed than near the ice. A similar tendency is also present in the root mean squares of the longitudinal velocities (u') (figure 4). Near the bed, values of u' are higher than near the ice, except for a few isolated cases. This observation, in addition to the steeper velocity gradient at the bed, means that the bed is the most important turbulence generating boundary and that turbulence tends to be less anisotropic under the smooth ice boundary.

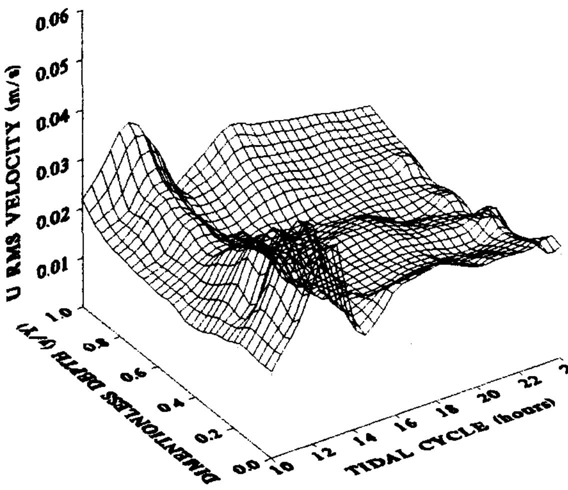

Figure 4: Vertical variations of root mean squared longitudinal velocity (u') over the tidal cycle

Vertical Velocities

Vertical velocities are always negative near the ice cover (figure 5). At the beginning of flood flow, velocities are all negative, reaching a value of -0.04 m/s near the ice and of -0.06 m/s in the centre portion of the flow. During flood flow, the values are up to three times higher than for the ebb flow. As the water level decreases, vertical velocities change gradually to positive values. The root mean squared vertical velocities (v') are higher than the root mean squared longitudinal velocities (u') at the beginning of the flood tide (during one hour) (figure 6). After that period, this tendency remains partially present until the end of the ebb tide in the centre portion of the profile, coincident with a region of high vertical velocities. Those high values of v' imply that exchanges remain important in the centre region during ebb flow. However, enhancement of vertical exchanges is linked mainly to flood flow penetration.

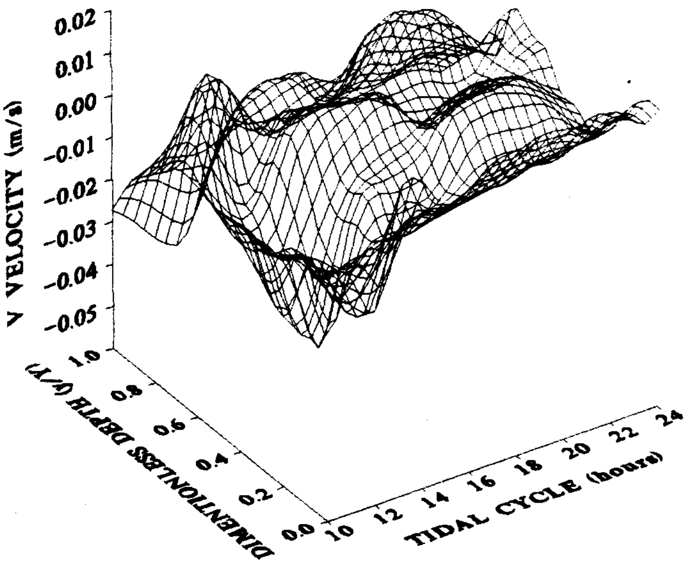

Figure 5: Vertical variations of vertical velocity component (v) over the tidal cycle

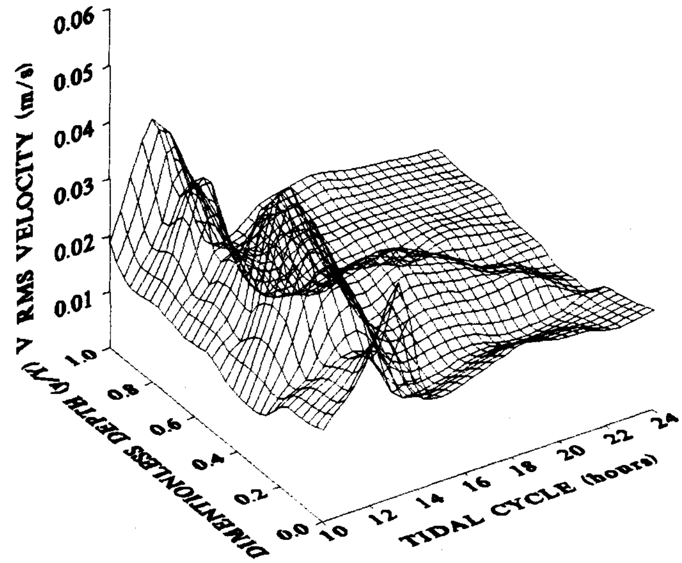

Figure 6: Vertical variations of root mean squared vertical velocity (v') over the tidal cycle

General Pattern Of Variability

General Characteristics Of Velocity And Turbulence Intensity Profiles In Relation With Tidal Phases

Three types of longitudinal velocity and turbulence intensity profiles are associated with the different tidal phases (figure 7). Each sensor is represented in both top and bottom positions. Mean values for each point of the profiles are estimated over the whole lenght of one typical measurement series that is representative for a specific tidal phase. The first velocity profile type corresponds to the fluvial flow (figure 7A). In this case, water level is low and velocities range from 0.1 to 0.2 m/s. The values of the turbulence intensity ratio (v'/u') range from 0.65 to 0.93. The second case (7B) represents the flood flow. Velocities shift from all positive to all negative values. During this phase, water level changes in the upward direction very rapidly and the absolute velocity values are the highest (from 0 to -0.45 m/s). The ratio v'/u' reaches a maximum value of 1.97 in this case. The last case (7C) includes the ebb flow and the fluvial flow. The water level decreases gradually and velocities range between 0.1 and 0.3 m/s. The values of the ratio v'/u' range from 0.61 to 1.31. The shapes of the velocity profile corresponding to fluvial flow (A) and the one comprising ebb and fluvial flows (C) are similar. The shape of the velocity profile describing flood flow (B) is different. The effect of the bed roughness is more apparent at the base of the profile as the drag linked to the bed surface is enhanced by the rapid velocities. Velocities in the top portion are higher than for the two other cases (A and C), with no apparent return to smaller values near the ice cover. The high flow velocity reduces the vertical development of the turbulent boundary layer near the hydraulically smooth ice surface. Also, the turbulence intensity ratios profile (v'/u') for the case of flood flow (B) presents a different pattern from A and C. During flood tide turbulence intensities are higher, especially in the centre portion of flow. This is a region of intense vertical mixing. There is a slight persistence of this tendency in the centre region of the turbulence intensity profile including ebb and fluvial flows. This concentration of high turbulence intensity values corresponds to a region of high vertical velocities.

Migration Pattern Of The Tomcod

Figure 8 presents temporal variations in the migration of tomcods in relation to mean longitudinal velocity (u) and tidal phases. This figure shows that upstream migration peaks mainly occur during flood flows. The most important upstream migration peak is related to the largest flow reversal where a maximum negative velocity value of -0.18 m/s is reached. The situation differs for the downstream migration that occurs mainly during ebb flow. Downstream migration is extended over a longer period than the upstream migration which is concentrated in few isolated peaks.

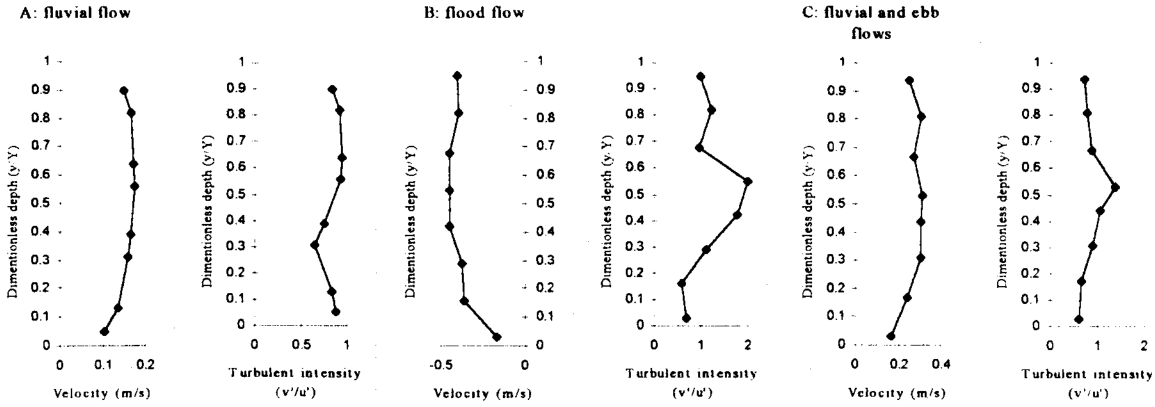

Figure 7: Typical U velocity and turbulence intensity profiles in relation with tidal phases

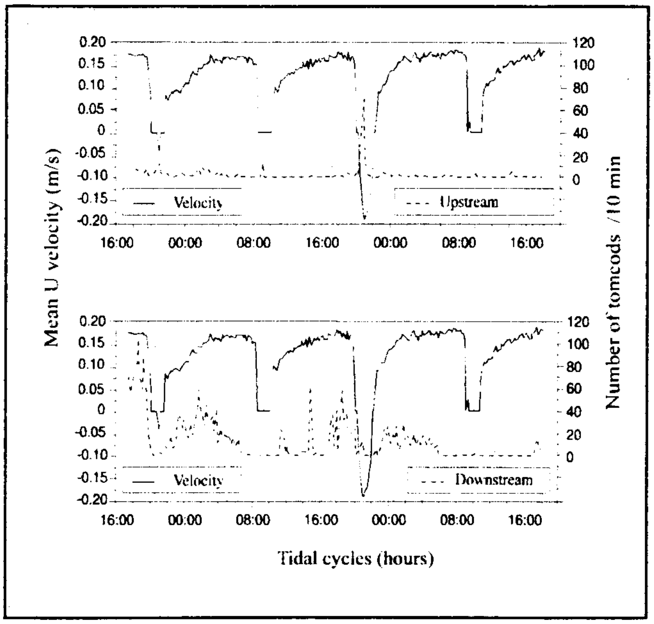

Figure 8: Migration pattern of tomcods in relation with mean U velocity direction and tidal phases (After Bergeron and al., 1994)

DISSCUSION AND CONCLUSIONS

This paper presents the first extensive velocity measurements recorded under an ice cover in a natural environment. Several turbulence parameters can be used to describe and compare the characteristics of both bed and ice boundary layers. These turbulent statistics provide valuable data to be used on a comparative basis in further studies on ice-covered river flow and of winter habitat characteristics and energy budget of fish during the winter time. From this study, the following conclusions can be made:

1. In a tidal environment and in the presence of an ice cover, the flow is separated in two boundary layers associated respectively to the bed and ice surfaces. The relative thickness of each turbulent boundary layer is related to the variations in flow depth induced by tides. As the water level decreases, the influence of the rougher bed surface turbulent boundary layer increases in the flow. The turbulence structure appears less anisotropic below the smooth ice boundary than at the bed where the velocity gradient is higher.

2. The flood flow penetration enhances turbulence intensity in both turbulent boundary layers. The vertical and longitudinal velocities are higher than during the ebb tide. During flood tide, v' values are higher than u' which means that vertical exchanges are important during this period. Shen and Harden (1978) have found that the addition of an ice cover inhibits vertical mixing. This is not what we observed at the Sainte-Anne river. In the present case, the role of tides is predominant in the vertical mixing enhancement. The turbulence intensity ratio v'/u' is up to 2.5 times higher than the highest value reported by West and Shiono (1988) and Kawanisi and Yokosi (1993) for a tidal environment without an ice cover. During flood flow, v' values are also more important near the ice than at the bed. This is linked to the mode of penetration of flood tide in the estuary. The flood wave invades the top layer of the river flow and induces a negative vertical movement which is clearly visible in the vertical velocities over the whole flow depth at the beginning of the flood tide. Thus, ice-covered tidal flow presents unique turbulent characteristics which are observed for the first time.

3. Though flood tide is marked by important turbulence intensity, it is the predilection upstream migration phase of tomcods. It seems that rapid upstream velocities is the most important factor driving the migrating behaviour of tomcods despite the important turbulence intensity of this phase.

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ON UNDER-ICE RIVER PLUMES IN JAMES AND HUDSON BAYS

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ABSTRACT

Knowledge of under-ice plumes is fairly recent. Most of the studies were undertaken on the east coast of James and Hudson Bays, motivated by the development of the hydroelectric potential of northern Québec. Such developments lead to a significant modification of the river hydrological cycle. Understanding of both physical components and coastal habitats ecology were key issues for the monitoring program of La Grande hydroelectric Complex and for the evaluation of the environmental impacts of the abandoned Grande-Baleine Project. Studies focused on under-ice plumes because flow modifications are much greater in winter.

An historical perspective of most of the studies undertaken on La Grande, Great Whale, Little Whale and Nastapoka rivers is presented and main conclusions are pointed out. Data from Nastapoka River will be used to illustrate that the horizontal and vertical extents of the winter plumes are much larger than their open water counterparts, even for lesser river flows.

An empirical relationship predicting plume surface area as a function of river flow (for a range of 60 to 4700 $m^3 s^{-1}$) and surface and ambient salinities is then discussed. This model is an improvement over earlier similar attempts. However, it should be applied with confidence only for plumes developping entirely under landfast ice in James and Hudson Bays, since it has been derived from data collected solely in those locations.

Along north-east coast of James Bay, landfast ice and drifting ice floes are often separated by a lead of open waters. The width of the lead seems to be related to wind conditions. Using satellite images, the influence of wind on ice conditions and on the surface area of the La Grande River plume will be discussed.

KEY WORDS: River plume / River flow / Hudson Bay / James Bay / Ice / Hydroelectricity / Monitoring / La Grande River / Great Whale River / Little Whale River / Nastapoka River.

INTRODUCTION

It has long been recognized that industrial development in Canadian North (oil and gas exploration, hydroelectric development) has stimulated scientific investigations (see Dunbar, 1982). In James and Hudson Bays (Figure 1) the pace of oceanographic studies rapidly increased under the pressure of the James Bay development, which in early '70 could include hydrological modifications of La Grande River and Nottaway-Broadback-Rupert Rivers in James Bay, and Great Whale River in Hudson Bay .

In 1971, when the La Grande hydroelectric project was first planned, there were so many gaps in current knowledge of the overall ecology of the region that it was difficult to predict with accuracy either the nature or the scope of the ecological impacts of any development project. A task force composed of specialists recognized throughout Canada recommended that priority be given to studies and research designed to provide a comprehensive body of basic data on the natural environment, data which would serve as reference in measuring temporary or permanent changes. Federal and provincial agencies joined the proponent of the project in that challenging task; among many fields of research activity, federal agencies were responsible for oceanography, estuarine sedimentology, marine and estuarine fauna and waterfowl and coastal habitats during the period 1972-82.

With the commissioning of power generating stations, the natural hydrological cycle of La Grande River has been changed: winter flows are 8 to 10 times higher and the spring flood is stored in the reservoir while summer flows fluctuate within the same range as under natural conditions (Figure 2). It became clear that most of the hydrodynamical modifications along the north-east coast of James Bay occur during the ice cover period. As soon as winter 1979-80, it was observed that river flows were high enough to push the river plume to the edge of the shore ice (Freeman *et al.*, 1982) and, for higher flows, even beyond (Ingram and Larouche, 1987a; Messier *et al.*, 1989; Anctil *et al.*, subm.). Monitoring of biological components was more puzzling, because biological processes are more active during ice free period. The main resource for Native along the coast of James Bay is migratory waterfowl. The monitoring program implemented by the proponent (1982-2000) is designed to take into account this resource, in respect for users. However, due to the mobility of highly valued resource, it was decided to study the habitat rather than the birds. The monitoring program in coastal James Bay is focused on hydrodynamical changes and its possible impacts on plant communities, the subtidal eelgrass and the intertidal marshes, in order to ascertain the changes related only to the project. The objective means that the main factors governing these communities must be taken into account : thus, the studies are more research oriented than simple monitoring (see also Julien *et al.*, 1996).

This paper discusses the main contributions on under-ice river plumes. Studies are not restricted to La Grande River plume, as increasing knowledge was also obtained through research programs on Great Whale River plume and environmental studies of the Great Whale Hydroelectric Project. Two elements are described in greater details : 1) a recent model to predict plume surface area as a function of river flow and surface and ambient salinities, and 2) the ice margin influences on the off-shore extension of the La Grande River plume. Since the model is based solely on river plumes developing entirely under an ice cover, observations of the role of a recurrent lead on the La Grande River plume illustrates the limit of the proposed model.

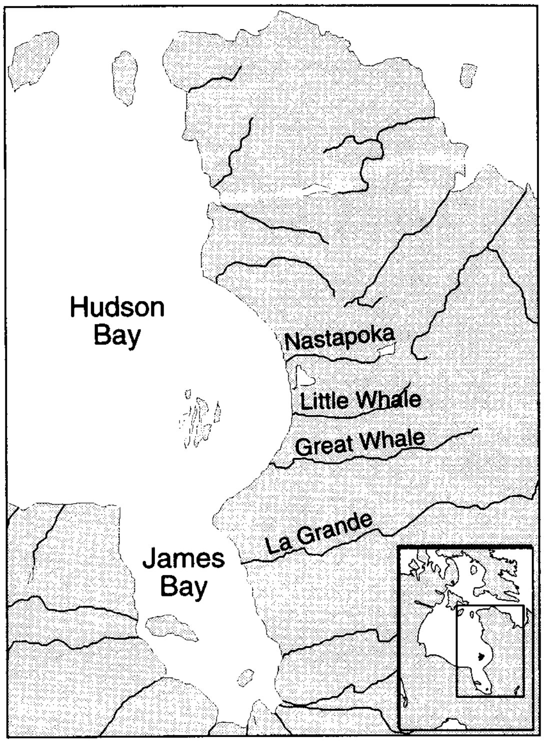

Figure 1: **Hudson and James Bay, showing positions of La Grande, Great Whale, Little Whale and Nastapoka Rivers.**

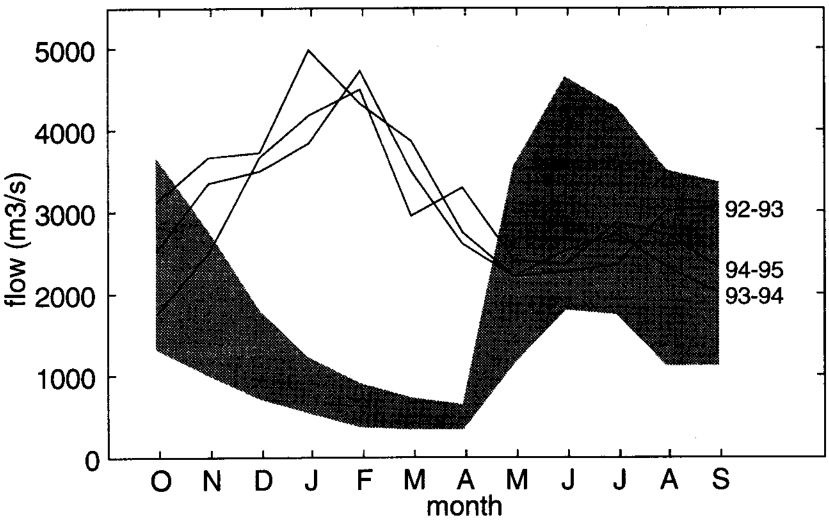

Figure 2: **La Grande River monthly average runoff. The shaded zone indicates the natural flow variation.**

AN HISTORICAL PERSPECTIVE

Knowledge of under-ice river plumes is fairly recent, since few oceanographic studies where conducted in James and Hudson Bays prior to 1960 (Dunbar, 1982), mainly because of technical difficulties associated with the presence of the ice cover and severe weather conditions. It is believed that the first winter oceanographic surveys of James Bay were performed in 1974, 1975 and 1976 (Croal and Tapiatic, 1974; Peck, 1978). In 1974, data were obtained only near the mouth of La Grande River, while those of 1975 and 1976 covered the entire James Bay, with a denser grid of stations close to La Grande River and Eastmain River. These data led to the first description of the La Grande River plume which was since intensively studied (La Grande River was the object of a major hydroelectric project). All in all, most under-ice river plume studies performed in the coastal waters of both James and Hudson Bays where motivated by the development of the hydroelectric potential of northern Québec. Since such development lead to a significant modification of the river hydrological cycle, the spring flood is suppressed and the maximum flow occurs from December to March (*i.e.* when the electric demand peaks), the prediction of the extent of under-ice plumes soon became an environmental preoccupation.

La Grande River Plume

Surveys by Peck (1978), with runoff from 400 to 500 $m^3 s^{-1}$, are the only significant information available on the La Grande River plume prior to the beginning of the engineering works that would alter its natural hydrologic cycle. During the winter of 1978-1979 the river flow was routed to fill the reservoir and the runoff to James Bay was limited to about 10 $m^3 s^{-1}$; during the winter of 1979-1980 the power plant operation began leading to runoff reaching a maximum of 1 800 $m^3 s^{-1}$ in March. The under-ice river plume over that period was investigated by Baird and Anning (1979), Brooks (1980), Freeman *et al.* (1982) and Freeman and Flemming (1982). From 1980 onward its winter runoff was gradually increased to reach a maximum daily discharge of 5 608 $m^3 s^{-1}$ in January 1995. Over that period, the under-ice plume was periodically surveyed to establish its extent for various runoff conditions (Ingram *et al.*, 1984; CSSA Consultants 1987; 1989; 1993). Knowledge gained over those years is summarised by Ingram and Larouche (1987) and Messier *et al.* (1989). Typical surface plume area defined by various isohalines are given in Table 1.

Great Whale River Plume

The Great Whale River plume, in the coastal waters of Hudson Bay, is the next most studied under-ice plume (see Table 1). The four studies performed over the years (Prinsenberg and Collins, 1979; Freeman *et al.*, 1982; Larouche, 1984a; 1984b) were all motivated by the Great Whale hydroelectric project which was recently abandoned. The Great Whale River is thus still unaffected by man-made discharge modifications. It is characterised by a mean flow of 640 $m^3 s^{-1}$ and spring floods up to 1 600 $m^3 s^{-1}$ which occur a few weeks before the Hudson Bay ice cover starts to melt. Ingram and Larouche (1987b) proposed an analysis of the most recent surveys.

Nastapoka River And Little Whale River Plumes

The Great Whale hydroelectric project called for the diversion of most of the flow (94%) of the Little Whale River into the future GB-1 Reservoir, while the Nastapoka River was left untouched. The under-ice plumes of those two rivers were surveyed once in March 1990 (CSSA Consultants, 1992). At that time, their flows reached 100 and 60 $m^3\ s^{-1}$, respectively (Table 1).

Table 1: Typical under-ice plume surface area (From Anctil *et al.*, submitted)

Date	Runoff ($m^3\ s^{-1}$)	Plume surface area (km^2) defined by isohaline					
		1	5	10	15	20	25
			La Grande River				
03 / 1980	1 600	295	665	993	1231	1642	
02 / 1993	4 700	970	1637	2055	2660	3522	
			Great Whale River				
04 / 1982	135	29	156	313	464	600	798
05 / 1982	910	240	672	1054	1379	1620	2010
03 / 1983	165	181	300	462	565	748	951
05 / 1983	505	252	486	654	867	1062	1337
			Little Whale River				
03 / 1990	60	53	80	117	160	185	210
			Nastapoka River				
03 / 1990	100	18	23	45	91	142	179

PROGRESS ON UNDER-ICE RIVER PLUMES

As the number of winter oceanographic investigations have increased, it has become clear that under-ice river plumes differ significantly from those observed during open water conditions. The horizontal and vertical extents of the winter plumes are much larger than their open water counterparts, even for lesser river flows (Ingram, 1981; 1983). The main justifications invoked for those differences are the suppression of wind mixing and the attenuation of tidal mixing (Ingram, 1981; Freeman *et al.*, 1982; Lepage and Ingram, 1991), which results in a much more stable interface between fresh and ambient water masses. Further studies have confirmed that the ice cover modifies tidal amplitudes and phases (Godin, 1986), significantly reducing the observed tidal, low frequency and residual currents (Prinsenberg and Ingram, 1991) and influencing the shape of the vertical variations of the tidal currents (Prinsenberg and Bennett, 1989). Overall, much less mixing energy is available under-ice than in open water in both James and Hudson Bays.

TYPICAL OBSERVATIONS: NASTAPOKA RIVER

A typical under-ice river plume data base consists of salinity and temperature profiles taken from conductivity-temperature-depth (CTD) samplings. A helicopter is used to transport personnel and equipment from station to station, where the CTD sampler is passed through holes drilled in the landfast ice or in floes. Occasionally some measurements are made in leads, if any. Given the large plume areas and the short winter daylight period, it may took up to 2 to 3 weeks to complete a survey of the largest observed single plumes.

Figure 3 and Table 2 present a comparison of the surface areas of an under-ice and an open water plume of the Nastapoka River in the coastal waters of Hudson Bay. The under-ice plume, observed in March 1990, corresponds to a river runoff of about 100 $m^3 s^{-1}$, while the open water plume, observed in October 1989, is associated to a runoff of 250 $m^3 s^{-1}$ (CSSA Consultants, 1992). Even if the open water runoff is 2.5 times higher than the under-ice runoff, its surface areas are about 40 times smaller. The presence of the ice cover thus dramatically change the mixing capacity of the Bay.

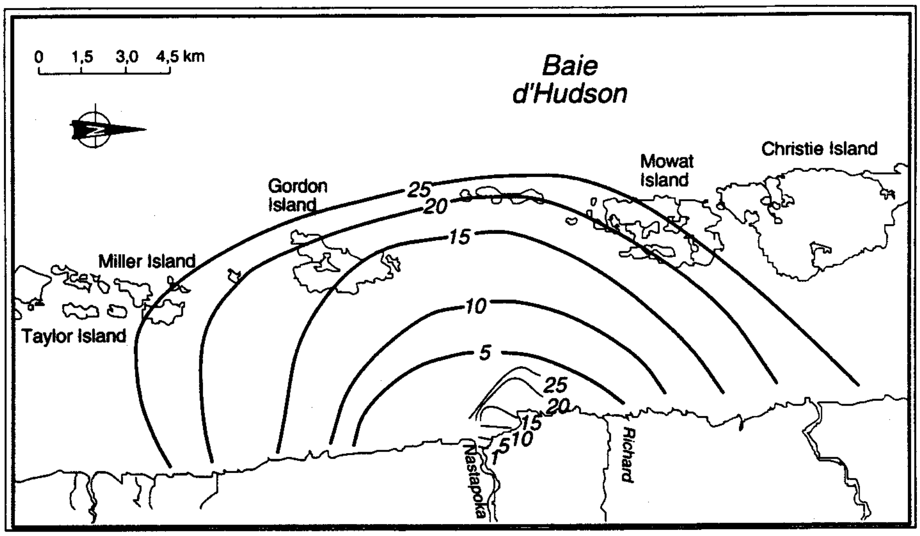

Figure 3: Comparison of Nastapoka under-ice and open water plumes

Table 2: Comparison of Nastapoka under-ice and open water plume surface areas

Date	Runoff ($m^3 s^{-1}$)	Plume surface area (km^2) defined by isohaline 1	5	10	15	20	25
			under-ice				
03 / 1990	100	18	23	45	91	142	179
			open water				
10 / 1989	250	0.3	0.4	0.8	1.8	3.7	4.6

PLUME EXTENT VERSUS RIVER RUNOFF

All the under-ice river plume surveys performed since 1980 in James and Hudson Bays had a common objective: to ascertain the extent of the under-ice plume for a specific river runoff condition in order to better understand the link between plume extent and river runoff. However, difficulties and costs related to salinity and temperature profile sampling in such a harsh environment and in such a remote area greatly limited the size of the available data base. For example, we have to wait for Ingram and Larouche (1987b) for the first comparative study. From six surveys of the Great Whale River plume in the coastal water of Hudson Bay and river discharge ranging from 135 to 910 $m^3 s^{-1}$, they proposed a relationship between plume surface area A_w (the area of the horizontal plane enclosed by a selected surface isohaline and the coast) and river runoff Q_0 of the form

(1) $$A_w = aQ_0^b \ .$$

Applied individually to plume areas delimited by surface isohalines 5, 15 and 25, exponent b (the slope) was 0.75, 0.59 and 0.39 with an r^2 of 0.64, 0.88 and 0.95, respectively (Ingram and Larouche, 1987b).

A similar work was recently carried by Anctil *et al.* (submitted) from selected conductivity-temperature-depth samplings describing eight plumes distributed among four locations along the eastern coast of both James and Hudson Bays, namely near La Grande River (LGR), Great Whale River (GWR), Little Whale River (LWR) and Nastapoka River (NR), for river discharge ranging from 60 to 4 700 $m^3 s^{-1}$ (Figure 4). In that case, the power law relationship (1) and surface areas delimited by isohalines 5 to 25 (in increment of 5) lead to slopes b lying between 0.67 and 0.79 (except for the 25 salinity for which a steeper slope of 0.85 was obtained) with an r^2 ranging from 0.77 to 0.85. These later results suggested the existence of the following unifying relationship between surface plume extent and river runoff (r^2 = 0.81):

(2) $$A_w = 17.7 \times 10^6 \left(Q_0 \frac{s_1}{s_a} \right)^{0.66} ,$$

where the river discharge is multiplied by the ratio of the surface (s_1) to ambient salinity (s_a), the later being the average value measured 3 m below the pycnocline for profiles with surface salinity ranging between 1 and 20 (Anctil *et al.*, subm.). Even if these empirical relationships should be used cautiously outside the eastern portion of both James and Hudson Bays, since mixing conditions may differ significantly elsewhere, they nevertheless show that the general features of under-ice river plumes can be empirically represented in a very simple fashion.

ICE MARGIN INFLUENCES

It has been mentioned that the coastal environment where James and Hudson Bays river plumes develop is much less energetic under-ice than for open water conditions. The suppression of wind mixing and the reduction of tidal mixing lead to considerably larger plumes in winter than in summer (see Figure 3), even for similar river flows. Thus, one can imagine the reduction effect of a important lead or polynya on the extension of an under-ice river plume. If a significant fetch of open water allows wind mixing to take its toll, the extension of the winter plume may be stopped quite abruptly.

Most of the authors who performed field works off La Grande River in wintertime (*e.g.* Croal and Tapiatic, 1974, Ingram and Larouche, 1987a; Messier *et al.*, 1989) mention the presence of a recurrent lead at the off-shore limit of the fast ice. With the important increase of La Grande River winter runoff, the plume extends beyond the fast ice (as observed in 1993), but it is believed that a favourable lead can stop the seaward progression of the plume (Ingram and Larouche, 1987a; Ingram *et al.*, 1987; Messier *et al.* 1989).

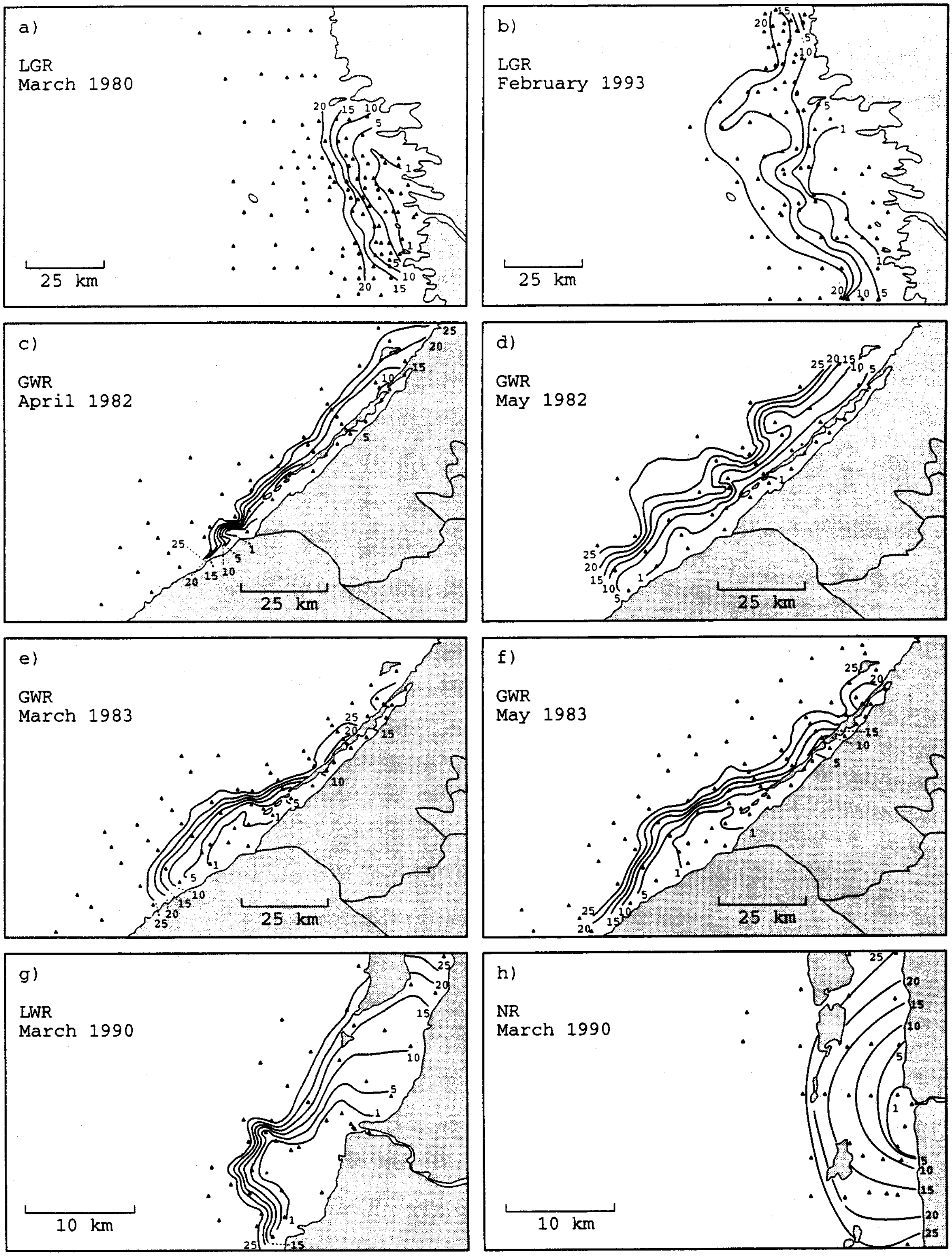

Figure 4: Surface isohaline contours. Triangles identify sampling location. (From Anctil *et al.*, subm.)

In that area, the fast ice is about 10 to 30 km wide, depending of the location along the coast, and its edge follows approximately the westward limit of coastal islands, shoals and reefs. This limit has been delimited repetitively since 1980 and no runoff influence has been detected. Beyond the fast ice, the bay is covered with ice floes for concentrations often reaching 8/10 and 9/10. At the interface of those two ice systems, favourable winds can open or close a coastal lead. Such a lead has been reported by many since Hare and Montgomery (1949) but its actual size and frequency is still not established. For instance, Freeman *et al.* (1982) believe the lead not to exceed 10 km wide, while Ingram *et al.* (1987) mention a frequency of about 4 to 5 days. In fact, some recent analyses of 6 Landsat-TM satellite images, from 1991 to 1995 in March and April, revealed coastal leads up to 60 km wide and confirmed that large leads are associated with north-east winds, while the absence of lead is observed under west or south winds.

The recurrence and the variability of the coastal lead offshore La Grande River make it difficult to establish its influence on the seaward progression of the plume beyond the fast ice. Moreover, it is not sure that the plumes observed in 1980 and 1984 would have been larger under a nearly complete ice cover, while no data were collected beyond the fast ice in 1987 and 1989 for security reasons. On the other hand, the 1993 observations clearly show that in the absence of a coastal lead, the plume may develop freely under the pack ice (floes concentration at that moment was believed to be 9/10). So, the influence of the ice margins on the La Grande River plume is still up in the air.

In spite the lack of conclusive observations, some explanations and hypotheses proposed in the past are worth discussion. For instance Freeman *et al.* (1982), that worked in the area for about three months in 1980, reported no lead larger that about 10 km which is, according to them, always too small to allow wave build-up and mixing. Moreover, the leads were refrozen at least half of the time. Thus, wider leads are needed to allow intense mixing. Ingram *et al.* (1987) support the idea of open water wind mixing but add to this phenomenon the effect of brine rejection associated with new ice production, following Schumacher *et al.* (1983). It is clear that James Bay climate is favourable to ice crystallisation; however, ice produced by plume water with salinities around 1 or 5 will not generate the large quantities of brine needed to mix 4 or 5 meters of plume brackish water.

Another hypothesis is proposed. Under favourable winds, ice floes are moved offshore and the coastal lead enlarges. At that time, the surface water is also displaced by the wind. Part of the plume is thus pushed away. Surface water carried offshore is replaced by deep saltier water, similarly to upwelling phenomenon in open water conditions. The surface plume is truncated in the lead and the part beyond the land fast ice will be transported northward with residual circulation. Referring to the observations of 1993 (see Figure 4) where about half of the plume develop beyond the fast ice, such a mechanism would over a short period of time eliminate a large part of the plume. Add to that mechanism the possibility of further wind and wave mixing whenever the lead gets large enough, the opening of the La Grande coastal lead becomes an effective plume reducer.

CONCLUSION

The James and Hudson Bays hydroelectric projects, proposed or built, in the seventies and eighties stimulated winter oceanographic surveys. It was believed that the main source of changes in coastal northern environments would be related to the modification of the hydrological regime of dammed rivers during winter and springtime, but no information was available on under-ice river plumes. The first surveys aimed to document these coastal features and to assess the physical effects of different river discharges on it.

Comparing under-ice La Grande River and Great Whale River plumes, Freeman *et al.* (1981) concluded that the volume of freshwater in the plume is directly related to river discharge and inversely related to tidal energy in the basin. Later on, Ingram and Larouche (1987b) proposed power relationships between plume surface area and river runoff for selected isohalines for Great Whale River plume. Similar work carried by Anctil *et al* (submitted) presented a unifying relationship for eight plumes distributed among four locations along the eastern coast of both James and Hudson Bays. Power law correlation shows that reasonable prediction could be achieved, since the horizontal extent of under-ice river plume is strongly related to the freshwater discharge and that local mixing influences can be approximated by regression. Further improvements on the model would be possible with some information about the vertical structure of horizontal currents inside and underneath the plume and with measurements of the vertical entrainment. The application of the proposed model outside the studied areas of James and Hudson Bays should be verified on suitable local observations, since mixing energy may significantly differ.

Another concern on plumes is the striking difference between open water and under-ice water mass characteristics. River plumes spread and expand under landfast ice reaching surface areas much greater than during open water conditions. For example, the Nastapoka River plume is about 40 times larger under-ice than for open water conditions, even if the river runoff is 2.5 times smaller. More difficult to study are transient conditions occurring during ice break-up or lead openings. Lepage and Ingram (1991) demonstrated that increased tidal action and inertial and low frequency phenomena provide sufficient kinetic energy to cause intense mixing when the ice sheet decays. Further progress could be achieved in understanding the role of open waters and floes motion on mixing during winter season when ice formation is still going on. We propose a mechanism involving surface water transport and upwelling in the coastal lead when it opens under favourable winds. A new set of observations of the La Grande River plume will soon be available for analysis. It may help better understand the role of the recurent La Grande River coastal lead on the extent of its under-ice river plume.

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WINTER FLOW CHARACTERISTICS IN COLD REGIONS

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ABSTRACT

The period of winter flow is critical in terms of aquatic habitat and water quality in the sub-Arctic. Unfortunately, relatively little is known of winter streamflow variability in sub-Arctic rivers. The cost of environmental monitoring may determine the monitoring strategy in remote regions rather than a knowledge of the relevant controlling processes. If low frequency sampling is an economic requirement then an improved knowledge of variability at relevant time scales is needed to assess the quality of winter flow estimates. In the absence of rainfall, snowmelt, or evapotranspiration, winter streamflow should be dominated by storage depletion processes resulting in a simple recession hydrograph. Discharge variability during the low-flow period, causing temporary habitat loss, may limit aquatic populations. Therefore, it is important to know what processes or events produce deviations from the storage depletion recession. Discharge measurements and water quality samples were collected frequently during the winter of 1994-95 at three sites in the southwest Yukon. These observations were complemented by electronic data acquisition to estimate flow between measurements and to ensure transient events were recorded. Analysis of these data shows that storage depletion from groundwater and lake sources dominates the pattern of winter streamflow. Discharge depressions associated with the formation of an ice cover occur as transient events and show as negative deviations from the recession model. These deviations correspond to distinct events in the specific conductance record. Additional periodic deviations in both streamflow and water chemistry, although small, indicate that other processes occur on a time scale of hours to several days. A monitoring strategy which is restricted to two or three measurements during the winter is relevant only on the time scale of the general recession. If interest is limited to the storage depletion process, measurements for calibrating recession models should be timed to avoid discharge depression events. If more specific knowledge of discharge variations on a daily or hourly time scale are needed, then time series observations of stage and velocity using electronic data acquisition are necessary.

KEY-WORDS : Winter/ice-cover/streamflow/recession/discharge depression/recession/storage depletion

INTRODUCTION

Annual low flows occur during the winter season in the sub-Arctic due to the absence of inputs from rainfall or snowmelt. Information about low flows are important both for design of hydraulic structures (such as reservoirs and water intakes) and for ecological concerns (water quality and aquatic habitat). However, knowledge of cold-season streamflow variability is hampered by a lack of continuous discharge data. During the open-water season discharge can be reliably produced from recorded stage data using a stage-discharge relation. However under ice-cover, stage has not proven to be a reliable predictor of discharge. Water Survey of Canada (WSC), the primary source of hydrometric data in Canada, bases winter flow records on as few as two observations of discharge during the ice-covered season which may last 6 months or longer. Daily discharge is typically estimated between measurements by a subjective process of interpolation with climatic comparison. Melcher and Walker (1992) found significant differences between individual hydrographers using this method and 42% of interpolated estimates on three streams in Iowa had errors greater than 25%. While new automated methods of determining discharge are being tested, there is presently no technology proven to produce reliable, continuous discharge data through the ice-cover period without frequent actual measurements.

During the sub-Arctic winter, streamflow variability should be dominated by groundwater storage depletion and changes in lake and channel storage because rainfall, snowmelt and evapotranspiration are negligible. Variability from the recession trend is thought to be due to temperature effects and to the frictional effects of the ice cover. Some technologists and hydrologists in both WSC and the U.S. Geological Survey (USGS) have adopted the assumption of a temperature effect on winter discharge, and air temperature is used as an independent variable in most methods currently used to estimate winter discharge (Melcher and Walker, 1992). An increase in river stage occurs in response to the frictional effect of the ice cover (Beltaos et al., 1993; Alford and Carmack, 1988) and it is conceptually feasible that a loss of discharge to channel storage is associated with this stage-up. The loss of discharge to channel storage may be exacerbated by a reduction of inflow from tributaries due to backwater effects resulting from stage-up response in the main channel. However, little is known about the magnitude or duration of temperature effects, if they exist, or of stage-up related discharge depressions. This lack of knowledge is due, at least in part, to monitoring strategies which operate on a time scale which is inappropriate for these processes. The objective of this research was to collect high resolution time-series discharge and water quality data from groundwater and lake-dominated sub-Arctic streams to provide a basis for addressing the following questions: (1) What is the magnitude of deviations from the recession trend for winter flow? (2) Can deviations from the recession trend be explained in terms of changes in channel storage and/or temperature effects? (3) What effect does lake storage have on variability of winter streamflow? and (4) What is a relevant time scale for observing winter streamflow?

METHODOLOGY

Field Sites

Stage, discharge and water quality data were collected from three sites in the Yukon River basin near Whitehorse, Yukon. Two sites were located on the Takhini River, one at the outlet of Kusawa Lake and the other 40 km downstream. The third site was located on the Ibex River, a groundwater dominated tributary to the Takhini. The sites were chosen on the basis of logistic simplicity and to provide definition of groundwater storage depletion, lake storage depletion and in-channel processes. Soils in the study area are a sequence of glacial, glaciofluvial and glaciolacustrine deposits. The climate is predominately continental with coastal influences in the upper Takhini River Basin.

The Takhini River near Whitehorse site has a drainage area of 6990 km^2 and is located 40 km downstream of Kusawa Lake. This site is a WSC gauging station which has been active since 1948 and was used by Chin (1966) in a study of the accuracy of winter streamflow records. The Takhini River at the outlet of Kusawa Lake site has a drainage area of 4070 km^2 and Kusawa Lake has an area of 140 km^2. This site was a WSC gauging site from 1952 to 1986. The Ibex River site has a drainage area of 457 km^2 and is located at a WSC gauging station which has been active from 1989 to the present. The Ibex River is confluent with the Takhini River 30 km downstream of Kusawa Lake.

Stage And Discharge Measurements

Vertical control for river and lake water level was obtained by running level surveys to WSC bench marks at each site visit. Mercury manometers connected to bubbler tubes were used for continuous measurement of stage at Ibex and Takhini River near Whitehorse sites. These stage measurements were logged on VEDAS dataloggers. A submersible Stevens transducer attached to a Stevens MultiLogger was used to record water level at Kusawa Lake.

Stream discharge measurements were made using WSC equipment and conformed to WSC standards and procedures (Terzi, 1981). Measurements were made a minimum of once per week with additional measurements during the freeze-up period. Discharge was determined by the area-velocity method, with a minimum of 20 measurements of depth and velocity across the cross-section. Velocities were measured with a Price current meter. The uncertainty of an open water discharge measurement under ideal conditions at the 95% confidence level may be as high as ±6% (Pelletier, 1988), and while the uncertainty of a discharge measurement under an ice cover has never been precisely quantified, the error may be greater than for open water discharge measurements (Pelletier, 1989, 1990). It is reasonable to assume that the uncertainty of a measurement under an ice cover may be at least ±10%.

A stage-discharge relation based on open-water measurements was used to convert recorded stage to discharge. WSC standards require that if a measurement does not plot within 5% of the stage-discharge curve a shift correction be applied to the stage-discharge relation, e.g. to accommodate changes due to frictional effects of the ice cover. Shift corrections were determined for each discharge measurement and linearly interpolated over the interval between measurements. Several variations of this method are widely used for estimating winter flow (Melcher and Walker, 1992). However, it should be noted that several assumptions implicit in the application of shift corrections cannnot be justified unless the effect of the ice cover on discharge is known to be stable over time. Hence, data estimated using this technique should be used with caution, though the high frequency of measurements in this study should help minimise this uncertainty.

Water Quality

Hydrolab H2O sensors located at each site were controlled by VEDAS dataloggers and programmed to sample specific conductance every three hours. The Hydrolabs were calibrated to KCl standards in a laboratory prior to deployment after which they were not touched for the duration of the study period. A fourth Hydrolab was frequently re-calibrated and operated in parallel with the primary Hydrolab at each of the three sites in order to evaluate instrument drift.

Meteorological Data

Daily temperature and precipitation data were obtained from the Atmospheric Environment Service (AES) station at the Whitehorse airport. This station should be representative of the climate in the lower elevations of the study catchments. The temperature data is used as a representative index of temperature variations throughout the study area.

Data Analysis

A fundamental assumption underlying the data analysis is that streamflow over the winter should be dominated by storage depletion, which should produce a concave-up pattern. The effects of channel storage should produce a dip below the recession trend, while temperature effects should produce variations about the trend line. The analysis proceeded as follows:

a. Visually inspect the hydrograph and identify whether a significant dip occurred at and following freeze-up. If so, locate the time at which flow recovered to the recession trend. It is acknowledged that identification of the time of flow recovery is subjective.
b. Fit a recession curve to the data, excluding data during a post-freeze-up depression, as well as data at the end of the winter which may be influenced by snowmelt runoff.
c. Estimate the volume of lake outflow which can be accounted for in terms of lake storage depletion.
d. Examine deviations from the recession trend to determine the magnitude and relevant time scale of other processes affecting winter discharge.

Model Fitting

The multiple linear-reservoir model is widely used to represent streamflow recession from groundwater dominated streams (Hall, 1968; Tallaksen 1995). A dual linear-reservoir model was used here to represent the recession at the Ibex River, the groundwater-dominated site, based on a diagnostic plot of discharge. In this model discharge is calculated as

(1) $$Q_t = Q_0 \cdot B \cdot e^{-k_2 \cdot t} + Q_0 \cdot (1-B) \cdot e^{-k_1 \cdot t}$$

where Q_t is discharge at time t, Q_0 is discharge at t = 0, k_1 and k_2 are recession constants and B is a calibrated parameter.

Parameters in the recession model were estimated using by a non-linear iterative algorithm in a commercial spreadsheet program (Excel). The loss function to be minimized was a sum of squared relative errors (SSRE), calculated as

(2) $$SSRE = \sum_{i=1}^{n} \left[(Q_{pi} - Q_{ai}) / Q_{ai} \right]^2$$

where Q_{ai} is observed flow on date i, Q_{pi} is predicted flow on date i and n is the number of data points used for fitting the model. Fractional residuals were used in the loss function, rather than the raw residuals, in order to avoid biasing the model to fitting the higher discharges. The use of fractional residuals is also consistent with the notion that errors in discharge measurements are a fraction of the actual value.

Lake outflow was modeled by coupling a model of lake inflow with the stage discharge relation at the lake outlet. Lake inflow was calculated from lake stage and measured outflow using the mass balance equation

(3) $$Q_i = Q_o + \frac{\Delta S}{\Delta t}$$

where Q_i is lake inflow, Q_o is lake outflow and $\frac{\Delta S}{\Delta t}$ is change in lake storage over time, which is calculated as

(4) $$\frac{\Delta S}{\Delta t} = \frac{H_{t+10} - H_{t-10}}{21} \cdot A \cdot 86400$$

where H_{t+10} is lake stage 10 days after time t, H_{t-10} is lake stage 10 days prior to time t, A is lake area, 86400 is the number of seconds in a day and 21 is the length (in days) of the time step. A three week time step was used to average out any error in the stage values. A value for Q_i was calculated for each day a discharge measurement was made at the lake outlet. The dual linear reservoir model was used to model Q_i on the basis of a diagnostic hydrograph. Modeled inflow was used in conjunction with the lake outlet stage-discharge relation to calculate daily change in lake stage which in turn was used to calculate lake outflow.

Discharge at Takhini River near Whitehorse was modeled as the sum of recorded lake outflow and estimated groundwater inputs downstream of Kusawa Lake. The dual linear-reservoir model was used to estimate groundwater inputs. The difference in discharge between Takhini River at the outlet of Kusawa Lake and Takhini River near Whitehorse on November 1 was used as Q_0. The model parameters used were from the calibration of Ibex River. Ibex River discharge accounted for 19% of inflow into this reach of the Takhini River on November 1.

RESULTS

Weather and Ice Conditions During The Study Period

Accumulated snowfall was slightly above normal at 146.7 cm (normal 141.5 cm) with heavier snowfall during November, February and March and lighter snowfall during December, January and April. Average temperature during the study period (November to April) was -10.4°C, fairly close to the normal of -10.8°C.

Anchor ice was observed along most of the streambed at Ibex River during a flight over the catchment Nov. 2. By Nov. 14 the stream was approximately 80% ice covered. Open leads persisted at the measurement section throughout the winter, due to the inflow of warm groundwater and physical removal of ice prior to each measurement. Flowing frazil ice was first observed on the Takhini River on Nov. 2 and by Nov. 16 frazil density was sufficient that it stopped running and formed an ice cover at the measurement section. Between Nov. 2 and Nov. 16 the density of the frazil run varied with atmospheric conditions with the heaviest runs occurring after clear, cold nights. Ice cover was first observed on the south end of Kusawa Lake on Nov. 30. By Dec. 10 ice cover had formed over the lake outlet for a distance of 300 m from the end of the lake. The river was open bank to bank 300 m downstream from the lake due to a riffle which kept this section ice-free all winter. The areal extent of the ice cover in the first 300 m of river varied considerably with changing atmospheric conditions throughout the winter.

Groundwater Storage Depletion

The winter discharge recession was non-linear (Figure 1) and therefore cannot be represented with a single recession coefficient. Only measured discharge was used for model calibration. However, results are presented including recorded discharge to provide a more complete picture of discharge variability (Figure 1). Measurements indicated with open circles were excluded from model calibration based on the subjective judgment that these measurements were affected by either the effects of formation of an ice cover or by snowmelt runoff. The dual linear-reservoir model provided a good fit to observed discharge at Ibex River with 25 out of 26 measurements between November 21 and April 10 plotting within ±10% of model predictions ($B = 0.699$, $k_1 = 0.0026$, $k_2=0.0255$).

Lake Storage Depletion

The inflow-storage-outflow time-series model for Kusawa Lake provides a reasonable approximation of the discharge trend at Takhini River at the outlet of Kusawa Lake (figure 2) and provides a good fit to the pre-calibration measurements. The event on December 1, with a deviation of more than 40% from predicted flow is consistent with observations of an ice cover over the first 300 m of river from the lake outlet at that time. The downward spike in conductance on November 30 was due to frazil ice accumulating on the Hydrolab sensor, indicating that anchor ice was forming at the start of the discharge anomaly. There is close agreement in the water balance for the period November 12 to April 21 between modeled discharge ($2.03 \cdot 10^8$ m^3) and recorded discharge ($2.06 \cdot 10^8$ m^3), a difference of 1.3%. From the similarity in the water balance it is reasonable to assume that the

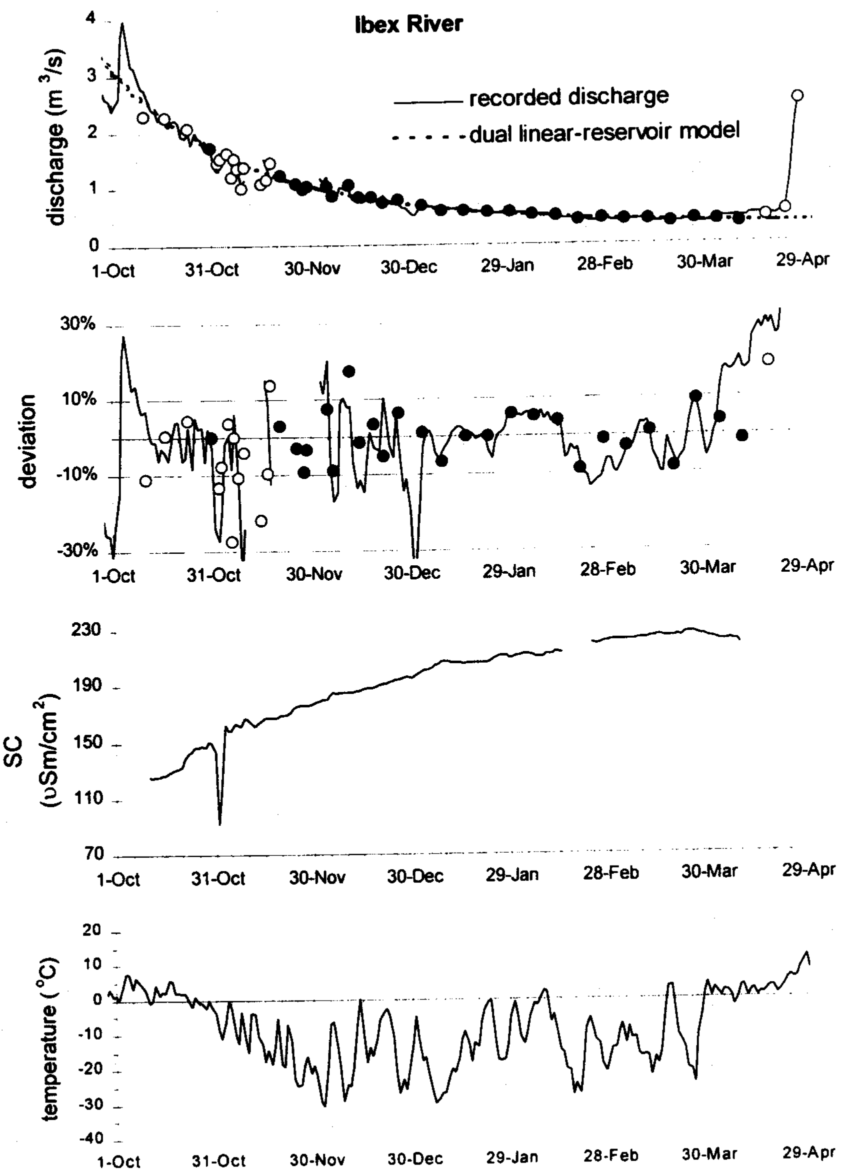

Figure 1: Ibex River discharge, deviation of discharge from the recession model, specific conductance and Whitehorse airport mean daily air temperature. Solid circles indicate measurements used for calibration. Open circles indicate measurements excluded from calibration.

positive residuals after Jan 11 are due to lake levels which are higher than they would have been without the discharge depression. The 7-day low flow occurred from April 15 to April 21, prior to an increase in lake stage April 21. The proportion of flow that can be attributed to lake storage depletion during the low flow period is 44% of lake discharge. This percentage is only slightly less than the proportion for the entire recession period of November 1 to April 21 (48%). Streamflow during the low-flow period was approximately 15% greater than predicted by the inflow-storage-outflow model.

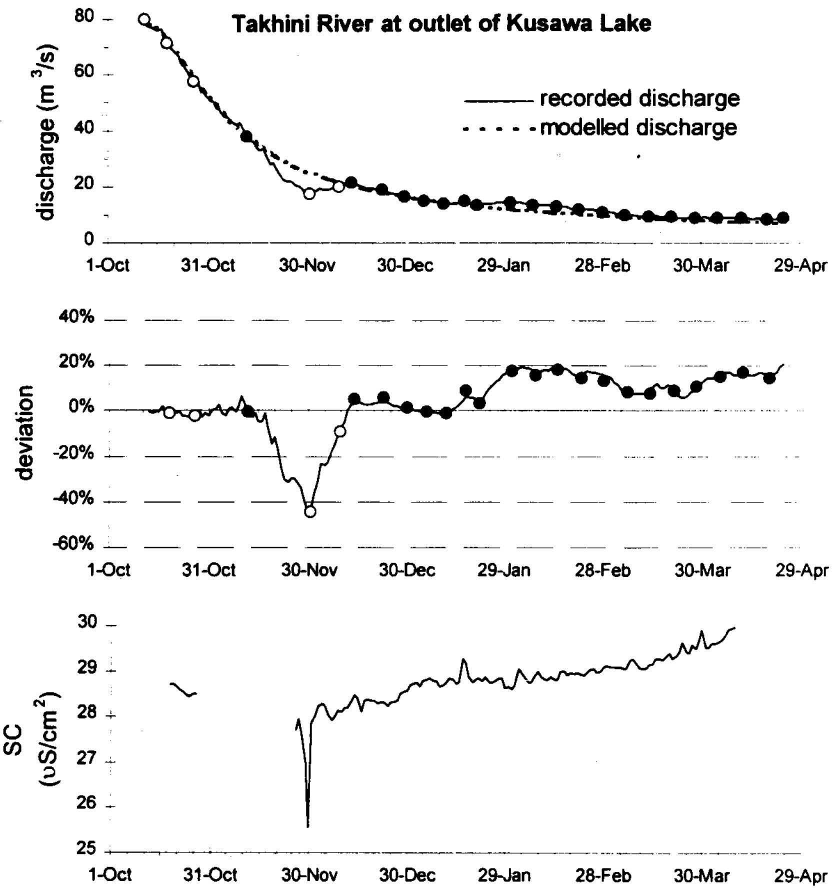

Figure 2: Takhini River at the outlet of Kusawa Lake discharge, deviation from modelled discharge and specific conductance.

It should also be noted that the positive residuals after January 11 are coincident with a slight decrease in specific conductance at Takhini River near Whitehorse (Figure 3), consistent with an increase in the relative contribution of dilute lake water, and an increase in stage at the Takhini River near Whitehorse (Figure 4). The negative residuals for the Takhini River near Whitehorse during the period of increased lake outflow must then be a result of negative deviations from the groundwater inflow model used. A reduction in groundwater inflow may be due to hydraulic damming as a result of the increase in channel stage causing a reversal of hydraulic gradient across the streambed (Hamilton and Moore, MS). Losses to bank storage, or backwater effects on influent streams may also explain some or all of these negative deviations.

Freeze-Up Related Discharge Depression

Negative residuals from the recession trend during the period of ice cover formation were found at all three sites, consistent with the notion of a freeze-up related discharge depression. Downward spikes in specific conductance were also observed at all three sites coincident with the start of discharge depression events. These anomalies are a result of frazil ice accumulating on the Hydrolab sensors. The observed discharge depression was unexpected at the outlet of Kusawa Lake, as it is generally assumed (e.g. Chin, 1966) that in the Southern Yukon, lake outlet polynia are persistent through the winter and, as a result, lake outflow should be monotonic through the winter. However, in the winter of 1994-95, ice was observed to form in the first 300 m of the river, retarding lake storage

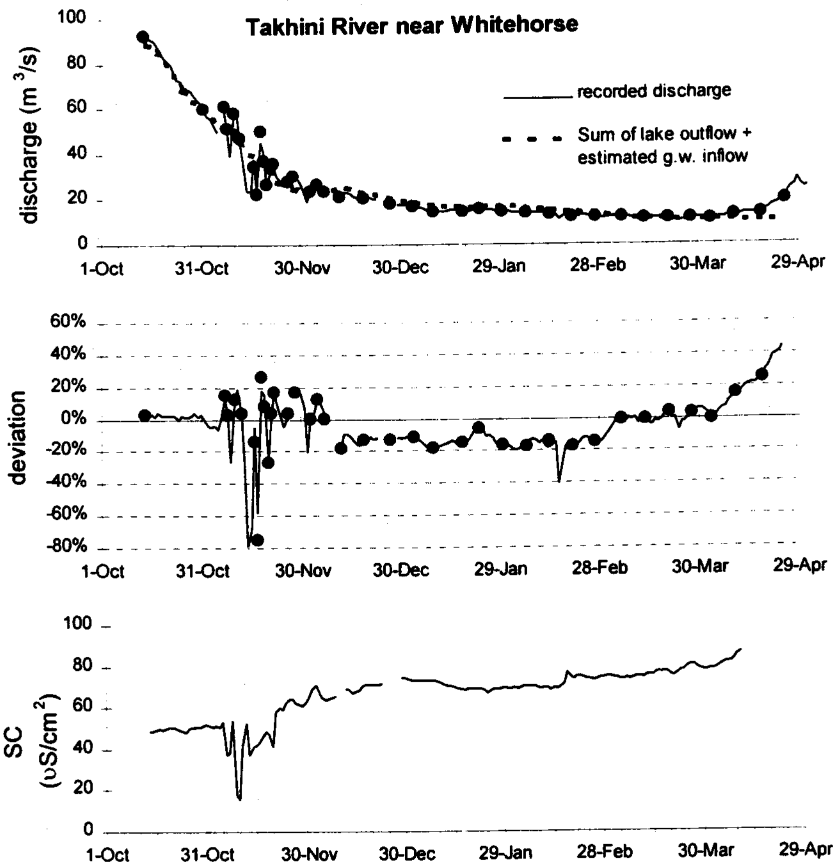

Figure 3: Takhini River near Whitehorse discharge, deviation from modelled discharge and specific conductance.

depletion. There was no increase in lake stage associated with this event whereas, river stage did increase at Takhini River near Whitehorse associated with freeze-up November 16 (Figure 4). The increase in stage at Takhini River near Whitehorse was approximately 50% of pre-freeze-up channel depth, somewhat more than the prediction of 32% of channel depth for a wide rectangular cross-section (Beltaos et al. 1993), possibly because of the presence of frazil ice in the channel (16% of cross-sectional area). At all three sites the minimum observed flow during freeze-up was substantially greater than the late-winter low flow.

Positive residuals from the recession trend (Figure 3) at the Takhini River near Whitehorse during the period of ice formation are likely due to the channel-storage mechanism being active on a diurnal time scale. These measurements were made during the day with flowing frazil ice present in the stream. The amount of frazil was observed to be light in the early morning and to increase as the day progressed. Sand and gravel up to the size of fist-size rocks could be seen in the flowing frazil. This portion of the floating ice formed as anchor ice during the night then lifted during daytime heating. As a result, there would have been transient channel storage during the night, with a release of that storage during the day. An observational bias in the data is a probable result of this process. The assumption of a linear interpolation of shift correction between measurements is not valid when channel conditions are changing this rapidly.

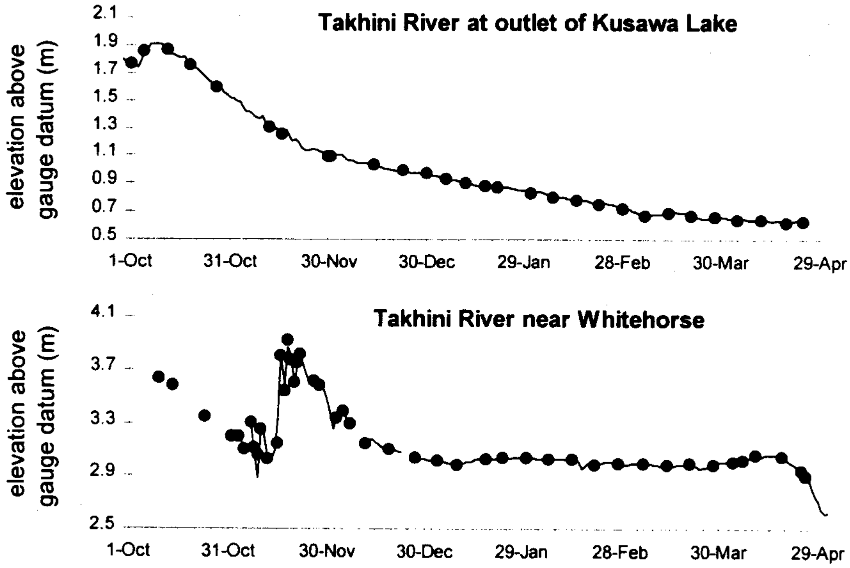

Figure 4: stage record for Takhini River at outlet of Kusawa Lake and Takhini River near Whitehorse

Temperature Effect On Winter Discharge

Linear regression analyses of model residuals against temperature were used to assess the significance of temperature on winter discharge. For Takhini River at the outlet of Kusawa Lake the effect is significant at the 95% confidence level ($R^2 = 0.268$, $P = 0.023$, $n = 19$). This result is consistent with observations of changes in the areal extent of the ice cover at the lake outlet. There is evidence for at least two other smaller events associated with cold temperatures. On February 14 a downward discharge spike can be seen in the discharge record for both the lake outlet and the Takhini River near Whitehorse. An upward spike in conductance recorded at the Takhini River near Whitehorse is consistent with a reduction in the relatively dilute lake water. Similarly, downward discharge spikes associated an upward spike in conductance at Takhini River near Whitehorse occurred on March 23. At Takhini River near Whitehorse ($R^2 = 0.019$, $P = 0.520$, $n = 24$) and Ibex River ($R^2 = 0.005$, $P = 0.722$, $n = 25$) the regression analyses show that the effect of temperature on discharge is not significant.

DISCUSSION

Magnitude of Deviations From Recession Trend

During the period of freeze-up, deviations from the recession trend ranged from -30% (Ibex River) to -80% (Takhini River near Whitehorse). However, at all three sites, the lowest flow during freeze-up was substantially greater than the late winter low flow. This is in contrast to observations of discharge depressions on the Liard and Mackenzie rivers where the low flow during the discharge depression event may be substantially lower than the late-winter low flows (Gray and Prowse, 1993; Burn, 1995). Initial ice conditions on large northern rivers tend to be very rough, with large amounts of frazil ice in the cross-section. These conditions would increase the stage required to pass a given volume of flow and therefore require proportionately greater channel storage than the small streams examined in this study. Large rivers also tend to have relatively large plan areas in comparison with smaller rivers and so would require relatively more volume per unit of stage increase.

For the groundwater-dominated stream (Ibex River), further deviations from the recession trend were within nominal measurement error (±10%) however, they appear to be auto-correlated. At the lake-dominated sites

(Takhini River at the outlet of Kusawa Lake and Takhini River near Whitehorse) deviations from the recession trend were substantially greater than measurement error (±20%).

Channel Storage and Temperature Effects

Discharge data collection during the freeze-up period has a positive bias, likely due to the assumption that a shift correction can be linearly distributed and applied to recorded stage data to produce discharge. This assumption cannot be justified when ice conditions in the channel are constantly changing. The positive bias would be a result of measurements which were conducted during the daytime, when there was a release of transient channel storage. For this reason no attempt was made to reconcile channel storage volumes with observed discharge depressions. At the Takhini River near Whitehorse the observed increase in channel stage associated with freeze-up is consistent with theoretical calculations of stage-up response if the presence of frazil ice in the channel is taken into consideration.

Temperature effects can account for some of the discharge anomalies observed at the outlet of Kusawa Lake. These events are associated with observed changes in the areal extent of the ice cover at the lake outlet. However, there is no simple cause and effect relation between temperature and discharge for groundwater-dominated flow.

Deviations from modeled flow at Takhini River near Whitehorse are thought to be due to the effect of channel stage on groundwater inflow. Specifically, an increase in channel stage may result in a reduction in groundwater inflow due to a reversal of the hydraulic gradient at the streambed. Further losses of flow may be caused by backwater effects on influent streams.

Deviations from modeled flow at Ibex River, though within nominal measurement accuracy show a quasi-periodicity which may indicate auto-correlation. Auto-correlation would indicate that some unexplained process is active.

Lake Storage Effects On Winter Streamflow

Janowicz (1991) found that in multiple regression analysis of annual low flows the inclusion of lake area index did not significantly improve low flow predictions. In contrast, analysis of the outflow from Kusawa Lake show that 42% of the low flow can be accounted for in terms of lake storage depletion. This disparity may be due to the discharge depression event in the winter of 1994-95 which had the effect of retarding lake storage depletion. This may indicate that the observed lake outlet discharge depression is anomalous. However, it should be noted that the temperature in November was only slightly colder than normal (- 13 °C compared to -10 °C) and December was slightly warmer than normal (-15 °C compared to -15.9 °C) and so it would be reasonable to assume that the conditions which contributed to a freezing of the lake outlet polynia may have occurred in other years. If this type of event is not unusual, then the assumption that lake stage is a good predictor of winter discharge at the lake outlet is not valid.

Time Scales For Winter Discharge Monitoring

If interest is limited to the storage-depletion recession and it can be shown that recession model parameters are stable between years, then 2 or 3 measurements per winter should be sufficient to validate a simple model such as the dual-linear reservoir model. The calibration measurements should be timed to avoid discharge depression events to avoid introducing bias into the model. The magnitude of the discharge depression events on these rivers contrast with results of other researchers indicating that the magnitude of the discharge depression event may be a function of basin scale. The effect of basin scale needs to be resolved in order to identify the time scale that is relevant for this process for any given river. If detailed knowledge of the discharge regime is required then continuous monitoring would be necessary to record transient events which may occur at on a time scale of hours (e.g. diurnal anchor-ice formation and release) or days (e.g. stream channel-aquifer interactions). Unfortunately, stage data alone is a poor predictor of winter discharge. Some measure of stream velocity is required, in addition to stage data, to provide reliable estimates of discharge under an ice cover.

CONCLUSION

Winter discharge is dominated by groundwater and lake storage depletion. Thé primary cause of deviations from the storage depletion recession is the frictional effect of an ice cover. Discharge depressions associated with the formation of the ice cover were observed at all three sites studied. Ice cover at the outlet of Kusawa Lake had a controlling influence on lake outflow. These changes in lake outflow had an effect on discharge at a site 40 km downstream. An increase in lake outflow was associated with a reduction in groundwater inflow in the reach between the lake and the downstream site. This is thought to be due to channel-aquifer interactions. There was no evidence to support the notion of a temperature effect on groundwater-dominated winter flow, though temperature was associated with discharge variability at the outlet of Kusawa Lake.

Further research into transient processes affecting winter flow is limited by traditional hydrometric technology. However acoustic flowmeters now under development may soon be able to meet the need for continuous, reliable winter discharge data (Fast and Wiebe, 1992). Studies of the magnitude and duration of freeze-up related discharge depressions should be conducted to determine the effect of basin scale on discharge depressions. Near-stream piezometric and channel stage data are required to determine the effect of stage-up on stream-aquifer interactions. Long term studies of winter flow are required to investigate the inter-annual variability of storage-depletion recession parameters.

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EFFECT OF ICE FORMATION ON THE PHYSICAL STREAM ENVIRONMENT AND THE OVERWINTER DISTRIBUTION OF PRESMOLT ATLANTIC SALMON PARR

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ABSTRACT

Winter carrying capacity of the stream environment is recognized as a bottleneck in smolt production. We assessed the effect of ice formation on water velocity and depth and on the nocturnal habitat use and distribution of yearling and older Atlantic salmon (*Salmo salar*) parr between December 2 ,1994 and April 4, 1995 in a 200-m reach of the Rock River, Vermont (USA). Ice formation reduced channel width, increased water depth, and altered the lateral distribution of low (< 40 cm/s) and high (> 80 cm/s) water velocities. Overwinter, 75% of the parr encountered were found at water velocities < 40 cm/s, even though these velocities represented only 33% of those recorded. During iced and post-ice conditions, parr were most abundant in areas ≥ 4 m from the stream centerline. Ice formation affected the distribution of water velocities commonly selected by parr overwinter. Low velocity, stream and ice-edge areas are important habitats for nocturnally active juvenile Atlantic salmon during winter.

KEY-WORDS: winter / Atlantic salmon parr / ice formation / edge habitats / spatial distribution / nocturnal activity / habitat selection

INTRODUCTION

Understanding the effects of ice formation on the overwinter distribution of presmolt salmon parr is important to the management of anadromous salmonids, because carrying capacity of the stream environment during the fall and winter has been recognized as an important bottleneck in smolt production (Cederholm et al., 1988; Nickelson et al., 1992). Ice formation affects both changes in, and the distribution of, physical space overwinter (Maciolek and Needham, 1952; Needham and Jones, 1959; Berg, 1994; Cunjak and Caissie, 1994; Prowse, 1994). Little is know, however, about the specific relation between changes in the physical stream environment caused by ice formation and carrying capacity for presmolt parr, because it is unknown how ice formation, physical space, and the overwinter use of physical space by presmolt parr interact. Generally, trout (Cunjak and Power, 1986; Chisholm et al., 1987; Riehle and Griffith, 1993; Bonneau et al., 1995; Brown and Mackay, 1995) and salmon (Hillman et al., 1987; McMahon and Hartman, 1989) occupy low velocity habitats overwinter and associate with instream cover. The need for low water velocities and habitats with cover increases the potential for space limitations overwinter, if such habitats are in short supply or if ice formation renders them unsuitable. Thus, understanding the relation between ice formation, physical space, and the use of physical space by presmolt parr is critical for identifying conditions and habitats important to their overwinter survival.

The activity patterns of overwintering Atlantic salmon (*Salmo salar*) parr that define the scope of their winter habitat needs are dictated by the low water temperatures ($\leq$ 0.5° C) that typically exist in winter and the associated physiological constraints that low water temperatures impose (Rimmer et al., 1985; Cunjak and Power, 1987; Cunjak, 1988a). In response to declining water temperatures in the fall (Rimmer et al., 1984) and low water temperatures in winter (Cunjak, 1988b), juvenile Atlantic salmon have been observed to shelter within the stream substrate and also to seek shelter in a laboratory setting under simulated fall-winter temperature conditions (Fraser et al., 1993; Fraser et al., 1995). The foraging and general activity of juvenile Atlantic salmon at low water temperatures (2° C) has been observed to be limited to nocturnal periods (Fraser et al., 1993; Fraser et al., 1995). Thus, the overwinter activity patterns that dictate habitat needs and in turn salmon parr/physical habitat interactions may differ between day and night. Because nocturnal activity has been related to foraging (Fraser et al., 1993) and thus may represent a behavioral-energetic relation (Metcalfe and Thorpe, 1992) critical to overwinter survival, our analysis of the parr distribution/ice formation/physical space interaction focused on habitat selection and distribution of presmolt Atlantic salmon parr at night. The specific objectives of our study were to determine the effect of ice formation on the physical stream environment and the overwinter distribution of presmolt Atlantic salmon parr.

METHODS

This analysis was undertaken in the Rock River, a tributary of the West River located in southern Vermont, New England, USA (43' 08' W, 73' 25' N). The West River is a tributary of the Connecticut River. Typical flows in the Rock River range from 0.15 to 2.5 m^3/s. Unfed Atlantic salmon fry stocked annually in the Rock River in May at target densities of 30 to 50/100 m^2 generally produce yearling parr densities of 3 to 10/100 m^2 (McMenemy, 1995). Although several adult salmon have been observed in the Rock River, the majority of parr production is assumed to be from fry stocking.

We selected a 200-m representative reach of the Rock River, which consisted primarily of riffle habitat dominated by coarse substrates. Within a 55-m section of the 200-m reach, we recorded mid-column water velocity (nearest ± 1 cm/s) and water depth (nearest ± 1 cm) on December 12, 1994, representing a pre-ice condition, and on February 1, 1995, representing an iced condition. We used a point sampling system based on a stream centerline axis anchored by steel rods that roughly divided the 55-m section longitudinally at mid-stream. This system was systematic in that sampling was conducted at either 1-m x 1-m or 1-m x 2-m scales, with the centerline axis serving as the baseline for measurement. The centerline axis transect system was also used to map surface ice in the 55-m section on February 1.

For the analysis of parr habitat selection and distribution, we identified three periods in reference to stream icing: pre-ice (< December 15, 1994), iced (December 15, 1994 to March 15, 1995); and post-ice (> March 15, 1995). For parr habitat selection, we completed twelve snorkeling surveys from December 2 , 1994 to April 4, 1995 in portions of the 200-m study reach, including the 55-m section. Snorkeling was systematic in that both mid-channel and lateral habitats were searched as the diver swam in an upstream direction (Thurow, 1994). Ice formation at times precluded safe access to stream edge habitats. All snorkeling was conducted at night (19:30 to 01:00 hours) using a hand-held white light; thus, only parr observed resting on or above the substrate were enumerated. Water temperatures during the study period ranged from 0 to 3° C.

Upon encounter, the position of each parr (yearling and older) was marked with a numbered weighted flag. Parr were collected with a small dip-net, measured for total length (nearest ± 0.5 cm) in a water-filled metered tube, and immediately returned to the location of capture by the diver. The total lengths of other parr were estimated to the nearest ± 1 cm using reference objects (Baltz et al., 1991). The total length of yearling and older parr in the study area ranged from 10 to 19 cm and, at the outset of the study, approximately 43% were sexually mature males. Mid-column water velocity and water depth associated with the position of each parr encountered were recorded immediately after the completion of the diving survey. The position of each parr observed in the 55-m section was recorded using the centerline transect system to assess changes in overwinter distribution.

RESULTS

The exposed wetted stream surface area decreased by approximately 52% between the pre-ice and iced condition, with a channel width decrease ranging from 4.5 to 11.5 m (mean decrease ± SE: 8.6 m ± 0.4) (Figure 1). Between the pre-ice and iced periods, a decrease in water velocity up to 88 cm/s was observed at 56% of the 52 recorded points, whereas an increase in water velocity up to 124 cm/s was recorded at 44% of the points. The mean (± SE) point water velocity decrease was 34 cm/s (± 4) and the mean (± SE) point water velocity increase was 44 cm/s (± 8). For water depth a decrease up to 62 cm was observed at 35% of the 49 recorded points, whereas an increase in water depth up to 83 cm was recorded at 65% of the points. The mean (± SE) point water depth decrease was 11 cm (± 4) and the mean (± SE) increase was 18 cm (± 3). Significant differences in the lateral distribution of < 40 cm/s and > 80 cm/s water velocities were detected both within and among survey dates in the 55-m section (Komolgorov-Smirnov two-sample test; all $P < 0.05$; Figure 2A-D). Generally, on both December 12, 1994 and February 1, 1995, peaks in the distribution of < 40 cm/s water velocities occurred lateral to the near centerline peak in the distribution of > 80 cm/s water velocities (Figure 2A, B). Ice formation resulted in a marked redistribution of < 40 cm/s water velocities (Figure 2C).

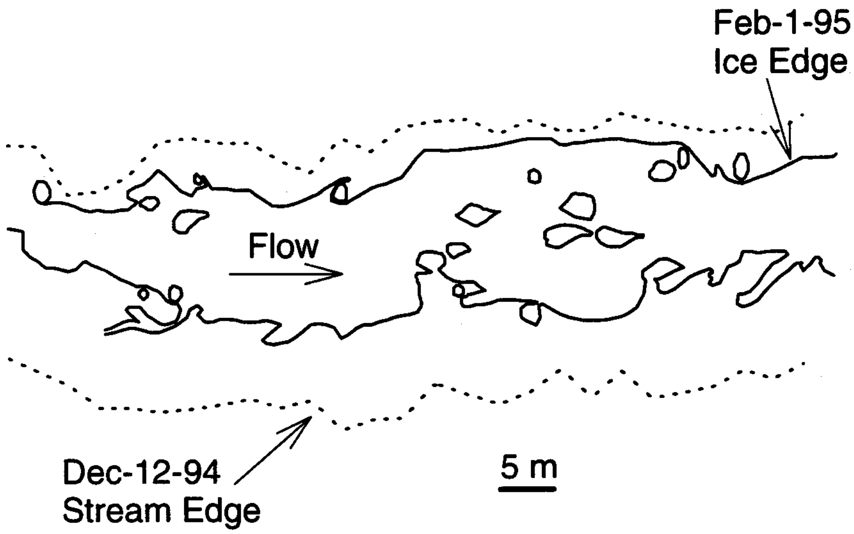

Figure 1: Overview of the 55-m study section on the Rock River, Vermont, showing the pre-ice stream edge on December 12, 1994 and the iced edge on February 1, 1995.

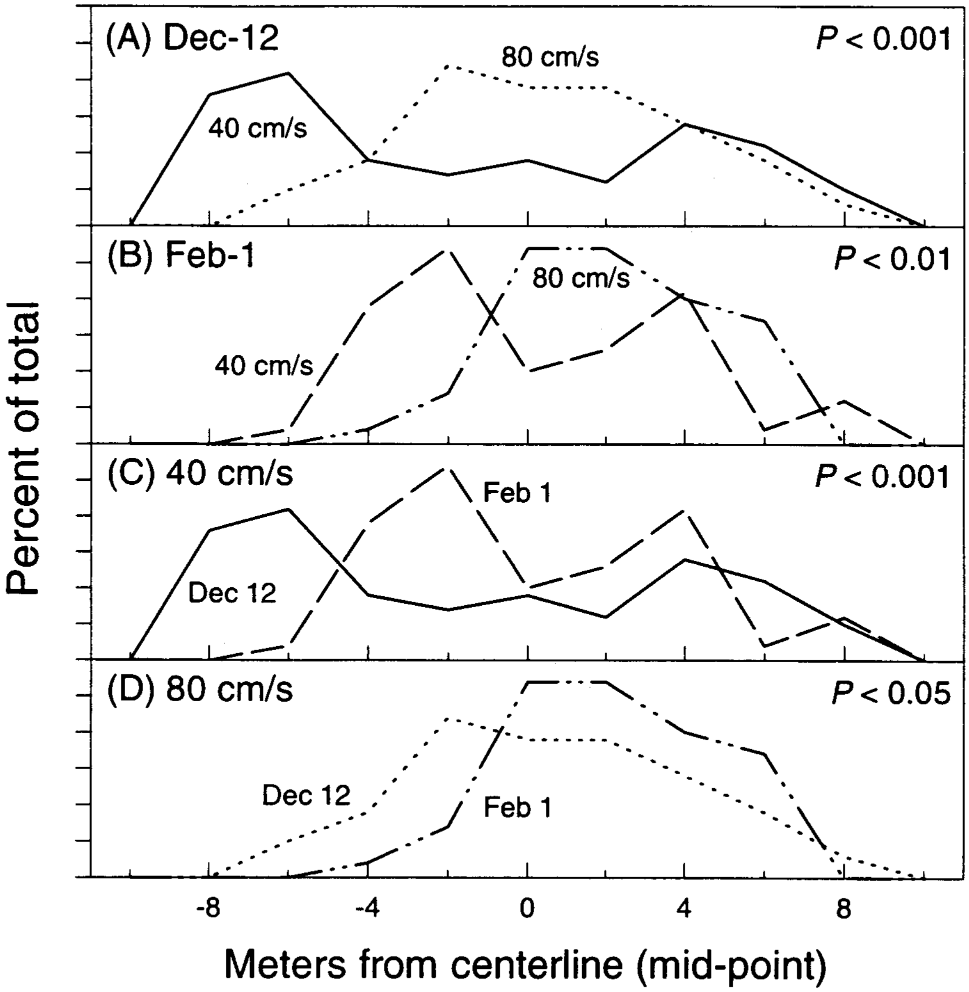

Figure 2: Lateral distribution (meters from stream centerline) of < 40 cm/s and > 80 cm/s water velocities recorded on December 12, 1994 (A) and February 1, 1995 (B), including a between date comparison of the distribution of < 40 cm/s (C) and > 80 cm/s (D) water velocities. Water velocities were recorded in a 55-m section of the Rock River, Vermont. P values are from the Komolgorov-Smirnov two-sample test for distributions. Sample size: Dec-12, < 40 cm/s, N = 107; Dec-12, > 80 cm/s, N = 74; Feb-1, < 40 cm/s, N = 52; and Feb-1, > 80 cm/s, N = 41.

The distribution of < 40 cm/s, ≥ 40 to ≤ 80 cm/s, and > 80 cm/s water velocities recorded in the pre-ice and iced periods (Table 1; I vs III) did not differ significantly nor did the number of salmon parr found at < 40 cm/s and ≥ 40 to ≤ 80 cm/s water velocities in the pre-ice, iced, and post-ice periods (II vs IV vs V). Overall, 75% of the salmon parr observed during the study period were found at water velocities < 40 cm/s, with only one parr recorded at a water velocity > 80 cm/s. In the pre-ice period, a significant difference was detected in the distribution of < 40 cm/s and ≥ 40 to ≤ 80 cm/s water velocities recorded and those used by salmon parr (I vs II), but this comparison for the iced period was only marginally significant (III vs IV).

Table 1: Distribution of water velocities recorded (habitat) and used (parr) by yearling and older Atlantic salmon parr during the pre-ice, iced, and post-ice periods. The percent of the row total is in parentheses and *N* = the number of observations. Results of χ^2 analyses for various comparison, as identified (ID) in the upper panel of the table, are reported in the lower panel. Water velocities were recorded in a 55-m section of the 200-m Rock River, Vermont, study reach. Only < 40 cm/s and ≥ 40 to ≤ 80 cm/s water velocity categories were used in the comparison of parr habitat use during the three study periods (II vs IV vs V), as well as the comparison of habitats recorded vs those used by parr in the pre-ice (I vs II) and iced (III vs IV) periods.

			Water Velocity Category			
Period	**Type**	**ID**	**< 40 cm/s**	**≥ 40 to ≤ 80 cm/s**	**> 80 cm/s**	***N***
Pre-ice	habitat	I	107 (34%)	134 (42%)	74 (24%)	315
	parr	II	25 (71%)	9 (26%)	1 (3%)	35
Iced	habitat	III	52 (32%)	68 (42%)	41 (26%)	161
	parr	IV	11 (69%)	5 (31%)	0 (0%)	16
Post-ice	habitat	---	---	---	---	---
	parr	V	24 (83%)	4 (14%)	1 (3%)	29

Comparison	**df**	χ^2	***P***	***N***
I vs III	2	0.26	0.877	476
III vs IV vs V	2	2.04	0.360	78
I vs II	1	10.13	0.001	275
III vs IV	1	3.67	0.066	136

Parr encountered in the 55-m section before stream icing were distributed across the stream width, with a peak in abundance occurring near the stream centerline (≥ -2 m to ≤ 2 m) (Figure 3A). Observations of parr during the iced and post-ice periods revealed a cross-sectional change in their distribution, with peaks in abundance occurring at $\leq$ -4 m and $\geq$ 4 m from the stream centerline. The difference in the number of parr observed in areas $\leq$ -4 m and $\geq$ 4 m and $\pm$ 2 m of the stream centerline in the pre-ice and post-ice periods was marginally significant (χ^2 test; $P < 0.07$), with the largest decrease in the abundance of parr occurring within $\pm$ 2 m of the stream centerline (Figure 3B).

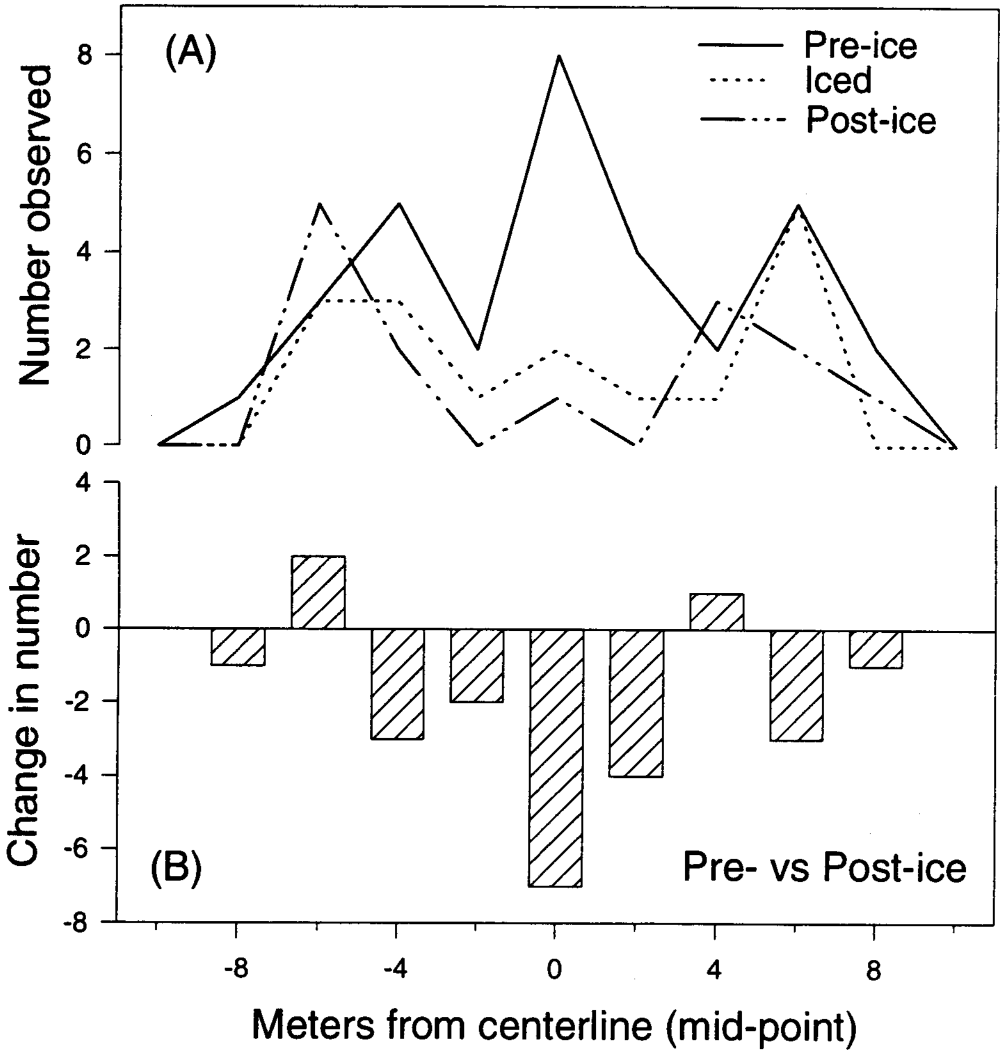

Figure 3: Lateral distribution (meters from stream centerline) of the number of yearling and older Atlantic salmon parr observed (A) in, and the difference in parr abundance between, the pre-ice and post-ice periods (B) in a 55-m section of the Rock River, Vermont. Sample size: pre-ice, $N = 32$; iced, $N = 16$; and post-ice, $N = 14$.

DISCUSSION

Significant changes in the physical stream environment resulted from ice formation, as the exposed stream channel width significantly decreased and both large increases and decreases in water velocity and depth were observed between the December 12, 1994 pre-ice and February 1, 1995 iced condition. The reduction in exposed wetted surface area due to ice formation is thus consistent with the frequently cited notion that physical space is likely to be an important factor affecting overwintering stream salmonids (Chapman, 1966). To evaluate this notion it is necessary to identify what aspect of the change in physical space wrought by ice formation may be most significant to overwintering salmon parr. For the iced condition that we considered, ice did not dramatically affect the proportional distribution of physical space in terms of water velocity categories, as the overall distribution of < 40 cms/, ≥ 40 to ≤ 80 cm/s, and > 80 cm/s water velocities was very similar in the pre-ice and iced condition. The reorganization and redistribution of physical space could be the most significant aspect of the effect of ice formation on the physical stream environment. Ice formation resulted in a reorganization of the physical habitat in terms of changes in water velocity and depth and also caused a significant lateral redistribution and compression of low (< 40 cm/s) and high (>80 cm/s) water velocities. We considered the understanding of the effects of ice formation on physical space to be a primary component in the identification of conditions and habitats important to the survival of overwintering salmon parr. That ice formation significantly altered the physical stream habitat implies that it is important to the overwinter stream carrying capacity.

The link between changes in the physical environment and effects on presmolt parr is largely determined by how parr use the available physical habitat. The distribution of water velocities selected by salmon parr was very similar overwinter and a large majority (75%) were observed in areas with water velocities < 40 cm/s even though these velocities represented only 33% of those measured. For the specific treatment of overwinter water velocity use/availability interactions, the selection of only a portion of the available water velocity profile by yearling and older Atlantic salmon parr is similar to brook trout (*Salvelinus fontinalis*) in high elevation streams in the western USA (Chisholm et al., 1987) and juvenile brown trout (*Salmo trutta*) at night in Norwegian streams (Heggenes et al., 1993). Because parr select only a portion of available habitats, there is a potential for a reduction in carrying capacity during iced conditions. To evaluate this potential, specifics of how Atlantic salmon parr use habitat and interact with conspecifics and other fishes in winter are needed. Our observations of juvenile salmon behavior at night in winter are similar to those reported for juvenile brown trout in the wild (Heggenes et al., 1993) and juvenile Atlantic salmon in the laboratory (Fraser et al., 1993). We found that parr were almost exclusively resting, essentially motionless, on the stream bottom, and seemed docile and tolerant of conspecifics that at times were in close proximity (within 10 to 20 cm). No instances of aggression were witnessed and feeding activity was rarely observed. Low water temperatures that reduce the scope of activity of Atlantic salmon parr may also reduce the amount of physical space required to both feed and shelter and that they are willing and able to defend. The limited activity and lack of aggression that we observed in salmon parr is consistent with the overwinter strategy of juvenile brown trout (Heggenes et al., 1993), which is one of minimization of net energy loss and basic maintenance rather than one of net energy gain and excessive activity. Measurements in terms of the scope of specific behaviors exhibited and habitats used are needed for relating habitat use and availability in a manner that is more meaningful to the analysis of carrying capacity/physical habitat interactions.

Changes in the lateral distribution in parr position that we observed between the pre-ice and post-ice periods may result from the movement of parr from centerline to lateral habitats and/or simply the maintenance of parr in lateral habitats and loss of parr in centerline areas. Losses of parr resulting from attrition through mortality are also likely important as overwinter mortality of juvenile Atlantic salmon may be substantial (Cunjak and Randall, 1993). We did not identify any centerline to lateral shifts in position of individual parr, nor were we able to relate the redistribution of parr to specific icing or flow events. Thus, it is unknown whether the decrease in abundance of parr near the stream centerline, as well as the maintenance of abundance in areas lateral to the stream centerline, were a function of avoidance, and/or selection, of specific habitat conditions. The lateral distribution of parr observed during iced and post-ice periods generally corresponded with the lateral distribution of < 40 cm/s water velocities the majority often selected. High flow events that lead to substrate scouring could also cause a redistribution of parr through an avoidance reaction, a factor that is exacerbated when edge ice or snow concentrates flow to open stream, mid-channel areas (Erman et al., 1989). Multiple anchor and frazil icing events were observed in the study area overwinter and both are believed to be important to overwinter redistributions of fish (Maciolek and Needham, 1952; Needham and Jones, 1959; Cunjak and Randall, 1993; Power et al., 1993; Cunjak and Caissie, 1994; Brown and Mackay, 1995). An important feature of the effects of high flow and ice formation is that they may be episodic and punctuated in nature, and thus redistributions of parr precipitated by them will likely be impossible to anticipate and difficult to quantify. Nonetheless, because conditions during such events may become limiting, their effects will need to be appreciated in the analysis of carrying capacity/parr distribution interactions.

The lateral distribution of parr during the iced and post-ice periods signifies the importance of edge habitats to overwintering Atlantic salmon parr and the role that ice formation plays in creating important habitat (Maciolek and Needham, 1952). The majority of parr observed during the post-ice period were found in positions that at some point during the winter period were covered by surface ice and a frazil ice curtain that extended to the stream bottom. Because we could not record water velocity below the surface ice nor safely access these areas by snorkeling, we could not specifically determine habitat conditions and parr habitat use in these areas. On several occasions, however, we observed parr moving within the limited space between the bottom of the frazil ice curtain and the stream bottom. Thus, while surface and frazil ice caused a significant redistribution of stream flows, ice itself may create important velocity refugia for overwintering Atlantic salmon parr.

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Impacts of logging operations

Impacts de l'exploitation forestière

IMPACTS OF LOGGING ON LAKE POPULATIONS OF BROOK TROUT (*SALVELINUS FONTINALIS*) AND THEIR HABITATS IN THE MASTIGOUCHE WILDLIFE RESERVE (QUÉBEC)

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ABSTRACT

Brook trout (*Salvelinus fontinalis*) fishing data time series, collected between 1971 and 1991, were analyzed "before", "during", and "after" logging operations for 20 lakes located within 200 meters of a logging area and 16 reference lakes undisturbed by logging. The fisheries parameters used as indicators of population dynamics, relative stock abundance and lake productivity are: mean weight of fish caught by anglers, catches per unit of effort (CPUE) and biomass per unit of effort (BPUE).

An analysis of variance (ANOVA) with three classification criteria (lake, year and logging period) free of interaction was conducted on each of the variables following a model similar to the "staircase analysis" of Walters *et al.* (1988). The mean weight of catches has remained unchanged over the three periods (before, during and after logging), while CPUE and BPUE have respectively decreased by 18% and 22% after logging operations in the area. These changes reflect a significant modification in population dynamics more than likely caused by logging. Results indicate that the negative impacts on aquatic fauna are felt more strongly on water bodies located in watersheds where deforestation is more severe; CPUEs are inversely correlated to a cumulative cutting index (CI) developed from physiographic parameters. A statistical analysis on the limnimetric data time series revealed an increasing interannual trend of the spring-flood discharge in the order of 8%. These results suggest that the observed variations in fishing success may be partially attributed to a decrease in recruitment caused by losses of potential reproduction habitats. The cause to effect links and the mechanisms associated with changes affecting fish populations following deforestation remain to be clarified.

KEY-WORDS : Biomass / Logging operation / Population dynamics / Exploitation / Habitat / Hydrology / Impact / Lake / Brook trout / Sportfishing / *Salvelinus fontinalis*.

INTRODUCTION

Disturbances affecting an ecosystem induce regulatory mechanisms which generally stabilize its intrinsect functions (Margalef, 1981). Thus, primary productivity and certain aspects of an aquatic community metabolism are remarkably homeostatic whenever a waterbody is modified by man-made disturbances (Odum, 1985). Fish populations have compensation mechanisms which allow them to adjust to environmental changes (O.M.N.R., 1983; Power *et al.*, 1994). This is the case for brook trout which shows a great stability and a high ecological valency potential.

These adaptation processes also apply in situation of population exploitation. In salmonid populations just recently fished, age at sexual maturity decreases, growth rate increases, numbers decrease and recruitment becomes more variable (Regier et Loftus, 1972). These adaptations are reflected in fisheries parameters such as total catches, average weight of catches and fishing success, which usually stabilize after a few years when the exploitation level reaches equilibrium with the population dynamics. Fish stocks are assumed stable when exploitation parameters do not vary for a constant fishing effort. But any changes in the average weight of the catches or in the relative abundance of individuals indicate that fish populations are responding to new life conditions or to a stressful situation. A fluctuation in fish biomass usually reflects a change in the productivity of the lake.

Although the impacts of logging operations on aquatic ecosystems and on salmonids vary in time and geographical regions, they are relatively well documented for lake environments (Lareau and Bérubé, 1992; Meehan, 1991; Salo and Cundy, 1987). Several studies indicate that logging may sometimes cause an increase of dry-weather flows (Dickison *et al.*, 1981; Kachanoski and De Jong, 1982) and of rain peak flows (Hetherington, 1982; Swanson and Hillman, 1977) and melting floods (Cheng, 1980; Dickison and Daugharty, 1982; Golding, 1981). According to Plamondon (1993), it is still difficult to generalize the impacts of logging on water flows because the characteristics of the watershed combine with local climates to produce different hydrological responses.

Logging may also induce a rise of water temperature in the summer (Beschta *et al.*, 1987; Feller, 1981; Plamondon and Thomassin, 1981) and a significant reduction of dissolved oxygen in streams (Hamelin, 1978 *in* Plamondon, 1993). Logging can also affect water quality by inducing an increase of nutrient concentrations, mineral salts and sediment transport (Durocher and Roy, 1986; Elouard, 1981; Hetherington, 1987; Nicolson *et al.*, 1982). These effects of logging may all have negative impacts on fish (Roberge, 1996).

While data are readily available for small streams, there is very little information available on lakes disturbed by logging conducted within the watershed, especially in Québec. This study attempts to fill in the gaps; its specific objective is to analyse brook trout (*Salvelinus fontinalis*) exploitation data, as an indicator of population status, in relation with logging operations conducted since 1978 in the Mastigouche Wildlife Reserve (Québec).

MATERIAL AND METHODS

Study area

The Mastigouche Wildlife Reserve, located northwest of Trois-Rivières, covers 1,573 km^2 between longitudes 73°02' W. and 73°40' W., and latitudes 46°28' N. et 46°53' N. The territory is not very mountainous, altitude ranging from 275 to 580 m. Mean annual temperature is 2.5 °C (Ferland and Gagnon, 1974). Mean annual precipitation is 1,120 mm, with a little more than one third as snowfalls (Villeneuve, 1967). Surface deposits consist for more than 90% of thin and deep tills. Other deposits are mostly sand of fluvial or glacio-lacustrian origin. Relatively well drained soils are comprised of 83% podzols and 17% brunisoils. The study area is in a mixed forest of deciduous trees, belonging for the most part to the sugar maple-yellow birch climacic domain (Thibault, 1985). Drainage of the Reserve flows through three watersheds: du Loup (876 km^2), Saint-Maurice (528 km^2) and Maskinongé (168 km^2) rivers. The total hydrographic system consists of 52 secondary watersheds ranging from 0.8 to 172 km^2.

The Reserve includes 497 lakes, about half of which are exploited by sportfishing. The preferred species is the brook trout (*Salvelinus fontinalis*), but some lakes are renowned for lake trout (*Salvelinus namaycush*) and land-locked salmon (*Salmo salar*) fishing. Several lakes, located in the central southeastern and western parts of the Reserve have many other fish species including some important competitors of the brook trout: the white sucker (*Catostomus commersoni*), the creek chub (*Semotilus atromaculatus)*, the pearl dace (*Semotilus margarita*) and the common shiner (*Notropis cornutus*). The indigenous and allopatric populations of brook trout are mostly located in lakes of the northern portion of the Reserve, upstream from lake Bourassa (Archambault and Lafleur, 1992).

For the most part, the Reserve has been logged since 1930. Logging gradually extended toward the north since the early 1970s; the most recent cuts (1980 to 1990) are found mainly in the northern part of the Reserve (Walsh, 1992).

Lake selection

Two groups of lakes were selected: disturbed and reference lakes. The first group includes 20 lakes where clear-cutting occurred within 200 m between 1978 and 1987. The second group comprises 16 lakes undisturbed by nearby logging. Reference lakes are in all manners comparable to disturbed lakes.

The lakes selected are inhabited by allopatric populations of brook trout alone or in a simple association with the northern redbelly dace (*Phoxinus eos*). As recommanded by Levings *et al.* (1989), disturbed lakes were selected on the basis of the availability of a long time series of brook trout fishing data which includes a minimum of two years for each of the periods before, during and after logging and spans from 1971 to 1991. Lakes smaller than 5 ha were rejected because they are unstable, the water replacement rate being generally too short. Some lakes are too acid for brook trout and the recruitment is unsuccessful; these were deleted from the list of 131 lakes where water physico-chemistry studies have been conducted in recent years by Tremblay (1991) and by MEF (unpublished data). In these acid lakes, the concentration of filtered aluminum is greater than 299 µg/l, the calcium content is less than 55 µeq/l and the pH is below 5.2. Lakes which have been stocked and those where poaching is suspected were also excluded in the analysis. Finally, lakes where one or more beaver dams occurred within one kilometer up a tributary or down the outfall were also rejected; these obstacle can restrain access to spawning grounds and therefore reduce the brook trout production yields.

The reference lakes are distributed on the territory in the same proportions as the disturbed lakes: about two thirds in the du Loup river basin, the remaining fraction in the Saint-Maurice river basin. Reference lakes have a fishing data time series generally complete for the 1971 to 1991 period; they are in the same size range (8.7 to 56.2 ha; $\overline{x} = 21.5 \pm 3.2$ ha) as disturbed lakes, which range from 5.0 to 65.4 ha ($\overline{x} = 16.7 \pm 3.1$ ha). A comparison of the mean surface areas of the groups showed no significant difference (Student "t" test = 2.45; P≥ 0.05).

Data processing

The five fisheries parameters used in the analysis are: fishing effort, total catches, mean weight of the catches, catch per unit of effort (CPUE) and biomass per unit of effort (BPUE). Fishing effort is estimated in number of hours fished, obtained from fishermen as they leave the Reserve. The total catch is the total number of fish caught in each lake during one fishing season. The mean weight of the catches (g) was calculated by dividing total weights (kg) by the number of fish caught. CPUE (catch/hour) is the average number of fish caught per fishing hour; it is basically a measure of fishing success. BPUEs (kg/h) represent the total quantity (in weight) of fish caught per fishing hour.

In order to determine whether one or the other of these parameters was influenced by logging, statistical analyses were first conducted on data from disturbed lakes to outline the major temporal trends of the fisheries parameters for each fo the three periods. The first period deals with fishing data collected in years preceding a clear-cutting. The second period consists of fishing data recorded during the logging operation and during the two years following the operation. This precaution ensures that the life cycle of brook trouts born during the final year of cutting is adequately covered, maturity being generally reached at two years in exploited populations. The third period groups together data collected after the second year following the end of logging. A series of analysis of variance with only one classification criteria (single ANOVA) was conducted for each lake individually, followed by multiple comparisons between the periods with a Tukey H.S.D. test for each parameter.

Disturbed and reference lakes were then classified in a matrix to verify which portions of the fluctuations of each fisheries parameters were related to interannual variations and to logging operations. This grouping of all lakes in a single matrix also increased the statistical power of the analysis and introduced an error factor representing fortuitous differences between the lakes and between years for a given lake. Since logging started at different times for different groups of lakes, this analysis model can be associated to the staircase analysis (Figure 1) described by Walters *et al.* (1988). This analysis model provides better control over the interactions of the time factor on the logging operations and it is also recommanded for estimating transitory responses to management measures or to environmental disturbances when they do not occur all at once.

The model includes a period before disturbance "t_c" (lakes before logging) and a period of disturbance "K" (K_1 = lakes during logging and K_2 = lakes after logging) for each stratum "m", where "n" represents the reference stratum (undisturbed lakes). In our case, strata were based upon the year when logging started around the lakes included in the study. Strata were subdivided in three-year periods (1978-1980: n = 11; 1981-1983: n = 2; 1984-1986: n = 7).

An analysis of variance with three classification factors (lake, year and logging period) was conducted on the data matrix for mean weight of the catches, CPUEs and BPUEs. The multifactorial analysis of variance was followed by multiple comparisons between the periods of cutting with a LSD test "Least Significant Difference" for each variable.

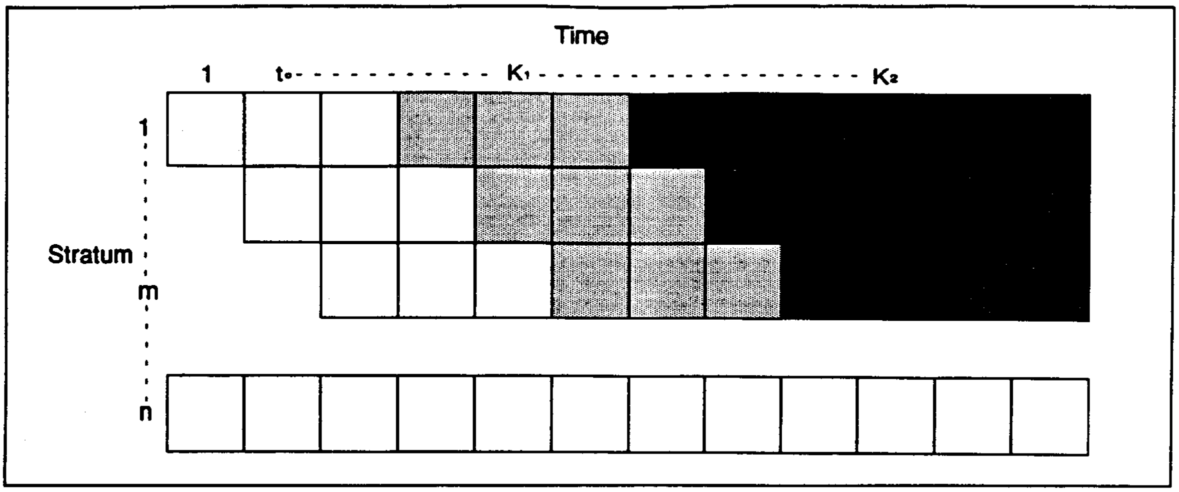

Figure 1 . Diagram of the staircase analysis used to measure the influence of logging on fisheries parameters. Units disturbed in time are represented by a shadowed area. Detailed explanations are given in the text (from Walters *et al.*, 1988).

A cumulative cutting intensity index was calculated from biophysical variables in order to assess if there was a relationship between a change of values in fisheries parameters in target lakes and the extent of clear-clutting. The index is calculated by:

(1) $$CI = CI_{bd}\,(CI_l + 1)$$

where, CI = cumulative cutting intensity index;
CI_{bd} = watershed cutting intensity index potentially affecting a given lake (zone located upstream from the lake discharge);
CI_l = lake cutting intensity index in the zone directly influencing the lake via surface runoff waters.

We have used $(CI_l + 1)$ to avoid obtaining a zero CI value when logging has not been conducted directly near the lake. CI can be calculated for a reference lake with a CI_{bd} value other than zero. Also, the addition $CI_l + 1$ counters the undesired effect (results lower than the actual indexes) induced by the multiplication of two values lower than 1.

Partial indexes CI_{bd} and CI_l can be estimated by the following equations:

(2) $$CI_{bd} = {}^{S_{ct}}\!/_{S_{bd}}$$

where, S_{ct} = total cutting area (ha) in the watershed upstream from the lake;
S_{bd} = total area (ha) of the watershed upstream from the lake.

(3) and $$CI_l = \left({}^{S_{c1}}\!/_{S_{lc}} \bullet {}^{\overline{p}_1}\!/_{\overline{d}_1}\right) + \left({}^{S_{c2}}\!/_{S_{lc}} \bullet {}^{\overline{p}_2}\!/_{\overline{d}_2}\right) + \ldots + \left({}^{S_{ci}}\!/_{S_{lc}} \bullet {}^{\overline{p}i}\!/_{\overline{d}i}\right)$$

where, S_{ci} = surface area (ha) of a cutting zone "i" where run off waters flow directly in the lake;

S_{lc} = surface area (ha) of the lake;
$\overline{p}_i$ = mean slope (%) between the cutting zone "i" and the lake;
$\overline{d}_i$ = mean distance (m) between the cutting zone "i" and the lake.

All these variables were measured from eco-forest maps MRN (1992), scale of 1:50 000, digitalized by MEF (1994).

Prior to the application of the parametrical analysis models, we tested the normality hypothesis with a Kolmogorov-Smirnov adequation test applied to each parameter. A F-max test and an examination of residuals dispersion confirmed that the variances homoscedasticity condition was respected. Log and square-root transformations were applied to normalize data not following a Gauss distribution. Statistical processing was conducted according to Sokal and Rohlf (1981). We used the SAS software (SAS Institute inc., 1990) and the WQSTAT II software (Phillips *et al.*, 1989).

RESULTS AND DISCUSSION

Lake Henry excepted,a singular case, there are three major patterns in the evolution of sportfishing data (Table 1). The first trend relates to the lakes in group A (categories 1, 2 and 3), where there are no significant variations of the fisheries parameters associated with logging, independant of fishing pressure. The second trend is characterized by a significant increase ($P < 0.05$) of the mean weight of the catches or by a significant decrease ($P < 0.05$) of the CPUEs and BPUEs calculated for five lakes in group B (categories 4 and 5), fishing effort having remained stable over all the fishing seasons. These changes indicate a notable modification of the population dynamics, likely caused by logging operations. In this group, the changes are more evident on lake au Tremble. The third trend shows a significant fluctuation ($P < 0.05$), in the average weight of reported catches in lakes of group C (categories 7, 8 and 9), often associated with a decrease in CPUEs or BPUEs. This result also suggests possible changes in population dynamics which may be attributed in part to the effects of logging on fish habitat. However, fishing effort has increased significantly ($P < 0.05$) and overfishing may have contributed to the decrease in fishing success over the years. This issue remains uncertain.

In general, results reveal that the negative impacts on fish populations are felt more strongly in watersheds where cutting intensity was greater. For lakes in group B, the mean CI ($\overline{x} = 2.298 \pm 1.807$) is 7.2 and 2.3 times higher than the mean CI recorded for lakes in group A and group C respectively. The mean rank (Wilcoxon score) of group B (CI=13.67) is statistically higher (Kruskal-Wallis "H" test by approximation du $\chi^2 = 7.31$; $P = 0.026$) than the group A ranking (5.57). These two values are not significantly different from the value for group C (CI=11.50). Also, CI is inversely correlated with the CPUEs (Spearman $r = -0.66$; $P = 0.042$) but it is not positively linked to BPUEs ($r = 0.43$; NS) and to the average weight of catches ($r = 0.35$; NS).

For reference lakes, the mean value of each exploitation parameter is significantly lower for "references alone" than for disturbed lakes alone "before" logging (*t*-test, $P < 0.05$; Table 2). This verification eliminates the possibility that some reference lakes may have been selected arbitrarily because of their greater fisheries potential. For disturbed lakes, the mean weight of the catches is 172 g and it remains unchanged ($F = 1.46$; NS) over the different periods, while CPUE and BPUE values decrease by 18% and 22% respectively with the occurrence of logging in the area (LSD-test ; $P < 0.05$). Figure 2 schematizes the results.

Table 1. Classification of the 20 disturbed lakes according to the evolution of brook trout fisheries statistics between "before", "during" and "after" logging operations in the Mastigouche Wildlife Reserve, between 1971 and 1991.

Group	Category	Fisheries parameters				Lake	Population status	Hypothetical cause	Cumulative cutting intensity index (CI)
		Effort	Weight	CPUE	BPUE				
A	1	→	→	→	→	Carroll	Stable	—	0.113
						Daphnies (des)	Stable	—	0.636
						Log	Stable	—	0.068
						Méta (du)	Stable	—	1.255
						Petit	Stable	—	0.061
	2	↗	→	→	→	Dernier	Stable	—	0.047
	3	↗↘	→	→	→	Huppé	Stable	—	0.070
									($\bar{x} \pm S = 0.321 \pm 0.463$) rank = 5.57
B	4	→	↗	→	→	Effilé	Changing	Logging?	0.125
						Moyen	Changing	Logging?	0.078
						Recto	Changing	Logging?	2.365
						Victoire	Changing	Logging?	3.210
	5	→	→	↘	↘	Tremble (au)	Changing	Logging	3.900
	6	↗↘	→	↘	→	Poivre (du)	Changing	Logging?	4.111
									($\bar{x} \pm S = 2.298 \pm 1.807$) rang = 13.67
C	7	↗	↗	↗↘	→	Baie (de la)	Changing	Fishing	0.664
	8	↗	↘	→	↘	Agonik	Changing	Fishing/logging?	0.702
	9	↗	→	↘	↘	Doré	Changing	Fishing/logging?	1.130
						Fox	Changing	Fishing/logging?	2.190
						Regnières (à)	Changing	Fishing/logging?	0.548
						Verso	Changing	Fishing/logging?	0.672
									($\bar{x} \pm S = 0.984 \pm 0.623$) rank = 11.50
D	10	↘	↗	→	→	Henry	Changing	Fishing	0.009

→ No significant difference in the evolution of the parameter ($P \geq 0.05$)

↗ Significant increase between the periods "before" and "after" logging ($P < 0.05$)

↘ Significant decrease between the periods "before" and "after" logging ($P < 0.05$)

↗↘ Significant increase between the periods "before" and "after" logging followed by a significant decrease during the period "after" logging operations ($P < 0.05$)

Table 2. Average weight, catches per unit of effort (CPUE) and biomass per unit of effort (BPUE) of brook trout yielded by sportfishing "before", "during" and "after" logging operations in 36 lakes of the Mastigouche Wildlife Reserve, between 1971 and 1991.

Cutting period	Average weight of catches (kg)	CPUE (catch/h)	BPUE (kg/h)
Reference group[1]			
References alone (n = 324)	0.160 ± 0.030	1.20 ± 0.36	0.19 ± 0.08
Before alone (n = 148)	0.174 ± 0.094	1.51 ± 0.70	0.24 ± 0.12
	(t = 1.33; P = 0.036)[4]	(t = 2.33; P < 0.0001)	(t = 1.46; P = 0.005)
Treatment group[2]			
Before[3] (n = 472)	0.173 ± 0.067	1.38 ± 0.59	0.23 ± 0.14
During (n = 68)	0.175 ± 0.058	1.27 ± 0.51	0.21 ± 0.12
After (n = 135)	0.169 ± 0.077	1.13 ± 0.69	0.18 ± 0.16

1 Arithmetical mean plus or minus the standard deviation.
2 Sample mean estimated by the ANOVA model plus or minus the standard deviation.
3 The period "before" logging ooperations includes the 16 reference lakes.
4 Student "t" test comparing the means of the parameters of the two entities in the reference group.

The more probable explanation to these results is that the disturbance of the aquatic environment caused by deforestation was sufficient to create a destabilization of population dynamics which resulted in a decrease of the relative abundance and the annual production of brook trout. The decrease in abundance may be the result of a decrease in the recruitment of the younger individuals in the population, as suggested by the decrease in CPUEs from 1.4 to 1.1 fish/hour. Similar results have been reported by Hartman (1987a) and confirmed by Hartman and Scrivener (1990) who recorded a decrease in adult trout and juvenile salmon densities following a logging operation near water bodies unprotected by a forest belt. Studies by Moring (1981) support the same conclusions: a population of reticulate sculpins (*Cottus perplexus*) was reduced for five years following logging.

During the last 21 years mean annual flow characteristics (12.3 m^3/s) in the study area did not vary significantly (seasonal Kendall test = - 2.38; NS). However, the analysis of the time series of peak flood flows ($\bar{x} = 20.5 \pm 11.3$ m^3/s) revealed a significant positive trend (seasonal Kendall test = 0.813; $P < 0.05$) of 1.7 m^3, an increase of 8.2% since 1978 (Figure 3), when logging began for the period covered by this study. Among possible changes affecting habitats and salmonid populations, Heede (1991) and Hicks *et al.* (1991) report that an increase in peak flow can induce mortality of embryos caused by progressive movements of stream beds, and that an increase in fine sediments over spawning grounds can reduce the reproduction success and the abundance of benthic organisms fed upon by fish.

It is possible that higher peak flows during the incubation period may reduce the survival rate of eggs until

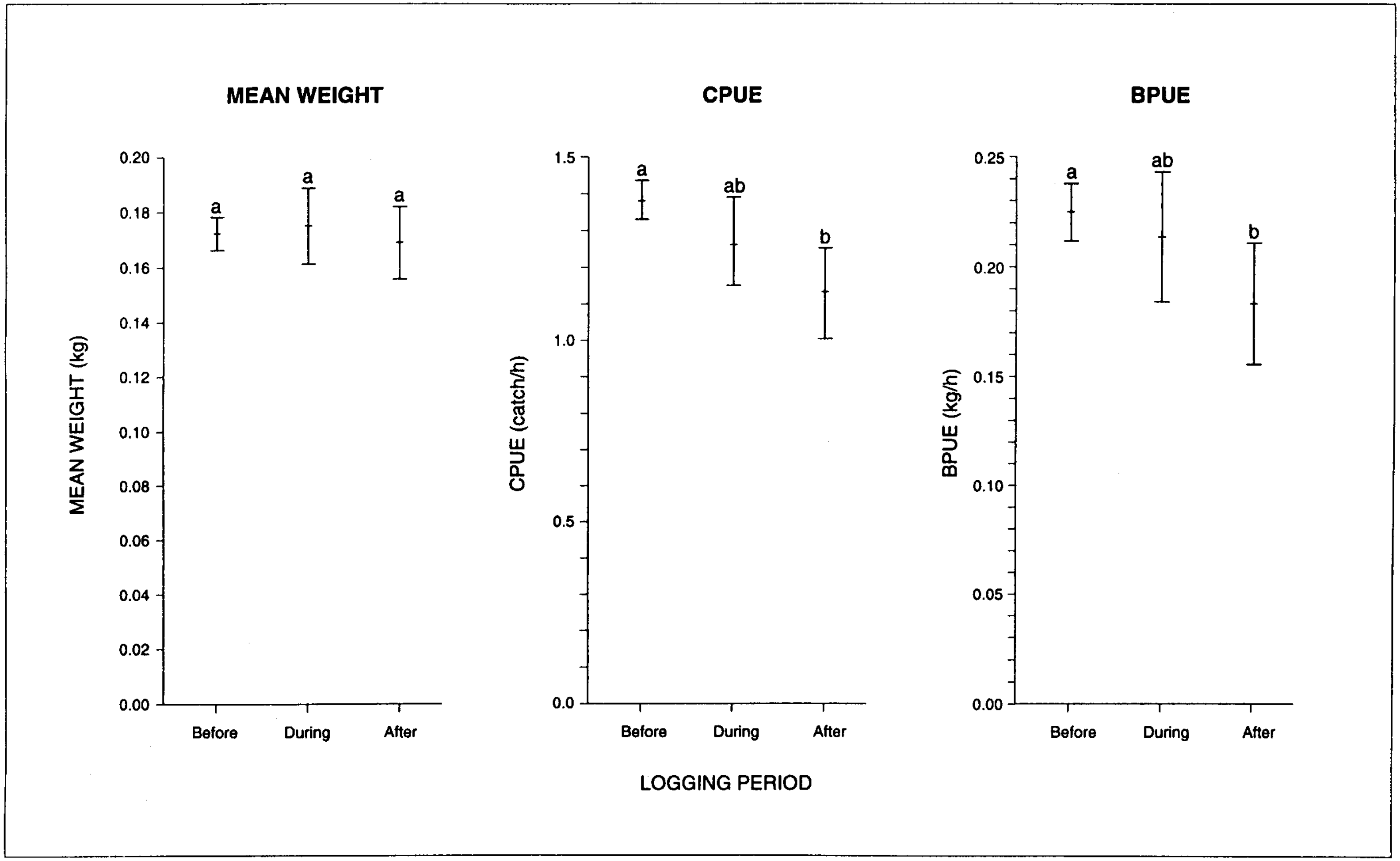

Figure 2. **Dispersion diagram representing the variation in average weight of catches, in CPUEs and BPUEs of brook trout relative to the periods of logging in the Mastigouche Wildlife Reserve, from 1971 to 1991. Data are the means with confidence intervals at 95%. Means indentified by the same letter are not significantly different as shown by a multifactorial ANOVA, followed by a multiple comparisons test (LSD test; $P \geq 0.05$)**

emergence. This hypothesis is supported by Cederholm and Reid (1987) and Scrivener (1987). According to Eaglin and Hubert (1993), the input and deposit of fine mineral particles in small tributaries to the lakes, caused by road construction and inadequate stream crossing works, may seal brook trout spawning beds which may result directly in a reduction of the recruitment rate in the lakes. This type of negative impact is conditioned by , among other things, the type of surface deposits and the width of the shore vegetation strip. Studies by Gregory *et al.* (1987), Hawkins *et al.* (1983), Murphy *et al.* (1981) and Scrivener et Brownlee (1989) underline the negative impacts of sediments on fish habitat.

Hartman (1987b) and Hicks *et al.* (1991) observed that logging may induce additional quantities of non-indigenous organic matters in streams, which could result in increasing primary productivity and stimulation of fish growth in oligotrophic environments. Although growth was not examined in our study, our results do not indicate any increase in fish size after deforestation, since the mean weight of the catches did not change markedly during or after logging.

Immediately after the cut, salmonid biomass tends to increase when the transitory gain in productivity may compensate for habitat disturbance (Bisson and Sedell, 1984; Burns, 1972; Grant *et al.*, 1986). Our results do not support these observations since the biomass of the catches decreased significantly after logging. The decrease in biomass may have resulted from a lower recruitment in the populations which was not counterbalanced by faster growth. It is a known fact that lakes disturbed by logging have a greater proportion of small-sized zooplankton (< 500 μm) than lakes undisturbed by logging (B. Pinel-Alloul, Univ. of Montreal, pers. comm.; Watson and Kalff, 1981). If we assume a change in the structure of the zooplankton community, it could very well be that the brook trout's diet was modified in terms of energetic efficiency, thereby explaining the drop in fish biomass.

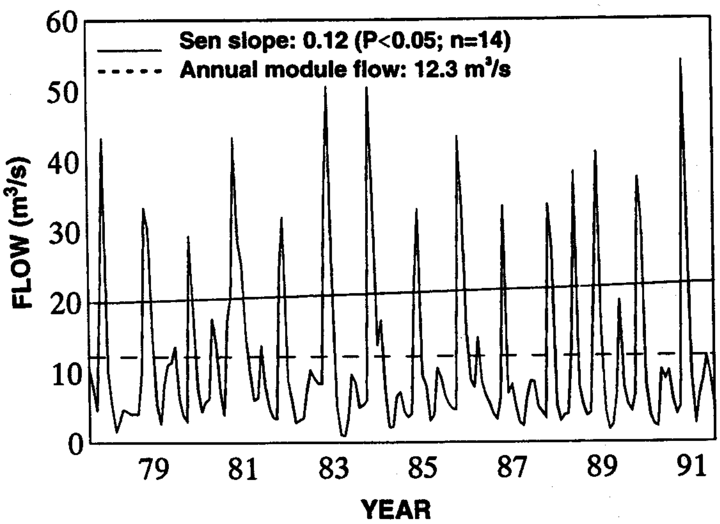

Figure 3. Interannual variation of spring flood flows recorded at gauging station n° 052805 on du Loup river between 1978 and 1991.

CONCLUSION

This study demonstrated that logging operations could have a marked influence on some indicative parameters of brook trout population dynamics in lakes. The decrease of CPUEs and BPUEs in certain lakes where fishing effort has been stable for the last 20 years confirms this statement. However, the cause to effect links or the mechanisms implied in these changes affecting fish populations are not yet clear.

Generally, it is accepted that the impacts of logging on the aquatic environment are usually felt over the long term (> 15 years). Thus, it is difficult to assess with any precision the direct effects on aquatic fauna. These impacts are complex and highly variable in a time-space frame, and depend upon the tolerance level of fish species (Holtby, 1987; Sullivan *et al.*, 1987). Water temperature variations, changes in primary productivity and habitat losses caused by erosion and sedimentation constitute the major problems most often mentionned.

The assessment of potential impacts on fish habitat of forest road construction and of stream crossing works deserves to be addressed in depth. Meanwhile, new ecological logging practices established in Québec in the past decade should help correct or improve certain deficient on-going situations.

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Conservation of wetlands and shore habitats

Conservation des milieux humides et côtiers

EFFECTIVE SHORE PROTECTION WITH INTEGRATED ENVIRONMENTAL ENHANCEMENT

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ABSTRACT

In an effort to arrest shore erosion at many coastal locations, it is necessary to construct structures in high wave energy zones. Current practice involves utilization of structures constructed using large armour stones, concrete and steel walls and a variety of other "hard" engineering techniques. Quite often, these structures do not add to the aesthetic and recreational attributes of a site and may impact significantly on the local environment.

Integration of bio-engineered components into the design of shore protection systems can be utilized, in certain cases, to enhance shoreline stabilization by providing enhanced biological habitat and ancillary water quality improvement. Thus the goal of a shoreline stabilization project changes to include not only the stabilization of the eroding area, but to increase the quantity and quality of habitat available to fish and waterfowl communities, while providing an effective and aesthetic control of shoreline erosion. Recent developments in the field of bio-engineering are outlined in this paper. Several specific applications are discussed.

KEY-WORDS: shore protection / bio-engineering / shore erosion / habitat enhancement / waves / bio-diversity / shoreline stabilization

INTRODUCTION

In both the engineered and natural environment, the flow of water often causes erosion. The causes must be understood before the problem can be addressed. In the coastal zone, the flow of water results from wave action, the associated runup and rundown, wave breaking, alongshore currents and the natural flow of water along-side and overtop of the shoreline. In addition to the interaction of the shoreline with water, a considerable amount of animal and human activity create additional stresses on the shoreline. Bio-engineering methods of shore protection offer a practical solution that can also create an aesthetically pleasing and environmentally beneficial "buffer zone". Bio-engineering, in this context, is the utilization of vegetation, either by itself or in combination with other defense mechanisms, depending upon the local environment. The other defense mechanisms may include the use of rock lining, offshore islands, wave screens and submerged shoals which limit the wave energy reaching a site. Quite often, these defense structure can be designed to provide significant enhancement to the environment, particularly in providing suitable fish habitat for spawning, feeding and hiding from predators.

The value of vegetation for protecting the soil depends on the combined effects of roots, stems and foliage. Roots and rhizomes reinforce the soil. Immersed foliage elements absorb and dissipate energy and may cause sufficient interference with the flow to prevent scour. In a sediment-laden environment, they may also promote deposition.

A shoreline requiring protection can be considered as two separate areas and thus habitat enhancement can be geared toward two communities; the high energy near shore environment and the onshore environment, which can be suitably modified to ensure low wave energy levels.

Enhancement of the near-shore zone can include construction of rock revetments as reef habitat, inclusion of submerged offshore structures to reduce wave energy levels reaching the shoreline and primary wave defense structures which provide habitat enhancement potential by the nature of their design (Figure 1). Selection of stone and design of its placement is developed in a manner to provide a reef like habitat beyond minimum stone placement required for the minimal shore stabilization. The effectiveness of both natural and artificial reef like habitats as fish community habitats has been well documented. Proper design and installation of rock will provide protective cover and feeding areas and will supply the needs for small aquatic and benthic organisms by providing protection from the high energy wave action and from larger predators.

Low wave energy areas can be created behind a primary defense such as an offshore rock structure or wave screen, or through the creation of lagoons behind stable control structures (Figure 2) . The development of constructed wetland pockets and areas for other shallow water plants can occur in these lagoons. This can be promoted by the establishment of "biological" rip rap in the form of brush and woody plant debris. These materials will provide a setting that will foster the accumulation of shore plants from wind blown seed banks. As the brush decomposes, it provides a limited release of nutrients to the developing plant community and is eventually replaced by living plants. The establishment of new habitat

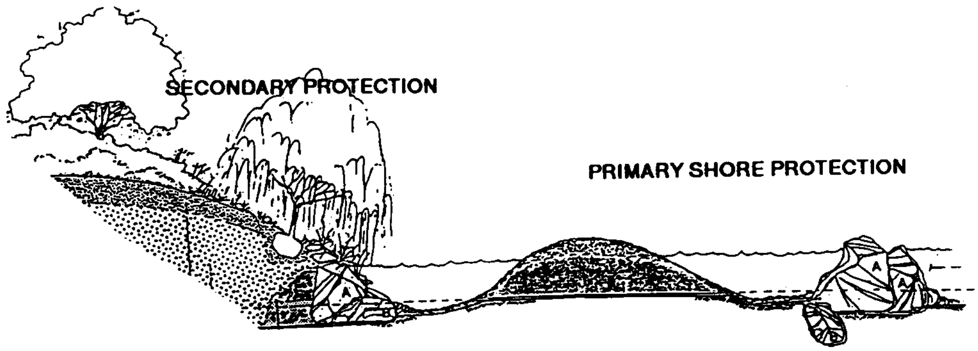

FIGURE 1 UTILIZATION OF SHOALS

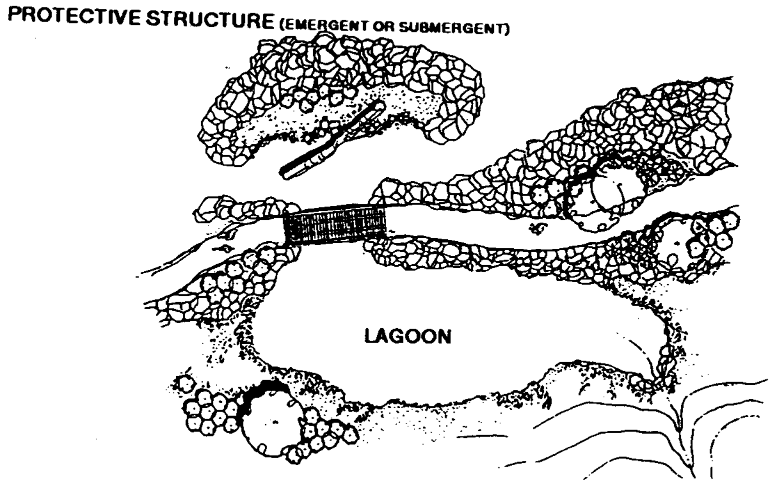

FIGURE 2 DEVELOPMENT OF A LAGOON CELL

will provide the opportunity for colonization by wetland plant and animal species that require quiescent waters. Transient use of the habitats by a variety of aquatic and migratory waterfowl is an additional potential for these environments.

GOALS AND OBJECTIVES

The designer is encouraged to consult specialists in the fields of coastal hydraulics, fisheries, geomorphology, biology, landscape architecture or any field that could make the project a success. The design of bio-engineered shore protection that functions environmentally requires a multi-disciplined approach. Usually, no individual has all the expertise required to ensure successful implementation.

The following geomorphological, hydraulic and biological changes may occur as a result of modification of the shoreline:

* Loss or elimination of aquatic vegetation
* Loss or elimination of backshore vegetation;
* Removal of specific nearshore bathymetrical features
* Modified substrate conditions;
* Modified hydrodynamic, flow, sediment and water quality regimes;
* Changes in nutrient conditions and reductions in food organisms;
* Aesthetic degradation;
* Reductions in habitat diversity and environmental stability;
* Increased water temperatures;

A shoreline is a dynamic system where impacts are difficult to predict . Engineered structures, when properly designed and constructed, can provide both species and habitat diversity and thereby mitigate potential adverse changes. However, the goals and objectives of the shore protection design must be correctly identified early in the design process. The designer must be aware of the design goals and objectives to correctly identify, size and locate the various functional elements within the system.

Bio-diversity within and adjacent to the shoreline is interrelated with the quality in updrift and downdrift areas. Changes to any one of these components may adversely impact on others.

Habitat Requirements

Aquatic life generally requires a habit that contains the following :

i) Sufficient water depth and volume for each life stage
ii) Adequate water quality with preferred ranges of temperature, dissolved oxygen, PH etc.
iii) A variety of continuous hydrodynamic conditions varying from deep water to shallow water for breeding and cover. Also flow conditions that sort bed load materials to provide a good environment for bottom dwelling organisms are advantageous.
iv) Adequate cover to provide shade, concealment, and orientation,
v) Adequate food to maintain metabolic processes, growth, reproduction.

Use of vegetation requires the following points to be considered.

Plant Selection

The principal plant groups that can be used are aquatic plants, grasses, shrubs and trees. Selection is based on consideration of the different roles to be performed by the vegetation as summarized in Table 1, taking into account the physical and chemical properties of the soil, the climatic conditions and the soil/water regime under which the plant must survive.

TABLE 1 ROLE OF VEGETATION IN BANK PROTECTION

Function	*Qualities required*
Interference	Above-ground parts which attenuate local flow velocity at soil surface
Protection	Dense surface cover of above-ground parts and surface roots which shield soil particles from erosion by flow
Restraint	Dense surface root structure which restrains surface soil particles from entrainment by flow
Reinforcement	Underground root structure which reinforces and improves strength of subsoil through dense root development at the required depth
Anchorage	Deep underground root structure which anchors the surface vegetation into the subsoil
Buttressing	'Propping' effect of individual plants which restrain a wider soil mass from movement
Transpiration	Large area of foliage which transpires soil moisture drawn in by root growth

Soil Requirements

Plants must be provided with the correct soil conditions to enable them to achieve their intended growth. The soil profile for the plant generally needs to be designed to at least 0.5 m depth, and sometimes considerably more, in order to optimize the root growth. Requirements for plant growth, such as low soil compaction, may apparently conflict with normal geotechnical requirements for stability and strength of the subsoil.

Surface Preparation

Proper ground preparation can enhance the establishment of vegetation. This may entail:

1. cultivation to produce an acceptable seedbed, or suitable trimming on steep slopes.
2. contouring to provide appropriate surface topography for drainage and future management.
3. scarification or ripping to relieve excessive subsoil compaction.
4. soil amelioration to improve soil structure, water holding capacity and/or fertility.
5. provision for short-term erosion control pending establishment of vegetation.

Vegetation Establishment

Five principal methods of seeding may be considered:

1. drilling, direct placement of seeds in the soil.
2. broadcasting, dry spreading seeds over the soil surface.
3. hydroseeding, spreading the seeds in a water slurry.

4. mulchseeding, wet or dry constituents with a heavy mulch applied dry, including pre-prepared seed mats).
5. hand seeding, including hand broadcasting and localized spot seeding of patches of seeds.

Pre-grown plants may be transplanted as turfs or planted individually as clumps. Vegetation establishment may take several growing seasons and is a seasonal activity that must be managed and maintained. The engineer must prepare and agree to specific management objectives and a management program with the owner/client. This is in order to ensure that the vegetation is maintained in a fit condition to perform its intended roles.

Zones and Horizons of Natural Protection

With natural methods of protection, and particularly methods involving the use of live material, the effectiveness of different materials is strongly dependent on their location in relation both to the dominant external water level and to the subsoil soil/water regime. To achieve effective protection using natural materials, the designer will almost inevitably need to use different methods of protection in different zones and horizons of the shoreline (Coppin and Richards, 1989.).

Use of Reeds

The emergent and marginal types of aquatic plants, such as the common reed, bulrush and great pond sedge, are frequently used for interference and protection purposes to form a protective margin along the shoreline at the waterline. They also encourage siltation by absorbing current flow energy, and thus reducing the sediment-carrying capacity of the flow. Reeds can be easily weakened by erosion and loosening of the soil around the rhizomes due to wave energy. It is therefore necessary to protect the zone containing roots from high-velocity flow or significant wave attack. Provided this is done, the stems and leaves will protect the shoreline bank above.

Uses of Shrubs and Trees

A limited range or trees are water-tolerant and can be used in bioengineering structures for bank protection in both the aquatic and damp zones. The willow, alder and black poplar are the principal water-tolerant species. Water tolerant trees and shrubs can perform any or all of the functions indicated in Table 1. In particular, a dense root structure is able to provide some protection as well as substantial reinforcement effect to enhance the stability of the shoreline bank both above and below the mean water level. The willow and poplar are particularly useful for bioengineering because they can be propagated from cut limbs. The cut limbs can be placed such that secondary root growth develops and shoots sprout from dormant buds.

Trees which are not water-tolerant do not have any major direct function in shoreline stabilization, although they may provide shade to control the growth of aquatic life as discussed earlier.

Use of Grasses

Grass is used very extensively in bank protection in the zones above the high water level. Grass roots cannot tolerate prolonged submergence periods. A wide variety of grass species and mixtures therefor are appropriate to satisfy the functional, environmental and management requirements for a protection scheme.

Shoreline improvements should be designed for the individual fish species. Specific requirements for reproduction, juvenile rearing and adult rearing with regard to feeding location, concealment from predators and competitors and sanctuary from flow extremes and ice formation varies between species.

Loss of the natural bathymetric features which are utilized by particular species as a result of implementation of shore protection could eliminate many of the requirements necessary to sustain significant bio-diversity along the nearshore area. In addition, removal of existing shoreline vegetation, in either the emergent of submergent zones would significantly reduce or eliminate the potential to sustain a fish population.

Utilizing Vegetation

In certain low wave energy environments, vegetation may be used by itself to provide suitable protection to an eroding shoreline. Reeds and other marginal plants can form an effective buffer zone by absorbing wave energy and restricting the alongshore flow velocity adjacent to the shoreline. They therefore have a protective value. Specific functions which they can perform include:

1. Absorbing and dissipating wave-wash energy.
2. Interference and protection of the shoreline bank from the flow.
3. Reinforcement of the surface soil through the root mat and prevention of scour of the bank material.
4. Sediment accumulation brought about by the dense plant stems.

Marginal plants require very wet ground and generally will not survive in water which is more than 0.5 m deep for long periods of time. They flourish in conditions of low flow velocity and their integrity is weakened by wave action in excess of 0.5 to 0.75 m. Different species offer different levels of protection with regard to wave energy dissipation. For incident wave conditions under 0.5 m, reed beds having a width of 2 to 2.5 m may dissipate 60 to 80% of the incoming wave energy. In areas with higher levels of wave energy, rip-rap and geotextiles may be used in conjunction with vegetation to provide effective bank protection (Figure 3)

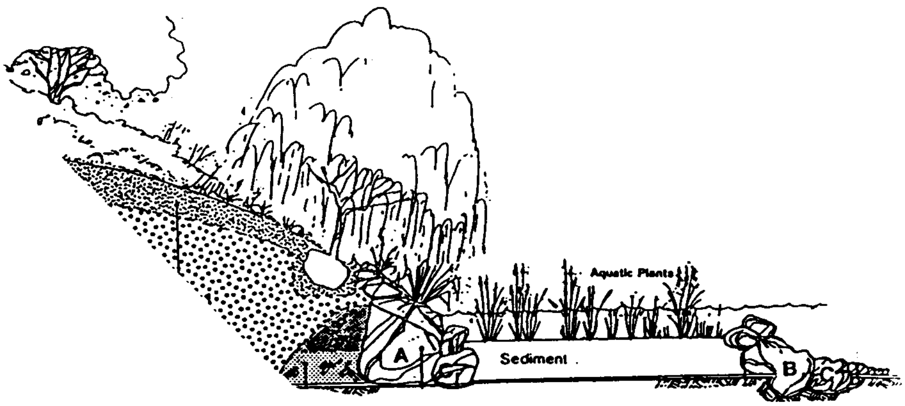

FIGURE 3 EMERGENT VEGETATION USED IN CONJUNCTION WITH STONE

In areas of high incident wave energy, an area of low wave energy can be created behind a primary defense such as an offshore rock structure or wave screen, or through the creation of lagoons behind stable control structures (Figure 4).

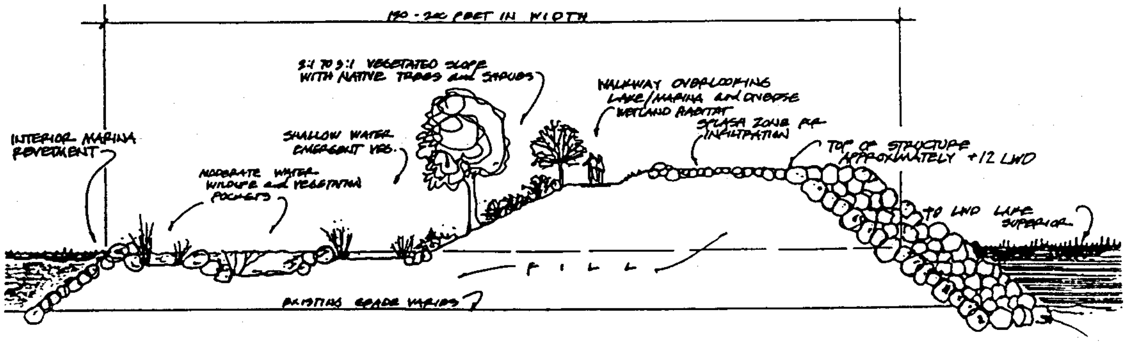

FIGURE 4 CONTROL STRUCTURE / LAGOON

The development of constructed wetland pockets and areas for other shallow water plants can occur in these lagoons. This can be promoted by the establishment of "biological" rip rap in the form of brush and woody plant debris. These materials will provide a setting that will foster the accumulation of shore plants from wind blown seed banks. As the brush decomposes, it provides a limited release of nutrients to the developing plant community and is eventually replaced by living plants. The establishment of new habitat will provide the opportunity for colonization by wetland plant and animal species that require quiescent waters. Transient use of the habitats by a variety of aquatic and migratory waterfowl is an additional potential for these environments.

Natural methods of protection, used by themselves, generally have low capital cost in comparison with conventional engineering methods. However, they may well have higher recurrent cost due to regular inspection, trimming and cutting and repair. In areas where a combination of conventional and bioengineered structures are required, recent experience at several sites on the Great Lakes has identified that these techniques may cost 20 to 30 percent more than conventional techniques alone.

Possible disadvantages are that natural protection schemes take time to mature and to become fully effective. Depending on the type, natural protection may take several growing seasons to reach the desired standard of protection.

Bio-engineering differs from other conventional forms of engineering in two key respects which strongly influence the design approach:

1. Bioengineering involves considerable practical experience and judgment, as opposed to the application of quantitative design theory or rules.
2. Careful management is required not only in the establishment of vegetation, but also in its aftercare over the initial growing seasons.

The principal functions which grass fulfils are those of interference, protection, root reinforcement and soil restraint. The surface root structure forms a composite soil/root mat which enhances the erosion resistance of the bare subsoil, and which is anchored into the subsoil by deeper roots.

The physical attributes of the grass plant which determine the effectiveness of grass for protection are:

1. length and stiffness of the sward.
2. surface area of grass blades.
3. strength and depth of root structure.
4. density of rhizomes, stolons and surface root structure.

The engineering function of grass may be augmented by the use of geotextile or cellular concrete reinforcement to form composite protection. With both types of reinforcement, the visual effect of grass is retained.

Erosion of grass cover by wave runup generally occurs by the scouring of soil from around the roots of a plant, thereby weakening its anchorage until the plant itself is removed by the drag of the flowing water. The effectiveness of grass protection can also be seriously reduced by any localized patches of bare soil or poor grass cover.

The rate of growth of different grasses varies considerably. Complete grass cover should normally be achieved by the middle of the first growing season while full protective strength of the sward is reached during the second season. Provision should be made for aftercare including mowing, fertilizing and weed control.

Grassland will slowly revert to scrub unless woody plants are removed, thus grass swards require continuous maintenance.

Use of Timber and Woody Material

A variety of timber and other dead woody materials can be used in the shore protection scheme usually fulfilling a reinforcement, protection and sometimes drainage functions (see Figure 5).

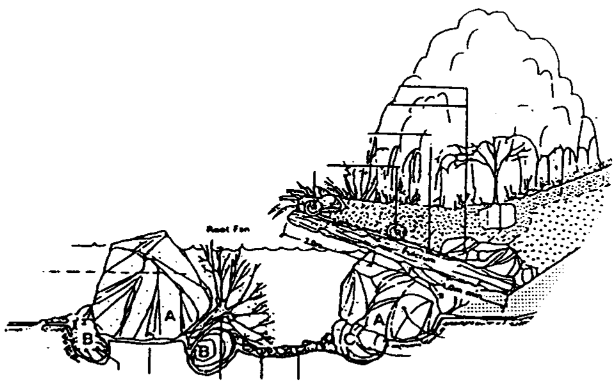

FIGURE 5 TIMBER (CANTILEVER) TO PROVIDE HABITAT AND SHADE

Natural hardwoods will retain their integrity for 5 to 10 years if built into the bottom of a bank below the water level. Out of the water they can last longer but the worst environment for timber is the alternately wet and dry zone around mean water level.

MONITORING

As part of the project design for the shoreline stabilization enhancements, a monitoring program is required. The purpose of the monitoring program is to measure the success and applicability of the enhancement methods to other shoreline projects.

Baseline habitat conditions should be assessed by observation and characterization of existing conditions. A plant survey and macroinvertebrate sampling of the near-shore benthic environment and a terrestrial plant survey should be performed to document existing plant and animal populations. Incidental observations of birds should be made as part of field work. Sampling of near-shore fish populations should be coordinated with local Natural Resource Ministries.

Post-construction monitoring of the establishment of biological communities should be completed to evaluate the success of a particular scheme.

EXAMPLE

Shore protection enhancements similar to those described in this paper have been successfully implemented at numerous sites on the Great Lakes, most notably in Canada at Red Rock Marina, Lake Superior; Thunder Bay Harbour, Lake Superior; Kingston, Lake Ontario; and various reaches of the St. Lawrence Seaway, and at Bender Park, Lake Michigan, Silver Bay, Lake Superior and in Louisiana (Gulf of Mexico) in the United States. The range of wave conditions appropriate at the sites range from 0.75 to 4.5 m at these various sites. Many other projects are in the process of implementation.

Bender Park Shoreline Habitat Enhancement

The Milwaukee County Department of Public Works (DPW) undertook a shoreline stabilization project at Bender Park, Oak Creek, Wisconsin, along approximately 6000 feet of Lake Michigan shoreline. This section of Lake Michigan shoreline is characterized by cliffs approximately 120 feet high, composed of silty clay tills and lacustrine deposits. The cliffs are unstable, and suffer severe erosion problems. Estimates from aerial photographs and historical maps indicate that the bluff line has receded up to approximately 500 feet over the last 70 years. Very little "beach" area exists at the toe of the bluffs, and the offshore environment consists of typically silt-laden water and patchy sand and gravel deposits over till on the lakebed.

The engineering design for the project called for recontouring of the shoreline slope and the use of rock revetment to dissipate wave energy to which the shoreline is exposed. This basic shoreline protection strategy will result in stabilized bluffs and a lake-level terrace constructed on lake fill and edged by rock rubble. Enhancement of the shoreline stabilization was implemented to provide enhanced biological habitat and ancillary water quality improvement. The goal of the shoreline stabilization enhancements is to increase the quantity and quality of habitat available to fish and waterfowl communities, while providing an effective and aesthetic control of shoreline erosion. Habitat enhancements were oriented toward the improvement of two communities: the high energy near-shore lake environment and the bluff toe lagoon system.

Enhancement of the high energy lake zone was achieved by the construction of the rock revetment fill as a reef habitat. Selection of stone and design of its placement was developed in a manner to provide a reef-like habitat beyond the minimum stone placement required for the minimal lakeshore stabilization.

Currently, the near-shore lake environment consists of a hard packed fine sandy clay bottom material, providing little habitat to sport fishes or populations of species that function as fish food (benthic invertebrates). Research by the University of Wisconsin Milwaukee Sea Grant Program indicated that the hard packed, high energy benthic environment supports only a few species of invertebrates, and low numbers of individuals of these species. The created environment will supply the needs for small aquatic and benthic organisms by providing protection from the high energy wave action and from larger predators. The protected, shallow-water nature of this habitat will allow the development of a food source, in the form of algae and microscopic algae-feeding invertebrates upon which the food chain of the fishery is based. The proposed habitat enhancement of the nearshore environment includes the design of a reef environment in three dimensions, such that the rock rubble extends lakeward from the shoreline in places to provide greater available reef habitat. The design will include a gradation of rock sizes to support protective cover for a variety of fish and fish food species.

Enhancements were also developed in the form of a bluff toe lagoon system. This zone is the area at lake level shoreward of the exposed rock rubble. Enhancement was provided for the area along the toe of the bluff, with the development of habitat in the area that has received the brunt of wind and wave action, and thus has not allowed the establishment of biological communities. Enhancement in this zone includes the development of pools near the toe of the bluff and the routing of stormwater runoff from the terrace into these pools. The pools and surrounding low land areas were developed into wetlands that will improve water quality and provide a shallow aquatic habitat protected from the wave action of Lake Michigan (Figure 6).

The benefits of the development of the bluff toe lagoon system are the creation of lake coastal habitats not present in this part of Lake Michigan. The establishment of new habitats will provide the opportunity for colonization by wetland plant and animal species that require quiescent waters. Transient use of the habitats by a variety of aquatic and migratory waterfowl is an additional potential for this environment. The lagoons will also provide a water quality improvement for runoff from upland areas to the west and from the lake-level terrace by providing filtering and settling mechanisms for stormwater drainage.

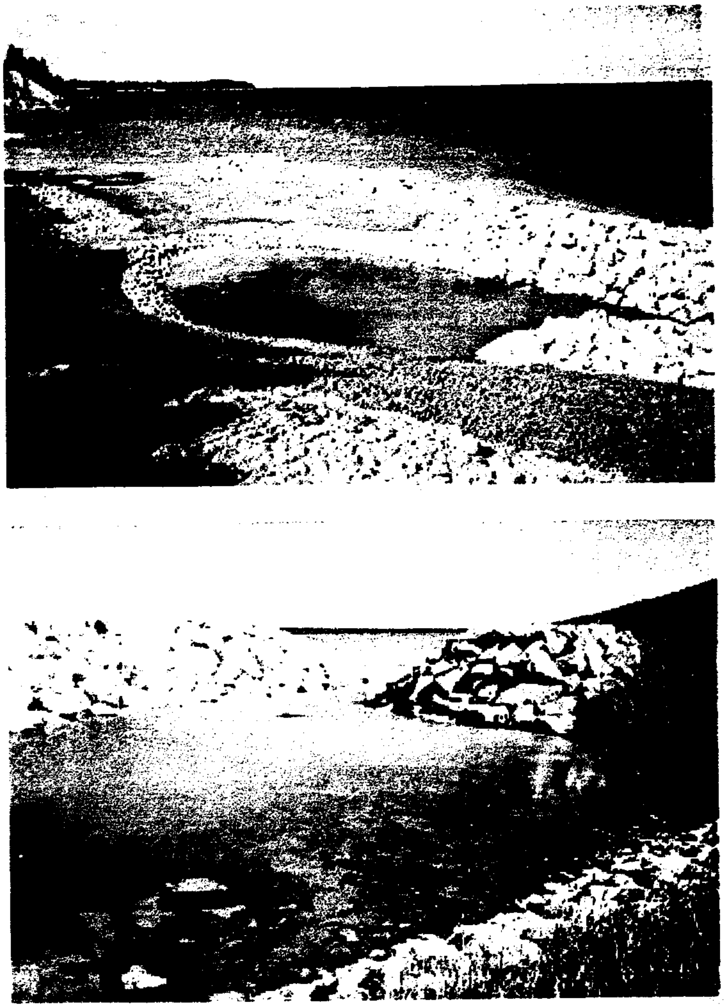

FIGURE 6 BENDER PARK LOW WAVE ENERGY CONSTRUCTED LAGOON

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HYDRAULIC AND ECOLOGICAL CONSIDERATIONS FOR COASTAL REALIGNMENT

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ABSTRACT

A progressive increase in relative sea level on the east coast of England is being accompanied by the erosion of the saltmarshes that form an integral part of many of the existing sea defences in the area. As the marshes fronting the seawalls erode, the probability of overtopping and breaching of the defences increases and the cost of maintaining the present line of defence escalates to the point at which it can no longer be justified economically for all sections of the coastline. Consequently, alternative solutions for coastal protection are being investigated in some areas. One such technique: coastal realignment (also known as managed retreat), involves relocating the existing line of sea defence further inland and encouraging the development of intertidal habitat, especially saltmarsh, in the zone between the old and new defence lines. By absorbing wave and tidal energy and moderating the waves impacting the shore, such saltmarsh can provide an effective, sustainable, low maintenance form of sea defence as well as a habitat of considerable conservation value.

In order for coastal realignment to succeed in these aims in a reasonable timescale, numerous interacting hydraulic and ecological processes must be optimised by careful design. By reference to two case studies, this paper describes some of the key hydraulic and ecological issues that need to be addressed in the design, construction and post-construction monitoring of coastal realignment schemes. In the first study, undertaken for the Ministry of Agriculture Fisheries and Food, and English Nature at an experimental site in Essex UK, computational modelling was employed to determine the likely effect of different configurations of breaches to the existing seawall on the morphology of the existing saltmarsh creek system adjacent to the breach. The study developed a method for better estimating water movements in low water channels and on and off intertidal areas, thereby providing improved calculation of the likely erosion that would take place in the existing estuary as a result of the increase in tidal volume passing through the creek network. The second study, also in Essex, conducted for the National Rivers Authority, addressed optimal design considerations in order to rapidly establish a stable morphology both inside and outside the retreat site. Four principle requirements were identified:

- appropriate elevation for saltmarsh colonisation
- the need for the site to drain completely during each tidal cycle to promote consolidation of deposited sediments
- construction of a skeletal creek network prior to breaching to facilitate distribution of incoming suspended sediments across the entire area
- Strategically placed sacrificial wave breaks to minimise wave disturbance within the developing retreat area and thereby promote retention of the settled sediment

This scheme was implemented in April 1995 when the seawall was breached at two locations. Monitoring is on-going and will continue until 1999. Initial results of the monitoring studies are presented.

KEY-WORDS: Coastal realignment/managed retreat/hydrodynamic modelling/saltmarsh/habitat creation/monitoring

INTRODUCTION

Saltmarshes are depositional coastal habitats found in areas of low wave energy, high in the intertidal zone. They are vegetated by halophytes (salt-tolerant plants) whose lower limit corresponds approximately with the level of mean high water of neap tides. The upper limit of saltmarsh occurs just above the level of mean high water spring tides. Typically, intertidal sand or mudflats lies seaward of the saltmarsh, whilst the landward margin is a transition zone of brackish and freshwater vegetation communities or a coastal defence structure such as an embankment or seawall. The network of creeks through which tidal flow is channelled are an integral part of the system. They are the conduits through which materials such as sediments and nutrients are exchanged between the saltmarsh and the coastal water. Moreover, they are the access route by which fish can enter the saltmarsh to gain shelter and food.

Saltmarshes play a significant role in sea defence. Saltmarsh vegetation enhances accretion rates, thus the elevation of saltmarshes is higher than the mudflat immediately adjacent to the vegetated zone. The width of saltmarsh acts as a berm so that wave height reduces as the wave propagates across the saltmarsh due to shallowing of the water. Moreover, the vegetation on the saltmarsh increases surface roughness and hence frictional resistance which dissipates wave energy. Thus saltmarsh situated in front of seawalls moderates the incident wave climate and thereby reduces the probability of flooding the hinterland by waves overtopping the seawall (Brampton, 1992), and/or wave attack damage to the seawall. As Table 1 shows, the capacity of saltmarsh to dissipate wave energy provides major cost savings for sea defence schemes. Indeed, in the county of Essex alone, 300 km of the 440 km of seawalls maintained by the National Rivers Authority (NRA) rely on saltmarsh vegetation as a 'first line' of sea defence. If the saltmarsh were lost the additional cost for sea defences would be many hundreds of millions of pounds.

Table 1: The effectiveness and economic value of saltmarsh as a natural form of sea defence (after NRA, 1994)

Width of saltmarsh fronting seawall (m)	Height of seawall required for adequate sea defence (m)	Cost of seawall per metre (£)
0	12	5000
6	6	1500
30	5	800
60	4	500
80	3	400

Unfortunately, however, in many areas of the UK saltmarsh is eroding and deteriorating a result of both natural and anthropogenic impacts. A high proportion of saltmarshes in Essex have suffered net erosion in the last 20 years with as much as 44% of the total marsh area being lost in the Stour Estuary (Burd, 1992). Indeed, Essex is particularly susceptible to saltmarsh erosion as a result of a progressive increase in relative sea level along its coastline. This is giving rise to the phenomenon of 'coastal squeeze' - the loss of intertidal area in areas experiencing sea level rise which are backed by sea walls. In areas where the rate of accretion of the intertidal zone is less than the rate of relative sea level rise, the increase in water depth exacerbates shear stress by allowing larger waves. This results in lateral, and more rarely, vertical erosion of the saltmarsh. In the absence of seawalls, this loss could, at least partially, be offset by the extension of the landward edge of the saltmarsh. However, embankments and seawalls prevent this from happening, resulting in a narrowing of the saltmarsh zone. The sea defences are thus rendered more susceptible to wave attack and the resulting damage necessitates substantial capital expenditure for seawall maintenance. In addition, the defences may need to be upgraded to maintain the past level of protection.

Coastal realignment, also known as managed retreat or setback, is being increasingly considered as a coastal management strategy in rural areas that are suffering from saltmarsh erosion where the cost of maintaining the current line of defence exceeds the value of the hinterland it protects. Essentially, the technique involves encouraging the development of intertidal habitat, especially saltmarsh landward of the existing sea defence. By absorbing wave and tidal energy, such saltmarsh can provide an effective, sustainable, low maintenance form of sea defence as well as a habitat of considerable conservation value.

The principle of coastal realignment is thus, relatively straight forward, however the practicality of such schemes is by no means universally accepted in the UK. Consequently, the pressure for the initial projects to succeed over a reasonable timescale is substantial. By reference to two case studies, this paper describes some of the key hydraulic and ecological issues that need to be addressed in the design, construction and post construction monitoring of coastal realignment schemes. It aims to identify those features essential to the rapid establishment of a stable morphology both inside and outside the realignment site and to provide a series of practical recommendations for the creation of saltmarsh habitat within such schemes.

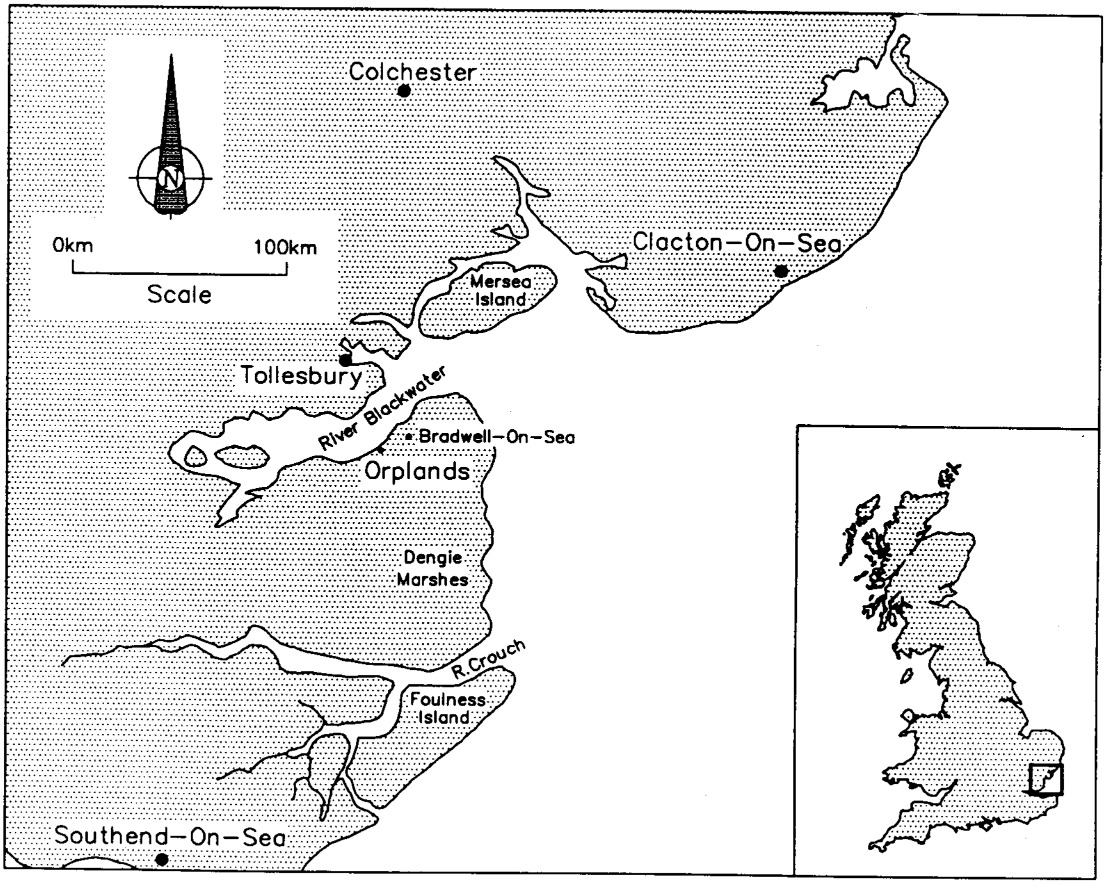

Figure 1: Location of Tollesbury and Orplands coastal realignment schemes

CASE STUDIES OF TWO COASTAL REALIGNMENT SCHEMES

Both of the realignment schemes chosen as representative case studies have been implemented in the county of Essex, England (Figure 1). The first, at Tollesbury, has been undertaken by the Ministry of Agriculture, Fisheries and Food (MAFF) and English Nature (EN). It covers an area of 21 ha. The second, is a 38 ha site that has been conducted by the National Rivers Authority (NRA) at Orplands. Each scheme has involved the removal of a section of the existing seawall to facilitate tidal inundation of the realignment area whilst retaining the protection afforded by the remaining seawall. At the Orplands site, two breaches were made. Both sites were breached in the spring/summer of 1995.

Hydraulic Considerations

It is essential that before proceeding with a coastal realignment scheme an assessment is made to ensure that it does not adversely impact adjacent areas of the coast. Opening up an area to tidal inundation increases the tidal prism of the estuary, which means that a greater volume of water has to pass through the part of the estuary downstream of the realignment site. The increased current velocities arising as the result of such a phenomenon could produce increased erosion downstream of the realignment site (the further upstream the realignment site is situated in the estuary, the greater the risk) , perhaps leading to the undermining of adjacent sea defences, thereby negating the intended benefits of coastal realignment and indeed, perhaps significantly worsening the existing situation. Such considerations are particularly crucial when addressing the cumulative impact of a number of realignment schemes in a given estuary. In such a case, detailed prediction of the likely hydraulic impact is required. Mathematical modelling is by far the most powerful tool for undertaking an assessment in this regard, as rates of water flow after the seawall is breached can be predicted and this data can then be put into another model to predict whether the increased shear stress would cause erosion. Nevertheless, most present modelling systems have difficulty resolving narrow channels or bed forms and simulating flows on and off intertidal areas such as are found in the complicated creek networks of saltmarsh systems.

The appropriate equations for studying water movements in tidal areas are the 2D, depth-averaged, shallow-water equations based on the well-established principles of conservation of mass and momentum. In theory, finite difference of finite element approximations to these equations can be made as accurate as required by suitably refining the computing grids, but in practise, due to limited budgets or computing resources, a minimum grid size has to be used which does not resolve narrow channels or bed forms as well as one would wish. This means that, for example, a deep channel in shallow water is represented in the model as a flat area of medium depth. The consequence of this is twofold: the directional effects of the channel upon the flow are lost, and the friction increases unrealistically.

Various methods have been tried to overcome this shortcoming, using non-uniform grid systems such as patching areas of progressively finer, but locally uniform grids, into areas of coarser grid, and non-uniform finite element or finite difference grids. Non-uniform grids are ideal when detail is required in a limited local area but less satisfactory for whole estuary models where detail can be required over large areas. Out of all such methods, finite elements seem to offer the best long term solution. However, these methods are notoriously slow to solve on the computer and there are problems re-configuring the grid as intertidal areas are covered and uncovered during the tide. Miles and Weare (1973) presented a method for improving model friction if sub-grid information was available. In the late 1970's HR Wallingford developed a method of patching together areas of different but locally uniform grid, and provision was also made to allow distorted, part grids to improve the representation of the shape of (fixed) no-flow boundaries around the edge of the model.

In 1985 a new approach was conceived (HR Wallingford, 1985), which was essentially an extension of the method used for 1D, area averaged models which employ look-up tables for the width, area and hydraulic radius of each 1D model section. This model was upgraded through ETSU-funded research (HR Wallingford ,1989), where it was found to give much improved predictions of tidal propagation in the Mersey Estuary, UK, compared to physical model measurements.

Due to the complex geometry and bathymetry of Tollesbury Fleet Estuary it was clear at the outset that accurate simulation of the tidal processes and associated sedimentology would be strongly dependent on the resolution of the estuary, including the areas of salt marsh and creeks. The TIDEWAY sub-grid modelling method was applied to improve model flow simulations in these complex creek systems without incurring prohibitively long run-times. Additional information regarding the salt marsh morphology was supplied in the form of a schematised geometry, and by specifying the estimated typical ratio of marsh coverage to creek area the water storage in the saltings could be represented.

As a precursor to the computational modelling, measurements were made of the bathymetry, tidal elevations, current velocities, and mud properties at a number of points in the creek. These allowed the model to be set up and validated in order to give confidence in the model predictions. With this model improved flow movements were achieved relative to a standard model output, resulting in improved bed shear stress estimations. This in turn allowed predictions of the pattern of mud erosion and accretion to be modelled so that the impact of different breaching scenarios could be assessed and an optimum configuration for breaching could be identified.

Habitat Considerations

Surface Elevation In Relation To The Tidal Frame

Saltmarsh can be expected to develop on low energy muddy shores between approximately MHWS and MHWN thus it is readily apparent that surface elevation in relation to the tidal frame will be a critical factor governing the success or failure of any saltmarsh restoration scheme. Elevation will also influence the composition of the halophyte species that will colonise the site, the pioneer species such as *Salicornia* grows lower down the shore than the upper saltmarsh species such as *Sueda vera.* One method to predict the character of the vegetation on the proposed coastal realignment site is to compare the surface elevation of the site with that of a neighbouring natural saltmarsh.

Both of the experimental realignment sites described in this paper were originally saltmarsh habitats prior to annexation for agricultural or pastoral purposes. Intuitively, it might be expected that the former saltmarsh areas would provide the best sites for realignment schemes. Typically, however, such sites tend to be at a somewhat lower level than the adjacent saltmarsh due to irreversible changes in the structure of the clay due to drying out, percolation of freshwater and compaction of the sediments by agricultural activity, whilst the remaining saltmarsh tends to increase in elevation as a result of sedimentation and peat formation. Large parts of the Tollesbury and Orplands sites are, indeed, somewhat lower than the neighbouring saltmarsh in each area, but the surface elevations still lie predominantly between mean high water of spring tides and mean high water of neap tides. Thus, in terms of surface elevation both sites can be expected to develop saltmarsh vegetation, although some areas are more likely to revert to mudflat. If this were not so, a site could still be suitable for realignment provided that the surface was built up to an appropriate level. One appropriate source of material with which to do this is the material removed during the excavation of a creek system on the site prior to inundation, alternatively suitable dredged material (which has a similar particle size range to the sediment on site) could be pumped onto the site. Construction of a creek system also provides other benefits as discussed below.

Topography

It is desirable that the chosen coastal realignment site has a seaward slope to provide good drainage of surface waters. Significant basins and depressions will lead to surface ponding after tides that inundate the saltmarsh surface, this waterlogging is not conducive to plant establishment.

Creek Systems

In natural saltmarsh systems creeks are essential for the conveyance of tidal waters and distribution of sediment across the whole saltmarsh surface. In the absence of creeks sedimentation rates would reduce up the saltmarsh due to sediment settling out of the water column with increasing distance from the sediment source. In time, this would lead to water logging and poor drainage of the back portion of the coastal realignment site as the landward part would be lower than the front portion.

The species composition and vigour of halophytes is controlled by the position of the water table and the residence time of the porewater in the sediment, which is governed by the degree of flushing and drainage. Work by Mendelssohn in North Carolina, USA has established a definite link between soil drainage and the growth of *Spartina alterniflora* (Mendelssohn, 1980; Mendelssohn and McKee, 1988). Plant stress is due to the build up of phyto-toxins (such as hydrogen sulphide, ethanol, ferrous and manganese ions) which are produced directly and indirectly by anaerobic bacterial respiration. Research in America has shown that there is a direct relationship between the concentration of sulphide and the height of *Spartina alterniflora* shoots. This is because hydrogen sulphide is a respiratory inhibitor and interferes with nutrient uptake in plants (Howes et al., 1981). Creeks facilitate flushing and drainage of the sediment, thereby reducing pore water retention time and lowering the water table and hence allowing oxygen to diffuse more freely through the unsaturated zone. This phenomenon combats reducing conditions in the saltmarsh sediments and makes them more conducive to plant growth.

Figure 2 illustrates the effect that creek proximity has on the position of the water table and redox potential using data derived from a natural saltmarsh adjacent to one of the realignment sites. Redox potential is the electrical charge of the sediment which is governed by diagenetic processes. It is a useful measurement because the value indicates whether the sediment is oxidising or reducing. Redox potential and pH are good indicators as to the concentration of phyto-toxins. The lower the redox potential, the more reducing and the greater stress exerted on plants and infauna. The data in Figure 2 emphasises the importance of creeks in controlling sediment chemistry and why creeks should be cut into managed retreat sites to aid plant growth.

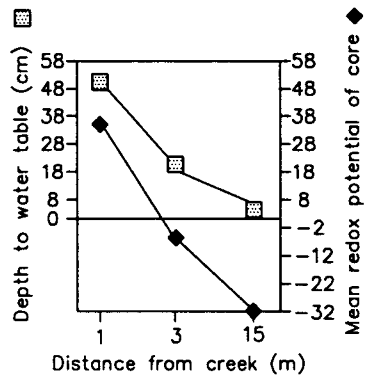

Figure 2: The relationship between creek proximity, depth to water table and redox potential

When the proposed realignment site is an area of previously enclosed saltmarsh the best approach is to re-open the original creek system. This is often readily discernable from aerial photographs even in areas subject to intensive cultivation over many years following enclosure. At the Orplands site, a creek network was cut, although no attempt was made to reestablish the old system.

Wave Climate

Saltmarshes can only be established in areas sheltered from direct wave action. Once established, however, a saltmarsh can, as already described, substantially moderate the wave climate. Inundation of both experimental sites via breaches and retention of the remaining sea wall will provide significant protection from wave action during the establishment of vegetation. Thus, the sea wall will perform an analogous function to an offshore bar or chenier at the seaward margin of a natural saltmarsh, i.e. dissipate wave energy before it reaches the saltmarsh.

It is also important to minimise wave action behind the seawall within the realignment site during the early stages of development before a good cover of vegetation has established. This is in order to optimise the settlement and consolidation of sediment brought into the site in suspension, by the tidal waters, and also reduce the risk of erosion. The sediment removed during the course of creek excavation, if placed in bunds strategically positioned around the site, can provide sacrificial wave breaks during the critical early phase of scheme development to moderate the internal wave climate.

Seed Source

Clearly, the most economically attractive method of achieving vegetation cover is natural colonisation. To achieve this there has to be saltmarsh close enough to provide seeds in sufficient quantity. Hence, coastal realignment sites adjacent to existing saltmarsh will establish more quickly and at less cost than if they are situated remote from any seed source.

Scheme Monitoring

As the concept of coastal realignment as a management strategy is relatively new, and the schemes at present are pilot studies, there is a considerable interest in monitoring the development of these sites to gauge their performance and to use this data to feed into future scheme designs to improve their effectiveness.

Vertical accretion/erosion rates and colonisation by halophytic vegetation are the key parameters to be investigated during HR Wallingford's five year monitoring programme for the Orplands realignment site which was begun following a pre-inundation baseline survey in 1995. Other variables being monitored include the physical and chemical nature of the sediment, creek development, utilisation of the site by animals, evidence of wave attack on the counterwall, and changes to the lateral extent of saltmarsh adjacent to the realignment site. In order to determine whether the recreated saltmarsh is functioning in the same way as a natural one, a natural saltmarsh close to the coastal realignment site is also being monitored to provide a control.

Vertical accretion/erosion

The sea defence value of the managed retreat area will decline if the surface level lowers after opening it up to tidal influence. Change in surface elevation is therefore a key parameter of the monitoring programme. Relative sea level is rising at a rate of between 3-6 mm/yr around the coastline of Essex. For the newly created intertidal area to function as a long term sea defence, accretion needs to keep pace with sea level rise. Failure to do so will result in continually declining capacity to prevent flooding.

It is important to recognise the difference between accretion and sedimentation rates. The former refers to an increase in surface elevation, whilst the latter is the depth of sediment which settles on the surface. These two differ in that the surface elevation of the saltmarsh can fall even if sedimentation is occurring. Elevation decrease could be due to:

- decomposition of organic matter, i.e. soil roots of terrestrial plants that were killed by salt-water flooding.
- land subsidence
- compaction of sediments after settling.

Both change in surface level and sedimentation/erosion are recorded in this monitoring programme. Three methods are employed: levelling of transect profiles, Feno marker pairs, and Kestner cores.

- Transect Profiles

To record changes in elevation on the managed retreat sites and the control saltmarsh, and changes in the morphology of the excavated creeks and natural creeks, permanent transects were set up; eleven on the coastal realignment site and three on the control saltmarsh. Profiles of the transects were determined using an electronic level and recording the surface elevation every 4m along the transect. Care is taken so that the repeat measurements are made at the same positions. Comparison of the profiles from subsequent years will enable topographic changes to be identified.

- Feno markers

Nineteen pairs of Feno survey markers were installed to enable precise, repeatable measurements of topography on a small scale. Figure 4 illustrates the appearance of these markers. The variation in sedimentation is complex. Rates differ over short distances due to: (a) changes in elevation which control the frequency of inundation; (b) proximity to creeks, sedimentation reduces with distance from the creek bank due to the settling of suspended sediment along the path of water movement during 'above-marsh' episodes (c) hydrodynamics, which affect settling and re-suspension Therefore it is necessary to collect accretion/erosion data at points that cover the range of elevations in the managed retreat and also a set of points of a similar elevation but at various distances from a creek. The locations of the Feno marker pairs were chosen to cover these ranges. These were arranged to cover the range of elevations on the site, and two sets of four pairs were installed in a transect perpendicular to a creek bank to investigate the effect that distance from creek had on accretion/erosion.

An aluminium frame, consisting of two sides supports and a horizontal bar was constructed. The vertical supports are pushed into the hollow centre of the Feno markers until the disc on the supports rests snugly on top of the marker. The distance between the lower side of the horizontal pole and the ground was measured at 10cm intervals along the pole (starting from the marker which is furthest from the sea) using a metre rule. A spirit level was attached to the metre rule to ensure that it was held vertically and to improve reading accuracy a sliding flange was attached to the metre rule. Three positions along the pole were selected at random and the measurements repeated to gauge the precision of this method; the variability of the measurements proved to be not more than 3mm.

Baseline measurements from the Feno pairs were taken on 8/4/1994. Repeat measurements were made post inundation in August, and will be conducted annually.

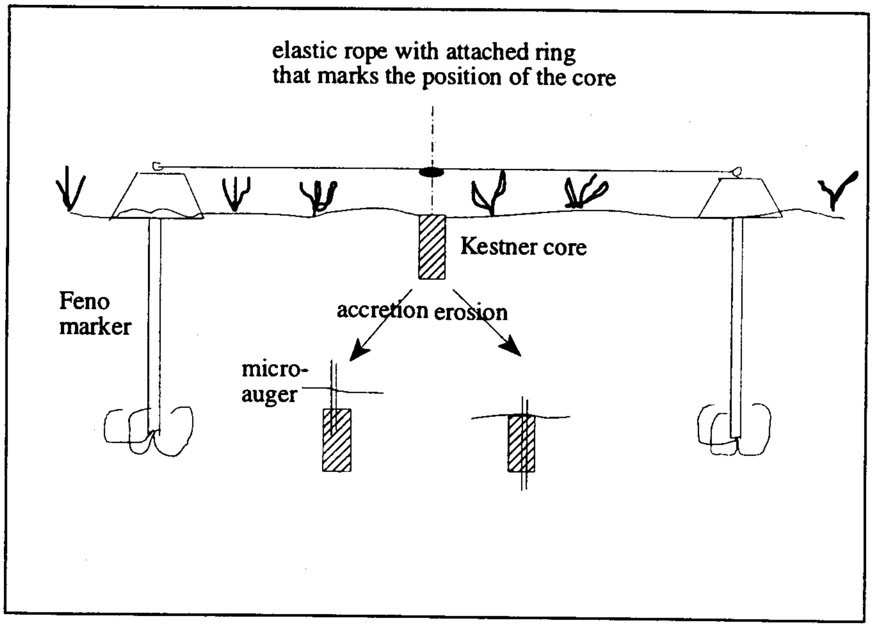

Figure 3: Illustration of how the Kestner core is relocated and measurements of sedimentation/erosion taken

- Kestner cores

Kestner cores were developed by Fritz Kestner at HR Wallingford for research into sedimentation/erosion of mudflats in the Wash, East Anglia. The technique measures the burial or erosion of a cylinder of white silica flour. Silica flour is chosen as it has similar properties and grain size to silt/clay and therefore should respond in the same way as the surrounding sediment to erosion and sedimentation forces. Figure 3 illustrates how Kestner cores can be used to measure accretion and erosion. A core of sediment, approximately 10cm deep and 4cm across was removed from each of the nineteen measurement sites. The base of the hole was flattened and the depth noted. Silica flour was packed into the hole and water added to wet the flour. More silica was added until the silica was level with the surrounding sediment. One Kestner core was placed in the centre of each pair of Feno markers.

Vegetation

Changes in vegetation cover will be monitored by analysing the photographs taken during the post-inundation surveys. The following three sets of photographs will be used.

- close up of ground around sampling station stake (photograph is facing seawards)
- view along transect from the original sea defence embankment
- aerial photographs.

Change in the plant community after the commencement of tidal inundation is being monitored by: a) producing an overall list of plant species present and b) evaluating the relative abundance of each species annually until the end of the monitoring period. Three survey methods are being employed:

1. Fixed quadrats on line surveys
2. Random quadrat sampling
3. Collation of total species list from a walk-over survey

Initial Monitoring Results

The first of the annual post-inundation surveys were conducted four months after the two breaches were made in the seawall, in August 1995. The results from the Kestner cores and Feno marker pair measurements show that the site has accreted. The rates of accretion/sedimentation calculated for this period were extrapolated to give annual rates, The annual mean rate from the Feno marker measurements was 14mm/year, which compares favourably with the rate of 10.5 from the Kestner cores. The topographic ground survey of the transect lines showed that the silt/clay bunds constructed from the excavated material from the cutting of the creeks and the breach in the seawall, had considerably diminished in size. Thus, this extra material may have enhanced the initial sedimentation rates. Annual measurements over the next five years will determine whether this is the case.

As expected, the terrestrial vegetation on the site died due to the onset of seawater inundation. This caused initial intense anaerobic conditions of the surface muds due to the decomposition of the plant matter in waterlogged conditions. Even so, within four months algal mats (a community of diatoms and cyanobacteria) had established and marine filamentous algae was growing prolifically in the pools. Halophyte colonisation on the part of the site that had been long grassland was probably hampered by the initial strongly reducing conditions, however colonisation had begun on the half of the site that had been a bare arable field; species present included: *Salicornia* and *Spergularia marina.*

CONCLUSIONS

- Coastal realignment has the potential to provide an effective, sustainable, low maintenance natural form of sea defence as well as a habitat (saltmarsh) of considerable conservation value.
- If successful, coastal realignment schemes could save many hundreds of millions of pounds expenditure on sea defence in the U.K.
- Care must be taken to avoid/minimise adverse impacts of coastal realignment on hydrodynamic processes in the estuary. Detailed prediction of the effect of different realignment scenarios should be undertaken using appropriate mathematical modelling systems with adequate grid resolution to simulate the complicated processes of water movement and sediment transport in the intricate creek systems of saltmarshes.
- The site should preferably be reclaimed saltmarsh
- The surface elevation of the site should be appropriate for the development of saltmarsh, with a seaward sloping gradient to enhance surface drainage, habitat diversity and hence conservation value.
- A suitable creek system should be provided from the outset to distribute sediment across the entire site surface and promote efficient drainage and flushing of the sediment to encourage healthy plant growth.
- The site should drain completely during the tidal cycle to promote consolidation of the settled sediments.
- Strategically placed sacrificial wave breaks within the realignment site are desirable to minimise initial wave disturbance and thereby facilitate sedimentation
- The long term success of coastal realignment schemes in providing a means of sea defence and new saltmarsh habitat will depend on surface levels accreting at a rate at least that of relative sea level rise, and the development of a healthy halophytic plant community. These are the key parameters for any monitoring programme.

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THE EFFECT OF SEAWALLS ON COASTAL MORPHOLOGY

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ABSTRACT

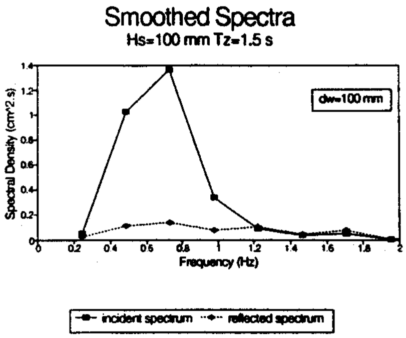
The hydrodynamics within the nearshore region and the subsequent evolution of beaches under wave attack are important elements governing the stability of coastal zone. Although there are some advanced models which predict the beach profile for natural beaches, the behaviour of the beaches in front of reflective structures (i.e. seawalls), still suffers from the lack of appropriate theoretical models and enough measured data. Seawalls are commonly constructed to prevent landward erosion of the shoreline and maintain the configuration of the area behind them against the wave action. It is clear that the effects of seawalls on beach profile evolution have not been studied sufficiently. The present study addresses this issue and considers the key aspects of the problem regarding the morphological changes in coastal area. The results of this investigation can be used for careful planning and implementation of sea defense works which can significantly reduce their impact on the environment.

KEY-WORDS: Seawall/Hydrodynamics/Beach Profile/ Sediment Transport/Reflective Structures/Wave/ Coastal Morphology/ Environmental Impacts/surf zone/sand.

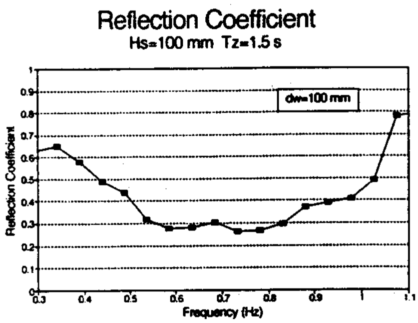
INTRODUCTION

An extensive literature review made by N.C. Kraus (1988), shows that only a small number of theoretical studies was found that treated the beach profile change in front of a seawall. Results have been derived for highly simplified conditions.

Numerous physical model laboratory experiments as well as field investigations have been performed to investigate the interaction between seawalls and the beach. It has been found that a beach with a seawall installed in the surf zone can respond to accretionary wave conditions as well as erosion. A number of researchers conducted experiments to consider the effects of seawalls on beach profile. It was found that the final profile configurations were remarkably similar with and without the seawall, suggesting that " The major transport process is not significantly influenced by the presence of the seawall " (Kraus, 1988). Weggel (1988) suggested that since the effect of seawalls on beaches and on coastal processes has not been well documented, there is a need for additional research into the effects of seawalls on coastal processes so that quantitative data on seawall performance will be available.



(a)



(b)

Figure 1:(a) Incident and reflected spectra (b) Variation of reflection coefficient

HYDRODYNAMICS

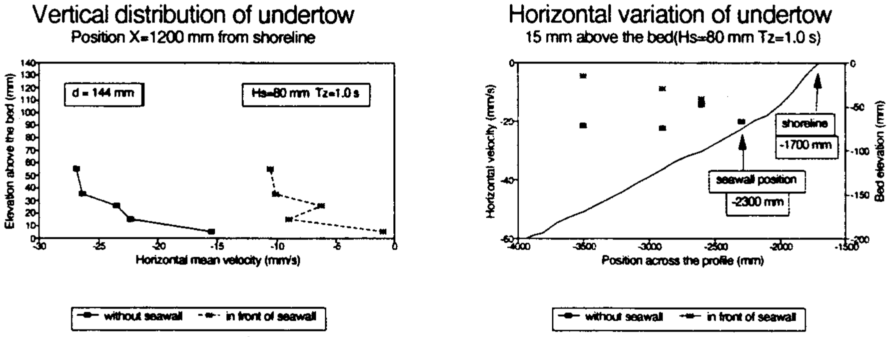
According to linear wave theory, for regular standing waves, water particles oscillate in a horizontal plane beneath the node and in vertical plane beneath the antinode. The extreme values of horizontal and vertical velocities therefore occur under the nodes and antinodes of the water surface profile. With assumption of linear theory, it is therefore possible to estimate the water particle velocities in front of a partially reflective structure for a spectrum of nonbreaking waves (Huges, 1992). In a real situation, however, the nonlinearity and random nature of incident waves as well as the breaking phenomena in front of structures contribute to create a complicated velocity field.

In order to determine the velocity field inside the surf zone, laboratory experiments were performed in a large two-dimensional wave tank at Imperial College hydraulics laboratory. The overal dimensions of the tank were 2.80 m wide (which was divided into three sections), 1.5 m deep and 60 m long. The width of channel used in experiments was 0.87 m. The water depth was 0.9 m for all experiments.

A beach profile according to an equilibrium profile equation (Vellinga, 1984) was built at the end of the tank. Irregular waves were generated at one end of the tank by a wave generator controlled by an electro-hydraulic system and the water particle velocities were measured in the surf zone using a one-component fiber-optic Laser-Doppler Anemometer (LDA). Four horizontal locations were chosen in the surf zone. At these locations measurements were made at several points between the bottom and still water level. The locations for all measuring points are given in Table 1.

Table 1: Specifications of measuring points

Positon	Distance from shoreline (m)	Water depth (m)	Elevation above the bed (mm)
1	0.6	0.097	5,15,25
2	0.9	0.1192	5,15,25,35,50
3	1.2	0.1438	5,15,25,35,55
4	1.8	0.187	5,15,25,45,65



(c) (d)

Figure 2: Comparison of vertical (c) and horizontal (d) distribution of undertow in the surf zone between an open beach and a beach fronting the seawall

The measuring points were in the middle of the flume (0.43 m from inside face of the sidewall). Additionally, a capacitance type wave gauge measured the water surface elevation (synoptic with the velocity measurements) at each location. Also, a deep water wave gauge was mounted further offshore to measure the deep water incident wave spectrum. Data was acquired and analyzed using a personal computer with a sampling rate of 25 Hz for each channel. Using a typical Jonswap spectrum, three different wave spectra were generated whose properties are summarised in Table 2.

Table 2: Descriptions of spectra

Deep water significant wave height (m)	Zero-crossing period (s)	Deep water significant wave length (m)	Deep water wave steepness
0.080	1.0	1.56	0.051
0.100	1.5	3.51	0.028
0.120	1.5	3.51	0.035

In order to consider the effect of reflective structures on beach hydrodynamics, the experiment was repeated in front a partially reflective seawall located in the surf zone. The main objective of this experiment was a quantitative comparison of near-bed velocities in the two cases (i.e. with and without the reflective structure). For this purpose, a permeable seawall was built at the end of the beach. The exact distance of seawall from the shoreline was 0.6 m resulting in 0.100 m water depth in front of the structure. A sampling rate of 25 Hz with a record length of 6 minutes, similar to the first experiment, was selected to provide the suitable conditions in order to compare the velocity field in two different experiments. In order to estimate the incident and reflected wave spectra and reflection coefficient from the structure, a three wave gauge method presented by Gaillard *et al* (1980) was used. For this purpose, three capacitance wave gauges were mounted at certain locations seaward of the beach in deep water.

Figure 1 shows the incident and reflected wave spectra as well as reflection coefficient for one of the spectrum used in the experiments. As it can be seen, the overall reflection coefficient around the peak frequency for entire experiment is approximately 30 %. Figure 2 contrasts the vertical and horizontal distribution of undertow on an open beach with that found in front of the reflective structure. As indicated in this

Figure, it seems that the presence of the reflective structure and the addition of reflected waves has caused significant changes in the mean flow particularly further offshore of the surf zone. Comparison of probability density functions for the near-bed horizontal velocities with the Gaussian distribution also shows the skewness due to reflected waves in front of the seawall. It can be therefore concluded that since the reflective structures modify the velocity field, they can contribute in terms of sediment transport resulting in beach profile evolution during storm conditions.

PROFILE EVOLUTION

The velocity measurements have been complemented by beach profile measurements performed with two different sizes of sand (fine sand,$D_{50} = 0.5mm$ and coarse sand,$D_{50} = 1.5mm$). The experiments were then repeated in front of the partially reflective structure located in the surf zone. The sand sizes were chosen so that the threshold of motion was exceeded for a significant period of time for both sediments. The results of the open beach experiments show the coarse sediment beach building a berm, while the finer sand beach erodes to form a longshore bar; whereas the results of the beach experiments in front of the seawall show the tendency of the beach to form a berm profile in front of the structure (Figure 3). This Figure also contrasts the change in bed elevation from the original profiles for both fine and coarse sand on an open beach with that measured in front of the seawall.

In the next stage of experiment, the change in profile was measured in front of a seawall with different water depths to assess the influence of the location of seawalls on beach profile evolution. For this purpose four water depths in front of the seawall were selected and the changes in profile were measured for three different wave conditions. Figure 4 shows the effect of seawall position on sediment transport and beach profile evolution for different wave conditions. It is quite interesting that the results show less sediment transport as the water depth in front of the seawall increases (in both cases of fine and coarse sand). Also, when the water depth in front of the seawall is too low, the beach behaviour is much the same as natural beach in which a longshore bar is formed. Obviously, as expected, in a particular water depth increasing the wave height causes the more sediment transport in front of the seawall.

As a conclusion, it seems that the extent to which a seawall affects the processes on the fronting beaches largely depends on its location relative to the active shoreface. A seawall located well landward of the active shoreface will behave in much the same way as a natural beach whereas seawalls located in the active shoreface will modify the near-shore beach profile because of the effects of reflected waves.

SUMMARY

Study of hydrodynamics in front of reflective structures shows that seawalls can modify the velocity field if they are located around the active zone (i.e. breaking zone). Therefore, it can be expected that seawalls can contribute in cross shore sediment transport resulting in the beach profile change during storm conditions. The results of profile evolution experiments are also in a good agreement with this fact and clearly show the different patterns in profile change with and without the structure. As a result, since the presence of sea defense structures will change the configuration of coastal area, the design of such systems should be developed in parallel with an environmental assessment and should take into account opportunities for environmental enhancement and appropriate measures for minimizing environmental impacts. The works described here is continuing with the development of a beach evolution model, using the measured probability distribution of near-bed horizontal velocities as input.

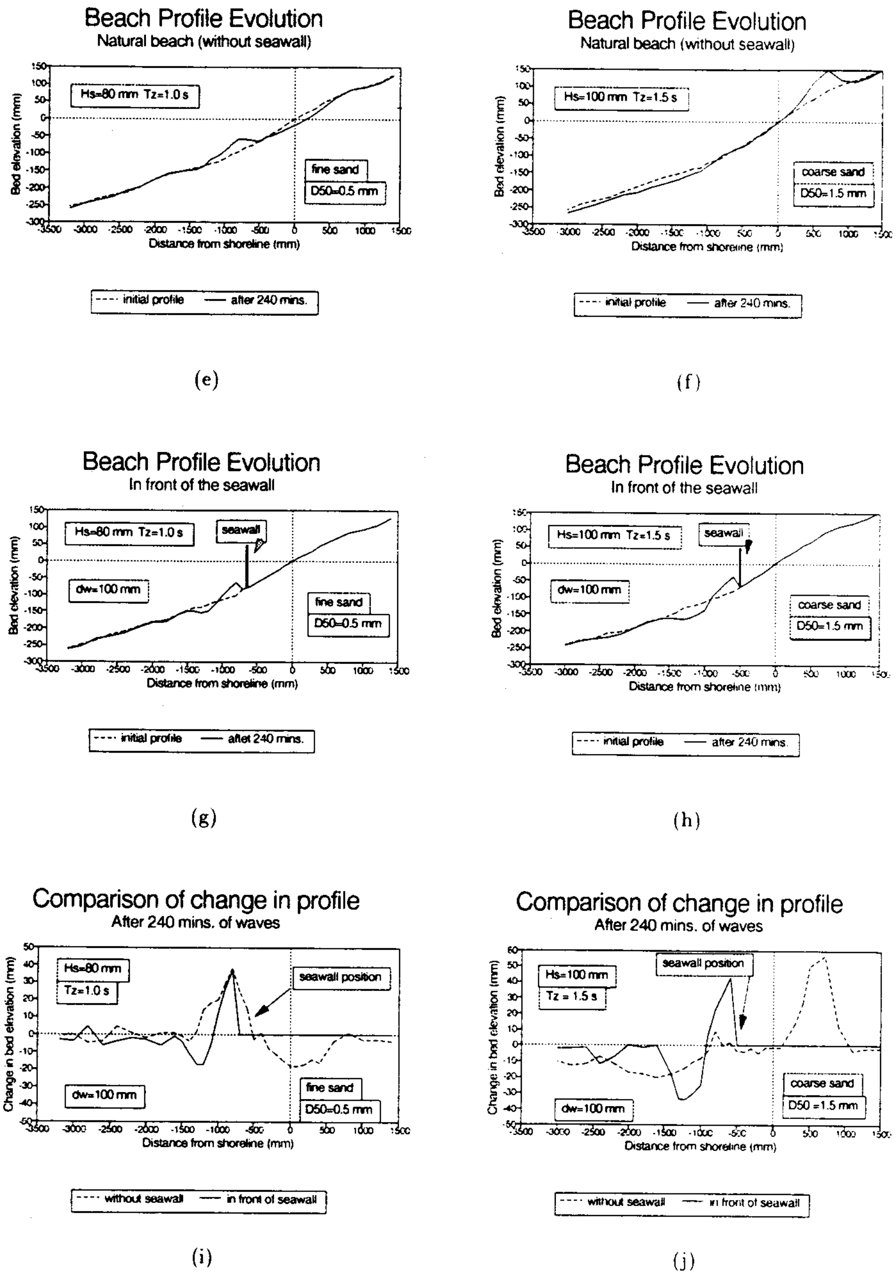
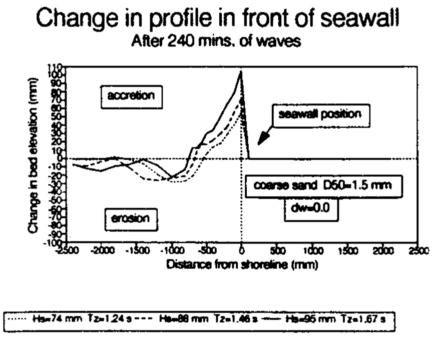


Figure 3: Comparison of beach profile evolution after 4.0 hrs. waves between an open beach and a beach fronting the seawall



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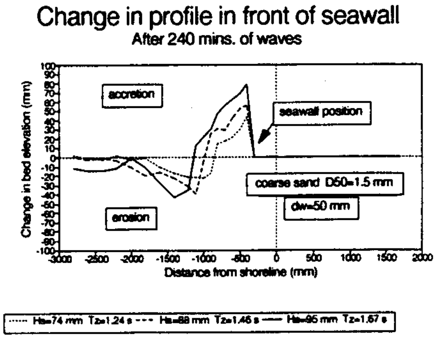

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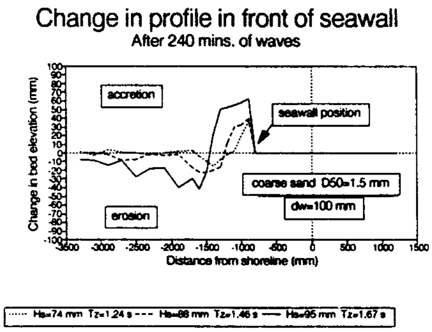

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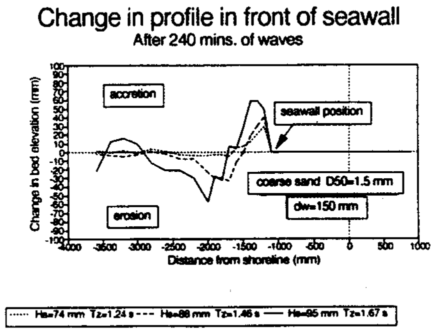

(n)

Figure 4: Comparison of changes in bed elevation for different water depths in front of the seawall (coarse sand)

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Minimum flow and adapted hydrological regimes

Débits réservés et hydrologie adaptée

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

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ABSTRACT

The Upper Salmon Hydroelectric Development in central Newfoundland, Canada, constructed in the early 1980's, resulted in the regulation of the West Salmon River, a major spawning and juvenile rearing river for landlocked Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*). A regulatory agency (Canadian Department of Fisheries and Oceans) and the regional utility (Newfoundland and Labrador Hydro) negotiated a controlled flow release strategy for the river which was developed using Tennant's Montana Method. A two-level strategy, employing seasonal modification, prescribed a release of 40% of the mean annual flow (MAF) (2.6 $m^3 s^{-1}$) between June 1 and November 30 and 20% of the MAF (1.3 $m^3 s^{-1}$) from December 1 to May 31.

Several studies have been conducted to address aspects of the regulation of this river including (i) a post-impoundment assessment of the actual and anticipated geomorphic and sedimentary characteristics, (ii) an effects monitoring study of juvenile fish populations under regulation, and (iii) a retrospective IFIM (Instream Flow Incremental Methodology) assessment. Studies provide evidence of the initial stages of river aggradation, including accumulation of fines from construction activities and natural processes within the watershed. Biological monitoring suggests no apparent effects of this sediment deposition on spawning and egg incubation as densities of Atlantic salmon fry are higher than prior to development. Conversely, densities of older age classes (parr, 1+ and greater) have declined under regulation and this is suggested to be related to poor over wintering conditions under the lower winter flow regimen. The IFIM studies, including development of microhabitat preference criteria, support these observations indicating that the existing flow regimen provides habitat conditions that would benefit salmon fry over older age classes. Flows under which optimum habitat conditions occur (based on estimates of weighted usable area) for salmon fry and parr were 0.75 and 3.5 $m^3 s^{-1}$, respectively. This retrospective assessment suggests that future proposals for flow regulation in Newfoundland should consider the need for a flushing flow component and attempt to include consideration of conditions necessary for over-winter survival of resident fish. Habitat-hydraulic models (e.g., IFIM) are preferable to standard setting approaches (e.g. Tennant's) where detailed analysis of habitat trade offs as related to flow regulation are required.

KEY-WORDS: Regulation/ Impoundment/ Fish populations/ Habitat-hydraulic modelling/ Atlantic salmon/ Geomorphology/Sedimentary characteristics/ Newfoundland, Canada

INTRODUCTION

Flow modification of rivers is one of the most widespread anthropogenic effects of hydroelectric power development and usually results in some form of modification of the spatial and temporal characteristics of the natural flow regimen (Ward and Stanford, 1979). Aquatic biological communities are related to habitat composition and stability which in turn is highly influenced by flow variability (Bain et al., 1988). River regulation can lead to decreased flood peaks, decreased sediment supply from upstream reaches, and a reduction or elimination of the ability of the water course to move bed materials (Kellerhals, 1982). These changes can result in reduced channel capacity, gradual aggradation of silt and organic debris, encroachment of vegetation both within the channel and in the riparian zone; all of which can have detrimental effects on habitat quality and fish production. Alternatively, a stable flow regime can benefit biota by increasing the potential productive capacity of the river (permits full utilization of the wetted area), increasing bed (substrate) stability and primary and secondary production, increasing (in some instances) winter flows which reduces damage from freezing, dampens the effects of floods and droughts, and lessening energy utilization (e.g., holding position under high flow periods) (Ward, 1976; Petts, 1980).

Water release below hydroelectric developments is a widespread means of mitigating the potential harmful effects of regulation on downstream aquatic resources (Rosenberg et al., 1987). There are wide variety of approaches to determine the appropriate flow release ranging from fixed flow standard setting methods (e.g., Tennant's Montana Method), utilizing analysis of historical flow data, to more complex incremental habitat based approaches (e.g., the Instream Flow Incremental Methodology or IFIM) (Bietz et al., 1985). Post-project evaluation of the effectiveness of controlled flow release from hydroelectric developments in protection of fish habitats and populations however are uncommon (Sale et al., 1991), particularly in Atlantic Canada. Flow release can be a major cost associated with mitigation of project impacts and surprisingly there have been few quantitative assessments on the benefits of measures taken.

The Upper Salmon Hydroelectric Development in central Newfoundland, Canada, was constructed in the early 1980's and was to de-water high quality spawning and rearing fluvial habitat for landlocked Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) in the West Salmon River. A controlled flow release strategy for the river, developed using Tennant's Montana Method (Beak, 1980b), was negotiated between a federal regulatory agency (Fisheries and Oceans) and the regional utility (Newfoundland and Labrador Hydro) and subsequently prescribed a two-level approach, employing seasonal modification, to protect fish habitat. A number of post-development studies have been undertaken to document the physical and biological effects of flow regulation. These studies have included (i) geomorphic and sedimentary characterization of the West Salmon River under regulation, (ii) an effects monitoring study to determine juvenile salmonid populations downstream of the project and (iii) an IFIM (Instream Flow Incremental Methodology) assessment of the West Salmon River. This paper critiques the existing regulated flow regimen (Tennant) based on the results of these various studies. Results are discussed in relation to the adequacy of the current flow regimen (in terms of magnitude and seasonal modification), the expected change in future habitat conditions under the existing arrangement, and regional considerations as to the effectiveness of controlled flow release as a mitigation measure for hydroelectric development in Newfoundland.

STUDY AREA AND FLOW REGULATION

The Upper Salmon Hydroelectric Development, located in central Newfoundland, was conceived to utilize existing reservoir and diversion systems of the previously constructed Bay D'Espoir Development. A dam constructed on the West Salmon River created an impoundment resulting in regulation of flows in the West Salmon River between Cold Spring Pond and Godaleich Pond (Figure 1). Pre-development studies estimated there were 2550 units (1 unit= 100 m^2) of high quality juvenile rearing and spawning habitat immediately below the dam site which would be completely de-watered, in the absence of flow compensation (Beak, 1980a). Studies subsequently confirmed extensive migrations from throughout the watershed to utilize these habitats consequently it was apparent the West Salmon River was critical to fish production and recruitment in the entire Salmon River drainage (Beak, 1981). Dominant fish species in the development area included landlocked Atlantic salmon or ouananiche (*Salmo salar*) and brook trout (*Salvelinus fontinalis*), while arctic char (*Salvelinus alpinus*) and threespine stickleback (*Gasterosteus aculeatus*) were also present (Beak 1980a).

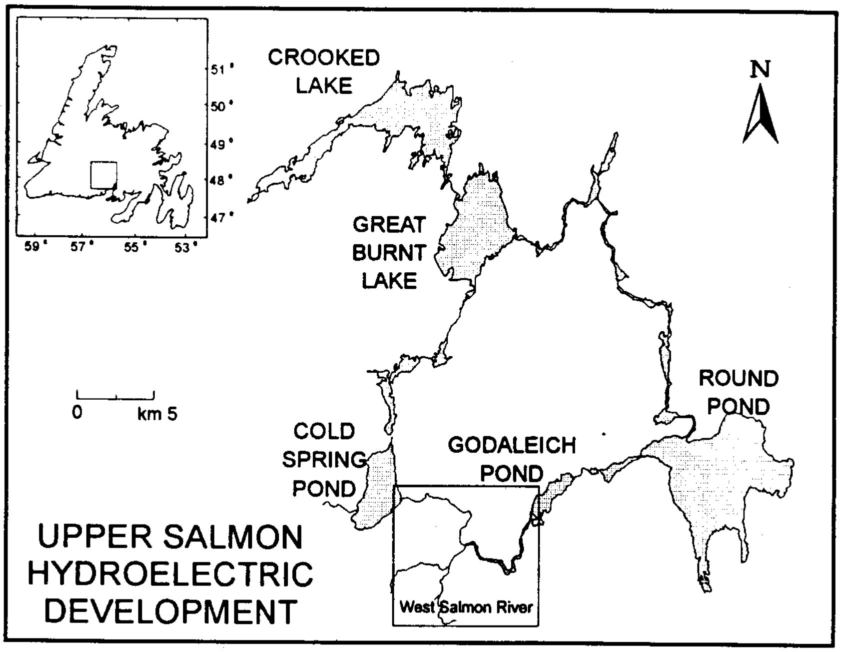

Figure 1. Map of the Upper Salmon Hydroelectric Development on the island of Newfoundland, Canada, showing the regulated section of the West Salmon River.

The West Salmon River is typical of boulder strewn rivers of southwestern Newfoundland and is characterized as wide and shallow with rapid flowing, steep bouldery sections alternating with sections of slower flow, greater width, and finer substrates (Rogerson, 1986). The pre-development hydrological regime of the West Salmon River was characterized by peak flow periods in the spring (associated with snow melt) and late fall (associated with a period of high rainfall) while the low flow period was in the summer months (July through to early October) (Figure 2). Winter flows from January through the end of March fluctuated around the mean annual flow (6.4 $m^3 s^{-1}$). A riparian flow release strategy, based on Tennant's Montana Method (Tennant, 1976) and modified by the professional judgement of the developer and regulators, was developed to preserve habitats and fish populations in the West Salmon River. This adopted 2-level regimen released 40% of the pre-development mean annual flow (2.6 $m^3 s^{-1}$) between June 1 and November 30 and 20% of mean annual flow (1.3 $m^3 s^{-1}$) from December 1 to May 31 (Beak, 1980b) (Figure 2).

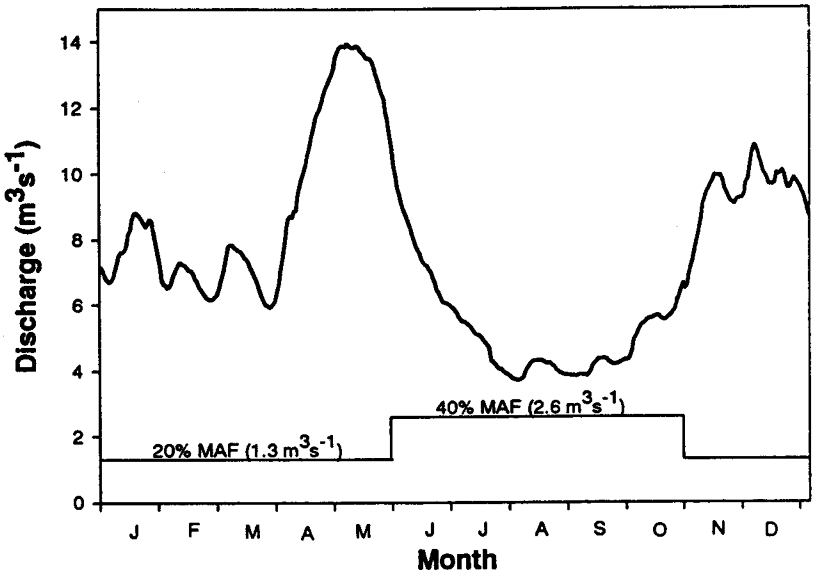

Figure 2. The pre-development hydrograph of the West Salmon River developed from the hydrographic records for the Salmon River at Long Pond (1945 to 1965), and adjusted for relative drainage basin size. The controlled flow release regimen developed from Tennant's Montana method is also included.

MATERIALS AND METHODS

Geomorphic And Sedimentary Characterization

A study of the post-regulation geomorphic and sedimentary environment of the West Salmon River was undertaken in 1985, 3 years after dam construction (Rogerson, 1986). This study (i) characterized the geomorphic and sedimentary characteristics of the channel environment, (ii) identified dynamic elements of the channel with a focus on spawning gravel habitats, and (iii) described the existing and expected evolution of sedimentary and geomorphic environments under flow regulation. Eight (8) cross sections described the depth and velocity distributions under the two-tiered regulated flow regimen and this data was used to discern the ability to move (scour) substrate materials. Substrate mapping was completed in seven (7) reaches to describe the distribution of substrates in relation to flow, with attention paid to areas with deposition of fines, including discussion as to origin of the material. Sediment samples were collected from the river bed using a McNeil type core sampler, inserted to a depth of 10 cm, and subsequently sorted with a set of standard nested sieves, dried, weighed in each of the four size fractions (< 1.40, < 0.85, < 0.50, < 0.09 mm diameter) and then analyzed for size distribution characteristics. Cobble and gravel substrates were also analyzed for shape (Folk, 1974) to determine likelihood of compaction and ability to be moved (rolled) by water flow and ice.

Juvenile Fish Populations

Fish populations were studied on four occasions: baseline (pre-development) data was collected in 1979 and 3 years of post-construction data were collected in 1985, 1987, and 1988 to compare salmonid standing stocks (numbers and biomass), age class structure, and species composition. Eight (8) sites were established on the regulated West Salmon River while control sites (n= 5 to 6) were established on unregulated streams within the same drainage basin.

Fish population estimates were determined by electrofishing using the fixed effort (successive) removal method (Scruton and Gibson, 1995) and sites were replicated precisely between years. Fish captured were anaesthetized, identified to species, length (nearest mm) and weight (nearest 0.1 g) measured, and ages were subsequently determined from scale samples. Abundance estimates and biomass were obtained using the MICROFISH DOS software program (Van Deventer and Platts, 1989). Separate population estimates were derived for: i) all salmonids, ii) ouananiche and brook trout separately, and iii) separately for fry and older age classes of each species. A Two Way ANOVA procedure was used to test for differences in both density and biomass between (i) regulated and unregulated (control) sites, (ii) between years, and (iii) interactions between years and effects of regulation (SAS, 1979). A One Way ANOVA procedure was used to test for differences before (1979) and after (1985, 1987, and 1988) regulation (SAS, 1979).

IFIM Studies

The fully regulated 4.2 km reach of the West Salmon River was designated for the Instream Flow Incremental Methodology (IFIM) study in 1994. The reach, mapped for meso-habitat types (after Gibson et al. 1987), consisted of riffles (28 segments, 28.6%), runs (23 segments, 28.7%), pocket waters (17 segments, 33.1%), flats (1 segment, 8%) and pools (2 segments, 1.5%). A stratified random selection approach was used to select representative transects based on the longitudinal mapping. The least available significant habitat type (e.g., flat at 8%) was used as the 'selector' and, using criteria that no more than 10 % of the habitat distance be represented by any one transect, the hydraulic transects were established as follows: riffle (3), run (7), pocket water (4), and flat (1). Hydraulic measurements were collected at three flows; high (5.2 $m^3 \cdot s^{-1}$), moderate (2.6 $m^3 \cdot s^{-1}$), and low (1.3 $m^3 \cdot s^{-1}$) following standard procedures as outlined in Trihey and Wegner (1981).

Microhabitat data were collected in the summer of 1994 at two flow conditions, 1.3 and 2.6 $m^3 \cdot s^{-1}$. Data were collected through snorkling observations along 10 to 30 m long transects established at a 60^0 angle to the bank, with the effort (number of transects) split equally between rifle, run, and pocket water habitats. A marker was placed on the stream bed immediately below the focal position of each fish observed were marked at the focal (nose) position and species, age class (as determined by size), activity, and distance off the substrate (cm) was recorded. Water depth (cm), velocity (mean column, focal and bottom) ($cm \cdot s^{-1}$), as well as substrate were subsequently measured at each marker location. Habitat availability data (for depth, mean column and bottom velocity, and substrate) were collected along each transect at 1 m intervals. Habitat preference curves were developed for the variables depth, mean column velocity, and substrate for Atlantic salmon fry (young-of-the-year [YOY]; n= 150) and parr (1+ through 3+ in age; n= 99). Frequency analysis with (polynomial) curve fitting (Bovee, 1986) was used in criteria development using two approaches: (i) constructing curves from the ratio of habitat use to availability and (ii) constructing curves based on habitat use only (i.e., considering that sampling all habitat types equally removed the requirement to adjust for availability) - curves developed using this approach were subsequently used in the IFIM analysis.

Physical Habitat Simulation (PHABSIM) modelling was completed using commercially available software (RHABSIM) to calculate weighted usable area (WUA) estimates. Simple straight multiplication of the component suitability rating for each of the three habitat attributes was used. Each transect used in the analysis was weighted in proportion to its representation of the total habitat area. WUA estimates were generated over the range from 0.25 to 15.0 $m^3 \cdot s^{-1}$ (4 to 230 % of the MAF, respectively).

RESULTS

Geomorphic And Sedimentary Characterization

The study of the geomorphic and sedimentary characteristics of the West Salmon River, under regulated flow conditions, was conducted largely to address concerns as to the present and future quality of the spawning habitats. Hydraulic transects indicated that the narrow, higher gradient stretches in the upper section of the river, were characterized by coarse substrates (boulder, rubble, cobble) and fast flows, with reasonable deep water (> 20 cm) areas evident even at low flow conditions (1.3 $m^3 \cdot s^{-1}$) (Rogerson, 1986). The wider, lower gradient reaches, were characterized by cobble/gravel substrates and slower flows, with up to 50% of the channel being either dry or covered by water of 10 cm depth or less. Velocities from the hydraulic transects, in consideration of the theoretical transport competency (Hjulstrom Curve, Richards, 1982), indicated the maximum size particle that could be entrained was fine gravel, while average velocities would be capable of transporting medium to coarse sand. Core sample analysis indicated the cobbles and gravels were compact and spherical in shape suggesting they would roll easily and pack loosely, which was subsequently confirmed by field observation.

Qualitative sediment mapping revealed that certain sections of the river were characterized by a high proportion of silt to fine sand which formed a thin discontinuous veneer over the underlying coarser substrates. Accumulation of fine organic sediments along the river margins and in low velocity areas was apparent, likely originating from the river banks and delivered by erosion and rainfall events. Subaqueous grasses were growing in areas of sediment concentration while grasses and flowering plants had colonized sheltered shorelines areas and gravel bars. Sediment coring indicated that some of these materials were clearly derived from 'artificial sources', likely from dam construction and road building, particularly in the upper reaches immediately below the dam.

Juvenile Fish Populations

Total salmonid biomass ($g \cdot 100\ m^{-2}$) on the regulated West Salmon River was greater in all post-construction years (183.3 - 233.7, $\bar{x}$= 221.8) than prior to development (176.7) while the trout contribution to this total was also higher under regulated conditions (89.0 - 120.2, $\bar{x}$= 102.5), as compared to pre-development levels (50.7 in 1979) (Figure 3). Salmon biomass ($g \cdot 100\ m^{-2}$) was highest in 1985 (150.2) and declined to 113.6 and 96.5 in 1987 and 1988, respectively. Salmon parr biomass increased from pre-development levels (111.7) to 131.7 in 1985, then declined to 85.4 and 52.5 in 1987 and 1988, respectively. Total salmonid and salmon densities ($nos \cdot 100\ m^{-2}$) initially declined in the post-impoundment years (1985 and 1987) and then increased substantially in 1988, primarily as a result of dramatic increases in salmon fry abundance (Figure 3). Salmon fry density was greatest in 1988 (about 3 times pre-development levels), similar to pre-development numbers in 1987 and were 41 % less than pre-project levels in 1985. Conversely, salmon parr have demonstrated a steady decline in abundance under regulation (from 36.0 in 1979 to a low of 10.7 in 1987).

Between year differences in population and biomass were examined in the post-construction period in both regulated and control sites. Regulated sites on the West Salmon River demonstrated significant differences ($p< 0.05$) for total fish density (+), total salmon density (+), and salmon YOY density (+) between the years 1985 and 1988 and 1987 and 1988 while there were no significant differences in trout (total, YOY, and older age classes) densities between years. No significant differences in density and biomass for all species and age groups between years were apparent on the unregulated control stations.

The proportion of salmon fry in the West Salmon River, as an indicator of the quality of spawning habitats and reproductive success, has increased relative to pre-impoundment baseline and has increased throughout the post-development monitoring. In 1979, prior to development, the proportion of fry to older age classes of salmon was about equal (50%) and has since increased from 60.4% (1985) to 71.1% (1987) to 90.2% (1988) of the salmon population. This same trend was evident, but not as dramatic, in the salmon populations on the unregulated, control stations where the fry component has increased from 38.1% (1985) through 61.6% (1987) to 65.2% (1988).

IFIM Studies

Habitat preference criteria were developed for Atlantic salmon fry (0+ , length less than 55 mm) and parr (1+ and greater, length greater than 55 mm) for attributes depth, mean water column velocity, and substrate (Figure 4). Generally, salmon parr utilized faster, deeper water than did fry. Optimums for depth for fry and parr were 25 and 35 cm, respectively. Optimum velocities for fry were mostly below 25 $cm \cdot s^{-1}$ while parr exhibited a wider range of velocity use. Results are generally consistent with previously published criteria in deGraff and Bain (1986), Morantz et al. (1987), Heggenes (1990, 1991) and LeDrew et al. (this proceedings) although velocity preferences are lower than previously published.

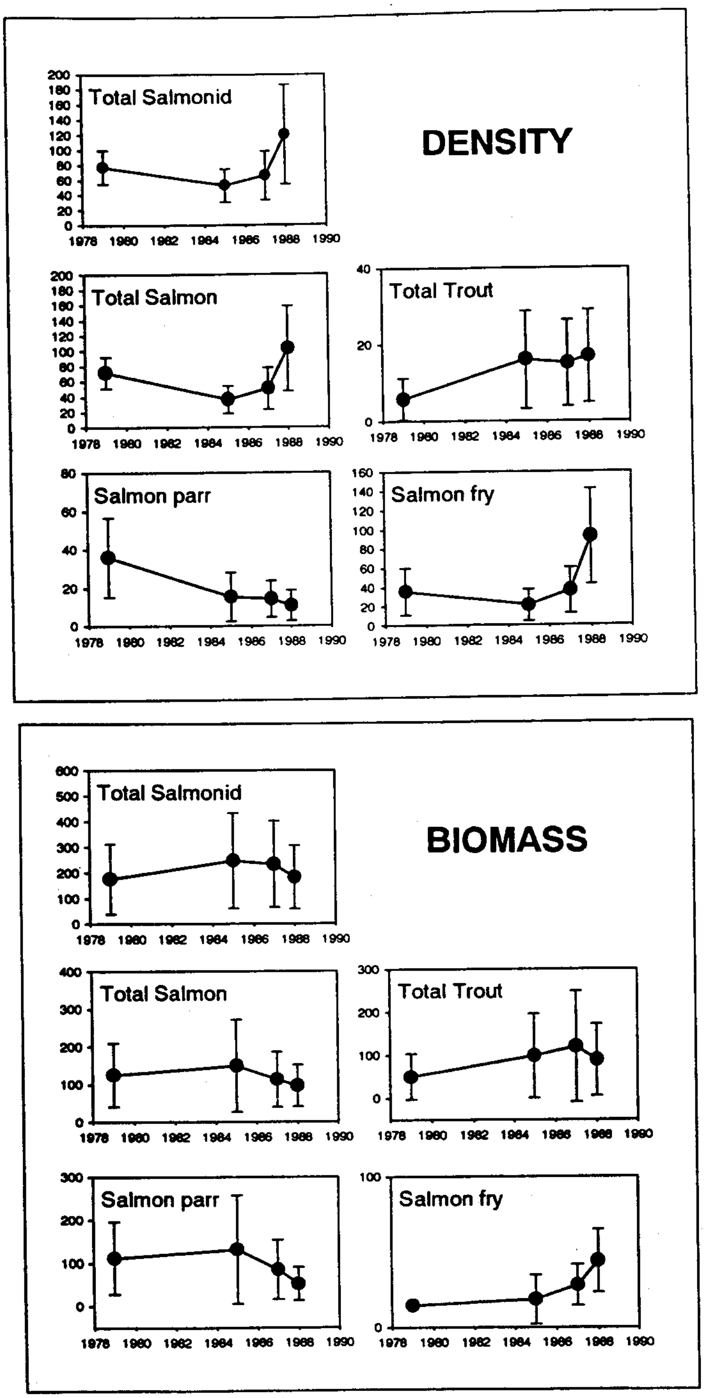

Figure 3. Mean (with standard deviation [bars]) density and biomass of (i) total salmonids, (ii) total salmon, (iii) total trout, (iv) salmon parr, and (v) salmon fry in the West Salmon River for the pre-development (1979) and post development (1985, 1987, 1988) periods.

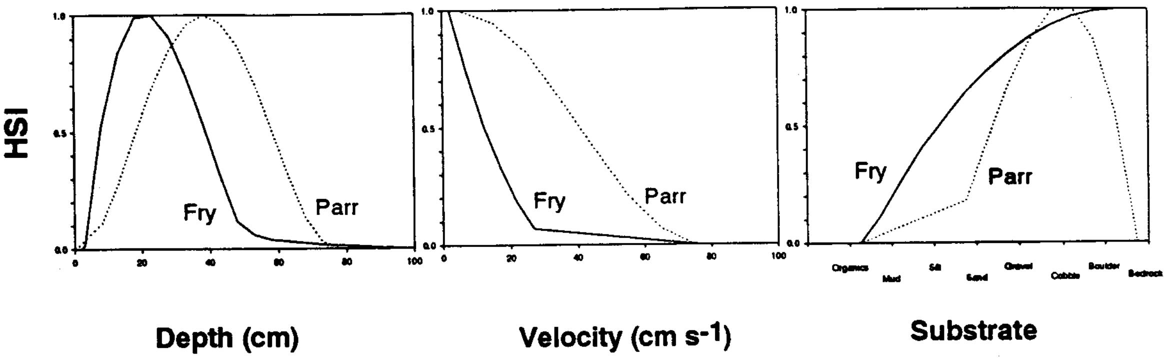

Figure 4. Habitat suitability index (HSI) curves for landlocked Atlantic salmon fry (solid line) and parr (dotted line) in the West Salmon River (1994) as used in the IFIM analysis.

The Weighted Usable Area (WUA) habitat index was calculated within the RHABSIM computer program by combining hydraulic-habitat simulations with species preference habitat criteria (above) over a range of modelled flows (Figure 5). For salmon fry, the optimum available habitat (WUA) was at 0.75 $m^3 s^{-1}$, with a modest but steady decline in estimated WUA above this level, consistent with expectations for a life stage that utilizes the lower ranges of available depth and velocity. At 2.6 $m^3 s^{-1}$ (the summer flow regimen), the WUA estimate was approximately 76% of the optimum for fry while the winter flow regimen (1.3 $m^3 s^{-1}$) was approximately 97% of optimum. Conversely, the optimum available habitat (WUA) for salmon parr was 3.5 $m^3 s^{-1}$ with sharper declines above and below this level, with WUA estimates at 1.3 $m^3 s^{-1}$ and 2.6 $m^3 s^{-1}$ of 65 and 98 % of the optimum, respectively.

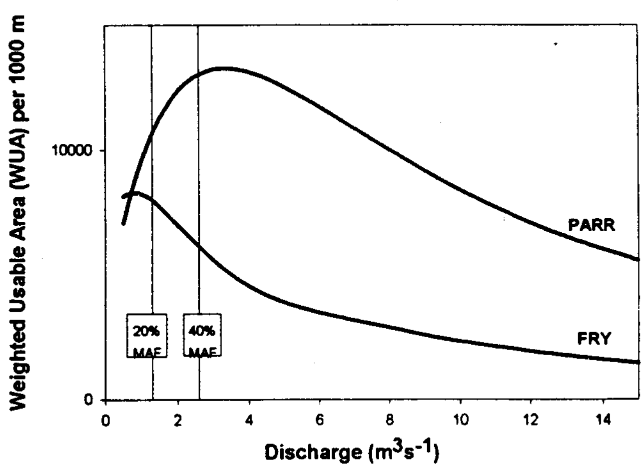

Figure 5. Estimates of weighted usable area (WUA) per 1000 m of stream for landlocked Atlantic salmon fry (FRY) and parr (PARR). The regulated flow regimen is also shown.

DISCUSSION

Studies on the geomorphic and sedimentary character of the river under regulation suggest it is incompetent with respect to much of its sedimentary calibre and unable to transport the predominantly cobble-gravel bedload. Regulation has led to a deterioration of the cobble-gravel substrate which could result in choking of interstitial spaces with fine sediment which would normally be flushed from the system during natural hydrological peaks. These locations are preferred habitats for rearing of salmon fry, and long term loss of these habitats could reduce the overall productivity of the river. The processes of aggradation and vegetation encroachment (succession) are occurring in a complementary fashion with accumulated sediment/organics providing the anchor for plant growth while the resulting vegetation is likely trapping additional fine materials. In the long term, it is expected accommodation will likely result in reduced river width which could result in loss of spawning habitat, depending on the use of, and reproductive success in, substrates along the margins. Consequently, the spawning habitat environment is not hydrologically dynamic and will be subject to future deterioration. This deterioration may be arrested for a considerable period due to lack of availability of fine sediments in the watershed and the fact that the Cold Spring Pond dam acts to trap upstream sediments. Further, the loss of width may be accompanied by increased flows in the wetted channel which may help reduce sedimentation of substrates.

The rationale for lower winter flows under the 2-level regime was based, in part, on the assumptions that juvenile fish would move to deeper waters to overwinter, that metabolic rates would be lower, and that reductions in flows from levels following spawning would not reduce incubation potential provided that redds are covered by water and oxygen levels are sufficient (Stalnaker and Arnette, 1976). The monitoring program was undertaken to determine if the winter flow regimen (20% MAF) would provide adequate conditions for incubation and survival of eggs and, in the absence of hydrological peaks (flushing flows) in the regimen, whether gradual aggradation of silt, increased growth of algae/macrophytes, and encroachment of riparian vegetation would impair the quality of reproductive habitats over the long term. Monitoring results indicate that the proportion of fry in the total salmon population has increased under regulated flow conditions and, with the exception of 1985, densities were equal to or greater than pre-development and results suggest no biological evidence of reduced quality of spawning habitats.

The densities of the older age classes of salmon, however, have declined under the regulated flow conditions. This is reflected in the decline in the proportion of salmon parr in relation to fry, both before and after project development and through the 3 years of post development monitoring, with a concurrent decline in parr density (from 36.0 to 10.7 per unit; 1979 to 1988). This same decline is evident for salmon parr biomass (excepting 1985). It is speculated that this decline is related to poor conditions for overwintering survival under the two-tiered flow regimen where flows are considerably less than natural pre-development levels (about 19 % of normal flows during this period, Figure 2). The West Salmon River is dominated by habitats characterized as shallow, low gradient, containing smaller substrates and moderate velocities that would be subject to significant effects from dewatering, particularly under winter conditions. Lower and regulated winter flows could lead to greater anchor ice formation and exposure of gravel bars and riffles which contribute to reduction of overwintering habitat.

The apparent decline in the older age classes of salmon under flow regulation is consistent with the change in habitat preferences of juveniles of the species as they grow. Habitat use varies with fish size/age (Heggenes, 1990) and, in general, as young salmon grow they select progressively faster, deeper water over coarser substrates (Symons and Helland, 1978, deGraff and Bain, 1986; Heggenes, 1990, 1991; Scruton and

Gibson, 1993). Salmon fry prefer pebble/cobble dominated substrates in shallow, slow water areas (Symons and Helland, 1978; deGraff and Bain, 1986; Gibson, 1993; Scruton and Gibson, 1993). Older juveniles (parr) become very territorial with territory size varying with velocity (flows), with larger territories associated with lower velocities resulting in less space available for fish.

The IFIM analysis includes consideration of differences in microhabitat requirements of the two age groups of juvenile salmon and results suggest that habitat conditions under the controlled flow regimen would favour salmon fry over the older age classes (Figure 6), which in turn are reflected in the post-development fish densities. Estimates of available habitat (WUA) for salmon parr are 65 and 97 % of the optimum at the flows specified in the Tennant's regimen (1.3 and 2.6 $m^3 \cdot s^{-1}$, respectively), and the analysis does not consider possible further habitat loss under winter conditions. The results of the monitoring of fish populations are therefor consistent with the predictions provided by the detailed habitat-hydraulic modelling completed within IFIM.

Much of our understanding as to habitat requirements of juveniles salmonids are based on summer, open water studies and there is limited knowledge of winter habitat preferences. Chapman (1966) suggests that habitat (space) may be the principal regulator of stream dwelling salmonids in winter. In winter, parr tend to move to areas of slow water velocity, often hiding beneath large substrate particles (Cunjak, 1988). Older parr tend to move into (shelter in) coarser substrates (rubbles, boulders) when temperatures decline to 9 °C and less and also move to lower velocity, deeper areas (pools) as temperatures decrease (Gibson, 1978). Ice accumulation, both on the surface and the formation of anchor ice, can significantly reduce available space in the water column and substrate and regulated flow conditions encourage both types of ice formation. The West Salmon River is characterized as broad and shallow with few well defined, deep pools and generally consists of habitats not well suited to overwintering of salmon and trout.

A major consideration that was not included in the flow regimen for the West Salmon River was the requirement for 'flushing flows' to periodically simulate natural peaks in the hydrograph. Extreme events often are a major factor in controlling habitat quality and diversity over the long term (Mundie,1991) and in regulation of substrate size and distribution, habitat types (e.g., pool to riffle ratio), and stream bank and cover conditions (Hynes, 1970). These flows are necessary to remove fines and sediments from the substrates and provide maintenance and dynamics of the channel and riparian habitats. Considerations for flushing flows include magnitude, duration, and timing of the release and a variety of techniques are available to assess this requirement (Reiser et al., 1989).

The Tennant Method has been one of the most widely used 'standard setting' techniques for planning and establishing minimum flows prior to wide scale implementation of IFIM (Reiser et al., 1987) and was selected as the preferred method for use on the Upper Salmon Hydroelectric Development, constructed in the early 1980s. The approach is easy to apply, however, it is recognized the method is intended as a reconnaissance level or planning approach, and not as a technique to prescribe a minimum flow for a specific river reach and species assemblage (Bietz et al., 1985). Further, the method recommends minimum flows and suggests the incorrect assumption that flow below these levels will be detrimental to fish populations without recognizing the incremental nature of changes in habitat quality and availability as a function of stream flow (Mordhart, 1986).

CONCLUSIONS

Geomorphic studies have provided initial evidence of sediment accumulation and channel accommodation over the early years after impoundment, with the materials from construction related sources and fine organic materials derived from within the watershed, residing in sedimentary repositories. Biological monitoring to 1988, as possibly the best indicator of a deterioration of the spawning environment, has suggested no apparent detrimental effects from this sediment deposition on spawning and egg incubation as evidenced by the densities of salmon fry in post-impoundment years. Salmon fry in the West Salmon River have increased from pre-construction baseline while, conversely, densities and proportion of older age classes of salmon (1+ and greater) have decreased under regulated flows. This decline has been suggested to be related to poor overwintering habitat conditions as a result of the lower flow regimen. IFIM studies suggest winter microhabitat conditions (depth, velocity) under regulated flows would favour salmon fry over the older age classes which support the results of monitoring of juvenile fish populations.

The Tennant (Montana) Method, a 'standard setting' approach, was used in conjunction with professional judgement to establish the regulated flow regime for this development and was considered an acceptable approach at that time (circa 1980). It would appear that the approach has recommended flow levels that are adequate for maintenance of reproductive success (successful spawning and egg to fry survival), however, flow conditions for older age classes of salmon, particularly during the overwintering period, may be limiting survival. In the absence of a flushing flow component to the regimen it is likely that there will be deterioration of the reproductive habitat over the long term. The results of these studies suggest that the Tennant Method, as applied in this project, may not be suitable, without further refinement, for setting detailed flow regimes for fish habitat protection in regulation of insular Newfoundland rivers. In projects requiring a high degree of protection for juvenile salmonid populations, approaches such as IFIM which allow for hydraulic simulation and prediction of incremental change in habitat quality with flow would be preferable to standard setting approaches. Adaptive instream flow management techniques, based on the results of biological monitoring, may also play an important role in setting of reservoir releases and in refinement of post-project flow regimens (Sale et al., 1991).

This study has pointed out a major shortcoming in the art of hydrological-habitat modelling, and in knowledge of fish habitat interactions - the inability to model flows and predict habitat parameters under winter conditions . In cold, northern rivers, where fish are exposed to harsh winter conditions for a large portion of the year, overwinter survival can be a major regulator of populations (Power, 1981). Knowledge and model development, with predictive ability, will need to improve in these areas before regulated flow regimes can be set that address habitat conditions affecting all life stages of resident fish.

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THE BASIC FLOW:
AN ALTERNATIVE APPROACH
TO CALCULATE MINIMUM ENVIRONMENTAL INSTREAM FLOWS

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ABSTRACT

Natural systems are organized and self-regulated according to information that is arranged in consecutive data series. Generally speaking, these consecutive data series are either gradients of spatial data or time series. Life itself is organized and transmitted by information arranged in data series: the DNA genetic code.

Three different approaches have been used to study the environmental instream flow of the rivers, resulting in what we call hydrologic (historical discharge methods), hydraulic (cross-sections methods), and hydro-biological (habitat simulation methods) techniques. Hydro-biological techniques have gained increased recognition and are widely accepted currently. They are based on simulation techniques that describe physical river habitats of some species as a function of streamflows. The main variables used to characterize habitats are distribution, composition and type of streambed materials, depth and water velocity, as well as other secondary parameters such as water quality, food availability, communities in the river bank. If we accept that these habitat variables are related to streamflow regimes, then we can define all of them as dependent variables in equations where the only independent variable is the river flow. Under this assumption, river flows and their variations determine physical and biotic factors of the river habitat. Streamflow time series, then, include all the needed information to explain the structure and biological requirements of a specific river or river reach. We only have to determine a good method to get that information from the series.

This paper proposes the application of the simple moving average forecasting model as the tool to get the information from hydrological series of daily mean flows. This technique is commonly used when studying irregularities in time series analysis. The model was applied to hydrological data from 11 Mediterranean rivers in the region of Catalonia (Spain). These rivers were selected because they are representative of the maximum variability of river regime patterns in this region (from snowmelt pattern to rainfall pattern). The results of this study show that there is a characteristic minimum flow for each river, which depends on its particular streamflow time series. This minimum flow, which is called "basic flow", goes from 5 to 50% of the mean annual flow, with 54% of the cases ranging within the 10 to 20% interval. This interval agrees with values obtained through other hydrobiological methods. The method proposed in this paper has the advantage over these other methods that it is easy, fast, and economical to apply.

The present article describes the methodology used and the constraints of the proposed model, and discusses the results obtained from its application and their interpretation. A research project is actually being carried on (1996-1998) to establish the biological meaning of the basic flow. Results will be compared to those obtained from the application of other methodologies currently used to the same study areas.

KEY-WORDS: Basic Flow / Environmental Instream flow / Regulated River Management.

INTRODUCTION

Environmental instream flow determination is a controversial environmental issue of major importance for water resources management in the Mediterranean developed countries, together with eutrophication, deforestation and erosion control (Prat, 1995).

Currently, there is not a universally accepted standard methodology to calculate environmental instream flows. All the current methodologies have three characteristics that reduce their credibility: subjectivity, arbitrariness, and lack of ecological significance in the calculation process. Three different approaches have been used to study the environmental instream flows of rivers, resulting in what we call hydrologic (historical discharge methods; i.e. Tennant, 1976), hydraulic (cross-sections methods; i.e. Collings, 1972) and hydro-biological (habitat simulation methods; i.e. Bovee, 1982) techniques. The latter techniques have gained increased recognition and are currently widely used, although they have had their share of criticism (Gan & Macmahon, 1990; Bovee, 1995). They are based on simulation techniques that describe physical river habitats of some species as a function of streamflows.

This paper proposes an alternative approach to calculate minimum environmental instream flow (MEIF), which is called Basic Flow (Qb; Palau, 1994), that sets forth theoretically valid statements to overcome the three constraints of current methods (subjectivity, arbitrariness, and ecological significance). The concept of MEIF is very simple to define: it is the flow that maintains a given level of biological functionality to preserve the continuity of natural communities and processes without any human intervention. So, the key of the problem is to establish which is that correct biological functionality level to be preserved. An analytical solution can be found in the relationship between functionality and habitability, as it is stated in the IFIM methodology (Bovee,1982).

BASIS OF THE PROPOSED METHODOLOGY

In order to quantify Qb, the first step was to define the direction of the research efforts. Some basic concepts of Ecological Theory were very useful in that goal.

Natural systems are biosphere compartments regulated and organized according to an information that is presented in a flow format. This information is arranged in consecutive data series (gradients of spatial data or time series), that are interpreted by the natural systems to define their abiotic characteristics, biological composition, and their organization and functioning (Margalef, 1991). Perhaps, the best example of this statement is that life itself is organized and transmitted by information arranged in data series: the DNA genetic code.

Space and time do not contribute in the same way in providing information to natural systems. Space usually brings information that is integrable (predictable). A natural system in a given space presents an organization level that depends on its environment, but this environment changes in time according to series of events. Therefore, the event time series are the independent and primary variable in providing information relevant to natural systems organization. Consequently, time series are the objects to study in order to establish Qb values.

Streamflow time series are, in a way, the DNA of a river (genotype). In the DNA, the combination and alternation of nucleotide determine protein synthesis and the organization and aspect of life. In the rivers, the streamflow temporal variability and magnitude are the variables determining the organization and aspect of the system. That means they determine river banks, streambed width, bed substrate types, water velocities and depths distribution, etc. And these factors condition the type of aquatic community that will live in the river.

River regulation results in great alterations and changes in rivers, as they would be produced by genetic code manipulation. The only advantage is that they are reversible. The simplest river regulation acts by controlling and reducing total streamflow and, therefore, eliminating floods. This regulation results in a drop of water availability for the natural environment and a lost of streamflow time series variability. The same thing happens when artificial high flows are generated downstream a reservoir. In all cases, the river genetic code is regularized resulting in very similar river characteristics (phenotype) downstream any reservoir.

A first important conclusion is that if we want to conduct proper river management actions, we have to maintain its "genetic code" as similar to the natural regime as possible.

Figure 1 shows all the variables that are frequently used to define environmental instream flows, as well as the simple dependence relationships among them. The main variables used to characterize habitats refer to fish communities. These variables are distribution, composition and type of streambed materials, depth and water velocity, as well as other secondary parameters such as water quality, food availability, and communities in the river bank.

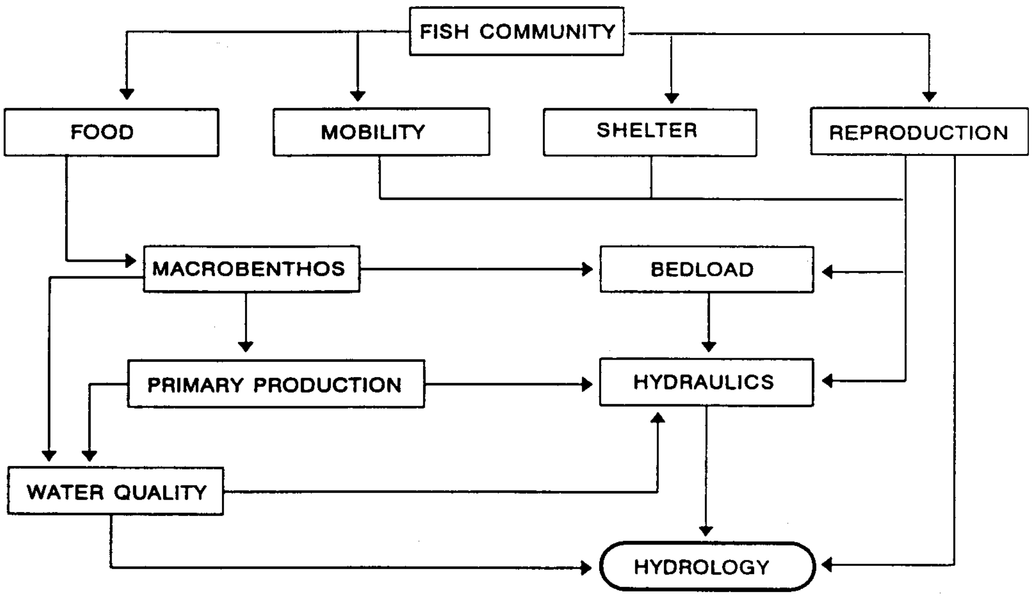

Figure 1. Flow chart of the organization and interrelationships among river variables.

If we accept that these habitat variables are related to streamflow regimes, then we can define all of them as dependent variables in equations where the only independent variable is the river flow. So, under this assumption, river flows and their variations determine physical and biotic factors of the river habitat. Streamflow time series, then, include all the needed information to explain the structure and biological requirements of a specific river or river reach, that means, everything needed to define the biological functionality of a system.

Figure 2 shows the theoretical relationship between habitability and streamflow. When a certain upper flow level (flood) is surpassed, the limnological organization in a river is altered. In the same way, there is a minimum flow level that if not reached, results in the limnological organization being also significantly modified. The latter one is what we call Basic Flow, and it is specific for each river and even river reach, depending on its particular characteristics. We only have to determine a valid method to extract that information from the time series and look for the adequate break-point in the relationship between flow and habitability.

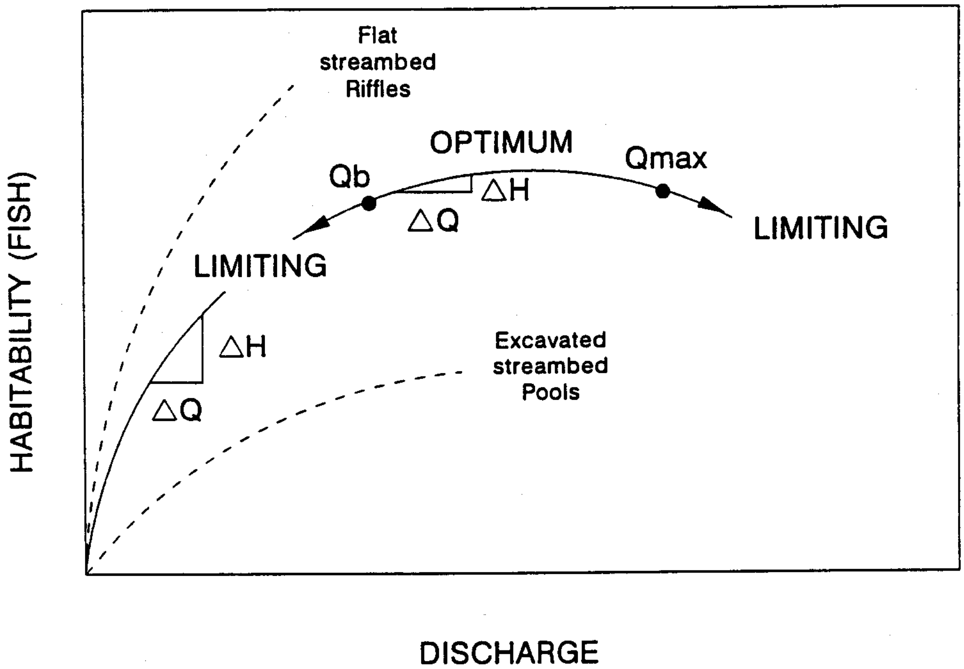

Figure 2. Theoretical relationship between discharge flow and fish habitability instream.

Finally, the way an aquatic organism "reads and interprets" the environment information is by integration. That means habitability and survival conditions of a species related to streamflow should be continuously modified with the information provided by the new daily streamflow data. A minimum flow would be defined from the continuous integration (cumulative information) of temporal values of the series. The appropriate mathematical approach to detect that minimum flow is the simple moving average forecasting model applied to increasing streamflow intervals, looking at irregularities in the generated series of accumulated flows. These irregularities would be the wanted break-points.

MATERIAL AND METHODS

Eleven rivers in Catalonian watersheds were selected for the study. These rivers were chosen because they are representative of the maximum variability of river regime patterns in this region (from snowmelt to rainfall-originated patterns). A control section on a flow gaging site was selected in each river to calculate the Qb. These sections were characterized by a natural regime (non regulated reaches).

Table 1 shows the main hydrologic characteristics of the selected river reaches. These river reaches can be classified as follows:

- Temporary rivers with rainfall-groundwater hydrograph regime: Siurana.
- Temporary rivers with mediterranean rainfall hydrograph regime: Caldes, Tenes, Tordera and Matarranya.
- Permanent rivers with mediterranean rainfall hydrograph regime: Fluvià.
- Permanent rivers with snow + rain hydrograph regime: Llobregat and Segre (1 and 2).
- Permanent rivers with snowmelt hydrograph regime: Noguera de Tor and Santa Magdalena.

Table 1. Main hydrologic characteristics of the river reaches studied.
(Qrm = real mean daily flow; Qm = mean daily flow of the 10 year series)

River	Watershed total area (km^2)	Watershed control area (km^2)	Altitude (m osl)	Qrm (m^3/s)	Qm (m^3/s)	Qmax (m^3/s)	Qmin (m^3/s)	Coef. var. (%)
Siurana	627	88	410	0.29	0.04	8.5	0.00	507.5
Caldes	112	110	56	0.26	0.21	14.9	0.00	328.6
Tenes	154	152	75	0.51	0.40	18.0	0.00	219.8
Tordera	894	802	35	5.65	5.67	690.0	0.00	395.5
Fluviá	1125	804	99	6.76	7.20	430.0	1.00	235.7
Llobregat	4948	23	929	1.36	1.29	20.0	0.23	98.4
Segre-1	22579	1023	868	10.4	12.34	147.6	0.22	87.9
Segre-2	22579	2700	440	32.5	24.78	1196.5	0.72	126.4
Noguera de Tor	249	235	910	7.5	6.57	51.9	0.08	89.8
Matarranya	1727	48	580	0.398	0.33	28.9	0.03	325.9
Sta. Magdalena	111	76	1305	1.55	2.23	70.6	0.05	185.0

Mediterranean-type regimes generated either by rain or snow precipitations are common in Catalonia, so more of them have been included in the sample.

This sample of selected reaches is sufficiently representative of the spectrum of watershed surfaces, altitudes, streamflows, and hydrological variability in the study area. Table 1 shows that some sites are close to the sea level (Tordera, 35 m osl) and other high up in the mountains (Santa Magdalena 1305 m osl); some have large watersheds (Segre-2, 2700 Km^2) or very small (Llobregat, 23 Km^2). Regarding hydrologic magnitude, some annual average streamflows are very low (Siurana, 0.04 m^3/s) or relatively high (Segre-2, 24.78 m^3/s), some behave as ephemeral at certain times of the year (Siurana, Caldes, Tenes, Tordera), and some suffer often great peak flows (Tordera, Fluvià, Segre-2), which causes a wide range in the values of the coefficient of variation, from 87,9% (Segre-1) to 507,5% (Siurana).

We used daily average flows series from the last ten years to calculate the Qb. Several reasons led to the selection of this time period. First, we considered that ten years time looked like a good compromise between sufficient information and manageability of the data. Secondly, in the Mediterranean environment, extremely variable hydrological conditions cause the present and future of a river to be determined by its recent past and not by events further behind. Moreover, ten years time is a longer period than the expected life span of the aquatic species in any

Mediterranean river. Finally, we checked the variability of the Qb calculated with data from an increasing number of years, and we found stable values from six to eight years on.

Average streamflow (Qmr) in the sites was computed from long series of data (n > 30) available at the gaging stations, and also from data in the last ten years (Qm) as shown in Table 1. Differences between this two values of average flow in the majority of sites reveals the critical drought period the study area suffered in the last years. These results point at the importance of considering more recent flow data versus long time series of hydrological data for river management; long time series may be more representative of hydrological conditions, but less realistic from an ecological perspective.

The Qb was computed independently for the ten years considered for analysis. We applied moving averages to increasing intervals of consecutive data (average daily flows) up to a maximum of 100 values. This limit was established in order to avoid large data matrixes and because an average of 100 consecutive daily flow values would include the period of lowest water levels on any study site (100 daily values cover more than 3 months, which is a longer period of time than the usual duration of low water levels in the summer months). Starting with a matrix Q (365 x 10) of average daily flows q_i^j where i = day in the year and j = years, we applied the simple moving average forecasting model as follows:

(1)
$$a_i^j = \left(\frac{1}{j+1}\right) \sum_{k=0}^{k=j} q_{i+j}$$

where a_i^j are the moving averages for $(1 \leq i \leq 365\text{-}j)$ and $(0 \leq j \leq 99)$ part of matrix A (365 x 99). Next, we obtained the minimum value for each column j in matrix A, resulting a vector V composed of values v_j such as:

(2)
$$v_j = \min(a_i^j) \qquad for \quad 1 \leq i \leq 365-j \quad , \quad 0 \leq j \leq 99$$

Repetition of this process for each study year (10) resulted in a matrix F (10 x 100). We obtained a new vector M by column arithmetic average. We calculated the relative increase between each pair of consecutive values as follows:

(3)
$$b_k = \frac{(m_k - m_{k-1})}{m_{k-1}}$$

Qb is defined as the flow m_k related to the largest relative increment h_{max} such as:

(4)
$$h_{max} = \max(b_k) \qquad for \ 1 \leq k \leq 99$$

RESULTS

Table 2 shows Qb values for each river reach. Some hydrological parameters used to evaluate the Qb are also presented in the same table.

Table 2. Basic flow and related hydrological significance.
(% of Qm = percentage of Qb over Qm; % days < Qb = percentage of days with flow lower than Qb; mean Qc = mean classified daily flow; Qmost freq. = most frequent flow interval in the series)

River	Qb	% of Qm	% days < Qb	Mean Qc	Q most frequent
Siurana	0.022	50	10.5	Q327	0-0.08
Caldes	0.012	5.7	16.3	Q305	0-0.15
Tenes	0.038	10	18.9	Q296	0-0.18
Tordera2	0.663	11.69	27.6	Q264	0-0.69
Fluvia2	1.500	20.8	7.6	Q337	1.72-2.15
Llobregat1	0.481	37.3	40.5	Q217	0.4-0.6
Segre-1	3.331	27.0	12.5	Q319	6.6-6.75
Segre-2	4.1	16.5	1.8	Q358	10.8-12
Noguera de Tor	1.072	16.3	1.2	Q361	2.08-2.6
Matarranya	0.091	27.3	16.6	Q304	0-0.29
Sta. Magdalena	0.331	14.8	0.79	Q362	0-0.71

The Qb varies between 5.7% and 50% of the mean annual flow, with 54% of the cases ranging within the 10 to 20% interval. The maximum Qb was obtained for the Siurana river site. Siurana is a temporary river mainly fed by groundwater. Therefore, its baseflow has a constant level, with very short low flow periods, and moderate peak flows. In fact, its estimated Qb value (0.022 m^3/s) is very close to the minimum registered flow different from 0 in the river (0.018 m^3/s).
As it would be expected, Qb values raises with increasing mean annual flow and watershed area of the control section.

Qb values for rivers with mediterranean rainfall hydrograph regimes were very similar (from 0.012 to 0.092 m^3/s), translating into 5.7% to 27.3% of mean annual flows.

Rivers with snowmelt + rainfall hydrograph regimes showed Qb values from 0.48 m^3/s to 4.1 m^3/s, whereas rivers with snowmelt hydrograph regimes showed Qb ranges from 0.33 to 1.07 . These values result in percentages of mean annual flows from 16.5% to 37.3%, and 14.8% to 16.3% respectively.

Rivers with rainfall and snowmelt + rainfall regimes show a tendency for more conservative Qb values (a higher mean annual flow percentage) when mean annual flow and flow variability are low (Figure 3). However, these results cannot be conclusive since sample size is too small to make any statistical inference.

Qb values fall within the most frequent flow range, in most cases, or below. This fact gives coherence to the methodology, since MEIF numbers higher than most frequent flows (biologically characteristic flows) would not be realistic.

Most of the Qb calculated were significantly higher than absolute minimums, as it is showed in Table 2 (% of days in a year under Qb flow level). This percentage is also expressed in classified flows (number of days in a year with Qb flow level equalled or exceeded), obtaining values ranging from Q_{217} to Q_{361}. As a reference for comparison, some european countries use classified flows Q_{330} or Q_{347} to establish MEIF levels (Palau *et al*, 1995).

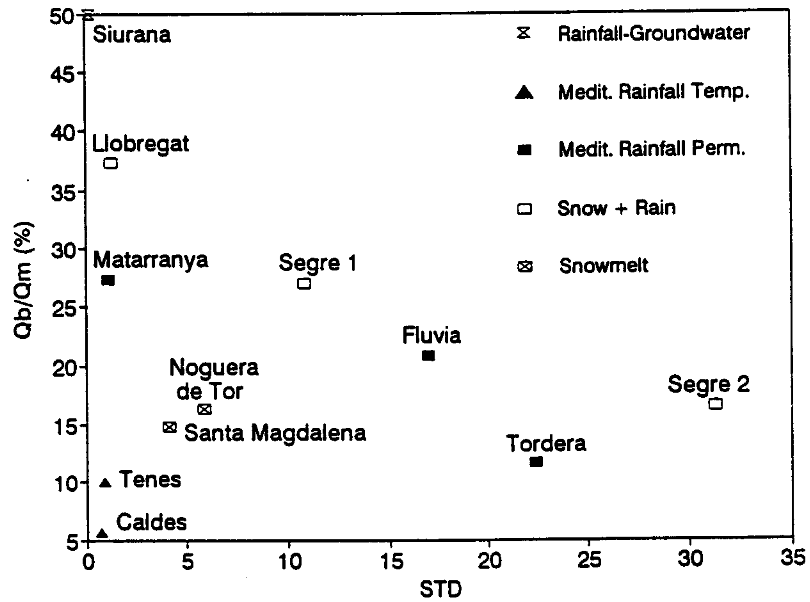

Figure 3. Relationship between flow variability (STD, standard deviation) and percentage of Qb over Qm for each type of river regime.

DISCUSSION

The calculation process suggests that Qb values depend on the number of days of low flow (usually summer flow in Mediterranean countries) and flow change increments. Qb is defined from the mean daily flow (Q1) ending the longest minimum low flow series in the average year. An exception of this definition occurs when there is a shorter minimum low flow sequence ended by a very high mean daily flow (Q2), which implies a bigger relative flow increment than Q1. In this case, Qb is defined from Q2.

The capabilities of the proposed method have been evaluated by mean daily flow series simulations (Figure 4 to 6), in order to assess the sensibility and mathematical functioning of the method. In the simulation, Qb has been calculated in different situations varying flow values of two consecutive peak flows, and the length of the low flow sequence between them (inter-peak flow). Figures 4 to 6 show the mean daily flow series used for the simulation, and the minimum flow series obtained from the simple moving average forecasting model application. They also show the values that define the largest relative increments used to calculate Qb (underlined values).

Figure 4 (A, B, C) shows the simulation of two consecutive peak flows (Q1<Q2) where the inter-peak flow length has been varied. Considering that the difference between Q1 and Q2 is small, Qb is always defined from Q1 in the three situations (A, B, C) regardless of the reduction of the low flow period between the peak flows.

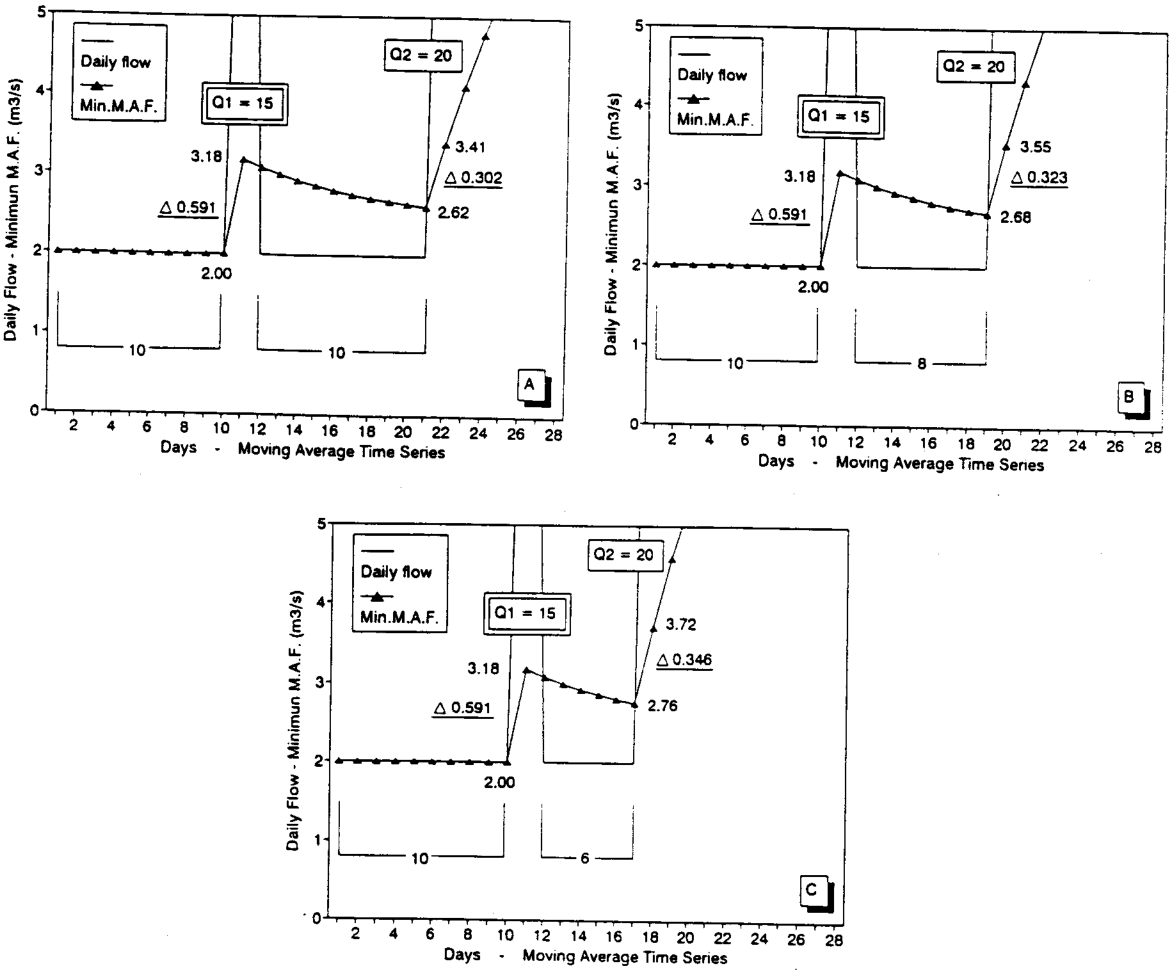

Figura 4 (A, B, C). Simulation of different inter-peak flow lengths between two consecutive similar peak flows.

Figure 5 presents the same situation that Figure 4, but with a higher difference between the peak values (Q1<<Q2). Results show that when the inter-peak flow length is not highly reduced, Qb is defined from Q1 (Figure 5A and Figure 5B). On the other hand, when the length of the inter-peak flow is highly reduced Q2 creates a larger relative increment and, therefore, Qb is defined from Q2 (Figure 5C) .

Figure 6 models a situation with exchanged peak flow values and an increase of the inter-peak flow. When Q2 is the flow ending the longest minimum low flow series, Qb is defined from Q2 (Figure 6A). In Figure 6B, Qb is defined from Q1 since its high value creates the largest relative increment. The latter situation is similar to that from Figure 5C, with the only difference that the low flow periods behind the peaks have been exchanged.

Another important fact revealed by the simulation is that Qb values are not influenced by extreme peak flows. It has been proved that if a peak flow is increased from a value of 10 to 80 (with a minimum low flow value of 2), the Qb only increases from 2.63 to 2.90.

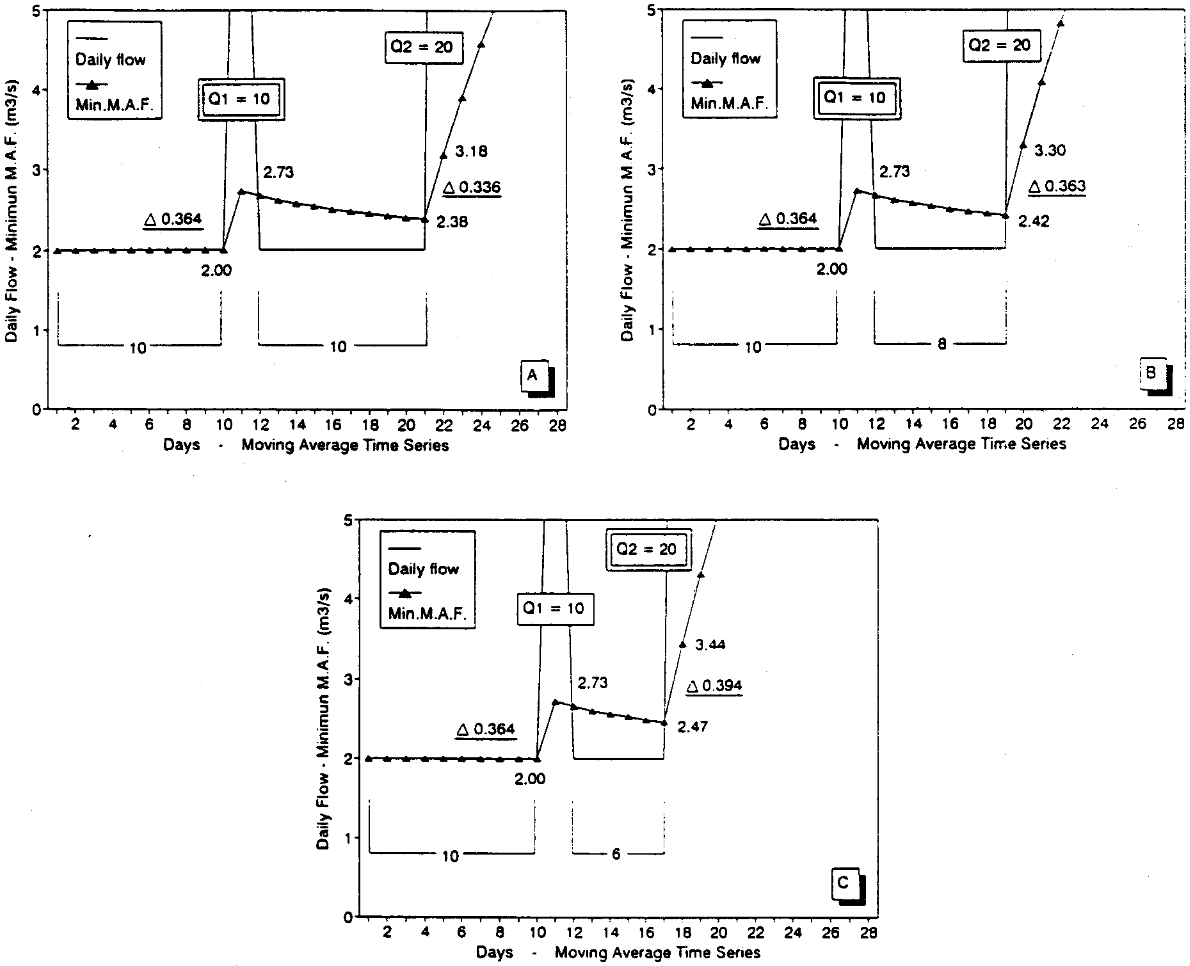

Figure 5 (A, B, C). Simulation of different inter-peak flow lengths between two consecutive dissimilar peak flows.

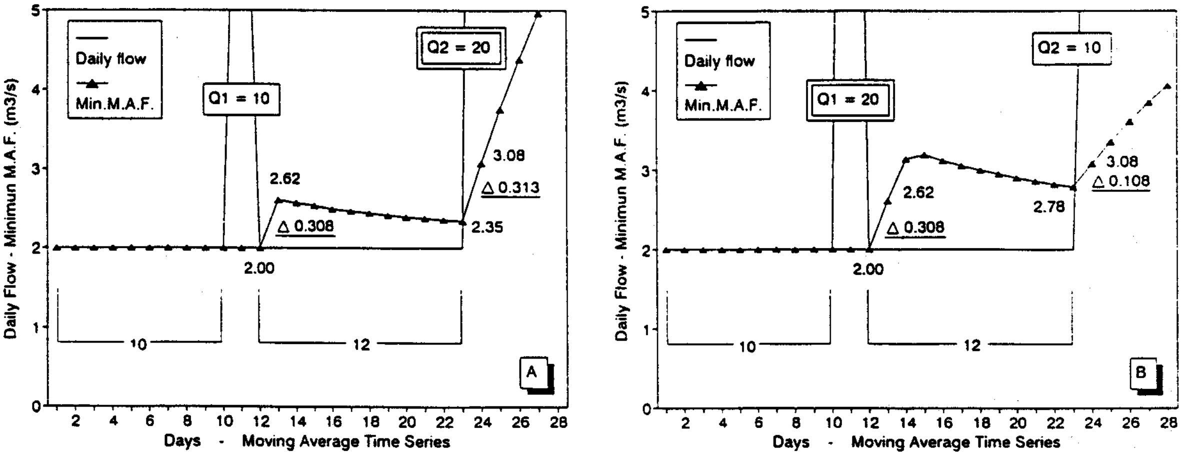

Figure 6 (A, B). Simulation of interchange peak flow values and increase of the inter-peak flow

The simulations carried out lead us to conclude that:

- Isolated extreme hydrological events, such as torrential precipitations (so frequent in Mediterranean areas), do not significantly affect Qb values.

- Qb is usually defined from the flows ending minimum low flow periods (summer low flows in Mediterranean countries), due to the use of relative increments in the calculation process instead of absolute increments. This makes sure that Qb is always higher than the most frequent flow in low flow periods. Furthermore, the location of Qb within the annual hydrograph suggests that Qb might have a biological meaning.

- In rivers with the same annual mean flow, short low flow periods produce higher Qb values than longer low flow periods. This is an important feature of the model since aquatic communities living in rivers with short low flow periods are not used to low flow regimes, regardless of the annual mean flow.

CONCLUSIONS

Qb calculation is based on 10 year mean daily flow time series. This is important if we consider that IFIM and other similar methodologies operate with daily flows from shorter series (less than 5 years), although the interannual variability is likely to be a relevant factor limiting fish abundance in streams (Davies, 1988).

The main characteristics of Qb that support its suitability for MEIF determination are:

- Qb values are easily replicated, objective, and not arbitrary, since the methodology used is based on real hydrologic time series data.

- It is sensitive to the hydrograph type and, therefore, the Qb is specific for each river or river reach. Despite the fact that this methodology is based only on hydrological data, it is more conservative when calculating MEIF for small rivers than other standard hydrologic techniques, which proves its sensitiveness and fitness to real life conditions.

- Qb values usually fall within the normal interval of MEIF values obtained with other methodologies currently used.

The method proposed in this paper has the additional advantage that it is easy to apply (it can be easily programmed in a PC), fast in obtaining results, and it is cost-efficient. The only input data needed are daily mean flows, which can be easily obtained at flow gaging stations. However, the Qb method is only a part of a more complex regulated river management proposal, which includes other parameters influencing the biological functioning maintenance of a river, such as the necessity of flow temporal variability, flushing flows, water quality, etc (Palau, 1994).

This method has not been validated with biological data yet. A new research project has just been initiated (1996-1998) with the aim of establishing the biological meaning of the Qb. Results will be compared to those obtained from the application of other methodologies to a large sample of rivers and river reaches in the Ebro Watershed, Spain. We expect that this project will provide statistical significance for the Qb capabilities pointed out in this paper.

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RECOMMENDING VARIABLE FLOW VALUES FOR FISH

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ABSTRACT

Instream flow needs (IFN) studies for the protection and management of fish have been conducted in Alberta over the past 10 years. Methods have ranged from the simple, Tennant, and the Tessman modification of the Tennant, to the more complex, Instream Flow Incremental Methodology (IFIM) using the PHABSIM group of models. The recommended flow values that are derived from these studies are used as input in basin-wide water planning exercises. Water planning is conducted using a water balance model which requires input from all water users, both consumptive, such as municipal water supply and irrigation for agriculture, and non-consumptive, such as fish and wildlife management, recreation and, where present, hydropower generation. Requested instream flow input values were originally limited to a "minimum" value, which were considered to protect basic water quality and instream flow needs at all times. Later in the water planning process, an "optimal" or preferred value was requested in addition to the minimum. The preferred value would protect desirable instream flow needs most of the time. While the single input value was convenient for the running of the water balance model, it gave the false impression that if this minimum value was met, the fisheries resources would be adequately protected. As this is not the case, it was necessary to provide to the water balance modelling process, a range of values, minimum to optimal, in order to effectively manage the fisheries resource. This paper presents the procedure that was used to take the principal output from the PHABSIM group of models, Weighted Usable Area (WUA) versus discharge, and convey that information into a range of recommended flow values that are linked to the naturally occurring availability of water in a river system.

KEY-WORDS: Instream flow needs / habitat / fish / modelling / flow / recommendations

INTRODUCTION

In the province of Alberta, instream flow needs (IFN) has increasingly become a major issue. As in other jurisdictions, social values have gradually shifted and in Alberta there is a growing need to define instream flow values to protect stream ecosystems. The Alberta Fisheries Management Division (AFMD) is actively participating in water management planning by first, carrying out instream flow needs studies and secondly, negotiating for suitable stream flows. The principal piece of legislation that governs water management in the province of Alberta, the Water Resources Act, is currently undergoing a comprehensive legislative review. This is the first major review of this statute since its inception in the late 1800's. Several issues are being debated, with instream flow needs for the protection of flowing aquatic ecosystems being one of the key items.

In the late 1970's, the South Saskatchewan River Basin Planning Program focused attention on the existing and potential conflicts between instream flow needs and the withdrawal and waste disposal uses of water in the basin. Later, in 1986, the Alberta Water Resources Commission released a report containing recommendations on water management in the South Saskatchewan River basin. The Commission's conclusions, based on public hearings held throughout the South Saskatchewan River basin during 1984, led to a number of recommendations pertaining to instream uses of rivers in the basin. In summary, the Commission recommended that, to be consistent with the philosophy of multi-purpose use, instream flow amounts should be established on a reach-by-reach basis for all major river reaches and tributaries in the basin. In the spring of 1990, the Minister of the former Department of the Environment released a policy statement which stated, among other things, that consideration must be given to the need to retain water in the river for instream needs, including recreation, fisheries, wildlife and the ecology of the river.

One of the mandates of the AFMD is to manage the fisheries in the province of Alberta. As the stewards of the resource, the AFMD was invited to participate in the instream flow needs process established by the government to: first, re-affirm the fish and wildlife values associated with free-flowing waters; second, to identify appropriate instream flow needs methods to be used in Alberta; third, to advise the government on flow regimes required to manage the fisheries resources and; fourth, to advise on the consequences of adopting various water management scenarios.

As with other jurisdictions, the AFMD went through the process of identifying methods that could be used to determine instream flow values for fisheries management. Fortunately, there was a considerable amount of experience and expertise from which to draw upon. Many natural resource agencies had gone through the exercise of investigating the various methods available and subsequently defining instream flows for fish in the early 1970's. The search for suitable techniques led to a summary document describing the entire range of approaches to define instream flow requirements for fish. Later, a selected few of these methods were examined in detail to see if they could be used in Alberta (Bietz *et al.*, 1985).

THE WATER PLANNING PROCESS

In the early 1980's, the AFMD set minimum flows for the main reaches of the rivers in the South Saskatchewan River Basin (Longmore and Stenton, 1981). The method that was used to determine these minimum flows was the Tennant Method (Tennant, 1976) which originated in Montana. The method is classified as a hydrologic method which is based on stream flow statistics and specifies, by month, some percentage of the mean annual flow. The method was used because it was inexpensive and could be readily applied to any reach of river where there were historical flow records. At the time, it was believed these minimums would protect the fisheries.

As the water planning process evolved, it soon became evident that only minimum flows would be met. The relatively large size of the water control structures on the main rivers and the perceived need to meet the "legal" licenced requirements meant that any flows above these minimums were allocated to consumptive uses. It was believed by others in the water planning process in Alberta that if minimum flows were achieved, then fish populations thrived. However, from the perspective of today's understanding, there are serious flaws with having only minimum flows available for fish habitat maintenance. In fact, if true minimum flows are maintained for extended periods of time, which is what the modelling results showed, fisheries would be seriously impacted and might cease to exist. The long-term effects of continuously maintaining these artificial minimum flows seldom are the same as the infrequent, naturally occurring, short-term effects that appear in the historical flow record. In addition to this problem, it seemed that regardless of the minimum instream flow value that was selected, it was regarded by some as impractical because it could not be "met" all the time even under natural flow conditions.

The AFMD soon recognised the need to use a better approach to determining instream flow needs for fish. A method was required that had to meet three basic criteria; first, it had to provide an answer other than a "minimum" flow value that would be used as a flow target in the water planning process; second, it had to be based on biology; and third, the results had to allow for meaningful interaction with the other water users. The search for such an approach led the AFMD down the same road that other agencies had gone before, eventually leading to a wide and varied collection of habitat-flow models. As was the conclusion reached by many others, the Instream Flow Incremental Methodology (IFIM) and the PHABSIM group of models emerged as a method that was most suitable for use to determine instream flow requirements for fish in Alberta.

The IFIM, and in particular the Physical Habitat Simulation system (PHABSIM), has had its fair share of controversy. Anyone familiar with the method knows there are critics and there are those who support it (Stalnaker *et al.*, 1995). Despite this controversy, the AFMD has used the PHABSIM on several rivers in Alberta largely because it met the three criteria set out by the Division; it provides more than a "minimum" flow value, it is based on biology and, it allows for meaningful discussion with other water users. To date, the IFIM approach has been completed or is underway in the South Saskatchewan River Basin on the Highwood River, the Oldman River, Pekisko Creek, Willow Creek, the St. Mary River, the Belly River, the Waterton River, the Bow River, the Sheep River, the Red Deer River, the Elbow River and the Kananaskis River (Figure 1).

The IFIM, developed by the U.S. Fish and Wildlife Service in Fort Collins, Colorado, is a concept which includes numerous models and a decision making process that is thoroughly described in several documents (Bovee, 1982; Milhous *et al.*, 1989). The principle output of the PHABSIM is a habitat index, expressed as a weighted usable area (WUA) versus flow relationship (Figure 2). It should be noted that reaching the generation of the WUA requires considerable time and effort relative to other IFN methods. However, as most practitioners involved in water negotiations know, once the habitat index versus flow relationships have been calculated, the real work has just begun.

The water planning process in Alberta was not originally set up to evaluate one scenario over another. Rather, it was designed to have all water users supply to the water balance model, flow requirements or flow targets. For the instream flow uses, the model only attempted to meet the targets and nothing more. Figuring out how to provide a range of flow targets, and thereby avoiding the problem of providing only a single minimum flow value from the numerous WUA versus flow relationships for the various species and life stages, was a challenge. In some of the studies carried out in Alberta there were times of the year when numerous WUA curves had to be considered for one study site.

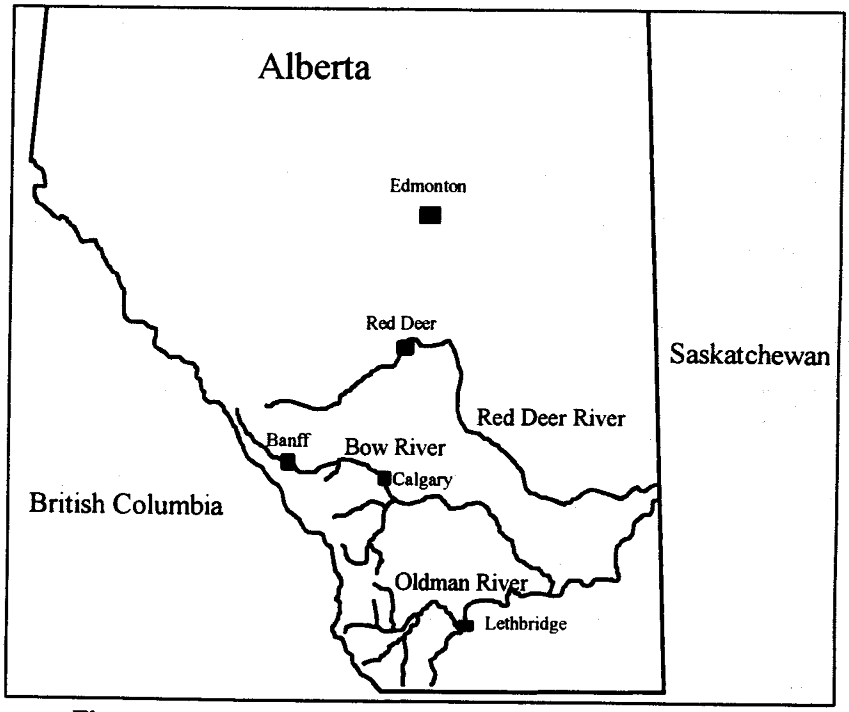

Figure 1. The South Saskatchewan River Basin in Alberta.

SELECTING A SINGLE RECOMMENDED FLOW

There are several approaches that can be used in selecting a single recommended flow from habitat quality indices such as Weighted Usable Area curves. A simple solution is to select the curve for one life stage and ignore the others or to select the "critical" life stage, based upon site specific criteria, for any given time period. The question that remains however, is how to select the recommended flow. One could conceivably select the peak of the curve as the recommended flow. Depending upon your point of view this could be thought of as either a

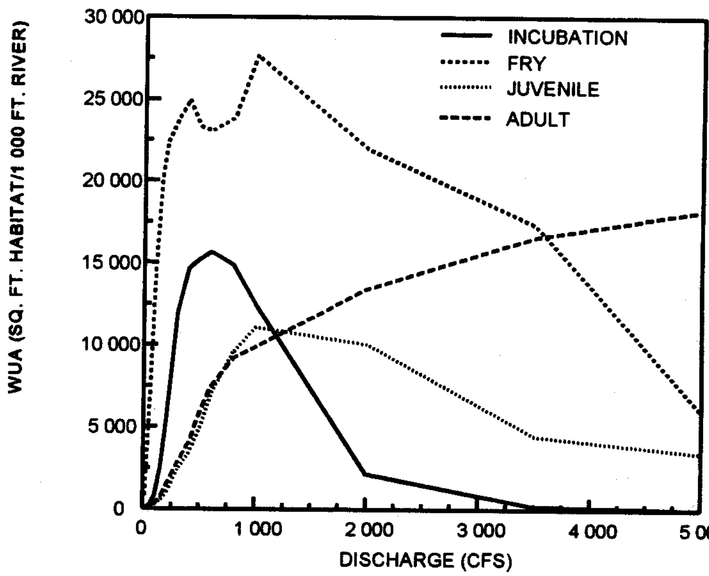

Figure 2. Mountain whitefish weighted useable area curves, Highwood River at Reach 2.

"minimum" or "preferred" flow value. This approach is oriented towards maximising WUA values and does not consider the stochastic habitat conditions that are experienced within the stream reach or the amount of water that is available. If there were only one life stage of one species that was being managed, this could be an appropriate approach. However, if there are multiple life stages and species with very different habitat requirements, using the optimal points to describe a minimum flow requirement does not seem to be practical. It could well be that the optimum habitat condition rarely exists even under natural flow conditions.

One method of dealing with the problem of having multiple life stages is to combine the various curves through a mathematical procedure (e.g., determine the arithmetic or geometric mean), then select the peak of the combined curve. An alternative to selecting the optimum habitat value would be to select a percentage of the maximum point. For example, one may consider the goal for the river is to select a percentage (i.e., 80% or 70%) of the maximum habitat that is obtainable.

Annear and Conder (1983) used a variation of this process to determine a recommended flow. They essentially established a range of acceptable flows based on the variability of the WUA curves. First, the curves were normalized then the mean of the normalized curves were plotted. The least significant difference (LSD) in WUA for the combined curves was used to determine the acceptable range of flows. The lower limit of the range was selected as the recommended flow.

Bovee (1982) suggested two techniques for choosing recommended stream flows for fish. These techniques are based on the concept of maximizing the habitat that is in least supply. The first consists of making a matrix of WUA for each life stage for flows in the 50% to 90% exceedence range, then selecting the flow that results in the maximum habitat for the life stage for which the least habitat is available. The second alternative is to weight the WUA curves by a factor which reflects the spacial requirements of the individual life stage. The coefficients by which the WUA curves are multiplied can be determined by a number of different ways.

Other investigators have used flow duration curves to determine the recommended flow. Geer (1980) and Hilgert (1982) used the criteria of the minimum recommended flow as being the lowest monthly discharge that provides the same habitat as the historic median flow for that month. The 50% exceedence flow is determined, and if this flow value is to the left of the peak of the WUA curve, then the median flow is the recommended flow. However, if the median flow is to the right of the peak on the WUA curve, then the flow to the left of the peak that produces the same amount of habitat becomes the recommended flow.

Sale *et al.* (1981) proposed using habitat duration curves instead of flow duration curves to select a recommended flow. They stipulated a habitat exceedence value, and therefore the corresponding flow value, as the minimum. The process used to determine this minimum flow is a two-step procedure. First, the habitat response curve (WUA) is used in conjunction with the flow record (historical or natural) to generate habitat values for each discrete flow value. The second step is to generate a habitat frequency plot, similar to a flow frequency plot. There are several ways to do this and the way suggested by Sale *et al.* (1981) is to use the formula:

(1) $$F(h) = 100 \times [r(h)/(n+1)]$$

where, $F(h)$ is the frequency at which habitat value h is equalled or exceeded,

r(h) is the rank of habitat value h, and
n is the number of events in the record.

The habitat exceedence plot and the WUA curve are then plotted on the same graph (Figure 3). The minimum flow is obtained by (1) selecting the habitat exceedence level to be protected (Sale *et al.* (1981) chose the 80% level), (2) dropping vertically to the point on the habitat frequency curve (based on a functional relationship between the habitat flow function and natural flow data) that corresponds to the desired exceedence level, (3) moving horizontally to the habitat response curve to the lowest discharge that produces the desired WUA value, and (4) moving vertically down to the discharge axis to find the stream flow to be used as the minimum flow recommendation.

The procedure described by Sale *et al.* (1981) is a significant improvement over the more deterministic approaches for determining minimum flow requirements. However, it is important to note that any single minimum flow value, whether it is generated by examining stochastic habitat events or by using a more simple deterministic approach, cannot duplicate all of the natural variability present in a natural state lotic ecosystem. If the recommended single flow value is all that will be provided in a regulated system, as was becoming evident through the water planning process in Alberta, then the single flow value will ultimately become the 100% exceedence value, not the 80% exceedence value. It is for these two reasons the AFMD developed a strategy to provide more than just one single flow value. First, it is widely acknowledged that one flow value cannot duplicate the natural variability in flows in which the aquatic organisms have evolved and secondly, if only one flow value was provided to the water planning process, then only this target would be met.

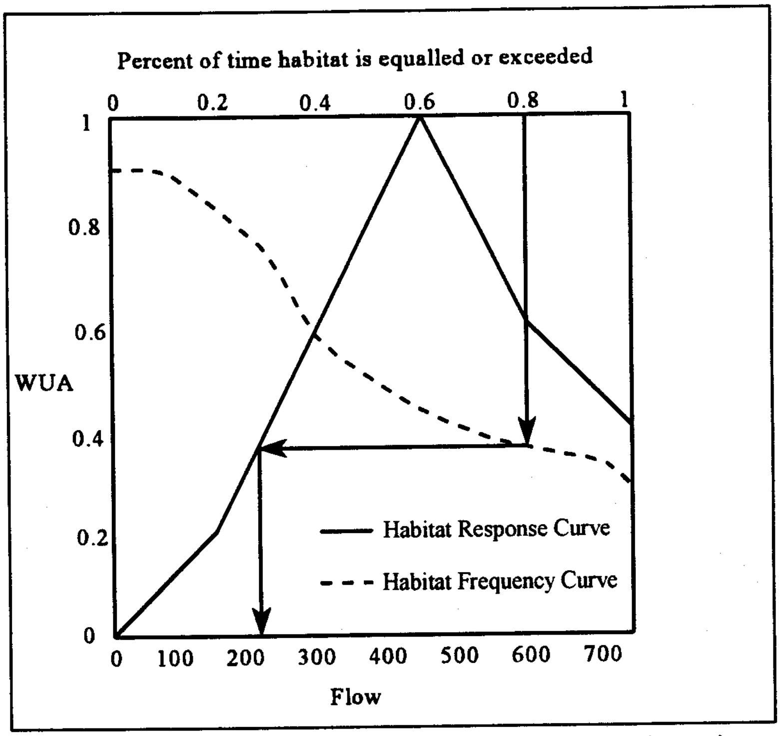

Figure 3. Calculating a minimum flow recommendation using habitat response and habitat frequency data. (After Sale *et al.*, 1981)

SELECTING A RANGE OF RECOMMENDED FLOWS

The AFMD determined that what was required was a hierarchy of flow targets for dry, normal and wet years. While this approach has been recognized as essential (Stalnaker, 1979; Sale *et al.*, 1981; Trihey and Stalnaker, 1985), there has been little effort in the instream flow needs community to incorporate this philosophy into setting flow regimes for fish. In an attempt to incorporate this philosophy, the AFMD used a combination of several of the previously discussed approaches and adapted the approach presented by Sale *et al.* (1981) to provide a range of flow targets which take into account the natural availability of water in the system.

The first step in selecting a range of recommended flows is to set the species periodicity chart. From this chart, weeks with similar groupings of life stages can be combined into a common biological period (Table 1). Geer (1983) referred to these as Biologically Significant Periods (BSPs). For each BSP it was necessary to combine all the WUA's for each life stage into one "composite fish". This was accomplished by using two approaches. The first approach was to simply select the life stage of greatest importance for the specific BSP. The selection of this life stage was done through consultation with the regional fisheries biologists and referring to any fisheries reports that were available for the river that was being studied. Once this was done, the curve was subjectively modified to ensure the incorporation of the requirements of all other life stages.

The other approach used was to weight the importance of each life stage and each species, using the same rationale described for the first approach. Then using a mathematical equation, one normalized curve for the "composite" fish was calculated (Geer, 1983). The two curves derived using the two approaches were then compared to see if there were any large or obvious differences. Any differences were reconciled through a consultative process with the regional fisheries biologists. This process was repeated for each BSP.

Table 1. Timing of biologically significant periods (BSPs) and species-life stages present.

BSP	Date	Julian Week	Rainbow Trout				Mountain Whitefish				Critical Life Stage
			F	J	A	S	F	J	A	S	
1	2 Apr - 29 Apr	14-17		X	X		X	X	X		RBTR Juvenile
2	30 Apr - 17 Jun	18-24		X		X	X	X	X		RBTR Spawning
3	18 Jun - 23 Sep	25-38	X	X	X		X	X	X		RBTR Fry
4	24 Sep - 28 Oct	39-43	X	X	X		X	X		X	MTWF Spawning

The next phase of the process was to take these composite fish WUA curves for each BSP and select a range of recommended flows for fish that would cover the entire range of "natural" flows that are available in the given reach of river. This process was carried out on a weekly time step. For lack of a better term, these recommended flow regimes were coined as Fish Rule Curves (FRC). The process is as follows:

Step 1

For each biologically significant time period, a minimum flow was selected that would provide a minimal or marginal habitat condition. As was the case in generating a composite habitat versus flow relationship, two approaches were used in selecting the minimum value. The first approach was to first examine the shape of the curve to determine if there were any obvious inflection points below which small incremental decreases in flow resulted in large decreases in habitat. In the absence of any obvious inflection points, the flow value that resulted in an 80 % reduction in habitat from the optimal habitat value (peak of the curve) for the critical life stage was identified. In all instances, only the part of the curve to the left of the peak is considered. Using the flow value which resulted in an 80 % reduction as a guideline, the effect of this flow on habitat availability for the other life stages was examined to ensure that at the very least, minimal habitat would be present for these life stages. Once again, selecting an 80 % habitat reduction value over another value, such as 75 % or 85 %, is not based on the results of extensive scientific experiments. Rather it is based on the assumption that a direct correlation exists between habitat and numbers of fish, and on the assumption that a reduction greater than this would be detrimental to the fish population. It should be noted, however, that the 80 % value was not strictly applied. Instead it was used as a guideline or starting point to estimate the habitat-limiting condition.

For some of the IFN studies, a second approach was carried out using the "composite" fish habitat-flow curve according to the procedure described by Sale *et al.* (1981). The minimum flow was obtained by selecting the habitat exceedence level to be protected, the 80% level (this is the amount of habitat that is equalled or exceeded 80 % of the time under natural flow conditions), taking this discrete habitat value and then determining the corresponding flow value from the WUA relationship. The minimum flows derived by the two approaches were compared and any large or obvious differences were reconciled through discussions with regional fisheries biologists. The selected habitat-limiting flow was then requested for a dry water year. Commonly, a dry year is defined as the 80 % exceedence flow. The habitat-limiting flow was set for flow exceedences in the 80 to 100 % flow exceedence range which covers the dry to extreme drought conditions (Figure 4).

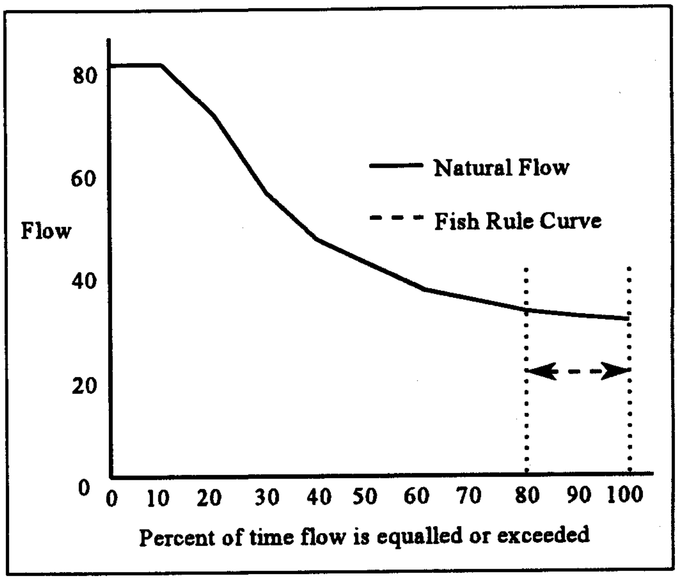

Figure 4. Setting the Fish Rule Curve for a dry year.

Step 2

The next step in setting the FRC was to select a flow that corresponded to an average habitat condition. As was done in Step 1, the first approach was to select a flow from the WUA curve for the chosen critical life stage that provided average habitat conditions. From this WUA curve, the flow value which resulted in a 50 % reduction in habitat from the optimal was used as a guideline. The effects of this flow on the habitat availability for the other life stages present were examined to ensure that close to average habitat conditions would be present for these other life stages.

Once again, for some of the IFN studies a second approach was carried out using the "composite" fish habitat-flow curve where the 50 % habitat exceedence level was selected. Taking this discrete 50 % habitat value, the corresponding flow value from the WUA relationship is determined. The average flows derived by the two approaches were compared and any large or obvious differences were reconciled through discussions with regional fisheries biologists. The selected average habitat flow was then requested for an average water year, that was defined as the 50 % exceedence flow (Figure 5).

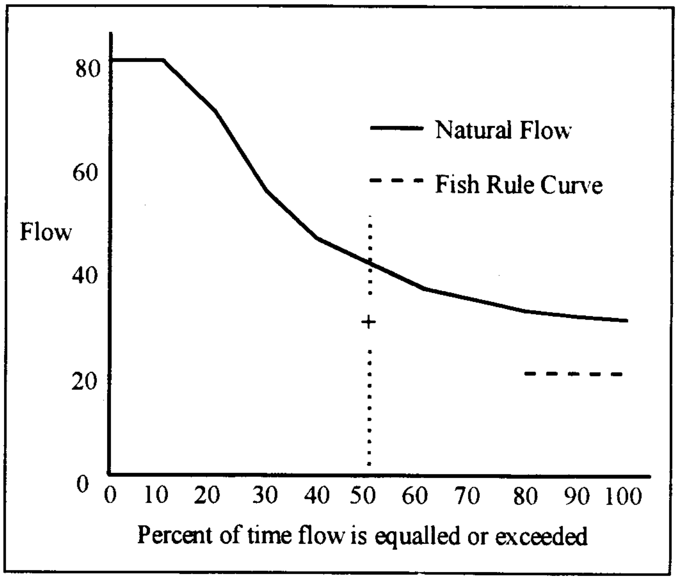

Figure 5. Setting the Fish Rule Curve for an average year.

Step 3

The final part of the FRC was set by selecting a flow that provided optimal habitat conditions. The first approach was to select the flow that provided the optimal habitat condition for the selected critical life stage. This is the peak of the WUA curve. The effects of this flow on the habitat availability for the other life stages present were then examined to ensure that close to optimal habitat conditions would be present for these other life stages.

Once again, for some of the IFN studies a second approach was carried out using the "composite" fish habitat-flow curve where the 20 % habitat exceedence level was selected. Taking this discrete 20 % habitat value, the corresponding flow value from the WUA relationship was determined. The optimal flows derived by the two approaches were compared and any large or obvious differences were reconciled through discussions with regional fisheries biologists. The selected optimal habitat flow was then requested for a wet water year. Hydrologists commonly define a wet year as one that corresponds to the 20 % exceedence flow range. This is a flow that is equalled or exceeded 20 % of the time or occurs two years out of 10 on average. The optimal flow value was then set for the 20 % to 0 % range of natural flows (Figure 6).

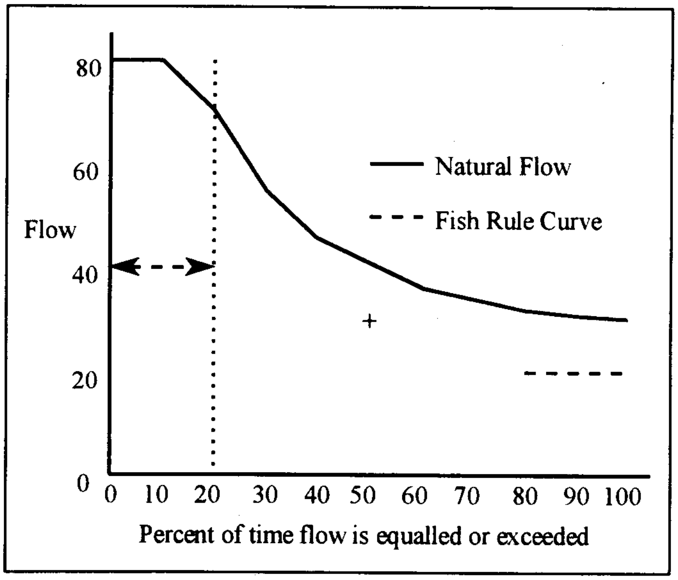

Figure 6. Setting the Fish Rule Curve for a wet year.

Step 4

The final FRC was defined by joining the wet, average, and dry-year points, and superimposing this operating rule on the naturalized weekly flow duration curve. In instances where the FRC value was higher than the natural flow value for any given exceedence range, the natural flow value became the FRC value. This ensured the requested flow values never exceeded the flow that was naturally available (Figure 7).

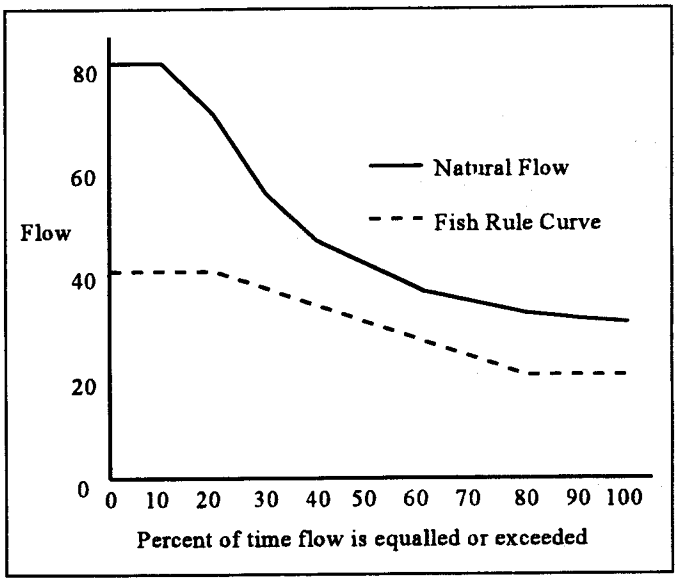

Figure 7. The final Fish Rule Curve.

DISCUSSION

Development of the Fish Rule Curve was based on physical habitat data, natural flow data, and professional judgement. Owing to the latter, the procedure may not be duplicated exactly by different investigators; i.e., it is not mathematically reproducible. However, the principles are reasonable and it is believed that other investigators would produce very similar fish rule curves if the same procedures were followed as outlined.

Many questions remain unanswered. First, the question of what habitat exceedence value should be used to determine the minimum flow is not resolved. Secondly, once the value is determined, how will it be incorporated into a water management plan. Specifying a flow without specifying a frequency could result in a minimum flow that is equalled or exceeded 100 % of the time. Stalnaker (1979) suggested a more realistic stream management strategy is to set a series of minimum flows for dry, normal, and wet years. The Fish Rule Curve employs this strategy, as well as considering the natural flow frequency as outlined by Sale *et al.* (1981), and specifies a frequency of exceedence.

It is important to recognize that the Fish Rule Curves represent targets for the amount of flow, and thus habitat, for particular time periods and specific reaches of river. The Fish Rule Curves do not represent minimum or preferred flows, rather they represent desired habitat conditions for the complete range of flows which may occur in the watercourse. Further, the intent of the Fish Rule Curve is that the flow requests are targets, against which the relative success of a particular operating plan can be evaluated. Habitats provided by a particular operating plan may be compared to those requested by the Fish Rule Curve. The differences between the two habitat values can then be compared and analysed.

In all instances when carrying out water quantity modelling runs, the Fish Rule Curves could not be met when existing consumptive licences were given first priority for the available water. However, the success of providing the Fish Rule Curves as targets was that flows much greater than minimums were achieved most of the time. As it turned out, the setting of the Fish Rule Curves represented the first step in the water planning process. It was apparent from the failure to meet the Fish Rule Curve that the environmental needs of the aquatic ecosystem and the consumptive demands in the basin could not be met. This precipitated the next step in the water planning process, namely the evaluation of alternative flow scenarios. At the outset, the AFMD was in a much better negotiating position since all model runs attempted, at least in part, to meet the Fish Rule Curve instead of a single minimum flow value. The method used by the AFMD to evaluate these flow scenarios was a standard habitat time series as described by Bovee (1982). For each scenario, there were three benchmarks against which the scenario could be evaluated; the historical (pre-project) flow regime, the natural flow regime and the recommended flow regime (Fish Rule Curve).

As with any technique used to prescribe flow regimes for fish, validation of criteria should be carried out. Many of the steps used in setting the Fish Rule Curves are based on subjective opinion. Any time subjectivity is part of the process, there is room for criticism. The setting of the Fish Rule Curve is no exception. In fact, a review of a recent IFN study carried out by the AFMD resulted in a strong criticism of picking a flow(s) from the WUA-flow function with the rationale being that it cannot be defended very well from a biological perspective (Ken Bovee, pers. com.). This is an acceptable criticism, depending on site specific circumstances, however, the intention of the Fish Rule Curve target was to move away from setting single minimum flows, and in Alberta this has been achieved.

The ultimate success of the Fish Rule Curve and water planning process in general, will only be known once an accepted water plan has been in place for a number of years and long term population monitoring indicates the fish respond as predicted by the habitat-flow model. Such a study is currently underway on the Oldman River where a water management plan has been in place since the Oldman River Dam reservoir filled and the dam became operational in 1993. Population monitoring began in the summer of 1995. Last year, there were reports of brown trout being caught by anglers at the confluence of the Oldman and St. Mary Rivers just upstream from the City of Lethbridge. Prior to the water management plan being in place, which was developed using Fish Rule Curves as input, this same reach of the Oldman River would, in drier years, cease to flow in late July and August.

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USE OF WETTED PERIMETER IN DEFINING MINIMUM ENVIRONMENTAL FLOWS

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ABSTRACT

In regulated rivers, the relation between wetted perimeter and discharge is often cited as an expedient technique for determining the minimum flow allowable for environmental purposes. The relation between wetted perimeter and discharge can be established by repeated field surveys at various discharge levels, or modelled from channel morphology survey data using a flow equation. The critical minimum discharge is supposed to correspond to the point where there is a break in the shape of the curve. Below this discharge, wetted perimeter declines rapidly. This critical point on the curve is almost universally, but incorrectly, termed an "inflection" point, and is usually determined subjectively by eye from a graph. The appearance of a break in the shape of the curve is very dependent on the relative scaling of the axes of the graph. This subjectivity can be overcome by defining the break in shape using mathematical techniques. The possibility of fitting a curve to the relation between wetted perimeter and discharge has already been firmly demonstrated in the geomorphic literature on "at-a-station" hydraulic geometry. The important break in the shape of the curve can be systematically defined by the point where the slope equals 1, or where the curvature is maximized. These techniques were applied to two headwater streams in Victoria, Australia. Armstrong and Starvation Creeks form part of the water supply network for Metropolitan Melbourne. Water is diverted from the weirs, but an environmental flow is passed to the stream below. Both streams had a lower diversity and abundance of sampled macroinvertebrates below the weirs compared with the unregulated streams above the weirs. The morphology of the channels was surveyed at ten cross-sections, at several flow levels, on both streams. The data were used to model the relation between wetted perimeter and discharge. Log functions adequately described the calibrated model data. The discharges corresponding to the critical point on the curves were the same or lower than the historical regulated minimum flows. Thus, in these cases, a regulated flow regime based simply on a minimum flow derived from the wetted perimeter discharge relation is inadequate for maintaining ecological integrity.

KEY-WORDS: Wetted Perimeter / Aquatic Habitat / Environmental Flow / Instream Flow / Inflection Point / Macroinvertebrate / Water Diversion / Weir / Velocity / Transect

INTRODUCTION

The relation between channel wetted perimeter and discharge is widely used to assist decision making when defining environmental (instream) flows in regulated rivers. This approach assumes that there is a relationship between wetted perimeter and habitat availability. While Annear and Conder (1984) considered this a logical assumption, the literature contains very few supporting field observations. As discharge decreases, riffles are the first locations to be exposed. It is known from "at-a-station" hydraulic geometry relations that as discharge decreases, stream velocity generally decreases at a faster rate (Leopold and Maddock, 1953). Riffles are generally more productive of invertebrate species than pools. Also, aquatic insect drift is a function of transport from riffles to pools, with a positive correlation between current velocity and the quantity of drifting insects (Stalnaker and Arnette, 1976). Pearson (1970) found that the production of macroinvertebrates was a function of the velocity of water through riffle areas, and the total amount of riffle area. It is reasonable then to assume that maintenance of aquatic macroinvertebrate habitat requires a minimum flow that covers a reasonable proportion of the bed with flowing water.

The wetted perimeter discharge relation is a basic tool in the "transect" approach to environmental flow evaluations (Gordon *et al.*, 1992: 428). One procedure is to derive the relation from channel cross-section surveys at several discharge levels. The transects are often located only at riffle sites, or at sites where fish passage is likely to be limited. Alternatively, the relation can be modelled using the channel morphology, and other data, using a flow equation such as the Manning equation (Gordon *et al.*, 1992: 171). Computer programs are available to perform these calculations (e.g. Grant *et al.*, 1992). A line is generally fitted through the surveyed or modelled points. The lowest breakpoint in the curve is taken to represent a critical discharge below which habitat conditions for aquatic organisms (usually fish or macroinvertebrates) rapidly become unfavourable. The breakpoint indicates where small decreases in flow result in increasingly greater decreases in wetted perimeter (Annear and Conder, 1984). The Instream Flow Incremental Methodology relies heavily on transect data for physical habitat assessment. While sophisticated hydraulic modelling techniques such as PHABSIM (Milhous *et al.*, 1989) are available, they require detailed hydraulic and morphological surveys, and knowledge of habitat preferences for the species of interest. For these reasons, the simpler approach based on examination of the wetted perimeter discharge relation is more appropriate in many locations.

The wetted perimeter discharge breakpoint has been used to define optimum or minimum flows for fish rearing (food production) in the U.S.A. (Collings, 1974; Cochnauer, 1976; Nelson, 1980) and Australia (Richardson, 1986). Stalnaker and Arnette (1976) reported that the breakpoints for some U.S.A. streams occurred at discharges corresponding to approximately 80% of the maximum available wetted perimeter. The Oregon Department of Fish and Wildlife recommend (among other criteria) that at least 50% of the maximum wetted perimeter be provided at riffles (Stalnaker and Arnette, 1976). Filipek *et al.* (1987) found that for Arkansas streams the breakpoint in the wetted perimeter discharge relation occurred at 50% of the mean flow. Below this discharge, riffle areas became exposed and unproductive, streambank cover for fish diminished, the water quality decreased, and fish overcrowding was possible. For Wyoming and Montana streams, Tennant (1976) found that the flow equivalent to 10% of the average flow provided about 50% of the maximum wetted perimeter, while flows greater than 30% of the average flow provided close to the maximum wetted perimeter.

Nelson (1980) found that wetted perimeter curves for streams in Montana had single, well-defined break points. The discharges at the breakpoints corresponded to the minimum flow levels required to maintain trout populations. Use of multiple transects resulted in less distinct breakpoints. Prewitt and Carlson (1977) reported that the wetted perimeter approach did not suit the unique fauna of the Upper Colorado River Basin.

The transect approach to defining environmental flows is widely used in Victoria, Australia. Tunbridge and Glenane (1988) defined an optimum flow as one which maintains >90% of the maximum available fish habitat, fish passage and wetted perimeter; a minimum environmental flow maintains >70% of the maximum available fish habitat, 10% of maximum fish passage, and 80% of maximum wetted perimeter; a survival environmental flow maintains >50% of the maximum available fish habitat, 10% of maximum fish passage, and 60% of maximum wetted perimeter.

Although the wetted perimeter approach is widely used in environmental flow evaluations, a systematic method for selecting the critical breakpoint on the curve has never been documented. The point is chosen solely on a subjective basis, and recommendations can vary between investigators (Annear and Conder, 1984). Also, complications can arise when no clearly defined breakpoint is found, or where multiple breakpoints occur. The only attempt to develop a systematic method of selecting the breakpoint was that of Annear and Conder (1984), who suggested a statistical technique to identify significant reductions in wetted perimeter from pre-defined reference flows.

This paper presents a method of characterising the wetted perimeter discharge relation. A systematic approach to defining the important break in the shape of the curve is given. The nature of the relation is examined for several different stream channel shapes. The suggested method of analysis was applied to two regulated headwater streams in Victoria, Australia. The minimum flows recommended by this method were compared with the unregulated and regulated flow regimes of the streams. The regulated flow regimes were assessed in terms of their ability to maintain populations of target macroinvertebrates.

CHARACTERISING THE WETTED PERIMETER DISCHARGE RELATION

The breakpoint in the wetted perimeter discharge relation is referred to in the literature almost universally as a point of "inflection" (Cochnauer, 1976; Stalnaker and Arnette, 1976; Prewitt and Carlson, 1977; Wesche and Rechard, 1980; Annear and Conder, 1984; Richardson, 1986; Filipek, 1987; O'Keefe *et al.*, 1989). This is an incorrect use of the term inflection. Analytical geometry texts define an inflection point as occurring where: a tangent at the point cuts through the curve; where the first derivative (slope) increases on one side of the point and decreases on the other; and where the second derivative at the point equals zero (Goodman, 1980: 200-201). None of these conditions occur at the point on wetted perimeter curves that is often referred to as an inflection point. What then is the nature of this breakpoint, and how can it be systematically defined?

Effect of Graph Scaling

The breakpoint on wetted perimeter discharge curves that is being sought is not an inflection point, but rather it is a point where the curvature is maximized, or where there is a marked change in the slope of the curve. It is not possible to reliably select these points by eye, as the appearance of the slope of the curve is strongly dependent on the relative scaling of the axes (Figure 1). Variability in the presentation of the graphs will cause unreliability in the selection of breakpoints.

Shape of the Wetted Perimeter Discharge Curve

The shape of the relation between wetted perimeter and discharge is a function of the geometry of the channel, and the manner in which discharge increases with depth. Channels have geometries that range from roughly triangular to roughly rectangular, although the latter shape is more prevalent. For rectangular and trapezoidal channels, as the depth of water in the channel increases from zero, there is an initial rapid increase

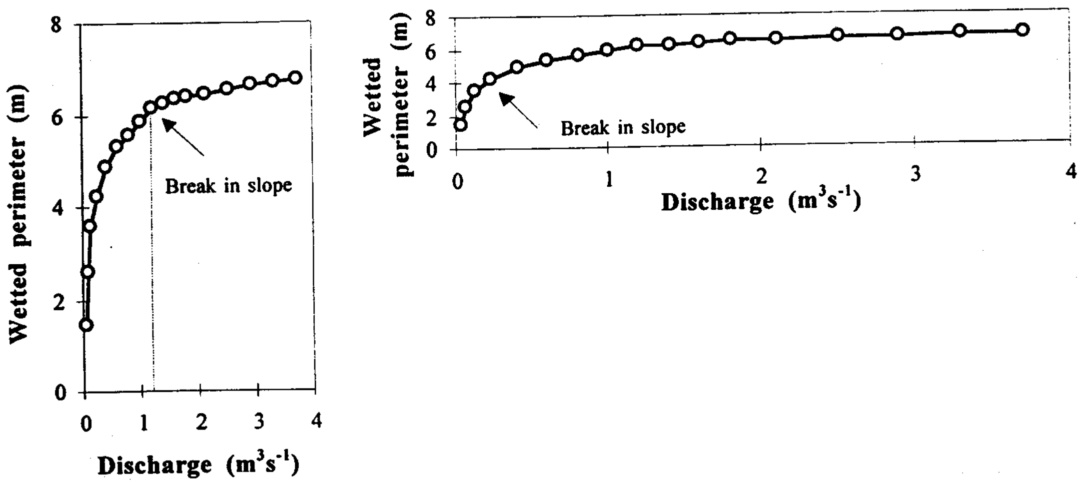

Figure 1: Hypothetical wetted perimeter discharge relation plotted on two differently scaled sets of axes. Different breakpoints are apparent, even though the same data are plotted on both graphs.

in wetted perimeter. As the bed becomes inundated, continued depth increases produce only relatively small increases in wetted perimeter. The shape of this relation is accentuated by the way discharge increases with water depth. At low flows the velocity is low; as depth increases, flow velocity increases, so that discharge increases at a faster rate than depth. The nature of this relation is described by the Manning equation:

(1) $$Q = \frac{1}{n} A R^{2/3} S^{1/2}.$$

where Q = discharge (m^3s^{-1}), n = Mannings roughness coefficient, A = cross-sectional area of the flow (m^2), R = hydraulic radius (m), and S = water surface slope.

For triangular channel geometries where the wetted perimeter increases proportionately with depth, from Equation 1 it can be seen that the wetted perimeter discharge relation is a power function. Such curves do not display a sharp break in slope. The power function relation does not apply to channel geometries that produce more rapid changes in wetted perimeter at lower discharges. Near-rectangular geometries produce a logarithmic relation between wetted perimeter and discharge. Figure 2 depicts four different channel geometries using hypothetical data. The cross-section and discharge plots are dimensionless, with each value being expressed as a proportion of the maximum value. This enables direct comparison of the shapes and slopes of the wetted perimeter discharge curves for the different channel geometries. It is apparent that triangular geometries produce a power relation between wetted perimeter (P_w) and discharge (Q) (Equation 2), while rectangular geometries produce a logarithmic relation (Equation 3).

(2) $$P_w = aQ^b.$$

(3) $$P_w = a\mathrm{Ln}(Q) + b.$$

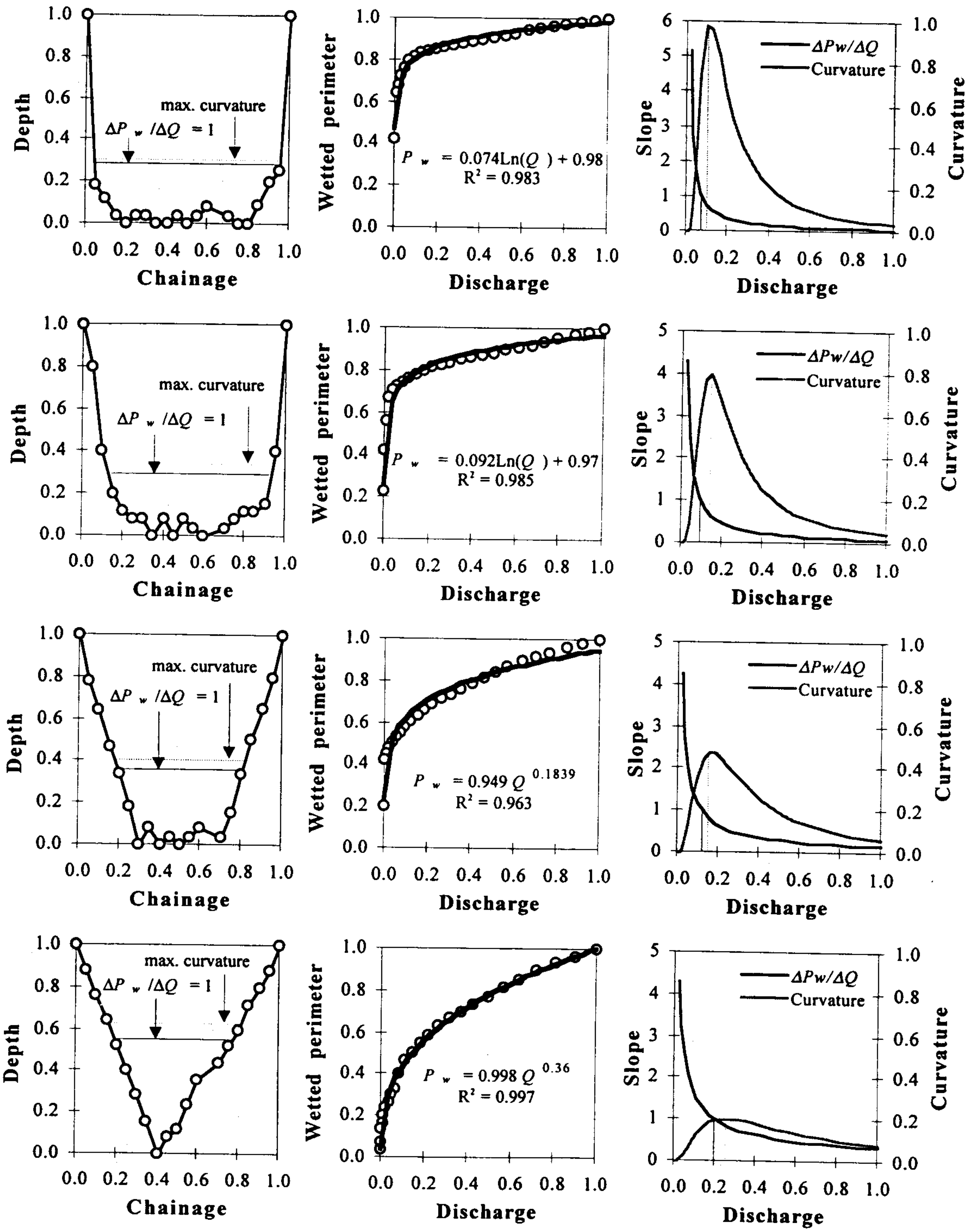

Figure 2: Cross-section, wetted perimeter discharge relation, and curvature of wetted perimeter relation for four channel geometries using hypothetical data.

It is relevant to note that Leopold and Maddock (1953) found that the general relation between channel width (closely related to wetted perimeter) and discharge (at-a-station) for U.S.A. streams was a power function of the form of Equation 2, with an average exponent of 0.26.

Defining the breakpoint

Assuming that changes in discharge and wetted perimeter should be equally weighted, then one method of selecting the breakpoint in the wetted perimeter discharge relation is to select the point on the curve where the slope $\frac{\Delta P_w}{\Delta Q}$ (second derivative $\frac{dy}{dx}$) equals 1 . This applies only when the discharge and wetted perimeter axes have been scaled to cover the same range (Figure 2). The equations for slope are given below for the power function (Equation 4) and logarithmic function (Equation 5). For the logarithmic function, the slope of unity occurs simply at $Q = a$ (Figure 2).

(4) $$\frac{dy}{dx} = baQ^{b-1}.$$

(5) $$\frac{dy}{dx} = \frac{a}{Q}.$$

In cases where it is not possible to fit a logarithmic or power function to the data, the slope of the relationship can be calculated for each plotted (surveyed or modelled) point. The breakpoint is taken as the point where the slope is closest to 1.

A second systematic method of selecting the breakpoint in the curve is to define the point of maximum curvature. The curvature (κ) is the rate at which a curve turns (Equation 6); it is a function of the angle that the tangent to the curve makes with the x-axis, and the arc length (Goodman, 1980: 610).

(6) $$\kappa = \frac{\frac{d^2y}{dx^2}}{\left[1+\left(\frac{dy}{dx}\right)^2\right]^{3/2}}.$$

The equations for absolute curvature are given below for the power function (Equation 7) and logarithmic function (Equation 8). Maximum curvature is best determined by plotting the functions given in Equations 7 or 8 as appropriate (Figure 2).

(7) $$|\kappa| = \frac{\left|\left(ab^2 - ba\right)Q^{b-2}\right|}{\left[1+\left(baQ^{b-1}\right)^2\right]^{3/2}}.$$

(8) $$|\kappa| = \frac{\left|\frac{-a}{Q^2}\right|}{\left[1+\left(\frac{a}{Q}\right)^2\right]^{3/2}}.$$

The slope = 1 and maximum curvature methods give similar breakpoints (Figure 2). The slope method of selecting the breakpoint is intuitively the most appropriate, and the simplest to apply. This approach assumes that changes in wetted perimeter and discharge are of equal importance. So, below the point where slope equals 1, wetted perimeter decreases at a lower rate with respect to decreasing in discharge; above the point where slope = 1, wetted perimeter increases at a lower rate with respect to increasing discharge. The selected breakpoints for the rectangular channel geometries are close to the bases of the banks (Figure 2). This is appropriate as a minimum discharge for maintaining flowing water over most of the bed. For the trapezoidal and triangular channel geometries (Figure 2), the breakpoint is determined more by the way discharge increases with depth (non-linear), than the way wetted perimeter changes with depth (linear).

CASE STUDY APPLICATION OF THE WETTED PERIMETER APPROACH TO DEFINING MINIMUM REGULATED FLOWS

Background

The water harvesting system that supplies water to metropolitan Melbourne consists of several large storage reservoirs, supplemented by a number of small diversion weirs, 1-5 m in height, located on headwater streams. The weirs divert water, via aqueducts or pipelines, to the larger water supply reservoirs. Typically, about half of the annual runoff is harvested from these small regulated streams (long-term mean daily flows range from 0.02-0.90 m^3s^{-1}). Although not legally obliged to do so, the managing authority (Melbourne Water) maintain minimum downstream releases from most of the weirs. The release discharges are nominal, and are not based on known environmental requirements.

A survey of the physical habitat in the vicinity of ten of the weirs found that the most obvious change in the downstream physical environment was a reduction in the area of channel having a mean velocity >0.4 m/s during times when water was being harvested (Rhodes, 1994). It was speculated that the flow reduction downstream of weirs would most likely adversely impact those aquatic species that prefer this type of habitat. Surveying three of the weirs examined by Rhodes (1994), located on Armstrong, Badger and Starvation Creeks, Gaynor *et al.* (1995) found that the abundance and diversity of target taxa (those preferring fast-water velocities) were significantly ($P < 0.05$) reduced below the weirs. The magnitudes of the reductions were not a simple function of the percentage of water harvested. There was no significant difference in the diversity or abundance of target macroinvertebrates at two control weir sites (weir present, but no water harvested).

Transect surveys were conducted downstream of the weirs at Starvation and Armstrong Creeks to determine the nature of the relationship between discharge and wetted perimeter. The objective was to determine the appropriateness of the current minimum environmental flows for providing adequate habitat for macroinvertebrates (i.e. a reasonable percentage of the channel bed covered by flowing water).

Channel Surveys

Rhodes (1994) measured the channel morphology and mean velocity across transects at five different discharge levels on both Starvation Creek and Armstrong Creek. Five transects spaced 20 m apart were measured at each site. In this study, the surveys were repeated at these sites at three different discharge levels, using ten transects spaced 20 m apart at each site. In both surveys, approximately 10 measurements were made across each transect. The channels ranged in width from 2-7 m.

The field data were used to model the change in wetted perimeter with discharge at fifteen discharge levels on Starvation Creek and fourteen discharge levels on Armstrong Creek. A value of mean wetted perimeter was calculated from the ten transects for each modelled discharge level. The modelled relations were calibrated to fit the surveyed values. The mean velocity and channel morphology data were used to define the relation between the area of channel (m^2) containing flowing water (> 0.01 ms^{-1}) A_f, and discharge Q.

Flow Analysis

Daily unregulated flow (i.e. upstream of the weirs) records were available for Starvation Creek from 1 Jan 1969 to 31 Dec 1995 and for Armstrong Creek from 1 March 1970 to 31 Dec 1995. These records were analysed to determine the mean daily flow, flow exceeded 95% of the time, minimum recorded flow, and flow exceeded 50% of the time (median flow). The flow indices were calculated for the four driest months (Dec-March), as these are the months when minimum flows are likely to be an issue.

In recent times, the minimum flows released from these weirs have been higher than the specified minimum environmental flows. The flows from the weirs are not regularly accurately gauged, but the minimum flow releases were estimated on the basis of gauging the streams on several occasions, and information provided by the weir keepers.

Results

The mean wetted perimeter discharge relation for Starvation Creek was of a logarithmic form (Figure 3); the breakpoint in the relation was calculated using the slope and curvature methods (Figure 3, Table 1). In contrast, the relation between mean wetted perimeter and discharge for Armstrong Creek was linear. This resulted from averaging data from a wide range of cross-sectional shapes surveyed at this site.

The relation between the area of channel containing flowing water and discharge was of a logarithmic form for both Starvation Creek and Armstrong Creek (Figure 4). The breakpoints in the relations were calculated using the slope and curvature methods (Figure 4, Table 1). The minimum discharges corresponding to the breakpoints calculated from the area of flowing water data were slightly higher than the breakpoints calculated from the wetted perimeter data.

Analysis of flow records revealed that the minimum unregulated discharges recorded in these streams occurred during a severe drought in the summer of 1982/83. The drought had a more severe impact on flows in Armstrong Creek (Table 1). Periodically, up to 40% of the channel bed did not contain flowing water. Despite the drought, at Starvation Creek, under an unregulated flow regime, not less than 80% of the channel bed would have been inundated with flowing water at all times. Droughts of this severity occur rarely; the flow exceeded 95% of the time is a better index of low flow. In Starvation Creek, at least 88% of the maximum available area of wetted channel was provided by the flow exceeded 95% of the time. In Armstrong Creek, at least 74% of the maximum available area of wetted channel was provided by the flow exceeded 95% of the time (Table 1).

The minimum flows passed from the weirs during recent times were similar to the minimum flows determined from analysis of the wetted perimeter and flowing wetted area relations (Table 1). The current nominal minimum regulated flows are much lower than the summer low flows normally experienced in these streams. At Starvation Creek, the nominal minimum flow is approximately one-third of the lowest unregulated flow recorded in the stream.

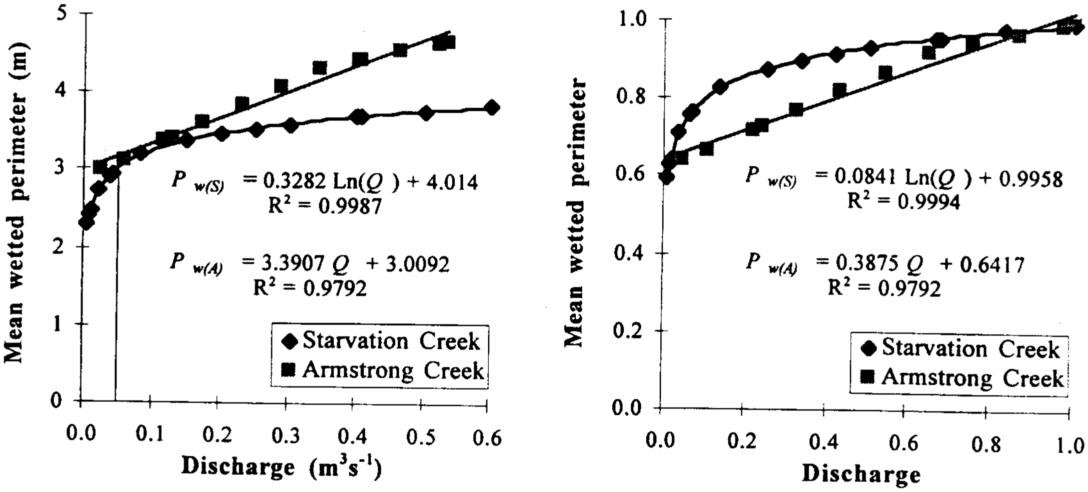

Figure 3: Wetted perimeter discharge relations for 200 m long reaches of Starvation Creek and Armstrong Creek expressed as raw values, and re-scaled dimensionless values. Vertical line is drawn at breakpoint where $\Delta P_w/\Delta Q = 1$.

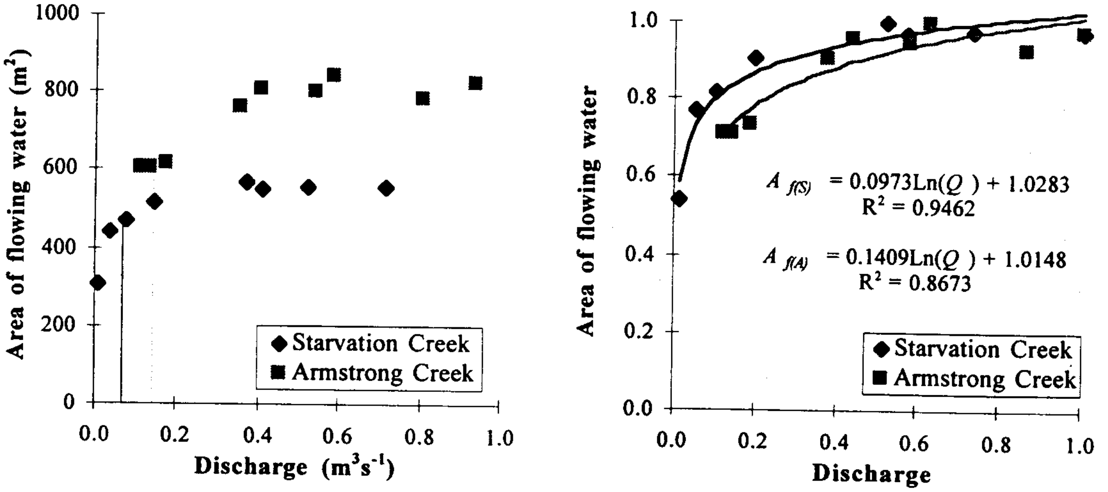

Figure 4: Area of flowing water discharge relations for 200 m long reaches of Starvation Creek and Armstrong Creek expressed as raw values, and re-scaled dimensionless values. Vertical lines are drawn at breakpoint where $\Delta P_w/\Delta Q = 1$.

Discussion of Case Study

The channel surveys conducted at Starvation and Armstrong Creeks were not restricted to riffle sites. Starvation Creek had a similar morphology along the surveyed 200 m reach. Armstrong Creek had a more variable channel morphology along the surveyed reach. This produced a linear relation between mean wetted perimeter and discharge. The implication of this relation (Figure 3) is that 60% of the maximum available

Table 1: Summary of analysis of flow records at Armstrong Creek and Starvation Creek. Included are the minimum flows suggested by analysis of the wetted perimeter and flowing wetted area versus discharge relations.

	Starvation Creek	Armstrong Creek
Wetted perimeter		
$\Delta P_w/\Delta Q = 1$	0.05 m^3s^{-1} (79%)	-
Maximum curvature	0.05 m^3s^{-1} (79%)	-
Flowing wetted area		
$\Delta A_f/\Delta Q = 1$	0 07 m^3s^{-1} (80%)	0.13 m^3s^{-1} (73%)
Maximum curvature	0.10 m^3s^{-1} (83%)	0.19 m^3s^{-1} (79%)
Historical mean daily unregulated flow		
	0.47 m^3s^{-1}	0.85 m^3s^{-1}
Historical 50% exceedance unregulated flow (% of max. flowing wetted area)		
Dec	0.44 m^3s^{-1} (98%)	0.56 m^3s^{-1} (94%)
Jan	0.34 m^3s^{-1} (96%)	0.36 m^3s^{-1} (88%)
Feb	0.28 m^3s^{-1} (94%)	0.27 m^3s^{-1} (84%)
Mar	0.25 m^3s^{-1} (93%)	0.23 m^3s^{-1} (82%)
Historical 95% exceedance unregulated flow (% of max. flowing wetted area)		
Dec	0.20 m^3s^{-1} (90%)	0.21 m^3s^{-1} (81%)
Jan	0.17 m^3s^{-1} (89%)	0.15 m^3s^{-1} (76%)
Feb	0.16 m^3s^{-1} (88%)	0.13 m^3s^{-1} (74%)
Mar	0.16 m^3s^{-1} (88%)	0.16 m^3s^{-1} (77%)
Historical minimum unregulated flow (max. duration in days)		
Dec	0.12 m^3s^{-1} (1 d)	0.10 (1 d)
Jan	0.14 m^3s^{-1} (3 d)	0.10 (1 d)
Feb	0.07 m^3s^{-1} (3 d)	0.05 (1 d; 24 d < 0.09 m^3s^{-1})
Mar	0.08 m^3s^{-1} (4 d)	0.05 (1 d; 21 d < 0.08 m^3s^{-1})
Estimated historical minimum regulated flow (% of max. flowing wetted area)		
	0.10 m^3s^{-1} (84%)	0.14 m^3s^{-1} (75%)
Nominal minimum regulated flow (% of max. flowing wetted area)		
	0.023 m^3s^{-1} (70%)	0.059 m^3s^{-1} (62%)

channel bed perimeter is available when discharge is zero. This is realistic, as pools would inundate large sections of the channel at zero discharge. Many unregulated streams in southeastern Australia occasionally cease to flow, yet they support healthy populations of native fish. River blackfish (*Gadopsis marmoratus*) have a limited home range and spend much of their life cycle in pools, so their abundance is unlikely to be greatly affected by naturally occurring periods of low or cease to flow conditions (Davies, 1989). In contrast, macroinvertebrate production is probably reduced by very low flows, when large areas of riffle habitat are exposed, and flow velocities are very low. Unlike the wetted perimeter model, the model of habitat availability based on the area of flowing water was non-linear (Fig 4). It is a more appropriate model for macroinvertebrate habitat.

The maximum curvature technique of selecting the breakpoints on the wetted perimeter (area of flowing water) discharge curves gave slightly higher recommendations than the values based on the slope of 1. For Starvation Creek, the recommended minimum flows were lower than the historical unregulated 95%

exceedance flow, and similar to the environmental flow that has been released in recent years. In Armstrong Creek, the recommended minimum flows were similar to the 95% exceedance flow and the environmental flow that has been released in recent years. The recommended minimum flows, and the minimum flows actually released from the weirs, provide 73%-84% of the maximum available habitat. Although this is within the range suggested in the literature as being adequate for environmental protection, the abundance and diversity of macroinvertebrates that prefer fast-flow environments have been severely reduced at these sites (Gaynor *et al.*, 1995). Application of the nominal minimum flows at these sites is likely to cause a reduction in habitat availability to levels substantially lower than would naturally be experienced, even during severe drought conditions.

One limitation of applying the wetted perimeter approach to the problem of environmental flow determination is that it recommends only a minimum flow. In reality, a flow regime should be specified. For example, in the case of the weirs at Starvation and Armstrong Creeks, the minimum flows suggested by the wetted perimeter method could be applied in the drier months, and only during naturally dry periods. During these times the main water supply storage reservoirs are low, and the non-stream demand for water is at its highest. During wet periods, provided the main storage reservoirs were reasonably full, more generous releases could be allocated for environmental purposes. It should be possible to develop a simple relation between the percentage of maximum storage capacity available and the amount of water available for release from the weirs for environmental purposes. The allocated flow could be reviewed and adjusted on a weekly or monthly basis.

CONCLUSION

The wetted perimeter method is suitable as a tool for providing recommendations on minimum environmental flows in regulated streams. However, it suffers some limitations, and it should only be used in conjunction with other techniques that together produce a recommended environmental flow regime. This paper provides a method of overcoming the major problem of subjective interpretation of the breakpoint in the wetted perimeter discharge curve. It should be possible to fit a curve (logarithmic or power function) to most wetted perimeter discharge data. The breakpoint of these relations can then be determined mathematically by calculating the point of maximum curvature or the point where the slope is equal to 1. Of these two approaches, the slope method is simpler to apply, and probably has more physical meaning.

The existence of a sharp breakpoint on wetted perimeter discharge curves is largely a function of the channel geometry; rectangular cross-sections produce a more defined breakpoint than do triangular cross-sections. On rectangular cross-sections the breakpoint defined by the suggested curvature or slope technique corresponds to a discharge level that inundates the channel bed with flowing water. The approach appears suited to suggesting minimum environmental flows in rectangular-shaped channels. Application of the technique to two regulated headwater streams in south-eastern Australia produced minimum flow recommendations that were similar to the historical minimum flows or flows exceeded 95% of the time. However, biological surveys revealed that these flows, when applied as a minimum for extended periods of time, do not maintain the macroinvertebrate community in its unregulated state.

For the headwater streams investigated here, the wetted perimeter approach did not suggest optimum environmental flows (as suggested in the literature), nor did it suggest a flow level that would maintain the macroinvertebrate community if it was applied for long period of time. The flows recommended by the wetted perimeter approach should be viewed as minimum flows to be applied only during dry periods when the flow in the stream would naturally have been low.

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LONG-TERM CHANGES IN FLOOD PROTECTION LEVEL AND FISH HABITAT QUALITY IN A MOUNTAINOUS STREAM AFTER A MAJOR FLOOD-DISASTER

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ABSTRACT

Long-term changes in a mountain stream were investigated, analyzed, and evaluated from a view point of both flood protection and fish habitat conservation. This study focuses on the upper reaches of the Tenryu-gawa, where a heavy rainfall in 1961 caused severe damage such as landslides, debris flows, and dike breaks followed by the typical development of flood protection works. Firstly, the improvement of the safety level against flood after the disaster was evaluated with the use of nonuniform flow computations. Then, the fish habitat quality, especially the effect on over-wintering caused by the long-term flood protection works was also evaluated using field data. The results show: 1)The safety level against flood has certainly improved in comparison with the condition before the 1961 disaster, even though some risks still exist. 2)Maximum depth/width ratio as a habitat variable index has increased in many places. 3)There is no clear evidence indicating a decrease of fish species due to this long-term change in the stream. 4)Even though the fish seem to prefer riprap-like micro habitat for over-wintering, historically steady pools in the main stream are much preferred by mid- to large- sized fish as an over-wintering place.

KEY-WORDS: Channel geometry/Nonuniform flow computation/maximum depth to width ratio/over-wintering

PURPOSE

In Japan, an unusual heavy rainfall or a major flood disaster often triggers the rapid construction of flood control facilities in the damaged stream basin such as erosion control works, debris barriers, check dams, large dams, and channel consolidation works. As a result, long-term and extensive changes, more rapid than the natural process, affect sediment discharge conditions and channel characteristics of the basin.

The changes are, however, not always or not sufficiently considered in the usual master plan for flood control. Although such changes may result in heavy damage to fish habitat such as the elimination of rapids and pools, it is only recently that the conservation of fish habitat has been seriously considered in stream modification works. In this study, such long-term and extensive changes in the upper Tenryu-gawa were investigated, analyzed, and evaluated from a view point of both flood protection and fish habitat conservation. A heavy rainfall in 1961 caused severe damage such as landslides, debris flows and dike breaks to this stream followed by the typical development of flood protection works.

In particular, this paper concentrates on:

1)How much did the flood protection works decrease the water surface level at high flows ?

2)With assumption that the fish habitat value for mid- to large- sized fishes is roughly proportional to the maximum water depth/width ratio at normal water stages, did the flood protection works cause the ratio to rise or fall ?

3)What kind of areas do mid- to large- sized fishes prefer as their over-wintering place in the regulated stream?

STUDY SITE

Location

The study site is situated in the upstream reach of a 22km length on the mainstream of the Tenryu-gawa River (C.A.=5,090 km^2) shown in Fig.1. The reach has always been a significant socio-economic and flood potential element in the region, because it is located downstream of almost all major tributaries in the upper area of the Tenryu-gawa basin which enter the mainstream. Major cities/towns in this region are concentrated beside this reach. Downstream of the study reach, a long deep canyon named Tenryu-kyo and the Sakuma-dam (height: 155.5 m) are located, so that all fishes are landlocked in the upper reaches.

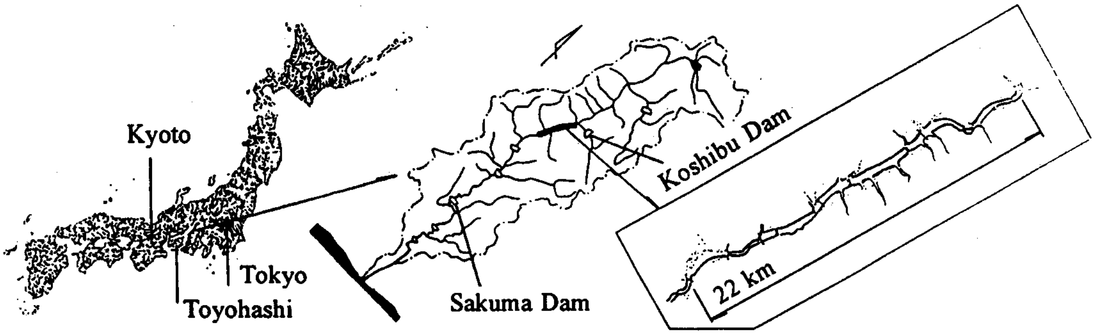

Figure 1: Location of study reach

Historic Flood Disaster in 1961

A heavy rainfall (maximum daily rainfall: 325 mm, total rainfall in a week: 565 mm) deluged this area on July 23-30 in 1961 (Kutsumi et al. 1995). The heavy rainfall caused the historic flood with 3,600 m^3/s at peak flow downstream of the Sakuma-dam and caused severe damage such as landslides, debris flows, and dike breaks in many places. One hundred and thirty-six lives were lost.

In the study reach, dikes and revetments were destroyed or damaged in many places and some bridges were lost (Fig.2(a)).

Flood Protection Works After the Historic Disaster

The historic disaster triggered the rapid progress of construction of various flood control facilities in the upper Tenryu-gawa basin, such as the Koshibu (flood control) dam (height: 105.0 m, completed in 1969) and many *sabo* (erosion and sediment runoff control) dams or debris barriers.

Also in the study reach, typical flood protection works such as levee construction, bank stabilization, channel widening and deepening were carried out along with the re-construction of bridges (Fig.2(b)).

The beneficial effect of these works can be seen in the 1983 flood event, i.e., the flood caused only slight damage in spite of a peak flow of 3,762 m^3/s (greater than peak flow in the 1961 flood).

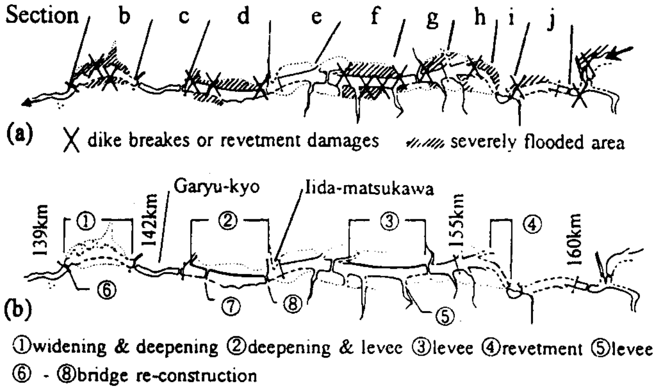

Figure 2: (a)Major damages by the 1961 flood in the study reach
(b)Major flood protection works after the 1961 disaster

Channel Characteristics of the Study Site

The study reach at the present time can be divided into the following sections (Fig.2).

a)upstream-end of Tenryu-kyo canyon: boundary of the sub- and super- critical flows during high river discharges

b)compound cross-section channel: normally narrow stream but wide at high flows exceeding 2,000 m^3/s approximately

c)small canyon named Garyu-kyo: formed by natural rocks; a scenic spot in this area

d)old open-levee section: the right-bank levee was partially opened to the flood plain until 1992

e)straight channel section: this was the only section where no occurred in the 1961 disaster; one of the major tributaries, the Iida-matsukawa, joins at the downstream-end

f)straight channel with open-levee: the right-bank levee is still opened at the downstream-end
g)transitional section between section h) and f)
h)bending and gradually widening channel section
i)narrow and sinuous section: very short canyon formed by rocks
j)upstream-end of levee reach (and the study reach)
The average slope is approximately 1/200 in the upstream of section d) as shown in Fig. 3.

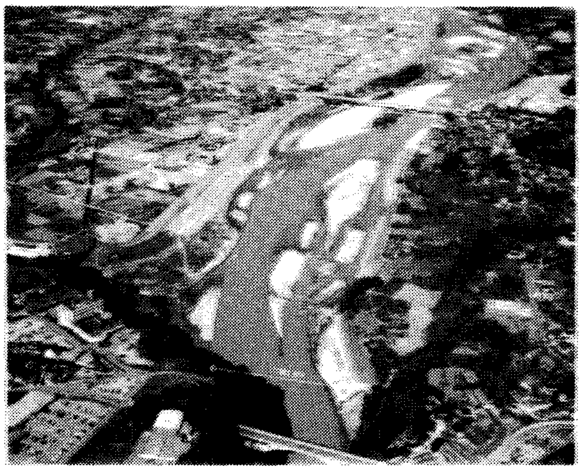
Photo 1: Section d

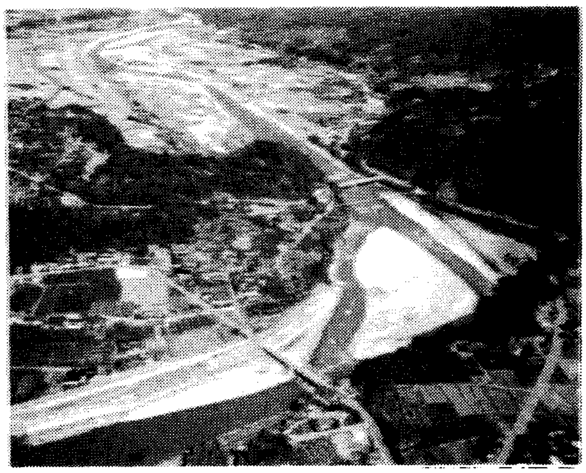
Photo 2: Section h-j

Fish Release by Local Fisheries Cooperatives

A lot of Ayu *Plecoglossus altivelis* and some common carp *Cyprinus carpio*, cruician carp *Carassius* spp., Japanese dace *Leuciscus hakonensis*, pale chub *Zacco platypus*, Japanese sculpin *Cottus hilgendorfi*, Japanese eel *Anguilla japonica*, Far Asian pond loach *Cobitis anguillicaudatus*, red-spot masu trout *Oncorhynchus masou macrostomus*, etc. are released in the stream every year by the local fisheries cooperatives, which are obligated to produce and stock fish in return for their fishing rights. Therefore, some diadromous species are living in the study reach, in spite of the fact that fish migration in this stream has been cut off from the ocean.

METHOD

Nonuniform Flow Calculation

Cross–Section Data, Equation and Flow Rate Used

In order to estimate the various changes in the macroscopic stream conditions after the 1961 disaster, one dimensional nonuniform flow calculations based on the following equation were carried out.

(1) $$i\Delta x - \Delta H = \frac{Q^2}{2g}\left(\frac{1}{A_1^{2}} - \frac{1}{A_2^{2}}\right) + \frac{n^2Q^2}{2}\left(\frac{1}{R_1^{4/3}A_1^{2}} + \frac{1}{R_2^{4/3}A_2^{2}}\right)\Delta x$$

where i=bed slope, Δx=distance between two computational cross-sections, ΔH=head loss, Q=water discharge, g=acceleration due to gravity, A=cross-sectional flow area, n=Manning's roughness coefficient, R=hydraulic radius, and suffix 1, 2 denote computational two cross-sections.

In the calculations, the cross-section data surveyed at 200 meter intervals in 1960, 1964, 1982, 1983, and 1988 were used. Flow rate of 3,762 m^3/s and 50 m^3/s were given as a high flow and a normal water stage respectively. The boundary conditions of the downstream-end are the observed water surface level at the peak of the 1983 flood for the high flow calculation and the uniform flow condition for the normal flow calculation.

Roughness Coefficient and Special Treatment

Manning's roughness coefficients in each section were roughly estimated by trial and error by comparing the calculated water surface and the observed high water marks after the 1983 flood.
In any cross-section, no dead zone was assumed and imaginary vertical walls were placed at both ends of the cross-section diagram which includes a sufficient width range of the river.
In the case of adjoining super-critical flow segments in the downstream-to-upstream progressing calculations, we assumed the water depth to be the critical depth plus 1 cm and continued the calculation to the next section.
Figure 3 shows (a)Manning's n used in the calculation, (b)calculated and observed water surface profiles showing the lowest riverbed profile and the location of super-critical flow, (c)water surface width at high and normal water stages, in which the thalweg is drawn as a straight line and the width is divided to the right and left side from the thalweg line, and (d)names of each section mentioned above.

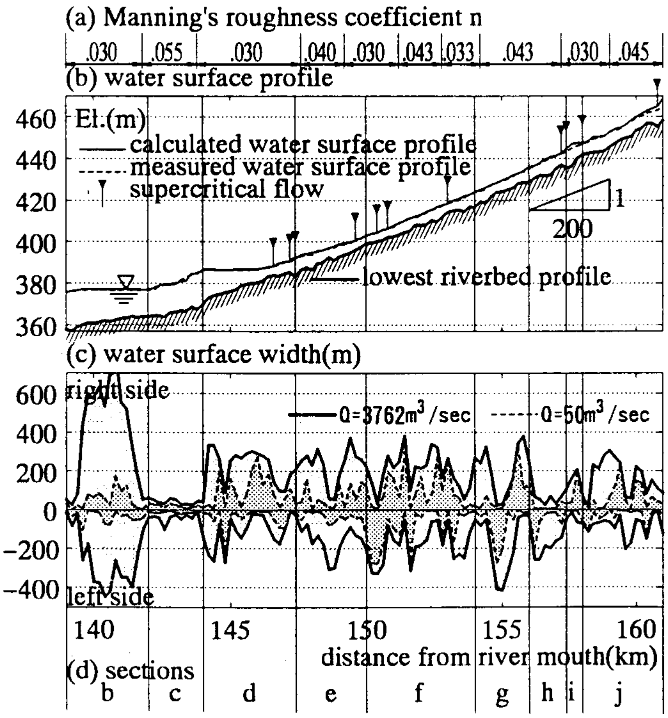

Figure 3
Manning's n, water surface profile, and water surface width at high flow

Fish Survey

Literature Survey

There are few historic and scientific data for fish in this area. Almost all literature mentions only the species living or confirmed by the local fishermen.
One of the rare exceptions is the report of the Stream Fish Census in the Tenryu-gawa 1992 (Cyubu Kensetsu-Kyokai 1992). In the census, fish surveys using fish nets were carried out at the three stations shown in Table 1.

Table 1: Outline of the surveyed stations in Fish Census 1992

St. No.	Location (km)	Survey Period	Water Depth(m)	Water Temp.(℃)	Bed Material Diameter(mm)	Riparian Condition
A	144.0-144.9	Aug.26	0.3-0.6	24.0	100-300	gravel bar
		Oct.20	0.4-0.7	14.3	100-300	gravel bar
B	150.6-151.4	Aug.26	0.4-1.5	24.1	sand-300	gravel bar
		Oct.20	0.5-1.5	14.8	sand-300	gravel bar
C	155.5-156.1	Aug.26	0.4-0.6	24.1	sand-100	concrete wall
		Oct.20	0.5-0.7	14.8	sand-100	concrete wall

Field Survey of Wintering Place

We carried out a field survey focussing on fish over-wintering places at six stations shown in Table 2, in February 1996. After the pre-investigation on foot, we chose six stations which appeared to be good places for over-wintering. In every station, fishes were counted by snorkelling at pools in the mainstream and caught by electro-shocker at shallow places near the tributary-junction.

Table 2: Outline of the Over-Wintering Fish Survey Stations

St. No.	Location (km)	Water Depth(m)	Water Temp.(℃)	Velocity (cm/s)	Bed Material Diameter(mm)	Riparian Condition
1	144.4-145.2	0.4-1.2	7.0	0.2-1.0	100-400	gravel bar
2	147.2-147.8	1.5-2.0	11.0	0.2-0.4	10-150	riprap-like concrete blocks
3	151.0-151.4	1.5-5.0	6.5	0.1-0.5	100-300	a few blocks + wall
4	153.1-153.5	0.5-1.5	6.9	0.1-0.4	0.07-50	step-like concrete revetment
5	157.0-157.8	2.5-3.0	3.7	0.1-0.2	2-200	riprap-like concrete blocks
6	158.2-158.6	0.5-1.2	3.7	0.2-0.5	100-250	a few blocks + wall

RESULT AND DISCUSSION

Changes in Water Surface Level at High Flow

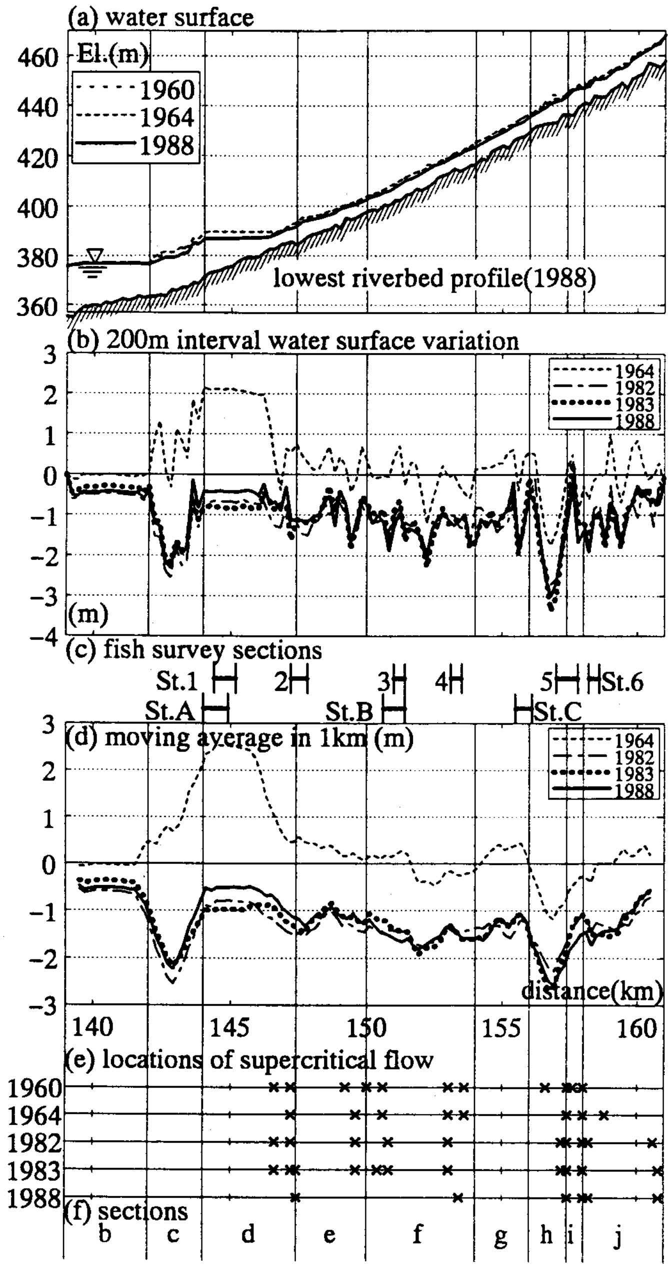

Figure 4
Calculated water surface profiles and the variation after the 1961 disaster, at high flow

Figure 4 shows: (a)water surface profiles of 1960 (before the 1961 disaster), 1964, and 1988, calculated for high flow (3,762 m^3/s) using the same values of Manning's coefficients as those shown in Figure 3, (b)differences between the water surface level of 1960 and the ones of 1964, 1982, 1983 and 1988 for each cross-section of 200m intervals, (c)location of the fish survey stations, (d)moving average in 1km (five cross- sections) of the

above b), (e)locations of supercritical flow, (f)names of each section previously mentioned.
From the figures, we may understand the safety level against flood has certainly improved in comparison with the condition before the 1961 disaster, at least from the view point of water surface level.
However, it should be noted:
1)there are some places with flood risks where the water surface level is very close to the 1960's level, for example at the upstream-end of section c (close to the fish survey station A and 1), near the boundary of section g and h (station C), and at section i (station 5),
2)in section d (including station A and 1), the water level has risen.

Changes in Maximum Water Depth and Water Surface Width in Normal Water Stage

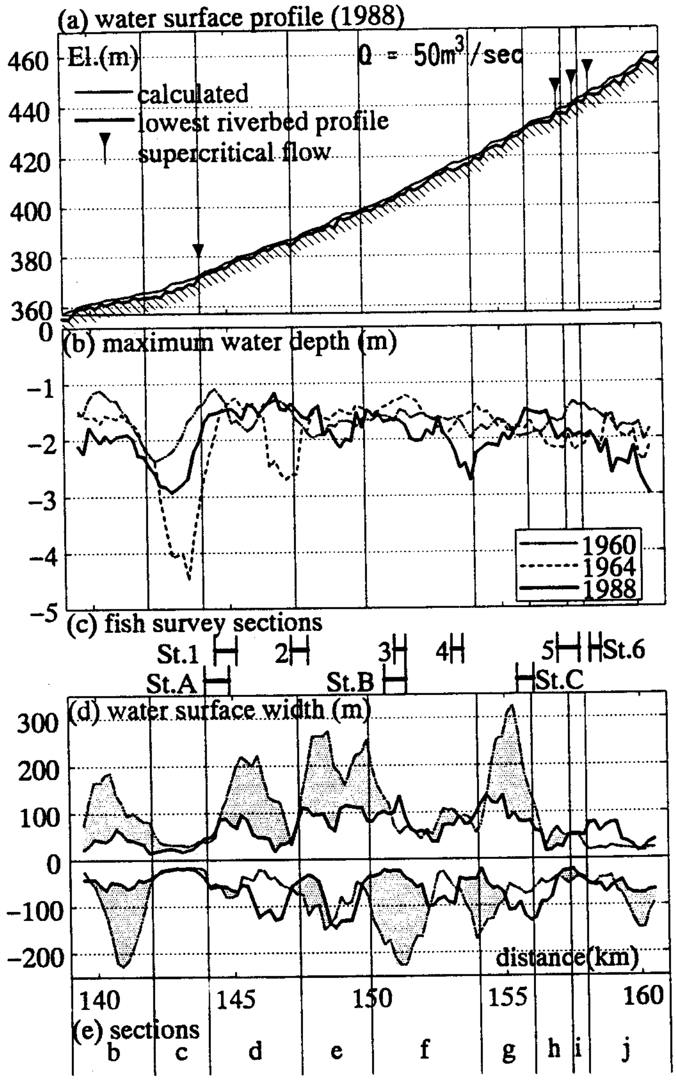

Figure 5
Changes in maximum water depth and water surface width at normal water stage

Figure 5 shows: (a)water surface profile calculated for normal water stage (50 m^3/s) using the same values of Manning's coefficient and cross-section data from 1988, (b)maximum water depth, (c)location of the fish survey

stations, (d)water surface width, in which both the width in 1960 and 1988 are drawn and any part where the 1960's width is wider than the 1988's is shaded gray, and (e)names of each section previously mentioned.
As shown in the figure (a), the number of the sections where supercritical flow occurred is less than in the case of high flow. This is because the calculation is one dimensional. Since the stream is braided, there are actually many supercritical sections with rapids or riffles in the normal water stage as shown in Photo 1 or 2.
Figure 6 drawn from the aerial photos also shows the braided streams in 1963, 1977, and 1992.

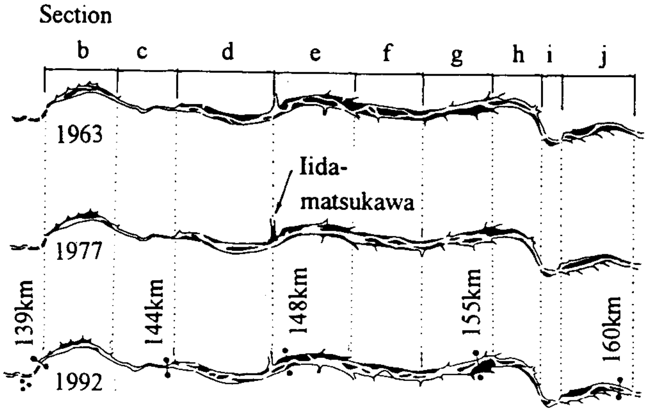

Figure 6
Changes of thalweg
at normal water stage

It may be seen from the figures (including Figure 6):
1)in section c and station A, maximum water depth is relatively large but unstable,
2)the water surface width is narrow and uniform in section c, boundary between section d and e (station 2), and section i (station 5),
3)the thalweg is historically steady at station A, 2, and 5.

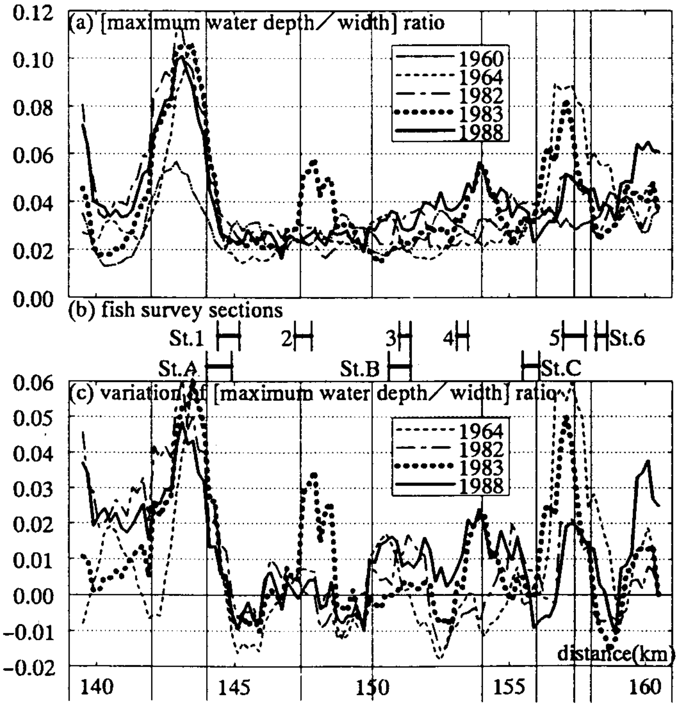

Figure 7
Changes in
maximum depth/width ratio

Changes in Fish Habitat Condition

Maximum Water Depth/Width Ratio

Typically in Japan when a flood protection project or dam is completed in an upper reach, normal discharge in the downstream reach decreases, so that the downstream riverbed becomes flattened by the stabilization of flow and the subsequent channel reconstructions. This less-variable stream flow consequently becomes unfavorably shallow for fish. Therefore, water depth, and water surface width in relation to the shallowness under the same flow rate, is one of the most important indices for examining habitat quality after channel reconstruction in Japan. This is similar to the frequent use of depth and width as indicative habitat variables in many habitat evaluation or prediction models in Europe and America (Fausch et al. 1988).
Looking again at Figure 5 from this point of view, we see that:
1)maximum water depth has increased almost everywhere, compared with depths in the 1960's,
2)water surface width has decreased also almost everywhere.

Width-to-depth ratio is also a frequently used habitat variables, but in Japanese streams, we have only limited experience in examining habitat quality by this ratio. According to our previous study in a channelized stream, the maximum depth/width ratio (inverse of width-to-depth ratio) shows a proportional relationship to the mid- to large- sized fish harvest (Nakamura and Tsukisaka 1994).
Figure 7 shows the changes of the maximum water depth/width ratio. In this figure, the variation in figure (c) indicates the difference between those of 1960 and other years.
From the figure, we may verify that the ratios have increased almost everywhere except a part of section d, e, h and j, and, at all fish survey stations, the ratio has been increased or remained at the same value as in 1960.

Number of Fish Species

Table 3 shows the results of surveys of major species in past literature from 1956 to 1981 which mentions fishes living in the study area (Cyubu Kensetsu-Kyokai 1992), along with the sampled fishes in the Fish Census of 1992 and the observed fishes in our winter-field survey of 1996.
As shown in the table, there seems no extensive change in the number of species before and after the 1961 disaster, although the number in our survey is relatively small because of the winter season. It may be concluded from the table that there is no evidence which suggests a decrease in the number of species caused by long-term changes in channel characteristics.

Fish Abundance in Each Station

According to the report of the Stream Fish Census in the Tenryu-gawa 1992 (Cyubu Kensetsu-Kyokai 1992), the number of the sampled fish is: 150 in Station A, 62 in Station B, and 69 in Station C. This result, in comparison with Figure 7, roughly supports the assumption that the fish habitat value for mid- to large-sized fishes is proportional to the maximum water depth/width ratio.
Table 4 shows the counted or caught fishes in our field survey. From the table, we see that Station 2 and 5 are extremely good as over-wintering places in the mainstream. This result, corresponding to the less-changeable places mentioned previously, suggests that historically steady pools are preferred as an over-wintering place by mid- to large- sized fish in the mainstream. The riprap-like micro habitat at Station 2 and 5, such as shown in

Photo 3, also seems to be attractive to fish.

Table 3: Major Species in the Study Reach

Japanese Name	English Name	Scientific Name	Species named in the literature of 1956	'65	'79	'81	'92*	'96**
Unagi	Japanese Eel	*Anguilla japonica*	○	○	○	○	○	
Amago	redspot masu trout	*Oncorhynchus masou macrostomus*				○	○	○
Iwana	Japanese char	*Salvelinus leucomaenis*	○	○	○	○		
Ayu	Ayu	*Plecoglossus altivelis*	○	○	○	○	○	
Funa	crucian carp	*Carassius* spp.	○	○	○	○		○
Koi	common carp	*Cyprinus carpio*	○	○	○	○	○	○
Kamatsuka	pike gudgeon	*Pseudogobio esocinus*	○	○	○	○	○	○
Higai	oily gudgeon	*Sarcocheilichthys variegatus*	○	○				○
Ugui	Japanese dace	*Leuciscus hakonensis*	○	○	○	○	○	○
Oikawa	pale chub	*Zacco platypus*			○	○	○	○
Dojyo	Asian pond loach	*Cobitis anguillicaudatus*	○	○	○	○	○	○
Namazu	Eastern catfish	*Silurus asotus*	○	○	○	○		○
Yoshinobori	freshwater goby	*Rhinobius brunneus*	○	○	○	○	○	○
Kajika	Japanese sculpin	*Cottus hilgendorfi*	○	○	○	○		
	number of other species		6	8	6	8	10	4
	total number of species		18	20	18	21	19	14

* Fish Census 1992, ** our winter-field survey of 1996

Table 4: Fish Data Obtained by the Field Survey

	Counted by snorkelling in pool Body length	Station No.						Caught by electroshocker near junction Body length	Station No.					
Name of fish	(mm)	1	2	3	4	5	6	(mm)	1	2	3	4	5	6
Japanese dace	60-250		35		1	18		27-135	70	4	20	75	3	1
pale chub	80-180	1	4					22-165	31	25	26	143		1
pike gudgeon	60-200	1	2		1	1		30-161	1			2		
crucian carp	100-200	1	12		2	23	3	54-132	9			1		
common carp	300-500	3			1			170	1					
Eastern catfish	450		1											
freshwater goby	30-70			2		15		39-54		3		2		
others		1	1						10	2	2	4		
Total		7	55	2	5	57	3		122	34	48	227	3	2

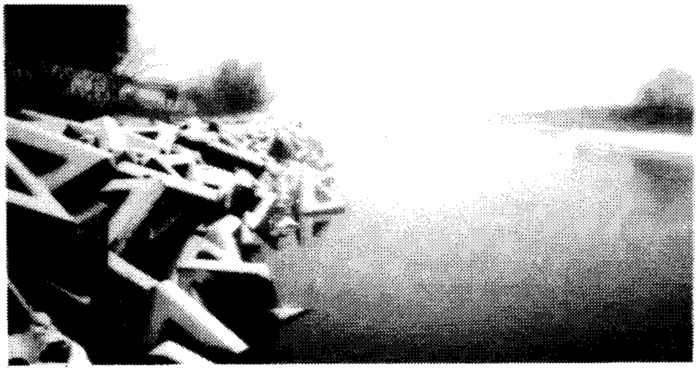

Photo 3
Riprap-like concrete blocks
in Station 5

CONCLUSION

Finally, we may conclude:

1)The safety level against flood has certainly improved in comparison with the condition before the 1961 disaster, even though there still remain some areas of flood risk.

2)Maximum water depth under the normal water condition has increased and water surface width has decreased, so that the maximum depth/width ratio as an important index of habitat quality has risen in many places.

3)There is no clear evidence indicating a decrease in the number of fish species due to long-term changes in the stream.

4)Even though the riprap-like micro habitat at pools seems to increase the preference of fish to over-winter, the results of the field survey suggests that historically steady pools in the main stream are much preferred by mid- to large- sized fishes as an over-wintering place.

Some of the good habitat areas are located near a natural small canyon which is a risky place for flood protection but could be developed as a scenic spot. How to improve such places, not only from the view point of flood protection but also from that of maintaining fish habitat and scenery, will be one of the future important challenges in this area.

ACKNOWLEDGEMENT

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ECOLOGICAL REHABILITATION OF THE RIVER MEUSE: A FRAMEWORK FOR ASSESSMENT OF ECOLOGICAL EFFECTS OF DISCHARGES

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ABSTRACT

Discharge characteristics largely determine the ecology of running waters. Ecological effects of low discharges are of special interest in the Grensmaas, a stretch of the River Meuse in the Netherlands. This rain-fed river, which flows through France, Belgium and the Netherlands, suffers from very low discharges in summer and autumn as a result of water extraction in Belgium and The Netherlands. International agreements have to be made regarding water economies in dry seasons, however, there is no quantitative ecological basis for these decisions.

In this paper, a framework is presented to assess ecological effects of low discharges. The framework provides the basis of a computer model which is presently being constructed.

When ready, the computer model will predict ecological effects by comparing habitat suitability at different discharges for a number of target species of the aquatic community. Habitat suitability is calculated for a number of determining variables (such as depth, stream velocity, substrate and water quality) from a GIS application of a hydromorphological model. The model is partially adapted from PHABSIM and modified for target species, spatial detail, hydrodynamics and water quality.

KEY-WORDS: River Meuse/ target species/ ecological effects/ low discharges/ PHABSIM/ HSI models/ GIS/ water management

INTRODUCTION

The catchment area of the River Meuse is managed by three water authorities in France, Belgium and the Netherlands to accommodate the changing requirements for water use for shipping, industry, agriculture, drinking water and other purposes. As a result of water extraction in France and Belgium, the Dutch section of the River Meuse receives very little water in the dry season. This water shortage affects the ecology of the Grensmaas, a stretch of the River Meuse with a relatively high ecological value. Allocation of Meuse water, therefore, is a point of discussion between the three countries. It is widely recognized that the international cooperation is needed to rehabilitate the ecology of the Meuse River. To assure that ecological rehabilitation has the same position in the discussion between the three countries as vested economic interests, ecological effects of low discharges must be assessed.

The present minimal discharge for the Grensmaas of 10 m^3/s is based on little ecological criteria set some time ago. In the late seventies, this value was established as the minimum discharge needed to dilute wastewater discharges from a chemical plant to the Grensmaas in order to prevent fish being poisoned. Since the late eighties, this value was also been used as the basic requirements for Simuliidae and Hydropsichidae, a group of rheophilous macroinvertebrates that are characteristic of the Grensmaas.

Sound criteria for minimum discharges need to be established to support future dicision making. A study was carried out to derive these ecological criteria by determining the habitat suitability for various aquatic organisms at different discharge characteristics. The study aims to develop an integrated model comprising geographical (GIS), hydraulics, and biological (HSI) information.

STUDY AREA

The River Meuse is 850 km long and drains a catchment area of 33,000 km^2 in France, Belgium and The Netherlands (Fig. 1). This rain-fed river has high discharges in winter and a distinct minimum flow in summer. Over the years, the river bed has been greatly modified by river regulation measures. Sluices and weirs have been constructed to enable navigation during low summer discharges. The Meuse catchment area has been described in detail by Descy and Empain (1984) and van Urk (1984). The Grensmaas is a non-navigable gravel reach of the River Meuse. The present status was created in 1930 when a lateral channel (Julianakanaal) was constructed for shipping purposes (Fig. 1). The river characteristics particularly the original morphodynamics have changed as a result of impounding of the upstream reaches, gravel mining and river training works in the Grensmaas. The armoured layer is only disrupted at peak discharges. The average width of the summer bed is 100 m. The mean summer and winter discharges of the Grensmaas at Borgharen are about 100 and 400 m^3/s respectively. The highest discharge ever recorded was 3100 m^3/s. The bankful discharge is approximately 1250 m^3/s. The summer is almost entirely inundated at a discharge of 100 m^3/s and about half of the cross-section of the summer bed is inundated at a discharge of 10 m^3/s,

The unfavourable conditions and the indifferent biotic community are due to a combination of factors. The Grensmaas is a harnessed river with steep river banks and hence does not have characteristic riverine habitats such as islands, snag, side channels, pools, coarse woody debris and floodplain forest. The water quality is poor and there are unnaturally short-term discharge fluctuations (Salverda *et al.*, 1996). Present oxygen conditions in the Grensmaas prohibit the occurrance of sensitive macroinvertebrate species (Bij de Vaate, 1995). Because of lack of characteristic riverine biotopes, the typical gravel river macrophyte Riverwater-crowfoot (*Ranunculus fluitans*) is absent (De la Haye, 1994). The aquatic benthos is dominated by dense growth of the algae *Cladophora* spec especially on riffles. Because the upstream and downstream reaches are impounded, most anadromous fish species have disappeared. Characteristic, rheophilous, fish species such as Barbel (*Barbus barbus*), Chub (*Leuciscus cephalus*), Ide (*Leusiscus idus*) and Dace (*Leuciscus leusiscus*) are found in low densities (Vriese, 1992).

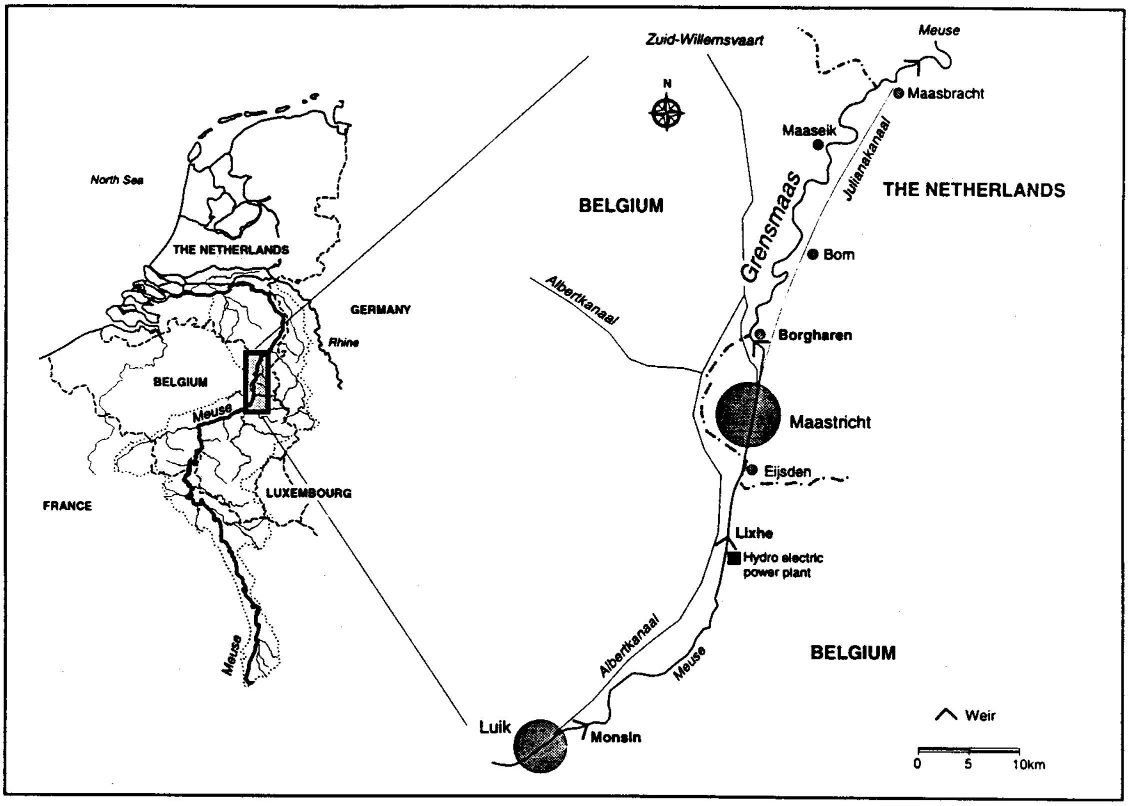

Figure 1: Map showing location of the catchment area of the River Meuse, Western Europe, and detail of the study area.

STRUCTURE OF THE FRAMEWORK

The effects of low discharges in the Grensmaas on selected target species can be assessed by simulating the changing habitat availability during different minimal discharges as reflected by the amount of weighted usable area (Bovee, 1982). The changing habitat availability of these target species is determined using the defined habitat preferences in HSI models (Milhouse *et al.*, 1984).

The application of physical microhabitat models (PHABSIM) to identify optimum flows to protect fish fauna is described by Orth and Leonard (1990). This method is based on an analysis of the habitat preferences of representative fish species and on hydraulic habitat simulation. The model is used to quantify the amount of potential fish habitat at various discharges as the basis of flow recommendations. The approach applied for the Grensmaas differs from that of Orth and Leonard (1990) in several ways:

- The selected target species need to be representative of the whole biotic community in the summer bed in order to assess ecological effects. Target species, therefore, include various groups of organisms present such as fishes, macrophytes and (macro)invertebrates.
- Unlike PHABSIM, our model simulates the changing physical habitat (hydraulic and morphological) of these target species during different discharges in the Grensmaas accurately and detailed relatively easily using an existing digital depth map (GIS) of the summer bed.
- The simulation of the physical habitat must be adapted to the specific hydraulic situation of the delta region (typical for rivers in the Netherlands).
- Water quality demands of the target species can be simulated using data on water quality parameters and discharges.

Comprising the principles of PHABSIM with the adaptations to the specific situation of the Grensmaas, a simplified framework of the model is presented in Figure 2. The compartments of the framework are elaborated here under.

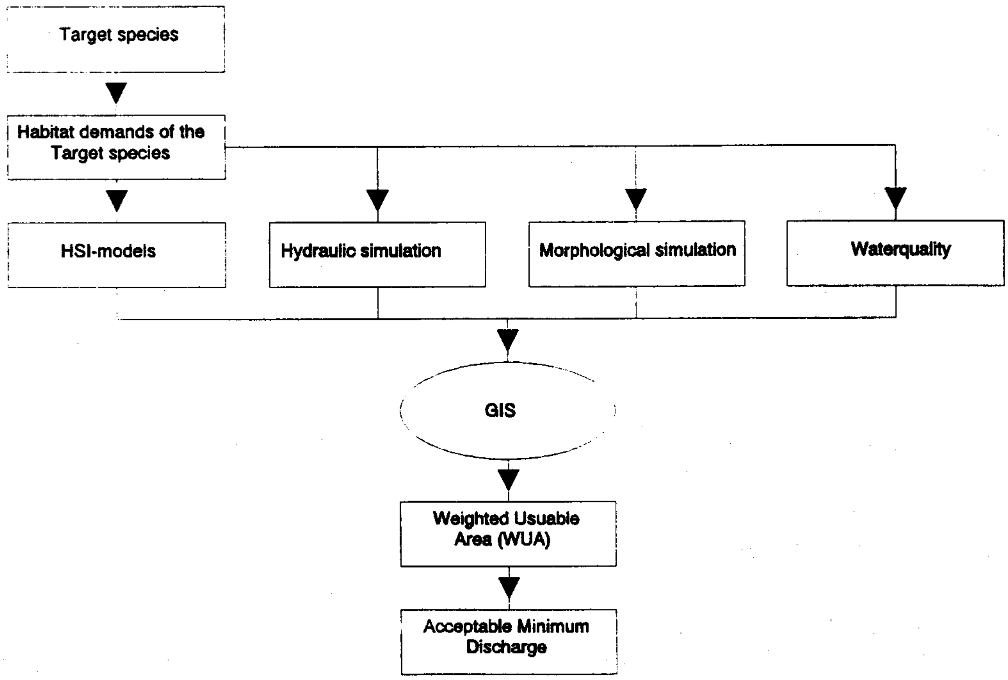

Figure 2: A simplification of the framework of the model

COMPARTMENTS OF THE FRAMEWORK

Target species

The target species have been selected from species living permanently in the inundated summer bed because they are most affected by low discharges. The selection was based on four criteria:

1) The target species must be representative of the biotic community expected after the implementation of a nature development plan and improvement of the water quality. The first selection of target species is based on the biotic community in the present situation, before 1900 and in more natural rivers resembling the Grensmaas.
2) For the target species adequate Habitat Suitability Index models (HSI) or (auto)ecological data must be available (so that suitability graphs can be developed).
3) The selection of target species must be representative of different groups of organisms (macrophytes, (macro)invertebrates and fishes) in order to estimate the effect of different low discharges on the biotic community.
4) The habitat suitability of the target species must be affected by changing minimal discharges.

Based on these criteria the following target species were selected (see Table 1): Loddon pondweed (*Potamogeton nodósus*); River water-crowfoot (*Ranunculus fluitans*); Yellow water-lily (*Nuphar lútea*); the caddis fly *Hydropsyche contubernalis* the mayfly *Palingenia longicauda*; Barbel (*Barbus barbus*); Bream (*Abramis - brama*); Chub (*Leuciscus cephalus*) Bullhead (*Cottus gobio*) Pike (*Esox lucius*) and Noble crayfish (*Astacus astacus*).

Habitat demands of target species

The habitat demands of the target species are defined in Habitat Suitability Index models (HSI; U.S. Fish and Wildlife Service, 1981). HSI models are composed of suitability graphs of all known habitat demands and are mostly optimum curves of the target species for physical or water quality habitat factors. The habitat demands used in the HSI models of the target species are presented in Table 1. The habitat demands are divided into physical (depth, flow velocity, substrate, cover and vegetation) and water quality habitat factors (temperature, oxygen, turbidity, secchi depth, trophic level, chloride and acid level).

HSI models of target species

The HSI models of the target species available are developed by research institutes in the Netherlands and the United Kingdom (TNO-BSA Working group planning, Delft; Organization for the Improvement of Inland fisheries, Nieuwegein and the University of Hull). These models describe the suitability of the habitat in different life stages for the most relevant habitat demands in suitability graphs (Figure 3).

Table 1. The most important habitat demands of the target species

	Physical					Water quality						
habitat factor / target species	depth	flow velocity	substrate	cover	vegetation	T (C°)	O_2	turbidity	secchi depth	trophic level	Cl^-	Ph
Potamogeton nodósus	x	x	x						x		x	
Ranunculus fluitans	x	x	x	x					x	x		
Nuphar lútea	x	x	x								x	
Hydropsyche contubernalis	x	x	x				x					
Palingenia longicauda	x	x	x			x	x					
Barbus barbus	x	x	x	x		x	x			x		
Abramis brama	x	x	x			x	x		x	x	x	x
Leuciscus cephalus	x	x	x	x	x	x	x	x		x		x
Cottus gobio	x	x	x		x	x	x					
Esox lucius	x	x			x	x	x	x			x	x
Astacus astacus	x	x	x	x	x	x	x		x			

The suitability of the habitat is presented as an suitability index (SI) on a scale from 0 (unsuitable) to 1 (suitable). These suitability graphs are used to estimate the suitability of the habitat of the target species in their different life stages on a scale from 0 to 1.

The suitability graphs from the HSI models will be incorporated in the computer model as well as possible. In some cases, these graphs could not be used because of the various descriptions of the habitat demands. To standardize the habitat demands in these cases, new suitability graphs were made on the basis of data in the HSI models.

Two HSI models were available for some species and in a few of these cases where the suitability graphs were different, the suitability graph with the best argued data were used.

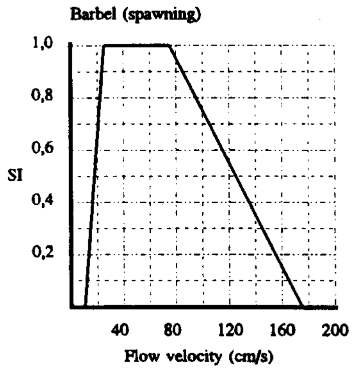

Figure 3: A suitability graph (flow velocity Barbel (Barbus barbus (spawning))

Hydraulic simulation

The most important physical habitat factors (depth, flow velocity and substrate) can be defined at different discharges using a quasi two dimensional hydrological model. The variation in the width has been included, using the cross-sections of the summer bed of the Grensmaas in the model. These cross-sections are made at 50 metre intervals in the river.

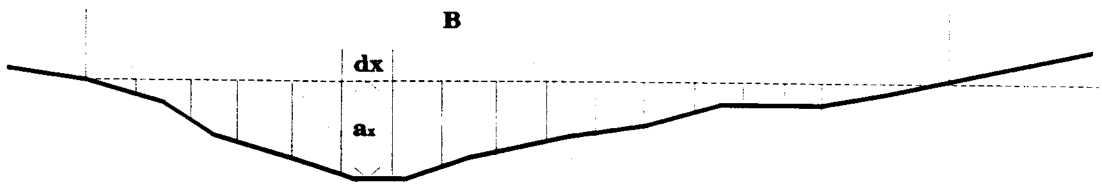

Figure 3: Schematic presentation of a cross section

Water movement in the Grensmaas will first be calculated, which delivers the variation in the length direction. In the model, the important parameters *A* (wetted cross-section) and *R* (hydraulic radius, according to Engelhund, 1964), will be calculated by integrating the water depth over the width of the cross-section. A quasi-steady state is assumed. This will result in the calculation of the flow profiles (1). If no significant changes of the flow profiles occur in the river stretch, the water level will be interpolated instead of using the hydrological model. This also delivers a relationship between total discharge (Q) and the water level (h) for each cross-section.

(1)
$$Q = C \int_0^B a^{3/2}\, dx \sqrt{i - \frac{dh}{dx}}$$

The variation in the discharge over the width of the river will be calculated by dividing the width of the cross-section (B) into parts (dx) with a minimal width of two metres (Figure 4). This gives the habitat factor depth (a_{xy}) in these parts at different discharges. The division of the discharge over the width of the cross-section (Q_{xy}) is related to the flow velocity (u_{xy}), the depth (a_{xy}) and the resistance of the river bed (C_{xy}), according to *Chézy*. The resistance of the river bed is related to the water depth and the bed roughness (k) is related to the seive diameter. The total discharge *Q* is divided according to (2):

(2)
$$Q_{xy} = \frac{\int_0^{dx} a^{3/2}\, dx\ 18\log\frac{12a_{avr}(dy)}{k}}{\int_0^B a^{3/2}\, dx\ 18\log\frac{12R}{k}} * Q$$

The habitat factor flow velocity in the cross-section (u_{xy}) will be determined with the following equation (3):

(3)
$$u_{xy} = \frac{Q_{xy}}{a_{xy}(avr)}$$

<u>Morphological simulation</u>

The flow velocity will be used in determining an indication of the modification of the substrate due to low discharges. During peak discharges in winter and spring, coarse local river bed sediment is transported and settles in the Grensmaas. At the same time, there is a large supply of fine sediment in suspension from the upper river basin (max. 500 mg/l). This sediment remains in suspension because of the high flow velocity. During low discharges in summer and autumn, coarse sediment is not transported in the Grensmaas. The diminished supply of fine sediment in suspension from the upper river basin (± 10 mg/l) partly settles. Three groups of fine sediment can be distinguished during low discharges: sediment transported in suspension (suspension transport); sediment transported along the river bed (river bed transport); and sediment that settles (this is the actual sedimentation). River bed transport and sedimentation of fine sediment modify the substrate and greatly affect the habitat suitability for macrophytes, invertebrates and fish spawning areas. To determine river transport and sedimentation at low discharges, the shear stress velocity u_* must first be calculated (4):

(4)
$$u_* = u \sqrt{\frac{g}{c^2}}$$

Aggradation of sediment is defined by the grain size of the sediment and the flow velocity. Shinohara and Tsubaki (1959) give $u_*/W = 5/3$ as a practical general indication for the start of suspension. In this equation W (m/s) is the rate of grain fall. The grain size D (m) for this fall rate is defined iteratively with one of the following equations (5-7):
for $D < 0{,}1$ m:

(5)
$$W = \frac{1}{18} \frac{\Delta g D^2}{\upsilon}$$

for 0,1 mm < D < 1 mm:

(6)
$$W = \frac{10\upsilon}{D} \left(\left(1 + \frac{0{,}01 \Delta g D^3}{\upsilon^2} \right)^{0{,}5} - 1 \right)$$

for D > 1 mm:

(7)
$$W = 1{,}1 \, (\Delta g D)^{0{,}5}$$

These equations will be used to define an indication of the grain size of sediment transported along the bottom instead of in suspension. Next an indication of the grain size of sediment that does not move and settles on the bottom will be calculated using the equation of Schields (1936):

(8)
$$\frac{u_*^2}{(\Delta g D)} = 0{,}03$$

In addition to the physical habitat, there are other physical habitat demands essential to specific target species such as cover and vegetation. Cover will not be defined in the computer model because major cover types such as boulders, overhanging banks and debris are absent during low discharges in the Grensmaas. There is little vegetation in the present situation and this does not affect the habitat suitability at different minimal discharges. Therefore, vegetation will only be defined as a potential habitat factor by estimating the theoretical presence of vegetation at different minimal discharges with the computer model.

Water quality

Over a ten-year period (1985-1995) water quality habitat factors such as temperature (C°), oxygen (O_2), turbidity, secchi depth, trophic level, chloride (Cl^-), acid level (pH) and discharge were measured daily at one location and monthly on five locations in the Grensmaas. These data will be used to describe the relationship between discharge and the concentration or level of the water quality parameters in the four seasons of the year. This relationship will be used to simulate water quality parameters during different minimal discharges. This simulation can then be used to define the suitability index of the water quality habitat factors at different minimal discharges. If there is no relationship between discharge and the water quality parameter, then the minimum or maximum values of the measured water quality parameters will be used.

Using a Geographic Information System for detailed spatial simulation

Unlike PHABSIM, it is possible to assess ecological effects on the study area with a high spatial detail using GIS. The habitat indices of about 90 suitability graphs (11 target species, 1-3 life stages and 5-10 habitat demands) will be estimated for every 2-5 metres in the cross- sections at about 10 different discharges. The distance between the cross-sections is 50 metres. As the Grensmaas is 50 km long and approximately 100 metres wide, at least 18 million habitat indices will be defined by computing the habitat factors in GIS. A digital topographic file (scale 1:5000) of the river bed will be used as base layer in a Arc/Info GIS file of the computer model. The height of the river bed is described in a computer file by the River Authority (RWSLOD). With the GIS application AMOR, these data can be processed to cross-sections and grid cells in a Arc/Info GIS format. In these cross-sections and grid cells depth and flow velocity can be calculated using the quasi two dimensional hydrological model referred to above. The substrate of the river bed is described in a digital geological map and a GIS file of the river bed. Changes in substrate for the grid cells can be calculated from the calculated flow velocity and the morphological simulation. The water quality habitat factors determined can also be linked to the grid cells in the Arc/Info GIS format.

Weighted Usable Area

In the GIS format all the habitat factors can be defined for each grid cell at different minimal discharges. The suitability graphs of the target species in their life stages are used to determine the suitability index of all habitat demands for each grid cell at different minimal discharges and periods in the GIS format. For each grid cell (i), a composite weighting factor (S_i) can be obtained by determining the minimal suitability index that limits the habitat suitability. An estimate of the amount of usable habitat, weighted usable area (WUA), can be calculated as (9):

(9) $$WUA = \sum_{i=1}^{n} S_i \ x \ A_i$$

where A_i is the area of cell i and n is the number of grid cells in the river bed (Orth and Leonard, 1990). This calculation can be repeated for each discharge, target species and life stage. However in evaluating the possible ecological effects of minimal discharges, it is not sufficient to calculate the effects on separate target species. At present, a method is being developed to estimate the effects on the biotic community.

Based on historic information and similar rivers in France, the desired biotic community is described (Helmer *et al.*, 1995) in the Grensmaas after the execution of a nature development plan. The abundance of species mentioned in this document can be used as a reference in measuring ecological effects. For this, the difference in the mean habitat suitability of the target species at different minimal discharges in a period and in the desired situation must be determined.

THE FRAMEWORK OF THE COMPUTER MODEL

The framework of the computer model in this paper is based on the principles of PHABSIM (Bovee, 1982; Milhouse *et al.*, 1984) and adapted to the specific situation of the Grensmaas. This computer model can be used easily to estimate the habitat availability of a limited number of representative target species in different life stages at different minimal discharges. The framework of the computer model is presented in Figure 5.

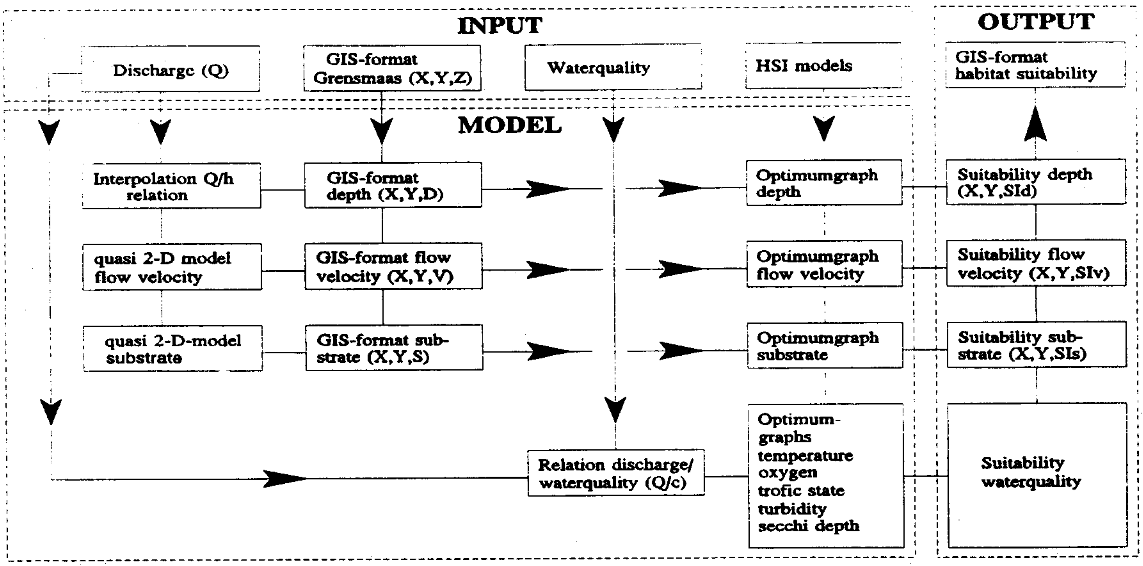

Figure 5: Schematic presentation of the model

The computer model will first interpolate the relationship between discharge (Q) and water level (h) for every cross-section. In combination with the Arc/Info GIS format, this will lead to a digital depth map for every minimal discharge required. With the quasi two dimensional hydrological and morphological model, these digital depth maps will be transformed in digital maps with flow velocity and after that substrate. The computer model compares these three habitat factors with the suitability graphs of the target species and calculates the suitability index for every point in the cross-section at every discharge. The suitability index of the water quality habitat factors at different minimal discharges can be defined using the relationship between water quality and discharge. These indices will also be coupled to a digital map. With the composite weighting factor of every point or grid in the Grensmaas, the weighted usable area (WUA) can be calculated.

CONCLUSIONS AND DISCUSSION

This study resulted in a framework of a computer model that can be used by the River Authority to quantify and evaluate the possible ecological effects of different water management options in times of water shortage. The quantification of ecological effects consolidates the position of ecology in the discussion between France, Belgium and the Netherlands on the allocation of Meuse water. In this way, the model contributes to the ecological rehabilitation of the River Meuse. The model can also be used to assess the effects on target species of other interferences in the river bed that affect the hydrological, morphological and water quality conditions such as dredging, construction of side channels, canalization and wastewater treatment.

The accuracy of the assessment of ecological effects by the computer model depends on the representativity of the target species, the correctness of the description of the habitat demands and suitability graphs and the accuracy of the quasi two dimensional hydro-morphological model.

A preliminary study (unpublished data) concluded that except for macroinvertebrates, the target species are sufficiently representative of the biotic community. To improve the framework, HSI models (or the relevant suitability graphs) for representative macroinvertebrates will be developed. The accuracy of the quasi two dimensional hydro-morphological model will be validated by measuring stream velocity, water depth and substrate at different discharges in different cross-sections in the Grensmaas. The correctness of the habitat demands and suitability graphs of the target species will be validated by comparing the present species composition and minimal discharge with the output of the model.

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Habitats and comprehensive management

Habitats et gestion intégrée des ressources

GOALS, INDICATORS, AND EXAMPLES OF RIVER BASIN STUDIES BASED ON ECOLOGICAL CONSIDERATIONS

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ABSTRACT

Frequently, goals of engineering studies with ecological considerations are not well defined and thus it is difficult to guide the analysis and also to monitor the degrees of project success. In this paper, the goals of these river basin projects are classified into the following three groups: Group 1 involves the enhancement of river basin according to the ecological requirements of certain dominant species; Group 2 considers ecological criteria of a diversity of species and thus the main goal is to restor the basin to a pre-determined historical condition; and river basins in Group 3 are aimed to achieve channel stability and aesthetic beauty and not necessarily according to any biological criteria. Often, a combination of these above three goals are used.

The second part of this paper is to discuss the types of indicators that can be used to guide the analysis and also to monitor the degrees of success of the project according to each established goal. Finally, two examples are used to illustrate ecological enhancement projects for the first two groups are presented. The first example is based on the restoration of the Kissimmee River, Florida, a project that was intended to restore the basin to a historical ecological conditions required by a variety of species. The second example is drawn from an engineering project designed to satisfy the ecological requirements of a single dominant species: migrating whooping cranes at the Niobrara River, Nebraska. In both cases, the ecological criteria were first determined through a combination of research by ecologists and engineers. These ecological criteria were then translated into engineering requirements for the design of these projects.

KEY-WORDS: Ecological Restoration / River Basin / Ecological Goals / Ecological Indicators.

INTRODUCTION

Decades ago, hydropower plants were considered the most environmentally sound means of generating electricity because they produce neither smoke nor nuclear waste. Gradually, however, the ecological consequences of dams and other river modifications have become appreciated. In particular, the environmental impact of the Aswan Dam in Egypt raised many concerns with regard to hydropower development. We learned that streams are not just conduits for supplying water for human needs; they must also serve as communities for other biological and botanical species. This paper discusses the general goal of attempts to improve the ecological properties of rivers, describes alternative specific objectives, and reviews two cases that provide insights into the potential for collaboration between ecologists and engineers.

GENERAL SCOPE

The general scope for the enhancement of a river basin is to achieve a sustainable condition so that human beings can live in harmony with their environment. The World Commission on Environment and Development (known popularly as the Brundtland Commission) defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (World Commission, (1987). Many international and national organizations, including the United Nations, UNESCO, World Bank, and the Earth Council have held meetings to discuss various concepts of sustainable development and sustainability indicators. In general, ecologists treated "sustainability" as preservation of the natural function and status of the ecological system, whereas economists emphasize the maintenance and improvement of *human living standards* as indices for sustainability.

Two key questions arise:

1. How much should we emphasize human interests in defining ecological goals?
2. How do we define and achieve sustainability in the ecological aspects of stream systems?

No precise answer can be given for either of these questions. When human interests conflict with those of another species, one must judge the appropriate goal and approach for each problem. Sustainability with regard to stream ecology is also difficult to define. Maintenance may be required to sustain certain aspects of streams. Perhaps the goals of modeling ecological constraints can be better described by the following examples.

SPECIFIC GOALS

It is possible to classify the basin ecological enhancement projects into three groups of specific goals as follows:

1. GROUP 1: Restore the river basin to a Natural Condition or a Selected Historical Condition. Ecological processes are dynamic: some processes are known now and others may not be known now or ever. Ideally, one wishes to enhance ecological conditions to satisfy future requirements, but unfortunately this is difficult to define and to achieve, especially for diversified species. A common approach is to attempt to restore a river basin to a

natural or selected historical condition. It is assumed that if the future hydrological condition is the same as that of the selected historical condition, then the majority of the ecological criteria will be satisfied. It is generally not feasible to return a particular stream in the United States to its condition during pre-Columbian time. Rather one is more likely to achieve success by returning the stream to a more recent historical condition. This general approach is particularly attractive when directed at diversified species. In this approach, one must investigate the historical conditions such as the frequency and the extent of flooding and the ranges of various flow parameters such as flow depths and flow velocity. The case study on Kissimmee River, as described later is a good example of this approach.

2.. GROUP 2: Achieve Certain Specific Ecological Criteria. If the ecological criteria for certain dominant species can be determined, then an approach would be to restore or maintain the river basin according to those specific ecological goals such as flow parameters required by different life stages, the magnitude and frequency of flooding, acceptable levels of sediment in the flow, and various water quality limits. A great deal of research is needed to understand the ecological criteria for many ecological elements under various hydrologic conditions. The case study of the Niobrara River (below) as given later is a good example of this approach.

3. GROUP 3: Maintain Stable Streams with Aesthetic Beauty: The third, less satisfactory, alternative is simply to design a stable and beautiful stream system. This may or may not satisfy various ecological criteria. Also, a dynamic rather than a stable stream may be more suitable in a dynamic ecological evolution in certain situations.

4. Often, the restoration of a river basin to its historical condition must also include considerations of ecological criteria and the stability of the streams such that any combination of the above three approaches may be used.

APPROACHES AND INDICATORS

After setting the goal as described in the previous section, the approaches and indicators to guide and to monitor the study should be formulated accordingly. Usually integrated efforts by specialists from various fields are required, and constant contact with the public would be helpful.

In most ecological studies, the essential ecological criteria for the basin should be determined first. This may be accomplished by a combination of reviewing current knowledge on fishery, riparian and other ecological requirements, the geographical and geological history of the basin, and engineering analyses of hydrological and hydraulic conditions. Field as well as laboratory experiments may be necessary.

Indicators can be rather useful to guide and monitor the project. Two different types of indicators are needed. The first type can be classified as diagnostic, the second as prescriptive. Diagnostic indicators are the indices for goal accomplishments and can also be divided into those for the public and those of the scientific community. Public diagnostic indicators can be an increase of fish and bird population, clean water, turbidity in the flow, etc.. These should be particularly simple and meaningful, and their goals achievable. Scientific diagnostic indicators are used by scientist to set their goals, such as salinity levels, ranges of flow velocities, etc. Prescriptive indicators, which include flows and salinity, are used to describe the controlling parameter for effecting the diagnostic indicators.

Certain elements can be used as different groups of indicators. For instance, delta smelt can be used as a public diagnostic indicator as well as a prescriptive indicator for the health of other species.

If restoration is the main goal, the degree of restoration to historical hydrological conditions can be used as a scientific diagnostic indicator.

EXAMPLES

Kissimmee River Restoration Studies

The channelization project in the Lower Kissimmee River Basin, Florida resulted in severe losses of river floodplain wetlands and waterfowl population throughout the river valley. An earthen channel, the C-38 canal, is the main element of this project. Immediately after the project completion in 1971, a strong effort was initiated to restore this basin to its pre-channelization status. This large Kissimmee River restoration project, if and when it is completed, will be a milestone in our journey toward ecological harmony because the only major goal of the project is ecological enhancement. In 1986 engineers and scientists from the University of California, Berkeley, working with the South Florida Water Management District, developed a set of restoration options. Ecological goals for the restoration plans were formulated. Alternative restoration plans were evaluated for their potential to satisfy the ecological goals. Analyses were based on a combination of field data collected, physical modeling, and numerical modeling. Finally, backfilling of certain reaches of canal C-38 was recommended. Details of this study are provided in Shen et al. (1994).

In most rivers, the ecological environment is the result of long-term adjustments by countless complex factors. Many of these factors and their interrelated processes are extremely difficult, if not impossible, to define. Thus, ecologists stress the need to restore a river basin to natural conditions after the occurrence of man-made changes. Unfortunately, in the Kissimmee River Basin, human activities have changed the upstream conditions so much that the regulated levels of Lake Kissimmee cannot be allowed to fluctuate as much as they did before channelization. Thus, it is necessary to establish a set of ecological criteria as the targets for restoration, rather than attempt to recreate prechannelization conditions, and the alternative restoration plans must be rated according to these criteria.

As in many other restoration projects, it is difficult to establish generally accepted ecological criteria because people representing various concerns stress different ecological goals. Conflicting requirements may even be proposed by the same group of people. Thus, the first task in our study of the Kissimmee River restoration was to search for a set of ecological criteria. These criteria must be ecologically justifiable and feasible, because if they are too restrictive then all feasible alternative measures can be eliminated. After many site visits, the principal investigator gradually satisfied the various interest groups that the study was sincerely searching for the most feasible plan for ecological restoration of the Kissimmee River. A symposium, suggested by the team and organized by the South Florida Water Management District, was held at Orlando in October 1988 to discuss various ecological and engineering concerns. This symposium was designed to focus on the restoration of the Kissimmee River ecosystem as a whole rather than on individual species. A set of ecological restoration criteria was established. These criteria will be discussed later. In essence, restoration requires that the floodplain receive flow relatively frequently to serve as a wetland. Also, floodwater should be directed back to the original river system slowly to revitalize the Kissimmee River. These ecological criteria must be satisfied while simultaneously meeting independent constraints for flood control, navigation, and sedimentation. The main engineering approaches were to divert flows to the river's original course and its floodplain by blocking the passage of flows in the C-38 canal with hydraulic structures such as weirs and earth plugs. Backfilling part of the canal was also considered.

The primary criterion for achieving environmental restoration goals is the reestablishment of prechannelization hydrology. Key characteristics of prechannelization hydrology were discussed in several papers presented at the Kissimmee River Symposium (Loftin et al., 1990a). Critical hydrologic determinants of prechannelization ecological integrity were reduced to a form that could be used to compare alternative restoration plans. Key hydrologic criteria are given below.

1. Continuous flow with duration and variability characteristics comparable to pre-channelization records

Historical data indicate that continuous discharge was a critical factor in the evolution and maintenance of biological communities in the prechannelization Kissimmee River ecosystem. Restoring the integrity of the Kissimmee River ecosystem depends on reestablishing the range of discharges of appropriate duration during representative (e.g., ten-year) post-restoration periods. Shen et al (1994) described the monthly discharge variations. The flow discharge characteristics required to restore biological communities that existed before channelization are: (a) continuous flow from July-October, (b) highest annual discharges in September-November and lowest flows in March-May and (c) a wide-range of discharge variability. These features should maintain favorable levels of dissolved oxygen during summer and fall, provide non-disruptive flows for fish species during their spring reproductive period, and restore the temporal and spatial heterogeneity of river channel habitat.

2. Average flow velocities between 0.24 - 0.55 meter per second (mps) when flows are contained within the river channel

Specifically, flow velocities within 60% of river channel cross sections must not fall below 0.24 mps for more than three consecutive days during July-September and ten consecutive days during October-June. These velocities complement the discharge criteria by protecting river biota, because too little flow results in low concentrations of dissolved oxygen and excessive flow could interfere with important biological functions (e.g., feeding and reproduction of fish).

3. Overbank flow along most of the floodplain when discharges exceed 40 - 57 cubic meters per second

This criterion reinforces and will reestablish important physical, chemical, and biological interactions between the river and its floodplain.

4. Flood recession rates on the floodplain

A slow flood recession rate of less than 0.3 meters per month is required to restore the diversity and functional utility of floodplain/wetlands, foster sustained river/floodplain interactions, and maintain river quality. Slow drainage is particularly important during biologically significant periods, such as the nesting season for wading birds. Rapid recession rates (e.g., rates that will drain most of the floodplain in less than a week) lead to fish kills and thus are not compatible with ecosystem restoration.

5. Floodplain inundation frequencies comparable to pre-channelization hydrology, including seasonal and long-term variability characteristics

Shen et al (1994) shows the prechannelization inundation frequencies for the floodplain adjacent to the Fort Kissimmee gaging station and provides guidelines for this criterion. For example, during a representative ten-year period, November stages should inundate 100% of the floodplain during 4 of the 10 years and 75% of the floodplain during 7 of the 10 years. When the entire floodplain is inundated, depths along the periphery of the floodplain should measure between 0.3 - 0.6 meter. These inundation levels will lead to redevelopment of floodplain structure and function and reestablish the floodplain as an integral component of the Kissimmee River ecosystem. Ecologically, the most important features of stage criteria are water level fluctuations that lead to seasonal wet-dry cycles along the periphery of the floodplain, while the remainder (approximately 75 percent) of the floodplain is exposed to only intermittent drying periods that vary in timing, duration, and spatial extent. As stated in Loftin et al. (1990, p. 26):

Reestablishment of ecological integrity requires that all restoration criteria are met simultaneously. A piecemeal restoration program in which some of the established restoration criteria are achieved in one segment of the system, and other criteria are met in another portion, will not accomplish restoration goals and may be of little or no value. Game fish populations, for example, may still be limited if appropriate flow characteristics are restored but production of potential food resources on the floodplain is limited by inadequate inundation, or inaccessibility to river fish because levees or beams block the connectivity (interaction) between the river and floodplain. Alternatively, restoration of floodplain inundation frequencies and the pre-channelization stage-discharge relationship probably would not benefit game fish species unless enough flow is reestablished to improve dissolved oxygen regimes in the river during summer and fall. Similarly, water level fluctuations in broadleaf marshes on the floodplain will be of no value to wading birds if reestablishment of peripheral wet prairie habitat is restricted or precluded by rapid stage recession rates.

Three alternative restoration plans have been selected by the District for analysis. In the Fixed Weir Plan, ten weirs would be installed along the canal to divert flows into the river oxbows adjacent to these weir locations. In the Level I Backfill Plan, the same canal reaches in which weirs would be installed in the Weir Plan would be completely backfilled between the two junctions with the oxbows. In the Level II Backfill Plan a specific, long, continuous canal reach would be backfilled. New river reaches would be created to maintain a continuous river reach with the same canal backfill reach.

Fixed Weir Plan

The advantage of using weirs is their flexibility in operation, especially with gated weirs. Gated weirs can be opened during floods to decrease the need for flood levees or additional flooding rights.

During high floods the oxbow flow velocities would be between 2-3 mps. These velocities would cause erosion and deposition of sediments in the river oxbow reaches and could interrupt navigation. Channel maintenance probably would be needed after major floods.

Approximately 40-50 percent of the oxbow length (revitalized river channels) adjacent to the weirs would have flow velocities within the ecologically acceptable range of 0.24 to 0.55 mps. These oxbows were part of the active river system before the construction of C-38 canal. However, many more reaches would have velocities greater than 0.55 mps (with maxima on the order of 1.5 mps) than below 0.24 mps. About 26 kilometers of banks (revitalized river channels) would be inundated for flow discharge of 40 cubic meters per second at the entrance of the river and 57 cubic meters per second at the downstream end of the river. About half of this inundation length would occur at oxbows. Several existing control structures in the river, together with their levees, can control the recession rates in the lower 40 percent of the pools. However, the stage recession rate after floods, in at least 60 percent of all upper reaches of pools, would far exceed 0.3 meters per month. These rates could even reach 0.3 meters in a six-hour period. Complex flow operation schemes might be devised to control both inflows and outflows from each pool to reduce the recession rate.

Level I Backfill Plan

For this plan, during high floods, the oxbow flow velocities would vary between 1.5 and 2.1 meters per second. These velocities can cause erosion and deposition in the oxbows. During normal flows, between 18-33 percent of the oxbow velocities were in the range of 0.24 to 0.55 meters per second. Except for low flows, more flow velocities were above 0.55 mps (with maxima in the order of 1.5 mps) than below 0.24 mps. About 16 kilometers of banks (in revitalized river channels) would be inundated for flow discharges of 40 cubic meters per second at the entrance of the river and 57 cubic meters per second at the exit end of the river. All internal control structures together with their levees can only control the recession for the bottom 40 percent of the respective pools. At the upper 60 percent, the stage recession rate after floods would far exceed 0.3 meter per month. It is nearly impossible to design complex flow operation schemes to control both inflows and outflows from each pool to reduce the recession rate because the capacity for flood flow under this plan would be greatly reduced by the earth plugs blocking the canal. The amount of time that given proportions of the floodplain would be inundated under this plan is slightly less than would be inundated under the Fixed Weir Plan, and these values are far from meeting the requirements for ecological restoration.

Level II Backfill Plan

The Level II Backfill Plan should produce flow velocity ranges close to the ecological criteria if the future precipitation regime matches the precanal conditions and the flow regulation scheme can be properly designed. Historical data suggest that the flow properties of the Kissimmee River were similar in 1901 and 1958. In addition there was no detectable change in the river's course. During high floods, the oxbow flow velocities would vary between 1 and 1.3 mps and probably would not interrupt navigation. During normal flows, in 40 to 52 percent of the lengths of the oxbows, the flow velocities would be in the range from 0.24 to 0.55 mps, and in very few oxbows would the velocities exceed 0.55 mps. The flow velocity in all oxbows would be below 0.76 mps.

It is recognized that the upper lake level flood control capacity should be provided by leaving a portion of the Canal C-38 intact in the upper reach. Perhaps the best approach in this plan is to completely backfill part of the river reach. This should satisfy the flood criterion for Lake Kissimmee upstream from the Kissimmee River. Sedimentation problems can be investigated with a careful monitoring system. If significant movement of sediment occurs, appropriate actions should be taken. Construction would be carried out in several stages over several years. The extent of backfilling would be governed by the amount of available funding, available soil, the extent of the vegetation growth, and the possible relaxation of the water level constraint at Lake Kissimmee during flooding.

Field tests should be conducted at different times to monitor growth of vegetation. Knowledge of vegetation growth should be useful both to protect against erosion and to analyze hydraulic resistance. The data collected in the field during future years would determine the exact extent of backfilling.

A certain amount of dredging in the oxbows may be needed. In the Level II Backfill Plan, the bank erosion and sediment deposition would be rather limited. Only a small amount of maintenance dredging in the oxbows may be needed after major flooding.

It is recommended that one type of plan be selected by the District from the three plans for final engineering design and construction analysis. Certain detailed engineering analyses such as the extent and sequence of the backfilling, as well as the number and location of earth plugs, or weirs, are still needed. The possible effects of each tributary (slough) during flooding should also be investigated. Currently sufficient data are not available to make these analyses.

In accordance with the study team's recommendations, the state of Florida has requested funding from the U.S. Congress. At the direction of Congress, the U.S. Army Corps of Engineers, Jacksonville District, has investigated the team's work and converted it into an engineering feasibility project, which is awaiting further federal funding.

Niobrara Whooping Crane Studies

The U.S. Bureau of Reclamation had a plan to construct a water supply dam on a braided Niobrara River near Norden, Nebraska. The primary objective was to supply irrigation water for local farmers. This plan was stopped by the courts after the U.S. Fish and Wildlife Services pointed out that Whooping Cranes, an endangered species, stop at that reach of the Niobrara River during their migration seasons. I was asked to develop an engineering design compatible with the requirements of the cranes. Because there are only about 100 whooping cranes, it is difficult to determine their ecological habitat needs, although their requirements are believed to be similar to those of the sandhill cranes. Sandhill cranes prefer to rest on sandbars that are submerged by less than 0.2 meter of slow flowing water amid vegetation shorter than 0.9 meter, and away from the riverbanks. The first requirement was imposed by their desire for food and perhaps for protection, to have water reach not above their joints on the feet. The second and third requirements are necessary to detect and evade predators. Based on these requirements, we determined that the braided feature (a river reach with multiple anastomosing channels) was critical to provide shallow, vegetated sandbars at sufficient distance from the riverbanks.

We identified the flushing flow requirements necessary to maintain the braided stream. Certain portions of the stream below the proposed Norden Dam would be unavoidably changed from braided to meandering by the release of flows with relatively small amounts of sediment particles from the reservoir. We planned to maintain the braided characteristic of the river farther downstream by regulating the flow, controlling vegetation growth, and even breaking the ice jams that form annually. We reported our conclusions to both the Bureau of Reclamation engineers and Fish and Wildlife Service biologists. After the conclusion of our study, we were pleased to learn that the Fish and Wildlife Service agreed to withdraw its objection to the construction of the dam if the Bureau of Reclamation would follow our recommended flow release plans. The Bureau of Reclamation agreed to follow our recommendations. In the end the Dam was never built due because of a lack of funding. Like the Kissimmee Restoration Project, this study indicated that engineers and biologists can work together, and engineers can provide plans to satisfy ecological needs.

CONCLUDING REMARKS

If possible, the goals, approaches and indicators for each project should be established clearly near the beginning of the study. The goals should be rational, simple and achievable. A great deal of knowledge such as the fish behavior at different stages of life spans are still not known. Specialists from various fields should integrate their efforts to establish a harmonic environment for all. This type of conference can bring us together.

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THE HABITAT EVALUATION PROCEDURE IN THE POLICY ANALYSIS OF INLAND WATERS IN THE NETHERLANDS: TOWARDS ECOLOGICAL REHABILITATION

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ABSTRACT

In the Netherlands, ecological rehabilitation of rivers, lakes and tidal water systems has a high priority in planning and policy for nature and water management. Recently, the Ministry of Transport, Public Works and Water Management has started the 'Aquatic Outlook' project to develop strategies to reinstate the ecological conditions and values of the inland and coastal waters, whilst improving the opportunities for functional use of these water systems. In order to assess the effectiveness and the impacts of the rehabilitation measures of the different water management strategies a framework for habitat analysis and evaluation of the inland waters has been developed. This framework is especially designed for policy analysis studies on water management. The framework is based on the concept of the Habitat Evaluation Procedure (HEP). Thus far applications of HEP in policy analysis studies on the national level are lacking. In this paper, the framework for analysis is elaborated. The applicability of the analysis framework will be illustrated by two cases of different watersystems, the river Rhine and the fresh water lake IJsselmeer. The usefulness of HEP as a new tool in the policy analysis for sustainable inland water management is discussed.

KEY-WORDS: policy analysis, water management strategies, ecological framework, impact assessment, ecotope, carrying capacity, habitat evaluation procedure, habitat suitability, AMOEBA approach, ecological rehabilitation.

INTRODUCTION

In the Netherlands, nature values of the inland waters are endangered or have disappeared in the last decades due to water pollution, habitat loss and deterioration of the quality of the remaining aquatic habitats (Rijkswaterstaat, 1990). Nowadays, habitat restoration is one of the major policy issues of inland water management. Rehabilitation of wetland habitats often aims at the improvement of the natural hydrodynamic processes, e.g. water level fluctuation, stream velocity, inundation frequency. Also, ecological rehabilitation includes reduction of agricultural activities in former wetlands and improvement of the water quality. It will be evident that ecological rehabilitation measures have an impact on other functions of the water system. Successful ecological rehabilitation and sustainable use can therefore only be accomplished through integrated management of the water resources (Marchand et al., 1995). As a part of this management process policy analysis is considered to be an effective tool (Marchand et al., 1992; Karssen et al.; 1994; Luiten, 1995).

Recently, the Ministry of Transport, Public Works and Water Management has started the so-called 'Aquatic Outlook' project to develop water management strategies to reinstate the ecological conditions and values of water systems, while improving the opportunities for functional use. The ecological objectives for inland waters are derived from water systems which have not or have only slightly been influenced by human activities: the ecological reference system. Three sources have been used to determine the reference system (Ten Brink e.a., 1991; Laane and Peters, 1993): (i) inventories which have been made in the past, (ii) comparative research involving other systems and (iii) ecological theory. In order to define quantitative and verifiable ecological objectives indicator species have been selected. The selected species are characteristic of the water systems considered and as a whole they provide a reasonably representative picture of the ecological conditions. The number of species used as an indicator is on the average of 25 species per water system.

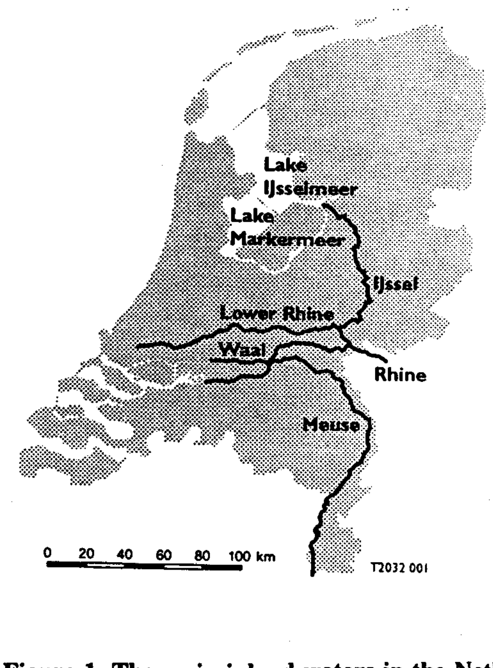

Figure 1. The main inland waters in the Netherlands.

For the assessment of the effectiveness of measures for ecological rehabilitation a habitat evaluation method is essential. In the Netherlands, the Habitat Evaluation Procedure (HEP) has recently become a standard approach for impact assessment and evaluation of measures causing changes in the environmental conditions of habitats for flora and fauna species (Duel et al., 1995). However, thus far applications of HEP in policy analysis on the national level are lacking. Therefore a framework for analysis based on the concept of HEP has been developed. Also, new concepts for ecological impact assessment and evaluation are implemented in the framework. The applicability of the framework for analysis will be discussed and is illustrated by two different main inland water systems in the Netherlands: the river Rhine and its branches and the fresh water lake IJsselmeer (figure 1).

FRAMEWORK FOR ANALYSIS

For assessment of the effectiveness and the ecological impacts of rehabilitation measures the HEP approach is used. Basically, HEP is a set of habitat suitability index models and analytical procedures to use the models for habitat evaluation in an appropriate way (US Fish and Wildlife Service, 1980). For the policy analysis a new set of analytical procedures has been developed and in addition ecological models have been implemented.

The structure of the framework for policy analysis of water management strategies for ecological rehabilitation of inland water systems is shown in figure 2. The framework consists of four blocks: (i) the ecotope classification of aquatic systems (ECLAS), (ii) the potential carrying capacity model (MORRES), (iii) the habitat suitability index models (HSI-models) and (iv) the AMOEBA-method to evaluate the effectiveness of rehabilitation measures.

Ecotopes are defined as spatial ecological units of which composition and development are determined by abiotic factors relating to hydrodynamics, morphodynamics and land use dynamics (Runhaar, 1986; Rademakers and Wolfert, 1995; Delft Hydraulics et al., 1996; Pedroli et al., 1996; Duel and De Vries, 1996). Ecotopes are comparable with cover types in HEP (US Fish and Wildlife Service, 1980; Terrell et al., 1982) and meso-habitats in the Instream Flow Incremental Method (Bovee, 1982; Stalnaker et al., 1994). ECLAS is elaborated for different types of inland water systems: rivers, lakes and tidal water systems. For each type of inland water system 15-20 main ecotopes have been distinguished. Measures concerning water management and habitat restoration have been worked out into ecotope classification criteria. By this way changes in the acreage and distribution of ecotopes are estimated.

For the assessment of potential carrying capacity of a water system for the selected species the model MORRES has been used. This software package, issued by Delft Hydraulics, consists of a comprehensive set of mathematical rules to calculate the areas of breeding, spawning or foraging habitats, based on the areas and distribution of the ecotopes and the hydrodynamics and morfodynamics of the water system (Duel & De Vries, 1996).

The EKOS software package, issued by Delft Hydraulics and the Ministry of Transport, Public Works and Water Management, has been used to calculate the habitat suitability of the different habitats of the selected flora and fauna species. This package has been developed for application of HEP in the Netherlands (Duel et al., 1995). EKOS makes use of habitat suitability index models for more than 60 flora and fauna species.

The AMOEBA approach is a method to evaluate the impacts of rehabilitation and water quality improvement measures on the habitat suitability and the carrying capacity of water systems (Ten Brink et al., 1991; Van Dijk &

Marteijn, 1993). In the AMOEBA approach, the impact of the policy alternatives on flora and fauna species is related to the ecological objectives. The score card for ecological rehabilitation is based on the similarity in population development between the ecological impact of a certain policy alternative and the ecological objective for each of the selected species. No distinction is made between the values for the various species. The score card figure obtained is the ecological development index scaled from 0 to 100 (Duel et al., 1996). The index value 100 indicates the ecological reference situation.

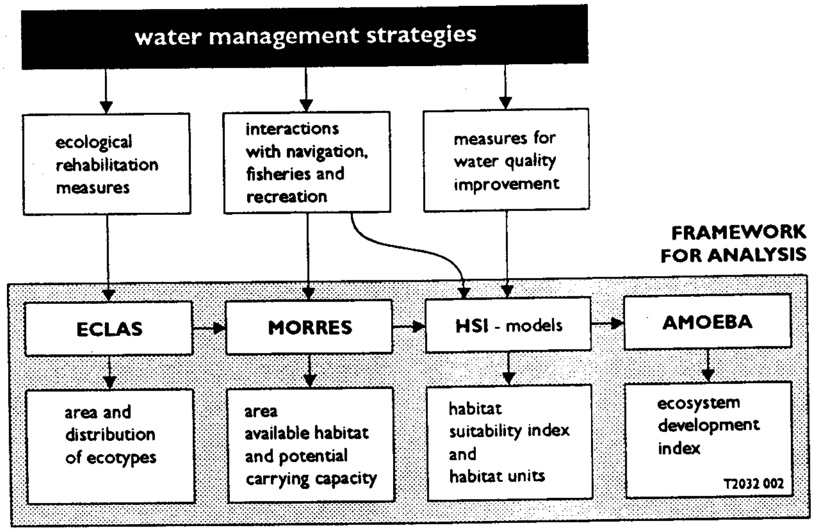

Figure 2. Framework for analysis of the effectiveness of water management strategies for ecological rehabilitation.

ECOTOPES, HABITATS AND HABITAT SUITABILITY

Habitat rehabilitation should not be understood as reconstruction of a former situation. Although the historical situation can be used as a reference, the current boundary conditions set by both nature and culture are different. New ecological objectives have to be found. This is a challenge for landscape planners, engineers and environmentalists. Because natural succession is, apart from human management, mainly dependent on habitat conditions, i.e. abiotic conditions of the site, abiotic site classification is required for proper habitat evaluation (Pedroli & Marchand, 1994). Since the ecotone concept concentrates on the gradients (Naiman and Decamps, 1990, Pringle et al., 1988), this concept is less suitable for classification purposes. The ecotope concept is more suitable concentrating on the differential key factors for habitat classification (Rademakers & Wolfert, 1994; Duel et al., 1994). An ecotope is a spatially discernable ecosystem type defined by the dominance of specific abiotic factors (Rademakers & Wolfert, 1994). For example, criteria concerning mean stream velocity, water depth at low river discharge, annual average duration of flooding and land use types are applied for the classification of the river ecotopes (Duel & De Vries, 1996). Examples of river ecotopes are deep riverbed, oxbow lake and river dune (figure 3).

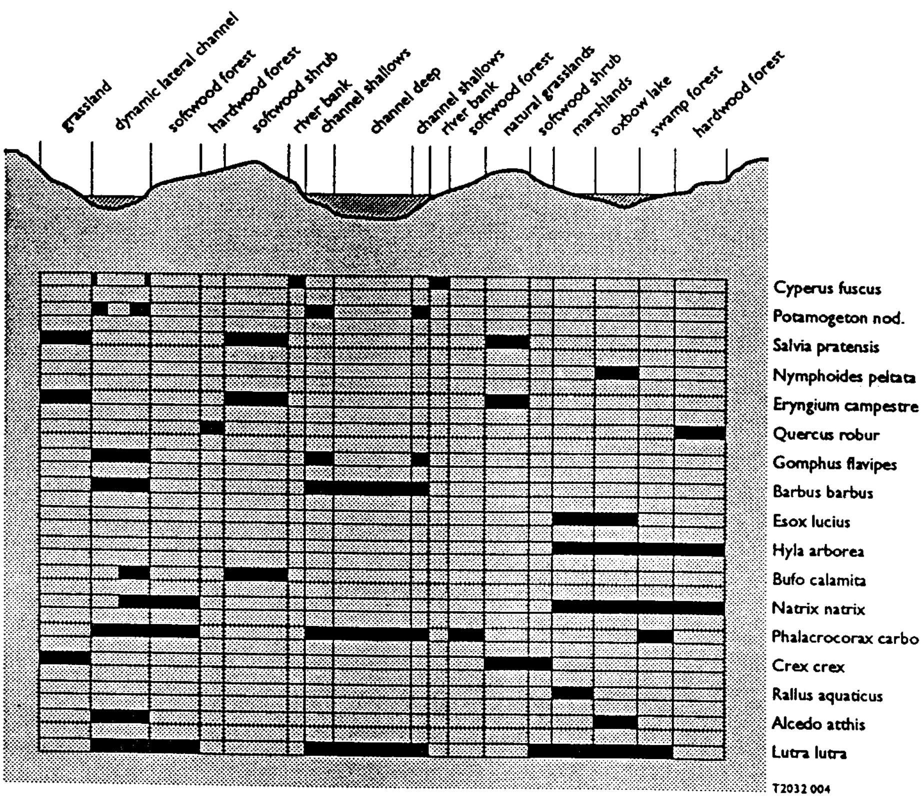

Figure 3. Illustration of the distribution of ecotopes and habitats in a cross section of a river

Since reactions of species and communities to changes in abiotic factors are difficult to predict, a specific set of habitat parameters is commonly used as the basis for prediction of potential ecotope succession. The size and distribution of ecotopes can be interpreted in terms of habitat for species of interest for nature conservation. To identify habitats for flora and fauna species the ecotope classification system is elaborated in more detail. For the classification of subecotopes a new set of criteria has to be distinguished, e.g. vegetation structure and soil type. MORRES calculates the total area of available habitat (see box 1). The total area of available habitat for the flora and fauna species includes all areas of ecotopes and subecotopes that provide lifecycle support to the species reviewed.

The habitat suitability of the available habitats is determined with HSI-models. In the Netherlands, habitat models are developed for flora species, macro-invertebrates, fish species, waterfowl, wetland birds and mammals (Duel et al., 1995). The HSI-models are based on the habitat requirements and preferences of the flora and fauna species. The models include only the main environmental factors limiting the population of the species reviewed. Examples of such factors are water quality, water depth, stream flow and vegetation cover (figure 4). The habitat requirements and preferences are derived from life history studies, field observation studies and statistical analysis of the environmental factors of the habitats used by the species observed (Duel e.a., 1995). The habitat models produce numerical ratings, which represent the carrying capacity of the habitats reviewed. The range of index rating is 0.0 to 1.0, expressing the range of unsuitable to optimal habitat conditions. The overall habitat suitability is determined by the suitability index ratings of the habitat factors limiting the carrying capacity of the habitat. The total carrying capacity of a water system for the selected species is derived from the areas and the suitability of the available habitats, expressed in habitat units:

$$HU = \text{sum}(HSI * A)$$

where HU = habitat units (ha)
HSI = habitat suitability index of available habitats
A = area of (sub)ecotopes which provide habitats (ha)

Box 1. Potential carrying capacity for Otter (*Lutra lutra*) in river systems

Total area of available nesting habitat (ANH) and foraging habitat (AFH), expressed in ha:
ANH = RS + SF + OLs
AFH = MC + LCw + FCw + OLw + ALw

Potential carrying capacity (PCC), expressed in potential number of individuals:
PCC = min (Dn * ANH, Df * AFH)

where RS = total area of ecotope reed swamp (ha)
SF= total area of ecotope swamp forest (ha)
OLb= total area of ecotope oxbow lake, subtype bank vegetation (ha)
MC= total area of ecotope main channel (ha)
LCw= total area of ecotope lateral channel, subtype water (ha)
FCw= total area of ecotope floodplain channel, subtype water (ha)
OLw= total area of ecotope oxbow lake, subtype water (ha)
ABw= total area of ecotope artificial backwater, subtype water (ha)
Dn= otter density (number of individuals per ha nesting habitat)
Df= otter density (number of individuals per ha foraging habitat)

WATER MANAGEMENT STRATEGIES FOR ECOLOGICAL REHABILITATION

Strategies for Ecological Rehabilitation

In the Aquatic Outlook project, three different alternatives for water management have been worked out: (i) the Economical Strategy, (ii) the Environmental Strategy and (iii) the Autonomous Development Scenario. In the Economical Strategy ecological rehabilitation measures will be taken only in combination with measures aiming at the improvement or maintenance of the economic functions of the water systems. Existing plans for improvement of the water quality will be executed. The Environmental Strategy aims at the restoration and development of wetlands at large scale. The opportunities for ecological rehabilitation are utilized with the preconditions of protection against flooding and the maintenance of infrastructure. The water quality of the inland waters is assumed to meet the habitat requirements of characteristic aquatic flora and fauna species. The Autonomous Development Scenario is characterized by a continuation of the present policy with respect to water and nature management. The existing plans for ecological rehabilitation will be executed. Also, measures will be taken to improve the water quality of the inland waters to the basic ecological level.

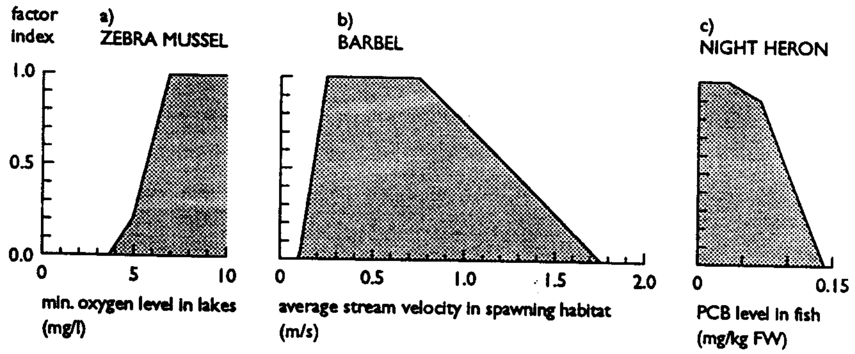

Figure 4. Examples of habitat suitability index curves for (a) Zebra Mussel (*Dreissena polymorhpa*), (b) Barbel (*Barbus barbus*), (c) Night Heron (*Nycticorax nycticorax*).

River Rhine

The river Rhine runs 1320 km long from its headwaters in Switzerland to the delta waters in the Netherlands. It has a catchment area of about 200,000 km^2. World wide the Rhine ranks among the inlands waters with the highest shipping traffic densities. To provide protection against flooding and to accommodate shipping and agriculture the river has been changed into one navigation channel of constant width, bordered by river banks on which brickyards and sand and gravel works are located. The adjacent floodplains are used as pastures. As a consequence characteristic riverine habitats have been disappeared. Water pollution has contributed to the ecological degradation as well. However, in the last decades the water quality is improved due to measures taken in the Rhine catchment as whole (Van Dijk and Marteijn, 1993). Nevertheless, to revitalize the Rhine ecosystem habitat restoration is essential. The water management strategies for ecological rehabilitation aim at the improvement of the secondary channels, the floodplains, the islands and the river banks:

(i) the creation of lateral channels and the improvement of the water flow conditions of secondary channels which are currently silted up, to provide and restore habitats for rheophillic macro-invertebrates and fish species which are seriously endangered along the Rhine, like may flies, caddish flies, Barbel (*Barbus barbus*), Ide (*Leuciscus idus*) and Dace (*Leuciscus cephalus*);

(ii) the development of floodplain forests to provide nesting habitats for birds species which have become very rare in the Netherlands, like Night Heron (*Nycticorax nycticorax*), Black Stork (*Ciconia nigra*), Black Kite (*Milvus nigrans*) and Osprey (*Pandion haliaetus*);

(iii) lowering of the floodplains to create reed swamps and shallow backwaters to support wildlife, amongst which rare species like Tree Frog (*Hyla arborea*), Grass Snake (*Natrix natrix*), Bittern (*Botaurus stellaris*), Purple Heron (*Ardea purpurea*) and Otter (*Lutra lutra*);

(iv) removal of river training works to restore natural river banks: the bare sand and gravel banks provide nesting opportunities for bird species like Common Tern (*Sterna hirundo*) and Little Ringed Plover (*Charadrius dubius*), on the other hand the steep banks provide nesting opportunities for Sand Martin (*Riparia riparia*) and Kingfisher (*Alcedo atthis*).

In the Economical Strategy, ecological rehabilitation plans will only be executed in combination with a lowering of the floodplain surface for clay mining purposes. Removal of clay in the floodplains restores the river dynamic processes in the floodplain, which are crucial for the development of characteristic floodplain habitats. On the other hand, the flooding risks for cities and villages adjacent to the river will be diminished (WWF, 1992; Silva and Kok, 1993). In the Economical Strategy, the total area for ecological rehabilitation in the floodplains of the Rhine and its branches is 2,300 ha, less than 10% of the total floodplain area.

In the Autonomous Development Scenario, the existing plans for ecological rehabilitation of the floodplains of the branches of the river Rhine will be executed. The total area for nature development and conservation projects is 8,260 ha, about 30 % of the total surface area of floodplains along the river Rhine in the Netherlands. The main feature of the ecological rehabilitation plans is to enhance the biodiversity along the Rhine. As a consequence the measures for ecological rehabilitation as mentioned above will be executed in all the projects.

In the Ecological and Environmental Strategy, ecological rehabilitation will cover the total surface area of the floodplains. The main channel will be accompanied by lateral channels in the adjacent floodplains. However, the total area for the development of floodplain forests is restricted in order to maintain the protection against flooding.

Lake IJsselmeer

Lake IJsselmeer is one the largest fresh water lakes in Western Europe: 1200 km^2. It is supplied with water by the river IJssel, the northern branch of the river Rhine in the Netherlands (Fig. 1). Lake IJsselmeer is a wetland of international importance (Wolff, 1989). Due to the large fish biomass and the high density of Zebra Mussels (*Dreissena polymorpha*) large numbers of piscivorous and molluscivorous birds are present in this area during the wintertime. However, marshes at the shoreland of the lake are almost entirely absent and as a consequence habitats and wildlife associated to lacustrine marshes are lacking. Therefore, ecological rehabilitation is focussed on the development of shallow wetlands around the lake. Strategies for the development of lacustrine marshes are (De Vriend en Iedema, 1995):

(i) elevating the surface level in shallow water areas to above the water level by suppletion of sand or clay;
(ii) removing the summer dykes in former marshes which are currently used as pastures;
(iii) allowing more natural water level fluctuations within the preconditions of protection against flooding and water management infrastructures.

In the Economical Strategy, ecological rehabilitation plans will only be executed in combination with dredging of the navigation channels. This results in the development of 135 ha marshland. In the Autonomous Development Scenario, the existing ecological rehabilitation plans in combination with sand extraction plans will be executed as well. The total area for nature development is 270 ha. The Environmental Strategy aims at the development of lacustrine marshlands at large scale as a result of a combination of ecological rehabilitation strategies mentioned above. The total marshland area will be enlarged with about 2,200 ha.

EVALUATION OF ECOLOGICAL REHABILITATION STRATEGIES

River Rhine

Not surprisingly, the Environmental Strategy is the best water management strategy to enhance biodiversity along the river and to meet the ecological objectives (figure 5). The lateral channels provide spawning and nursery habitats for riverine fish species and habitats for rheophyllic macrofauna species. For these species the lateral channels form refugia; the main channel has predominantly a shipping function. As a result of restoring river dynamic processes in the floodplain by reconnecting former side arms to the main channel, the available suitable habitat for fish species of stagnant water, like Pike, consists of oxbow lakes and floodplain channels which are connected only at the downstream end to the main channel. The availability of eroding river banks supports Sand Martins and Kingfishers. The floodplain forests, reed swamps and floodplain grasslands provide habitats for characteristic river birds and mammals.

In the Economical Strategy only a few lateral channels will be created and therefore the restoration of habitats for the riverine fish species and macro-invertebrates will hardly occur. Due to ecological rehabilitation projects the total area of backwater and reed marshe habitats will increase strongly. The enhanced availability of these habitats will have a positive effect on species like Pike, Tree Frog, Grass Snake, Bittern and Water Rail (*Rallus aquaticus*).

In the Autonomous Development Scenario floodplain forests will develop and lateral channels will be created. For characteristic bird species of lowland rivers, like Cormorant and Night Heron, the rehabilitation measures result in an increase of the availability of nesting and foraging habitats. The impact of the creation of lateral channels on riverine fish species and macro-invertebrates is shown in figure 5.

Lake IJsselmeer

The creation of lacustrine marshes provides habitats for species like Marsh Harrier (*Circus cyaneus*), Greylag Goose (*Anser anser*), Bittern, Spoonbill (*Platalea leucorodia*), Cormorant (*Phalacrocorax carbo*), Otter and Beaver (*Castor fiber*). It is clear that the Environmental Strategy will increase the opportunities for wetland birds and mammals strongly (figure 5). For an optimal ecological situation the total marshland area has to be, however, twice as much (Vanhemelrijk et al, 1993). In the Environmental Strategy changes in the fish stock will take place due to an improvement of the water quality and a strong reduction of the yearly catch of Pikeperch (*Stizostedion lucioperca*), Perch (*Perca fluviatilis*) and Ale (*Anguillia anguillia*). The impact of a reduction of the fish stock on piscivorous birds (e.g. cormorants, goosanders) needs no explanation.

In the Economical Strategy and Autonomous Development Scenario the development of marshes is limited to several small areas. Little improvement of the ecological conditions for marsh species can be noticed. Water quality will hardly improve and there are no yearly catch limitations for the inland fisheries. As a result, changes in the fish population are neglectable.

Ecological Development Index

The overall impact of the water management strategies on the ecological conditions of the Rhine and Lake IJsselmeer is shown in table 1. As mentioned above, the Environmental Strategy is of the best water management strategy for ecological rehabilitation. The Economic Strategy and the Autonomous Development Strategy improve the ecological conditions slightly.

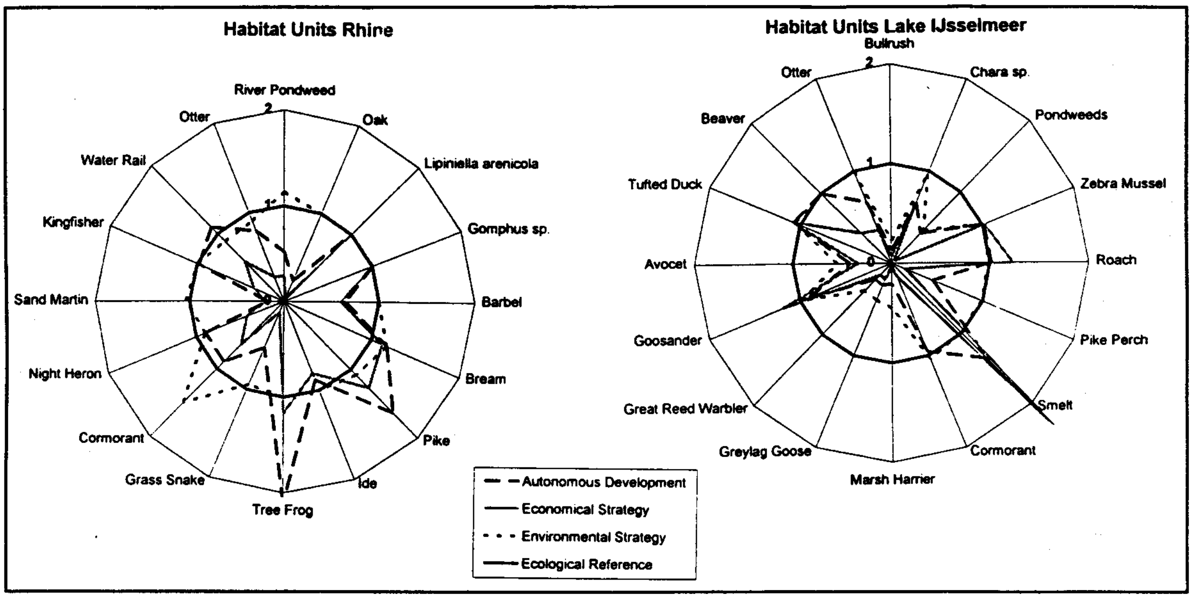

Figure 5. The AMOEBA for the Rhine (a) and Lake IJsselmeer (b) for different rehabilitation strategies. The impact of the policy alternatives is related to the ecological objectives derived from the ecological reference system.

Table 1. Score card for ecological rehabilitation of different policy alternatives on water management; ecological reference situation has an index value of 100.

water system	current situation (1995)	Autonomous Development Scenario	Economical Strategy	Environmental Strategy
River Rhine	71	88	77	98
Lake IJsselmeer	77	83	81	97

CONCLUSIONS

The framework for analysis based on the concept of HEP is an effective tool for habitat evaluation of water management strategies. The implementation of the ecotope concept and the AMOEBA approach is a very important reinforcement of HEP as a case analysis tool for water management. The policy analysis study showed that not only the impact of ecological rehabilitation measures on the availability of habitats of characteristic species can be assessed easily, but also the impact of the environmental conditions on the habitat suitability.

Future challenges for the improvement of the ecological analysis framework are related to the implementation of geographic information systems for spatial distribution effects and to the improvement of ecological modelling for temporal aspects. Ecological rehabilitation measures include also measures concerning defragmentation of existing habitats by enlarging core areas for nature rehabilitation and creating islands or corridors between existing core areas. Recent years numerous studies have been published on the meaning of spatial aspects of habitats on the opportunities for population development (e.g. Merriam, 1985; Opdam, 1991). The impact of environmental stochasticity on the habitat suitability is very important for fauna species living in dynamic ecosystems, such as river ecosystems (Verboom et al., 1992; Stalnaker et al., 1994). Therefore understanding of the temporal aspects of habitat suitability is essential to develop ecological sound water management strategies.

In any case, further development of habitat evaluation methods is an urgent need in water management policy studies. This includes the development of habitat suitability index models for all the indicator species for water management and nature conservation, the improvement and validation of existing models by field studies and habitat time series analysis. Uncertainty analysis of the results will improve the validity of the conclusions. Natural variation in population dynamics, the assumptions made and the status of the HSI-model influence the range of uncertainty in the final figures and have to be considered more thoroughly.

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TOWARDS A REGIONALIZATION OF AQUATIC HABITAT : DISTRIBUTION OF MESOHABITATS AT THE SCALE OF A LARGE BASIN

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ABSTRACT

Integrated management of water resources and aquatic ecosystems is becoming a common concern for managers as well as for researchers. The will for preservation of a good ecological functioning of aquatic ecosystems at the scale of a large basin creates needs for the study of these systems. This paper aims at describing physical habitat behaviour at the scale of the Loire basin (France). The basin has been partitionned in geomorphological regions, the validity of which we would like to test in terms of physical habitat. We tested the following hypotheses at the scale of mesohabitat :

- differences in mesohabitat composition exist between regions,
- the longitudinal structure of mesohabitat composition is different between regions,
- the factors governing composition and evolution of habitat distributions are different between regions.

We found that the four tested regions behave in different ways in terms of distribution and evolution of mesohabitats. Valley slope and stream order, the two hypothetical control variables, do not play the same role in each region. If the considered region contains mainly alluvial rivers, slope and/or order explain or predict mesohabitat distributions. If the region contains cohesive rivers, these factors do not or badly explain mesohabitat distributions.

Since predictive models can not be developped in most regions for mesohabitat distributions, it is necessary to build descriptive models at the regional scale. In addition of one predictive model, this paper provides such results for three regions of the Loire basin.

KEY-WORDS : integrated management / aquatic ecosystems / regionalization / mesohabitat / river Loire / slope / order /

INTRODUCTION

Since a few years, there is an increasing need for delineating spatial arrays in which river management could be held in an homogeneous way (Gallant, 1989). This need is consistent with the French "Loi sur l'Eau" (Water Act) (1992) which makes compulsory an integrated approach of aquatic ecosystems and water resources management. This act makes provision for the preservation of a good ecological functioning of freshwater ecosystems as part of the water resources management. But to achieve the general purpose of integrated management, we need new tools to get a global knowledge of stream ecosytem functioning at the scale of a large basin. Fish-habitat relationships have been well studied at the local scale for which efficient models and methods are available (Souchon, 1994). But the existing knowledge at larger scales is insufficient to get good prediction of habitat and fish distibutions at those scales. Longitudinal zonations are well documented (Sheldon, 1968 ; Statzner and Higler, 1986 ; Przybylski, 1993), but their predictive capacities are relatively poor. The purpose of this paper is to explore the ability of a regional approach to improve the efficiency of fish-habitat models at a large scale. The first step was to explore the regional patterns of physical habitat.

As part of this effort, an ecosystemic approach was started in France on the whole river Loire basin (117,000 km^2), which was divided in 11 geomorphological regions. The valley morphological type, according to Cupp (1989), was the key for delineating those regions.

The problem is to evaluate this macroscopic classification realised on topographic maps at a lower scale. Is the realised partition relevant in terms of fish habitat characteristics ? The choice of mesohabitats (or morphodynamic units) to test this hypothesis was natural for several reasons :

• morphodynamic units have a good hydraulic, morphological and ecological significance (Takahashi, 1994). This is mainly due to differences of microhabitat characteristics offered by each category of morphodynamic units. They have their own ecological functionings and relationships between habitat units and fish or invertebrate communities have several times been described (Bisson *et al.*, 1988 ; Lobb and Orth, 1991 ; Kershner and Snider, 1992).

• habitat types are characterised by a good homogeneity of physical habitat variables (water depth, velocity and substrate particle size). Thus, mesohabitats are a scale at which one can aggregate data gathered at a lower scale (microhabitat) with several modelising methodologies. In the past few years, many numerical models dedicated to physical habitat assessment have been developped at the microhabitat level. But most of those techniques need a hierarchical procedure to allow a good interpretation of results. For example, IFIM methodologies (Bovee, 1982 in the USA, and Souchon, 1994 in France) suggest that results should be interpreted on at least two scales : reaches and segments. Viewing IFIM results when pooled at mesohabitat scale is the first step in aggregating data from microhabitat level to reach level : this is the bottum-up approach proposed by Bovee (1982) as an application of instream flow studies.

• Mesohabitats distribution and genesis are influenced by morphodynamic processes controled by large scale factors. Climate (which controls hydrology), geology (which controls topography, lithology, sediment transport) and thus energy are varying at the regional scale (Morisawa, 1985). In this orientation, mesohabitat is a good scale to study the influence of regional-scale structures and phenomemas on physical habitat structure : this is a top-down approach. Slope and stream size are often presented as control factors of mesohabitat distributions (Richards, 1982 : Knighton, 1984), but only for alluvial systems. We were not able to find in bibliography global models or models for non-alluvial systems. Thus, the question of the validity of models including general factors such as slope an size is still present.

Our hypothesis is that belonging to a geomorphological region induces differences in the mesohabitats distribution. The aims of the study are, first, to describe the physical habitat composition at the mesohabitat scale in the four main geomorphological regions of the Loire basin. Second, we will test the hypothesis that differences exist in the longitudinal structure of mesohabitats composition. Third, we would like to identify the main factors governing the longitudinal evolution of habitat structure in each region.

METHODS

Study segments

Sampling strategy

The map of geomorphological regions which makes the basis of the sampling design was realised after a typology of valley morphology according to Cupp (1989 ; see also Naiman et al, 1992). Through the whole river Loire basin, France (117,000 km^2), twenty-two valley types have been censused visually (1/100,000 topographic maps) using the five morphometric variables proposed by Cupp (1989) : those variables are valley bottom slope, valley sides slope, channel meandering, valley bottom width and channel width.

The whole hydrographic river system was described visually with this method on 1/100,000 topographic maps. Geomorphological regions were defined as areas in which valley types distribution is as homogeneous as possible. Each region is then characterised by a specific association of a small number of valley types (2 to 6). This approach was quantified and validated on a sample of 900 segments on which the above mentioned five variables have been measured on 1/25,000 topographic maps. This work has allowed to ajust the limits of the regions in order that each of them have proper distributions of the five variables describing the valley morphology. For details about methodology and results about valley types, see Wasson *et al.* (1993) and Malavoi and Andriamahefa (1995).

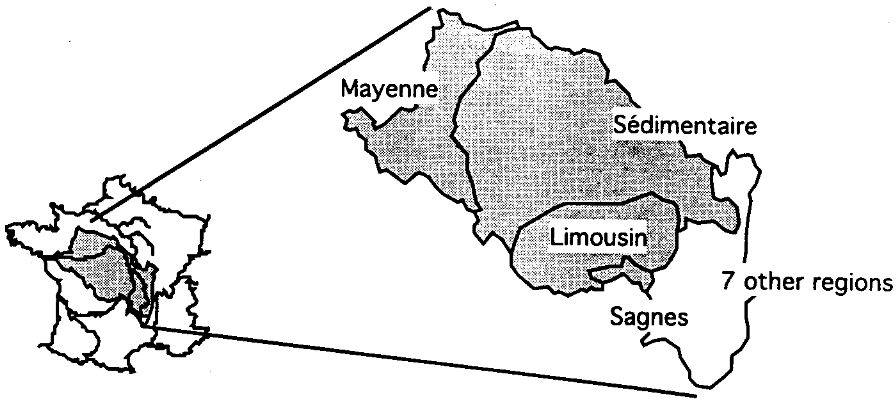

Figure 1 : the Loire basin and the four tested geomorphological regions

Out of the eleven regions obtained by this method on the Loire basin, we kept the four largest ones to test our hypotheses about morphodynamic units distributions (figure 1). We choose those regions because they represent 3/4 of the basin surface and because they are the most important regarding the management criteria. Those regions are :

• Sagnes : ancient crystalline massif, mainly granitic, part of the Massif Central. The relief is paleoglacial, dominated by smooth plateaus. Longitudinal profiles of valleys are made of steps and terraces.

• Limousin : in the same granitic massif as Sagnes. Valley sides are steeper : gorges are more frequent than in Sagnes and are intercalated with low-gradient reaches.

• Sédimentaire : very flat and tabular relief on calcareous tables. Valleys are wide and their gradient is low, although some of them may be steeper when following cuestas. The drainage density is poorer than in the other three regions.

• Armoricain : an ancient hercinian massif on granitic rocks. Although the relief is quite flat, it is possible to find areas of strong relief. Valleys are lightly incised, due to generally low altitudes.

A general characterisation of the morphology of the valleys by regions is available on table 1. We measured habitats on 196 segments of 22 rivers in the 4 tested regions of the watershed. A segment is defined as a portion of the river homogeneous by size (estimated by stream order) and valley slope.

Table 1 : General description of the valley morphology of the 196 studied reaches.

Regions	Order	Number	Altitude (m.a.s.l., mean)	Valley slope (‰, mean)	Sides slope (mode)	Valley width (times MSW, mode)	Mean Stream Width (m.)	Sinuosity
Sagnes	2	11	749	15.6	< 10%	>24w	2.7	* * *
	3	13	746	12.6	< 10%	12 to 24w	7.9	* *
	4	7	563	8.7	30 to 60%	1 to 3w	14.1	* *
Limousin	2	11	417	13.1	< 10%	>24w	2.0	*
	3	8	398	12.2	< 10%	12 to 24w	5.3	* *
	4	17	332	5.4	> 60%	1 to 3w	11.6	* *
	5	12	164	2.2	30 to 60%	1 to 3w	21.3	*
Sédimentaire	2	8	183	3.9	< 10%	>24w	2.1	*
	3	17	159	2.0	< 10%	>24w	5.8	*
	4	31	156	1.1	< 10%	>24w	10.8	*
	5	16	146	0.9	< 10%	>24w	15.9	* *
Mayenne	2	12	165	7.3	10 to 30%	>24w	2.2	*
	3	15	129	2.9	< 10%	>24w	6.1	* *
	4	15	98	0.8	< 10%	>24w	12.2	*
	5	3	90	0.3	< 10%	>24w	20.7	*

* : ≤ 1.05 ** : 1.05 to 1.25 *** : 1.25 to 1.5

Control variables

Those variables are segment slope and stream order. Both of them were measured on 1/25,000 topographic maps for each segment. Stream order was determined with Strahler method (1957) whereas segment slope is defined as the division of valley bottom length by the difference in elevation of the segment. Segments were then split into four classes of valley slope : < 1‰, 1 to 3‰, 3 to 10‰ and ≥ 10‰. When crossing the two variables, we obtained a contingency table (table 2) in which numbers of observations are not equally distributed. This is due to the fact that some combinations of modalities are not often encoutered : for example, there are very few high-slope and high-order segments. In order to reduce the amount of empty combinations, we pooled streams of order 1 and 2 and streams of orders ≥ 5.

Table 2 : Sampling design, whole Loire basin.

		Stream order			
		1 and 2	**3**	**4**	**≥ 5**
Slope classes	**< 1‰**	1	8	28	17
	1‰ to 3‰	5	21	21	11
	3‰ to 10‰	22	12	17	2
	≥ 10‰	14	12	4	1

Field measurements

We sampled 196 segments : in each of them, we choose a reach at random. We avoided reaches obviously affected by human activities : the most encountered alterations are dams, channelization, recalibration and dredging. Despite of our efforts to avoid artificialization, some of the sampled reaches might have been affected by ancient channelization works, mainly in Sédimentaire and Mayenne regions. A lot of low-gradient rivers are hydraulically controlled by ancient weirs and small dams. And in some cases, the sampled reaches might be under control of a downstream impoundment.

The length of the studied reaches had to be undimensionned to avoid scale biases. To describe reaches, we used a minimum of 35 times the mean stream width (MSW). When studying mesohabitats, Kershner *et al.* (1992) and Kershner and Snider (1992) used a minimum of 30 times the MSW. Simonson (1993) and Simonson *et al.* (1994) recommended that study reaches be at least 35 times the MSW to obtain good profiles of water depths and velocities.

Mesohabitat units were classified according to Malavoi key (1989) which is very similar to Bisson *et al.* key (1982) but adapted to high-gradient rivers of the Alps (France). We used the 7 main habitat units described in Malavoi (1989) which are : lentic channel (LEC), pool (POO), riffle (RIF), run (RUN), glide (GLI), rapid (RAP), and lotic channel (LOC). The length of each unit was recorded (nearest 0.5 m) as well as mean wetted width (5 transects across each unit, nearest 0.1 m) and average depth (several random locations, nearest 0.01 m).

Statistical analysis

We wanted to determine differences in mesohabitat induced by the three hypothetical control factors. We thus calculated the mean of each habitat type length after percent transformation on each reach. Working on percent length to avoid scale biases, we had to perform Kruskal-Wallace (non-parametric) statistical tests to determine if habitat composition was different between regions, stream orders or slope classes (Siegal, 1956).

Due to missing data in some combinations of modalities, we were not able to perform multiple-way statistical analyses such as analysis of variance. We used a factorial analysis to underscore the effects of the three hypothetical control variables on mesohabitat composition. This method is interesting in the way it allows an analysis of the composition of a whole reach, whatever the number and type of units encountered. The most appropriate analysis is the after row-percent transformation PCA (Principal Composants Analysis). The analysed table is then 196 rows (reaches) and 7 columns (habitat types). The sum of each row is 100%. This kind of analysis allows to condense most of the information on a biplot factorial map : row scores and column scores can be displayed simultaneously in order to interpret row (reaches) positions thanks to column (habitat types) vectors. This generalization of a triangular graphical display of a canonical base is described and discussed in Gabriel (1971) and Ter Braak (1983).

Factorial analysis do not allow to develop predictive models. Then, to help in understanding the quantitative role played by valley slope and stream size (measured by stream order) in mesohabitat distribution, we tested some multiple regression models on our data. We plotted percent length of fast units against slope and stream order for the whole data set and for each region. In order to simplify the interpretation of results, we had to sum the lengths of fast units (riffles, runs, rapids and lotic channels). Automated stepwise regression was used to avoid problems due to correlation between the independent variables. At each step, this kind of regression technique selects a variable to enter the model : if the partial correlation ratio is not significant, the variable does not enter the model.

RESULTS

Mesohabitats distributions

By morphological regions

Results are presented in table 3. Apart from GLI, which is the least represented habitat type on the whole basin (max 8 % in Sédimentaire), each type of mesohabitats shows significant differences in percent length between regions (p values < 5 %). LEC is dominant in Sédimentaire (61 %) and Mayenne (59 %)

regions. LOC is highly represented in Sagnes (20 %) but poorly in the three other regions. POO and RIF show the same pattern of distribution, opposite to the LEC one : more represented in Sagnes and Limousin than in Sédimentaire and Mayenne. As to the RAP, they are only present in Sagnes and Limousin.

Table 3 : Mean percent lenghts of mesohabitat units in each tested region and Kruskal-Wallace statistic

	Sagnes	Limousin	Sédimentaire	Mayenne	p value (%)
LEC	0.05	0.14	0.61	0.59	0.0
LOC	0.20	0.06	0.02	0.03	0.0
POO	0.08	0.08	0.03	0.04	2.0
RUN	0.28	0.45	0.22	0.22	0.0
GLI	0.01	0.04	0.08	0.03	86.0
RIF	0.27	0.20	0.04	0.10	0.0
RAP	0.10	0.03	-	-	0.0

By stream order

Table 4 : Mean percent lenghts of mesohabitat units by stream order and Kruskal-Wallace statistic

	1 and 2	3	4	≥ 5	p value (%)
LEC	0.07	0.32	0.41	0.70	0.0
LOC	0.10	0.07	0.08	-	0.2
POO	0.11	0.07	0.02	-	0.0
RUN	0.44	0.25	0.27	0.23	0.0
GLI	0.07	0.06	0.04	0.03	71.7
RIF	0.19	0.16	0.16	0.04	0.0
RAP	0.01	0.08	0.01	-	77.4

RAP and GLI show no significant differences between stream order. GLI can be said to be equally represented along the longitudinal gradient, but the result for RAP may be influenced by the low number of such encountered habitat units. Habitat diversity is the highest in order 1 and 2 : apart from RAP, all habitat types are represented at at least 7 %. Orders ≥ 5 exhibit a very poor diversity : only LEC and RUN are well represented in those downstream reaches.

By valley slope

Table 5 : Mean percent lenghts of mesohabitat units by valley slope and Kruskal-Wallace statistic

	< 1‰	1‰ to 3‰	3‰ to 10‰	≥ 10‰	p value (%)
LEC	0.79	0.52	0.04	0.09	0.0
LOC	0.01	0.04	0.11	0.12	0.0
POO	0.03	0.04	0.08	0.07	0.7
RUN	0.12	0.25	0.48	0.31	0.0
GLI	0.03	0.07	0.06	0.01	78.0
RIF	0.02	0.08	0.22	0.26	0.0
RAP	-	-	0.02	0.13	0.9

Table 5 shows that valley slope is a control factor of mesohabitat distributions, as well as stream order or geomorphological region. Only GLI shows no significant differences, as for the other factors. Lotic habitat types are mainly found in high slopes : RIF and LOC, and above all RAP, are almost pledged to slopes ≥ 3 ‰. Channels, which have the same geometry, are mainly lentic in slopes ≤ 3‰ and lotic in slopes ≥ 3‰.

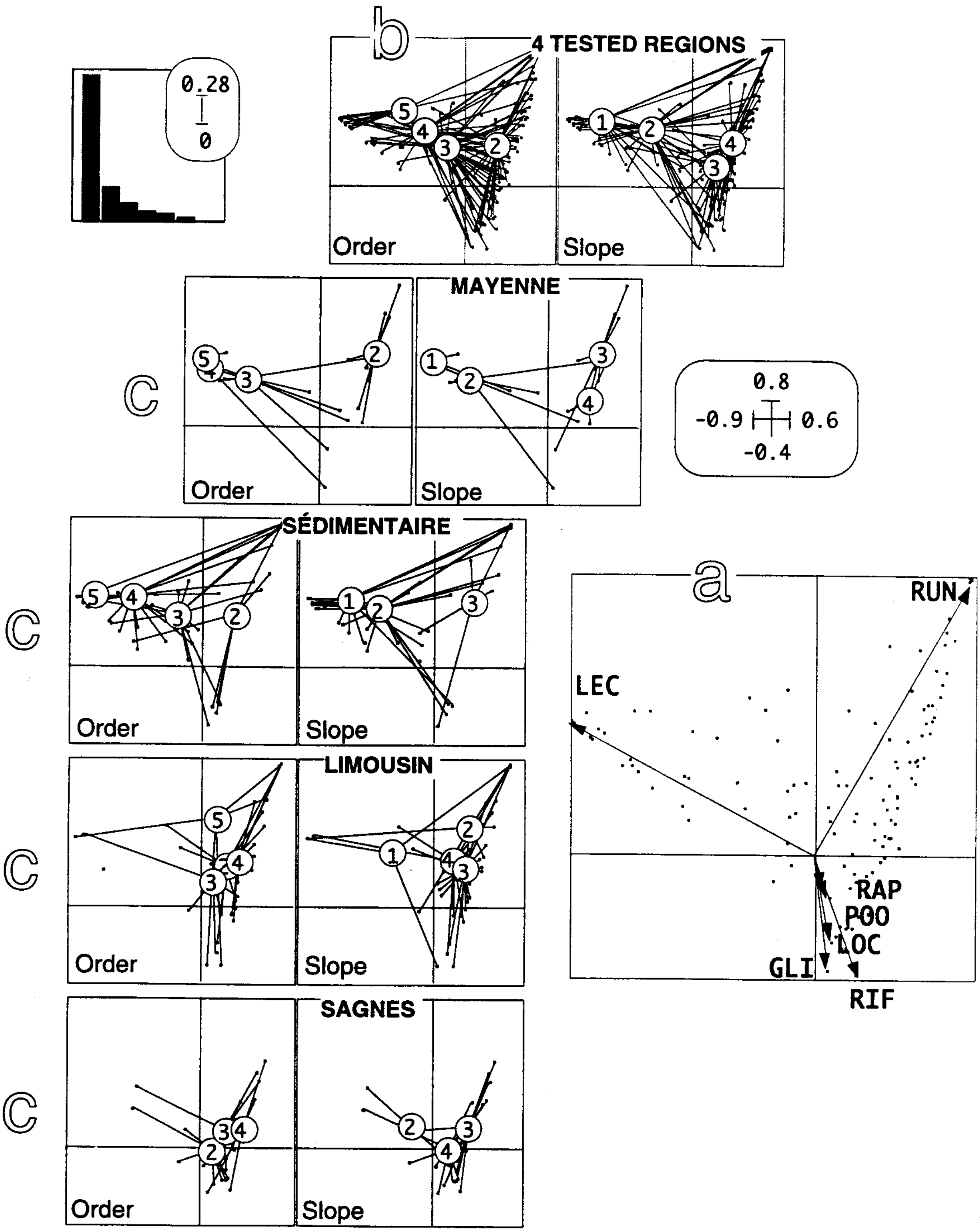

Figure 2 : F1-F2 factorial maps of the after row-percent transformation PCA

a : whole data set (196 reaches) + vectors of variables (i.e. habitat types),
b : same map as a, reaches splitted by categories of slope and stream order,
c : same map as b, only one region represented on each figure.

Factorial analysis

Given the distribution of eingen values, we must only take into account the first two axes. On figure 2a, the whole data set has been plotted : reaches and variables are represented on the same F1-F2 factorial map. The closer a station is from the end of a vector, the higher the percent length corresponding to the habitat type is. The factorial map is organised by some variables (i.e. habitat types) which are LEC, RUN and to some extent RIF and GLI. Stations are separated into three main groups : dominated by LEC, dominated by RUN or more diverse.

On figure 2b and 2c, the stations have been scattered into categories of valley slope or stream order for each region. Sagnes and Limousin regions exhibit similar patterns regarding slope and order. The effect of those control variables is low : the ordination of the stations along gradients of slope or order is not very pronounced. For Sagnes region, the stations are plotted on the center of the map and this means that habitat diversity within a station is high. To a lesser extent, this is true for Limousin region, but the proportions of LEC and RUN are higher in this region : diversity is reduced, especially for high orders and low slopes.

According to the same criteria, Mayenne and Sédimentaire are showing quite poor within-station diversities : LEC and RUN are dominant to the detriment of the other habitat types. But unlike the two other regions, the mean pattern of habitat distribution varies regularly along gradients of slope and order. In these regions, LEC percent lenght is inversely proportional to RUN : it increases as slope decreases and order increases. Conversely, RUN percent length increases with slope and against order.

Regression models

Although R^2 is low (R^2=0.24), the regression model performed on the whole data set is highly significant (p<0.01%). The percent length of fast units increases with slope and decreases with stream order. But those general results become different when studying each region individually. Table 6 shows that stepwise regression model is highly significant for Sédimentaire and Mayenne regions. The model is not significant for Limousin whereas Sagnes model is just significant (p=4.74%). The role played by slope and order to explain the percent length of fast units is not the same in the 4 regions. If the regions are considered together, the two factors enter the model. It is not the case for three regions. Slope and order do not explain mesohabitat variability in Limousin. In Sagnes, stream order enters the model and Sédimentaire percent length of fast units is partially explained by slope. The higher R^2 is observed in Mayenne (0.69), the only region in which slope and order are both took into account by the regression model. This value of R^2 is high enough to consider this model as a predictive model.

Table 6 : Stepwise regression results : percent length of fast units vs slope and/or order

	Slope	Order	final R^2	p (%)	Model
Sagnes	-	*	0.12	4.74	% fast = 0.59 + 9.4e-2 order
Limousin	-	-	-	-	-
Sédimentaire	*		0.20	< 0.01	% fast = 0.11 + 0.11 slope
Mayenne	*	*	0.69	< 0.01	% fast = 0.96 - 0.24 order + 4.25e-2 slope
Whole basin	*	*	0.24	< 0.01	% fast = 0.65 - 7.73e-2 order + 2.24 e-2 slope

* : variables entering the regression model

DISCUSSION

The first analysis of the present data set clearly shows that each region is characterised by its own mesohabitat units distribution. This is the answer to our first hypothesis. On the one hand, Sagnes and Limousin are regions of medium altitude : hence, valley slope distribution is wide and centered around high values (12 ‰). Those regions tend to be more diverse, due to the wide range of slope values. Whittaker and Jaeggi (1982) reported that some types of mesohabitats are found only in mountainous regions due to the high slopes encountered. On the other hand, Sédimentaire and to a lesser extent Mayenne, are regions of low gradient. Thus, the slopes are generally more gentle : we found less habitat types than in mountainous regions. This result is not surprising, due to the differences in morphological characteristics of the valleys exhibited by each region. Although expectable, this result had to be quantified, and it is now available for management concern. The aim of the first works on the Loire basin was to define regions the limits of which had a good aptness in terms of management. The results of this paper show that the proposed regions have different physical habitat functionings and must then be managed in different ways.

Geomorphological regions is a factor affecting mesohabitat distributions. But valley slope and stream order might be sufficient to predict those distributions : the observed differences could be only due to differences in slope distributions between regions for example. Hubert and Kozel (1993) reported that channel slope and stream size were two determinants of physical habitat features. Kershner and Snider (1992) found that mesohabitat distributions differed between rivers belonging to three different types based on channel gradient among other parameters. The underlying question is to evaluate the improvement in terms of description and prediction brought by the regional scale. A first part of the answer is given by the second analysis. The expected upstream-downstream evolution of patterns of mesohabitat distributions is not obvious for all regions. When regarding the effects of slope and stream order on mesohabitat patterns, the behaviours of Sagnes and Limousin regions are opposite to Mayenne and Sédimentaire. The first ones exhibit no reactions to slope and order, whereas the second ones evolve along the gradients of the presumed control factors. Those differences in upstream-downstream evolutions are the justification of developping quantitative models which would enable predictions with a few control factors.

The patterns of mesohabitat distributions, i.e. the way habitat types are associated, do vary at the regional scale, as shown by the second analysis. The results of regression models prove that valley slope and stream size do not play the same role in each region. The factors governing habitat distributions are not the same at the scale of a large basin.

The rivers of Sédimentaire and Mayenne regions are mainly made of alluvial deposits. A lot of them show well developped meander processes. Then, morphodynamic units distributions is generated by the equilibrium between erosion and deposition and between carrying capacity and solid load. Successions of pools and riffles is more predictable for this kind of non-cohesive substratum streams than for rivers flowing on the underlying bedrock. This is the main reason why slope and/or stream order, a means to measure stream energy, better explain habitat distributions in Sédimentaire and Mayenne than in Sagnes and Limousin. Since Leopold and Wolman (1957), who related for the first time that pools and riffles were spaced on an order of 5 to 7 widths, many papers have explored the description and genesis of riffle-pool sequences in alluvial rivers. Yang (1971) and Keller and Melhorn (1973, 1978) developped models for the formation of such channels features. Those region meet the assumptions of those general morphodynamic models.

The slope of most of the rivers in the Sagnes and Limousin regions is usually steep as well as valley sides slope. Although high, due to high slopes, river energy is not sufficient enough to reshape longitudinal profile and sinuosity. Geological substratum is granitic and mean substrate particle size is coarse, even in low-slope valleys : as valley sides slope is high, the supply of particles of large diameter is sustained. Valley bottom is made of steps and terraces mainly generated by topographic accidents. Moreover, breaks in slope are frequent. In these regions, distributions of morphodynamic units can not be explained with models developed for alluvial rivers : they must be largely influenced by underlying rock or topographic accidents. This is the reason why slope and stream order weakly or do not explain distributions of habitat features in those mountainous regions.

This study shows that at the scale of a large basin, it is not possible to explain habitat distributions for a whole basin. Regional patterns do exist in habitat distribution. Moreover, the available models describing mesohabitat distributions and genesis are accurate in only one region which represents a small part of the basin (< 20%). This reinforce the usefullness of the descriptive models presented above for the three other regions, in which predictive models can not be developped.

Artificialization may be in some cases a forgotten control factor, despite of our efforts to avoid affected reaches. In Sagnes and Limousin, this effect must be non-existent : existing channelization is always obvious and then can be avoided. And the slope is high enough to avoid the hydraulic control of a whole segment in case of damming. In Mayenne region, small dams and weirs are quite numerous. But the strong effect of slope on mesohabitat distributions prooves that the "natural" morphodynamic behaviour is still present. In Sédimentaire region, some of the reaches might have been affected by ancient channelization or far downstream damming. But those effects can at worst only reinforce the regional tendency, which is a high percent lenght of LEC.

CONCLUSION

The usefulness of the approach described in this paper is demonstrated at the scale of a large basin for management as well as for ecological concern. This regionalization of physical habitat could be linked to regional patterns of fish assemblages or aquatic ecosystems, the validity of which having been prooved several times (Larsen *et al.*,1986 ; Whittier *et al.*, 1988). The statistic description realised here is a good basis for the characterisation of physical habitat for fish or invertebrates, because relationships between mesohabitat availability and its use by communities is getting well described (Bisson *et al.*, 1988).

This paper shows, as others, that slope and order do play a role in mesohabitat distribution and composition. They are control factors of available physical habitat. But this role is varying in the four tested regions of the Loire basin. In one of them, mesohabitat composition can be estimated, knowing slope and order. In two others, those variables can explain these distribution. In the fourth, slope and order neither explain nor predict mesohabitat percent length. This prooves that geomorphoregions have a good ability to descriminate river functioning. Developing descriptive models on a regional basis is then of a great importance.

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A REFERENCE FRAMEWORK FOR THE INTEGRATED ECOLOGICAL MANAGEMENT OF THE SAINT-CHARLES WATERSHED, QUÉBEC, CANADA

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ABSTRACT

Located in the heart of the Québec Urban Community (QUC), the Saint-Charles River watershed is characterized by great ecological diversity (geology, geomorphology, plant life, landscapes) and a wide range of land uses (dense urbanization, industrial activity, farming, and uninhabited forests). Moreover, the river provides almost 300 000 people with their water supply.

In 1992, at the request of the Québec Urban Community, the Québec department of the environment and wildlife (MEF) produced an ecological reference framework for all 520 km^2 of the Saint-Charles River watershed, to be used as a basic tool for formulating QUC management proposals and policies. This ecological reference framework is based on a conceptual model in which the factors controlling ecological function, such as bedrock geology, topography, landforms, and climate, are located upstream of the aquatic ecosystem. The territory is analyzed taking a holistic and hierarchical approach that proposes maps and typologies based on four levels of resolution: the ecodistrict (1:250 000); the topographic complex (1:100 000); the topographic entity (1:50 000); and the topographic element (1:20 000). Mapping units delineate natural spatial entities which are easily recognizable in the field and usually expressed in terms of physical geography.

When processed by geographic information systems, this information is analyzed in terms of land capability, terrain sensitivity and fragility. When socioeconomic and land use data are factored in, it is possible to produce a complete ecological profile of the units, enabling the formulation of land and resource management practices tailored to the needs and requirements of each unit. This paper will describe how the ecological reference framework was used to produce genuine hydroecological units.

KEY WORDS: Ecological management / ecological mapping / ecosystem / water system / socioeconomic / watershed

BACKGROUND

The Saint-Charles River watershed covers 520 km² and includes all or parts of the municipalities of Québec City, Charlesbourg, Lac Saint-Charles, Saint-Adolphe. The Saint-Charles River, which issues from Lake Saint-Charles, provides some 300 000 people with their water supply. The lower third of the basin is heavily urbanized, the middle third has a high concentration of suburbs, and the upper third (northern) is forested.

With a view to conserving water quality and quantity, the Québec Urban Community mandated the Québec department of the environment and wildlife to devise and produce an ecological reference framework that would enable urban planners to more effectively orchestrate urban development so as to harmonize the varying and often conflicting land and resource uses. These imperatives include controlling flooding and logging, conserving environment quality, and curbing the development of potentially harmful infrastructures.

Traditionally, pollution and species habitat analyses of aquatic environments are carried out locally. Very rarely are they approached from a broader base, that is, at the watershed level, integrating the permanent physical parameters of the terrestrial environment, as is the case in the ecological mapping developed by MEF (Ducruc *et al.*, 1995a; Gerardin *et al.*, 1995). The purpose of this paper is to describe the contribution of ecological mapping to our understanding of water systems, land use planning, and water management. We will also briefly touch on the integration of the socioeconomic parameters enabling management units to be defined.

BASIC CONCEPTS

The main purpose of the SCW ecological reference framework is to promote a broad understanding of the dynamics of the watershed. It is founded on a unique concept, i.e., studying terrestrial and aquatic environments from an ecosystem-based approach (Tansley, 1935). The defining features of this concept are (Ducruc *et al.*, 1995):

- a holistic approach to the environment whereby the whole takes precedence over individual parts;
- a hierarchical system of levels of resolution which moves from the general to the specific and in which the higher levels control the lower levels;
- a spatial organization whereby each level of resolution has its own heterogeneity.

Terrestrial and aquatic ecosystems are closely interconnected through the circulation of water and organic and mineral matter, making the water system dependent on the basin (Amoros and Petts, 1993; Wasson *et al.*, 1993) and the floodplain controlled by the hydraulic and hydrological forces of the watercourse. Furthermore, they also have certain control factors in common, such as climate, geology, relief, and geomorphology. The impact of human activity carried out in the watershed on the aquatic environment and, vice versa, the impact of modifications in the aquatic environment on adjacent areas (semi-aquatic and terrestrial), bring these functional links clearly into focus. These principles, along with the equally critical, and perhaps more crucial one, of studying "the factors that control ecosystem function rather than focusing on the restricted results of biological population studies" (Wasson *et al.*, 1993) [Translation] are the centrepieces of the SCW ecological reference framework.

THE SCW ECOLOGICAL REFERENCE FRAMEWORK

An ecological reference framework is a set of decision-making tools for land use planning and resource management in the form of mapping and typology of the nature, capability, and limitations of the area under study. Mapping serves as a spatial framework for typologies without depending on them. Ecological mapping is based on the principle that the nature and spatial organization of ecosystems is controlled by physical geography (relief, geomorphology, and hydrography). Mapping delineation is therefore based on the identification of major physiographical shifts indicating significant ecological changes at the level of resolution being considered.

We began by producing the ecological reference framework for the terrestrial environment and then integrated the hydrological parameters required for understanding the Saint-Charles water system. However we are convinced that in future projects both aspects should be considered together.

Land Mapping

Several levels of resolution were mapped in the SCW, from the natural province (1:4 000 000) to the topographic element (1:20 000). The topographic complex mapped on SPOT stereoscopic images at 1:100 000 was the level of choice for defining natural management units. A topographic complex is a land parcel generally presenting highly varied relief features, best mapped at a scale of 1:100 000 (Figure 1), and a surface area of roughly 10^1 km^2. Thirty-six topographic complexes, defined by their physiography and geology, are described in terms of bioclimate, relief, surficial deposits (origin, texture, stoniness, thickness) and soil moisture content (Figure 2, Table 1).

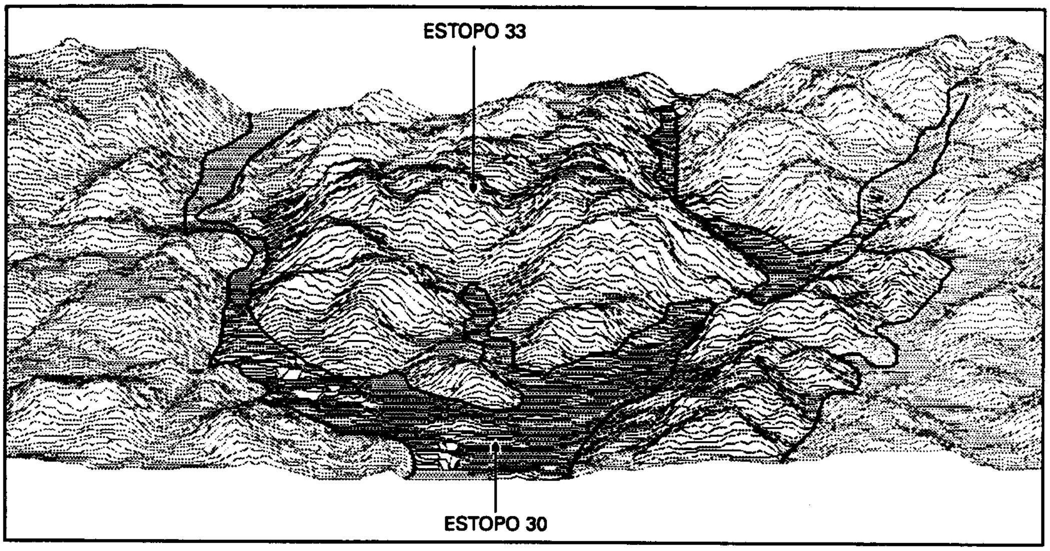


Figure 1: Digital elevation model illustrating the level of resolution for the topographic complex.

Integration of Hydrological Parameters

In keeping with the approach proposed by Cupp (1989) and subsequently adopted by Wasson *et al.* (1993) and Andriamahefa (1994), we characterized valley sections based on their cross-sectional profile and their level in Stralher's hierarchical system (1957). The valley section, the first linear hierarchical level, appeared to be controlled by geological and, by extension, physiographic forces (Wasson *et al.*, 1993). This was confirmed when we characterized the topographic complex by valley type (Table 1).

THE SOCIOECONOMIC FRAMEWORK

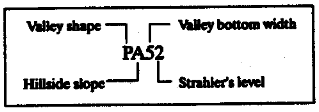
The ecological reference framework is used to map and describe distinct territorial units based on the nature and organization of stable ecosystem elements (physical environment supporting biological components). However, these units are subject to more or less intense human pressure which in turn influences ecosystem function. To understand the interaction between ecosystems and social systems, we intend to map social systems in such a way as to explain the nature and impacts of human activity on the SCW. As a preliminary step, we characterized the topographic complex based on five categories of area use.

Table 1: Characterization of a few topographic complexes.

ECOLOGICAL UNITS					PHYSIOGRAPHY					HYDROGRAPHY	
					RELIEF						
Natural region (1)	Land district	Topographical complex	Bioclimate (2)	Geological bedrock (3)	Type (4)	Morphology (5)	Slope (6)	Superficial material (7)	Soil drainage (8)	Network density (9)	Valley type (10)
B2	02	02	ERS-Ti	S3	PN	--	A	5A	2	2	PA52(7)
B2	03	03	ERS-Ti	S2	TE	ON	A	5C	2	1	PA51(7)
B2	04	04	ERS-Ti	S1	TE	--	A	5D	2	3	--(1)
B2	04	05	ERS-Ti	S1	CU	--	A	5M	3	2	VC31(3)
C8	05	06	ERS-Ti	G2	BCS	MA	C	1A	2	2	VE21(3)
C8	06	11	ERS-Ti	G2	PN	--	A	4BL	3	1	PB52(7)
C8	06	12	ERS-Ti	G2	PN	--	A	6C	2	1	VB41(5)
C8	06	13	ERS-Ti	G2	TE	--	A	6C	2	3	VD31(2)
C8	06	14	ERS-Ti	G2	TE	BO	A	2B	2	2	VB41(4)
C8	06	08	ERS-Ti	G2	TE	BO	B	1Y	2	3	VD41(2)
C8	06	09	ERS-Ti	G2	BSC	MO	C	1A	2	3	--(1)
C8	06	15	ERS-Ti	G2	MC	--	C	1A	2	3	--(1)
C8	07	16	ERS-Ti	G2	CT	--	A	6C	2	2	VC41(4)
C8	09	19	ERS-Ti	G2	FV	--	A	2B	2	1	VC42(5)
C8	09	22	ERS-Ti	G2	FV	BO	B	4BL	2	1	LC42(6)
C8	09	20	ERS-Ti	G2	BCS	MA	B	1A	2	2	VD31(3)
C8	09	23	ERS-Ti	G2	BCS	MO	C	1A	2	2	VC31(3)
C8	11	30	ERS-Boj	G2	FV		A	2B	2	1	UE31(5)

(1) (Ducruc *et al.*, 1995b) B2= middle St-Lawrence plain; C8= lake Jacques-Cartier massif.
(2) ERS-Ti= sugar maple-basswood domain; ERS-Boj= sugar maple - yellow birch domain.
(3) S1= Neuville calcareous rocks; S2= shales; S3= schist; G2= igneous metamorphic rocks.
(4) BCS= complex of low hills; CT= knoll, CU= hollow; FV= valley bottom; L= lake; MC= medium hill; PN= plain; TE= terrace.
(5) secondary morphology BO= bumpy; MA= hilly; MO= rolling; ON= undulating.
(6) A= 0-5%; B= 6-15%; C= 16-30%; D= 31-50%.
(7) 1A= regional till; 1Y= reworked till; 2B= fluvio-glacial sand and gravel; 4BL= lacustrine silt; 5A= marine clay; 5C= estuarian silt; 5D= tideland clay; 5M= marine silt; 6C= holocene beach sand and gravel.
(8) 2= good; 3= poor.
(9) 1= dense; 2= moderate; 3= sparse.
(10)

Valley shape	Hillside slope (%)	Valley bottom width (m)	Strahler's order
V = V shape	A = 0-2	1 = 0-50	1 = 1 or 2
U = U shape	B = 3-5	2 = 51-100	2 = 3 or 4
G = gorge	C = 6-15	3 = 101-300	
P = flat	D = 16-30	4 = 301-1000	
L = large lake	E = 31-60	5 = >1000	



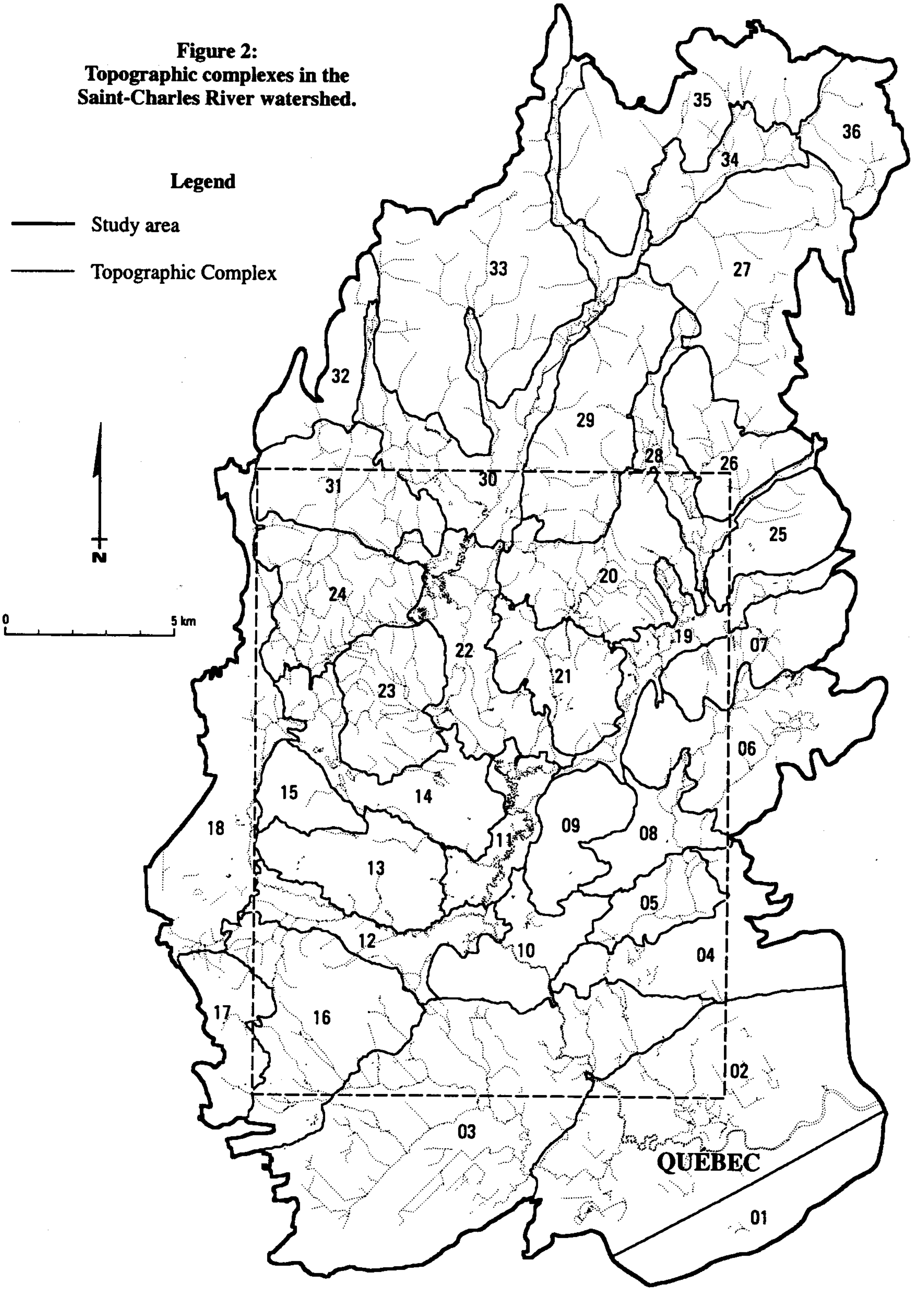
The figure in ( ) refers to map D in figure 3.

Figure 2:
Topographic complexes in the Saint-Charles River watershed.

Legend

Study area

Topographic Complex



SUPPORT CAPABILITY OF THE NATURAL ENVIRONMENT AND INTEGRATED MANAGEMENT UNITS

Ecological mapping enables distinct spatial entities to be delineated from adjacent ones based on their ecological features. Several polygons can have identical ecological features, while others are sufficiently alike to enable similar results to be obtained with regard to a specific objective. It is therefore necessary to translate these ecological units into reference units for land management. Natural management units (NMU) are spatial entities characterized by productive capacity, support capability, and environmental sensitivity per se. These properties are derived and interpreted from the nature of the ecosystems at the level of resolution chosen. NMUs are determined by classifying topographic complexes according to the two basic ecological characteristics of climate and geology, as well as the values obtained for 1) land capability in terms of forest and agricultural production, 2) likelihood of urbanization based on the feasibility of building roads, water supply and sewer systems and individual water purification sites, 3) groundwater sensitivity to pollution, and 4) the nature of the water system (type of valleys, density, size of floodplain). Figure 3 shows four of these values, while Figure 4 and Table 2 illustrate some of the natural management units.

NMUs come under pressure from social systems. The integration of these two organizational levels is necessary to provide decision makers and land use planners with a comprehensive view of all the relevant issues, constraints, and proposed solutions. The integration of social, economic, and ecological imperatives may appear more feasible for traditionally structured units such as rural areas. In recent decades we have seen technological development run rampant at the expense of the ability of the environment to support it, with dire environmental consequences which are only too well known. Despite this, we have only to look at how human settlement and activities are organized in the Saint-Charles basin to realize that they are nonetheless closely linked to the natural organization of the area, especially with regard to topography and surficial deposits. Figure 5 and Table 3 present the main preliminary social system categories and integrated management units (IMU) where ecological and socioeconomic limits converge. IMU VI comprises two topographic complexes with similar relief features, dominated by the same geomorphological material and a moderate-to-underdeveloped water system, features which make the water table moderately sensitive to contamination. In addition they are both undergoing a dynamical urbanization. IMU VII, whose water table is also moderately sensitive, has no potential for urban development because of its steep slopes and thin soil layers, although there are some signs of spot urbanization wich, in addition to the fact that these territory have already undergone urbanization, will bring soon or later environmental problems. Forests and landscapes are its main assets. IMU VIII is distinctly different from the units VI and VII owing to its highly sensitive water table, strong capacity for urbanization, and well-developed water system which makes it highly attractive (recreation and landscapes) but also extremely vulnerable to industrial and urban encroachment. Any further development undertaken in this unit should take this duality into account.

DISCUSSION AND CONCLUSION

Ecological mapping provides comprehensive knowledge of the area by identifying ecological limits most often expressed by the topography resulting from the area's geological history, bedrock, and geomorphology. Subsequent integration of the water system's descriptive elements (valley type) showed a strong correlation with the TC level of resolution. Thus, for 45 pairs of consecutive valley sections included in a topographic complex, the section change coincided in 80% of cases with the limit of a topographic complex. Moreover, the topographic complexes all had a limited number of valley types (Table 1). This is no surprise; what is astonishing, however, is that to date, very little use has been made of ecological mapping. Furthermore, there have been few attempts to show the relationship between water systems and the terrestrial environment of their watershed on which these systems depend.

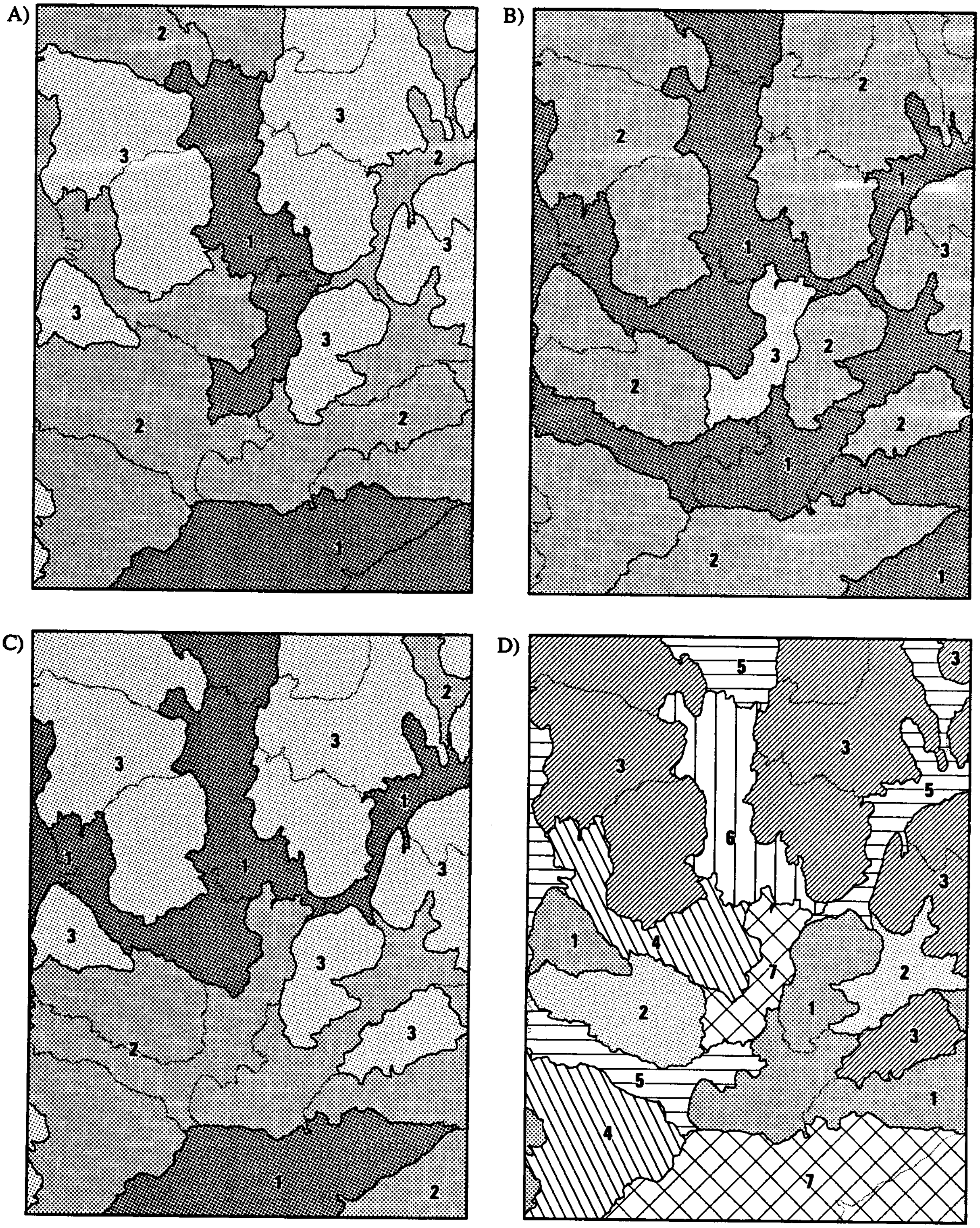


Figure 3 : A sampling of values for topographic complexes (TC):
A) agricultural potential; B) likelihood of urbanization; C) groundwater sensitivity to pollution; D) dominant valley type and drainage density. The legend for maps A, B, and C is found in Table 2; the map D legend is found in Table 1.

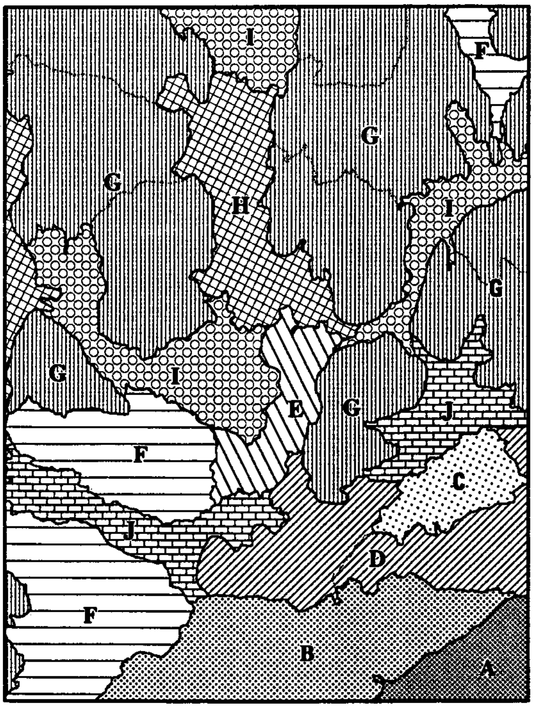


Figure 4: Natural management units (NMU) in the area around Lake Saint-Charles.

Table 2: Description of natural management units (NMU) in the area around Lake Saint-Charles.

RELIEF Entity (1)	Slope (1)	Landform (1)	Valley type (1)	Topographical complex	Ground water pollution potential (2)	Urbanisation capability (3)	Agricultural capability (3)	Natural management unit
PN	A	5A	PA52	02	3	2	1	A
TE	A	5C	PA51	03	2	1	1	B
CU	A	5M	VC31	05	2	2	2	C
TE	A	5D	--	04	1	2	2	D
PN	A	4BL	PB52	11	3	2	1	E
TE	A	6C	VD31	13	2	2	2	F
CT	A	6C	VC41	16	2	2	2	F
BCS	B	1A	VD31	20	2	3	3	G
BCS	C	1A	VE21	06	2	3	3	G
BCS	C	1A	VC31	23	2	3	3	G
BCS	C	1A	--	09	2	3	3	G
MC	C	1A	--	15	2	3	3	G
FV	B	4BL	LC42	22	1	1	1	H
FV	A	2B	UE31	30	1	1	2	I
FV	A	2B	VC42	19	1	1	2	I
TE	A	2B	VB41	14	1	1	2	I
PN	A	6C	VB41	12	1	2	2	J
TE	B	1Y	VD41	08	1	2	2	J

(1) Refer to table 1 legend. (2) Preliminary evaluation by the DRASTIC method of the first aquifer (confined or unconfined) met (Aller *et al.*, 1987, Champagne, 1990); drastic class 1= 150 to 190, 2= 100 to 149, 3= < 100. (3) Capability 1= high, 2= moderate, 3= low.

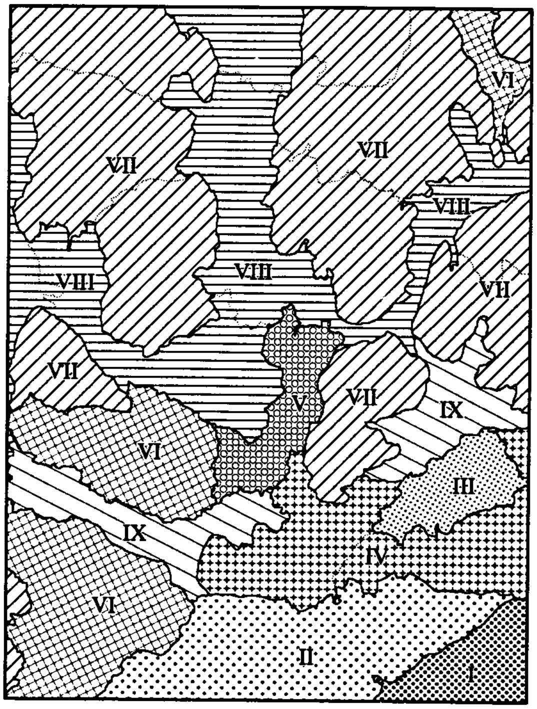


Figure 5: Integrated management units (IMU) in the area around Lake Saint-Charles.

Table 3: Integrated management units (IMU) in the area around Lake Saint-Charles.

TOPOGRAPHICAL COMPLEX	02	03	05	04	11	13	16	20	06	23	09	15	22	30	19	14	12	08
Ground water pollution potential (1)	3	2	2	1	3	2	2	2	2	2	2	2	1	1	1	1	1	1
Urbanization capability (1)	2	1	3	2	2	2	2	3	3	3	3	3	1	1	1	1	2	2
Agricultural capability (1)	1	1	2	2	1	2	2	3	3	3	3	3	1	2	2	2	2	2
Natural management unit	A	B	C	D	E	F	F	G	G	G	G	G	H	I	I	I	J	J
Socio-system (2)	UI	UB	F	U	B	B	B	N	N	N	N	N	B	B	B	F	B	B
Integrated management unit	**I**	**II**	**III**	**IV**	**V**	**VI**	**VI**	**VII**	**VII**	**VII**	**VII**	**VII**	**VIII**	**VIII**	**VIII**	**VIII**	**IX**	**IX**

(1) Refer to table 2 legend.
(2) B= suburban and urbanizing fallow; F= fallow; I= industrial; N= natural environment, mainly forest; U= densely urbanized.

This approach by ecological unit and valley section is the first level in understanding the organization and nature of habitats. For example, in the Saint-Charles River watershed, NMUs located in cold, rainy regions presenting rugged landforms and acidic bedrock, support only highly oxygenated, oligotrophic lotic habitats. NMUs with a gentler relief on bedrock composed of limestone and fine material feature slower-flowing watercourses with greater turbidity but which are richer and more diverse from a biological perspective.

These NMUs are under heavy human pressure due to the high productivity of their terrestrial and aquatic environments, their hydrolicity, and their urbanization potential. While these areas have reached the saturation point in terms of industrial and urban development, other areas which have remained more natural but are nonetheless vulnerable, are being sought out. This is why it is crucial, indeed urgent, to integrate ecosystems and social systems in order to "spatialize, interpret and, if possible, foresee human intervention" (Wasson *et al.*, 1993) and manage the different land and resources uses in a sound manner.

The hierarchical system containing the topographic complex chosen here enables the same questions to be addressed but at different levels. The ecodistrict generally expressed at a scale of 1:250 000, and even the natural region (1:1 000 000), both delineated and characterized by the same physical parameters underlying the organization and nature of water systems, are particularly suitable for analyzing vast areas.

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ELABORATION OF AN ECOLOGICAL INFORMATION SYSTEM WITH SPATIAL REFERENCE EXAMPLE OF THE ANTI-LARVAL MOSQUITOS CONTROL

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ABSTRACT

Managing a landscape means predicting its evolution according to internal modifications or the appearance of external perturbations. To archieve that you must be able (i) to assess and follow the evolution of the significant parameters; (ii) to establish functionnal relationship between the constituent elements and the flux of matter and energy; (iii) to anticipate the effects resulting from the development at different times (dams for example); and at last (iv) to develop mathematical models for reliable simulations.

For a few years, under pressure of the computer technologies, systems able to stock, manage and use geographical information have been developed. In fact the objective of these systems is to process any type of information spatially referenced and not only geographical informations. Then they become adapted tools for landscape management. These sytems are called Geographic Information System and are used for many different themes, from the district to the continent, and the solutions may be relevant in one case but not in the others. However in such a complexe field as environment they only give an incomplete answer to the landscape manager's questions.

In this paper the authors present a EISSR based on a conceptual model conceived from a global and systemic approach of the field of application. With this EISSR it is possible to go farther than the simple superimposition of the spatial data. They turn towards more complex operations of spatio temporal analysis (taking into account previous facts and elaborating evolution predictions) allowing a better understanding of the hydrosystem dynamic according to the variations of the matter and energy fluxes. An example of how this EISSR is used in the case of mosquitos larval population control in the Rhône valley is presented.

KEY-WORDS: Landscape Ecology / GIS, Floodplain / Wetlands / Environmental planning / Rhône river / France

INTRODUCTION

To elaborate a plan of management , it is necessary to aquire homogeneous quantitative and qualitative information for the whole concerned territorry. To achieve that, French organisations such as national and regional parks or natural reserves,which are responsible for the territory management have developped programs to aquire these informations in order to produce maps. These maps objective is to visualize a given situation at a given moment and to predict different scenarios of intervention. Data haven't been collected homogeneously in these different organisations, from the minimum information that is to say an inventory of the flora and the fauna to a very complete and systematic inventory. Since 1966, when we began the first investigation, we have been thinking about the needs in the mapping field in the light of all the management problems which have occured during the last three decades. Managing a landscape means managing populations and ecosystems but also taking into account the evolution at different times, crisis situations (floods for example) and innovation phenomena (appearance of new biological sets).

The object of this paper is to present the method our laboratory has followed to develop a system of information on hydrosystems development. This approach depends on two main elements:(i) the lab experience in the field of ecological and phytoecological mapping and (ii) the technical competences for the development of image and spatial analysis tools.

INTEREST AND LIMITS OF PHYTOECOLOGICAL MAPPING.

The phytoecological map is relevant to reveal the different relationships between the biotic parameters (types of structure, specific richness) and hydrologic parameters (groundwater depth and range of variations, flooding length), geomorphological parameters (alluvial forms are getting younger), pedological ones link between mineral elements and organic matters and hydrochemical ones (availibility of biogene elements). The map has different types of functional units, each one becoming individual thanks to practical details which affect the functioning of biogeochemical cycles.

The data collected must allow the elaboration of models which give details on the structure, the functioning and the dynamic of the territory concerned but which also specify the relations with its environment in order to identify the connectivity, permeability and percolation phenomena.The implementation of the eco-complex (Blandin and Lamotte, 1988) concept has the advantage of explaining the relations between the elements which compose the vegetal cover (that is to say the different herbaceous and ligneous ecosystems) and the fluxes of water, sediments, organic matter, diasporas nutrients as well as the terms of auxiliary energy clearing.

High scale ecological maps (from 1/5000 to 1/25000) have been established for the whole floodplain between Geneva and Lyon. The methods used to make such documents (Ozenda, 1986; Pautou, 1985)are well known and it doesn't seem to be necessary to remind their principles. They propose a functional typology, give an evaluation of the quantitative ratios, between the different taxons and the assembling terms and give their position to the circuits followed by the flux of matters and energy.

In the Rhone hydrosystem, the ecosystems have privileged positions in a tridimensional space which has (i) a longitudinal component (from upstream to downstream) which corresponds to a gradient reduction, an increase in the quantity of energy cleared, a strong representation of materials with a small diameter,

(ii) a transversal component (from the main channel to peripheral marshes) which corresponds to an increasing charge in organic matter, (iii) a vertical component (from aquatic ecosystems along high speed axes to alluvial forests which lie in the highest steps of the floodplain and are only flooded by a high rise in water level.

The constraints due to water excess must be kept (hypoxy, anoxy, habitats instability) and this must be the force idea of the plan of management. We go out of the fluvial system when the vegetal roots apparatus doesn't benefit temporarily or permanently from capillary rises from groundwater. The ecological map allows to simulate the future of the ecosystems when hydrological conditions are modified because of the cumulative effects of anthropic actions. One of the major objective of this plan must be to carry out the processes reproducibility to ensure permanent cummunities and a maintained number of populations with a major biological interest. To manage a landscape it is necessary to intervene on the transition between a previous situation which is often unknown to the land manager and a future situation which characteristics are difficult to fathom out when the changes tendencies under way only are taken into account. It is thus necessary to know if the plan foreseen is going to accelerate or slow down the evolution, deflect it or provoke a junction. The building of four dams since 1980 in the Rhône floodplain is going to modify quite deeply the structure of the vegetal cover and the nature of its components for the next few decades. The multitemporal mapping, from aerial photos and satellite data, allows to assess the changes intensity and to project them into the future (Garguet-Duport *et al.* 1994) The land manager can then have two opposite attitudes: either he finds the means to freeze a state opposing the evolutions in progress or he adjusts his plan taking note of the new properties the ecocomplex has aquired.

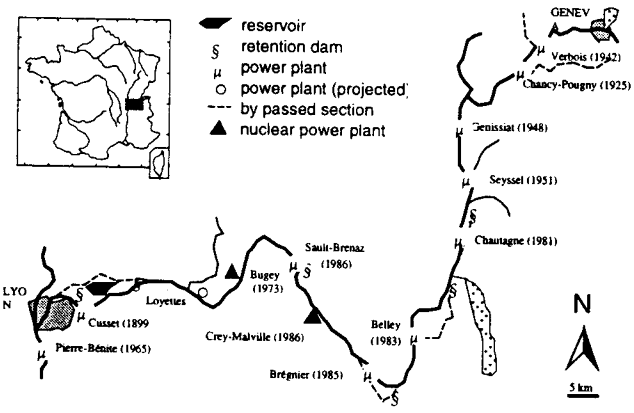


Figure 1: Location of different developpements along the Rhône river between Geneva and Lyon

ELABORATION OF AN ECOLOGICAL INFORMATION SYSTEM WITH SPATIAL REFERENCE

To manage a landscape you have to predict its evolution according to its internal modifications or the appearance of external pertubations. To achieve this you must be able (i) to measure and follow the evolution of significant parameters; (ii) to establish functional relationships between the constituent element and the fluxes of matter and energy; (iii) to anticipate the effects provoked by the building of works (dams for example) at different times and at last (iv) to elaborate mathematical models allowing reliable simulations. For a few years, under the pressure of computer technologies (Aronoff, 1989), we have been witnessing the development of systems able to stock, manage and use geographical information. In fact, the objective of these systems is to go beyond the only geographical information by processing all kinds of information spatially referenced. Then, these systems become privileged tools for territorry planning. They are called Geographical Information Systems (GIS) and cover a great variety of subjects, from district to continent, and solutions may be relevant in one case but not necessarily in the others. The role of the current GIS resembles a set of software tools gathering around five functions: aquiring, archiving, extracting, analysing and displaying information. In the environmental field, they only give an incomplete answer to the land manager's questions because of their low level of technological potential.

ISSR pertinence for the environment management and planning.

The ISSR technology was born at the beginning of the 1960s, caused by the progress in mapping and photogrametry fields. Canada was the first country to aquire an ISSR, the Canadian Geographic Information System (Terence, 1987). Since then, the applications first based on a simple superimposition of spatial data have been moving towards far more complex management and analysis operations which objective was to plan and help to take decisions.

However, the GIS are not the solution to all problems even if we could first have had that impression. Many applications of the GIS have probably failed because of that illusion, quite naive we must admit. Thus, it is important to identify at first what the contribution of GIS in planning and managing fields may be. Three different types of contributions appear: the GIS allow (i) to describe the territorry, (ii) to analyse the territorry and (iii) to feed models which simulate spatial phenomena (localisation of mosquitos larval shelter for example).

It is important to precise that the use of GIS only can't permit the achievement of these contributions. In nearly every case it is necessary to use tools or methods which complement the use of the GIS. Thus, the conception of a geographical database for the territory concerned may follow the methods of database conception such as REMORA (Rolland *et al.* 1988) or MERISE (Gabay, 1993). To be efficient in the territory analysis, it is necessary to use both the GIS and external specific softwares.

Conceptual model.

The territorry management is efficient if it is based on a global and systemic approach (Bédard, 1989)and the creation of associated data coming from various domains, relevant to each others and functional (de Sède, 1995). The approach concerning these ISSR is carried out by the Hydrosystemes Alpins group in a research and analysis scientific context. It concerns the development of a specialized Information System with Spatial Reference (ISSR) that we define as an Ecological Information System with Spatial Reference (EISSR). The methodological context of this EISSR implementation relies on the three mainstays defined by technology, research and applications as Dangermond[1] had already defined it (Figure 2).Research is the central mainstay of this system development. This research is carried out from the field of application (hydrosystems study) on the one hand, and from the technological tools

[1] Jack Dangermond, ESRI President, URISA congress, 1990 Edmonton

available or specifically developped on the other hand.

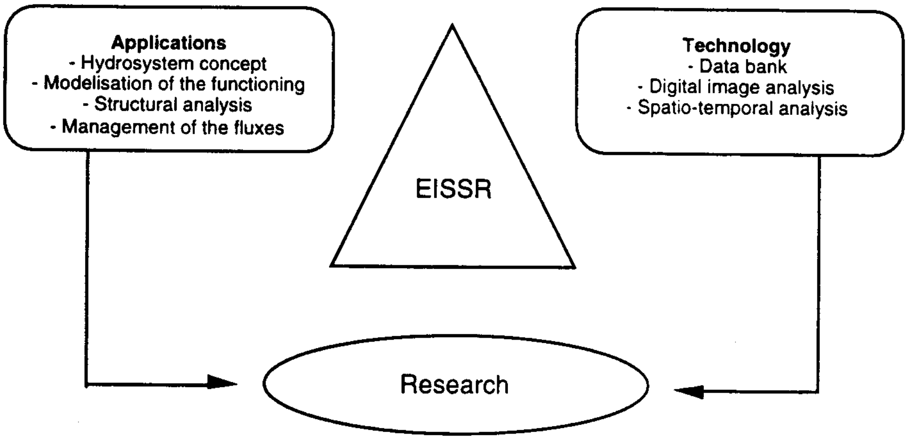


Figure 2: EISSR schematic representation (after De Sède, 1995)

The ISSR quality and efficiency are closely linked to the care taken from the beginning over the conceptual analysis and the modelisation done from the issues (Burrough, 1986; Burrough, 1991) The establishment of a conceptual model must be done from a global and systemic approach in the field of applications.

The analysis carried out by these systems offers an adapted set of concepts and methods. Indeed simplification and abstraction are necessary to understand the system, contrary to exhaustivity. Very often, the profusion of information gives a faulse impression of richness and usually goes together with a poverty of relevant and useful indicators (Voiron Canicio, 1992). Moreover it is a handicap in highlighting fundamental phenomena because it hides the essential with a background noise. This systemic approach allows a better understanding of the environment and its intrinsic characteristics.

The next step after the systemic analysis is the modelisation which allows to formalize the results of the analysis. The aim of the model is to reproduce in a more simple way the system fundamental characteristics which will permit to simulate its functioning and consequently its evolution. This evolution depends on the variation between the different internal parameters or the appearance of external perturbations.

These two steps are the precondition for the setting up of an Ecological Information System with Spatial Reference. They also permit a better analysis of the issues thanks to the definition of the thematic objects which constitute the system and with which we are going to work.

The objectives of the EISSR are to go beyond a simple superimposition of spatial data and to move towards more complex operations of spatio-temporal analyses (that is to say which takes into account previous facts and the elaboration of evolution predictions) which would allow a better understanding of the hydrosystems dynamic according to variations of the flux of energy and matter.

AN EISSR APPLICATION MODEL : THE ANTI-LARVAL MOSQUITO CONTROL

Specific problems occur in each type of ecocomplexe . The hydrosystems are characterized by the presence of human-biting mosquitos. By 1950, (Jullin *et al.*) underlined the consequences of hydrological changes upon mosquito populations along the lower Columbia river. By 1955, (Mihalyi *et al.* 1955) quoted the engineering-works impacts on mosquitos populations within the Danubian valley near Budapest. By 1969, (Ain and Pautou, 1969) highlighted the relationships between change of water, matter-flows and strong decreases in Aedes populations. By 1971, (Pautou and Ozenda, 1971) recognized the consequences of the abandonment of agricultural practices (mowing, grazing, ditches dredging) and various other anthropogenic factors such as of pond excavation or peat extraction on man-biting mosquitos populations along the French upper Rhône valley, between Lyon and Geneva. By 1980, (Maire and Aubin, 1980) measured the role of anthropogenic interventions in the St Lawrence river floodplain between Montreal and Quebec. (Knoz and Vanhara, 1982; Minar and Ryba, 1981) analysed the effects of hydroelectric developments in the Southern Morava; (Petts, 1984) highlighted the consequences of river regulation on invertebrate populations.

The adjustment of control programs concerning man-biting mosquitos population highlight the scientific basis on which the EISSR must be founded in order to succeed in its attempt.The species responsible for a high nuisance are belonging to the genus Aedes which is characterized by a 3-staged life-cycle (i) a terrestrial stage as eggs which are able to stay on wet soils for several years, (ii) an aquatic stage as larva and nymph and (iii) an aerial stage as imago. The mosquito control is carried out during the aquatic stage. Precise environmental information is necessary for (i) surveying and mapping large floodplains which can spread over 10 to 100 km long stretches of valleys (ii) to treat the habitats with larva - killing chemicals. The mosquito control must be efficient: actually, 10% of resistant larvae in a population are generally enough to produce an invasion over a region within a radius of 10 km and more.

So, an efficient EISSR must be able to consider (i) the biological type diversity (Aedes are characterized by larval cycles having various durations and developing during varying periods; besides Aedes, other genus such as Anopheles, Culiseta, Culex or Coquillettidia may have specific habitats), the spatio-temporal variation of water inputs (from a year to another, rainfall and snowcover volumes can double). Factors causing floods are numerous: river high-flows, floodings, high rainfall within the floodplain or on the watershed.

Each mosquito population has its own perception of the physiographic environment. Only mapped documents taking into account the determining parameters concerning each population are really useful for the mosquito-control operator. In order to efficiently plan the intervention programs over potential habitats, environmentalists need information concerning larval habitats sites and suitable periods for the control. This information can be provided from two elements exactly defined in the EISSR the functional sets and the functional units.

The functional sets

The functional set or, in other words, the biohydric compartment was defined in order to map the areas in which it was possible to find both standing water and stimuli (e.g. a favourable water temperature) allowing to trigger the hatching processes of species that develop their life-cycle during the cold period (winter and beginning of spring) such as *Aedes vexans, A. sticticus, A. cinerus*, and *Culex pipiens*). The origins of water inputs are varying. Flooding can be caused through a river overflowing or a ground water table rising. In this case, it generally occurs since may to the end of july and it is followed by a mosquito breeding characterized by a short larval stage, less than 10 days (Table 1). It is underlined that in this case-study, the EISSR must be particularly efficient in order to control mosquitos in the whole floodplain.The habitats depending on the functioning of the river Rhône cluster in the functional set. An other case-study may occur; flooding can be the result of tributaries overflows supplied from upper catchments (Southern Jura and Prealpes ranges). In this case, flooding generally occur before, owing to a early snowmelt period (in Jura and Prealpes mountains elevations are generally lower). The same larval habitats may be found in two different functional sets . It was true for a tall-grass community (a sedge meadow with *Carex elata*) which contained populations of *Aedes cantans* and *A. annulipes*. These were present within both types of compartments, yet, their flooding patterns were not synchronous. Insofar as the water table is not deep, roots can easily use the water supply; hence, plant communities are not good indicators of the timing "dried periods/flooded periods". By contrast, within functional sets, the whole sedges-communities were simultaneously flooded, thus, eggs hatching and larval development became synchronous.

Table 1: Biotic and hydrologic characteristics of the functional sets located to the North of the Bourget lake.

Functional sets	Origin of the water	Flooding period	Mosquitos species
of Rhône	Snowmelt in the upper catchment of the river	from june to august	*Aedes sticticus, A. vexans*, A. du groupe cinereus
of Séran	Snowmelt and rain in the middle watershed	from march / May to october / November	*Aedes* du groupe *cantans, A. vexans*
of Bourget lake	lake overflowing, Rhone water input, snowmelt and rain in the lake drainage basin	d'avril à juillet décembre et janvier	Anopheles du groupe *maculipennis*
of Lavours forest	local rainfall	from october to April	**Aedes rusticus**, *A.* du groupe cantans,
of Ceyzerieu	local rainfall	from october to April	*Aedes* du groupe *cantans*
of Béon-Talissieu	local rainfall and reappearances	from october to June	*Aedes rusticus*, A. du groupe cantans *Anopheles claviger*
of Chautagne and Lavours	ditch with permanent water ditch regularly overflooded		*Coquillettidia richiardii, Anopheles* gr. *maculipennis, Aedes rusticus, Culiseta annulata*

The Figure 3 shows the functional sets which were delineated within the floodplain area located at the Northern end of the Lac du Bourget. Functional sets can be easily mapped from remote-sensed data. A merging method of SPOT P and SPOT XS images has been developed (Garguet-Duport *et al.* 1994; Garguet-Duport, *et al.* In Press) Landforms, channel patterns (braided, anastomosed, meandering, straight...) and quantitative ratio concerning wooded, herbaceous or nude areas allowed to define configurations characterized by specific patterns of water-flow transits.

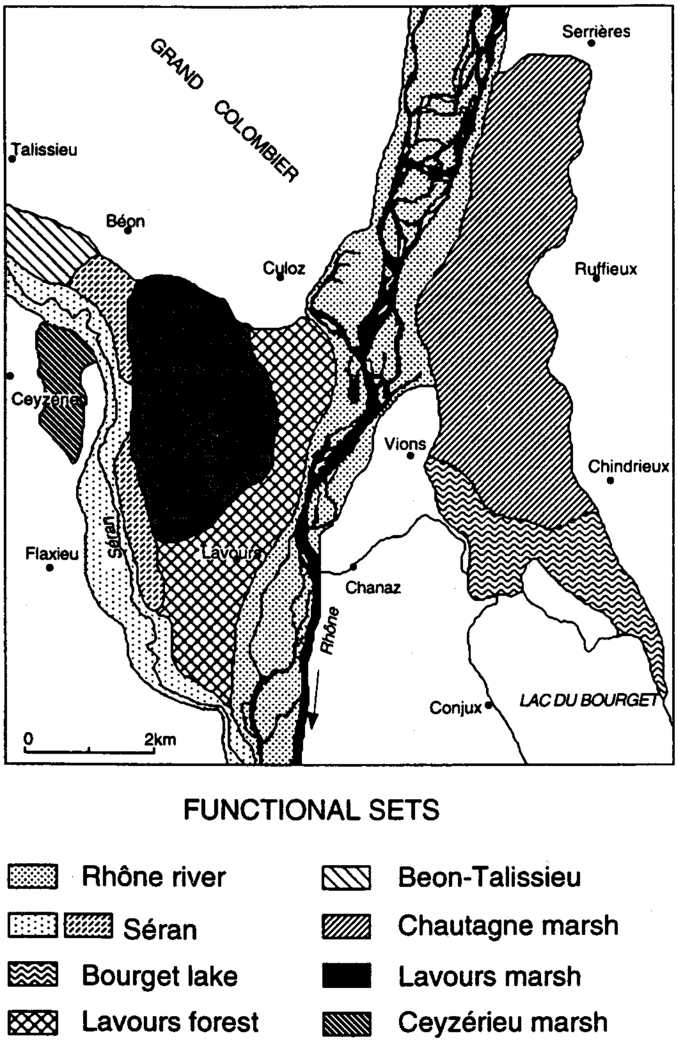


Figure 3: Functional sets delimited to the Noth of the Bourget lake.

<u>The functional units</u>

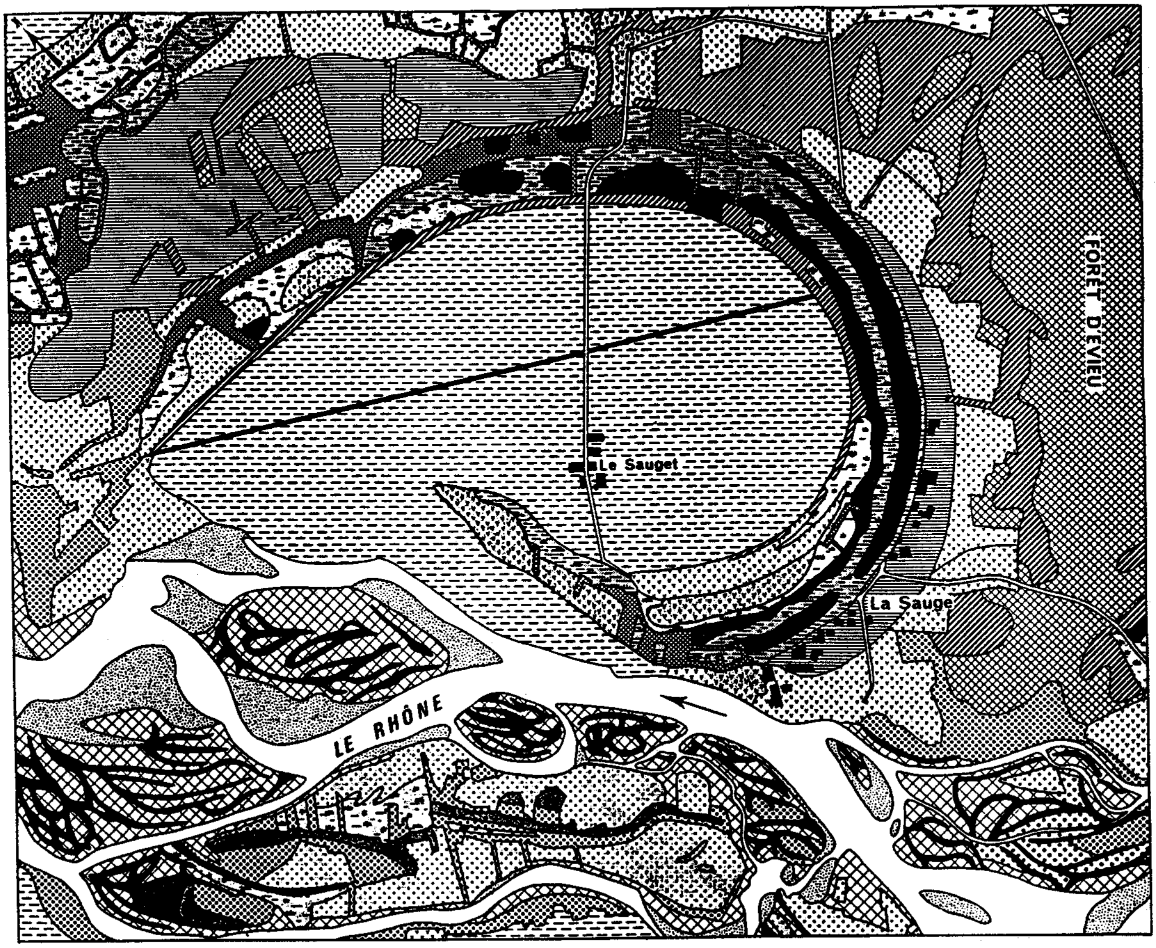
A functional unit is a part of floodplain characterized by specific associations (plants and animals), a more or less flooding likelihood and a period favourable to eggs hatching and larval stage achievement. Correlation between functional unit types and river discharges (e.g. white willow bushes with sedges and reeds growing on river islands which are flooded when the water discharges reach 400 $m^3 s^{-1}$ can be convenient biotopes for *Aedes sticticus* and *A. cinereus*) allowed to do some predictions. These were depending on the water inputs supplied from the upper catchment (snowmelt); they enabled us to delineate the functional units which could be positive, one month before the eggs hatching (Table 2). It will be noted that the functional units ranged along a water-flow gradient.

Table 2: Rhône river discharges and functional units flooding;

River Discharge	FONCTIONAL UNITS						
	Nymphaea alba	Phragmites communis	Carex elata Carex acutiformis	Salix alba, Carex acutiformis	Salix cinerea	Ulmus campestris	Crataegus monogyna
300 m3/s	+						
400 m3/s	+	+					
450 m3/s	+	+	+	+			
500 m3/s	+	+	+	+	+		
600 m3/s	+	+	+	+	+	+	
800 m3/s	+	+	+	+	+	+	+

The functional unit map (Figure 4) which was made from the EISSR is to used in a "dynamical way", according to hydrological patterns. During the years characterized by low snowcovers, only the units located on the lowest levels are functioning. A functional unit can be delineated through its physiognomy and environment (presence of a circuit, of a permanent water body in the vicinity) and, like the functional sets, it can be delineated from remote-sensed documents. Nevertheless, the functional unit cannot be superimposed exactly to plant communities areas; in other words, it may correspond to a particular woodlot, to a type of community (e.g. white willows woodlands) or to several plant communities characterized by similar flooding patterns (e.g. Black Alder woodlands and *Salix cinerea* bushes). Hence, the EISSR must enable us to perform delineations or clusterings in order to highlight the functional homogeneity, in other words, the habitat like it is detected by the mosquitos. The functional unit map (Figure 4) concerned the upper Rhône floodplain area. It was composed of two levels (=functional units) ranged along a water gradient. The functional unit characterized by *Salix alba* and *Carex acutiformis* is one of the best developed. It corresponds to braided channels cut-off from the main channel and more or less filled by fine sediments (presence of a 0.30 to 0.40 m. deep silt layer). This unit was flooded when the river discharge reached 400m^3/s. This type of unit may produce 4 to 6 larval cycles (*Aedes sticticus*) per year. The functional unit characterized by elms (*Ulmus minor*) was established within secondary channels in which the filling processes were not advanced (the silt/clay layers covering the coarse gravel deposits were 0.60 to 0.80 m. deep) . This functional unit was flooded when the flow discharge reached 800m^3/s once a year. The map was used in dynamic way; then it was easy to correlate the hydrologic position and the larval habitats. The prediction concerning water-flow inputs in relation to snowcovers and temperature allowed us to plan the mosquito control over space and time.

Figure 4: Functional units map



(functional units)

- Niveau à *Nymphaea alba*
- Niveau à *Phragmites communis* et *Carex acutiformis*
- Niveau à *Carex acutiformis*
- Niveau à *Ranunculus repens*
- Niveau à *Salix cinerea*
- Niveau à *Alnus glutinosa*
- Niveau à *Alnus glutinosa* et *Fraxinus excelsior*
- Niveau à *Fraxinus excelsior* et *Quercus pedunculata*
- Niveau à *Salix alba* et *Carex acutiformis*
- Niveau à *Ulmus campestris*

(natural vegetation and cultivited areas) ;

- Taillis à *Alnus incana*
- Taillis à *Salix alba* et *Impatiens roylei*
- Bois de *Fraxinus excelsior* et *Populus nigra*
- Prairie à *Arrhenatherum elatius*
- Prairie à *Molinia coerulea* . Lande
- Prairie - Maïs - Pommiers - Tabac
- Prairie - Maïs - Céréales - Vigne
- Bancs de graviers

CONCLUSION

The floodplains are privileged conflictual sites. The elaboration of modern analysis tools which rely on spatio temporal database is necessary to establish programmes for the planning of alluvial spaces and the management of renewable resources relying on a harmonious balance between the different functions (production of food and electricity resources, conservation of the animal and vegetal populations, industrial development, construction of communications, maintaning of the quality of the underground water, urbanisation, water stocking when high floods, ...). The establishment of maps from this database is a fundamental step to plan the use of the alluvial space in the large fluvial systems.

The development of such tools should allow to elaborate predictive scenarios about the hydrosystems functioning modalities during paroxistic events such as one-in-fifty-years or one-in-a-hundred-years floods. The partition of the floodplain in compartments, more or less well connected by the construction of works (railways, highways), the impermeabilisation of large surfaces for urbanism, the lack of alluvial forests likely to stock the excess of water and the vegetalisation of confined beds, are not likely to accelerate the flow of surface waters.

We may wonder if the disastrous flood which have taken place in France for a few years do not result from heavy rainfalls but also surround the disaster towns which would have favoured the concentration of huge amounts of water on small surfaces in a very short time. Scenarios relying on the same principles could be carried out for different hydrological situations (year with an imposed low-water-level, functioning after a series of dry or wet years for example). The optimum ratio between the areas reserved for the different functions of the floodplain could be modelised from a large range of hydrologic situations.

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AQUATIC GAP ANALYSIS: TOOL FOR WATERSHED SCALE ASSESSMENT OF FLUVIAL HABITAT AND BIODIVERSITY

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ABSTRACT

Methods for the conservation of stream habitat and biodiversity at the watershed scale have not been developed. Watersheds span large land areas, encompass a connected range of stream sizes, and integrate natural and altered properties of a drainage area. Methods are needed to identify the locations of high biodiversity in watersheds, compare aquatic biodiversity distributions among regions, and provide watershed-scale information useful for targeting conservation measures. The National Biological Service (USA) in cooperation with other Federal and State agencies developed geographic information system (GIS) methodology called Gap Analysis to identify the distribution of biodiversity over large spatial areas. To date, it has been used to address only terrestrial conservation needs. We are developing an aquatic version of the Gap Analysis in the Allegheny River drainage in western New York State to define the methodology and evaluate the feasibility of predicting biodiversity distribution at the watershed scale.

Our standardized stream reach accounting system is based on the U. S. Environmental Protection Agency Reach File 3 System. Each stream reach is classified into one of 18 habitat types for fish faunal predictions and one of 8 habitat types for invertebrate faunal predictions. Habitat types were defined using the following sets of physicochemical attributes: stream size (headwaters, large streams/small rivers, large rivers), physical habitat (dominated by natural geomorphological processes, moderately altered, and dominated by human structures and controls), water quality (suitable for life support, biologically stressful), gradient (steep, low slope) and riparian forest cover (closed canopy over channel, open channel). Stream size was determined from drainage area using the GIS. Physical habitat, reach gradient, and riparian forest cover were classified from topographic and land use maps. Physicochemical data from the U. S. Environmental Protection Agency STORET database provides a means to classify water quality. Using our habitat typing system, we predict that the highest fish diversity will be found in medium size streams with natural fluvial channels and good water quality, whereas the most reduced fish faunas will be found in large rivers with highly modified channels and poor water quality. For invertebrates, we predict that the greatest diversity (in terms of ecological function groups) will be in small and medium size streams with primarily a closed canopy, steep gradient, and good water quality.

Our GIS modeling effort succeeded in predicting the expected distribution of fish and invertebrate diversity at the watershed scale. Adequate biological and physicochemical data appear available and compatible with watershed-scale GIS programs. We also have extensive biological survey data that provides an independent means to testing the validity of our biodiversity predictions.

KEY-WORDS: Biodiversity / Conservation / Gap Analysis / GIS / Allegheny River / Biotic indices / Stream habitat / Watersheds / Water quality

INTRODUCTION

Methods for the conservation of stream habitat and biodiversity at the watershed scale have not been developed. Watersheds span large land areas, encompass a connected range of stream sizes, and integrate natural and altered properties of a drainage area. Methods are needed to identify the locations of high biodiversity in watersheds, compare aquatic biodiversity distributions among regions, and provide watershed-scale information useful for targeting conservation measures. The National Biological Service (USA) in cooperation with other Federal and State agencies developed geographic information system (GIS) methodology called Gap Analysis (Scott et al. 1993) to identify the distribution of biodiversity over large spatial areas. To date, it has been used to address only terrestrial conservation needs. We are developing an aquatic version of the Gap Analysis in the Allegheny River drainage in western New York USA to resolve two main questions: (1) is there adequate biological information to link faunal composition to stream reaches (tributary confluence to confluence) on a large scale and (2) Can useful physicochemical data be assembled for stream reach habitat classification? This study seeks to create a methodology to answer these questions and evaluate the overall feasibility of predicting biodiversity distribution at the watershed scale.

METHODS

The basic elements of our GIS model are stream segments, where a segment is the portion from tributary confluence to tributary confluence or in lakes from stream mouth to stream mouth or outflow. A River Reach File version 3.0 Arc/Info layer was obtained from the U.S. Environmental Protection Agency to numerically and graphically catalog the stream segments in the Allegheny River basin. Each stream segment in this file is numbered and geospatially referenced at the 1:100,000 topographic (U.S. Geological Survey [USGS]) map scale. There are 1340 stream segments in the New York portion of the Allegheny River basin. All stream segments in the Reach File were highlighted on 1:24,000 USGS maps to ease identification of segments and help delineate drainage divides. The divides were then digitized and edge matched to form a complete map layer. A program to calculate accumulated drainage area can use this layer to obtain the total drainage area for any stream segment. The accumulated drainage area measurements are used in classifying stream size.

Models for predicting biodiversity

Each stream segment was designated as one of 18 habitat types which were used to classify flowing water habitats to make predictions of characteristic fish species. Habitat types were defined using the following sets of physicochemical attributes: stream size (headwaters, large streams/small rivers, large rivers), physical habitat (dominated by natural geomorphological processes, moderately altered, and dominated by human land use and controls), and water quality (suitable for life support, biologically stressful). As mentioned above, the drainage area GIS layer was used to determine stream size. Topographic and land use maps were used to determine physical habitat, reach gradient, and riparian forest cover. Physicochemical data from the U.S. Environmental Protection Agency STORET database provided a means to classify water quality.

A list of fish species recorded from the Allegheny River basin was developed to classify fish species by habitat type. Information was gathered from various "Fishes of" books and previous bioassessment and fish distribution studies (Bramblett and Fausch, 1991; Karr et al., 1986; Ohio EPA, 1989; Schlosser, 1982). Stream size preference was categorized as headwaters, large streams/small rivers, and large rivers. In cases where a species was associated with more than one stream size, both categories were used with the first designation indicating the primary stream size. Tolerance to habitat degradation was categorized as intolerant, moderately tolerant, or tolerant. Three water quality tolerance ratings have been used in past classifications (intolerant, moderately tolerant, tolerant). These were regrouped into two categories (intolerant, tolerant) by moving moderately tolerant species into the intolerant class. Thus, only the most tolerant fish species are considered tolerant in this study.

The classification of habitat types for invertebrate taxa was handled in a similar way as fish. Stream segments were classified into one of eight habitat types for invertebrate diversity predictions using the following sets of physicochemical attributes: gradient (steep, low slope), riparian forest cover (closed canopy over channel, open channel), and water quality (suitable for life support, biologically stressful). All large rivers were considered to have open channels regardless of riparian status. Physical habitat, reach gradient, and riparian forest cover were classified from topographic and land use maps for each stream segment. Physicochemical data from the U. S. Environmental Protection Agency STORET database was used to classify water quality.

The approach for predicting characteristic invertebrate taxa by habitat type was the same as the approach for fish. Eight hundred and seventeen taxa (mostly benthic insects) were recorded in stream and river samples in the New York portion of the Allegheny River basin. Almost half of these taxa were identified at the species level. Sample data is from bioassessment surveys completed in 1981 and 1989-1990 by the New York State Department of Environmental Conservation [NYDEC], Division of Water (Bode et al., 1991, 1993). Feeding guilds were assigned from Bode et al. (1991) as follows: predator, collector-gatherer, collector-filterer, scraper, or shredder. Species and genus-level tolerances for degraded water quality were based on Bode et al. (1991) using the categories: tolerant and intolerant. Family-level tolerances were taken from Hilsenhoff (1988). When taxa were not assigned a family tolerance by Hilsenhoff (1988), their species tolerance was used. The life style habit of each taxa was taken from Merritt and Cummins (1984) for the insects, and from Pennak (1978) for the other invertebrates. Life habit categories were: burrower, clinger, swimmer, climber, sprawler, and not specific. When several habits were listed for a taxa, the first was used for our classification.

<u>Testing the biodiversity predictions</u>

We are relying on GIS layers of measured data for fish and invertebrate assemblages to indicate biodiversity priority areas. Two map layers, fish species richness and biological integrity (a community quality index), were developed to test fish assemblage diversity. For the fish species richness layer, we assembled all known data in recent decades that reports on whole fish community collections. The primary sources were: intensive surveys of the French Creek portion of the Allegheny River basin and recent (late 1980s and 1990s) surveys conducted by the NY Department of Environmental Conservation. A total of 114 fish species have been recorded in the New York portion of the Allegheny River basin. For the index of biotic integrity map layer, we modified the original index of biotic integrity (Plafkin et al. 1989; reviewed in Karr et al. 1986; Karr 1991) for our study area. We need further refinement of the our index of biotic integrity because quality ratings are too low for species-poor, coldwater streams in forested areas. A headwater index of biotic integrity will soon be published (Lyons et al. 1996) to correct this problem.

We gathered data from the 1980s covering invertebrate assemblage collections (largely 100-organism, benthic kick-samples) to test the invertebrate biodiversity predictions. The primary sources were: intensive surveys of the French Creek portion of the Allegheny River and bioassessment surveys conducted by the NYDEC (Bode et al., 1993). The NYDEC bioassessment collections are the most extensive, and field sampling was done with multiplate samplers (1981) and 100-organism kick netting (1989-1990). Details of the sampling protocol and processing is fully described by Bode et al.

(1991). We are relying on GIS layers of taxa richness and six measures of community quality to indicate biodiversity priority areas due to the high taxonomic diversity of aquatic invertebrates. The six community parameters calculated for each invertebrate collection are taxa (species) richness, EPT index, biotic index, family-level biotic index, diversity (Shannon-Wiener), and percent model affinity. Methods and scoring criteria for each described in Bode et al. (1991) except for the family biotic index which was taken from Hilsenhoff (1988) with some scoring modifications.

RESULTS AND CONCLUSIONS

In the beginning of the study we were concerned that the Environmental Protection Agency River Reach File version 3.0 base layer of stream segments may not meet our needs. After extensive use it was determined to have many advantages for our purposes. The streams and lakes in the Reach File matched what was printed on the 1:100,000 scale USGS topographic maps. This was important for matching reach numbers from the Reach File to habitat classifications from the topographic maps. The reach numbers also served as a useful link to other data sets and coverages. In addition, the navigational attributes of the Reach File were useful for tracing upstream and downstream directions from any point on a stream and hence determining cumulative drainage area.

Our predictions for fish diversity indicate that mid-size streams should have the highest diversity (Figure 1). Although large rivers generally have longer species lists than smaller waters, the middle size class of streams had the greatest number of characteristic species. Water quality appears to diminish fish diversity to a much greater degree than habitat degradation. However, degraded water quality is much less common on the watershed scale than habitat alteration. Therefore, we expect that habitat alteration will have a more pervasive effect on fish diversity than water quality degradation, but water quality has a stronger effect in fish diversity when it is stressful.

As for fish, water quality degradation appears to have a much greater impact on invertebrate diversity than physical habitat alteration (Figure 2). Degraded water quality is much less common on the watershed scale than habitat changes like riparian deforestation and channelization. Riparian vegetation removal causes open canopy stream segments shifting the community to a new faunal composition more oriented to use of algae as a food source. The combination of a diverse food base (allocthonous input from riparian forest) and coarse substrate provides the widest range of feeding guilds and life habit groups. Consequently, we predict that shaded and high gradient streams will have the highest anticipated taxonomic diversity for invertebrates.

Combining predicted patterns for fish and invertebrates, we expect that overall biodiversity is likely to be highest in mid-sized streams with a high gradient, forested riparian zones and valley, and suitable water quality. Stream segments with this combination of attributes will be expected to have the highest diversity of fish and invertebrates. These high diversity stream segments are likely to be sparsely distributed throughout watersheds with abundant human land uses.

The measured data used to test the biodiversity predictions show a scattered distributional pattern of diverse, quality fish communities for both species richness and biological integrity. The highest rated stream segments are on mid-size streams spread across the river basin. Therefore, field data does not indicate clusters of high diversity and quality stream segments for fish in the Allegheny River basin. Rather, what appears to be priority conservation sites are well dispersed in the region.

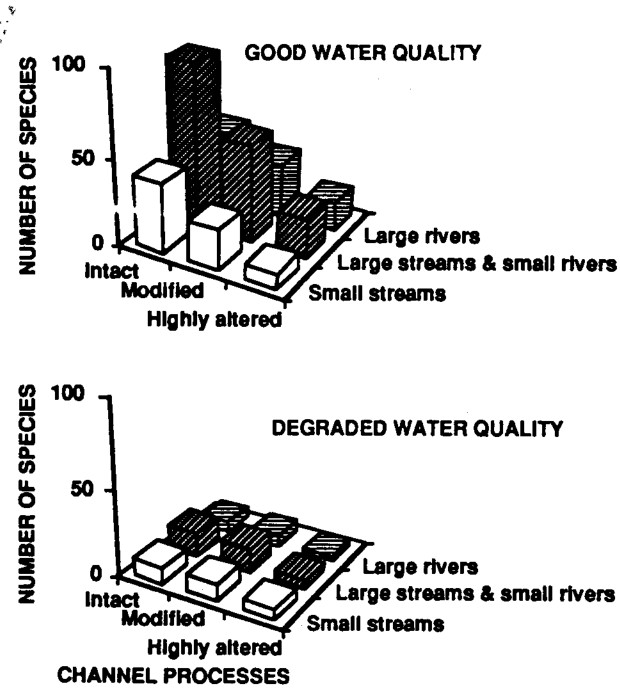

Figure 1: Predicted Fish Fauna

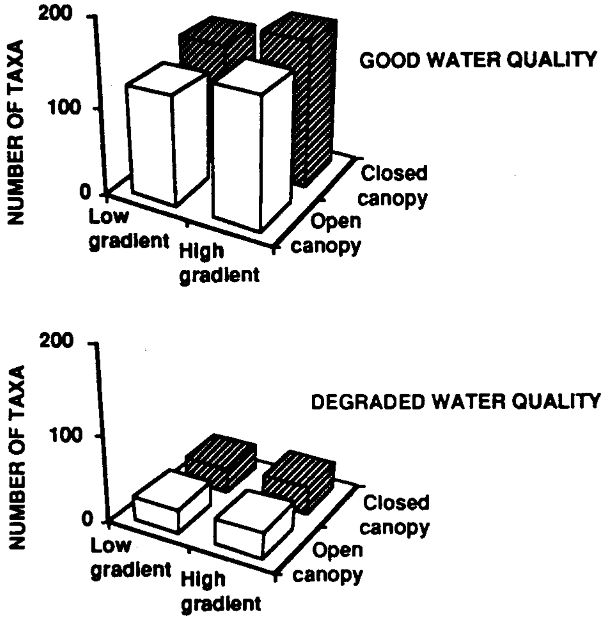

Figure 2: Predicted Macroinvertebrate Fauna

Our GIS modeling effort succeeded in predicting the expected distribution of fish and invertebrate diversity at the watershed scale. Adequate biological information and useful physicochemical data appear available and compatible with watershed-scale GIS programs. However, more work needs to be done to develop a sound method for the conservation of stream habitat and biodiversity at the watershed scale. Further studies are planned to determine the predicted biodiversity of mussels in the Allegheny River basin and create an erosion model to link land cover to biodiversity.

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Peak flow management

Effet des éclusées et débits de pointe

STUDY OF THE JUVENILE COMMUNITY IN THE BROWN TROUT
(*Salmo trutta fario* L.,1758)
IN HYDROPEAKING SITUATIONS

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ABSTRACT

The Oriege River, in the central part of the French pyrenees, has its source at an altitude of 1700 m. Over its total length of 11 km, it presents two distinct profiles: the upper part, with a mountain torrent-type flow (slope over 1%) and the lower part which is calmer (slope less than 1%) with successions of different morphologies including pools, riffles and rapids. In 1958 a hydroelectric station was built in the central part of the stream - at an altitude of 912 m. The plant uses 4°C water stored in a dam at an altitude of about 1900 m and operates with up to 11 $m^3.s^{-1}$ going through the turbines and into the stream which has a mean flow of 4 $m^3.s^{-1}$. The annual flow pattern of the stream is mainly nival so the stream is subjected to periods of severe low flow in autumn and early winter (less than 1 $m^3.s^{-1}$).

Downstream of the power station, the Oriege River must withstand large variations in flow: the amplitude, duration and frequency of which are dependent on the peaking management of the plant. The fluctuations in flow and temperature could have a negative impact on the fish populations and in particular on the youngest stages. Therefore, we carried out research on the population dynamics of juvenile of the brown trout (0+ and 1+) in relation to the habitat and the characteristic peaking patterns.

From the three sites selected for the study, (1 upstream of the power plant, one just below the plant's water outlet, and one further downstream), we showed that the flow surges acted significantly (ANOVA, $p< 0.05$) on densities of 0^+ and 1^+ trout. Moreover, 0+ and 1+ under hydropeaking conditions lossed their logic for mesohabitat in term of spatial distribution.

KEY-WORDS: Hydropeaking / Disturbance / Juvenile community / Brown Trout / Mesohabitat.

INTRODUCTION

The impact of the regulation of flow in rivers has been qualitatively studied in the past by numerous authors (Cushman, 1985; Lauters, 1993). However, the importance of habitat for populations of stream salmonids is generally not well understood, despite the large amount of research that has been conducted (Hicks *et al.*, 1991). On the other hand, relatively few researchers have experimentally mesured changes in salmonid abundance and habitat caused by hydropeaking downstream plant producing electricity during peak power consumption periods (Heggenes, 1988a; Gore *et al.*, 1989; Kinsolving and Bain, 1993; Lauters, 1995; Valentin, 1995).

In France, 144 hydroelectric plants are known to operate in the discontinuous mode causing flow variations. They involve 80 streams and rivers and strongly affect 800 km of riverbed (Lauters, 1993). These fluctuations in the flow are due to the sudden release, into the river, of water stored in a reservoir. The flow surges caused are often called hydropeaking. Whatever the type of river or the type of plant, researchers often attribute hydropeaking management to negative impact on the ecosystem. This can be gathered into four categories:

- modification of the physical and chimical parameters of the water (temperature, level of nitrates, etc.) (Ward and Stanford, 1979);
- processes of erosion of the mineral and organic substrates (Gore, 1994);
- disturbances of the population of benthic invertebrates such as reduction of density, modification of the populations and increase of drift; (Garcia de Jalon, 1988; Moog, 1993; Sabaton *et al.*, 1995),
- disturbance of the fish populations (Crisp, 1993; Moog, 1993; Liebig, 1994; Lauters, 1995; Valentin, 1995).

The aim of our study is to determine impact of hydropeaking from the hydroelectric plant of Orlu in the Oriege River, on density and spatial distribution of the youngest stages of brown trout. The effects can be accounted for by analysis of juveniles trout populations parameters, and by the study of the mesohabitats used.

MATERIALS AND METHODS

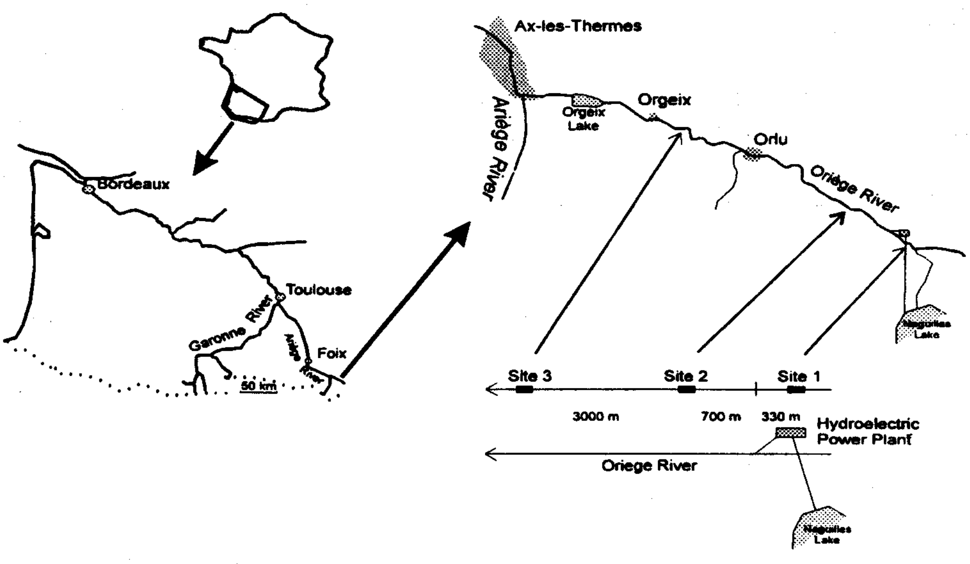

Figure 1: Location and hydroelectric impoundement of the three studied sites

The Oriege River is an affluent of the Ariege (Pyrenees mountains, south of France), having its source at 1700 m altitude (Figure 1). It is a mountain torrent 11 km long, is fed essentially by melt-water, has a mean annual discharge of 4 m^3 s^{-1} and an average slope of 8.6 ‰. The morphology is that of a torrent with falls, rapids, riffles, channels and runs (Malavoi, 1989). 6 km from the source (altitude 910 m) a hydroelectric power station, built in 1958, takes its water from the Naguilles dam at an altitude of 1890 m. It is bottom water that is drawn off from the dam. The plant has two 40 megawatt Pelton generators which can turbine up to 11 $m^3.s^{-1}$. The water is released into the river through a 200 m long canal.

Water-quality was similar for both lake and river (Dauba and Tourenq, 1986) and was characteristic of salmonid rivers (Nisbet and Verneaux, 1970). The reservoir is oligotrophic and did not develop anoxic hypolimnia. Waters released were high in dissolved oxygen and low in organic pollution. Only the temperature regimes differed between Naguilles Lake and Oriege River.

Three sites were selected according to the following criteria: presence of a run with suitable spawning characteristics (called spawning area) and different morphological units such as channel, riffle and rapid (Table 1). The first site representative of the natural stream, is located 330 m upstream of the impoundment, the two others respectively at 700 m and 3000 m downstream of the rejection impact point

Table 1: Sites and morphological characteristics.

	Site 1	Site 2	Site 3
Altitude (m)	913	880	825
Length (m)	75	70	70
Mean slope (%)	2.6	2.57	1.45
Numbert of survey	4	4	4
Morphological units (codes)	channel (cha1) rapid (rap1a) rapid (rap1b) riffle (rif1) run (spw1)	riffle (rif2a) riffle (rif2b) run (spw2) channel (cha2)	run (spw3) riffle (rif3a) rapid (rap3a) channel (cha3) riffle (rif3b)

Water temperatures were monitored with three-month continuous recording thermometers ("Indic 8000" from Pekly Hermann Moritz, Groupe Befic) installed at Site 1 and 2. Recording was interrupted in September 1995.

Sedentary brown trout (*Salmo trutta fario*, L. 1758) was the only fish species in the Oriege River. Surveys were undertaken in July (summer) and September (early autumn) 1994 and 1995 by electroshocking ("Grebe Huppee" from Dream Electronic, 220-380 V / 7.6 A, transformed into 180-1000 v / 1 A DC) with particular attention being given to catching the youngest life stages. A two-pass removal procedure was used. The spawning season was dated by visual observation of maximum spawning activity, like in many Pyrenees Rivers (Delacoste, 1995) as being in mid of December (taken as December 15th). Hourly flow rates between Sites 2 and 3 were provided from the hydrological database of EDF (Direction de la Production et du Transport - Division Technique Generale - Grenoble, France). The chronology obtained was smoothed, eliminating flow due to hydropeaking, to reconstitute a chronology of discharge in the area upstream of the restitution. Three curves of potentially available physical habitat (called Weighted Usable Area or WUA) versus discharge were available, for the juveniles at each site from Lauters (1995) which used PHABSIM model. The theoretical number of trout in each cohort was estimated with the Carl and Strub (1978) method. Trout population parameters were studied with Winfish Software ®, release 2.7 (Ichthyo Sys).

Growth and mortality rates were calculated from September 1994 to September 1995 ($G=\ln(w_t/w_0)$ where w_o is the starting weight (g) and w_t is the weight at the end of the period; $Z=\ln(N_t/N_0)$ where N_0 is the starting number and N_t is the number at the end of the period).

To consider the spatial distribution of Young-of-the-Year (called YOY or 0+) and yearling trout (called 1+), samplings were pooled at each site.

Univariate and graphic statistical analysis were performed with SPSS Software ® (release 6.1). We used analysis of variance (ANOVA) to test changes in density of 0+ and 1+ groups of trout for each morphological unit at each site and during each sampling period. The data were transformed (logarithm) to passed Lilliefors test of normality. Multivariate analysis was performed with STATLAB Software ® (release 2.1). Correspondence analysis was made using the data on density.

RESULTS

Temperature and flow remained the two most important factors controlling biota of unpolluted streams (Ward and Stanford, 1979). The thermal regime of the unregulated site is characteristic of a mountain river i.e. cold winter and warm summer with seasonal and daily changes of about 3°C for winter to 5.5°C for summer. Both flow regulated sites showed an increased daily fluctuation and a reduced seasonal one. 4°C water release as impoundment peak caused temperature elevation in winter and decrease in summer (Table 2 and Figure 2).

Table 2 Mean thermal characteristics for unregulated and regulated sites (Sites 1 and 2).

	Site 1	Site 2
Mean annual temperature (°C)	7.2	6.9
Mean winter temperature (°C)	4.3	4.1
Mean summer temperature (°C)	11.3	9.9
Mean winter difference (°C)	3	5
Mean summer difference (°C)	5.5	9

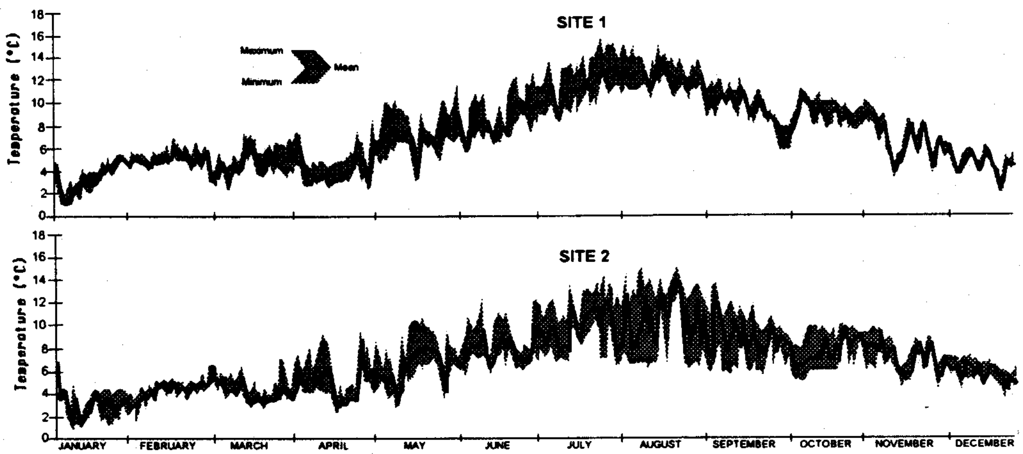

Figure 2: Thermal regimes of the unregulated (Site 1) and the regulated sites (Site 2) of the Oriege River during 1995.

Cumulative temperatures built up at each station were compared for the most important periods in the life of YOY (Table 3).The difference between Sites 1 and 2 for the under-gravel period was about 6 degree-days, i.e. less than one day, for Site 2. So, we assumed that all YOY theoretically emerged from the gravel on 5th May. There being no significant difference between Sites 1 and 2 for the date of emergence. For the period of spring growth, there is no difference between Sites 1 and 2. During the maximum growth period, there are 111 degree-days more for the fry at Site 1.

Table 3: Cumulative degree-days at Sites 1 and 2 for three periods

	Site 1	Site 2
Incubation and under-gravel life	637	643
Spring growth	611	611
Summer growth	746	635

As a typical mountain stream, the Oriege River presented a nival flow regime with a snow-melt maximum in May-June (about 6 $m^3.s^{-1}$) and lower flow during end of summer and winter (about 0.5 $m^3.s^{-1}$). Flow peaks due to rainfall are also visible (Figure 3). The natural flow patterns were similar for the two years, with just an earlier snow-melt for 1994 (mid-April) than for 1995 (start of May). For the downstream curves, the maxima of discharge caused by hydropeaking are well represented (maximum about 13 $m^3.s^{-1}$). High discharge and daily fluctuations becoming stronger. The regulated flow patterns are similar for the two years, except for the low discharge period occurring in early May 1995, during the emergence period.

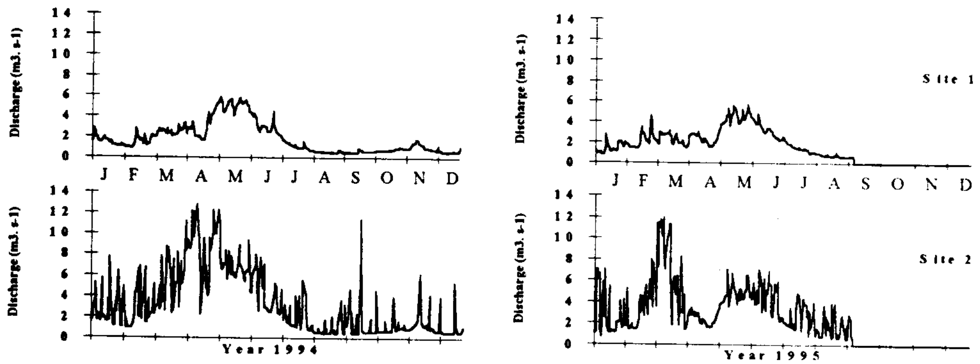

Figure 3: Flow regimes of the unregulated (Site 1) and the regulated sites (Site 2) on the Oriege River during 1994 and 1995.

Comparative study of 0+ and 1+ trout population characteristics

Biomass, density, mean weight and mean length for trout of ages 0+ to 1+ for each site and at each sampling period are represented in Table 4. The highest YOY density for Site 1 occurred in 1994 (659 Ind./100m). Comparable values were found for Site 3 in 1995. Density of YOY next September and of yearlings next year did not follow the high initial density. This should be considered in the context of the limited number of suitable territories (rest or hunting).

Table 4: Biomass and density for age 0+ and 1+ trouts in River Oriege.

		July 1994			September 1994			July 1995			September 1995		
Sites		**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**	**1**	**2**	**3**
Biomass	**0+**	309	106	406	315	398	755	108	130	436	463	295	1025
(g/100m)	**1+**	621	237	1042	542	242	1135	328	156	1074	631	168	1032
Density	**0+**	659	184	370	202	181	277	258	250	693	278	133	381
(ind/100m)	**1+**	86	25	77	56	26	69	51	21	88	71	18	62

The mortality coefficient (z) calculated from September 1994 to September 1995 was -1.045 at site 1. It was much higher at site 2: -2.308 and presented an intermediate value at site 3 of -1.497. This increase in the rate of disappearance of the fry at the downstream sites i.e. actual mortality but also emigration, should be viewed with respect to the frequent abrupt changes in the hydrological conditions at site 2, already more attenuated at site 3.

Growth in weight showed an increasing trend going downstream. But at the same time, growth rate (G) presented a lower value for site 2 (Table 5).

Table 5: Mean weight parameters for both the life stages at each site

	Site 1		Site 2		Site 3	
Life stages	0+	1+	0+	1+	0+	1+
July mean weight (g)	0.45	6.9	0.55	8.45	0.85	12.7
September mean weight (g)	1.58	9.3	2.2	9.3	2.7	16.5
Growth rate	1.740		1.449		1.811	

Growth in length estimated by Petersen's method was verified by scale analysis ($p<0.05$). Length, like weight, presented an increasing trend going downstream (Figure 4).

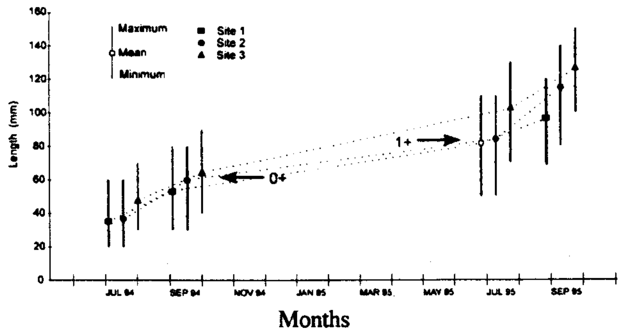

Figure 4: Length increase of the 0+ trout group born in 1994 on the River Oriege

Spatial ditribution of juvenile 0+ and 1+.

Analysis of variance (ANOVA) was used to determine and test explicative factors of the variation in density and mortality (Table 6).

Table 6: Result of the ANOVA tests calculated on the density of different stages at different sites for three factors (years, seasons and morphological units). * : significant difference at 95%confidence level; ** : significant difference at 99% confidence level.

Life stages	YOY			Yearling		
Sites	1	2	3	1	2	3
Years	0.979	0.910	0.953	0.402	0.377	0.465
Seasons	0.766	0.339	0.984	0.095	0.001**	0.796
Morphological Units	0.013*	0.189	0.003**	0.021*	0.310	0.017*

No significant difference was found between the two years (i.e. no effect of year on density variation), for both the life stages of brown trout. Significant difference between seasons was only shown for yearlings at Site 2 ($p<0.01$). This result assumed that there was no diminution of density between two seasons for any of the YOY or for yearlings at Site 1 et 3. Actual meaning was different because the variation in density could be due to mesohabitat shifts in fish inside the station. This behavior disappeared for yearlings under hydropeaking conditions.

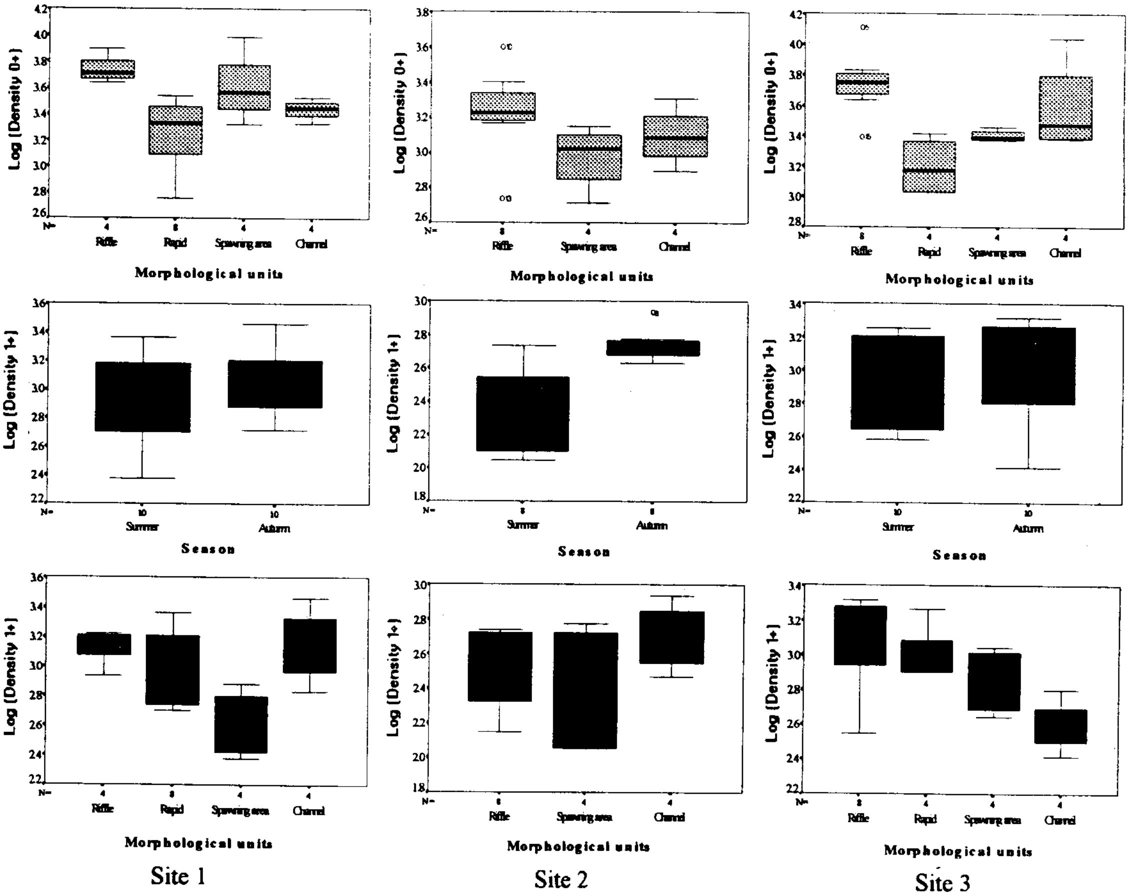

Figure 5: Box-plots of 0+ and 1+ trout density at three sites for three factors (years, seasons and morphological units).

Significant differences were found between the densities of YOY and yearlings at Site 1 and the type of morphological unit ($p<0.05$). Similar results were found for Site 3 ($p<0.01$ for YOY and $p<0.05$ for yearlings). This means that young trout used some morphological units better than others. At Site 1 YOY preferentially occupied riffles and runs (Figure 5). Comparable results appeared for Site 3 with extension to channels. Yearlings showed preferences for riffles, rapids and channels at Site 1, for riffles and rapids at Site 3. For Site 2 there was no significant difference between densities, for the various morphological units. This suggests therefore that stronger hydropeaking conditions in Site 2 compared to Site 3 disrupted the choice of site of the young trouts

Correspondence analysis was used with density data to show the dynamic factors involved. Results are shown in the Figure 6 (F1xF2 factorial plane). The first axis (F1) gathered 50.91% of the information. This axis presented a significant negative correlation with the density of 0+ in July 1994 (*0+j94*, Rs=-0.569) and a positive one with 0+ density in July 1995 (*0+j95*, Rs=0.501). Spawning area of Site 1 (spw1) and those of Site 2 and 3 (spw2 and spw3) were associated with the first axis. Spw1 was representative of YOY production in 1994. Spw2 and spw3 were representative of YOY production in 1995, which was more efficient for both the regulated sites, but lower for Site 2 than Site 3.

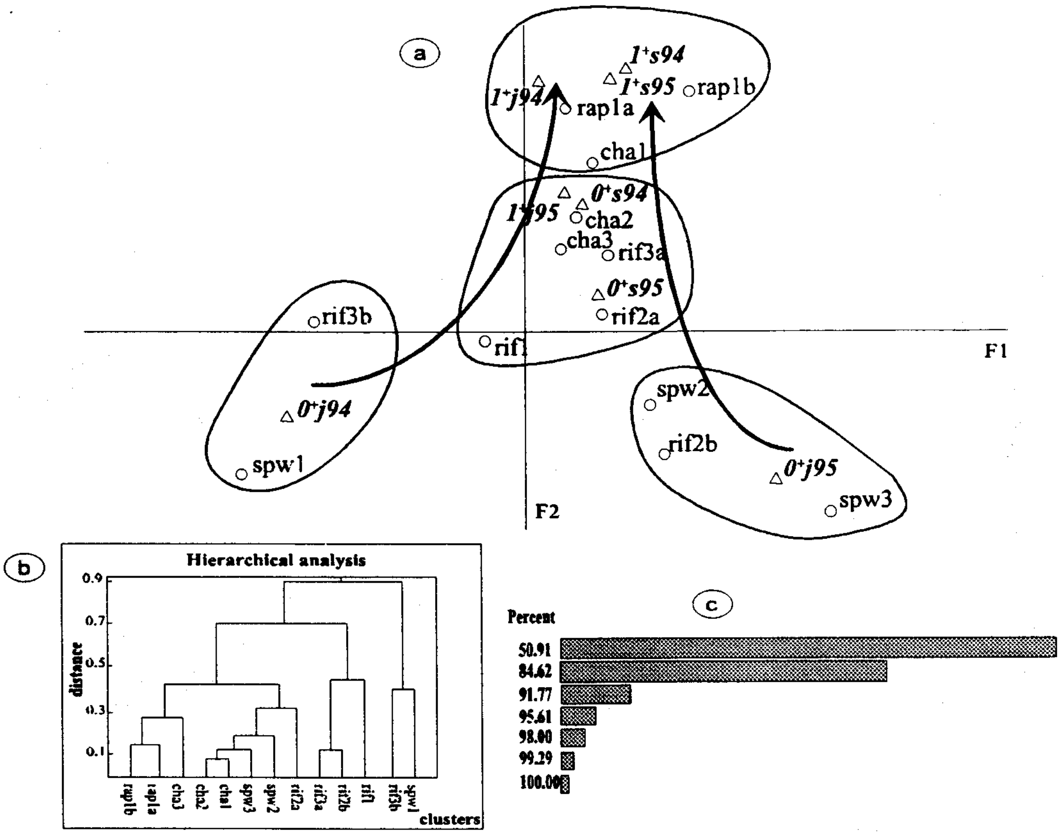

Figure 6: Ordination of habitat type by correspondence analysis. a: distribution of morphological units and stages of trout on the F1xF2 factorial planr; b: cluster analysis of the two firts coordinates F1, F2; c: histogram of eigenvalues with cumulative percent.

A significant correlation was found between the second axis (F2) and 1+ density in 1994 (1+s94, Rs=0.637). This axis opposed all spawning areas characterized by high 0+ density and morphological units with high current velocity and depth, well occupied by 1+ (rap1a and 1b, cha1).

Examination of the cluster analysis showed that four groups of morphological units can be separated: spw1 and rif3b; spw2, spw3 and rif3b; rap1a, rap1b and cha1 and cha2, cha3, rif2a, rif3a and rif1. Similar trends occurred for the two years.

Spatial and temporal variation of the YOY production was associated to the chronicle of monthly minimum WUA at each site and for the two years (Figure 7). As indicated for the discharges study, minimum WUA was noted in 1994 after emergence while it occurred before emergence in 1995.

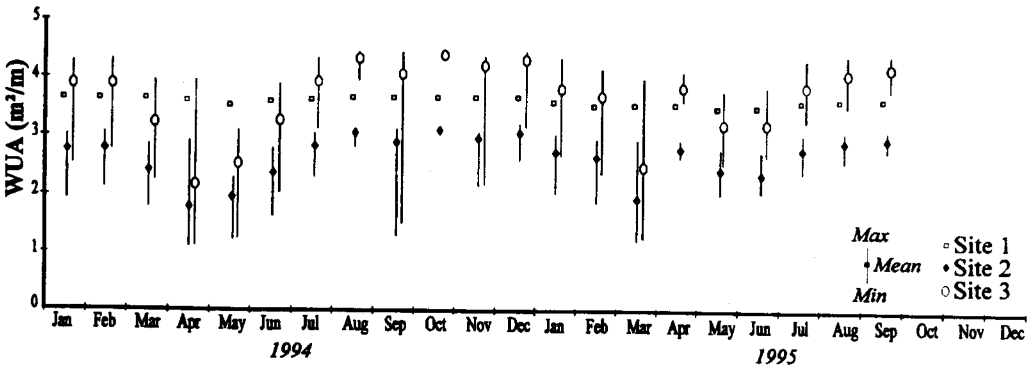

Figure 7: WUA time series for two years at the three sites.

Diminution of monthly minimum WUA for the regulated sites, in 1994, corresponds to low YOY production at these stations. At the same time, Site 1 shown the highest production. In 1995, high values of monthly minimum WUA, due to the early end of snow-melt in addition to absence of power generation, are associated with better fry production in downstream sites while that of natural site was low. For unregulated stations, monthly minimum WUA were virtually constant and could not be used to account for fry production.

DISCUSSION

Effects of the variation in temperature, flow and habitat have been studied and described by many authors (Garcia de Jalon *et al.*, 1988; Bain *et al.*, 1988; Gore *et al.*, 1989; Moog, 1993; Valentin, 1995; Lauters, 1995). The results of our study can be generalized into such adverse effects on early life history, population parameters and behavioural responses of the youngest brown trout stages.

On Oriege River trout life history at each site depends on the thermal regimes they were submited to. Even though mean temperatures were comparable, general thermic patterns differed between unregulated and regulated sites. However, they showed no difference in terms of cumulative degree-days undergone by eggs and larvae. So, the main emergence occurred for the whole river at about the same time. The three sites were therefore subjected to the same hydrological conditions and were only distinguished by their morphodynamic characteristics and the periods of power generation by the plant.

Growth rate indicated lower values in the regulated reaches than at Site 1. This agrees with the findings of Casado (1989). The thermal decreases recorded dowstream could be implied by their sudden fall to around 4°C. Exposure to this temperature for varying periods has been given, by Elliott (1994), to be the lower limit of growth bading to considerable as the fish size was small. However, the temperature fluctuations cannot be held responables for the mortality increase at Site 2. Indeed, thermal values were allways within the thermal tolerance polygon presented in Elliott (1994) for brown trout.

Under hydropeaking conditions, we found a lower density of YOY than that expected in an unregulated site. Nevertheless, in 1995 after a short reducing discharge period, summer densities were high in the downstream reach. Such results could be added to the critical period concept well developed by Elliott (1989). This authors demonstrated that during a short period after emergence called t_c, higher rate of density-dependent mortality occured. An explanation was found in establishement of feeding territories soon after emergence. However it was clear that physical factors were involved too (Ottaway and Forest, 1983; Heggenes, 1988b). On emerging from the gravel, young trout are weak and ungainly in the flows (Proctor, 1980). They rapidly learn static-swim (Heland, 1991) and are able to establish hunting territories characterized by minimal net-energy-intake (Fausch, 1984). Then they developed agonistic behaviour and defended social territories in hierarchical structures (Kalleberg, 1958; Mortensen, 1977; Elliott, 1990). During this first short stage of life, whereas density was dominating, habitat parameters play an important role in population regulation. Low discharge levels as encountered in May 1995 permitted YOY to used all territories available (high YOY WUA). A period of suitable conditions during the first week after emergence may reduce the high early mortality to a threshold corresponding to the maximum carrying capacity allowed by the suitable habitat available. However, such habitat is limited. Grant and Krammer (1990) reported that only 2 to 20% of the habitat was occupied by territories. So the number of fishes supported by a given reach is limited even so Jowett and Richardson (1994) reported sub-optimal habitat use for fishes under flooding conditions.

This explains the reduced difference in density found in our study between all sites after few months: fishes which were unable to occupy or defend a territory were displaced or washed out dowstream (Ottaway and Clarke, 1981; Titus and Mosegaard, 1992). For regulated stations, final densities may correspond to habitats or territories which remained suitable even under peaking generation (minimal YOY WUA).

This availibility of territories was closely related to flow and to the physical parameters of the reach and therefore of the morphological units. Our results indicated that young trout (0+ and 1+) in natural reaches (Sites 1 and 3) showed preferences for morphological units, i.e. obeyed mesohabitat logic (Baran, 1995). This logic depended on intrinsic trout preferences for certain hydrological and physical parameters (Bovee, 1978) found in certain morphological units (Orth and Leonard, 1990). These preferences are dependent on fish age and therefore, on swimming ability. Young-of-the-year appear on spawning areas (runs). They rapidly colonise adjacent riffles, then at the end of summer, they rather choose channels and riffles. The year after (being 1+), their preferences more closer to those of adults: higher depth and velocity (Cunjak and Power, 1986), they shifted to rapids.

Under strong hydropeaking conditions (Site 2 in opposition to Site 3) mesohabitat shifting behaviour disappeared for yearlings and all the young trouts were so disrupted that they were unable to keep their preferences and showed no distibution logic for the mesohabitat. We hypothesize that the youngest trout stages use a territory pattern (on a microhabitat scale), which remained suitable under all hydrological conditions. Such a microhabitat template could exist in every morphological unit even in low numbers and allow a basic density to persist in each reach. Research based on this hypothesis should now focus on the parameters of the microhabitats as well as on the biological responses of the juveniles on a more individual level.

The difference between Site 2 and 3 can not be explained by broadening of the of flow peak wave, the distance between them being too short. However the Site 3 shows a lower mean slope and is wider than Site 2. So for the same hydropeaking generation occurring at the two sites, snout velocities and depth will be lower at Site 3 causing a flattening of the surgeand explaining the decreased adverse effects at Site 3. Moreover, both varying factors (mean depth and mean slope) are important in the calculation of Tractive force (Newbury,1984; Death and Winterbourn, 1994). In further studies we should use this indicator of stream bed stability (providing flow shelters to young trout) to confirm those results.

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THE EFFECT OF MANAGED HYDROPOWER PEAKING ON THE PHYSICAL HABITAT, BENTHOS AND FISH FAUNA IN THE BREGENZERACH, A NIVAL 6TH ORDER RIVER IN AUSTRIA.

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ABSTRACT

The lower reaches of the Bregenzerach, a river in western Austria, have been, for several decades, affected by heavy hydro-peaking discharges (up to 60 m^3/s). In combination with a plan to construct a new power plant, mitigation measures were developed to reduce the adverse effects of hydro-peaking in this river reach.
The new powerplant Alberschwende, completed in 1992, includes a re-regulation reservoir to improve flow management. An impact assessment study examined the effects of this new construction and flow management system, comparing data collected before and after the construction of the new powerplant. The study concentrated on three main components: abiotics (morphology, hydrology, hydraulics), fish, and benthic fauna. Four representative sites were examined within the hydro-peaking area and two sites within a reference area located upstream.
Prior to mitigation, both fish and invertebrate biomass was less than 5% of what would be expected. After mitigation, when the amplitudes of the surge releases were reduced, benthic biomass recovered to about 60% of the expected value. There was no significant change in the benthic community composition except for a slight increase of grazing organisms. No post-mitigation improvement was found in respect to fish biomass. Although the amplitudes of the surge releases were reduced their frequency remained the same. This frequency apparently has a strong negative influence on the fish community.
Flow velocity distribution data showed that rising surge releases had two distinct phases, a "bed-filling" phase and a phase of accelerating flow velocities. The present flow management system, based on a dual flow logistic, results in increased base flows and reduced peak flows, but does not alter the ramping rates. The reduction of the adverse effects of the "bed-filling" phase was apparently responsible for the recovery of the benthic fauna. However the unaltered ramping rate of the "acceleration" phase seemed to inhibit the development of the fish community.
An analysis of the hydrograph demonstrated the possibility of adjusting the present management strategy to reduce the ramping rate of the "acceleration" phase, smoothing out the peak duration curve. This modification, accomplished with existing facilities, could better mitigate the hydro-peaking impacts and should be evaluated in the future.

Key-words: Peaking Hydropower / Artificial Floods / Impact Mitigation / Flooding Frequency / Physical Habitat / Fish / Benthic Invertebrates / Austria

INTRODUCTION

Hydropower peaking operations produce artificial flood events that, due to their unpredictability and intensity, can be classified as disturbances (sensu Resh et al., 1988) directly affecting aquatic ecosystems of the tailwaters. Frequent, high fluctuations in depth, velocity and the wetted perimeter, which follow unnatural schedules of power generation, cause entrainment and stranding of many aquatic organisms. These spates often result in an impoverishment of aquatic biota. Numerous studies document the effects of these hydraulic changes on the behavioral response and distribution of aquatic fauna (Barwick and Hudson, 1985, Cushman, 1985, Gore et al., 1990, 1994, Hudson and Nichols, 1986, Layser et al., 1989, Niemela, 1989). Zones of fluctuation may not be suitable for terrestrial or aquatic fauna. (Fisher and La Voy, 1972). Long-term, regular, peaking operations also typically change substrate, depth and velocity patterns in the tailwaters. These changes in substrate have direct effects on the composition of aquatic community (Gore and Pennington, 1988). Temperature and water quality changes may also have a strong influence on the tailwater fauna (Hunter, 1992, Cushman, 1985).

Hydro-peaking impacts can be classified based on the intensity and duration of surge releases (Resh et al., 1988). The negative correlation between the benthic biomass and the base/generation flow relation is presented by Moog (1993). However, the impact of the peaking wave is not constant over time. The direct impacts of the falling limb of a surge release are better described than the potential impacts of the unnatural flow increase (Cushman, 1985, Hunter,1992). This may be due to methodological problems.

The initial rise in discharge is characterized by turbulence and rapid changes of physical habitat, which often result in much higher drift density than during the generation flows (Matter et al., 1983a,b). Often, habitat changes occur faster than the organisms can adjust to the new conditions (Gore et al., 1989). Natural floods supply „warning signals" e. g. the rise in ground water level before flood wave, that would allow organisms to make appropriate behavioral responses (Bretschko and Moog, 1990).

Gore et al. 1994 analyzed the change in shear stress during peaking, recognizing a general pattern of changes. Within the low water thalweg the FST hemisphere (Statzner and Müller, 1989) values at first increased dramatically and then decreased when the generation flow was reached. However in isolated pools and places that are minimally wetted during base flows, shear stress increased with the surge and remained high during generation flows.

The rising limb of the hydrograph causes critical hydraulic changes; increasing flow velocities and a broadening of the wetted area. Rapid decreases in flow, affect the fauna by reducing the wetted area. This means that the frequency of the peaking events and the ramping rates are more important than the absolute duration of the surge release.

The possibility of mitigating these impacts by reducing the described intensity and frequency of peak releases is considered in this paper. Two peaking flow management options, single release (pre-mitigation) and dual flow release (post-mitigation), is evaluated as a part of an environmental assessment study. The results will be used to develop an adaptive flow management plan.

MATERIALS AND METHODS

Project area

Bregenzerach (BA) is a 4th to 6th order alpine river with a nivalic discharge regime, located in the most western part of Austria. It originates at the elevation of about 2000 m above sea level (ASL) and flows 68 km before emptying into Lake Constance (ca. 400 m ASL). The river's discharge regime is typically very dynamic. The mean annual flow (MAF) ranges, through the length of the study reach (the lowest 40 km of the river), from 13 to 47 m^3/s, with min./max. ratio of 1:400.

The main human impacts on this system are caused by hydro-power operation, weirs and some channel regulation. Until 1986, the lower part of the BA was primarily affected by two hydropower plants. The uppermost plant, Andelsbuch, is located at river kilometer 28 (counted upstream). This plant used the entire flow of the BA (up to 29 m^3/s, or, 200% MAF) and produced peaks up to 30 m^3/s a few times a day. Seven kilometers downstream another

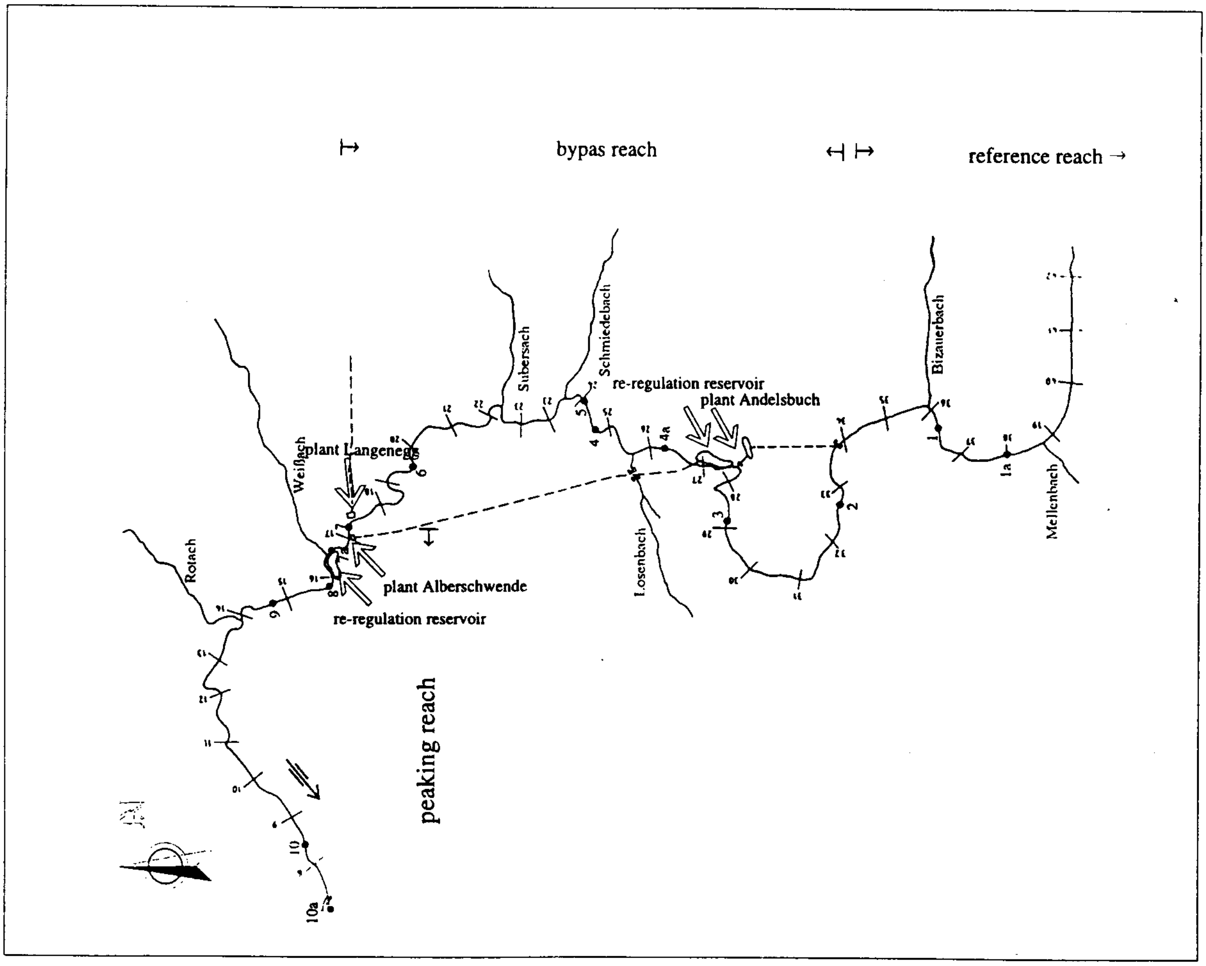


Figure 1: The situation plan of the study site on Breganzerach

plant, Langenegg, still uses the water of a tributary of BA - Bolgenach - adding another 30 m³/s surge to the total discharge. The peaking amplitude below Langenegg was thus 1:60. These peak flows heavily impacted the aquatic fauna in downstream reaches (Jungwirth et al., 1990). The water quality of the tailwater was not affected by the plants.

The following management and development plan was realized in 1992 (fig. 1). A re-regulating reservoir (350 000 m³/s) was built below Andelsbuch. The water is used for a new run-of-the-river plant, Alberschwende, constructed beside Langenegg. The bypass reach receives seasonally varied compensation flows (Schmutz et al, in prep). Below these two plants the second re-regulating reservoir was constructed. The capacity of this impoundment is 150 000 m³ with a maximum in and outflow of 60 m³/s. Additionally, a flow management system for the reduction of peak amplitudes was introduced. The concept of this system is based on the dual flow logistic (Milhous, 1991) to assure a higher base discharge for at least one day before peaking. Figure 2 shows the relation between permitted generation flows and base flows; these discharges were based on professional judgment.

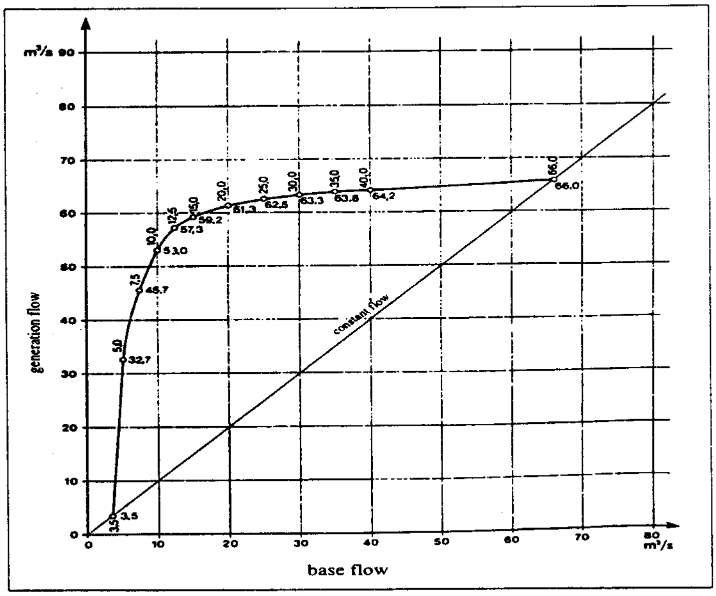


Figure 2: The dual flow generation schedule

Study sites.

Several representative sites were located within study area, where abiotic and biotic data were collected (fig. 1). The reference sites 1 and 1a are located by kilometer 38 and 36 of BA, upstream from the hydro-power plants. Sites 2 - 6 are located in the present abstraction area and will be not discussed further in this paper.

The first site influenced by peaking operations (site 7), is between the outlets of Langenegg and Alberschwende which are only 300 meters apart (fig. 1). This site receives rapid surges from upstream plant, Langenegg. These surges are often mitigated by backflow from generation discharges of Alberschwende at the lower end of the site. The site 7a is in the bypass reach of the re-regulation reservoir about 100 m below site 7. Sites 8 and 9 are located about 1 km and 2 km respectively, below the outlet of the re-regulation impoundment, and site 10 is 10 km downstream. The most downstream site, 10a, is located shortly before BA enters Lake Constance.

Benthos

Benthic invertebrates were collected in four sites (1, 7a, 8, 10a) with a modified surber sampler (box-type sampler, 1000 cm² sampling area and 112 µm-sized mesh). Cobble areas were sampled down to a depth of about 30 cm into the sediment. At least 6 replicates were taken. With the aid of vertical sampling it could be shown that this depth range covers most of the benthic invertebrate distribution.

The vertical distribution of benthic animals was determined with a modified Stocker-Williams-N2-frezeeing corer (Bretschko and Klemens, 1986). Seven replicate samples were taken at sites 6 and 7 during September 1986 (Bretschko and Moog, 1986). In 1994, a total of 14 samples were taken. The distribution range of benthic invertebrates is shown in figure 3 and clearly demonstrates that most of the animals inhabited the first 20 cm of the bed sediments

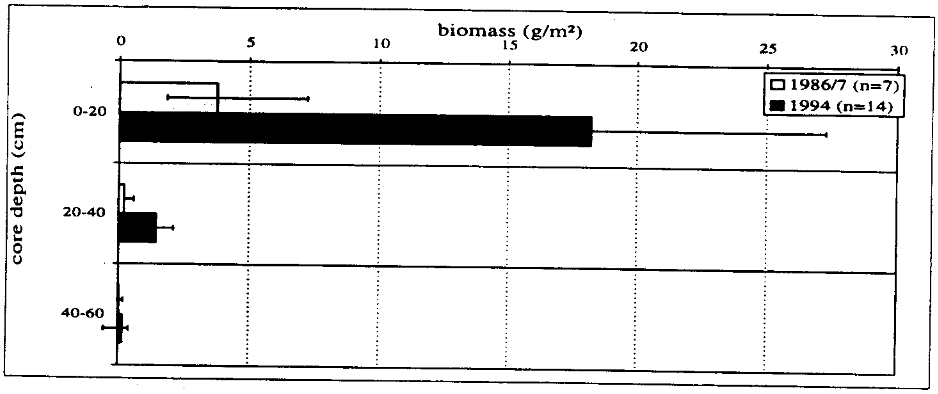


Figure 3: Depth distribution of macroinvertabrates

Samples were processed in 4% formalin. At the laboratory, the organisms were separated, identified to the lowest determinable taxonomic level, counted and weighted. For biomass determination, the fresh weight was determined, using filter paper to remove the excess water. The organisms of the second sample series in 1994 were determined to a higher taxonomic level only.

Fish

Surveys of the fish fauna were carried out to evaluate the hydro-peaking impacts before (1992) and two years after the new power plant Alberschwende was in operation (1994).

In 1992 and 1994 5 stretches were sampled (1,7,8,9,10). In 1994 an additional stretch was added (1a) to increase the representativness of the samples; all stretches were fished at low flow conditions in autumn. A 3 pass removal method with 3 backpack electro-shockers (DC, 600 V, 1.5 kW) was used. The upper end of the stretch was blocked with an electric fence. Areas deeper than 1.5 m (in stretches 7, 8, 9) were additionally sampled by boat with a hand-held anode (DC, 600 V, 2.5 kW). Stock densities estimates (ind./100 m, ind./ha) and total biomass (kg/100 m, kg/ha) were done according to De Lury (1947).

Morphology / hydraulics

The reference sites (1, 1a) and tailwater sites (7, 8, 9 and 10) were examined at base low flow in autumn 1993. Morphology was surveyed using systematic irregular sampling method (Parasiewicz, in press.). For description of channel velocity distributions, one representative cross-section per site (impacted sites only) was measured. Measurements were taken with a OTT propeller at four different flows within the peaking range. Typical pre- and post-mitigation peaking regimes were simulated. Changes in the water surface elevation (WSE) were recorded at temporary gauging stations located within each transect. This gave the stage/time and discharge/time relations during peaking events.

Additionally, the change in the WSE and mean cross-sectional velocities were modeled for each site using the step-back-water model.

RESULTS

Benthos

Prior to the mitigation project the benthic fauna was dominated by ephemeropterans, plecopterans and dipterans. The dominance structure of the 105 taxa sampled clearly matches that of a „*Baetis alpinus-Rhithrogena hybrida*-assemblage“ according to Braukmann (1987) or, a submontanous community type B and C (Kawecka et al., 1971)

Post-mitigation sampling in the reference site revealed no significant change in the taxa composition. In the downstream reach oligochaetes and chironomids increased. Figure 4 shows the taxa composition, and figure 5, biomass for the BA during the study period.

In 1986/87, biomass in the reference reach was much higher (10 g/m²) compared to tailwater sites (1.35, 2.68 and 5.55 g/m²). Two years after the new operation system was in place, biomass in the tailwater sites increased dramatically. In sites 7 and 8 biomass increased three-fold and in site 10a two-fold.

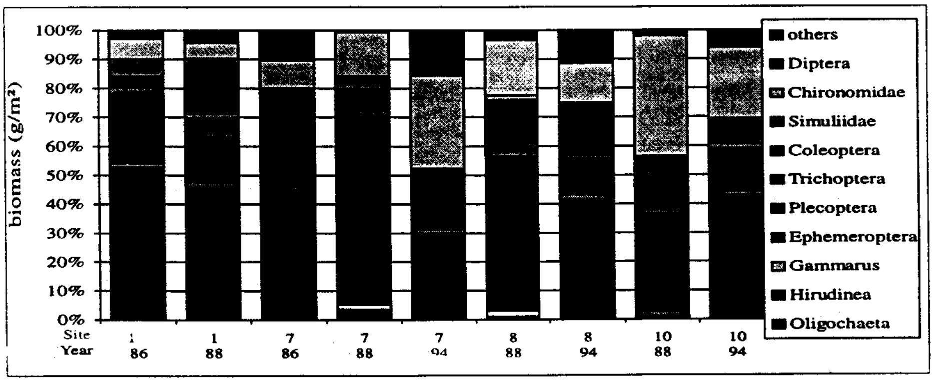


Figure 4: Taxa composition of benthic fauna in the study sites

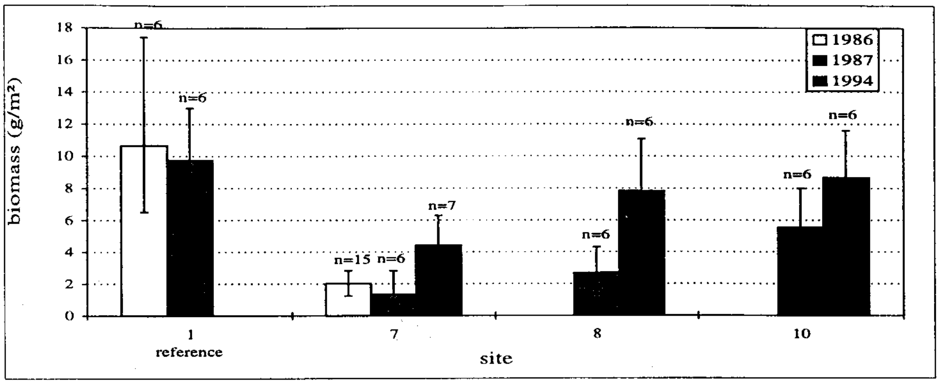


Figure 5: Benthic biomass in the study sites

<u>Fish</u>

Fish assemblage zonation is apparent in the investigation area. The upper reach (sites 1-3) is a typical headwater river where brown trout (Salmo trutta f. fario, L.) and bullhead (Cottus gobio, L.) occur exclusively. In the middle reach (sites 4-6) these species still dominate and additionally a few specimens of rainbow trout (Oncorhynchus mykiss, WAL.), European grayling (Thymallus thymallus, L.) and stone loach (Noemacheilus barbatulus, L.) are found. In the tailwater (sites 7-10) barbel (Barbus barbus, L.) and blageon (Leuciscus souffia agasizzi, RISSO) additionally contribute to the assemblage indicating a slight shift to the barbel region.

Figure 6 shows the estimated population densities before and two years after the initial operation of the power plant Alberschwende. Figure 7 shows the biomass of study sites for both years. In 1992, total density and biomass in the tailwater reach varied between 0-59 ind./ha and 0-12.4 kg/ha, demonstrating the tremendous adverse effects of hydro-peaking. Two years later the densities (7-50 ind./ha) and the biomass (1.5-3.1 kg/ha) remained, in these sites, at a very low level. Site number 8, with a biomass of 26.4 kg/ha represents an exception. A large deep pool with rocks along the bank obviously provides a refuge area during peaking events.

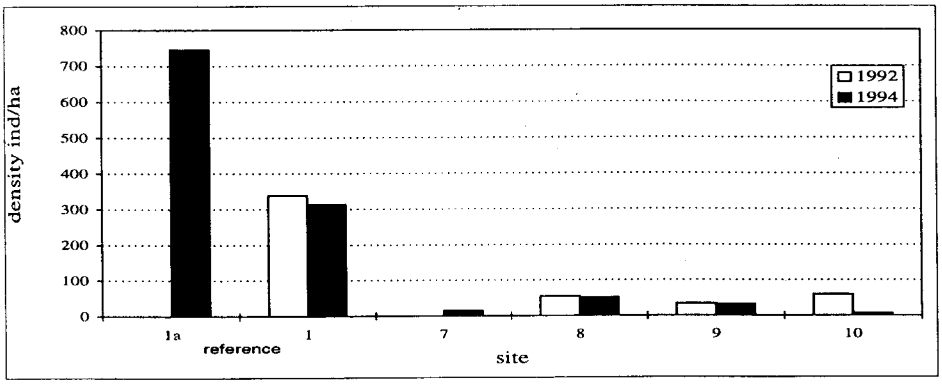


Figure 6: Estimated population dencities of the fish fauna in the study sites

In 1994 the stock in the reference site 1 remained at the 1992 level (314 ind./ha or 21 kg/ha). Attempts to support the fish populations with stocking failed. The annual stocking rate per 100 m included: about 4.7 cacheable and 40 1+ brown trout, 3.2 cacheable rainbow trout and 39 0+ European grayling. The estimated population densities however are only small fractions of the number of stocked fish.

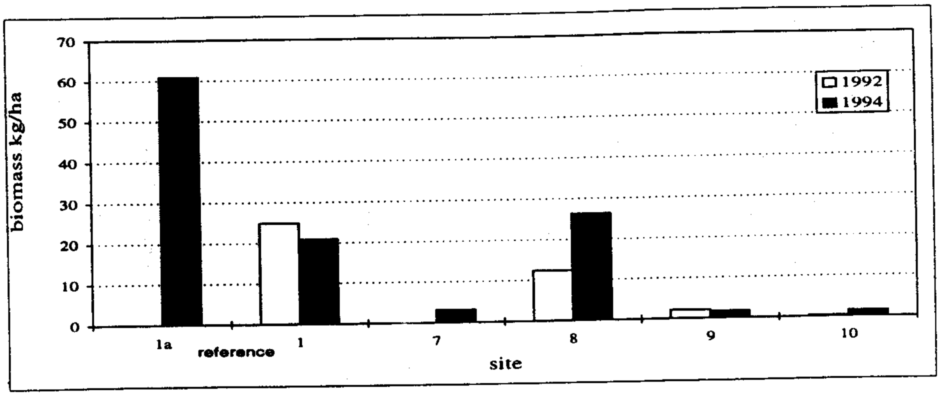
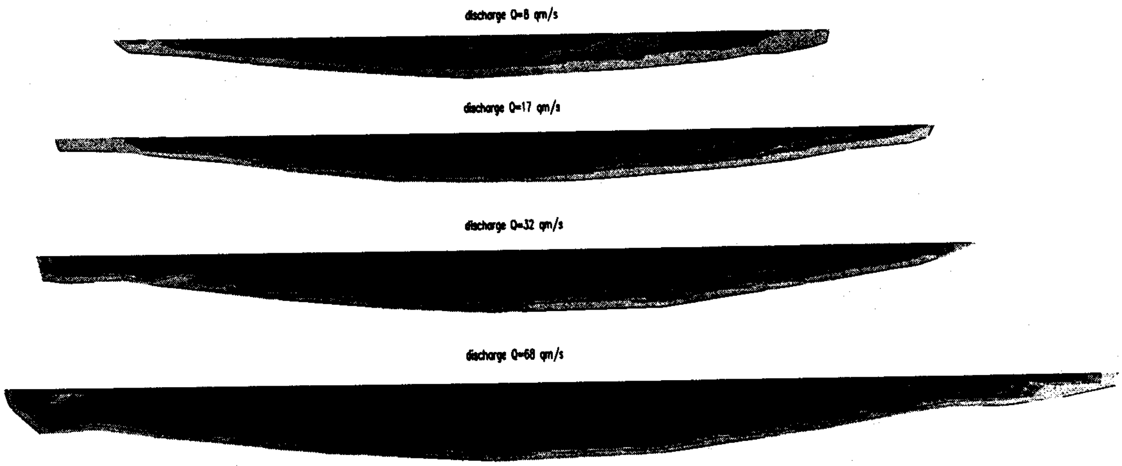


Figure 6: Estimated biomass of the fish fauna in the study sites

<u>Morphology / hydraulics</u>

Based on our habitat utilization curves (Schmutz et al., in prep) as well as the literature (Bainbridge, 1960, 1962, Jones et al., 1974, Heggeness, 1988), the wetted transect area was divided to three velocity classes. Velocities between 0 and 40 cm/s were the most suitable; between 40 and 100 cm/s were acceptable; and critical velocities were found to be above 1m/s. Critical velocity areas are shown in black (example diagram Figure 8). There is a rapid increase in these areas between base (10%) and maximum flows (70%) which is visible in all measured transects. At base flows, suitable



			Q=8 qm/s		Q=17 qm/s		Q=32 qm/s		Q=68 qm/s	
shade	from (m/s)	to (m/s)	area (m2)	%	area (m2)	%	area (m2)	%	area (m2)	%
	0.00	0.20	2.42	15.6	2.02	9.2	2.25	7.5	2.08	4.6
	0.20	0.40	2.54	16.4	2.04	9.2	1.52	5.0	1.70	3.8
	0.40	0.60	6.61	42.7	2.40	10.8	1.62	5.4	1.71	3.8
	0.60	0.80	3.90	25.2	3.68	16.7	2.45	8.1	1.74	3.8
	0.80	1.00	0.00	0.0	7.41	33.5	3.02	10.0	2.22	4.9
	1.00	3.40	0.00	0.0	4.55	20.6	19.25	63.9	35.69	79.1

Figure 8: Velocity distribution at measured discharges in representative cross-section of site 8

velocity areas can be found in 40% of the profile compared to only 10% of the profile at generation flows. This optimal area remains only at the bottom and littoral zones.

In general, change in the river's geometry (primarily in width) is more intensive at the very beginning of the surge, before the riverbed is filled with water. Afterwards, flows change primarily in terms of velocity. This is visualized in curves fitted to the velocity and volume increase data calculated for all sites with a step-back-water model (fig 9). Growth in volume is, in this case, primarily lateral as opposed to vertical. The increase in water surface area is also presented in the figure 9. Differential curves show the rate of change for each parameter (fig. 10); these expressed rates allow the following generalizations.

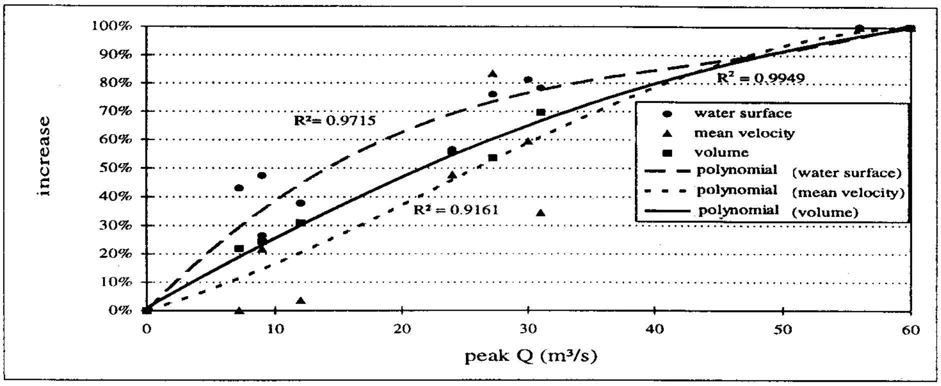


Figure 9: Curve fit of increases of main parameters during the surge

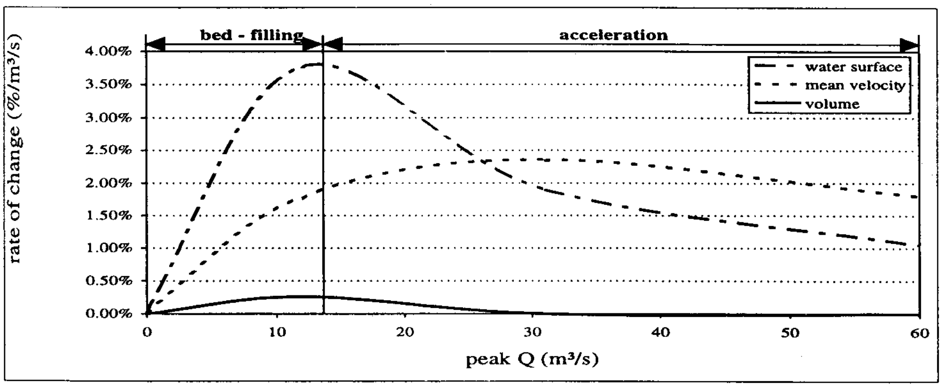


Figure 10: Rate of changes of main parameters during the surge

The rising limb of the peaking event can be divided into a „bed-filling" phase, occurring during the first 15 m³/s of flow increase, and, an „acceleration" phase occurring afterwards.

DISCUSSION

In Austrian rivers, hydro-power peaking has resulted in three distinct kinds of impacts on the benthic invertebrate composition and density (Moog, 1993). Total extinction, except of the deep interstitial species (river Drau); significant change of the community structure without biomass reduction (river Salzach); reduction of biomass without a change in the taxa composition as it was a case at Bregenzerach.

Morgan et. al. (1991) observed that less intensive ramping rates result in increased abundance of macroinvertebrates in downstream sections. This same tendency could be observed, studying the biomass of benthic invertebrates, in the Bregenzerach.

Based on studies of 49 alpine Austrian rivers, in which there was no hydro-power peaking influence, an inverse correlation between altitude and benthic biomass was found (Moog, unpublished data). This calculation allows estimation of expected biomass values along the similarly structured study reaches. The biomass values of the reference site are within the confidence interval of expected . Pre-mitigation biomass values in tailwater reaches (1986/87) were only 5% of expected values. Post-mitigation (1994) benthic biomass in the tailwater was about 60% of expected values and also within the confidence interval. In addition, higher abundance of drift-sensitive oligochaetes and chironomids, suggests that new flow management benefits this portion of the fauna (Irvine and Henriques, 1984).

Fisher and La Voy (1972) show that periodically exposed areas of the riverbed have lower invertebrates density and diversity than communities in constantly flooded areas of the riverbed. Apparently, on the BA, reducing the intensity and frequency of the „bed-filling“ phase of the surge (hence the exposure frequency of littoral area) had a positive effect on the benthal organisms.

In near-natural Austrian rivers, with a stream order of 4-6, we expect a total fish stock density of about 1000-2000 ind./ha. At a stream order of 4 or 5 the expected biomass is about 170 kg/ha and increases to more than 400 kg/ha at stream order 6. In general 4th and 5th order streams are inhabited by grayling which represents about 1/3 of the total biomass (Schmutz, 1995).

When we compare the actual data with the expected stocks at the BA, it is apparent that in 1992 the total stock in reference site 1a, which is typical for 4th order river reach, has only half of the expected density or one third of the expected biomass. Brown trout populations meet expected values but grayling are found in comparatively low numbers. The stock density and biomass in site 1 (1992 and 1994) are very low, probably due to the low diversity of instream structures. Therefore we suggest that site 1a represents a more realistic estimation of a natural brown trout population in a 4th order reach of the BA.

Pre- and post-mitigation sampling in site 7-10 reveal very few fish. Some larger brown trouts and barbels were caught at site 8 presumably due to the presence of large flow refuge areas. However, the expected biomass of about 400 kg/ha, which would be typical for a 6th order stream can not be achieved.

Although Weisberg and Burton (1993) reported the positive influence of a higher base flow on the fish fauna, we could not demonstrate any benefit to the fish fauna on the BA. Apparently, the frequent and rapid reduction of suitable habitat due to rapid changes in flow velocity, is responsible for the impact on this biota. It is clear that the area of critical velocity starts to expand very rapidly with the onset of „acceleration“ phase (fig. 8). At the end of this phase the remaining low velocity areas are primarily found in shallow littoral zones, which are often unavailable to fish. These areas are also of high risk due to the subsequent and rapid flow reductions, which can potentially strand organisms.

The introduced dual flow release management divides the ramped releases into two parts. First, base flows are increased in proportion to the planned generation flow within a 24-hour period prior to a surge release. The magnitude of this base flow is approximately set to fill the bed. Since the planned surge releases are relatively frequent, this base flow remains relatively stable compared to the pre-mitigation situation. The "acceleration" phase, takes place with the same frequency as before modification, but has a lower absolute magnitude.

The average (discharge-based) ramping rate of pre-mitigation peaks, was up to 4 m³/s /min. Due to the lower flooding intensity, the ramping rate of the „bed-filling" surge is reduced by more then 75%. Because the river bed is already full, turbulence and flow acceleration along the bed is also far less then under the prior management scheme. Nevertheless the ramping rate of the "acceleration" phase is only reduced by 25%. Further reduction of this ramping rate could therefore have positive effect on the fish fauna. The example below shows that this could be reached using existing resources without an economic loss.

The analysis of the 1993 hydrograph showed that flows were not managed to their fullest mitigative potential. The following plot shows the daily flow from 29. January 1993. This curve is typical for winter, a critical period due to low flows and high energy consumption. Without changing the energy production for this day and adhering to all existing restrictions a more efficient flow management was calculated. It results in the discharge curve for the same day that is much more stable and has much less intensive peaks. (fig. 11)

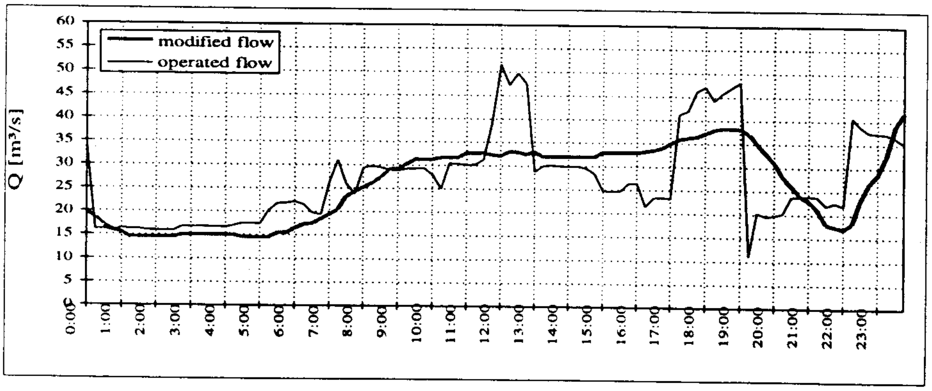


Figure 11: Regulated discharge in the downstream area on 29. January 1993 and suggested modifed flow

This curve reduces the ramping rate and frequency of the „acceleration" phase. The optimized use of existing resources could significantly mitigate the human impact on the Bregenzerach and should be examined in a following monitoring project.

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INFLUENCE OF HYDROPEAKING ON THE STRUCTURE AND DYNAMICS OF INVERTEBRATE POPULATIONS IN A MOUNTAIN STREAM

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ABSTRACT

The invertebrate communities, the population structure, life history and larval growth of 22 common invertebrate species (including Ephemeroptera, Trichoptera, Plecoptera and Diptera Simuliidae) were studied in the River Oriège (Pyrénées, France), 700 m upstream and respectively 700 m and 3000 m downstream from a hydroelectric power plant. The river receives hypolimnetic water diverted from a nearby reservoir lake and the natural flow may be enhanced several times a day from 1 to 11 m^3 s^{-1} in summer and winter, and from 5 to 15 m^3 s^{-1} during spring spates. During hydropeaking, streamwater is slightly warmed in winter and strongly cooled in summer. Peak flows decreased both larval density and biomass of most species below the hydroplant by increasing the frequency and intensity of bed scour, which lead to a high catastrophic drift. Significant differences in mean annual population structure were found in 10 species with a shift towards larger larvae 700 m downstream of the plant in 8 species and a shift towards smaller larvae in 2 species. The losses resulting from catastrophic drift were partially compensated by larvae drifting from the unregulated zone during spring spates. This explained the dominance of larger individuals for some species immediately downstream of the outlet. None of the life history was fundamentally modified below the power plant. Rapid and frequent changes in water temperature and current velocity caused slight changes, affecting hatching, emergence periods, and growth rates of several species. In spring, snow melt floods minimized the impact of peaking flows as the growth of most species occurs during this period at this elevation. The new thermal regime below the outlet, and particularly an increase in winter temperatures allowed some species to have a low but continuous winter growth, whereas their growth above the hydroplant stopped in winter (e.g., *Rhithrogena semicolorata*). These species occurring preferentially usptream, we suggest that such thermal modifications could advantage stenothermic species, and help them survive in the lower course of the stream. Specific ecological changes due to hydroelectric facilities depend on operation and construction variables; the repeated return to the natural flow in the River Oriège reduced the downstream effects of the peak flows.

KEY-WORDS: Aquatic insects / Benthos / Hydropower / River regulation / Stream flow / Temperature / Population structure / Life history / Growth / Pyrenees.

INTRODUCTION

Studies of peak flows in stony streams have chiefly examined the effects of water discharge on the density and species richness of macroinvertebrate assemblages (e.g., Kraft and Mundahl, 1984; Cobb and Flannagan, 1990; Troelstrup and Hergenrader, 1990; Cobb *et al.*, 1992). However, although population structure is related to life history patterns, only few studies address the influence of hydropeaking on the larval development of invertebrates (Mundahl and Kraft, 1988). When the hydroelectric facility creates changes in flow and temperature regime, the life history of the invertebrates can be affected (Sweeney, 1984; Raddum and Fjellheim, 1993). Temperature is the most important factor influencing growth and development of lotic species (Elliott, 1978; Sweeney, 1978; Humpesch, 1979; Ward and Stanford, 1979, 1982; Vannote and Sweeney, 1980; Kondratieff and Voshell, 1981; Sweeney *et al.*, 1986; Pritchard and Zloty, 1994). High current velocities are believed to suppress aquatic insect growth rates, as increased physiological demands for position maintenance result in less energy channeled into growth (Hynes, 1970; Kovalak, 1978; Mundahl and Kraft, 1988). Moreover, flow regime provides the spatiotemporal variability of stream habitats that influences the community structure and the demography of the populations (Feminella and Resh, 1990; Moog and Janecek, 1991; Robinson *et al.*, 1992). However, the specific changes due to impoundments depend on a complex series of interactions resulting from operation and construction variables (position of outlet, purpose of the reservoir, frequency of releases). Streamwater below most hydroelectric impoundments consists entirely of reservoir water (Ward and Stanford, 1979, 1983; Brittain and Saltveit, 1989). The setting for our study is quite different: the hydroelectric power plant preserves the natural flow of the river, which is only supplemented by hypolimnetic water releases diverted from a nearby reservoir. The purpose of this work was to compare the abundance, biomass, life history and larval growth of several common invertebrate species upstream and downstream of such a hydroelectric facility, which generates high daily fluctuations in river temperature and flow, but maintains the natural conditions when the hydroplant is inoperative.

STUDY AREA, MATERIAL AND METHODS

The River Oriège is a torrential stream in the French Pyrenees (Fig. 1). Its source is at 1700 m above sea level, and it flows into the River Ariège at 815 m a.s.l. At 912 m a.s.l., the Oriège's natural flow is supplemented by the discharge from the Orlu hydroelectric plant, which is fed by hypolimnial water from a high altitude reservoir (Lake Naguilles, 1855 m a.s.l.). Three sampling sites were selected: a reference site 700 m upstream of the hydrostation's discharge point (site A, 920 m a.s.l., stream width=8m), and two sites respectively located at 700 m (site B, 880 a.s.l., stream width=17 m) and 3000 m downstream of the plant (site C, 820 m a.s.l., stream width=21 m).

Discharge was recorded hourly in 1991 using hydrographs at site A and at 1500 m downstream of the hydroplant. Water temperature was recorded hourly from June 1991 to July 1992 at sites A, B, and C, using submerged thermographs.

At least five benthic samples were collected monthly at each site from June 1991 to July 1992 using a Surber sampler (sampling area 0.1 m², mesh size 0.3 mm). The samples were taken from the various substrate types: gravel (2-20 mm), pebbles (20-200 mm) and cobbles (>200 mm) and were distributed proportionally to the relative abundance of the substrate types. Thus, the sampling provides the best qualitative (all substrates were sampled, maximum species richness) and quantitative (Surber sampler, replicate samples) estimates of the populations, taking into account the rather low numbers of samples. Simuliidae larvae have a contagious spatial distribution and studying their life cycles requires a specific sampling technique. They were collected by hand from rocky substrates

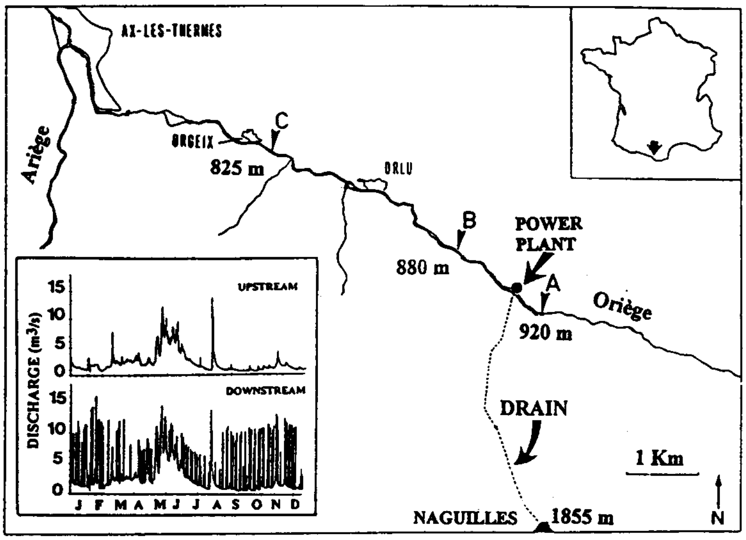


Figure 1: Map of the River Oriège, location of the three sampling sites (A, B, C), and hourly discharge recorded at site A and 1500 m below the power plant (year 1991).

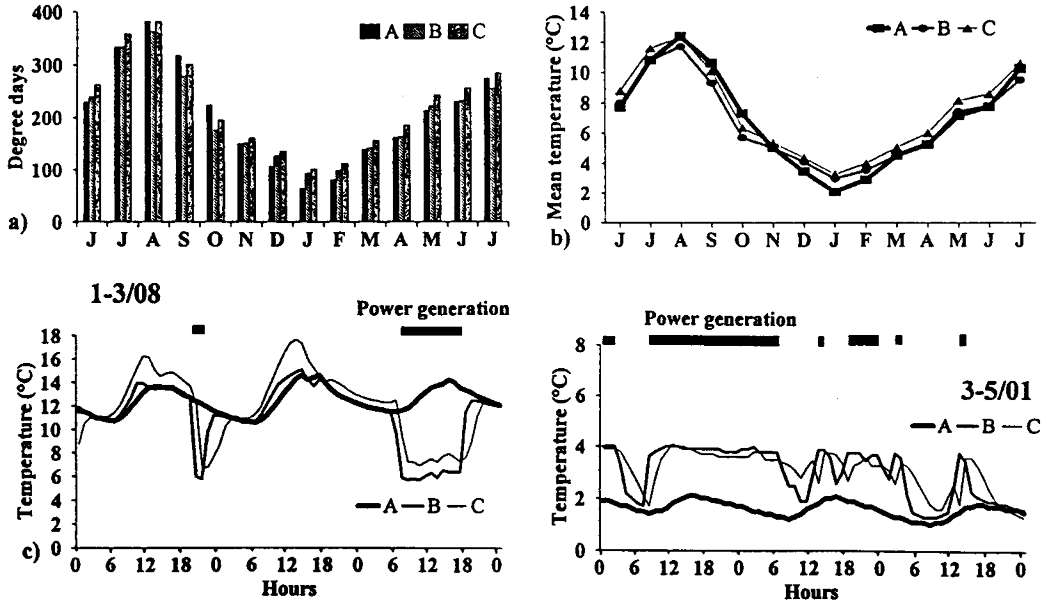


Figure 2: Thermal regime of the Oriège and influence of hypolimnetic releases (June 1991-July 1992): a) total number of degree-days monthly accumulated at sites A, B, and C; b) mean monthly temperature (°C) at the three sites; and c) examples of diel fluctuations in streamwater at sites A, B, and C over 3 days in summer and winter.

over one more year in 1995, monthly from January to April, and bimonthly from May to December. Unfortunately, these exclusively qualitative samples could not provide density estimates. The invertebrates were preserved in the field in 5 % formaldehyde, identified in the laboratory and preserved in 70 % alcohol. All larvae were divided into 0.1 mm interval size classes. The size criterions were chozen to reduce the variability resulting from sexual dimorphism or by convenience: tibia length of the third leg (*Rhithrogena* species), pronotum width (*Perla grandis*), head capsule length (Trichoptera and Simuliidae), head capsule width (other species). The distributions of the size classes were used to determine the five larval instars of Trichoptera species, the life cycles and the mean annual structure of the populations of all species at the three sites. The mean biomass of individuals within size classes was obtained by drying at least 10 individuals at 60°C for 24 hours.

Drift was used to determine the influence of peak flows on animal transport and bed scour. 24 samples were collected using a drift net (mesh size 0.3 mm) at 1-hour intervals, over two 24-hour periods (in June and October 1991) at sites A and B. The results are expressed as intensity of drift (Bournaud and Thibault, 1973), ind m^{-2} 30 min for invertebrates, Kg m^{-2} h for mineral material. Water conductivity was monitored at both sites during drift sampling.

RESULTS

Discharge And Temperature

The natural mean flow of the Oriège varied from 1 $m^3 s^{-1}$ in winter and summer to 15 $m^3 s^{-1}$ during snowmelt (Fig. 1) with hydroplant discharge supplementing this flow by 5 $m^3 s^{-1}$ (in spring) or 10 $m^3 s^{-1}$ (in winter or summer). The mean water depth increased from 30 to 80 cm at site B and from 30 to 60 cm at site C. The mean bottom current velocity increased from 30 to 120 cm s^{-1} at site B, and 30 to 80 cm^{-1} at site C. The natural temperature of the river ranged from 0 to 13.5 °C at site A, from 0.4 to 13.8 °C at site B, and from 0.7 to 17.5 °C at site C. During hydropower generation, the input of hypolimnetic water (4 °C) downstream of the plant slightly increased the winter temperature and strongly reduced the summer stream temperature (see examples in Fig. 2). Thus, mean water temperature and total degree days (dd) at sites B and C where lower than at site A from August to October 1991 (927 dd at site A, 818 dd at site B, 881 dd at site C), and higher from November to July 1992 (1425 dd, 1486 dd and 1640 dd at site A, B, and C respectively) (Fig. 2).

Invertebrate Drift And Mineral Transport

Site A showed a constant invertebrate drift due to accidental dislodgement, and behavioral drift was mainly nocturnal (Fig. 3). The mineral transport was very low (< 5 Kg m^{-2} h). At site B, peaking flows increased bed scour. Their flushing action added to this a catastrophic invertebrate drift, and the mineral transport could reach 270 Kg m^{-2} h. Finally, the input of reservoir water resulted in a decrease in water conductivity downstream of the outlet.

Spatial Distribution Patterns

The 22 studied species (Table 1) are characteristic of torrential Pyrenees mountain streams (Berthelemy, 1966; Décamps, 1967; Vinçon and Thomas, 1987; Clergue-Gazeau 1991). In the river Oriège, they showed three different spatial distribution patterns in density (Fig. 4): 1) the mean annual density decreased from site A to C (e.g.,

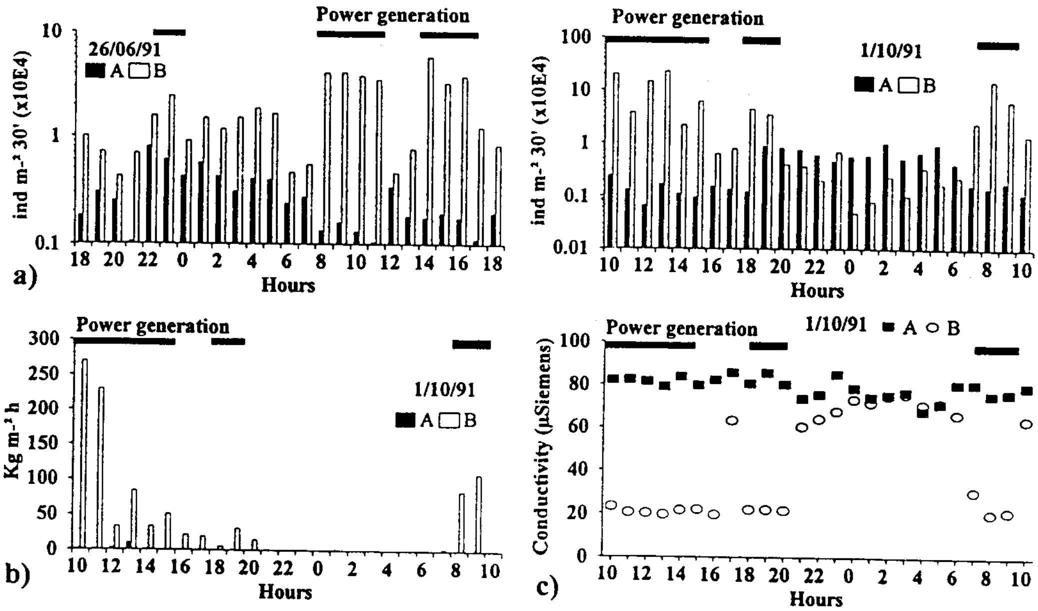


Figure 3: a) Invertebrate drift; b) mineral transport; and c) example of fluctuations in water conductivity due to reservoir releases over 24 hours at sites A and B.

Table 1: Mean annual densities (ind m^{-2}) and biomasses (mg m^{-2}) of studied species at the three sampling sites, with annual variability (min-max). Simuliidae were collected qualitatively.

	A		B		C	
Species	ind m²	mg m²	ind m²	mg m²	ind m²	mg m²
EPHEMEROPTERA						
Baetis alpinus	667 (8-2750)	534 (11.3-2420)	578 (5-2568)	488 (3.5-1823)	429 (15-1196)	549 (5.9-1662)
Baetis rhodani	61 (5-407)	44 (3.4-272.7)	74 (0-482)	54 (0-416.1)	321 (17-2370)	231 (10.3-1896)
Rhithrogena semicolorata	311 (80-682)	500 (48.8-1207)	154 (20-506)	566 (74-2110)	589 (80-1748)	548 (107-556)
Rhithrogena sp. hercynia	88 (0-317)	311 (0-268)	24 (0-86)	166 (0-468)	10 (0-50)	95 (0-281)
Rhithrogena kimminsi	43 (10-150)	74 (14.6-259)	10 (0-30)	20 (0-107.7)	3 (0-27)	1.8 (0-5.4)
Ephemerella ignita	24 (0-69)	11 (0-17.4)	68 (0-735)	36 (0-272)	263 (0-2050)	210 (0-1086)
PLECOPTERA						
Siphonoperla torrentium	46 (9-107)	32 (1.3-73.6)	27 (10-90)	28 (1.8-149.4)	63 (20-115)	21 (4.2-25)
Isoperla acicularis	76 (6-267)	50 (0.6-258)	17 (3-75)	19 (0.2-97)	3.1 (0-74)	2.1 (0-6.6)
Perla grandis	17 (6-30)	24 (12.1-92.9)	4 (0-10)	5.5 (0-41.4)	9 (4-13)	18 (4.1-39.4)
Amphinemura sulcicollis	35 (0-230)	25 (0-246)	-	-	46 (0-182)	33 (0-187.5)
Protonemoura beatensis	114 (0-797)	100 (0-558)	73 (0-625)	66 (0-1019)	3 (0-15)	3.9 (0-8.8)
TRICHOPTERA						
Hydropsyche instabilis	38 (5-189)	98 (0.6-177.5)	13 (0-43)	31 (0-140)	69 (13-246)	898 (47-3415)
Rhyacophila occidentalis	21 (2-76)	107 (0.2-460)	11 (0-25)	35 (0-442)	9 (0-21)	25 (0-61)
Rhyacophila meridionalis	4 (0-6)	13 (0-22.1)	25 (2-56)	134 (2.6-300.1)	20 (0-39)	450 (0-578)
Drusus rectus	37 (0-73)	164 (0-507.5)	7 (0-12)	4.1 (0-19.9)	13 (0-32)	51 (0-29.2)
Anomalopterygella chauviniana	66 (1-243)	24 (0.1-78)	9 (0-30)	5 (0-19.4)	51 (0-138)	68 (0-194.6)
Micrasema longulum	14 (1-52)	20 (0.1-9.3)	2 (0-9)	5 (0-7.5)	2 (0-4)	3.4 (0-4.7)
DIPTERA: SIMULIIDAE						
Prosimulium rufipes	++	++	-	-	++	++
Simulium argenteostriatum	+	+	-	-	+	+
Simulium variegatum	++	++	-	-	++	++
Simulium sp. monticola	+++	+++	+++	+++	+++	+++
Simulium cryophilum	+	+	+	+	++	++

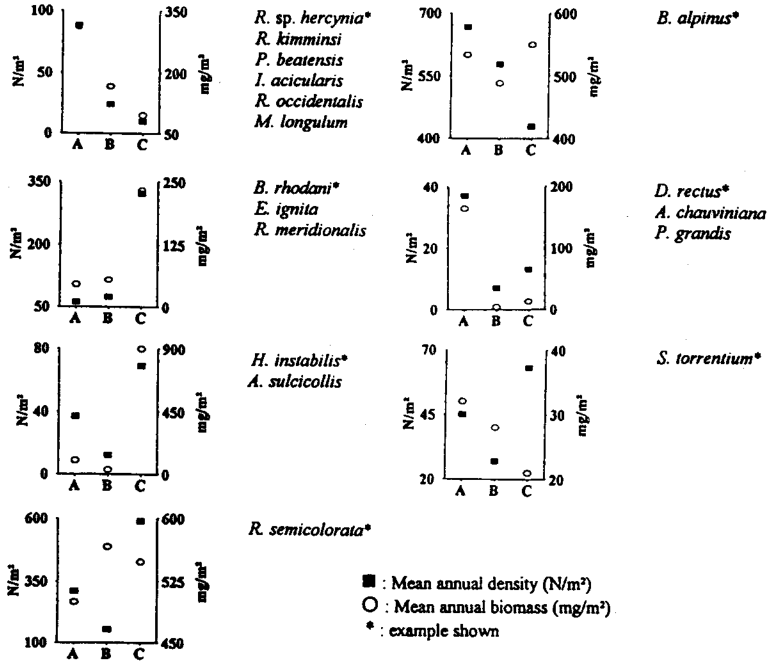


Figure 4: Different spatial distribution patterns in mean annual density and biomass for the various species. In each type, an example is given (species with « * »). See text for further details.

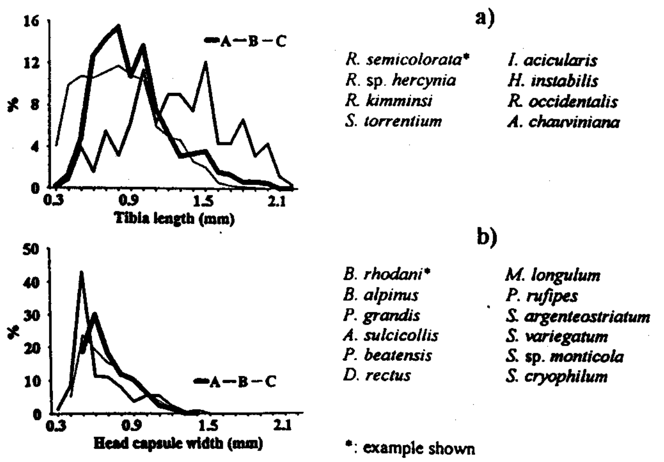


Figure 5: Frequency distribution patterns of size measurements for larvae of the studied species at the various sites, with the examples of *Rhithrogena semicolorata* and *Baetis rhodani*. See text for comments.

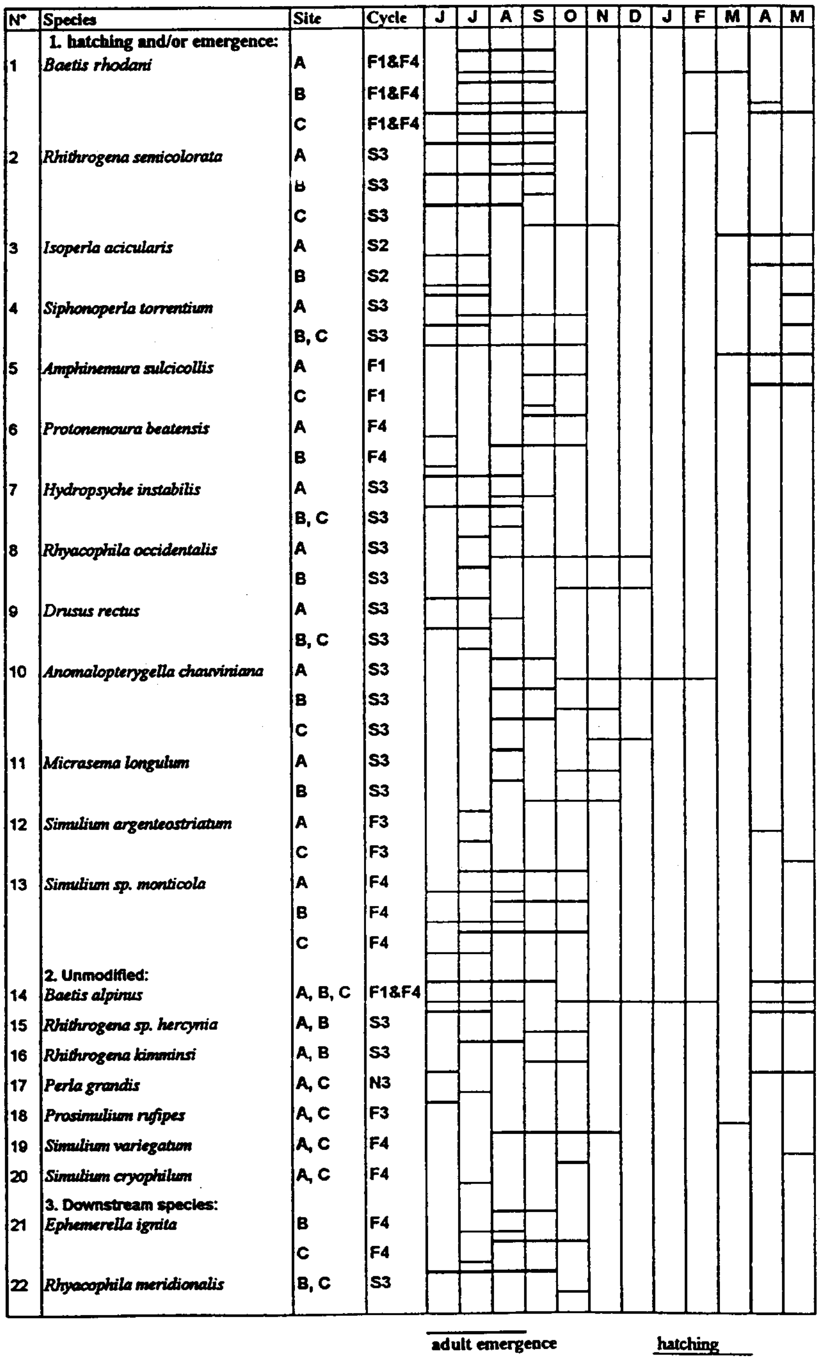


Figure 6: Periods of hatching and emergence of the 22 studied species. Life cycle types (F, S, N) are identified according to Hynes (1970).

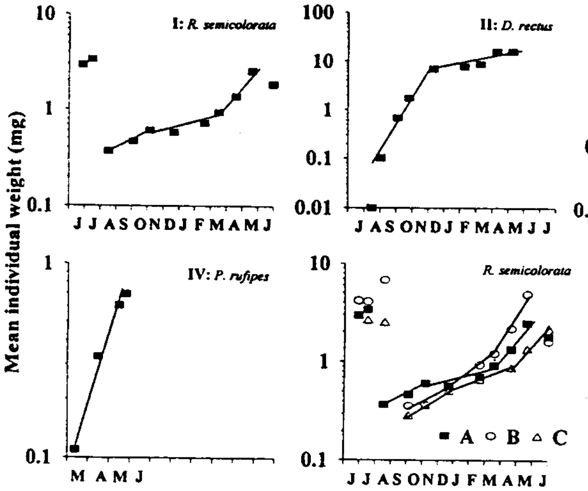
Rhithrogena sp. *hercynia*, *Baetis alpinus*); 2) the mean annual density decreased from site A to B, then increased to C where it was higher (e.g., *Hydropsyche instabilis*, *Siphonoperla torrentium*, *Rhithrogena semicolorata*) or lower (e.g., *Drusus rectus*) than at site A; and 3) the mean annual density increased from site A to C (e.g., *Baetis rhodani*). In most species, the mean annual distribution of sizes classes (Fig. 5) showed no significant differences from one site to another ($p > 0.05$, Kolmogorov-Smirnov two samples tests), and the spatial distribution patterns in biomass were similar to those in numbers. Conversely, the biomass of several species did not change in proportion to their abundance, e.g., the highest biomass of *Rhithrogena semicolorata* and *Baetis alpinus* corresponded to their lowest density; the opposite pattern was observed in *Siphonoperla torrentium*. The non similar changes in density and biomass resulted from significant differences ($p < 0.05$) in mean population structures between the three sites. The numbers of early instars larvae of the three *Rhithrogena* species, *Siphonoperla torrentium*, *Isoperla acicularis*, *Hydropsyche instabilis*, *Rhyacophila occidentalis* and *Anomalopterygella chauviniana* was lower at site B than at site A. The opposite pattern was observed in *Amphinemura sulcicollis* and *Micrasema longulum*. Only *Rhithrogena semicolorata* returned significantly to the reference population structure at site C

Life Histories And Growth

The percentage size distribution of approximately monthly collections of the 52 populations of the 22 species can be classified into 4 main types of the life cycles according to Hynes (1970) (Figs. 6). 1) The non-seasonal life cycle of *Perla grandis* ended in summer; it lasted 3 years and individuals of all stages were present at all times (N3). 2) Seven species exhibited fast seasonal cycles with hatching periods in spring or early summer, a short growth period and adult emergence in summer or autumn (F3, F4). 3) *Simulium* gr. *monticola* was clearly bivoltine. *Baetis alpinus* and *Baetis rhodani* had two overlapping generations which were difficult to separate; the winter generation (F1) being slower than the summer generation (F4). 4) Eleven species showed slow seasonal cycles. The eggs began to hatch soon after laying and growth occurred over a long period before late spring (S2) or early summer emergences (S3, most of the species). The life cycle patterns of each species remained rather similar along the stream despite the slight differences in hatching and/or emergence periods. Growth was observed to be exponential either over the whole period of larval growth (F4) or over 2 or 3 successive periods during which the growth rate was fairly constant (S2, S3). During these periods, there was a linear relationship between the logarithm of the mean individual dry weight of the larvae and time. Four main growth patterns types could be perceived (Fig. 7): I) growth occurred mainly in autumn and spring; it was low in winter (e.g., *Rhithrogena semicolorata*); II) growth was higher in autumn than in winter and spring (e.g., *Drusus rectus*); III) growth was low or moderate but continuous over autumn and winter; it increased during spring (e.g., *Amphinemura sulcicollis*); and IV) the whole larval development occurred over 3-4 months in spring, summer , or autumn (e.g., *Prosimulium rufipes*). In several species with slow-seasonal cycle, the larval growth patterns were modified according to the sites (see Fig. 7). Growth was generally less slowed dowstream than upstream from the hydroplant during the winter period. Thus, the larval growth of *Drusus rectus* showed the type I at site A and the type II at site B. A good example is *Rhithrogena semicolorata* (Fig. 7) whose winter growth was clearly more continuous downstream than upstream from the power plant. A quite similar model of growth can be applied to the development of Baetidae larvae.

DISCUSSION

Changes in flow and temperature are obviously the two main environmental factors modified by hydropeaking. Flow and temperature are also the two main controlling factors for macroinvertebrate populations (Ward and



Growth patterns	Species, sites
I	9A 10C 11A 15AB 22BC
II	2B 3AB 4AC 5AC 9C 10A 11?C 19AC
III	1?ABC 2AC 4B 7AC 8AB 12A 14?ABC 16AB 17AC
IV	6AC 12C 13ABC 18AC 19?AC 20AC 21BC

Figure 7: Different growth patterns (I-IV) of the various species and *Rhithrogena semicolorata* as an example. Species are identified with the numbers indicated in Fig. 6. In each type, an example corresponding to site A is given.

Stanford, 1979) either directly by acting on the energetics of stream insects and/or indirectly by modifying the substrate type and/or the available food resources (Anderson and Cummins, 1979; Sweeney, 1978, 1984; Sweeney *et al.*, 1986; Moog and Janecek, 1991; Giberson and Rosenberg, 1992; Robinson *et al.*, 1992; Pritchard and Zloty, 1994). Peak flows are known to decrease both larval densities and biomasses downstream of hydroplant outlets (Cushman, 1985; Moog, 1993) by increasing the frequency and intensity of bed scour (Newbury, 1984; Cobb *et al.*, 1992) which leads to a catastrophic drift (Fig. 3) (Brooker & Hemsworth, 1978). In the river Oriege, hydropeaking added intermittently to the natural flows so that the disturbance was even higher since the amplitude between baseflow and peaking flow was elevated. Moreover, the impact of hydropeaking upon invertebrate communities added to the faunal changes due to the longitudinal zonation. Indeed, a study based on the whole fauna (unpublished data) showed a faunal discontinuity between the farthest downstream site C and sites A and B. At the three sites, there was an overlap of the epi-metarhithral fauna (species whose numbers decreased naturally from A to C) and of the hyporhithral fauna (species whose numbers decreased naturally from C to A). Nevertheless, the effect of peaking flows on invertebrate density and biomass were obvious in the Oriège. Taking into account the longitudinal zonation of the fauna, it appeared clearly that site B was the most affected for all the studied species, and the disturbances were strongly weakened at site C, i.e., 3000 m downstream of the plant. Not all larval stages responded to environmental changes in the same way. Significant differences in mean annual population structure were found in 8 species with a shift towards larger larvae at site B. Similar patterns were reported in *Hydropsyche* species by Cellot and Bournaud (1986). They concluded that newly-hatched larvae were easily flushed by peak flows. In contrast, Raddum (1985) reported a shift towards smaller individuals in several mayfly and stonefly species in Norway, due to the drift of the largest individuals out of the area. In the Oriège, the dominance of large larvae at site B resulted from: 1) a decrease in abundance of early instar larvae in autumn and winter due to a catastrophic drift when baseflow was the lowest (about 1 m^3 s^{-1}); and 2) an increase in numbers of older larvae in spring which colonized the site by drift when baseflow ranged from 5 to 7 m^3 s^{-1}.

The dynamics of the populations which survived the hydrologic conditions was chiefly controlled by temperature. Hypolimnetic water discharges (4 °C) caused high temperature fluctuations several times a day. As a result, streamwater was cooled in summer and early autumn, reducing the natural temperature gradient between unregulated and regulated sites. Conversely, streamwater below the plant was warmed in winter and the differences in temperature between upstream and downstream sites increased slightly. Because insect metabolisms react immediately to fluctuations in temperature (Sweeney, 1978), and although mean temperature over a given period remains an approximative parameter when frequent fluctuations occur (Sweeney, op. cit.), mean temperature is currently used when explaining the growth patterns of aquatic insects. Despite the reservoir releases, the timing of development of the species was not fundamentally modified below the power plant. At the three sites, the growth rates of most species were positively correlated with mean water temperatures, whatever the diel temperature and flow regimes. Moreover, the growth of some species was low but continuous in winter below the hydroplant, whereas it stopped at site A. As these species occur preferentially usptream, such thermal modifications could be advantageous to stenothermic species, particularly when they reach their lower altitudinal limit. Considering the theoritical relationships between activity rate and temperature (Pradan, 1945 in Sweeney, 1984), an increase in low winter temperatures would stimulate their growth - sthenothermic species having a low threshold temperature - and a decrease in high summer temperatures would have a minor incidence on their growth rate. Further experimental studies would be necessary to test this hypothesis.

In the River Oriège, the repeated return to baseflow reduced the downstream effects of the hydroelectric facility. Moreover, snow melt floods in spring minimize the impact of peaking flows as the growth of most species occurs during this period at this elevation. Snow melt floods also allow the losses by catastrophic drift to be compensated by larvae drifting from the unregulated zone when the power plant is inoperative.

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TOWARDS A METHOD TO RECONSTRUCT A NATURAL HYDROGRAPH FOR DEFINING ECOLOGICALLY ACCEPTABLE DISCHARGE FLUCTUATIONS

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ABSTRACT

Management of the hydro-electric power plant at Lixhe in Belgium leads to short-term discharge fluctuations in the Grensmaas, a non-navigable section of the Meuse River with a relatively high ecological value. These discharge fluctuations negatively affect the ecology of the Grensmaas. Recent in situ studies on macroinvertebrates have shown that the ecological effects of discharge fluctuations cannot be demonstrated in the Grensmaas. In the present study, the ecologically acceptable discharge fluctuations are based on the reconstruction of a natural hydrograph for the Grensmaas, from which the discharge fluctuations are derived from the relevant hydrological parameters. These fluctuations in the natural hydrograph are acceptable for the characteristic flora and fauna of the Grensmaas. A natural hydrograph was derived from the historical hydrograph, before the construction of the Lixhe hydro-electric power plant, a comparable hydrograph of another river in a more natural situation, and a reconstructed hydrograph by filtering out the artificial discharge fluctuations from the present hydrograph using the moving average and spectral analyses. A natural hydrograph based on a historical hydrograph is the most accurate, but cannot be used for most rivers because hydrographic measurements are not available for the natural situation. Use of a comparable hydrograph has the disadvantage that a natural hydrograph of two rivers is never identical. Therefore, in this study a natural hydrograph was derived from a reconstructed hydrograph using the moving average and spectral analyses. For 95 % of the period, the amplitude must not be above 15 m^3s^{-1} and the rate of increase not above 20 %. It is assumed that when these values are exceeded in 5% of the time the flora and fauna can recover.

KEY-WORDS: Maas/ Grensmaas/ ecological rehabilitation/ macro invertebrates/ discharge fluctuations/ natural hydrograph/ spectral analyses/ water management / ecological engineering/ control management

INTRODUCTION

The ecology of many of the major rivers in Western Europe has declined in the last hundred years or so as a consequence of river regulation measures and pollution. This is also the case with the Meuse River. As the river flows through France, Belgium and the Netherlands, an international approach is required to rehabilitate the river ecology. Any rehabilitation measures must take into account economic interests.

In addition to factors such as river regulation and pollution, the ecology of the Meuse river is further deteriorating because of discharge fluctuations caused by the operation of hydro-electric power plants. The hydro-electric power plant in Lixhe, Belgium causes short-term discharge fluctuations in the Meuse River in the Netherlands, which are well in excess of the natural situation. As well as obstructing shipping, these discharge fluctuations negatively affect the ecology of the Grensmaas, a non-navigable section of the Meuse with a relatively high ecological value. Thus their reduction is an essential step in the river's ecological rehabilitation.

The first step in this process is to determine the ecologically acceptable discharge fluctuations. Recent in situ studies on macroinvertebrates have shown that the ecological effects of discharge fluctuations could not be demonstrated in the Grensmaas. The present study therefore aimed to establish ecologically acceptable discharge fluctuations by reconstructing the natural hydrograph of the Meuse River. From this, the relevant hydrological parameters were determined to describe the discharge fluctuations of the natural hydrograph. The discharge fluctuations in a natural hydrograph are expected to have no effect on the characteristic flora and fauna of the Grensmaas.

STUDY AREA

The Meuse catchment area covers approximately 33,000 km^2 extending through France, Belgium and the Netherlands (Figure 1). The present situation in the Meuse River is largely determined by river regulation measures. To make the river navigable at low discharges, locks, weirs and navigable lateral canals such as the Julianakanaal, have been constructed. A detailed description of the Meuse catchment area is given in Descy and Empain (1984) and van Urk (1984).

The Borgharen weir distributes the Meuse discharge to the Grensmaas and the Julianakanaal. The objective is to ensure sufficient water to maintain the navigability of the Julianakanaal, a minimum discharge for the Grensmaas flora and fauna, and to reduce the discharge fluctuations caused by the Lixhe hydro-electric power plant. Despite considerable efforts, discharge fluctuations in the Grensmaas have only been reduced partly because priority has been given to maintaining the water level for navigation.

Since 1980, the Lixhe hydro-electric power plant has generated electricity with four fixed turbines each with a capacity of about 80 m^{3}s^{-1}. River discharge is the determining factor in the operation of the turbines. At high discharges, the turbines can operate continually while at discharges of less than 320 m^{3}s^{-1} the turbines are frequently switched on and off. This results in a block-shaped hydrograph of the Meuse at Eijsden directly downstream of the Lixhe electric power plant (Figure 2). Weir management at Borgharen reduces the discharge fluctuations in the Grensmaas somewhat. In the Grensmaas itself, discharge fluctuations are further reduced in a downstream direction so that at Maaseik considerable fluctuations cannot be discerned on the hydrograph (Figure 2).

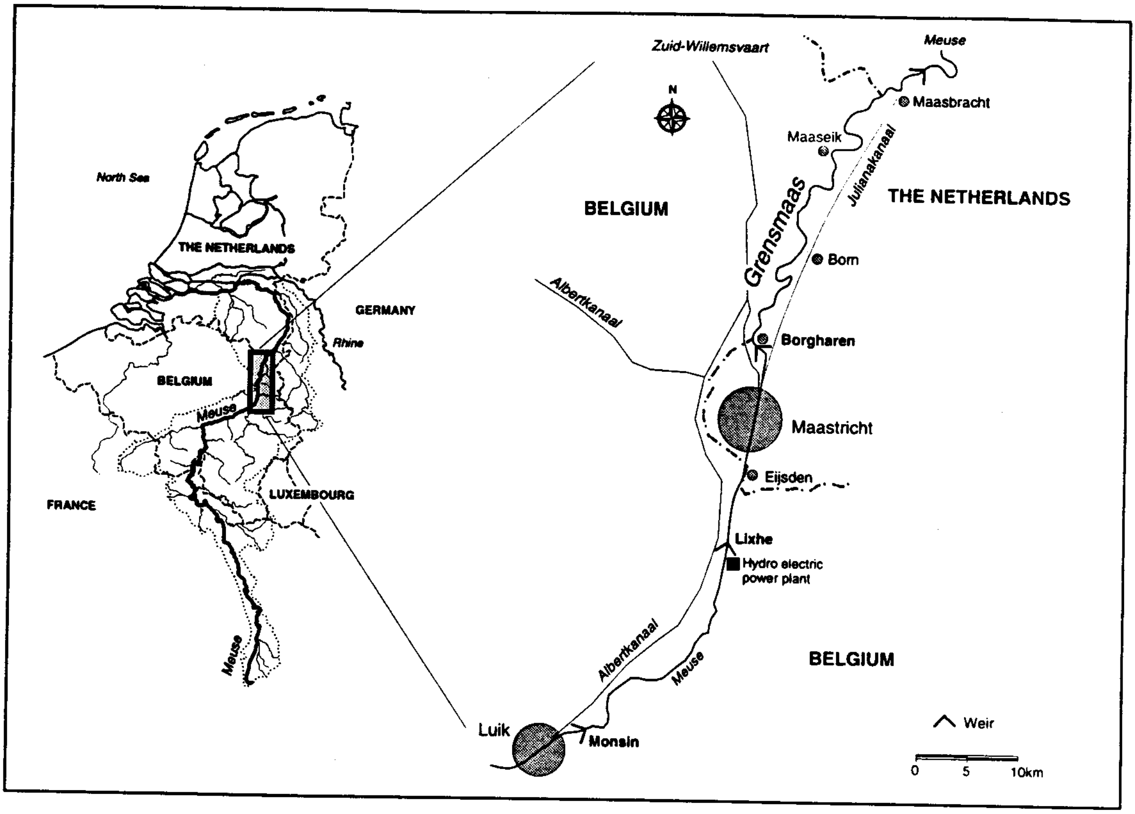

Figure 1 : Map showing the location of the Meuse catchment area and a detail of the study area

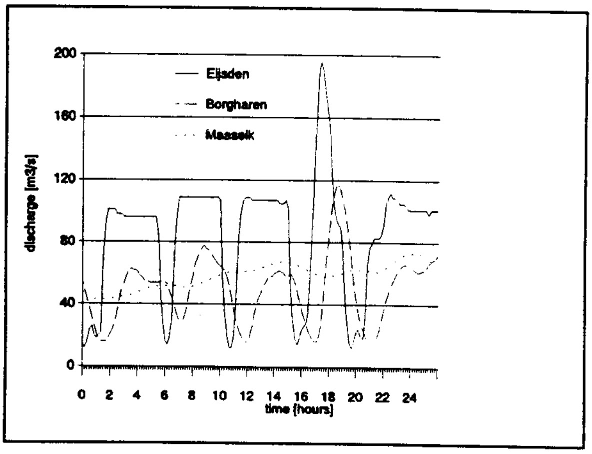

Figure 2 : Representative hydrograph for Eijsden, Borgharen and Maaseik on 10 June 1993

MATERIALS AND METHOD

(a) Reconstruction Of The Natural Hydrograph

In this study, a natural hydrograph has been derived from: **(1) an historical hydrograph** of the Grensmaas before the construction of the Lixhe hydro-electric power plant; **(2) a comparable hydrograph** of another river in a more natural state; and **(3) a reconstructed hydrograph** by filtering out the artificial discharge fluctuations from the present hydrograph. The present hydrograph is determined by the natural and artificial discharge fluctuations. The natural discharge fluctuations include the seasonal variation and the precipitation differences. The artificial discharge fluctuations are related to the Lixhe hydro-electric power plant. These discharge fluctuations vary in frequency (number of times occurring) and amplitude (size). The natural discharge fluctuations are long-term (low frequencies) because the rainfall discharge process is relatively slow. The artificial discharge fluctuations are short-term (high frequencies) as a result of the frequent switching on and off of the turbines. Because of this difference in frequency, the artificial discharge fluctuations can be filtered out and a natural hydrograph generated. The following methods were used for this purpose: **(3-a) a reconstructed hydrograph using the moving average**, and **(3-b) a reconstructed hydrograph using spectral analyses**. In addition, **(4) the present hydrograph**, after construction of the Lixhe hydro-electric power plant, was used as a reference framework.

(1) A Historical Hydrograph

A natural hydrograph can be derived directly from the historical hydrograph for the situation before the Lixhe hydro-electric power plant was constructed in 1980. The historical hydrograph was derived directly from the hourly discharge measurements at Borgharen in 1975, 1977 and 1979.

(2) A Comparable Hydrograph

The natural hydrograph can be derived indirectly from the hydrograph of another river in a more natural situation. The Mosel River (a tributary of the Rhine) was used because its catchment area is similar to that of the Meuse with regard to morphology, hydrology and climate. As Mosel discharge is not as regulated as that of the Meuse, it is more of a natural situation. A comparable hydrograph was derived directly from the 15-minute discharge measurements at Trier for the period 1987 to 1995.

(3-a) A Reconstructed Hydrograph Using The Moving Average

Using the moving average, the present short-term artificial discharge fluctuations only were smoothed out, not the long-term fluctuations. To develop the reconstructed hydrograph using the moving average, the 10-minute discharge measurements at Borgharen were used for the period 1987 to 1995. The moving average was calculated over a 18-hour period.

(3-b) A Reconstructed Hydrograph Using Spectral Analyses

In a spectral analyses, the hydrograph is approached as a Fourier series. The hydrograph is seen as discharge fluctuations, which are described as "sinus-components" with different frequencies and amplitudes (Box and Jenkins, 1980). This is represented as a spectrum of discharge fluctuations with the variance density on the y axis and frequency on the x axis. The variance density is a measure of the occurrence of the discharge fluctuations. If a certain frequency of discharge fluctuations is clearly discernible, and is thus the major determining factor of the hydrograph, then there is a high variance density in the spectrum (a peak). Natural discharge fluctuations are discernible in the spectrum as peaks at low frequencies, and artificial discharge fluctuations as peaks at high frequencies (Figure 3). In a natural hydrograph, peaks are expected at low frequencies. For the present hydrograph, peaks are also expected at high frequencies. Possible peaks at high frequencies cannot be explained by natural processes and are therefore attributed to the Lixhe hydro-electric power plant.

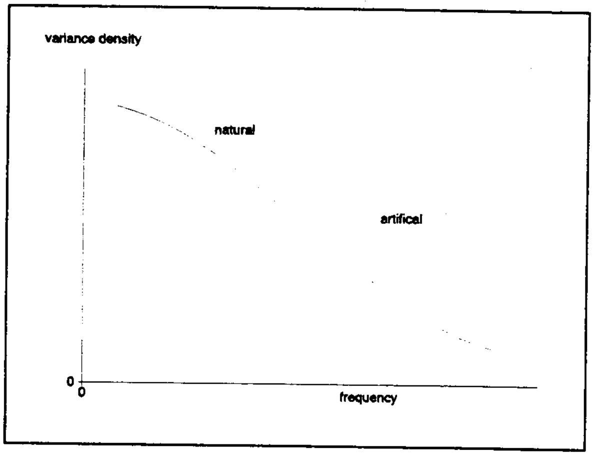

Figure 3 : Sketch of a spectrum with natural discharge fluctuations and artificial discharge fluctuations

A corrected spectrum can be established by suppressing the discharge fluctuations caused by the Lixhe hydro-electric power plant. A new reconstructed hydrograph can be generated without the effect of the Lixhe hydro-electric power plant. To use this method, there needs to be a clear peak in the spectrum at high frequencies because otherwise this artificial discharge fluctuation cannot be filtered out. In the spectrum of the present hydrograph at Borgharen, the artificial discharge fluctuations are not discernible as clear peaks but as a gradual increase of a larger band of frequencies (see Results, Figure 6). A definite peak is visible at Eijsden but the discharge measurements at this location are not reliable enough to reconstruct a natural hydrograph. Therefore, an alternative method was used to determine the corrected spectrum. The Mosel spectrum was scaled up to the total Grensmaas variance density. A natural hydrograph was generated for the Grensmaas with the characteristics of a Mosel comparable hydrograph. The reconstructed hydrograph using spectral analyses was determined from the 10-minute discharge measurements at Borgharen and the 15-minute discharge measurements at Trier (Germany) for the period 1987 to 1995.

(4) The Present Hydrograph

The present hydrograph was derived directly from the 10-minute discharge measurements at Borgharen for the period 1987 to 1995.

The method used is presented schematically in Figure 4.

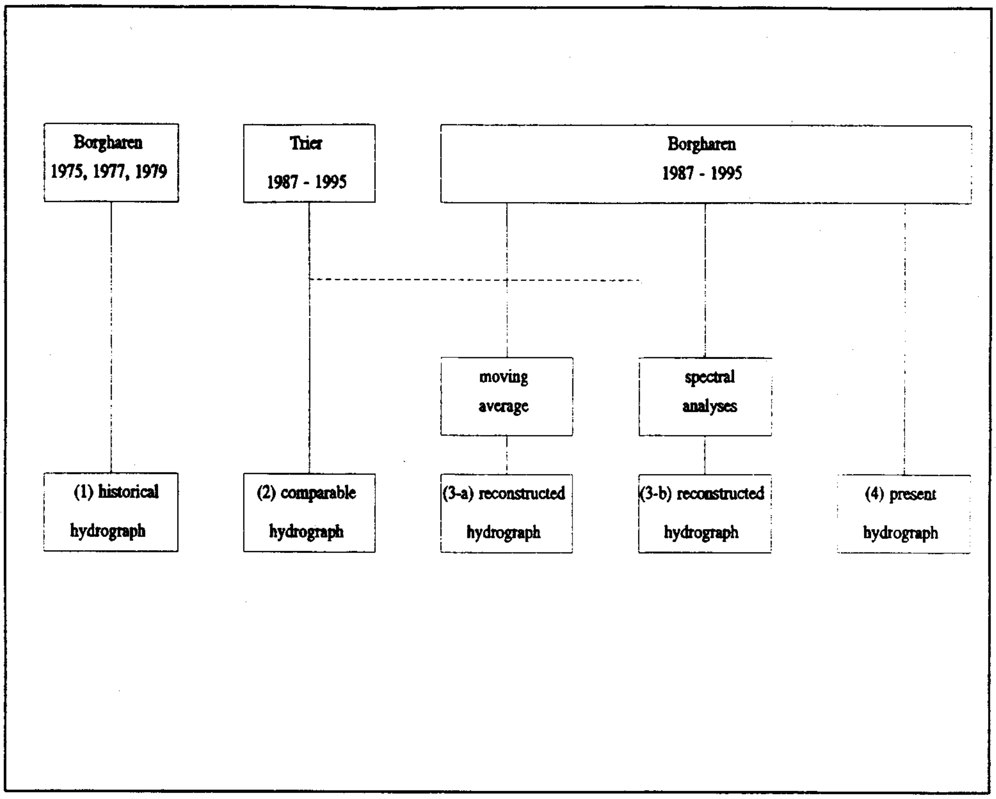

Figure 4 : The method employed

<u>(b) Determination Of The Hydrological Parameters</u>

After reconstructing a natural hydrograph using the different methods, the relevant hydrological parameters were determined. The hydrological parameters which characterize the discharge fluctuations in a natural hydrograph were identified as (Figure 5): **(1) the frequency**, expressed as the number of discharge fluctuations in the hydrograph per day, which is consistent with the switching on and off of a turbine in the Lixhe hydro-electric power plant; **(2) the amplitude**, expressed as the difference between the smallest and largest discharge within a day (in mathematics, amplitude is defined as the half of this difference); and **(3) the rate of increase**, expressed as the largest increase in discharge within a day. This rate of increase is expressed as the increase in discharge within an hour in relation to the average discharge in the previous hour. This was calculated for each period and the largest value in a day selected.

In determining the hydrological parameters only the summer periods from 1 June to 30 September were considered. In these periods, there are mostly small discharges with a high number of discharge fluctuations and a high rate of increase, so that this is the most representative period.

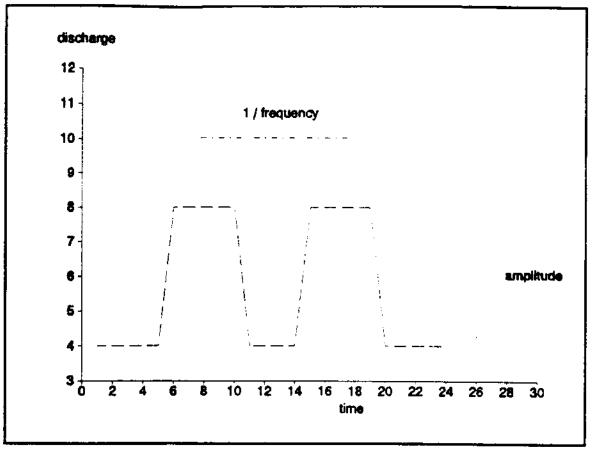

Figure 5 : Definition of the hydrological parameters of discharge fluctuations

RESULTS

(a) Reconstruction Of A Natural Hydrograph

The results are the natural hydrographs as determined by the different methods. These are not presented as figures because the hydrographs are only used for further processing. In this section, only the spectrum as used for a reconstructed hydrograph with spectral analyses is discussed because this is an essential part of the further processing.

A Reconstructed Hydrograph Using Spectral Analyses

The spectra for the present hydrographs at Eijsden, Borgharen and a comparable hydrograph at Trier are presented in Figure 6. In the spectrum for Eijsden, the discharge fluctuations caused by the Lixhe hydro-electric power plant are clearly discernible as an increase in the variance density for frequencies between 3.10^{-5} and 3.10^{-4} Hz (per second), equivalent to 2 to 26 times a day. At Borgharen, this increase is not as easily detected because the Borgharen weir management partly reduces the discharge fluctuations. The artificial discharge fluctuations are visible, however, as a higher variance density than in a comparable hydrograph for Trier. Interference in the discharge measurements is clearly visible in the spectrum as a freakish graph. The Mosel spectrum was scaled up to the total variance density of the Grensmaas in order to generate the spectrum for a natural hydrograph of the Grensmaas.

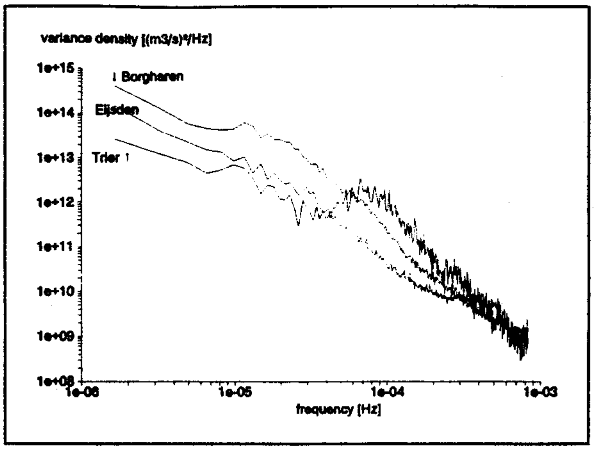

Figure 6 : Spectrum of the variance density for Eijsden, Borgharen and Trier

(b) Determination Of The Hydrological Parameters

The frequency of the discharge fluctuations in the present hydrograph is presented in Figure 7. Only the frequency for the present hydrograph is given because in the natural hydrographs, the frequency is less than once a day.

The amplitude and the rate of increase were calculated per day (see materials and methods). The results are presented as cumulative values, expressed as a percentage of days on which the value is not reached (Figure 8 and 9). The average values of the frequency, amplitude and rate of increase are presented in Table 1.

Table 1 : Average frequency, amplitude and rate of increase of the discharge fluctuations

hydrograph	frequency per day	amplitude (m^3s^{-1})	rate of increase (%)
(1) historical hydrograph	< 1	43	40
(2) comparable hydrograph	< 1	31	9
(3-a) reconstructed hydrograph (moving average)	< 1	29	9
(3-b) reconstructed hydrograph (spectral analyses)	< 1	9	16
(4) present hydrograph	8.5	84	86

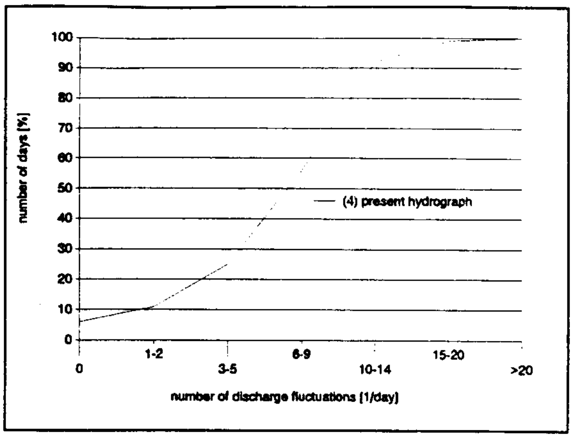

Figure 7 : Cumulative distribution of the frequency of the discharge fluctuations

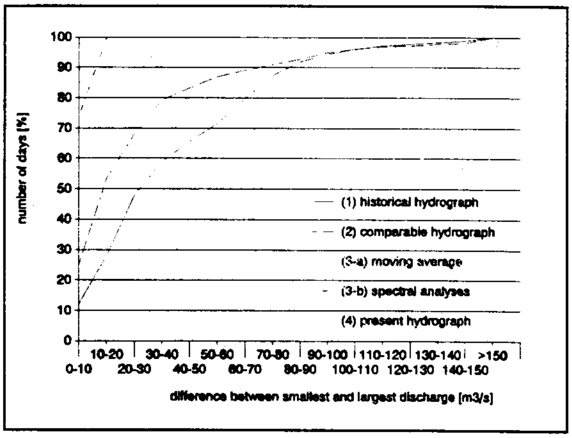

Figure 8 : Cumulative distribution of the amplitude of the discharge fluctuations

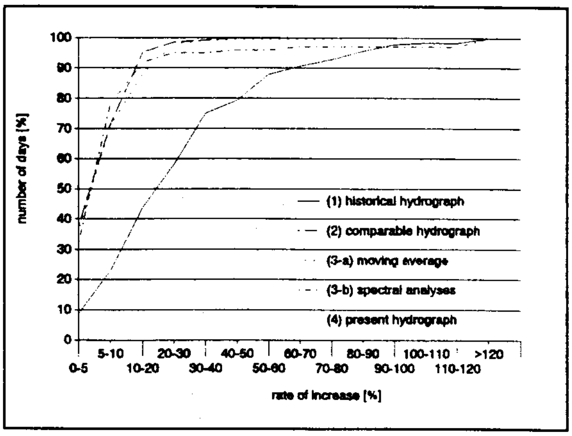

Figure 9 : Cumulative distribution of the rate of increase of the discharge fluctuations

CONCLUSIONS AND DISCUSSION

Comparison Of Methods

The most direct method to describe a natural hydrograph is by means of an historical hydrograph but this has two limitations. Firstly, an historical hydrograph is not identical to a natural hydrograph because of discharge fluctuations caused by locks and weirs, navigation canals and increased paved areas, for more than a century are detectable in the Meuse hydrograph. Secondly, for this period, only hour measurements are available so that the hydrological parameters of discharge fluctuations cannot be determined within an hour. This is confirmed by the relatively large amplitude and rate of increase, from which it appears there are artificial fluctuations in the situation before the construction of the Lixhe hydro-electric power plant. While this method is the most suitable, it cannot be used for most rivers because discharge measurements are not available for the natural situation.

The use of a comparable hydrograph has the disadvantage that a natural hydrograph of two rivers is never identical because there are always differences in the morphology, hydrology and the climate between catchment areas. The comparison with the Mosel hydrograph also has limitations because the average discharge of the Mosel in the summer is significantly higher than that of the Grensmaas.

With the use of the reconstructed hydrograph using the moving average, the artificial discharge fluctuations are smoothed instead of filtered out. The result depends on the period averaged. If the period is too short, then the artificial discharge fluctuations are not completely smoothed out. If the period is too long then the natural discharge fluctuations are smoothed out. For the 18-hour period of the moving average, the average amplitude is 29 m^3s^{-1}, which indicates that the artificial discharge fluctuations are smoothed out. The distinguishing strength of this method is, however, limited because it is difficult to indicate the boundary between smoothing artificial and natural discharge fluctuations. Nevertheless, this method produces satisfactory result because there is a relatively large difference in frequency between artificial and natural discharge fluctuations.

With the reconstructed hydrograph using spectral analyses, a natural hydrograph of the Grensmaas was generated with the characteristics of the Mosel. The mean amplitude was 9 m^3s^{-1} which is significantly lower than that produced with the method using the moving average. This method can be used to reconstruct a natural hydrograph despite the effect of river regulation, locks and weirs, navigation canals and the increasing paved area. This method can only be used when a comparable hydrograph of another river in a more natural state is available, but this is not often the case.

There are two possible reasons for the difference in amplitude between the method using the moving average, 29 m^3s^{-1}, and spectral analyses, 9 m^3s^{-1}. Firstly, the moving average does not smooth out the artificial discharge fluctuations enough, or secondly, a deviation occurs in the conversion to the Mosel hydrograph in the spectral analyses. The amplitude with spectral analyses is lower than the amplitude in a comparable hydrograph in the Mosel itself because of scale effects (the Mosel discharge is significantly higher than that of the Grensmaas discharge). Both the moving average and the spectral analyses can be used to develop a reconstructed hydrograph. The present results do not indicate which method is preferable. The moving average method cannot make a clear distinction between natural and artificial fluctuations. For use of the spectral analyses to determine the natural hydrograph for the Grensmaas, a comparable river is needed and therefore spectral analyses is of limited used.

Comparison Of Discharge Fluctuations In The Present Hydrograph And A Natural Hydrograph

The frequency of discharge fluctuations caused by the Lixhe hydro-electric power plant of 8.5 times a day is much higher than the frequency in the natural hydrograph, which is less that once a day. The amplitude of the present hydrograph is often about 80 and sometimes 160 m^3s^{-1}, equivalent to the capacity of the turbines (the consequence of being switched on and off). The average amplitude of 84 m^3s^{-1} and rate of increase of 86 % in the present hydrograph are much higher than the 9 m^3s^{-1} and 16 % in the natural hydrograph (Table 2).

Table 2 : Discharge fluctuations in the present hydrograph and a natural hydrograph

hydrograph	frequency per day	amplitude (m^3s^{-1})	rate of increase (%)
natural hydrograph	< 1	9 - 29	9 - 16
present hydrograph	8.5	84	86

Ecologically Acceptable Discharge Fluctuations

The ecologically acceptable discharge fluctuations are derived from the natural hydrograph of the reconstructed hydrograph using the moving average and spectral analyses. The values of the hydrological parameters in a natural hydrograph vary with time so that the ecologically acceptable discharge fluctuations cannot be expressed as single values for the amplitude and for the rate of increase. It is proposed to relate the ecologically acceptable discharge fluctuations to a value which, for example, is not exceeded in 95 % of the days (Table 3). Thus the amplitude may not be above 15 m^3s^{-1} and the rate of increase not above 20 % for 95% of the selected period. It is assumed that when these valves are exceeded in 5% of the time the flora and fauna will recover. This 95 % value can be further substantiated by ecological studies and from practical experience.

Table 3 : The ecological acceptable discharge fluctuations with a 95 %

frequency per day	amplitude (m^3s^{-1})	rate of increase (%)
< 1	15	20

Recommendations

On the basis of the results of this study, measures can be designed to reduce the discharge fluctuations to an ecologically acceptable level. The effects of the measures can be calculated with a hydro-dynamic model. The ecologically acceptable discharge fluctuations are described in terms of hydrological parameters. The effect on the ecology can be assessed in the field because the density and species composition of macroinvertebrates and fish have been monitored once every one to four years since 1992.

ACKNOWLEDGEMENTS

The authors are grateful for the suggestions about the derivation of the natural hydrograph from spectral-analyses made by Sjors van der Kamer and Floris Schulze from the Institute for Inland Water Management and Waste Water Treatment, and further processing by Tjerk Zitman from Witteveen+Bos Consulting engineers. We are grateful to the many people who have assisted in implementing this study and in preparing this paper.

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Impact mitigation measures

Mesures d'atténuation d'impacts

CONTROLLING ALGAL BLOOMING IN WITHDRAWAL ZONE USING VERTICAL CURTAINS: AN APPLICATION TO TERAUCHI DAM RESERVOIR, JAPAN

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ABSTRACT

Terauchi Dam Reservoir, one of largest reservoirs in Kyushu, Japan has been supplying water for paddy field irrigation and domestic consumption. During recent years it has shown serious deterioration of water quality specially in summer season. Destruction of thermocline has been prohibited to maintain desirable temperature downstream of the reservoir for paddy field irrigation, thus, most of the existing control devices cannot be applied to the reservoir. As one of the ways of reducing algal blooming in the reservoir, two vertical curtains, having depths to cover the epilimnion thickness, are installed across the reservoir in order to curtail the nutrient supply from nutrient-rich inflow to the downstream epilimnion of the reservoir. Withdrawal level is also regulated to keep the downstream epilimnion off the nutrient supply.

The experimental results of this study identified some of the possible mechanisms and reasons for the reduction of algal blooming at the downstream zone of the reservoir. During early spring up to the middle of April, inflow was lighter than the water at the surface of the reservoir and floats over the reservoir surface. Thus, the presence of curtains prevents the direct intrusion of the high level of nutrients to the downstream zones. Higher concentration of algae at the upstream zones consume large amount of inflow nutrients, which cause a reduced nutrient supply to the downstream zone of the reservoir. During late spring and summer, presence of curtain prevents the dispersion of nutrient from upstream zones to the downstream zones. It is also found that the inflow penetration level reduced close to the withdrawal level which played a major role by withdrawing nutrient rich interflow. Curtain method is most suitable for reservoirs or lakes having elongated morphology which reduces length of the curtain.

The process in the reservoir ecosystem have been modeled, by assuming stratified layered structure at each zone of the reservoir, to predict the water quality and algal species composition in the reservoir. The state variables, temperature, four types of phytoplankton as chlorophyll-*a*, soluble phosphorus, nitrate, ammonium, dissolved oxygen, biochemical oxygen demand, internal nitrogen, internal phosphorus, and when diatoms are modeled explicitly silica, are considered in the model. The model is calibrated with the measurements taken in the field, with good agreement.

KEY-WORDS: Algae / Curtain / Entrainment / Epilimnion / Eutrophication / Nutrient / Phytoplankton / Reservoir / Riverine / Stratified.

* Corresponding author

INTRODUCTION

Terauchi Dam Reservoir, one of the reservoirs in Kyushu, Japan has shown a serious deterioration of water quality, caused by cultural eutrophication, specially during summer. Eutrophication process accompanied by excessive phytoplankton blooms are responsible for the characteristic greenish color, reduced transparency, and hypolimnetic oxygen depletion of the reservoir water (Fig. 1), and thus certainly impairs the recreational use of the reservoir. Hence, as a management implication, an environmentally sound, technically viable, and easily constructable control device has to be used to curtail the eutrophication process.

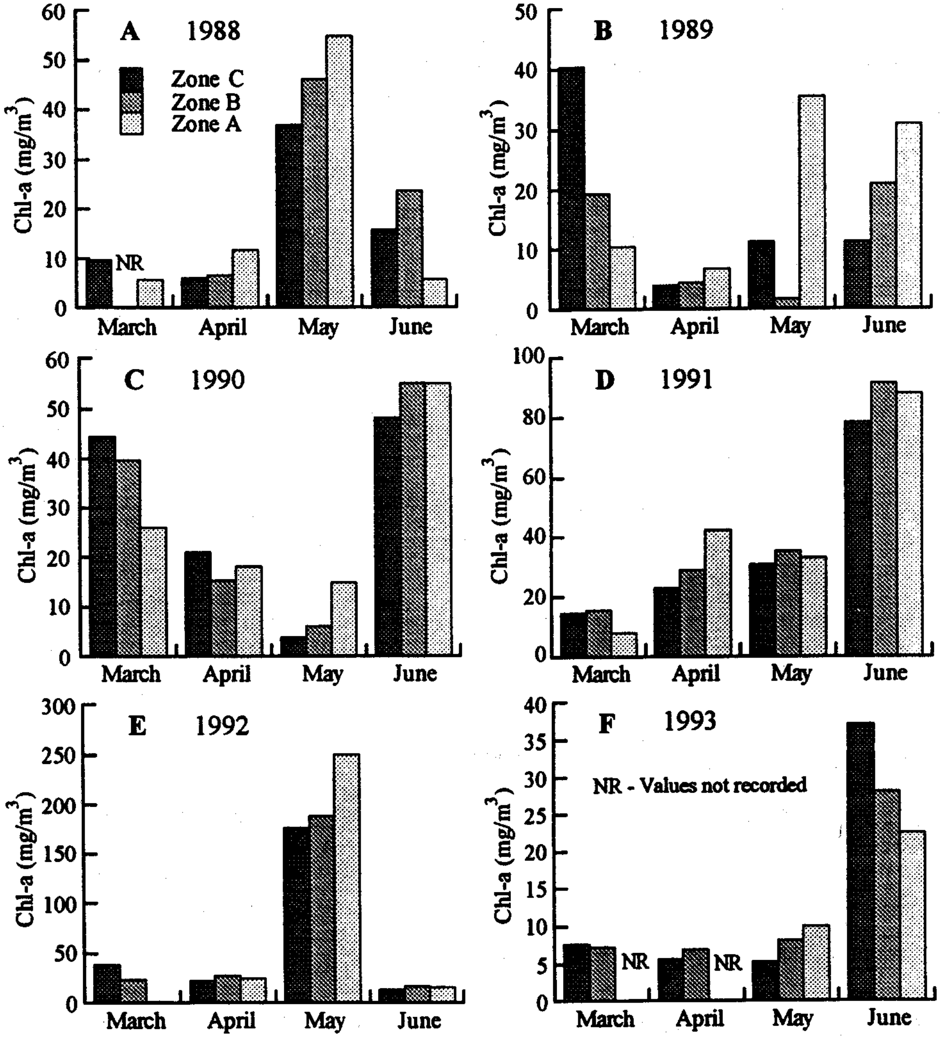

Figure 1: Past records of chlorophyll-a concentrations in the surface layer of Terauchi Dam Reservoir before curtains were installed

One of the important roles of Terauchi Dam Reservoir is to supply water to paddy fields in the region. Paddy field irrigation requires water warmer than 17 ^{0}C. Thus, there are various drawbacks for conventional existing control devices to be applied due to the destruction of the thermocline partially or completely. The reservoir outlet also works incorporating selective withdrawal device. Although various water quality parameters are considered, the most important objective is to maintain a desirable river temperature downstream of the reservoir (Asaeda et al. 1996).

INFORMATION ON STUDY SITE

Terauchi Dam reservoir is located in the northern part of Kyushu, the west island of Japanese archipelago, 33^0 25' N in latitude, and 130^0 43' E in longitude. The reservoir is 2.5 km long, 400 m wide, 900 ha of total water surface area, and 35 m of maximum depth. Total volume of the reservoir is 18 million m^3. Three rivers enter the reservoir; the main river called Sata River, enters the upstream end of the reservoir, whereas Teishakuji River and Small River enter the middle of the reservoir (Fig. 2). The Small River has the highest concentration of nutrients but its discharge is small compared with other two rivers. Phosphate phosphorus concentration reported in the three rivers during the experimental period are given in Table 1. The morphology of the reservoir is such that it has an elongated shape as shown in Fig. 2.

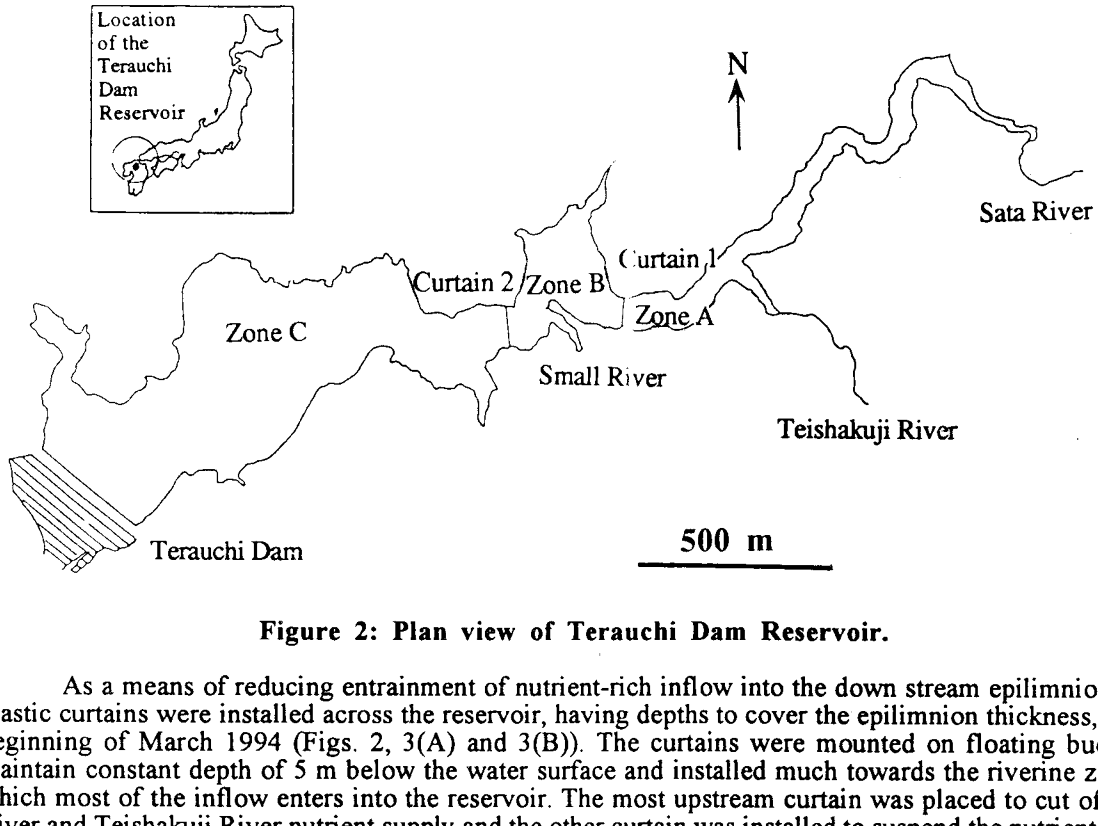

Figure 2: Plan view of Terauchi Dam Reservoir.

As a means of reducing entrainment of nutrient-rich inflow into the down stream epilimnion, two plastic curtains were installed across the reservoir, having depths to cover the epilimnion thickness, at the beginning of March 1994 (Figs. 2, 3(A) and 3(B)). The curtains were mounted on floating buoys to maintain constant depth of 5 m below the water surface and installed much towards the riverine zone in which most of the inflow enters into the reservoir. The most upstream curtain was placed to cut off Sata River and Teishakuji River nutrient supply and the other curtain was installed to suspend the nutrients from Small River.

The three parts of the reservoir separated by two curtains were named as Zone A, Zone B, and Zone C (Fig. 2). Water samples were taken every week from four depths; 0.5 m, 2.0 m, 4.0 m, and 6.0 m from each zone for chemical and biological analysis such as chlorophyll-*a*, total phosphorus, soluble phosphorus, suspended phosphorus, nitrate nitrogen, ammonium, suspended sediment, DO, COD, BOD, conductivity, pH, and turbidity. Temperature was recorded every 1 m interval of the depth and transparency (Secchi depth) was measured at the three zones. Meteorological data such as solar radiation, temperature, humidity, rainfall, and wind velocity were measured continuously at the dam site.

Table 1: Phosphate phosphorus concentration reported in 1994

River	Date March 17	April 27	May 27	June 24
	mg/m^3			
Teishakuji River	89	68	160	111
Sata River	14	19	24	26
Small River	81	176	405	84

The experiments were conducted over three months from the beginning of March to the end of June, 1994. The withdrawal level of Terauchi Dam reservoir was maintained at 1.5 m below the water level from March 3 to March 22, then was lowered to 12 m on March 23, 15 m from March 25, then was raised to 10 m on April 25, and from June 13 it was changed daily between 6-9 m to supply warmer water for irrigation.

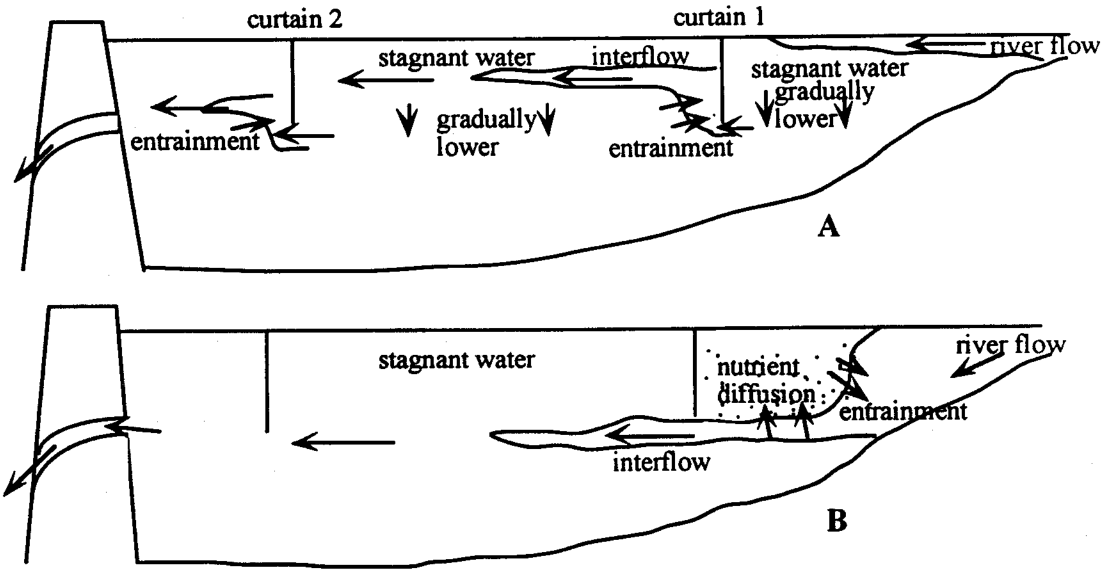

Figure 3: Illustration of possible mechanisms after installing curtains (A) during early spring (B) during late spring and summer.

Predominant species of phytoplankton in the reservoir was *Melosira* in March, which was replaced by *Synedra* at the beginning of April. In the late April, *Phormidium* and *Cyclotella kuetzingiana* propagated instead of *Synedra*. *Fragilaria* dominated in May. However, in this paper total biomass is expressed in terms of chlorophyll-*a* regardless of the variety of species to look into the aspects pertaining to curtain method on algal blooming. The inflow rate to the reservoir and outflow rate from the reservoir during the experimental period are shown in Fig. 4 . Average air temperature and precipitation at the reservoir are given in Fig. 5 (Saitoh and Gotoh, 1994).

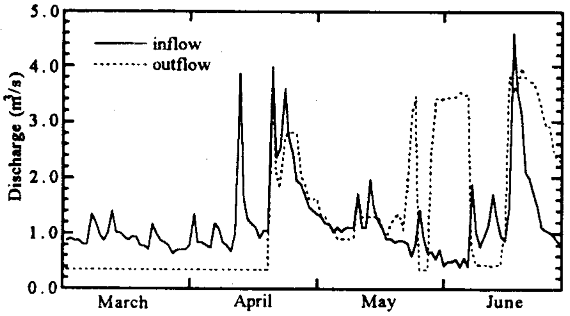

Figure 4: Inflow and outflow hydrograph of the reservoir in 1994.

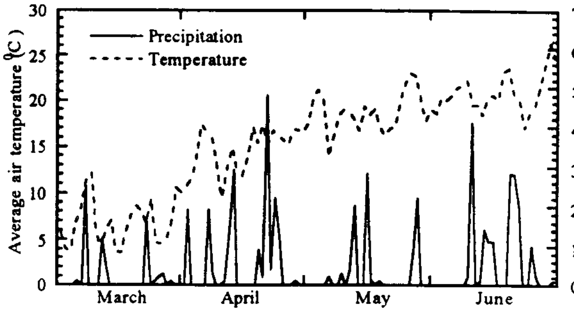

Figure 5: Average air temperature and precipitation at the reservoir in 1994.

EXPERIMENTAL RESULTS

Until the beginning of April the difference of chlorophyll-*a* concentration between the three zones were not substantial (Figs. 6(A-C)). A remarkable change occurred in the middle of April, after which chlorophyll-*a* concentration reduced significantly in Zone C (Figs. 6(C-H)). In spring, chlorophyll-*a* at the surface layer of Zone A increased, which may be due to the high nutrient supply from inflow water to the surface layer during this period. The possible mechanisms and reasons for the reduction of algal blooming at the downstream zone of the reservoir are yet to be identified. It is evident from chlorophyll-*a* counts that the impacts of the curtains on each and every process in the reservoir ecosystem have a great effect on algal blooming. During early spring up to the middle of April, inflow was lighter than the water at the surface (Fig. 7(A-C)) and floats over the reservoir surface (Fig. 3(A)). Thus, the presence of curtains prevents the direct intrusion of the high level of nutrients to the Zone C. With the high concentration of algae at Zone A and Zone B (Fig 6(A-H)), large amount of nutrients were consumed within these zones, which cause a reduced nutrient supply to the downstream zone of the reservoir.

Homogeneous temperature in Zone A surface layers (Figs. 7(A-C)) indicates that the water in the upper layers gradually settled down until the bottom of the curtain replacing the newly inflow water in Zone A. Since the euphotic layer was only about 4 m deep in this zone, below this depth the algal production was less until it flows up into the Zone B after crossing the curtain. The water flows up along the downstream of the curtain, entraining cool heavy water in Zone B. During this process, entrainment leads to an increase in the density of the upward flow until at some level, where the upward flow and reservoir densities balance and the upward flow penetrates the Zone B. This process increases the submergence level of nutrient-rich interflow, as shown in Fig.8(A), in which phosphorus is highest at 2 m deep in Zone C rather than the surface.

In late spring and summer, epilimnion of the reservoir was heated up, thus inflow becomes heavier than the water at the surface of the reservoir (Fig. 6(C-I)) and plunges beneath the reservoir. This plunge-flow will move down the river channel entraining warmer reservoir water. The entrainment leads to a decrease in density of the plunging inflow until at some level, where the reservoir and inflow density balance and inflow penetrates the reservoir at a level below BC as shown in Fig. 3(B). During this period, presence of curtains reduced the dispersion of nutrients from upstream zones to the downstream zones of the reservoir (Figs. 8(D-H)), which might contributed significant reduction of algal concentration after the middle of April. Figures 6(F-I) indicate that during May and June inflow penetration level reduced close to the withdrawal level. Thus, during this period outflow played a major role by withdrawing nutrient-rich interflow water. However, this process is independent of the presence of curtains, but presence of curtains might have some impact on the penetration level of the inflow.

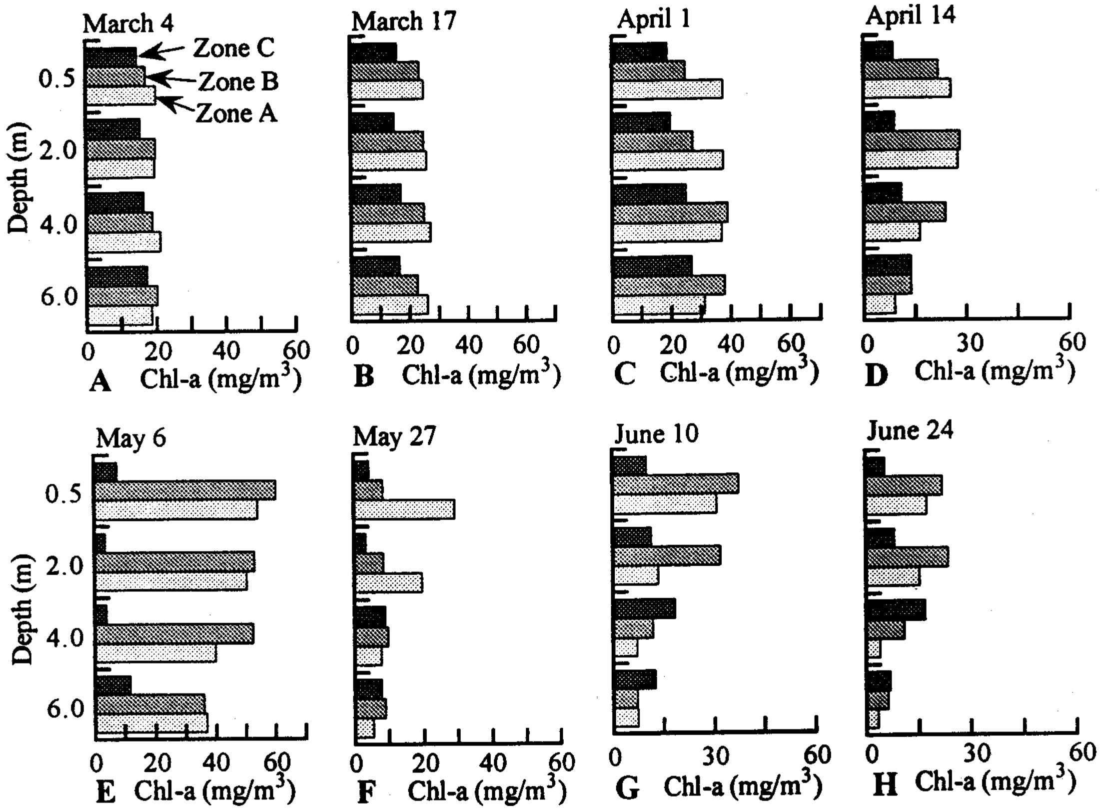

Figure 6: Depth vs. chlorophyll-a concentration in 1994

MODEL FORMULATIONS

Physical Process Sub-Model

To simulate physical state variables such as temperature, Quasi-Two-Dimensional Reservoir Simulation Model (Hocking and Patterson, 1991) is used after including the processes occurring due to the presence of the curtains which are identified by experimental results. The model uses Lagrangian layers that expand, contract, combine, move vertically, and divided in response to the physical process within the reservoir, and horizontal parcels of variable size are included within each of the horizontal layers. Processes included in the model are surface layer deepening, surface heat, mass and momentum exchange, mixing in the hypolimnion, inflow, and outflow.

The length of a time step for each processes varies; a variable time step which is selected internally by the model based on the dynamics of surface layer for the process of surface heat transfer and wind mixing, and a fixed daily time step is used for inflow and outflow. Boundary condition at the curtains is such that there is no flow across the curtains only up to the curtain depth. Flow under the curtains from upstream side to downstream side is also considered to be insensitive to fluctuations on time scale less than one day. Flow is withdrawn under each curtain from the upstream side using the same method described in outflow dynamics in DYRESM by Imberger et al. (1978). If the withdrawal density is less than the density of the parcel downstream of the curtain at the level of curtain bottom, the withdrawal water will flow up entraining reservoir water until at which its density equals that of the reservoir. This process is observed during early spring. Vertical buoyant jet equations described by (Jirka and Harleman, 1973) are used to calculate entrainment to the upward flow. The maximum measured turbidity at the deepest part of the reservoir (Zone C) is 4.0 mg l^{-1}. Hence, the particulate matter settling and the influence of turbidity on the light penetration are assumed to be negligible in the model formulations.

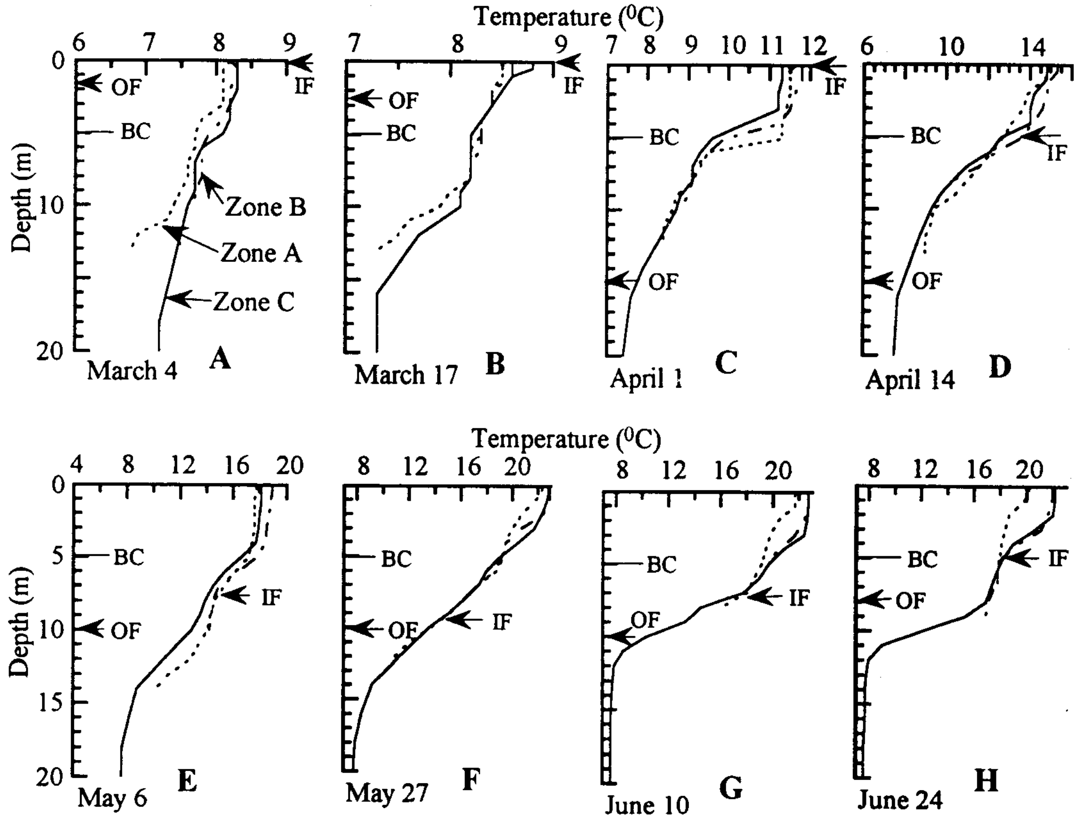

Figure 7: Depth vs. temperature distribution in 1994 (BC - Level correspond to bottom of the curtains, IF - Level correspond to inflow temperature, OF - Outflow level)

Ecological Process Sub-Model

Figure 9 shows the interrelations between the state variables in the ecological sub-model which described the phytoplankton production, dissolved oxygen budget, and nutrient cycling. Vertical diffusion of biological state variables in the hypolimnion is estimated using the turbulent diffusion algorithm described by Imberger and Patterson (1981). Four major groups of phytoplankton are identified in Terauchi Dam Reservoir. Hence, the model considers four phytoplankton groups; diatoms, cyanobacteria, green algae, and flagellates.

The state equations of the ecological sub- model are given below (After Riley and Stefan, 1987, and Hamilton and Schladow 1994).

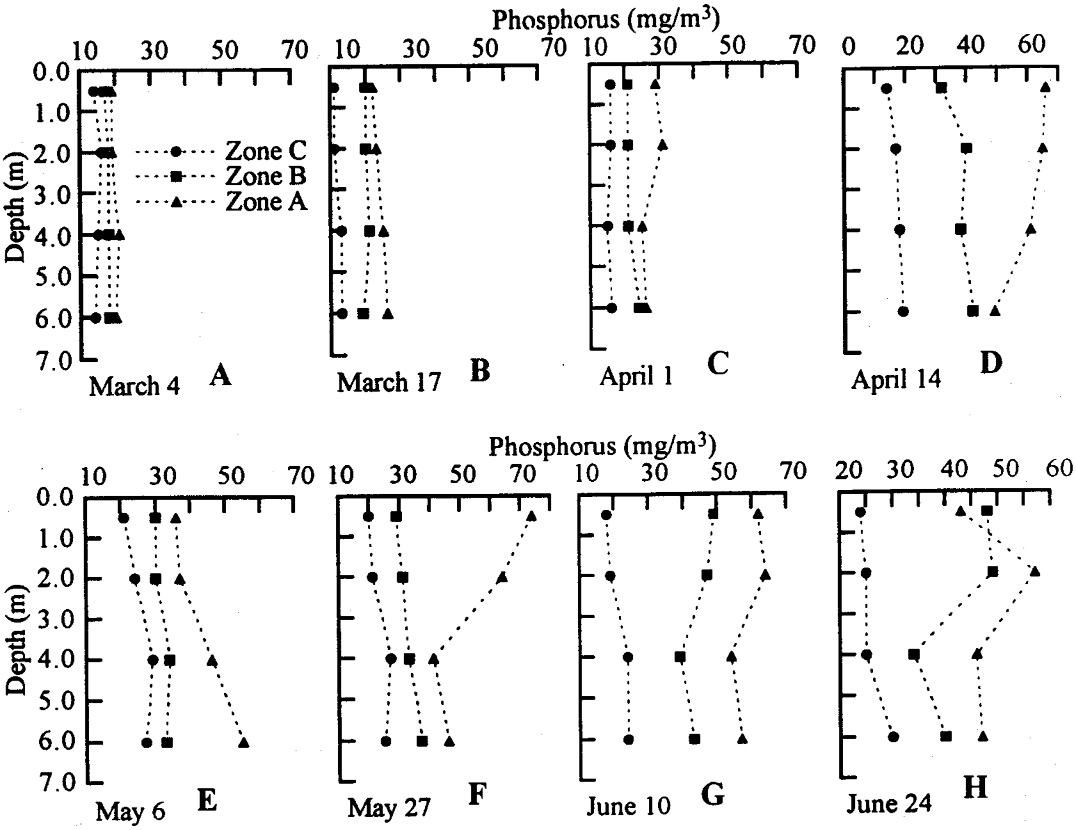

Figure 8: Depth vs. total phosphorus concentration in 1994

(1) $$\frac{\partial Py_i}{\partial t} = \left\{G_{max}\theta_p{}^{T-20}\, min\left[f(IP_i) : f(IN_i) : f(Si) : f(I)\right] - R_i - M_i - S_i\right\} Py_i - G_{Z_i}$$

(2) $$\frac{\partial IP_i}{\partial t} = Up_{max}\theta_p{}^{T-20}\frac{IP_{max}-IP_i}{IP_{max}-IP_{min}}\frac{PO_4}{K_{PO_4}+PO_4}Py_i - R_i IP_i - M_i IP_i - k_{gz}\theta_{gz}{}^{T-20}\frac{Py_i}{K_{gz}+Py_i}P_i\frac{IP_i}{Py_i}Z_p$$

(3) $$\frac{\partial IN_i}{\partial t} = Un_{max}\theta_p{}^{T-20}\frac{IN_{max}-IN_i}{IN_{max}-IN_{min}}f(N)Py_i - R_i IN_i - M_i IN_i - k_{gz}\theta_{gz}{}^{T-20}\frac{Py_i}{K_{gz}+Py_i}P_i\frac{IN_i}{Py_i}Z_p$$

(4) $$\frac{\partial PO_4}{\partial t} = \sum_{i=1}^{4}\left[-Up_{max}\theta_p{}^{T-20}\frac{IP_{max}-IP_i}{IP_{max}-IP_{min}}f(P)Py_i + R_i IP_i + M_i\left(IP_i - IP_{min}\right)\right.$$
$$\left. + k_{gz}\theta_{gz}{}^{T-20}\frac{Py_i}{K_{gz}+Py_i}P_i\frac{IP_i - IP_{min}}{Py_i}Z_p\right] + S_p\theta_s{}^{T-20}\frac{DO+K_{DO}}{DO}\frac{A_s}{V_L} + k_{op}\theta_{bod}{}^{T-20}Y_{Pbod}BOD$$

(5) $$\frac{\partial NO_3}{\partial t} = \sum_{i=1}^{4}\left[-Un_{max}\theta_p{}^{T-20}\frac{IN_{max}-IN_i}{IN_{max}-IN_{min}}f(N)\left(1-P_{NH}\right)Py_i\right] + k_{NO}\theta_{NO}{}^{T-20}\frac{DO}{K_{NO}+DO}NH_4$$

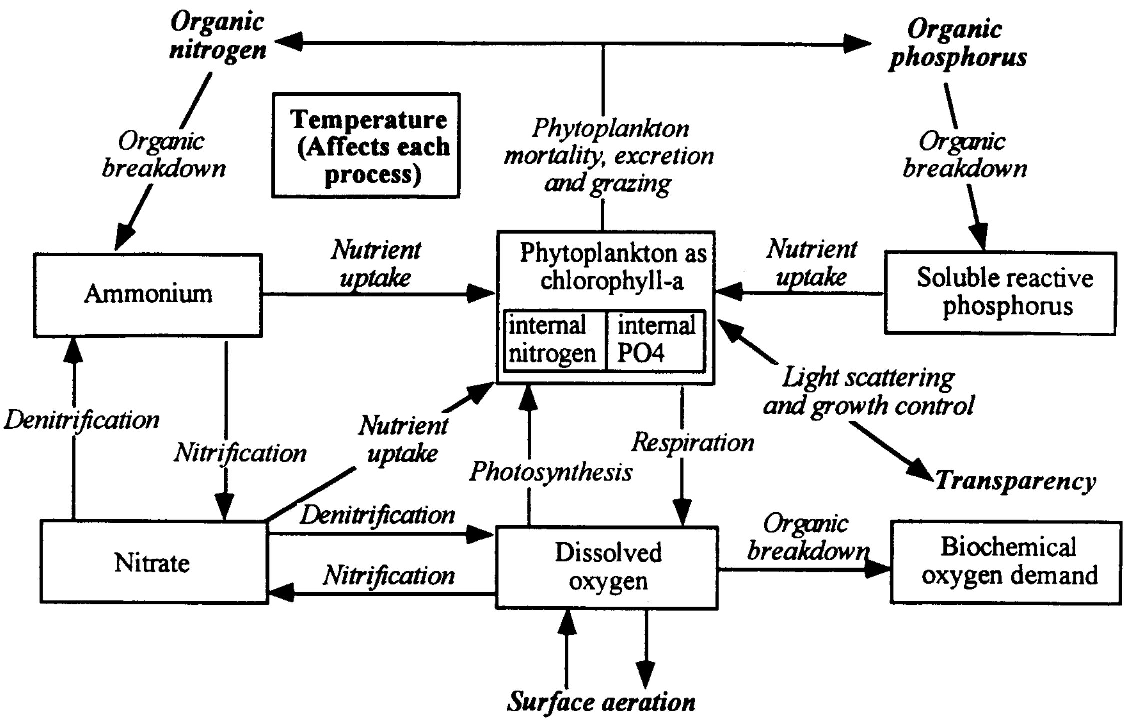

Figure 9: Diagrammatic representation of interactions between main ecological state variables (PO4 - Phosphorus).

(6)
$$\frac{\partial Si}{\partial t} = \left[\left(-G_i + R_i + M_i\right) Py_i I_{Si} + k_{gz}\theta_{gz}^{T-20} \frac{Py_i}{K_{gz} + Py_i} P_i I_{Si} Z_p \right]_{i=1} + k_{USi}\theta_{bod}^{T-20} U_{Si}$$

(7)
$$\frac{\partial NH_4}{\partial t} = \sum_{i=1}^{4} \left[-Up_{max}\theta_p{}^{T-20} \frac{IN_{max} - IN_i}{IN_{max} - IN_{min}} f(N) P_{NH} Py_i + R_i IN_i + M_i\left(IN_i - IN_{min}\right) \right.$$
$$\left. + k_{gz}\theta_{gz}{}^{T-20} \frac{Py_i}{K_{gz} + Py_i} P_i \frac{IN_i - IN_{min}}{Py_i} Z_p \right] + S_n\theta_s{}^{T-20} \frac{DO + K_{DO}}{DO} \frac{A_s}{V_L}$$
$$- k_{NO}\theta_{NO}{}^{T-20} \frac{DO}{K_{NO} + DO} NH_4 + k_{on}\theta_{bod}{}^{T-20} Yn_{bod} BOD$$

(8)
$$\frac{\partial DO}{\partial t} = W_s - W_E - k_{NO}\theta_{NO}{}^{T-20} \frac{DO}{K_{NO} + DO} NH_4 Y_{on}$$
$$+ \sum_{i=1}^{4} \left\{ G_{max}\theta_p{}^{T-20} \min[f(P) : f(N) : f(I)] Py_i Y_{op} - R_i Py_i Y_{op} \right\} - k_{BOD}\theta_{bod}{}^{T-20} \frac{DO}{K_{bod} + DO} BOD$$

$$(9)\quad \frac{\partial BOD}{\partial t} = k_{BOD}\theta_{bod}{}^{T-20}\frac{DO}{K_{bod}+DO}BOD - \sum_{i=1}^{4} k_m\theta_m{}^{T-20}Py_iY_{op}$$

where, A_S= area of sediment in contact with a layer (m^2); BOD = biochemical oxygen demand (mg m^{-3}); DO = dissolved oxygen concentration (mg m^{-3}); $f(I)$ = light limiting function for growth; $f(IN)$ = internal nitrogen limitation function for growth; $f(IP)$ = internal phosphorus function for growth; $f(Si)$ = silicon limitation function for growth considered only when diatoms is modelled (i=1); G_i = gross growth rate of phytoplankton (day^{-1}); G_{max} = maximum growth rate (day^{-1}); G_{Z_i} = loss of phytoplankton from zooplankton grazing per day (mg $m^{-3}day^{-1}$); i = phytoplankton group, 1-diatoms, 2-green algae, 3-cyanobacteria, and 4-flagellate; IN = internal nitrogen concentration (mg m^{-3}); IN_{max} = maximum internal nitrogen concentration (mg m^{-3}); IN_{min}= minimum internal nitrogen concentration for no growth (mg m^{-3}); IP= internal phosphorus concentration (mg m^{-3}); IP_{max} = maximum internal phosphorus concentration (mg m^{-3}); IP_{min} = minimum internal phosphorus concentration for no growth (mg m^{-3}); IS_i = internal silica concentration (mg Si (mg $Py_{i=1}$)$^{-1}$); k_{BOD} = rate coefficient for detrial decay on dissolved oxygen (day^{-1}); k_{gz} = rate coefficient for grazing (day^{-1}); k_m = rate coefficient for mortality (day^{-1}); k_{NO}= rate coefficient for nitrification (day^{-1}); k_{on} = rate coefficient for organic decay of nitrogen (day^{-1}); k_{op} = rate coefficient for organic decay of phosphorus (day^{-1}); k_{USi} = rate coefficient for mineralisation of unreactive silica (day^{-1}); K_{bod} = half saturation constant for dependence of detrial decay on dissolved oxygen (mg m^{-3}); K_{DO} = factor regulating sediment nutrient release with dissolved oxygen concentration (mg m^{-3}); K_{gz} = half saturation constant for zooplankton grazing (mg m^{-3}); K_N = half saturation constant for nitrogen uptake (mg m^{-3}); K_{NO} = half saturation constant for dependence of nitrification or denitrification on dissolved oxygen (mg m^{-3}); M_i = natural mortality except grazing by zooplankton (day^{-1}); NH_4 = concentration of ammonium (mg m^{-3}); NO_3 = concentration of nitrate (mg m^{-3}); P_i = preference factor for specific phytoplankton group; P_{NH} = preferential ammonium uptake; PO_4 = concentration of soluble reactive phosphorus (mg m^{-3}); Py_i = phytoplankton concentration of group i (mg m^{-3}); R_i = respiration of phytoplankton (day^{-1}); Si = reactive silica (mg m^{-3}); S_n = sediment nitrogen release rate (mg m^{-2}); S_p = sediment phosphorus release rate (mg m^{-2}); T = temperature (^{0}C); Un_{max} = maximum rate of nitrogen uptake (day^{-1}); Up_{max} = maximum rate of phosphorus uptake (day^{-1}); US_i = unreactive silica concentration (mg m^{-3}); V_L = maximum rate of phosphorus uptake; W_s = Oxygen flux from surface applies only when considering surface layer of depth Δz_I (mg m^{-3} day^{-1}); W_E = Oxygen demand in sub-euphotic zone applies only wen considering sub-euphotic zone for a layer of depth Δz (mg m^{-3} day^{-1}); Yn_{bod} = ratio of nitrogen release to oxygen utilized in organic decay; Y_{on} = stoichiometric ratio of oxygen to nitrogen for nitrification; Y_{op} = ratio of mass of oxygen produced or respired to mass of chlorophyll-*a*; Yp_{bod} = ratio of phosphorus release to oxygen utilized in organic decay; z = depth measured from water surface to a layer (m); Z_p = zooplankton concentration (mg m^{-3}); Δz = thickness of the layer (m); μ = light extinction coefficient (m^{-1}); μ_b = background light extinction coefficient (m^{-1}); μ_{Py} = specific light extinction coefficient for phytoplankton (m^2 mg^{-1}); θ_{gz} = temperature multiplier for grazing; θ_m = temperature multiplier for mortality; θ_p = temperature multiplier for growth; θ_r = temperature multiplier for respiration; θ_s, θ_b = temperature adjustment coefficient; θ_{NO} = temperature multiplier for nitrification.

Model Simulation

The physical sub-model is independent of calibration of the ecological component of the model. Figure 10 shows the measured and simulated temperature distribution of surface layers of Zone A, Zone B, and Zone C. A reasonable match was observed between measured and simulated temperature distributions. The ecological sub-model is calibrated over 115 days available data, from the beginning of March to the end of June, 1994. The model was initialized using the measurements taken on March 3. Values other than temperature below 6 m depth were set equal to the values measured at 6 m. Total chlorophyll-*a* concentration was converted into each group by considering the phytoplankton counts, assuming linear relation between cell numbers and chlorophyll-*a* for each group. Measured and simulated total chlorophyll-*a* concentration at the surface layer of the three zones are shown in Fig. 11. In Zone A, estimated values showed variation from some measured values. This variation could be due to several reasons. One reason is that it may be due to chlorophyll-*a* concentration of the inflow. The inflow chlorophyll-*a* was measured once a month, hence interpolated values between two months were used. These interpolated values might have higher deviation from their real values. Another reason could be due to the average parameter values. One parameter represents the average values of several species. As each species has its own characteristic parameters, the variation in the species composition inevitably gives a corresponding variation in the average parameter used in the model.

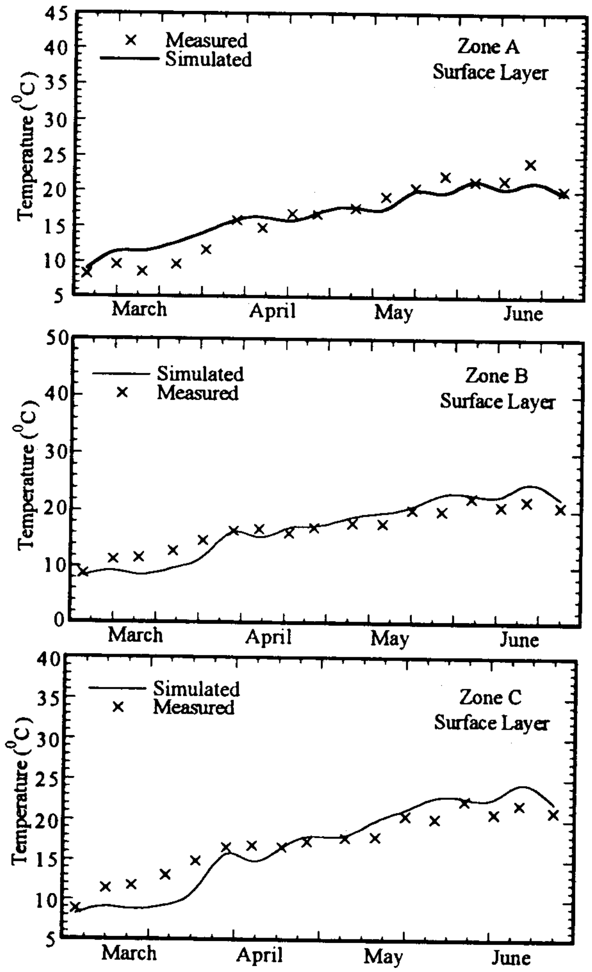

Figure 10: Measured and simulated temperature distributions in 1994.

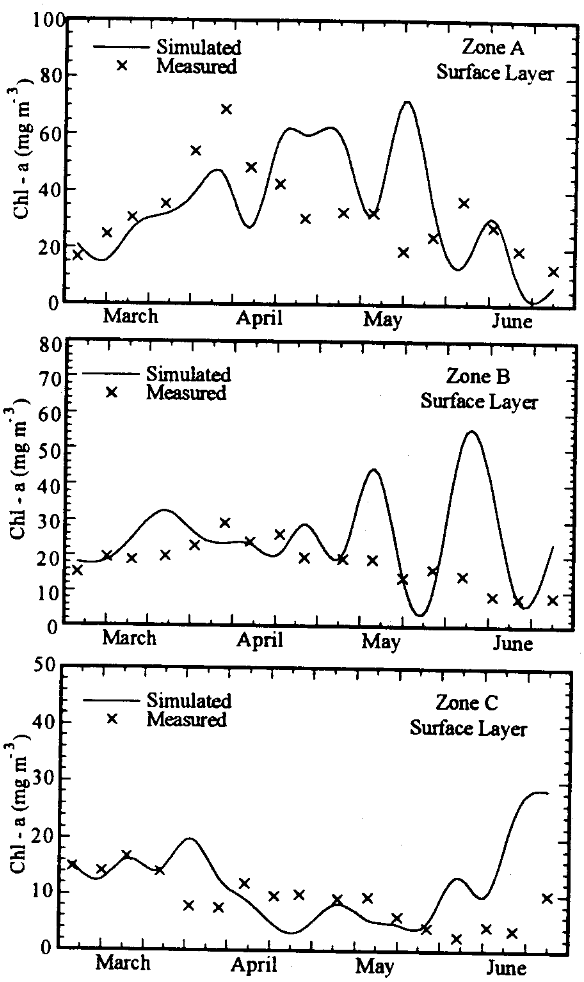

Figure 11: Measured and simulated chlorophyll-*a* distributions in 1994.

CONCLUSION AND DISCUSSION

The experimental results of this study shows an ecotechnology that could be used to reduce algal blooming in downstream withdrawal zone of the reservoir. The algal blooming at withdrawal zone (Zone C) reduced significantly, compared to the upstream zones (Fig. 8) and pre-curtain years (Fig.1), after installing the curtains. The possible mechanisms, for the reduction of algal blooming in withdrawal zone, identified by the experimental data are;

(1) During early spring up to the middle of April
- (a) Inflow is lighter than the water at the reservoir surface and floats over the reservoir surface. Presence of curtains prevent intrusion of this surface flow to the downstream zones.
- (b) Algal concentration is higher at upstream zones. Thus, within upstream zones algae consume high amount of inflow nutrient which cause a reduced nutrient supply to the downstream zones of the reservoir.
- (c) Entrainment leads to an increase in the density of the upward flow which increase the submergence level of nutrient-rich interflow.

(2) During late spring and summer
- (a) Curtail the dispersion of nutrients from upstream zones to the downstream zones of the reservoir.
- (b) Outflow plays a major role by withdrawing nutrient-rich interflow water.

The simulation study has been presented a model that combines the mixing dynamics of a stratified lake with the biological processes, diffusion, and flow under the curtains. The model has been run using lake data from the Terauchi Dam Reservoir in Kyushu, Japan. The ecological sub-model is calibrated over 115 days available data, from the beginning of March to the end of June, 1994. The model was initialized using the measurements taken on March 3. The measurement include each zone profiles of temperature, total chlorophyll-*a*, phosphorus, nitrate nitrogen, ammonium, BOD, and silica. The model simulate the chlorophyll-*a* concentration in the three zones of the reservoir with reasonable accuracy.

Curtain method can mostly be applied for a reservoir or lake having an elongated shape. This method is particularly a low-cost technique with higher degree of reliability and simplicity compared to other existing control measures. Installation cost of the curtain in Terauchi Dam Reservoir is about 3/4 of the installation cost of bubble plume.

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AERATION AND MINIMUM FLOW ENHANCEMENTS AT HYDROPOWER DAMS

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ABSTRACT

The Tennessee Valley Authority is completing a five-year program to improve the dissolved oxygen content of hydroturbine releases and to maintain minimum flows downstream from sixteen multipurpose dams. The objective of this program is to enhance water quality and aquatic habitat in the rivers downstream from the dams. Several aeration techniques have been developed and utilized to increase dissolved oxygen. These techniques include turbine venting, forced air injection, diffusion of oxygen into the turbine penstock, surface water pumps, diffusion of air or oxygen into the upstream reservoir, and construction of aerating weirs downstream from the dam. Techniques applied for maintaining minimum flow include the installation of small supplemental hydroturbines, pulsing operation of the existing turbines, and construction of reregulation weirs downstream.

This paper describes the various aeration and minimum flow techniques and discusses the effects of these applications on project operation and downstream water quality.

KEY-WORDS: Aeration / Diffusers / Dissolved Oxygen / Dams / Minimum Flow / Reservoir / Hydroturbine Venting / Water Quality / Weirs.

INTRODUCTION

The Tennessee Valley Authority (TVA) has built or acquired 48 dams which are operated to provide multiple public benefits. The system was originally designed to provide navigation, flood control, and the production of hydroelectric power. Operating the dams for these purposes, especially for hydropower, has resulted in intermittent low flow, or no flow, from some of the dams and low dissolved oxygen (DO) levels for these discharges during the summer and fall months.

During the summer and fall months, water in the reservoirs created by the dams becomes thermally stratified. Over time, chemical and biological processes use up the DO in the hypolimnion. Hydroturbine intakes draw water from the hypolimnion and thus the hydroturbine discharges become low in DO. In addition, the operation of the dams to provide economical hydropower generation, when needed, results in intermittent flows. This creates periods when the downstream reaches of the river may remain dry for extended periods of time.

In February 1991, the TVA Board of Directors approved a five-year Lake Improvement Plan which included a commitment to improve the quality of the releases at sixteen dams by increasing DO levels and sustaining minimum flows. Minimum flow releases were initiated as soon as the plan was approved and techniques have been implemented to extend river reaches with instantaneous minimum flows. Aeration systems have been implemented at 12 dams and work is underway to install aeration systems at the remaining dams during 1996.

DISSOLVED OXYGEN ENHANCEMENT TECHNIQUES

The techniques used to increase DO in the turbine discharges include turbine venting, compressed air injection, surface water pumps, penstock oxygen injection, forebay aeration, and aerating weirs.

Turbine Venting

Turbine venting utilizes suction or low pressure to induce air into the water passages of the hydroturbine (Figure 1). This method is usually the least expensive, least obtrusive, and most maintenance-free method of improving the dissolved oxygen concentrations in hydroturbine releases. Therefore it is usually the first choice. However, the technique is site-specific and its success depends upon the geometry of the turbine and draft tube; the elevation of the hydroturbine with respect to tailwater; and the size, location, and flow resistance of the air supply passages (Carter, 1995). TVA has utilized turbine venting to supply all or part of the needed oxygenation at 21 turbines operated at eight projects. At most of these projects, the natural suction created by the flowing water has been increased by the addition of baffles over the thrust relief or vacuum breaker ports on the turbine hub. In some cases, it was also necessary to move the location of the ports. At all projects, additional openings have been cut into the turbine headcover and piping has been added to produce a path for the air which by-passes the existing vacuum breaker system. The disadvantage of turbine venting is a slight decrease (usually less than 0.5 percent) in turbine efficiency during venting operations.

TVA, in cooperation with Voith Hydro, has incorporated autoventing techniques into the manufacture of turbine runners (Hopping, 1996). This new design utilizes low pressure areas identified in scale model testing to induce air into the turbine releases. These model results were incorporated into the design for new turbine runners for Norris Dam. Testing of these new runners has shown that DO increases of 5 mg/L or more are possible. Thses are the first replacement turbines in the world designed specifically to aerate turbine releases, with the additional benefit of increased capacity and efficiency of the generating units.

Compressed Air Injection

At two projects, high volume/low pressure blowers are used to force air into the water as it passes through the turbines. The blowers are located in separate buildings adjacent to or near the powerhouse and the air is piped to the water passages. At the Nottely project, the air is injected under the turbine headcover through a vacuum breaker by-pass system. At the Tims Ford project, air is injected both under the turbine headcover and through a manifold installed in the draft tube just downstream from the turbine. About 4 mg/L of DO are added by the blowers at Nottely and about 2 mg/L are added at Tims Ford.

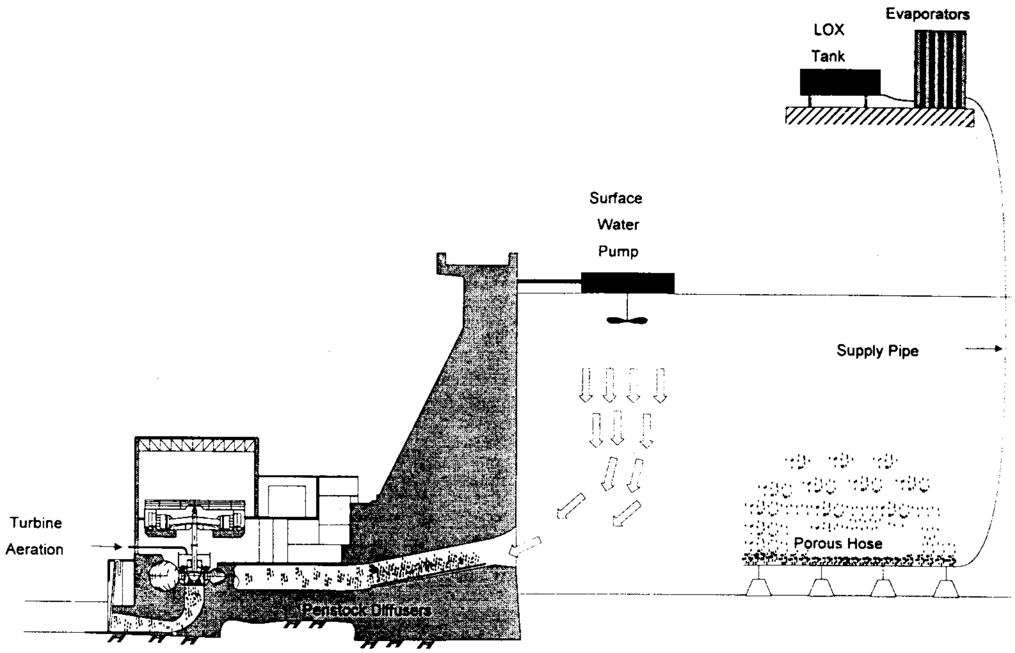

Figure 1. Reservoir Dissolved Oxygen Enhancement Options

Surface Water Pumps

Surface water pumps are being used to improve the DO in the releases at two TVA projects, Cherokee and Douglas. The pumps resemble huge ceiling fans that are positioned above the turbine intakes at the dam (see Figure 1). The pumps and the electric motors used to drive them are mounted on floats that are attached to a rail system which allow the system to rise and fall with the water surface. Variable speed controllers for the pumps are located in a gallery inside the dam. Surface water pumps improve the DO in turbine releases by moving warm, oxygen-rich surface water downward to the turbine intakes. Test data indicates that the pumps can improve the DO downstream of a dam by up to 3 mg/L, depending upon the DO profile in the reservoir. The disadvantage of surface water pumps include an increase in discharge water temperature and a decrease in their effectiveness if the surface water is low in DO.

Penstock Oxygen Injection

At the Tims Ford project, in addition to the compressed air injection, aeration is supplemented by injecting gaseous oxygen into the penstock (Harshbarger et al., 1995). A schematic of the penstock aeration system is shown in Figure 1. Gaseous oxygen from an on-site bulk storage facility is bubbled through porous hoses attached to the walls of the penstock. There are two sets of hoses each about 15 meters long, installed over the bottom half of the penstock beginning approximately 12 meters downstream from the penstock entrance. Each set contains 132 hoses connected to a 5-cm diameter header attached to the penstock wall. The hoses are commercially available, 1.3-cm diameter porous hoses made of recycled automobile tires. This system is capable of diffusing about 0.4 Nm^3/s of oxygen into the water flow. The DO uptake is about 2 mg/L.

Forebay Aeration

TVA has developed an efficient and economical reservoir oxygen line diffuser design that has been installed and operated successfully at five TVA hydropower projects, one TVA nuclear plant, and two non-power reservoirs (Mobley et al., 1996). The line diffuser is a two pipe system, with a gas supply header pipe and a buoyancy chamber pipe, as shown in Figure 2. The diffusers can be supplied with air or oxygen, either from a bulk liquid oxygen storage tank, an onsite air separation plant, or air compressors.

The line diffuser can be assembled and deployed without divers because the buoyancy pipe supports the entire weight of the diffuser in water, including the concrete anchors. Once the assembled diffuser is positioned on the water surface above the desired location, the buoyancy pipe is flooded to allow the diffuser to sink, in a controlled manner, to the reservoir bottom. The process is reversed to retrieve a diffuser for repositioning or maintenance.

The line diffuser has also been applied to meet other water quality requirements within reservoirs. TVA has installed a line diffuser system supplied with compressed air at Normandy Dam, a non-power project, to precipitate hydrogen sulfide, iron, and manganese in the reservoir. Another diffuser using compressed air was installed to provide aeration of an intake embayment at Sequoyah Nuclear Plant to prevent fish kills. A TVA-designed line diffuser system was also installed in a water supply reservoir near Madrid, Spain, to reduce the treatment required for drinking water.

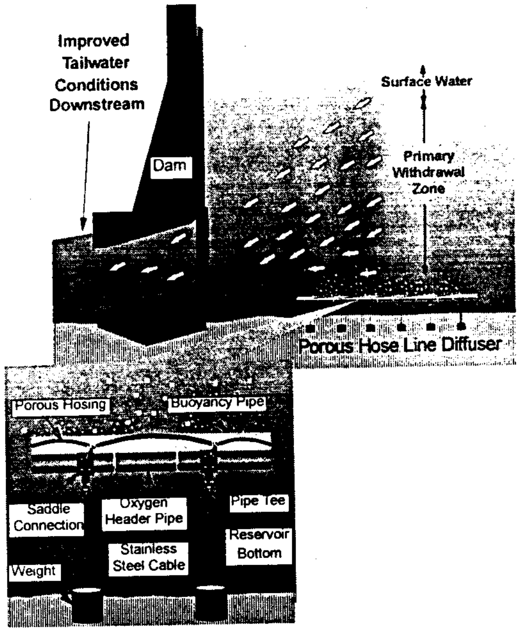

Figure 2. Line Diffuser

Aerating Weirs

Aerating weirs are often favored over other methods because they are passive, reliable, low maintenance, and they aerate flow from the dam regardless of outlet of origin (when constructed from bank to bank). Aeration of oxygen-deficient turbine releases is achieved by water overtopping the weir and plunging into a downstream pool in a process similar to natural waterfalls. TVA research has focused on ways to improve weir aeration beyond that of conventional weirs while maintaining safe flow conditions in the vicinity of the structure. Two different kinds of aerating weirs have been utilized: the infuser (US Letter of Patent, 5,462,657, Rizk and Hauser), and the labyrinth.

Infuser Weir - Chatuge

An infuser is a hollow, broadcrest weir with specially-designed transverse openings in its crest that create a series of transverse water curtains that fall through the crest to a plunge pool below (see Figure 3). The Chatuge weir employs a slotted infuser deck attached to the downstream face of a conventional linear weir to achieve these flow patterns. The weir component is a stepped timber crib filled with loose rock and lined with tongue-and-groove timbers along its upstream face to make it impermeable. Timbers are also used to support the steel foot grating on the attached infuser deck. The spaces between the deck timbers allow flow through, creating a series of turbulent waterfalls. These deck openings increase in size in the downstream direction to maintain approximately uniform flow in each as the head on each opening decreases in the downstream direction. The timbers creating the blockages between deck openings are supported by concrete beams and columns. Aeration from the Chatuge weir has been exceeding 6 mg/L with limited operation and maintenance. For a drop height of 2.1 m, aeration tests for this weir showed an aeration efficiency of 68 percent during normal turbine discharge of 37 m^3/s (Hauser et al., 1996).

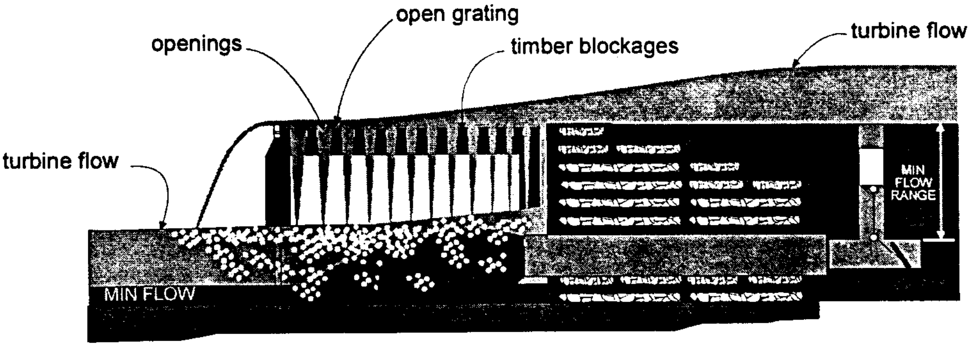

Figure 3. Elevation View of the Chatuge Infuser Weir

Labyrinth Weir - South Holston

The South Holston labyrinth is a long, serpentine weir that creates a 640-m wide waterfall that is overtopped during turbine operations. The substantial crest length required for a labyrinth weir is necessary to reduce the specific discharge to a level that reduces the intensity of the downstream recirculation, or "roller," to a safe level. The required labyrinth crest length is that needed to yield a safe specific discharge--i.e., turbine flow divided by the "safe" specific discharge of 0.14 m^2/s (Hauser, 1991). The South Holston weir has been meeting or exceeding the aeration target of 6 mg/L with minimum operation and maintenance. For a drop height of 1.5 m, aeration tests for this weir showed an aeration efficiency of approximately 65 percent during normal turbine discharge of 68 m^3/s.

Figure 4. South Holston Labyrinth Weir

MINIMUM FLOW TECHNIQUES

Pulsing

Pulsing is a simple technique whereby the turbine(s) are operated periodically to supply water downstream. The drawback with this method is that, for the reaches of the river near the dam, the flow is very unsteady and fluctuating. It frequently requires several miles for the flow from the pulses to become steady. Pulsing by itself is now in use at seven TVA projects.

Reregulation Weirs

A reregulation weir is a relatively low structure built in the river downstream of the dam to create a pool of water which can be used to provide a minimum flow when the turbines are not operating. Periodic pulsing of the turbines is necessary to fill the pool, but the pulsing frequency is greatly reduced from that needed without the weir. When the turbines are operating, the weir is overtopped. When the turbines are off, water from the weir pool is discharged through low level pipes. Most of these pipes are equipped with float-operated valves so that the flow remains constant as the pool level drops. TVA's Norris project has a reregulation weir for minimum flow and the Chatuge and South Holston projects have combination aeration and minimum flow weirs. Typical low level pipes and float-operated valves are shown in Figure 3.

Supplemental Turbines

At the Tims Ford, Blue Ridge, and Nottely projects, TVA has installed small turbines that are operated to provide minimum flows whenever the main turbines are not needed. These small turbines discharge approximately 2.3 m^3/s, 3.4 m^3/s, 1.7 m^3/s, respectively. This flow provides a greatly increased wetted area in the river while generating electricity for the hydropower system.

SUMMARY

TVA has developed several techniques for increasing the dissolved oxygen content of turbine discharges and for providing minimum flow downstream from hydropower dams. The Lake Improvement Plan included significant research and development efforts during its implementation. This has resulted in state-of-the-art systems which include the development of new technologies and the improvement in existing aeration concepts. These concepts include autoventing turbines, conventional turbine venting, compressed air injection, surface water pumps, penstock oxygen injection, forebay aeration, and aerating weirs. Downstream minimum flow is provided below the dam by either turbine pulsing, the addition of a small hydrounit, or a reregulation weir. Application of these techniques is site-specific. Twelve minimum flow projects have been completed and four others are underway.

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EKOS - A SOFTWARE PROGRAM FOR ANALYZING BANKS USED BY ANIMAL SPECIES

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ABSTRACT

In the design and management of banks the requirements of animal species should be taken into account. Therefore there is a need of accessible information of wetland species and a method to translate this information into the most optimal measures to be taken in a bank. EKOS (Ecological Quality of Banks for the Benefit of Animal Species) is a software program which provides for these needs.

KEY-WORDS: HSI-models/ river bank/ fauna/design/ management/habitat/compensation/

INTRODUCTION

According to the Dutch policy documents on water management, the natural values of banks have to be restored and improved (Ministry of Transport and Public Works, 1989). In the past years many efforts have been taken to enlarge wetland vegetation along Dutch banks. For the fauna however are hardly any specific measures taken. This is mainly caused by the fact that designers and managers of banks do not know the living conditions of wetland bound fauna.

The tasks of the Road and Hydraulic Engineering Division of the Ministry of Transport, Public Works and Watermanagement are to advise bank managers how to improve their banks into more natural systems and to develop instruments for these managers to predict the effects of these improvements.
By developing the software program EKOS (Ecological Quality of Banks for the Benefit of Animal Species) our division made an instrument to predict the suitability of a bank for a species, and to make clear which aspects limit the possibilities for a species.

EKOS is based on Habitat Suitability Index models. These models contain a description of variables which affects the suitability of an area for a species. When the value of a variable has a optimum for the species, the value gets a HSI-value of 1.0. Values which correspond to an unlivable situation for a species get a 0.0. To translate the separated values into a main HSI value, each of the HSI values is weighed against one another.

THE SOFTWARE PROGRAM EKOS

EKOS is the title of a software program containing a comprehensive database of animal species that use banks. Using location data, the program can compute or show the following:

1. a general description of the ecology of the species and the used model.
2. a description of the geographical distribution of the species in 18 regions in the Netherlands.
3. a description of landscapes types the species normally occurs in.
4. a description of the variables of the bank and immediate surroundings affecting the occurrence of the species.
5. a description of the variables of the bank that limit the opportunities to settle and to reproduce.
6. a calculation of the suitability-index of the habitat.
7. a calculation of the number of habitat units.

<u>1. General information of the model and the species.</u>

The user of EKOS needs some information about the presented model before using. A notebook containing additional information fulfil these needs.

For each species more habitats can be described. For example: the foraging habitat, the breeding habitat, the winter habitat and resting habitat. It is therefore possible to have more models for one species and necessary to describe the used habitat. A model containing the needs of the foraging habitat is incomparable to a winter habitat. A clear description of the described habitat is therefor necessary.

Some models are limited in use because not all relevant aspects are described in the model. It is know that these aspects can affect the suitability for a species but there are no clear methods available to describe this aspect. For instance, some species are sensitive to disturbance but it is unclear how to measure this sensitivity. In the general information of a model the user is informed which aspects are not described in the model but could affect the occurrence of the species in the bank.

Besides the description of the model, also a description of the species is given. General information of the ecology, the importance of the species in policy and some information about the population density is given. Table 1 lists the species (and models) described by our division (Winden et al., 1996).
This background information is added to provide the user a guide for selecting models and species.

2. Geographical distribution

Using mainly differences in soil types, the Netherlands is divided in 18 regions. The occurrence of most species is well related to these regions. The occurrence of each species within a region is described using terms as main area, minor area and area in which species is absent.
For most species we used the current distribution of the species. However, for species still colonizing new areas the potential distribution is described. The user can apply this information to select the species and the models. For the user only measures for species living in his region are relevant.

3. Types of landscape.

Although the region can belong to the main distribution area of the species no information is given about the distribution pattern within the region. The occurrence within a region is mainly restricted to the available landscape.
For the Netherlands 90 different types of landscape have been recognized. For each model, containing one species and one habitat type, the landscape types corresponding to the described habitat are listed.
Besides the geographic distribution, also the different landscape type can be used to select relevant models. Choosing species which does not live in the type of landscape surrounding the bank is not advisable. Occupation of the bank by these species is not likely.

4. Main variables

EKOS can generate a list of the variables having affect on the suitability of an area for a species. With this list the user gets insight into the variables described in the model and how useful the model can be for his aims. It also is an indication of the complexity of the model (and species).

5. Average Habitat Suitability Index

For each variable (and each species) the relationship between variable value and its suitability is expressed in index values between 0.0 to 1.0. If an index value of 0.0 means that the variable is unsuitable for the species, values of 1.0 mean that the variable has an optimum.
To become the average HSI value, all index values of the individual variables have to weighed against one another (figure 1). The average HSI gives a general indication of the suitability of the area. Interacting variables and independent variables are balanced on different ways: the indices for interacting variables affect each other

Table 1: HSI-models developed by the Road and Hydraulic Engineering Division

class	species	habitat type	version
Amphibians	Common Toad	T	2
	Natterjack	T	2
	Common Tree Frog	T	2
	Pool Frog	R	2
	Lake Frog	R	2
	Edible Frog	R	2
	Common Frog	T	2
Reptiles	Grass snake	T	2
Mammals	Daubenton's bat	F	2
	Pond bat	F	2
	Root vole	T	1
	Water vole	T	1
	Muskrat	T	1
	Beaver	T	1
Birds	Bittern	R	2
	Greylag Goose	R	2
	Garganey	R	2
	Shoveler	R	2
	Water Rail	R	2
	Moorhen	R	2
	Little Ringed Plover	R	2
	Snipe	R	2
	Black Tern	R	2
	Common Tern	R	2
	Kingfisher	R	2
	Bluetroat	R	2
	Marsh Wabler	R	2
	Bearded Tit	R	2
	Reed Bunting	R	2

R	= reproduction habitat	1	= literature
T	= whole life cycles	2	= validation specialist
F	= foraging habitat	3	= validation with field data

and are therefore averaged. In case of independent variables the lowest index value is used to describe the average suitability.

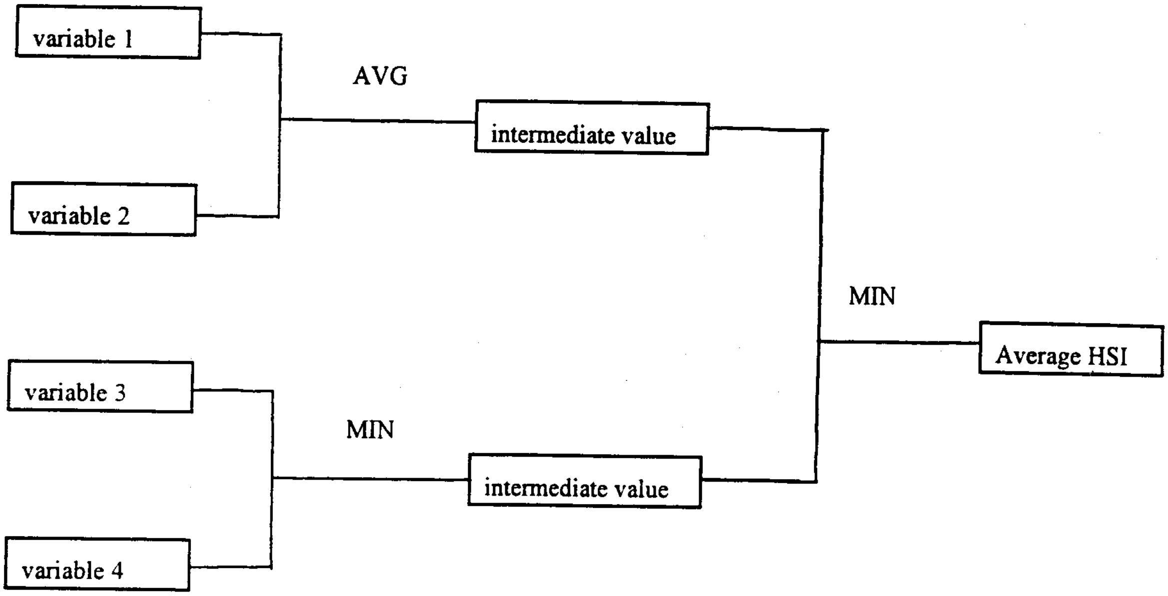

Figure 1: Technique to determine the average Habitat suitability Index. Variable 1 and 2 are interacting variables; variable 3 and 4 are independant variables.

The index values give the user general information about the suitability of his bank for the species. When a species gets an index of zero it is clear that somehow his bank is unfit for the species and changing the management or the design of the bank is necessary. Values of 1.0 indicate no necessary changes in management or design.

The average HSI can also be used to compare different scenarios of bank design and for management.

6. Limiting variables.

With EKOS it is possible to reconstruct which variable causes the averaged HSI-value. This is the limiting variable. Depending on the average HSI-value and the importance of the species for the manager of river banks, the manager can consider changes in management or design. The limiting variable is the guideline to improve the life conditions of the species.

7. Habitat units

In the Netherlands the idea is growing that loss of nature should be compensated. Nowadays compensation takes place based on the area size. The quality of this area is not taken into account. EKOS gives a (first) possibility to compensate also for the change of quality. This by using the formula:

$$HSI * A = HU$$

in which

HSI	= Average Habitat Suitability Index
A	= size of the area
HU	= Habitat unit; area optimal habitat

An area of 6 km² with an average HSI of 1.0 becomes a HU of 6. This area should be compensated by an area with the same HU-value, like an area of 10 km² with a HSI of 0.6, or an area of 7.5 km² with a HSI of 0.8.

This method however, is not (always) the best thinkable solution for compensation questions. Especially areas with well staggered habitat suitability indices gives doubts: it is not likely that an area with a HSI of 1.0 is comparable to an area 10 times that area but now with a HSI of 0.1.
As long as there is no better method available this tool prevents compensation practices only based on the size of area without any reference to the quality of the area. Further research on this aspect of EKOS is needed.

Limitations of EKOS

EKOS is, at least at this moment, program which provides basic knowledge about animals and gives guidelines for species related bank design and management.
However, we can imagine the use of EKOS to solve other fauna related problems, like compensation. Many of these problems are hard to solve because the knowledge of the ecology of the species is not adequate. A few examples:

Occupation chance

The main purpose of EKOS is to give a general opinion of the suitability of a bank (and its surroundings) for a species. It also gives a description of the limiting variable(s).
However, EKOS does not predict the chance of getting these species into the bank zone. To predict these chances more information is needed about the distribution pattern of the surrounding populations, the size of these populations and the migrating capacity of the species. In most cases the information is insufficient to incorporate these variables into the HSI-model. We advise to make banks habitable for species living in the direct environment. The chances of occupation of the bank by the species is largest in this situation.

Reliability of prediction.

The reliability of the HSI-models is questionable. Validation of HSI models with field data is scarce (Lancia et al., 1982; Cole and Smith, 1983; Mc Crain, 1992). Therefore 10 of our models of reptiles and amphibians will this year be validated. An inventory of these animals and the variables of their models within 300 potential reproduction areas will take place.
A comparison between the reproduction success in the field will be compared to the calculated HSI. At this moment we can not present results of this comparison.

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Water quality and habitats

La qualité de l'eau et les habitats

WATER QUALITY ASSESSMENT AND MANAGEMENT USING NUMERICAL SIMULATION TOOLS

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ABSTRACT

Numerical modeling of water flows is a versatile tool for the study and comprehension of phenomena affecting water quality. Current models combine different numerical techniques with associated software for analyzing the results and presenting them visually to the user. Three different examples underline the potential of the numerical approach. Such a tool will eventually be of great help to the non-specialist local administrator who has the responsibility of assessing environmental impacts or emergencies related to among others, such things as abnormal bacterial concentrations or toxic product spills.

KEY-WORDS: Water Quality/Environmental Impact Assessment/Quality Management/Environmental Emergency/Numerical Simulation method.

INTRODUCTION

The province of Quebec is blessed with an important portion of the total water resources on the planet, due to its strategic location within the basin containing the Great Lakes, its long maritime coast and its climate. Available in quantity and very accessible, this water is of unequalled quality. However, economic, industrial and demographic pressures have contributed to a certain deterioration of the aquatic habitats and their associated populations (Ministère de l'Environnement, 1993).

Numerical modeling is an effective tool (1) To understand the behavior of a waterway, a bay, or an estuary, (2) To study their evolution, (3) To quantify some impacts of human intervention on this water resource and (4) To investigate an abnormal situation and study remedial measures. Numerical models generally treat three aspects of water behavior: the hydrodynamics, mixing processes and qualitative evolution phenomena and the interaction with the flow bed. It generally appeals to two methods of analysis: the so-called Eulerian description consists in observing the characteristics of fluid particles traversing given points in space; the Lagrangian method on the other hand consists in following the evolution of the characteristics of the sample particles during their movement (GRAF, ALTINKAR 1995). The two methods are complementary and coupling between the two approaches is nowadays possible due to the strides in numerical modeling techniques recently realized. The purpose of this paper is to briefly explain these techniques and to show how they may be used to analyze different scenarios. Three concrete examples are presented.

NUMERICAL MODELING AND TECHNIQUES

The quantitative and qualitative aspects of the hydrodynamic behavior and mixing processes at a site, such as the interaction between the flow and the bathymetry, are described by well-known physical laws. They are generally written in the form of a system of non linear partial derivative equations and are therefore not amenable to exact solutions except in the most trivial of cases (CUNGE et al, 1980, LOUCKS et al, 1981).

Recourse must therefore be made to numerical techniques so that cases of physical interest may be treated. The numerical method consists in transforming the governing equations into a system of linear algebraic equations applied along the nodes of a computational mesh superposed on the region to be studied. The finite difference and finite element methods have been used for the past 20 years to obtain these numerical solutions. The advantages and inconveniences of each of these techniques evolve from year to year depending on the development of hardware and software support available.

The horizontal two dimensional form of these equations is usually retained in the study of habitats since it respects the preponderance of horizontal phenomena over vertical phenomena (speed, exchange, constraints, etc) (LECLERC et al, 1994).

A numerical model (MIMES which in French stands for Modèle Informatisé du Mouvement des Eaux de Surface) has been used to obtain the results summarized in this article. It consists of four interconnected modules:

- C2E	2D water circulation
- D2C	2D contaminant dispersion
- T2E	2D water heat
- M2F	2D sediment transport.

They are capped with an analysis and flow modeling software (LAME) whose role is to automate data entry and to produce various numerical and graphical forms of results synthesis (Figure 1). It can be used under DOS or WINDOWS in a micro-computer environment (FAUBERT 1995).

Continuous coupling of calculation exists between C2E and M2F to take into account the erosion and deposit effects in the flowing calculation, and LAME manages automatically the transfer of analysis in Euler variables to analysis in Lagrange variables according to the user's needs.

APPLICATION EXAMPLES

The simultaneous computation of the Lagrange and Euler formulations in the numerical modeling enables three distinct study objectives to be realized. Each will be illustrated with a realistic case which is of current concern to a significant number of inhabitants that live in close proximity to a river shoreline.

1. Calculation of the instantaneous and cumulative exposure of piscicultural communities to high bacterial concentrations.

The global dispersion of two independent sources of microbial pollution in the waters inside a partially closed bay called "barachois" locally has been calculated over various tidal cycles according to the seasonal discharge of the four main rivers flowing into the barachois.

Figure 2 shows as an example the superposition of predictable dilution zones at high tide for low water flow and provides the precise location of a mud clam bed (Mya arenaria) not currently exploited.

The S_1 source in the south is essentially diffused and makes up a strong dilution. The influence of the S_2 source on the constraint exerted on the shoal is of primordial importance. The analysis of successive situations encountered over a given period allows prediction of the accumulation of bacterial concentration (expressed in "coliform . hour/100 ml") imposed by each of the discharges while allowing superposition of the natural autopurification rhythm of the mud clam that comes about when the pollutant source is intermittent.

2. Identification of the probable zone containing the source of an observed contamination

The source of temporary bacterial contamination is not always easy to identify. On the one hand, there may be multiple sources, while on the other, detection is long and may necessitate a high sampling frequency. Numerical modeling allows one to define the zone to be sampled and studied.

Even though its wastewater is dispatched to a purification pond by a vacuum-type system, the municipality of Maria notices on rare occasions, unacceptably high bacterial concentrations on its beach. The beach forms part of the shoreline of the "Chaleur Bay" whose currents change according to tides, fluvial discharge and dominant winds.

The modeling of local hydraulic conditions has allowed a temporal backtrack of the particle paths arriving at the beach at each stage of the tidal cycle while taking into consideration the life span of the coliforms and the estimated progressive dilution of the contaminant.

Figure 3 indicates the influence of the pollution sources to the pollution levels which would be measured on the beach. Within this zone, many sources of diffused pollution and two seagull colonies have been observed. This figure indicates the order of magnitude of the concentration source to seek depending on the distance to the beach.

3. Establishing priority intervention zones in case of environmental emergency

Transport by trucks, trains or boats multiply the risks of water contamination of rivers, lakes or estuaries by spillage of toxic products of hydrocarbons.

In the later instance, the spills produced are transported by the currents and their trajectories will also be sensitive to the wind patterns. Numerical modeling allows synthesis of a local data bank of the predictable current fields according to the observed conditions of flow, wind and detailed tidal characteristics.

The tracing by advection can then be used to rapidly identify the path of the spill, the marine zone on which partial dissolution or deposition may occur and the coastal zone which is at risk. Such an analysis thus enables the target zone to be quickly identified, so that an appropriate level of intervention may be chosen (confining booms, dispersion, repumping, etc) and put into operation at the earliest opportunity.

Figure 4 illustrates the path, calculated over 2 tidal cycles, of a spill detected at high tide in front of Carleton, with an east wind prevailing.

CONCLUSION

The previous examples show the versatility and usefulness of this tool in the quality control of the environment. The numerical hydrodynamic model allows the understanding of the circulation and exchange mechanisms particular to water masses. It becomes a planning, identification of critical zones and choice of intervention mode tool in an emergency situation.

Its use within current models necessitates modest computational and processing capacities. Municipal authorities among others, could use it for fast and efficient management of the waters in their territories.

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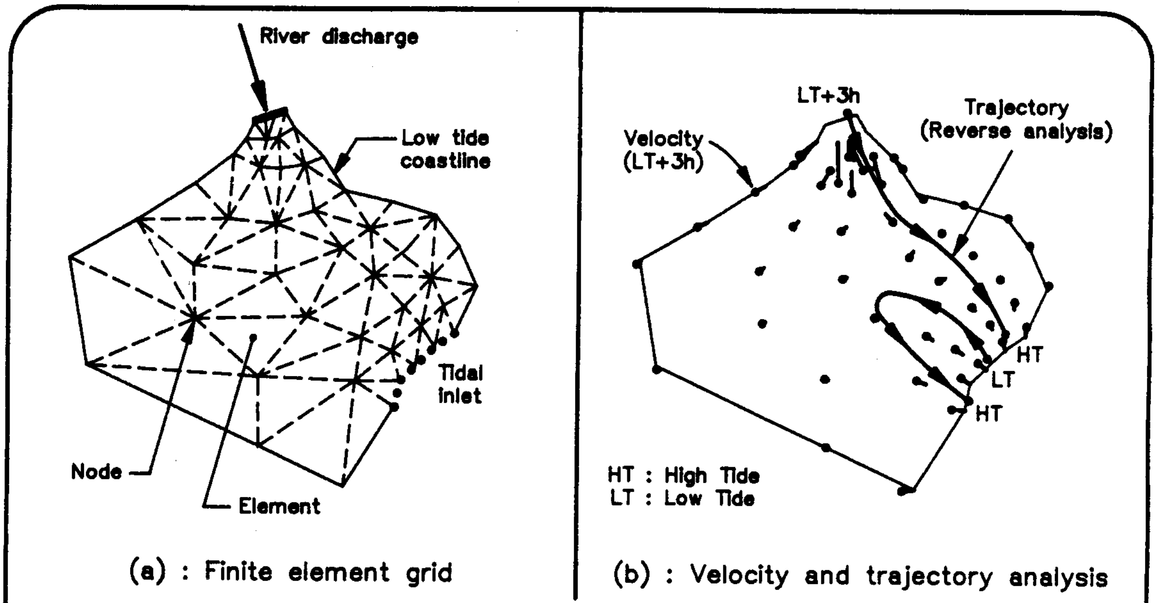


Figure 1: Graphical representation associated with the mathematical model

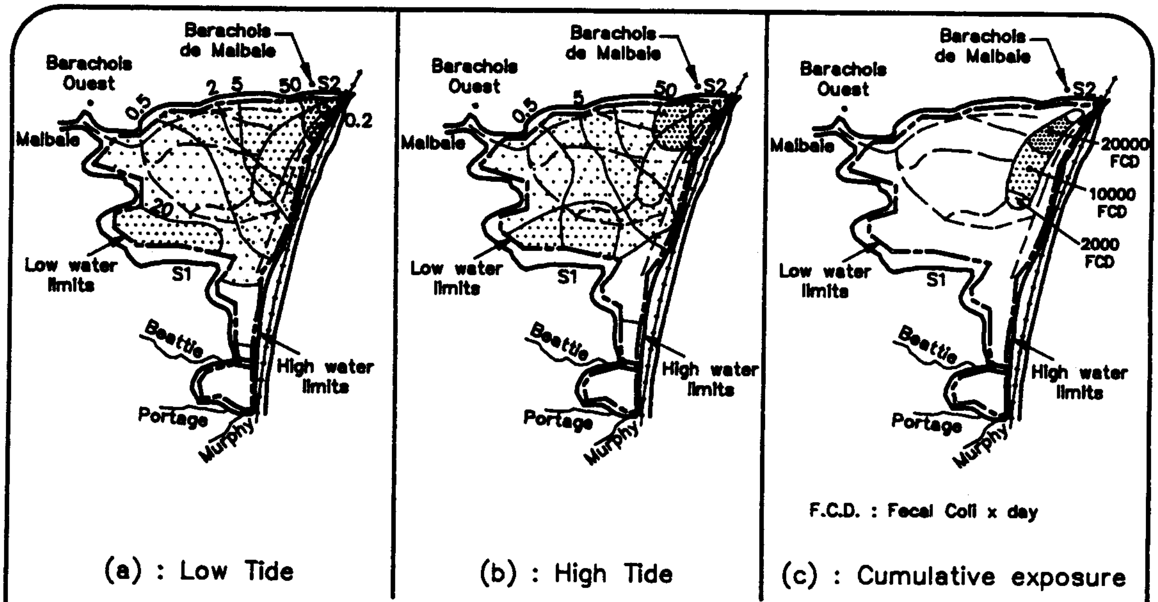


Figure 2: Example of contamination analysis

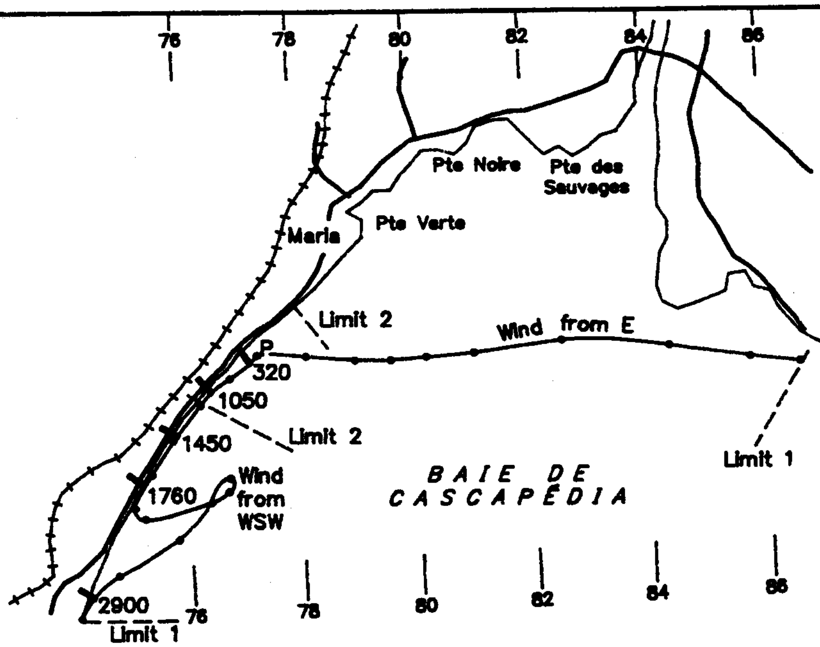
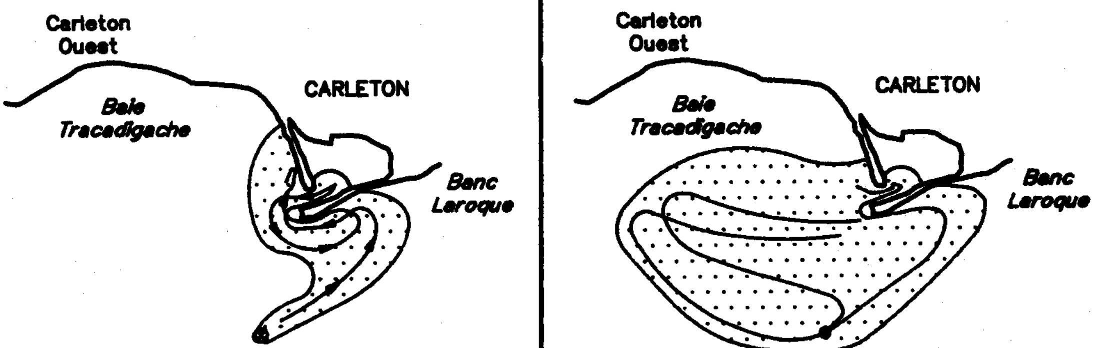


Figure 3: Delimitation of zone of suspected contaminant sources



(a) : Limited spill
Two tidal cycles

(b) : Continuous spill
One tidal cycle

Figure 4: Impact zone of a spill

Intermittent release of sewage disposal for ecosystem protection in the coastal zone.

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ABSTRACT

Some of the most productive ecosystems of the world oceans are found in the coastal zone. On the other hand, a large portion of the world population lives near the coastal zone. Traditionnaly, most industrial and urban effluents from these agglomerations have been discarded in this coastal zone. As a result, coastal ecosystems are now threatened, and careful management procedures are required to protect them from further deterioration. This study introduces the concept of intermittent effluent release in the coastal zone, as a function of the tidal mixing potential, in order to protect threatened nearby coastal ecosystems. The concept is applied to a case study, the Rimouski marine outfall diffuser, in the St. Lawrence estuary. Field measurements of currents, sea levels, and rhodamine tracer concentrations are used to calibrate and validate two-dimensional hydrodynamical and advection-dispersion numerical models of the MIKE21 system. The models are then used in worst cases scenarios to identify those tidal phase intervals during which effluents can be released without serious threat to a nearby coastal ecosystem. Finally, a real-time operational system is proposed to control effluent releases as a function of those tidal phase intervals.

KEY-WORDS : Coastal zone; environmental hydraulics; numerical modelling; coastal outfall diffuser; ecosystem protection; coastal hydrodynamics; rhodamine dispersion; operational oceanography; real-time control system.

1. INTRODUCTION

The coastal zone of the world oceans includes some of the most productive ecosystems on earth. It provides more than 90 % of the world's marine fish catch and other living resources associated with biological communities such as coral reefs, mangrove forests, and coastal marshes (Postma and Zijlstra, 1988). On the other hand, about 50 % of the world population lives within 60 km of the coastal zone, and traditionally, a large portion of urban or industrial effluents from these communities have been discharged without treatment into the coastal zone. This poses a serious threat to coastal ecosystems. In an effort to protect their coastal resources (fisheries, corals, beaches, etc.), several industrialized countries have imposed environmental laws which require some form of primary, secondary, or tertiary treatment to be applied to the effluents before their release into the coastal zone.

Normally, a sewage outfall diffuser is constructed at a coastal site where effluents can be well diluted or advected away from sites to be protected (ecosystems, marshes, reefs, etc.). This is verified by a site selection study where environmental constraints are optimized against economical ones, such as the offshore distance, or the depth of the underwater outfall pipe. The concept proposed in this work applies to those cases where a selected site must be rejected because of its adverse effects on local resources, or to the case of an existing diffuser which must be upgraded at high costs to meet environmental regulations. The concept is that of intermittent effluent release. In other words, it may still be possible, in some instances, to retain these sites provided ambient hydrodynamical conditions ensure adequate mixing of the effluents for an extended period of the day, every day.

Most coastal areas of the world oceans are subjected to the influence of tides. Tidal currents change intensity and direction throughout the tidal period, such that favourable mixing or advection conditions may exist during an interval of the tidal cycle (e.g. during flood or ebb phases). In addition, tides being deterministic in nature, they can be predicted with high accuracy; this then makes it possible to set up some kind of real-time control system at the site.

This paper describes the approach used to determine those tidal phase intervals during which effluents could possibly be released, and how their release can be controlled in real-time. A case study is presented for the city of Rimouski (figure 1), in eastern Canada. In this case, two environmental sensitive sites are to be protected from the effluents, namely an intertidal marsh visited by tourists for its rich fauna and flora, and an underwater intake for a nearby marine aquaculture station. The approach consists of long-term field measurements of sea-levels and currents, supplemented by fluorescent dye concentration measurements around the release site (section 2). These measurements are then used to extract tidal phases and amplitudes for the region, and to calibrate and validate a set of numerical models (section 3). Once operational, these models are then used to simulate the dispersion of sewage effluents under most adverse hydrodynamical conditions (section 4). These results should identify, if any, those tidal phase intervals during which effluent discharge can be activated. Tides being predictable, a control system can be set up in real-time (section 5).

2. FIELD OBSERVATIONS

The Rimouski coastal zone (figure 1) is characterized by semi-diurnal tidal oscillations whose amplitudes reach about 2 meters during spring tides, and by winds predominantly blowing alongshore from the south-west (Koutitonsky and Coté, 1993). The Rimouski urban sewages undergoe secondary treatment (organic material oxidation) in settling ponds prior to their continuous release, at a rate of about 40,000 m^3/s through the orifice of

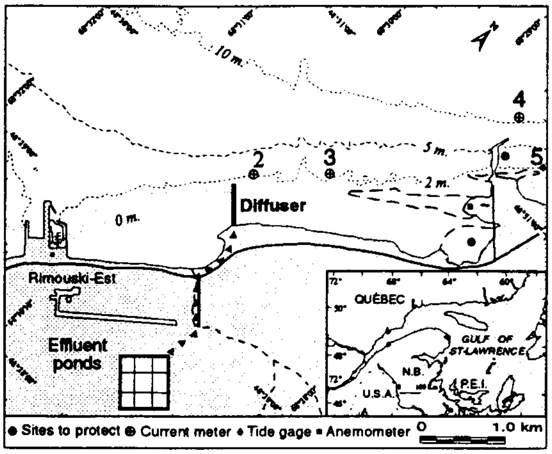

Figure 1: **Rimouski coastal zône, with positions of measurement stations, diffuser pipe, effluent ponds, and sites to be protected.**

orifice of a 0.5 m diameter underwater pipe. The orifice is located 500 m offshore, at a depth of 2 m below lowest low water. Aanderaa current meters and tide gages were moored at several locations (figure 1), for periods ranging from one to four months, starting July 1991. Time series analysis of the records revealed a coastal circulation dominated by tidal currents, superimposed on low-frequency wind-driven currents and a seasonal large-scale estuarine circulation (Koutitonsky and Coté, 1993). The principle axes of variability of the currents also revealed some spatial variability, in response to the local topography. Winds, sea levels, currents, temperatures and salinities measured near the diffuser outlet (station 2) are shown in figure 2, for a period including the day the dye dispersion experiment, October 30, 1991.

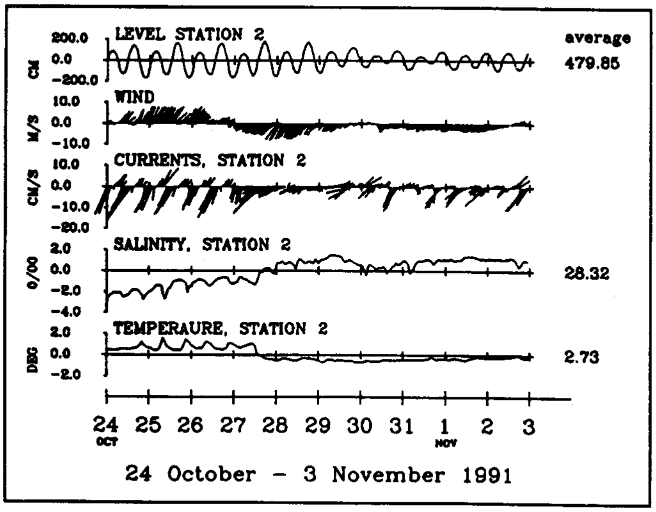

Figure 2: **Time series of sea levels, winds, currents, temperature and salinity at station 2.**

On that day, Rhodamine WT dye was injected continuously, at constant concentrations, into the effluents entering the outfall pipe at the ponds site. Dye concentrations at the offshore diffuser orifice, and across the dye plume, were measured from 8:00 to 16:30 hours, using simultanuous Trisponder positioning and running water fluorometry aboard a research vessel. The tracks followed by the vessel throughout the day, and the dye concentrations measured as a function of distance from the orifice are shown in figures 3a and 3b, respectively. Results indicated that dye dilutions of the order of 1/100 were generally reached within 2.5 km of the diffuser. Vertical profiles of salinity and temperature measured throughout the day also showed that the water column was well mixed. This supports the choice of horizontally two-dimensional numerical models for the hydrodynamic (HD) and advection-dispersion (AD) dye dispersion simulations.

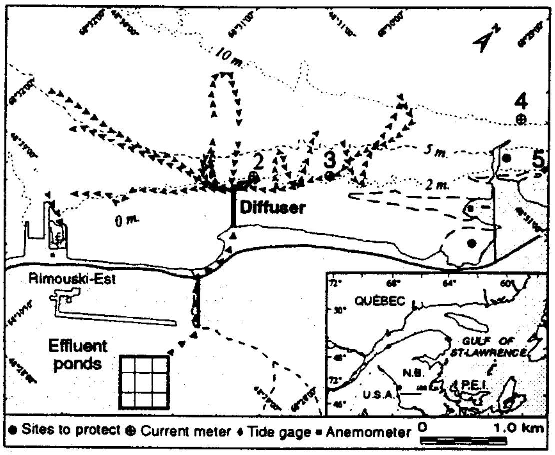

Figure 3a: Vessel tracks during Rhodamine dye concentration measurements on October 30, 1991.

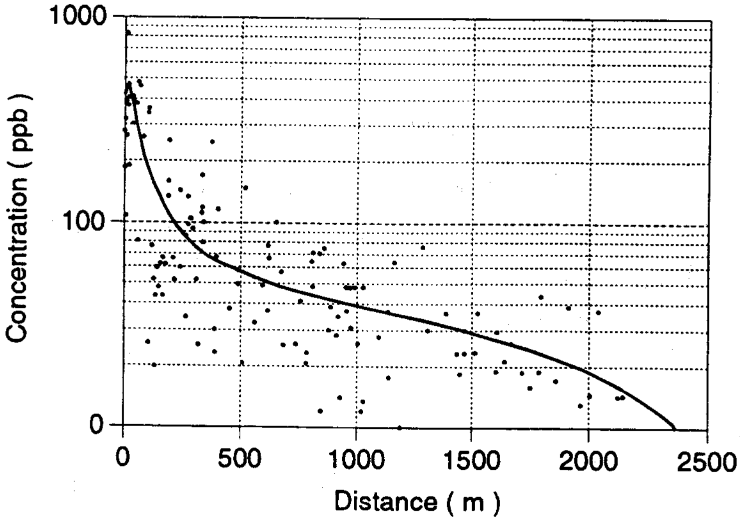

Figure 3b: Rhodamine concentrations (ppb) as a function of distance from the diffuser outlet.

3. NUMERICAL MODELLING

The HD and AD numerical models used are modules of the MIKE21 system developped by the Danish Hydraulic Institute (Warren and Bach, 1992). These are non-linear, time dependent, finite difference semi-implicit two-dimensional models, which allow for flooding and drying of grid cells located in the intertidal zone. The numerical grid of the region consists of 170 x 89 grid cells, each 100 m x 100 m in size. Models set-up, telescopic boundary conditions, calibration and validations have been described in details elsewhere (Koutitonsky, 1994; Gidas et al., 1995).

Results indicated that the HD and AD numerical models did reproduce sea levels and currents measured at the diffuser site (station 2, figure 4 a), and the dye concentration measurements (figure 4 b) on October 30, 1991, with good accuracy.

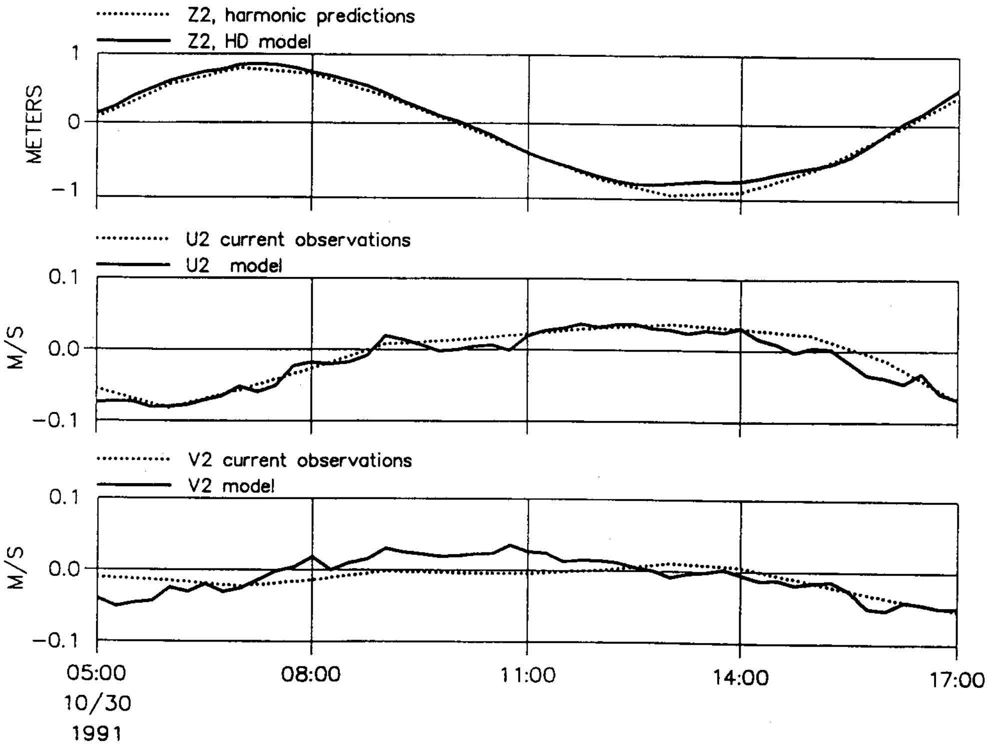

Figure 4a: Comparison between sea levels, alongshore (U) and cross-shore (V) currents measured at station 2 and computed by the HD model.

Results of the HD and AD simulations are shown in figure 5 at 8:00 hrs (left, high tide), and 14:00 hrs (right, low tide). Inspection of such results on an hourly basis revealed that, due to the local bathymetry, waters start leaving the salt marshes just after high tide (e.g. 8:00 hrs), such that the effluents released 2 to 3 hours before will not enter the marshes, but will be advected in a northeastward direction. At low tide, curents start flooding the marshes, and effluents released 2-3 hours earlier would start entering the marshes at that time. Therefore, effluents should be released from flood tides (e.g. 5:00 hrs) to ebb tides (e.g 11:00 hrs) in order not to enter the marshes. This local hydrodynamical feature will be instrumental in selecting the intermittent release interval.

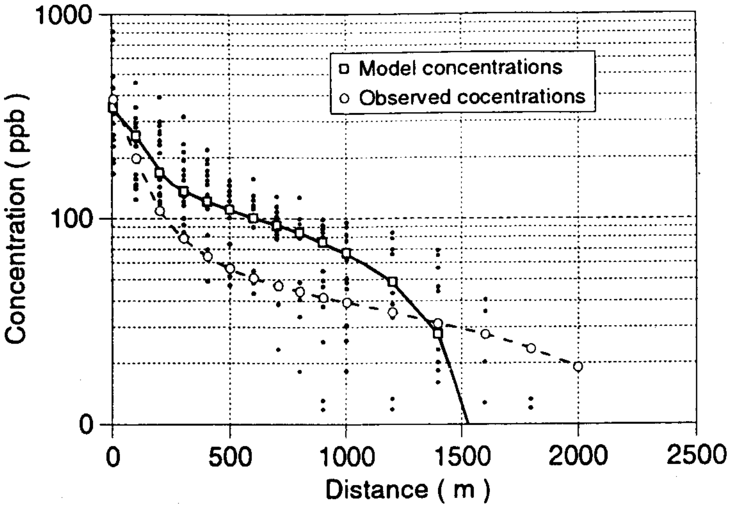

Figure 4b: **Comparison between Rhodamine WT dye concentrations measured on October 30, 1991, and the maximum concentrations simulated by the model around the diffuser outlet during the same period.**

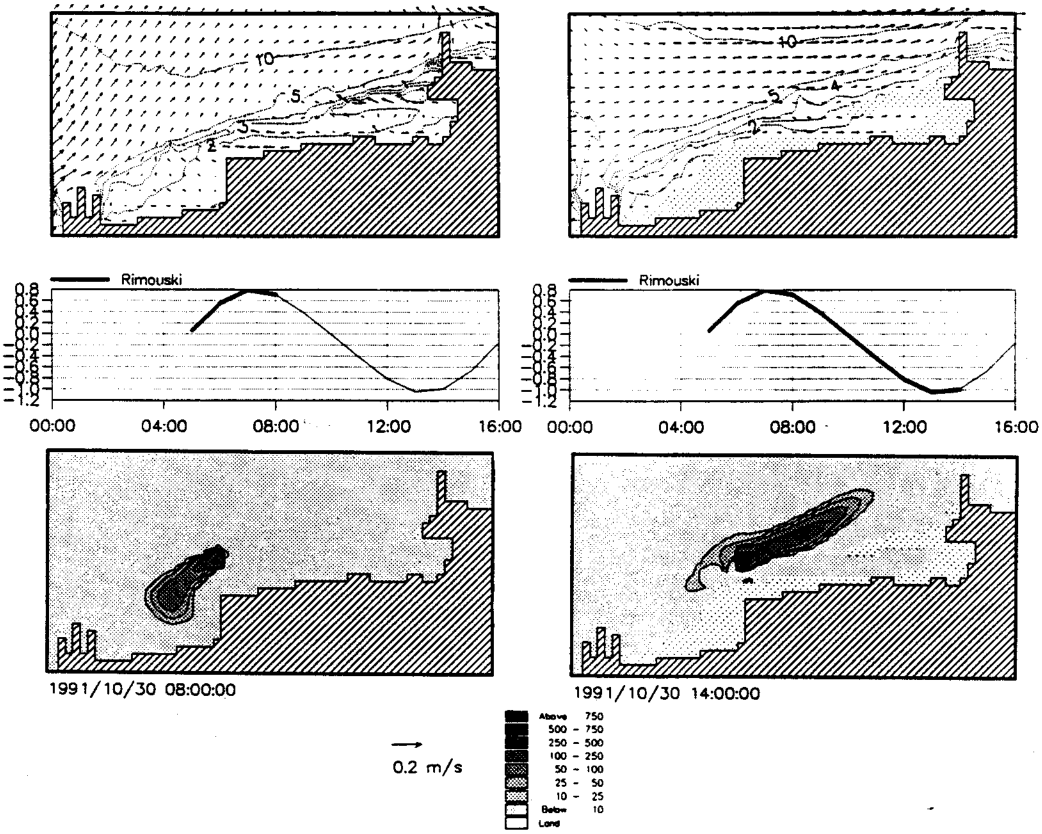

Figure 5: **Results from the hydrodynamical (top) and advection-dispersion (bottom) models, at 8:00 hrs (left) and 14:00 hrs (right), with sea levels shown in between, for October 30, 1991.**

4. CONTINUOUS VERSUS INTERMITTENT RELEASE

Once validated, numerical models can be used to simulate worst case scenarios. One such scenario is currents are strongest and directed towards the sites to be protected. That should occur during spring tides (October 25-26, 1991, figure 3a), under strong winds (e.g. 10 m/s) blowing towards the marshes. Results of AD numerical simulations under these conditions, for a continuous effluent release, are compared in figure 6 to those for effluents released intermittently, from flood to ebb tidal phases (hatched periods on tidal signal in figure 6). The simulations, shown at three hours intervals starting at 6:00 hrs, clearly show that when released continuously, the effluents reach the salt marshes and remain there for some time, whereas when they are released intermittently, they always by-pass the marshes and are considerably diluted as they reach the aquaculture water intake. The marsh ecosystem is thus protected.

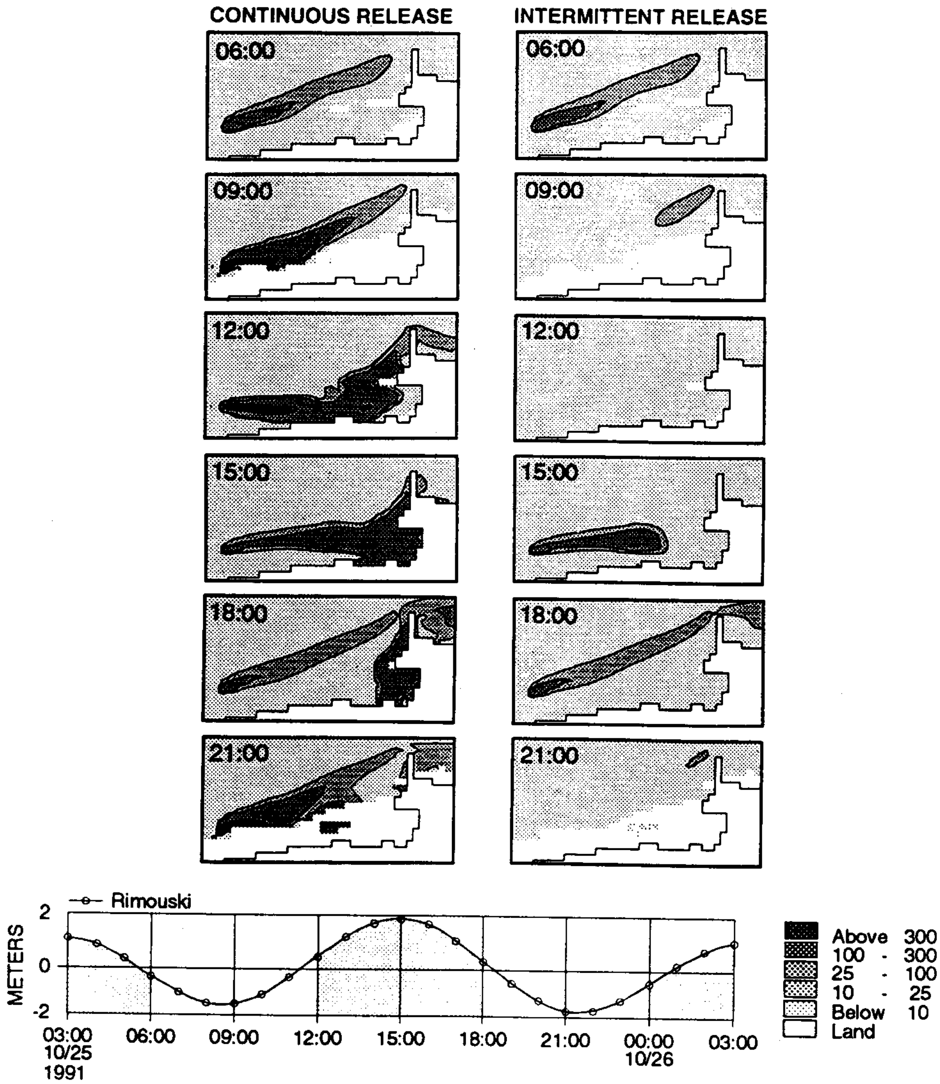

Figure 6: Advection-dispersion of effluents released continuously (left) and intermittently (right), as a function of tides (below). Simulations are shown at 3 hrs intervals, starting at 6:00 hrs (top).

5. REAL-TIME CONTROL SYSTEM

Being deterministic in nature, tides are the most predictable feature of the ocean's dynamics. However, the prediction of tides for a particular location is normally performed once the harmonic constituents of the tides at that location are determined. These are obtained by harmonic analysis (Foreman, 1977) of sea levels measured at that location for an extended period of time. Once available, these constituents can be stored in a microcomputer. A tidal predictor software (e.g. Foreman 1977) can be set up in the computer to predict the tides in real-time for that location using the computer clock. A sketch of this real-time control system is presented in figure 7.

Having determined in section 4 that most favourable conditions for the release of effluents in the Rimouski coastal zone are from flooding tides to ebbing tides, a software can be set up in the computer (A in figure 7) which would issue to the parallel port an "on" analog signal as tides reach their flood phase, and an "off" signal as tides reach their ebb phase. This signal then becomes an input signal to some electrical control system (B), which in turn opens or closes the release gate (C) at the effluent pond (D), respectively allowing or preventing the effluents to enter the diffuser pipe (E).

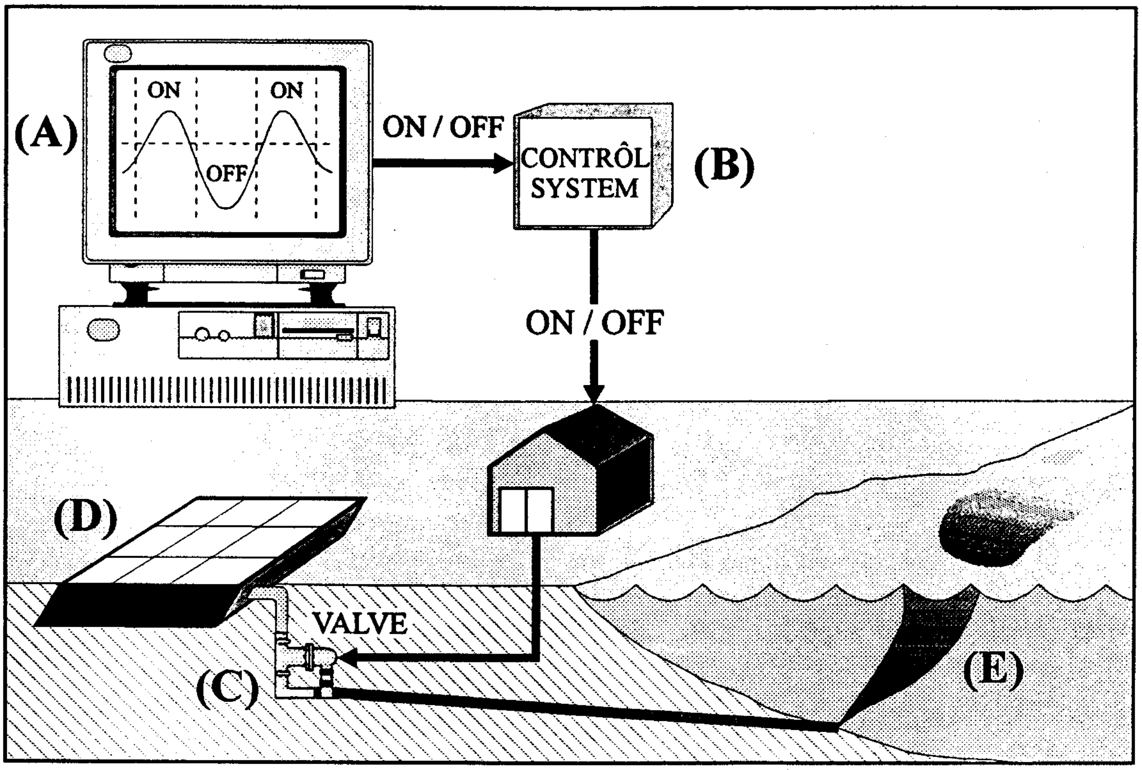

Figure 7: Skech of a real-time control system for sewage disposal, with a microcomputer (A) running a tidal-predictor software, feeding a control box (B) which sends the ON/OFF signal to a valve gate (C) at the ponds outlet (D), which in turn controls the release of effluents into the diffuser pipe (E).

6. DISCUSSION

Many factors must be considered when developing an intermittent effluent release control system. Ponds must be large enough to contain those effluents which are being retained when the release is suspended. The outfall diffuser pipe diameter must also be larger to allow for increased discharge rates during the effluents release period. A back-up electrical generator must be available to provide power to the computer and the release gate mechanism in case of power shortage.

In some instances, hydrodynamical factors other than tides could be used to control the release. One example are trade winds in some equatorial coastal areas (e.g. Venezuela and Colombia) which intensify in the morning and weaken at the end of the day. In all cases however, a detailed hydrodynamial field experiment, coupled to careful numerical modeling of effluents dispersion are definitly required to set-up the real-time control of the effluents in the coastal zone, when possible.

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A DECISION SUPPORT SYSTEM FOR MANAGING AN URBAN AQUATIC ECOSYSTEM

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ABSTRACT

Increasingly the effective management of water resources in an urban environment requires an ecosystem approach. The information provided to the manager must not only describe the individual systems components and processes, but also determine their inter-relationships in a comprehensive format. To satisfy these management needs, a decision support system is required that integrates the following components: data acquisition, storage and management; information interpretation; multi-media information display; a predictive model to assess the potential impact of their decisions and provide option evaluation techniques. The Regional Municipality of Ottawa-Carleton (RMOC) and Environment Canada have combined resources to develop and implement an integrated environmental monitoring program, along with a decision support system to assess the management options for a portion of the Rideau River Watershed within the municipality's jurisdiction.

In Phase I, GIS data processing was used with raw survey data to create station location maps, contour maps, and drogue tracking maps. GIS data interpretation was used to interpolate water quality parameters in a spatial and temporal context. Finally, the multi-media reporting abilities of GIS were incorporated into this study. Phase II will incorporate neural networks to aid in data interpretation and hydrodynamic models for process modeling and decision formulation. The GIS will have a role in the decision testing module. The models that are being developed will be linked to the GIS for spatial representation and analyses purposes. Also, improved data communication between system components will be developed by automating interface functions. Phase III will focus on the development of the decision support component of the system which will incorporate socio-economic information, expert systems to assist planning, information access and decision making and, improve accessibility to the system through enhancements to the human interface.

KEY TERMS: municipal water resource management; decision support system; ecosystem management; integrated data management; urban water environment; Rideau River.

THE MOONEY'S BAY PROJECT

In recent years, Regional Municipality of Ottawa-Carleton's municipal water resource managers have witnessed an increase in the need for more complete and comprehensive environmental information. Interest in water quality has evolved from the singular assessment of human health risks from bacteriological contamination at a public beach, to the evaluation of the inter-relationships and effects of a complete spectrum of urban-generated contaminants on the aquatic ecosystem. This requires the integration of temporal and spatial data. In addition, the data are from several sources and in various formats, further straining the existing data management capabilities.

However, the public expects managers to have this information and make decisions crucial to the aquatic ecosystem as if it were any other piece of municipal infrastructure. Gradually managers are having to incorporate policies of sustainable development and an ecosystem analyses into municipal decision and planning processes. The development of a decision support system was initiated to assist managers in making informed decisions in their area of jurisdiction with Mooney's Bay as the primary geographic focus of this development.

The main objectives of the Mooney's Bay Project are as follows:

1) Develop an aquatic ecosystem approach to monitoring
2) Design a relational database to manage discrete and time series data;
3) Apply the most current analytical tools for data processing, integration and interpretation;
4) Develop an understanding of the dynamics of the aquatic ecosystem;
5) Provide advice and present strategies for managing the ecosystem that will ensure its integrity; and,
6) Incorporate all of the above elements into a decision support system to effectively manage the aquatic ecosystem.

Background

The Rideau River watershed drains a portion of Eastern Ontario (3,830 km^2). During the summer months, daily average discharges are typically in the 10-20 m^3/s range. Within the watershed, the land use is quite diverse and includes: agricultural, residential, commercial, industrial and recreational uses. The 200 km long river is managed as one continuous system in order to meet the diverse needs of navigation, municipal water supply, sewage treatment, hydroelectric generation, recreational, wildlife habitat and flood control.

Within the Rideau River watershed, Mooney's Bay is one of the prime riverine urban recreational areas located in Canada's National Capital Region.

As an initial response to beach closures in the 1970's, two hydraulic pumps were installed to circulate water across the beach to improve the water quality. Since then, monitoring efforts have focused on bacterial levels and establishing a protocol for determining beach closure (RMOC, 1983; OMOE, 1984; OCHD, 1991). Considerable effort was placed on identifying all potential sources of bacterial contamination and initiating mitigative measures, with particular emphasis given to stormwater drainage systems. The estimated costs of improving the bacterial water quality, through the upgrade of the infrastructure, was in the order of $20 million (RMOC, 1992).

Overview

Prior to 1993, data acquisition in the Mooney's Bay area had concentrated on providing bacteriological and chemical information on the water quality. This included data from time-series as well as discrete samples. However, data describing the hydrodynamics of the Bay were insufficient to determine the source, transport and fate of contaminants within the Bay to any degree of confidence. It became apparent that due to the complexity of the system a much broader study was required to properly formulate management decisions and justify any large remedial expenditures (RMOC, 1993).

In 1993, the Regional Municipality of Ottawa-Carleton and Environment Canada entered into a partnership to address the broader issue of aquatic integrity in the Rideau River, and in particular Mooney's Bay. An environmental monitoring program was designed and the results incorporated into a prototype decision support system, of which data acquisition and management is a key component.

An ecosystem management approach requires that information be available which describes the physical (both static and dynamic) characteristics of the system, the quality of the water within and entering the system, and the biological state of the system. Since the individual parameters for each of these components vary in time and space, the data must be collected at frequencies and locations that reflect these changes.

Data collection programs conducted through the summers of 1993, 1994 and 1995 consisted of the following continuous time series measurements:

- flow velocities, volumes and water levels using an AFFRA acoustic flow meter;
- water levels and velocities (magnitude and direction), for future model calibration, using AFFRA's;
- storm sewer outfall flow volumes obtained using ISCO flow meters and stage triggered sampling;
- water quality parameters from Hydrolab multi-probe sensors
- wind speed/direction and precipitation using RM Young and RIMCO instrumentation.

The data from the above mentioned sensors were recorded on site by data loggers and transmitted automatically, for immediate use into a project relational database, via telephone or satellite. In addition, discrete water quality measurements and samples were obtained to provide detailed spatial information. These samples were analyzed for nutrient, bacteria and trace metal content. At selected sites analysis included phytoplankton and zooplankton biomass and species identification. Fish habitat and populations were also assessed.

ENVIRONMENTAL DATA MANAGEMENT AND DECISION SUPPORT SYSTEM

A multi-disciplinary approach implies studying entire systems to understand the complex interactions. Recently, hydroinformatics has been defined as the application of advanced information technology to the solution of problems in the field of hydraulics and related areas. Advances in data capture, simulation, knowledge-based and computer-related technologies have and continue to produce an impressive array of new tools used by many disciplines to understand and solve environmental problems. An integrated generic framework for hydroinformatics currently under development by Environment Canada and the Nation Research Council Canada will be used for environmental data management and decision support. One of the applications that it will be configured to support is *Urban River Management*.

The primary consideration in the design of the decision support system is that it be generic, in that it should be independent of any specific management environment. To meet this criterion, the hydroinformatics system (HYFO) must:

- automatically acquire data from all pertinent sources, convert them to standard format(s) and, store them in appropriate data tables;
- integrate and interpret the available information through the interactive use of analytical tools; and,

- test and display results of possible decision scenarios through the interactive use of predictive models and multi-media reporting tools.

Figure 1 illustrates the prototype system. There is a high reliance on commercially available software to provide the desired system's functions. This development approach serves two purposes. First, it allows the system to be used almost immediately and to be expanded as required. Second, new or more efficient modules can be easily incorporated into the system as they become available.

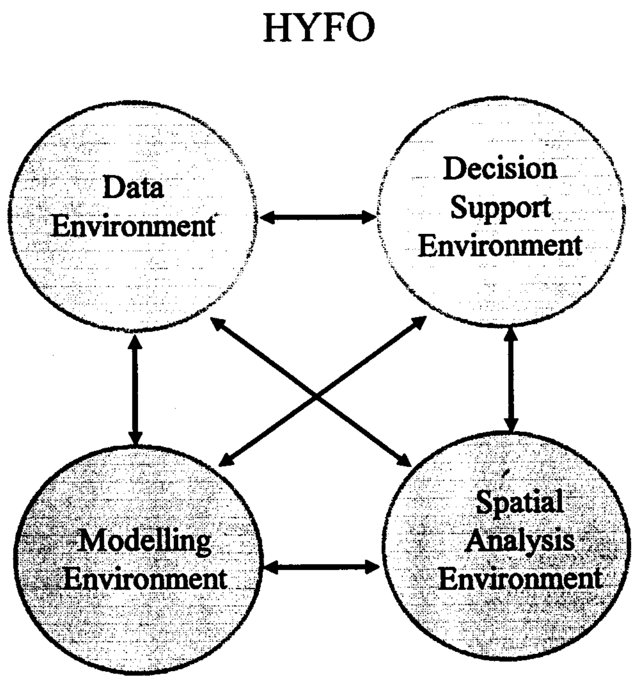

Figure 1 Components of a Hydroinformatics System

The effective management of an ecosystem requires that decisions are able to be made within the response time of the system. This requires accurate processed information from existing archival databases in conjunction with the analysis of data from real time data acquisition networks. The Mooney's Bay project provides the necessary data sources for the HYFO application.

An archival relational database was developed to store water quality and quantity data in standard formats. This allowed for the integration of the flat files provided by the various field instruments and the discrete water quality data. The information could then be efficiently extracted to the project database to be combined with additional real time data inputs or linked to process components. A critical requirement is the ability to easily move information into or out of the system as well as between system components. This capability is provided through interface drivers, a combination of commercially available and custom built software packages.

The process components, from data processing through to reporting, contain the tools to transform data into information for decision making. The nature of these tools depends on analytical needs of the ecosystem in question and can include the range of applications from commercially available statistical software packages to numerical models, neural networks and expert systems.

A spatial analysis component (i.e. GIS) has a role in the data processing, data interpretation, decision testing, and multi-media reporting components. In Phase I, GIS data processing was used with raw survey data to create station location maps, contour maps, and drogue tracking maps. GIS data interpretation was used to interpolate water quality parameters in a spatial and temporal context. Some of the commonly used planning databases such as ; land use, vegetation, soils and groundwater were included. Finally, the multi-media reporting abilities of GIS were incorporated into this study as an essential tool to communicate the information to the public and decision makers.

The HYFO concepts described above, were implemented in the Phase I development of the support system to be used in the assessment and ultimate management of Mooney's Bay. Phase II will incorporate neural networks to aid in data interpretation and hydrodynamic models for process modeling and decision formulation. The GIS will have a role in the decision testing module. The models that are being developed will be linked to the GIS for spatial representation and analyses purposes. Also, improved data communication between system components will be developed by automating interface functions. Phase III will focus on the development of the application support component of the system which will incorporate socio-economic information, expert systems to assist planning, information access and decision making and, improve accessibility to the system through enhancements to the human interface.

PRELIMINARY INTEGRATION OF AQUATIC ECOSYSTEM INFORMATION

The major components of an aquatic ecosystem as defined in this study are: physical , static and (hydro)dynamic, water quality and biology.

Physical-Static and Hydrodynamic

The key physical element is the bathymetry of the Bay which affects flow velocities and circulation patterns. The hydrographic information is useful for the selection of sampling sites and is fundamental to understanding thermocline development. The Bay has relatively steep side slopes, a large sand shoal in the middle of the southern section, and a deep trench (~11 m) in the section close to Hog's Back Falls.

Flow rates, wind speed and wind direction, in addition to the physical setting all play an important role in establishing the overall circulation patterns in the Bay. Acoustic flow meters were installed at four locations within the Bay to provide time series measurements of flow velocity and direction. These data inputs, along with the wind velocities and direction, were mapped and then animated to show the conditions in the Bay for the entire summer. It was then possible to determine when significant hydrologic events were occurring and the impact that they had on circulation patterns within the Bay.

To supplement the AFFRA and meteorological data, drogue tracking was performed in the Bay under specific wind conditions. These three data types were integrated and used to map the general flow patterns for each day. The water temperature stratification indicates that a thermocline typically develops by mid - summer. In terms of the hydrodynamics, the deeper colder water remains relatively undisturbed and does not readily mix with the warmer water above the thermocline.

In Phase II of this project, a three-dimensional hydrodynamic model will be calibrated with these data sets. The model will provide a complete picture of the circulation patterns under varying inflow conditions and stages of thermocline development.

Water Quality

Once the hydrodynamics have been documented, information can be generated to provide an understanding of key chemical and biological processes.

An oxygen content of > 5 mg/L is required to support aquatic life. In the Bay, the < 5 mg/L zone increases marginally over the summer, while the anoxic zone found below the thermocline (at depth > 6 m) expands considerably to cover one third of the Bay area by late August. With the onset of cooler temperatures, during the nights, the thermocline breaks up by late summer and the anoxic zone is easily dispersed by an increase in river flow.

The phosphorus measurements in the Bay typically exceed the Provincial Water Quality Objective of 0.03 mg/L. The highest reactive and total phosphorous levels are found in the anoxic zones. For example, in the southwest area of the Bay, which generally is anoxic the entire summer, the average reactive phosphorous level was 0.35 mg/L and total phosphorous level was 0.41 mg/L. These levels are approximately an order of magnitude higher than those measured near the surface. The deep zones had low Redox Potential (~100 mV) with the northern deep trench getting as low as -200 mV. Under these conditions, internal phosphorus loading may be contributing to the high phosphorous levels in the Bay. This is a major concern since elevated phosphorus levels promote heavy weed growth, algae blooms and reduced clarity.

Figure 2 depicts the extent of anoxic zone development during the latter half of July, 1994. The darker area of the bay represents the depth at which anoxic conditions exist. The square platelets represent the total phosphorus results for the discrete samples taken at two metre depth intervals. The graphical presentation at the top of the figure represents the time series data of river inflow in cubic metres per second and rainfall (bars) in millimetres. The figure is a snapshot of an animated presentation of 1994 data. The exact time of the snapshot is provided by the leading edge of the shaded area of the graph.

Figure 3 is a similar rendition of data on E. coli concentrations. Since E. coli are normally transported to the Bay via local storm sewers, the storm sewer flow volumes in cubic metres per day for the two largest sewers are shown as a time series in the graph. Note that one of the storm sewer outfalls (depicted as pyramids on the southern edge of the Bay) is discharging concentrations of E. coli >8000 counts per decilitre immediately after an intense rainfall event. The threshold for E. coli concentrations in the beach area which would force beach closure is equal to 100 counts per decilitre. This animation of data has proven useful for the integration and interpretation of the data as well as an effective tool to communicate information to decision makers.

Biology

In 1995 the spatial and temporal delineation of algae (phytoplankton) communities was evaluated through chlorophyll-a analysis. At the same time, integrated samples were collected and preserved for phytoplankton identification and density. This component of the study extended from May to October 1995 covering three phytoplankton blooms and two major summer rain events. Preliminary chlorophyll information, an indicator of productivity and biomass, indicate higher concentrations in Mooney's Bay than other sections of the river just upstream. The concentrations of Chlorophyll-a were found to be positively related to the total phosphorous levels.

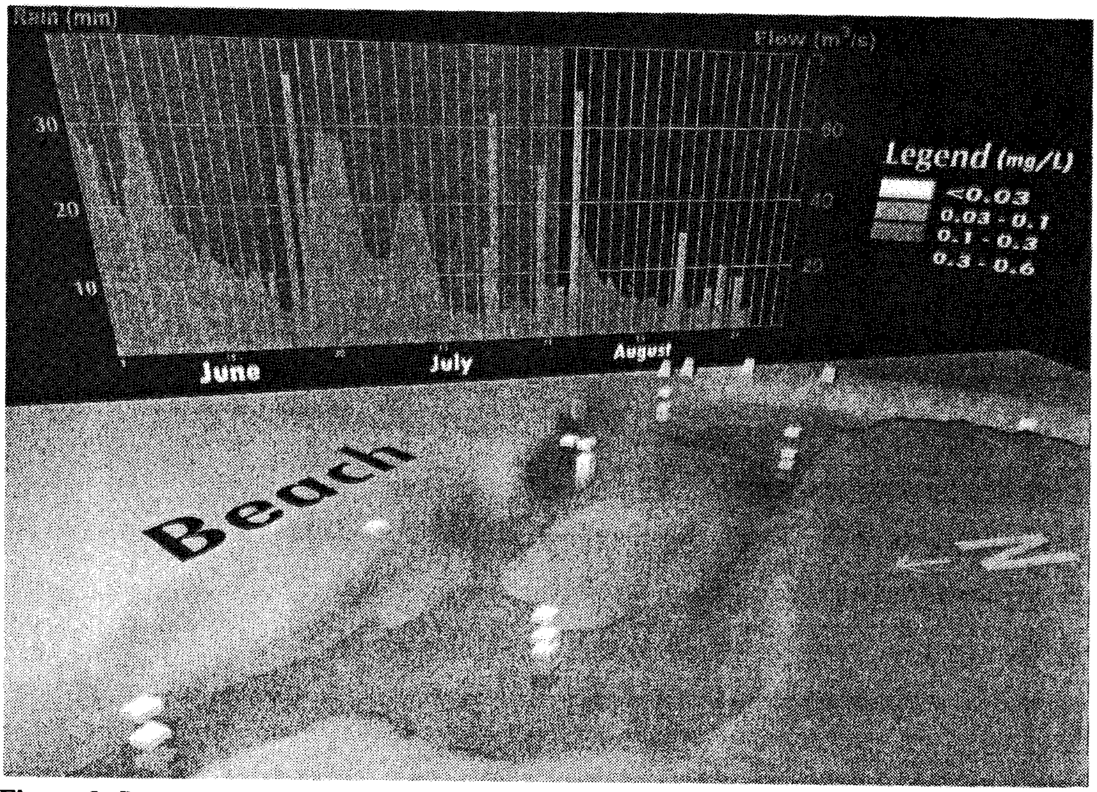

Figure 2: Interrelationships between Flow, Rainfall, Dissolved Oxygen & Phosphorous

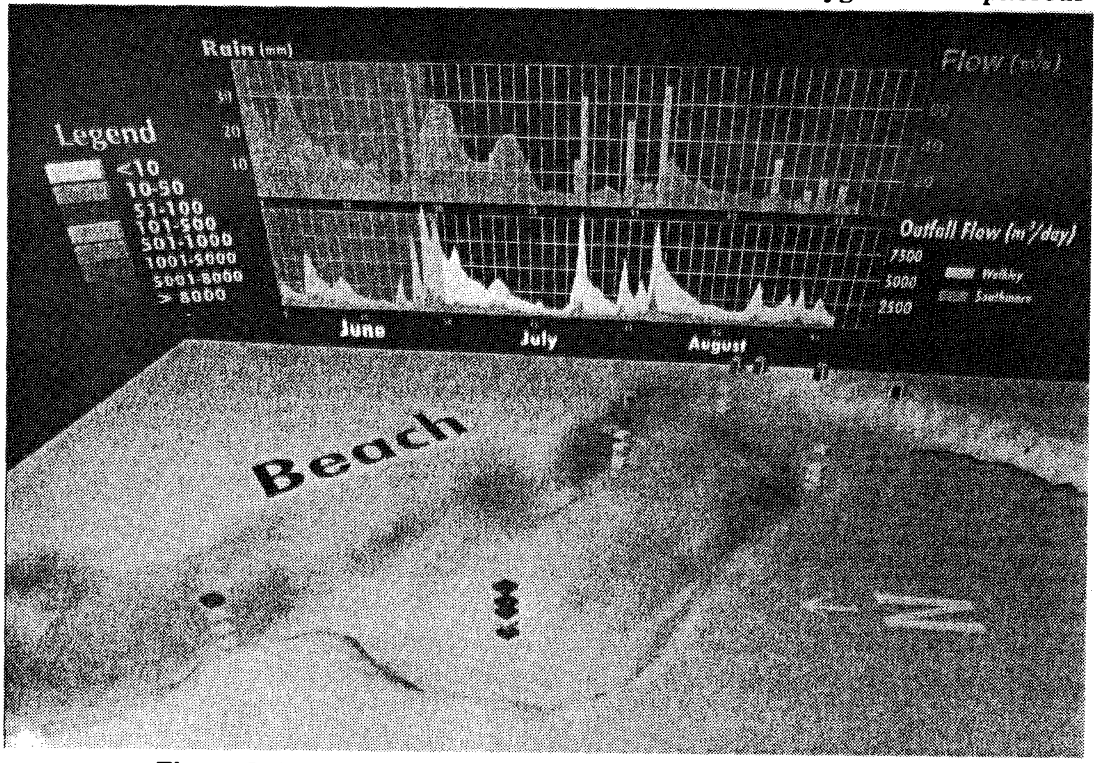

Figure 3: Interrelationships between Flow, Rainfall & E. Coli

As well, in partnership with the University of Ottawa and the Museum of Nature the evaluation of the phytoplankton communities showed a high diversity of species within Mooney's Bay. Many species were typical of lakes or slow moving waters. However, in other areas of the Rideau where the flow is faster, the reduction in the algae was reflected in the lower chlorophyll a analytical results.

The purpose of the seasonal investigation is to establish the reproducibility potential of the phytoplankton seasonal patterns and develop a simple model for changes in Mooney's Bay as a result of the exponential growth of Zebra Mussels. Biomass levels were very high in 1995(up to 6000 mg/m^3) during bloom periods dominated by large diatoms. Throughout the summer season biomass levels were comparable to 1994 (2200 mg/m^3). The early spring blooms associated with increasing water levels were dominated by "stick " diatoms. After the spring blooms a summer community of a cryptophyte/centric diatom/chlorophyte developed. A major rain event in August disrupted the summer development of the phytoplankton and triggered an early fall bloom. The flora was similar to that observed in 1994. Spans GIS mapping was used to show the spatial & temporal changes in response to the physical and seasonal factors (Figure 4).

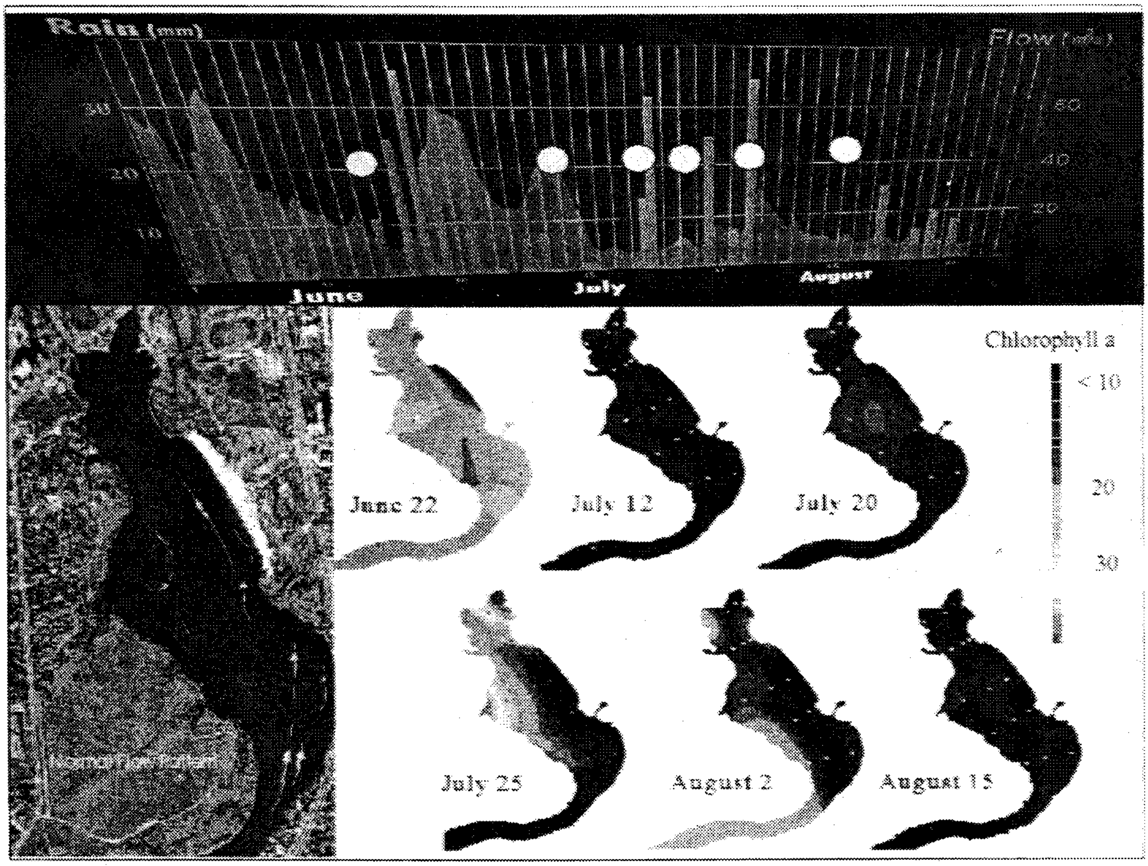

Figure 4 Spatial and temporal changes in Chlorophyll a

Thirteen distinct local rain events occurred from May 1st to August 15 th 1995. During the rain events the benthic filamentous blue green algae are disrupted and displaced into the water column. The removal of these benthic forms appears to range from 3-11 days depending on the severity of the storm. This was an important observation due to the increase in Chlorophyll-a and total phosphorus (TP) levels in the water column. To date increases in TP during rain events have only been attributed to the various inputs to the river.

The visual impact of algae, specifically algal mats along the shoreline created an interest in the nearshore algal growth. The littoral communities are diverse. The three major floating mat problems were represented by Spirogyra spp., Cladophora spp. and water -net. Water-net is the most significant contributor to the floating mats. Knowledge of the algal species now allows for development of control strategies to be considered.

Zooplankton samples were collected at each of the phytoplankton sites. Each sample was analyzed for total biomass(dry weight) and species identification. Their critical role in the ecosystem may be threatened by stormwater inputs, increased river flows or by the filter feeding of zebra mussels. Initial results show that the species change over time throughout the Bay.

Parts of the Rideau River have been recognized for their high recreational fishing value. However, very little information was documented on the fish communities in the Mooney's Bay reach. Thus, in 1995 the SWQB in conjunction with the Ontario Ministry of Natural Resources initiated an assessment of this part of the river. Detailed information was collected on the abundance and size of the fish species present. Fish habitat and substrate mapping has been carried out to evaluate the potential for fish habitat improvement. This will enable the RMOC to recommend sound management practices to protect or enhance this resource. Local areas of significance for young of the year refuge and spawning have been identified and have thus altered one of the preliminary proposed stormwater options.

The RMOC, in conjunction with the Museum of Nature has developed a monitoring programme to estimate the colonization and growth of zebra mussel in the Rideau River. Since last year their population has more than doubled on the Rideau. The potential for changes to occur could be quite significant if the zebra mussel population grows to the densities it has in Lake Erie. If these densities are realized in the Rideau we should be prepared to see changes in the biological community structure and changes in several water quality parameters. In some areas of the river the zebra mussels are affecting the chlorophyll a concentrations and increases in soluble reactive phosphorous have been observed. Fortunately, the Region has collected enough water quality and biological data prior to the accelerated zebra mussel population growth. This will assist in understanding the impact of the zebra mussel invasion on the Rideau River.

The most significant physical factor impacting on the lower Rideau River is the 3m lowering of the water level every fall and fill-up every spring. This has an engineering function of reducing the potential impact of ice damage on some of the locks. Biologically, the drawdown has disrupted the littoral vegetation and increases the shoreline instability. Increased sedimentation has a negative impact on the benthic organisms and native clams. Rapid draw-down traps many young of the year fish along the shoreline where they die. One positive impact of draw-down is the possible impact on the zebra mussel populations left exposed during the winter. Over the next few years the question of draw-down will receive more attention with ecosystem stability as the primary concern.

The source, fate and effect of various contaminants is another key aspect of this project. The importance of the storm sewer inputs into the Bay relative to the upstream river input is critical when considering remedial action. The 1994 and 1995 field programs focused on monitoring a number of significant storm events, mapping stormwater fate and determining its impact on the Bay. Once the hydrodynamic model for the Bay has been calibrated, it will be used to model the stormwater outfalls under varying conditions and predict their impact on different areas of the Bay. The modeling, when combined with real-time monitoring capabilities, will allow for a better assessment to be made on the impact of contaminants on the aquatic ecosystem.

MANAGEMENT CONCERNS

The complexity of processes affecting this aquatic ecosystem provides a range of problems to be addressed and many possible management options to be evaluated. These options need to be practical, supportable, and provide an economic benefit. It is no longer acceptable to use large capital expenditures to provide partial solutions. However, consideration must be given to the extensive effort that has been spent to date in developing pollution control options balanced with the expanding public, political, environmental and infrastructure pressures.

In the case of the RMOC, it cost the public $400 K to develop a remedial action strategy that solely responded to bacterial contamination . In 1995 dollars this singular remedial strategy is estimated to cost $60M. Senior management is now deciding whether to modify the existing strategy to respect the various concerns, or evaluate a new matrix of options with an understanding of the true problems.

Creativity shall become the operative word. The public's sensitivity to what is perceived as another "expensive" study, will be a difficult hurdle to overcome. However, a reliable decision support system allows managers to evaluate the cost/benefit of alternate strategies and effectively communicate these results to the public and decision makers. In any case, the management of this ecosystem is potentially feasible and with a functional monitoring network and decision support system, informed decisions can be made and their effects monitored.

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CHEMICAL AND BIOTIC INDICES IN RIVER ENVIRONMENT QUALITY ASSESSMENT. THE CASE OF BELICE RIVER (WESTERN SICILY)

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ABSTRACT

On the basis of field data collected during some surveys in lower River Belice, several *quality indices (QI)* were applied to assess both the quality of river water itself and of the river environment. The values taken by the indices were then compared, and the differences explained. A method was also proposed for replacing missing data.

KEY-WORDS: River Environment/ Water Quality Index/ Biotic Index/ River Belice/ Sicily.

INTRODUCTION

The Physical Environment

Belice River flows into Mediterranean Sea after its course southward, through western Sicily. Its watershed, 964 km^2 wide, is the fourth major in this Island; the annual mean rainfall is below 600 mm. Only a small part of runoff reaches the mouth, because two reservoirs and several diversion dams store river water, and supply irrigation and drinking water systems with it. At the time this research was carried out (Winter 1993 - Spring 1994), December and January were almost dry, whereas November before and (markedly) February were rainy.

The lower Belice was largely modified in the Sixties to middle Seventies by regulation works, such as embankments, checkdams and shortcuts. Going up the river from the mouth, reaches are found with trapezoidal-shape main section (with or without side sections), 60 m wide overall, diminishing by step to 30 m. Straight trenches are built in earth; bends in gabions. Several checkdams, 1 m high, have reduced and leveled natural bed slopes, so slowing down the stream, and making suspended solids settle as sandy or silty bed. Regrading and alignment were often carried out together.

Aims and Methods of the Present Research

During the development of a broader study of this river basin, three surveys were carried out, namely, in Winter 1993, and in Spring 1994. A considerable amount of phisico-chemical, bacteriological and bionomic data were collected; the complete set consists of 224 data for DO, BOD, T, NH_4, NO_3, PO_4, pH, Electric Conductivity (EC), Turbidity, Total and Faecal Coliforms, and Faecal Streptococci, plus abundance and diversity of benthic macroinvertebrate population (Spring only). Turbidity was actually replaced with more reliable TSS; TDS (mg/l) were calculated as 0.7 EC (μS/cm).

This research work was aimed at comparing strict water quality assessment with the broader assessment of river environment quality, based on living species found (actually, benthic macroinvertebrates). These are known to depend on the structure of river bed and flow velocity, as well as on water composition and temperature.

In the lower Belice 7 sampling sites (marked C_1, ...C_7) were set for the physico-chemical survey (Fig. 1). Three of them (marked as couples from S_1 - S_2 to S_5 - S_6) were selected also for benthic macro-invertebrates collection. Field data *(quality indicators)* collected in these 7 stations enabled the writers to apply several *quality indices (QI)* in order to assess both the quality of river water itself and of the river environment. In detail, stations C_1 and C_2 were set downstream the discharge of treated wastewaters from the small towns Poggioreale and Salaparuta; C_3 and C_4 downstream raw wastewaters outfalls from Montevago and Partanna, respectively. The remaining ones were conveniently distributed over the river length. All stations' sites are shown in Figure 1.

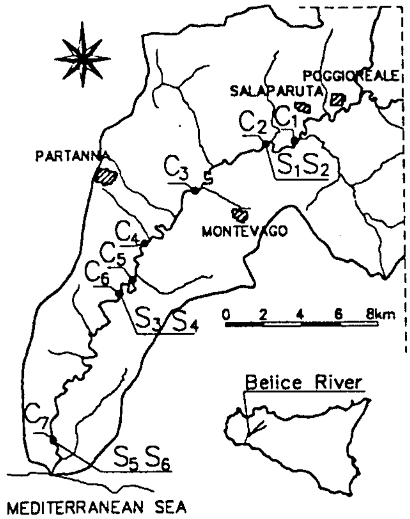

Figure 1: The sampling sites in the lower Belice

<u>Physico-chemical Indices</u>

Several Authors have proposed water quality indices (House, 1989; House and Newsome, 1989; Tyson and House, 1989; Halen and Descy, 1992; Lascombe, 1992). In this study the following were employed:

1) Water Quality Index after National Sanitation Foundation (as reported by Ott, 1978), *WQI-s*, where -s stands for -sum of the individual variables *(sub-indices)* q_i, 9 conventionally, previously multiplied by appropriate *weights* w_i: $\text{WQI-s} = \Sigma_i (q_i w_i)$;

2) as above, *WQI-p*, where -p stands for -product of the same *sub-indices* q_i, previously raised to their *weights* w_i: $\text{WQI-p} = \Pi_i (q_i)^{w_i}$;

3) Chemical Index *(C.I.)* adopted in Federal Republic of Germany since Eighties: $\text{CI} = \Pi_i (q_i)^{w_i}$. It is based on 8 sub-indices q_i, not completely coincident with the former ones, as CI ignores Turbidity, and replaces Coliforms with Ammonia, which is easier to detemine, and more reliable too;

4) Extended Biotic Index *(EBI)* by Woodiwiss, as modified for Italy by Ghetti *(see ahead)* (Ghetti, 1986).

Figure 2 (in Appendix) contains all the curves required to apply WQI after NSF. The shaded areas represent the margins of allowance for values to be assigned to q_i. Such allowance is inherent in the method itself (Ott, *cit.*). The values of the respective weights have been superimposed; their sum is 1.

According to the value of physico-chemical quality index awarded, the writers generally rank river waters in the following five classes, where ecological and economical criteria are linked:

<u>Class 1</u> (100 down to 91): excellent water, suitable for highest-value uses, such as drinking, sensitive fish species rearing, etc.; treatment required, if any, consists only of grit removal and disinfection;

<u>Class 2</u> (90 to 71): good water, high-value uses, low sophistication and cost required for appropriate treatment;

<u>Class 3</u> (70 to 51): fair water, high-value uses possible after treatment of mean cost and complexity;

<u>Class 4</u> (50 to 26): poor water, probably polluted, requiring complex and expensive treatment for high-value uses other than acclimatized wildlife conservation;

<u>Class 5</u> (25 down to 0): very poor water, severely polluted, requiring treatment for any use other than navigation and landscape conservation.

Biotic Indices

Biotic indices rest on the principle of greatest richness of most demanding species associated with healtiest environments. To obtain a numerical value for them, two variables are correlated: the sensitivity of established faunistic groups to extraneous chemical species, and the number of species that make up the macrobenthic community in the riverine biotope investigated. Extended Biotic Index *(EBI)* by Woodiwiss, as modified for Italy by Ghetti, employs a conventional table where the groups of macroinvertebrates, which can be found, are listed as decreasingly sensitive from top to bottom. The identification of macroinvertebrates proceeds down to level of families or genera (*Systematic Units* S.U.). EBI can range between 1 and 14; accordingly, water quality falls into one of five classes. Class I groups EBIs 10-14, describing an environment not polluted or not considerably altered; Class V, on the opposite, EBIs from 1 to 3 (severely polluted environment).

REPORT ON THE SURVEYS

Experimental data: physico-chemical and microbiological

The complete set of measured data is shown in Table 1 (commented below).

Table 1 - Physico-chemical data and Quality Indices resulting

Date	Site	DO % sat.	BOD_5 mg/l	T °C	NH_4 mg/l	NO_3 mg/l	PO_3 mg/l	pH	EC µS/cm	TDS mg/l	Faecal MPN	TSS mg/l	WQIp	WQIs	CI
	C_1	104	2.8	14.0	0.58	5.13	0.20	7.59	2440	1743		15.6		**76**	72
	C_2	102	2.1	14.7	0.64	3.32	0.12	8.08	2370	1693		3.6		**80**	72
20	C_3	106	1.9	13.2	4.90	4.42	0.15	8.22	2200	1571		36.8		**75**	57
Dec.	C_4	88	2.6	21.5	0.27	3.34	0.14	7.98	2100	1500		56.0		**63**	69
'93	C_5	90	2.1	21.0	0.28	3.16	0.17	8.05	2100	1500		28.0		**65**	71
	C_6	92	2.2	20.5	0.37	1.44	0.57	8.12	2150	1536		6.4		**66**	71
	C_7	95	2.1	18.0	0.44	2.11	0.22	8.07	2170	1550		20.4		**77**	73
	C_1	113	3.4	9.5	0.44	6.33	1.07	8.15	2160	1542	18000	85.2	**43**	**56**	69
	C_2	113	4.8	10.0	0.51	6.59	1.48	8.08	2260	1614	10000	109.2	**39**	**54**	63
25	C_3	115	2	9.0	0.53	6.71	1.05	8.14	2230	1592	3500	187.6	**42**	**57**	71
Jan.	C_4	102	2.8	14.6	0.62	5.18	1.26	7.9	2130	1521	110000	111.6	**37**	**56**	70
'94	C_5	104	2.7	14.0	0.50	5.15	0.87	8.07	2140	1529	8000	96.8	**47**	**59**	71
	C_6	104	2.2	14.0	0.60	4.47	0.53	8.07	2160	1543	5000	140.8	**45**	**60**	72
	C_7	104	3.4	14.0	0.35	4.41	0.77	8.04	2210	1579	11000	184.8	**42**	**58**	71
	C_1	102		15.0	0.84	4.25	0	7.40	1340	957	1700	33.2		**68**	70
	C_2	100		15.7	0.37	4.09	0.06	8.33	1016	726	200	13.6		**70**	73
15	C_3	102		14.6	0.34	4.09	0.21	8.32	1035	739	70	26.0		**69**	74
Mar.	C_4	102		14.9	0.37	4.05	0.3	8.13	1169	835	6500	40.4		**63**	73
'94	C_5	104		14.4	0.48	4.05	0.11	8.12	1187	848	1900	34.0		**66**	72
	C_6	104		14.1	0.40	4.21	0.11	8.11	1195	854	1500	52.8		**65**	73
	C_7	102		14.7	0.36	4.27	0.21	8.15	1221	872	5000	66.0		**63**	73

As in March river discharge was greater, lower values for EC and Coliforms resulted. DO, BOD, NH_4 and NO_3 depict slightly polluted waters in any country. EC is always and everywhere high in comparison with watercourses where both WQI and CI were originally conceived and validated. This results in low scores, although fluvial ecosystem is a viable one *(see EBI)*; such low scores are reasonable only inasmuch they refer to expensive treatment for water utilization, if required. Temperature Deviation was assumed zero everywhere, as no particularly hot or cold wastewaters are discharged into the river.

Considerable biological pollution was measured in C_1 and C_4, downstream raw wastewaters outfalls from Poggioreale and Salaparuta (C_1) and Partanna (C_4). The values of FC to FS ratio (Table 2) point out that cattle pollution is widespread over the entire river length investigated, whilst human waste pollution is restricted to few reaches downstream untreated sewage outfalls.

Table 2 - Total Coliforms (TC), Faecal coliforms (FC) and Faecal Streptococci (FS), in MPN/100 cc.

Site	TC	FC	FS	FC/FS	TC	FC	FS	FC/FS
	25 Jan. '94				15 Mar. '94			
C_1	30000	18000	5800	3.1	7000	1700	3900	0.4
C_2	20000	10000	11000	0.9	2000	200	320	0.6
C_3	10000	3500	6000	0.6	700	70	64	1.1
C_4	150000	110000	43000	2.6	15000	6500	2000	3.3
C_5	12000	8000	4600	1.7	3000	1900	1500	1.3
C_6	14000	5000	2000	2.5	6000	1500	2300	0.7
C_7	60000	11000	43000	0.3	9000	5000	1600	3.1

Experimental data: bionomic

Stations for biological survey were set in 3 pairs, all lying in regulated reaches of Belice. A good reproducibility of measures was observed (Table 3). During the sampling, carried out in May only, the faunistic group *Plecoptera* was never found, which is known to be the most sensitive to water pollution or habitat changes. In S_1 and S_2 just as many as 5 and 6 S.U. were found; the responsibility for that could be shared by civil and farm wastes, on one hand, and by flat sandy man-made river bed on the other. In S_3 and S_4 less degraded conditions were found, as 16 and 17 s.u. were counted, respectively, in spite of Partanna sewer outfall, over 9 km upstream. Such recovery can be explained with rich riparian vegetation observed, which enhances biological diversity. In the last stations, S_5 and S_6, S.U. number dropped to 8 and 14; bed uniformity and non-point farm pollution, revealed by microbiological analysis, are believed to contribuite both to degrade the river.

CALCULATING INDICES

Physico-chemical Indices

All of three physico-chemical indices introduced above were calculated and reported in the last three columns of Table 1. The empty places of BOD and Coliforms in Table 1, however, reveal that only the second survey rendered data sufficient for application of indices. In fact, the first one gave no values for Faecal Coliforms (FC), and unfortunately the same happened in the third for BOD. To deal with this partial deficiency of data, the writers devised a replacement criterion, as described below.

First, indices WQI-s, WQI-p and CI for second survey (complete set) were calculated. **Second**, the set of data from second survey was deprived of indicator FC. The authors attempted to maintain unchanged the values of indices above, distributing the weight of the suppressed indicator over the remaining ones (*criterion a*) or adding it to another single indicator, chosen as *related* to the missing one (criterion *b*), thus substitutable for it. **Third**, the experiment was repeated with BOD. Indicator BOD was considered related to FC, and Ammonia to BOD.

Tab. 3 - Systematic unit collected in sites S_i

MACRO-INVERTEBRATE GROUPS	SYSTEMATIC UNIT	S_1	S_2	S_3	S_4	S_5	S_6
TRICHOPTERA (Family)	Hydropsychidae			L		1	1
	Hydroptilidae					I	
EPHEMEROPTERA (Genus)	Baetis	L	L	U	L	L	U
	Caenis			1			
COLEOPTERA (Family)	Hydrophilidae	1	I		1		
	Dytiscidae	2	L	L	L	L	L
ODONATA (Genus)	Sympetrum		L				
	Lestes Viridis				L		2
	Pyrrhosoma						2
DIPTERA (Family)	Chironomidae	I		L	U	L	L
	Limoniidae				1		
	Ceratopogonidae			L			
	Simuliidae	U	L	I	L		L
HETEROPTERA (Genus)	Plea				2		
	Notonocta				L		
	Micronecta			I			2
	Naucoris			I	I		I
CRUSTACEA (Family)	Atyidae						I
GASTROPODA (Genus)	Ancylus			I	2	I	1
	Lymnaea			1	1	1	
	Physa		1	I	I		
BIVALVIA (Genus)	Pisidium			1	1		
HIRUDINEA (Genus)	Limnatis			2	1	1	1
OLIGOCHAETA (Family)	Lumbriculidae			I	1		1
	Lumbricidae			I	1		
	Total S.U.	**5**	**6**	**16**	**17**	**8**	**14**
	E.B.I.	**4**	**5**	**7**	**7**	**6**	**6**
	Quality class	**IV**	**IV**	**III**	**III**	**III**	**III**

I=from rare to common; L=from common to abundant; U=numerically dominant; 1e2= number of individuals captured

It was apparent that (see Table 4 below):

- *criterion a* (distributing weight) keeps the quality indices, calculated from the deprived set of data, closer to the values calculated from the entire;
- WQI-s appeared to be less sensitive to data deficiency than WQI-p;
- Chemical Index (C.I.) performs similarly, in spite of its inherent resemblance with WQI-p.

This behaviour was explained as follows: indicators FC and Turbidity are unfavourable throughout the river Belice, thus, if they are not included in an index (as it happens with CI) or they are replaced with another, more favourable indicator, total score will improve. Obviously, this case cannot be generalized.

Table 4 - Deviation of *calculated* values of Indices from *true*.

	WQI-s		WQI-p		CI	
	BOD lacking	CF lacking	BOD lacking	CF lacking	BOD lacking	CF lacking
Criterion a	- 4%	+ 16%	- 7%	+ 37%	± 7%	( )
Criterion b	- 12%	+ 17%	- 22%	+ 45%	± 8%	( )

Table 1 shows that, applying WQI-s and CI, water of lower Belice falls everywhere into Classes 2 and 3 defined above, with few seasonal shifts in some reaches. Applying WQI-p, waters are down-graded to Class 4. All Water Quality Indices applied exhibit good *stability* over space and time; this dampening effect tends to hide individual phenomena or forms of pollution, which could be better observed through the appropriate indicators.

In fact, CI yields always the highest scores, whilst WQI-p the lowest, because, a) the latter includes among basic indicators F.C. and Turbidity, which are fairly high nearly everywhere in lower Belice; and, b) by the very nature of product of WQI-p, low scores in individual indicators lower the aggregate index.

Biotic Indices

In stations S_1 and S_2, EBI's calculated values are 4 and 5, respectively; IV Class of Quality (severe pollution) was thus awarded. In S_3 and S_4, downstream, EBI rises to III Class (slight pollution). As water quality does not improve from the former reach to the latter, the merit should be attributed to almost natural river bed, with banks rich in vegetation. In S_5 and S_6 River Belice remains in III Class, in spite of diminished S.U., a symptom of lessened biological diversity.

The river environment, according to EBI values, falls into classes of quality everywhere lower than it appears from the physico-chemical and bacteriological indices. The writers' opinion is that the existing vast regulation works have simplified the fluvial habitat to such extent that most sensitive macro-invertebrates have disappeared in spite of fair water quality.

ACKNOWLEDGEMENTS

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APPENDIX

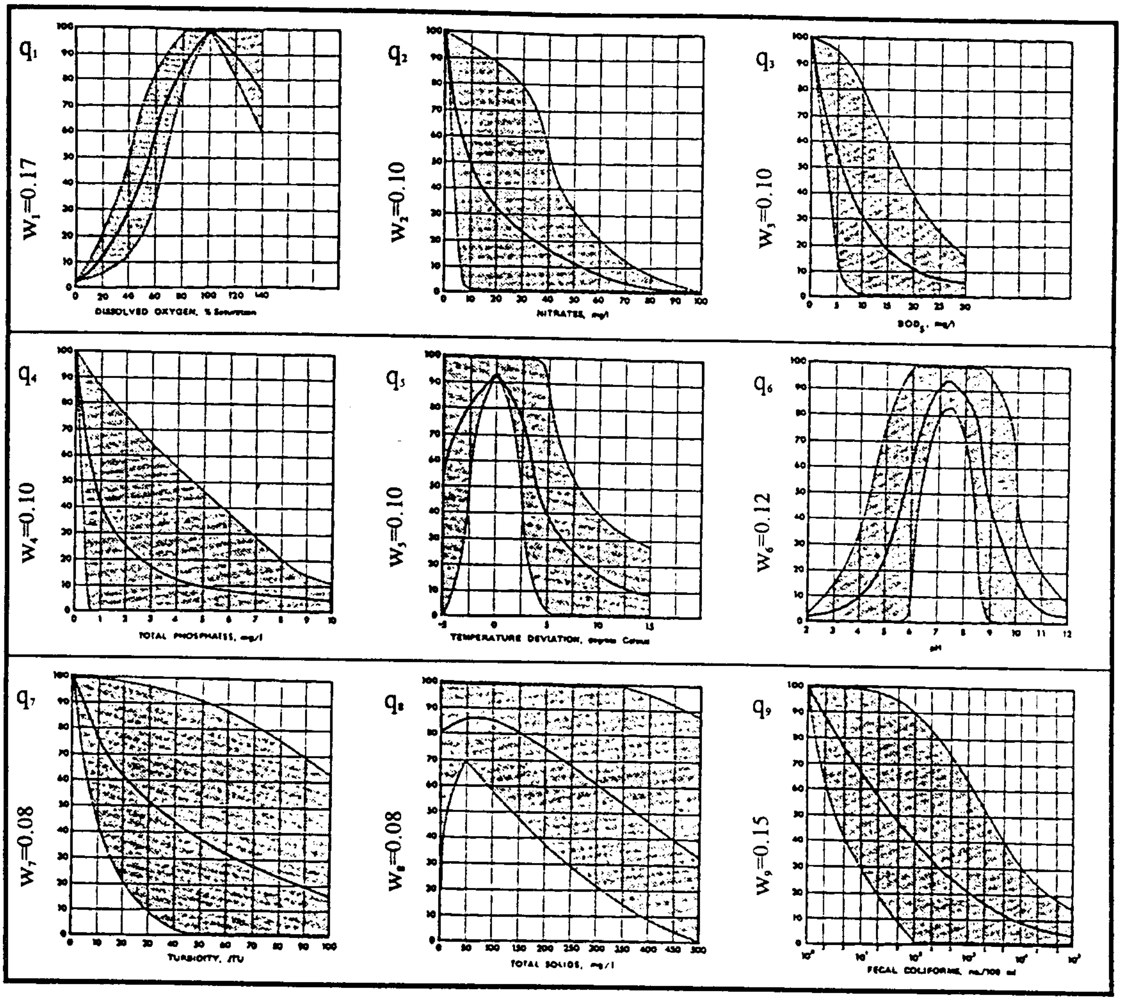

Figure 2: Subindex functions for WQI after NSF (Ott, 1978)

Interdisciplinary approaches - Partnership

Approches interdisciplinaires - Partenariats

ECOHYDRAULICS 2000
SYMPOSIUM IN QUEBEC, CANADA, 11-14 JUNE 1996

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MANAGEMENT AND USE OF RIVERS IN NORWAY: MEASURES TO ALLEVIATE THE IMPACT OF HYDROPOWER DEVELOPMENT

ABSTRACT

Norway measures 1800 km from south to north. The coast is long and the country is intersected by numerous large and small rivers. The annual runoff from individual catchment basins varies from 160 l/s/km2 in the wettest parts of the country to 10 l/s/km2 in the driest.

Norway is one of the few countries in the world that obtains all its electrical energy from hydroelectric power stations. 63% of the water power potential has been developed and 20% is protected from being exploited. When permission is given for a development scheme, considerable emphasis is placed on alleviating its possible impact on nature and the environment.

To determine the effects of a development, investigations must be undertaken before and after the development. The most detailed studies generally concern fish. Attempts are made to limit negative effects by improving biotopes along the river. Compensatory release of fish has also been widely used, but recently attempts have been made to limit this because of the potential negative consequences involved.

In recent schemes, the minimum flow has been experimented with for a five-year trial period, generally with specified lower and upper limits. Biological studies, which the developer must fund, accompany the experiments. The costs of investigations and measures attached to a development amount to 2-3% of the total cost of the development.

KEY-WORDS: Management of rivers/Legislation/Hydroelectric plants/Watercourses/Impacts of development.

INTRODUCTION

Norway is situated on the Scandinavian peninsula and stretches 1800 km from south to north. A warm ocean current, the Gulf Stream, flows along the coast keeping it ice free all the year. Large parts of the country are mountainous, with some peaks reaching nearly 2500 m. The average altitude of the country is estimated to be around 500 m. The tree limit in southern Norway is at about 900 m a.s.l. The population mostly lives on the coast and in inland valleys.

South Norway is divided by a long mountain range which forms a watershed between east and west. The dominant wind directions are westerly and southwesterly and the winds carry damp air from the Atlantic Ocean across the country. West of the watershed the annual precipitation may be around 2500 mm, but in the rain shadow on the east side the driest areas only receive 300 mm. Figure 1 shows the variation in annual precipitation from one part of the country to another. (NVE, hydrologisk avdeling, 1986)

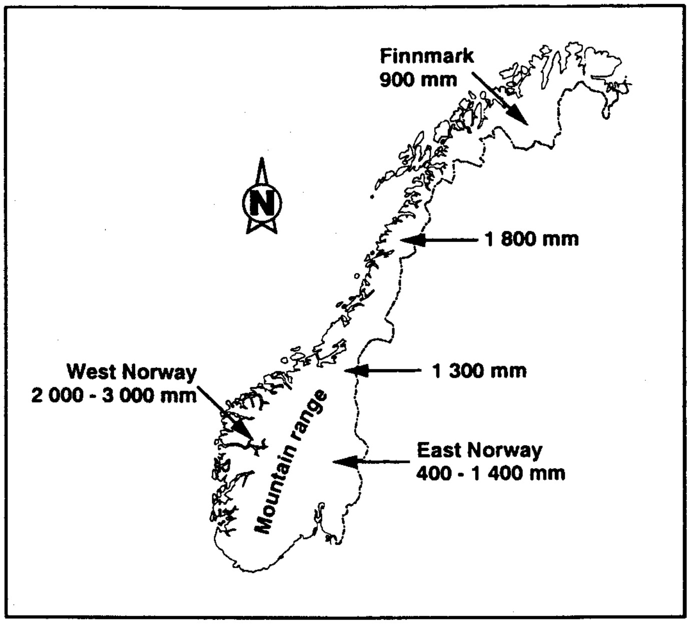

Figure 1 Map of Norway, including precipitation figures for different regions.

The northern part of the country also has a high annual precipitation along the coast, except in Finnmark and Troms where it is low everywhere. The precipitation mostly falls as snow in winter and rain the rest of the year, least in summer. The considerable amount of melting snow in spring means that the water level in the rivers is at its highest then.

Norwegian watercourses are relatively small, but contain a great deal of water. Only five have a catchment area in excess of 10,000 km2. the Glomma being the largest at 41,000 km2. On the other hand, they have a substantial fall in altitude, many lakes and, in some cases, very large specific discharges.

Rivers in western Norway are often short, but contain a great deal of water. In southeastern Norway, they are larger and have a more gentle profile, the main fall frequently being concentrated in their upper reaches. Some rivers in Finnmark are large, but they have a gentle fall Figure 2 shows profiles along rivers in these three parts of the country.

Numerous lakes at various altitudes along rivers provide good opportunities for regulating the rivers. The largest lake is Mjøsa with an area of 362 km2.

PAST AND PRESENT UTILISATION

Norwegian rivers have been put to a variety of uses for a great many years. There have been numerous watermills for grinding corn, and their remains can still be seen on many rivers. They were simple devices where the water was led along a channel to a waterwheel which was directly linked to the grindstone. It is estimated that there were nearly 30,000 such mills 200 years ago. (Hydropower Development, 1995)

The rivers have also been important means of transportation for both people and timber. Timber floating has now ceased, and logs are transported by road to sawmills and factories which no longer need to be situated on rivers.

Access to groundwater is limited in many parts of Norway. Consequently, rivers have been, and still are, important sources of raw water. Except for their lower stretches, contamination levels are low and the water can be used without resort to expensive purification.

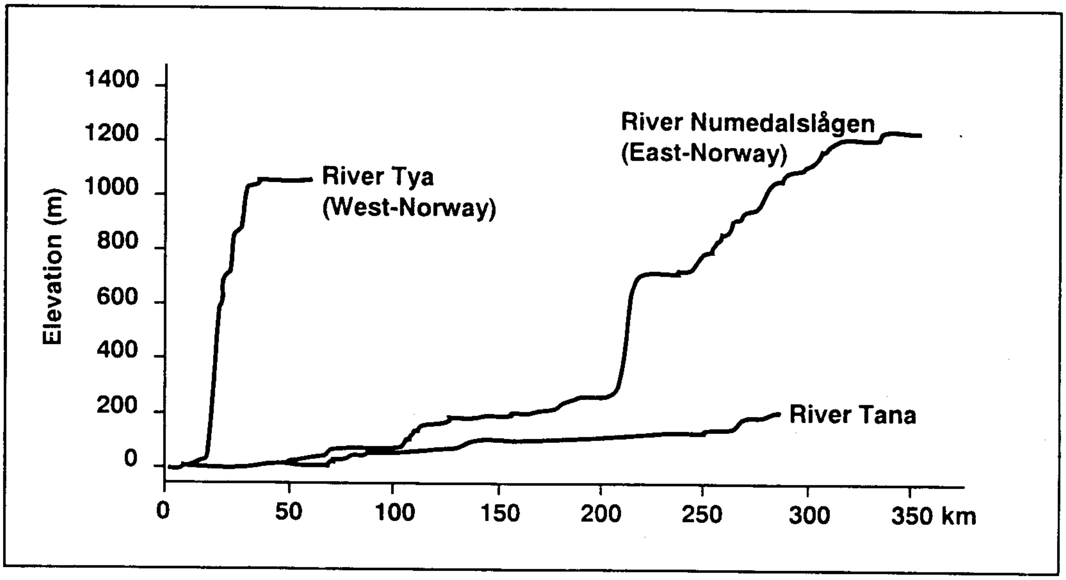

Figure 2 Three typical longitudinal river profiles in Norway.

Most rivers contain anadromous fish. At present, 959 Norwegian rivers carry salmon and sea trout. Fishways have been built on many rivers, to open up new stretches for migrating fish. A total of 420 fishways have been constructed in Norway. The letting of fishing rights has been, and still is, an important source of income for many owners of riverside land. The fishing of non-migratory brown trout and char is also very important in many lakes and stretches of river where anadromous fish are absent. (DN-notat 1995 - 1)

The most important use to which rivers are put today, apart from recreation, is to produce electric power. Almost all Norwegian power is produced in hydroelectric plants. 63% of the hydropower potential has been developed, and about 20% is in locations where power development schemes are forbidden. When permission for a scheme is given, great efforts are made to reduce the negative impacts which the development might have on nature and the environment. About 112 Twh are now produced in Norway.

LEGISLATION

The use of the rivers is regulated by, in the main, two Acts, the Watercourses Act and the Watercourse Regulation Act introduced in 1940 and 1917, respectively.

The Watercourses Act concerns the ownership of rivers and how they can be utilised through development. The Act has been particularly important for setting limitations on the exploitation of gradients to produce hydroelectric power. Watercourses are deemed to be owned by the owner of the adjacent land.

The Watercourse Regulation Act makes provisions concerning the regulation of the discharge of a watercourse. Every regulation which involves the inundation of land or the lowering of a reservoir is dealt with under the terms of this Act. Permission (concession) for such undertakings may be given for a period of 60 years, or for an unlimited period if the applicant is a publicly-owned company.

New development plans must also be dealt with under the terms of the Planning and Building Act. A specifically defined environmental input assessment is required in the case of projects that have an annual production in excess of 40 Gwh. This is intended to form a good decision-making basis for groups presenting submissions and the authorities awarding concessions. (DN-notat 1992-7)

HANDLING APPLICATIONS FOR POWER DEVELOPMENT

A concession must be applied for if it is desired to construct a new power station in Norway, and in the case of major projects, this must be decided by the government. Before permission is given, a long and wide-ranging deliberation process is enacted. This procedure is shown schematically in somewhat simplified form in Figure 3. It is divided into three phases: (NVE, nr. 13-1993)

-planning
-application
-total appraisement and decision-taking.

The time this takes varies greatly depending on the complexity; it may be anything from 1 year to 5 or 6 years. When permission is given, the limits for the regulation of the reservoirs and requirements for storing and releasing water are laid down in a set of manoeuvring regulations which are part of the terms of the concession.

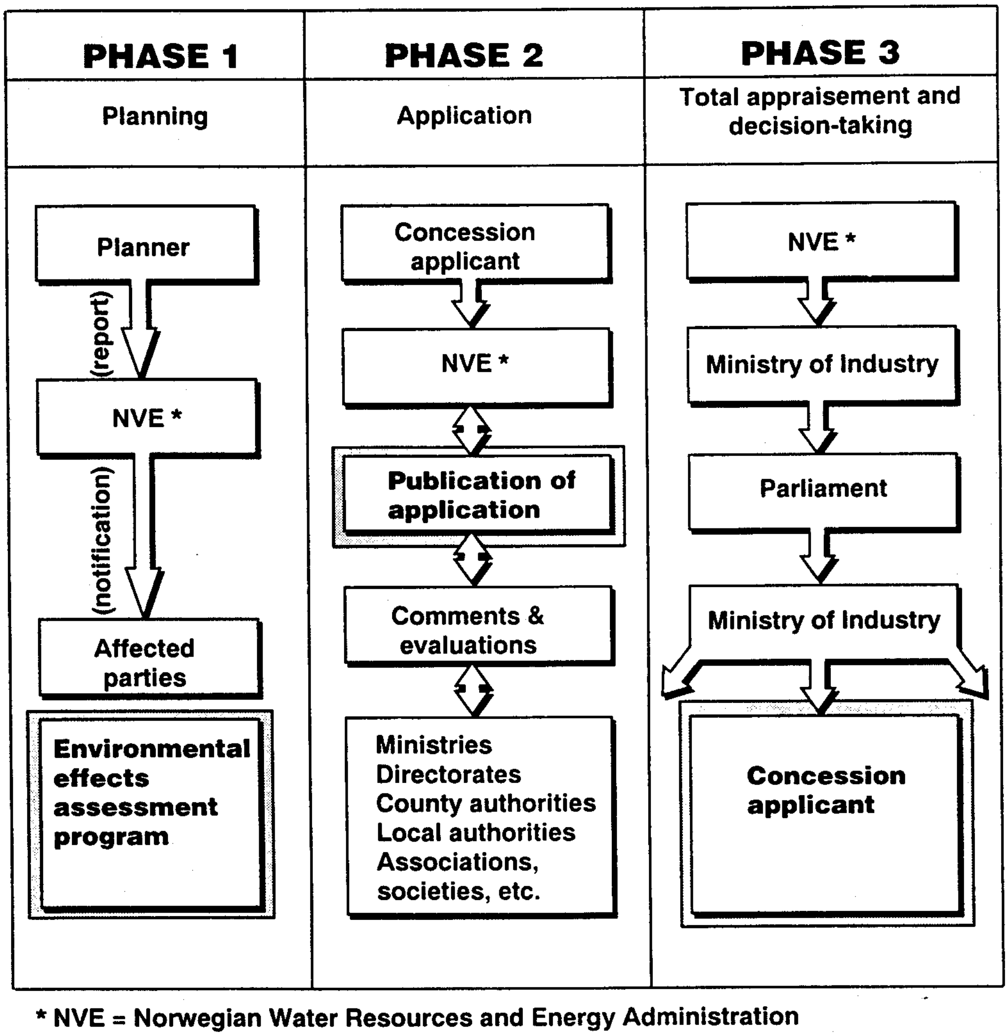

Figure 3 Procedure (in simplified form) for dealing with applications for hydropower development

The concession also includes terms stating what investigations and measures a developer can be required to undertake to safeguard the natural environment, or to compensate for damage which the development causes. The management authorities, following careful evaluation of what is needed, can require such investigations and measures as:

-biological investigations
-construction of fishways and cultivation plants
-release of fish
-building of thresholds and biotope improvement measures
-measures relating to open-air recreation.

TYPES OF POWER STATIONS

The power station projects in Norway can be divided into three groups:

-river power stations in dams
-power stations with intake from a single reservoir
-"gutter" projects, involving several intakes.

River power stations are generally located at a waterfall or rapids. They exploit the discharge of the river without regulating it.

A power station with an intake from a single reservoir exploits the drop between that and the river. The distance between the intake and the outlet from the power station is often great, with the water being transferred through tunnels. The discharge is determined on the basis of the size of the catchment basin and the capacity of the reservoir.

The term "gutter" project is used to describe a scheme whereby water is led to a power station after being collected through several intakes from a number of reservoirs and tributaries. These are often high altitude lakes and streams, each of which has a limited discharge, but often a substantial drop in height. In such projects, a minimum flow for the main river is stipulated which must not be violated.

Figure 4 shows sketches of these three types of hydropower schemes. (Norske kraftverker)

IMPACTS OF DEVELOPMENTS ON NATURE AND THE ENVIRONMENT

The construction of a hydropower station impacts on many aspects of the natural environment. Changes in discharge and water levels have the greatest effect on the habitats of aquatic invertebrates and fish. Other wildlife may also often suffer. The migratory routes of animals may be cut through inundation of land for reservoirs or the construction of new roads, and transmission lines may reduce the value of unspoilt nature. The development may also produce climatic changes which affect the vegetation.

Lake regulations may impose substantial changes on the living conditions of aquatic invertebrates and fish. If the water level is lowered, the opportunities for trout to reach their spawning grounds in rivers and streams entering the reservoir will be ruined. Other species may find their spawning grounds left dry and the benthic fauna in the littoral zone will lose their means of survival through scouring or erosion. (MVU-rapport A-12a,b,c 1989)

A reduction in the discharge along the river will lead to a loss of production areas for aquatic invertebrates and fish and make it difficult for migrating fish to reach their spawning grounds. Moreover, rapid changes in discharge may leave fry/small fish and aquatic invertebrates stranded on dry land if the water level is lowered too quickly.

A development scheme may also have a negative effect on fishing along a watercourse. When the discharge is lower than normal, migrating fish will often remain stationary waiting for a higher discharge before running up the river.

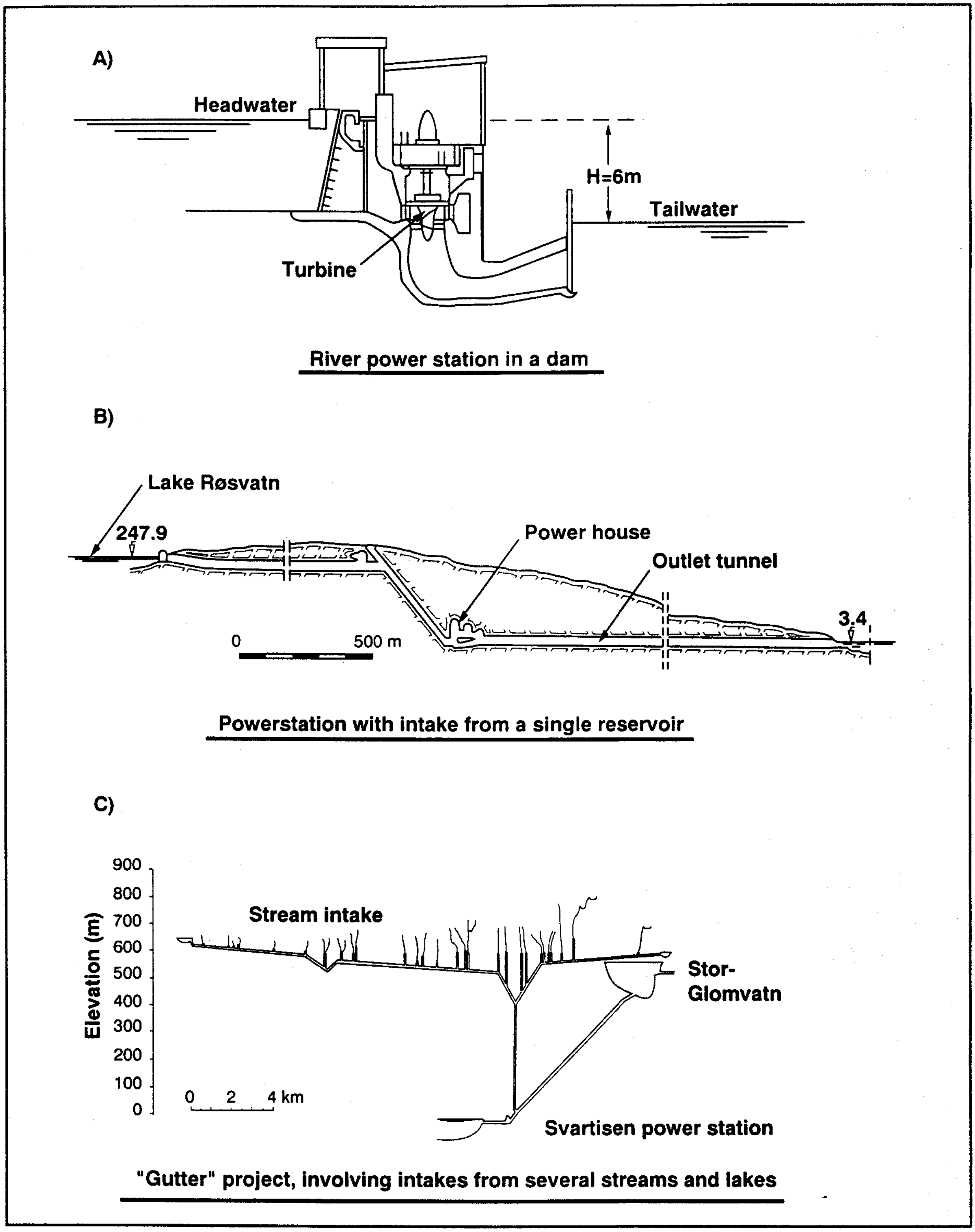

Figure 4 Sketches showing the three main types of hydropower schemes.

The qualities of the fishing spots used by anglers will change when the river has to adapt to new discharge regimes following a regulation. (Statistics of Norway, 10/1984)

MEASURES TO ALLEVIATE IMPACTS

To alleviate the impacts of a power development scheme it is vital to have the chance to influence its design. Plans to safeguard the natural environment need to be put forward sufficiently early in the planning process to be taken into consideration. This applies to the positioning of intakes, outlets and dams, and their design.

After the scheme has been completed a number of measures are still called for. Nowadays, attempts are generally made to avoid undertaking measures of a physical nature, if conditions following the completion of a hydropower scheme can be maintained some other way, such as through flexible regulations for the release of water. In Norway, such regulations are of great importance for fish. It is claimed that if the discharge requirements for fish are satisfied, the conditions are also satisfactory for many other parts of the ecosystem.

For fish, the volume and flexibility of the discharge are important for several reasons. The demands which fish place on the environment in which they live differ for different species and stages in the life cycle. A particular discharge may be favourable for one life stage, but unfavourable for another (satisfactory for growing up, but not for migrating upstream). A discharge that is favourable for one species may be unfavourable for another (satisfactory for carp, but not for trout). The life stages and the species may require different levels of discharge at different times of the year (high in spring, but not in winter). A discharge that is favourable in one part of a river may not be favourable in another part (appropriate in a narrow profile, but not where the profile is wide).

To be able to determine the effects of a development on nature and the environment it is usual to carry out investigations both before and after the development. They comprise mapping:

-the species composition and the occurrence of flora and fauna
-the stock sizes and migration routes of the fauna
-the relationship between the discharge and the environment for wildlife to live in.
-landscape and outdoor recreation

When hydropower development takes place, a large proportion of the water is often removed from the watercourse, or the discharge is reduced over stretches of the watercourse. When permission is given for such a development, a minimum discharge is stipulated for the river and this must not be violated. This discharge generally varies from one time of year to another.

In recently implemented schemes a minimum discharge has been stipulated for a trial period of five years. Upper and lower limits are generally set for this experimental discharge. When the minimum discharge is being determined, the following factors are considered with regard to a stretch of river:

-area covered by water
-the need for spawning grounds and hatching areas
-the need for periods of rest, pools
-oxygen-rich water, water exchange
-requirements regarding water temperature
-transport of nutrients from lakes
-run of fish, floods to tempt them
-emigration of fish (smolt)
-pursuit of fishing.

The measures used to alleviate the negative effects may be:

-construction of thresholds, biotope improvement
-construction of fishways
-construction of cultivation plants and release of fish.

In recent years, attempts have been made to reduce the release of fish because of its possible negative side-effects.

The costs of the measures implemented and the investigations linked to a development scheme may amount to 2-3% of the total costs of the development.

SUMMARY

To attain better management of the natural history of Norwegian watercourses a master plan for water resources has been drawn up. This has resulted in about 20% of the nation's water power potential being protected from hydropower development. In practice, this means that these watercourses are also protected from other forms of exploitation. In these watercources applications for other form of encroatchments is to be handled more restricted than in other watercources. National guidelines have been drawn up for encroachments in these watercourses, in order that their conservation assets shall not be depreciated.

63% of the nation's total hydroelectric potential has been developed. The remaining 17% of watercourses which are still open for possible development are categorised and grouped according to costs and consequences. This has enabled a ranked list to be drawn up for exploiting the remaining hydroelectric power potential of the country. Even though this list has been prepared, any development requires an application and its approval in accordance with the legislation that is in force. This system seems to have adequately clarified the situation for those interested in development, and reduced the degree of conflict in the case of new projects.

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THE SUCCESSION OF THE MARCHFELD CANAL - A NEWLY CREATED WATERCOURSE

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ABSTRACT

The Marchfeld Canal represents a quasi natural watercourse (18,8 km, 2-15m³/sec) in the east of Austria (Europe) with the following functions: recharge of ground water, recreation of existing rivulets and irrigation of agricultural areas. In 1991, one year before it was filled, an interdisciplinary research program started to investigate its succession. The aim is to show the development of a complex ecosystem, the influence on the surrounding areas and to describe the essential interactions.

The focal points are limnology (nutrients, algae, makrophytes, evertebrates), fish ecology (colonization, migration, habitat preference, fish pass function), hydrology (current patterns, current model), sedimentology (surveys, types of structures, quality and quantity of the sediment), and bioclimatology (bioclimate, physical limnology). The terrestrial/amphibious part includes the investigation of the establishment of vegetation at the water-level and in xerothermic sites, the succession of mammals, birds, amphibians and some groups of insects and the function of the canal as a recreational area for human visitors.

The program will be finished in 1997. Now preliminary results for each single project are available. They allow a first synthesis dealing with complex interdisciplinary questions (e.g. fish habitats-current, makrophytes-current-sedimentation).

From an ecological point of view, the Marchfeld Canal can be seen as an artificial branch of the River Danube (MQ = 1900 m³/s). The smaller dimensions and the artificial construction of the canal have caused a dramatic change of the abiotic situation, and subsequently of the biotic situation. The preliminary results of the different focal points are discussed in detail with the aid of different theories and concepts of successions and river-ecology (e.g. Vannote's River Continuum Concept). The main differences between the newly created watercourse and a natural river are brought out.

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KEY-WORDS: Artificial / Channel / Succession / Colonization / Limnology / Fish-ecology / Hydrology / Sedimentology / Bioclimatology / Interdisciplinary Research.

INTRODUCTION

The Marchfeld Canal System

The Marchfeld Canal System includes the newly created Marchfeld Canal (MC, finally filled in 1992; length: 18.8 km; slope: 0.2 o/oo) and Obersiebenbrunner Canal (finally filled in 1995; length: 5.4 km) and the re-widened parts of the small rivers Rußbach and Stempfelbach (lengths 39 km and 26 km). Water (2 - 15 m^3/s) is taken from the river Danube (HQ1 = 5.300 m^3/s, HQ10 = 7.300 m^3/s) north of Vienna and transported by the MC in a free-flow to the Rußbach. When the water level of the Danube is too low, pumps are used to raise the water into the channel. A retention basin (approx. 100 ha) in the upper part of the MC (km 3.0 - 3.6) serves to improve the water quality. Via the Obersiebenbrunner Canal a part of the water of the Rußbach is diverted into the Stempfelbach. The Rußbach runs directly into the Danube south of Vienna , the Stempfelbach runs into a tributary of the Danube. The Marchfeld Canal System was constructed to improve the water conditions of the Marchfeld (approx. 1,000 km^2), an important agricultural area of Austria (Europe). On the one hand water can be taken directly out of the MC and used for irrigation, on the other hand water seeps into the ground at favourable points to stabilize and raise the falling ground water level, which makes it necessary to control or lower the amount of particles transported. The water quality of the Rußbach and the Stempfelbach is improved by the additional supply of water.

Problems

The MC - the main object of investigation - was to be built in harmony with nature as far as shape, riverbed-design and plantings are concerned (Grubinger and Ernegger, 1994). But during the planning stage, several questions and problems very soon arose out of this aim: How naturally can an artificial river be designed, and what would be the appropriate measures? Would it be better to predetermine as much as possible or to give nature plenty of space to develop the watercourse its own way?

Initial Situation

Right from the beginning there were some differences between the MC and a natural river system. By law the river beds of all new and re-widened parts of the Marchfeld Canal System had to be sealed using either a mineral or a synthetic layer. In order to shape the river bed and to protect the layer it was covered with gravel up to a height of at least 60 cm. But such a gravel bed is not at all typical of a low-land river of such a location and such small dimensions. The discharge varies only between 2 m^3/s and 6 m^3/s - in exceptional circumstances it can reach 12 m^3/s. Four weirs regulate the level and flow of water in the MC. In order not to interrupt the continuous flow of water and to allow organisms to pass, bypassing channels were constructed. Another great difference is the fact that the MC does not have a normal source - it receives its water from the potamal of a large river.

Succession

Only very few papers dealing with the succession of natural river systems exist. It is even difficult to find universal patterns of common structures and courses of succession. A good criterion of assessment to test the MC and its

Table 1: The individual projects of the Research Program Marchfeld Canal

(Sub-) Project	Topic	Interdisciplinary Aspects (examples)
Bioclimatology	microclimatological and physical-limnological measurements	effect on organisms
Hydraulics	flow velocity measurement, flow patterns	interaction with water plants; effect on sedimentation
Morphology	surveying and classification of the research sections	interaction of habitat structures and species composition
Sedimentology	sediment quality, sediment transportation	effect on spectral radiation transmission and organisms
Limnolog. Chemistry	nutrients, temporal and spatial distribution	development of phytoplankton
Phytoplankton	succession, spatial distribution	development of zooplankton
Periphyton	succession, spatial distribution	interaction with sediment and flow velocity
Macrophytes	succession, spatial distribution	interaction with sediment and flow velocity
Evertebrates	succession, spatial distribution	food supply for fish
Fish-ecology	succession, microhabitats, fish pass function	population density depending on food supply and flow velocity.
Amphibious Vegetation	establishment in the riparian fluctuation area	development of habitat structures
Semiterrestrial and Terrestrial Zoology	dragonflies, amphibians, reptiles, birds, small mammals	population densities depending on the vegetation
Recreational Use	function of the MC as a recreational area	anthropogenous influence

'ecological abilities' seems to be a comparison with natural river systems of the same size and within the same geographical region. Such a comparison will only be possible when the MC has had enough time to develop. Nevertheless, from a scientific point of view it is very interesting to investigate the development of this newly created water-course right from the start to improve our knowledge about succession and the ecology of rivers.

METHODS

The Interdisciplinary Research Program Marchfeld Canal

To investigate the various problems that derive from the questions mentioned above, an interdisciplinary research program was founded (Grubinger and Ernegger, 1994). The emphasis had to be limited to the aquatic area. Associated projects examine the amphibic and terrestrial parts. Tab. 1 shows the individual projects, their fields of research and interdisciplinary connections. There is not enough space to give all their different methods. Detailed information is included in the individual papers (see references). A basic project organisation connects the different parts of the research program and supports the interdisciplinarity and co-operation. The investigations started in 1992, before the canal was finally filled and should last until 1997. The investigations of most projects are concentrated in 11 research sections of an average length of 200 meters. Six of them are within the MC, 5 within the Rußbach. They were chosen by all participants to take the following factors into account: rate of urbanisation, shape and structure of the waterway and overgrowth of the banks.

Models of Succession and Running Waters

The basis for a comparison between the results and already existing models are ecological concepts which were designed especially for running waters, such as Vannote's the River Continuum Concept (Schönborn, 1992; Lampert and Sommer, 1993). As far as the problems of succession are concerned, the papers written by Klötzli (1993) and Conell and Slayter (1977) are used. A recent overview is given by Pahl-Wostl (1995).

RESULTS

Because of the limited extent of this paper, it is only possible to present the most important preliminary results.

Thermal ecology

The water temperature of the MC is highly influenced by the thermal character of the Danube as one of the greatest rivers in Europe (Eitzinger *et al.*, 1995; Krisa *et al.*, 1996). But with increasing distance from the inlet the amplitudes of the daily and annual water temperature increase significantly. During the warm season the water warms up greatly when it flows through the channel (by up to 9 °C in 1995), and on the other hand it can cool down by up to 5.5 °C in winter. The maxima of the daily water temperatures show a clear time lag as the water flows along the channel. With the aid of a simple regression analysis it was possible to determine that the monthly average of the water temperature is highly correlated (95 - 98 % at the different locations) with the monthly average of the air temperature, whereas the water temperature and the global radiation show no great correspondence (55 - 75 % at the different locations).

Radiation Climate

The measurements of the spectral radiation transmission in the channel were carried out using an under-water photometer (Eitzinger *et al.*, 1995; Krisa *et al.*, 1996). In summer the total transmission diminished with increasing distance from the start of the channel, the maximum transmission changed to longer wavelengths. In winter these phenomena are much less pronounced. This could be due to increased biological activity in the water and an increasing concentration of light-absorbing matters (chlorophyll and other organic matters) during summer (Eitzinger *et al.*, 1995). Additionally, higher discharges and the resulting mobilisation of sediments could be of importance for the transmission. More detailed investigations are planned.

Discharges

As was mentioned above, the MC was built with a maximum discharge of 15 m^3/s (bankful discharge). The regulation of the discharge by the 4 weirs corresponds to that. In practice, the mean discharge varied between 2 and

7 m^3/s. The maximum discharges in the years 1993 - 1995 were 8 m^3/s, 7 m^3/s and 12 m^3/s. The resulting mean velocities were 0.3 - 0.7 m/s, the maximum velocity was 0.9 m/s (12 m^3/s). No extreme variance of the annual discharge occurred. In general, depending on the water levels of the Danube, it was possible to raise the discharge during late spring and early summer, whereas especially in autumn and winter it was often necessary to transport the minimal amount of water (2 m^3/s) into the channel with the aid of pumps.

Morphology

To classify the singular research sections by means of morphological similarities, in a first step 'conventional parameters' such as the variance of width and maximum depth, wetted bed area, approximated length of shore line and others were used for factor and cluster analyses (Wick *et al.*, 1996; Nachtnebel *et al.*, 1996). First results show that poorly structured sections can be separated from ones with varying cross section parameters and from others showing high variability with regard to cross section and area. In a first approach a combination of these parameters was compared to the number of fish species and the number of individuals. A positive correlation could be observed for the first time in the year 1994.

Sediment Transportation, Accumulation Processes and Sediment Quality

Generally speaking, more sedimentation than erosion occurred, especially in the retention basin (Schreiner *et al.*, 1995; Nachtnebel *et al.*, 1996; Reitner and Kralik, 1996) . In the deep part of this area a thick layer (average 13 cm, maximum 18 cm) of fine grained sediment was observed nearly two years (June 1994) after the channel had been finally filled. The remaining reference sections did not exhibit any dynamics at this time, so that all in all the quantity of the dynamic processes is relatively low. The average concentration of suspended sediment showed a characteristic longitudinal decline in 1993. The influence of the retention basin was quite obvious. Compared to the input (100%), the concentration diminished after the retention basin to less than 60 %, and at the end of the channel to app. 40 %. Whereas the situation was quite less clear in 1994, a similar pattern occurred in 1995, even though the concentration at the end of the channel was higher than in 1993 (80%). This may be due to the improved control and regulation of the discharge in the channel: To ensure a sufficient water quality (relating to dissolved and particular mineral and organic substances) for the groundwater infiltration, the discharge was reduced to a minimum when the water quality of the Danube was low (e.g. at high waters). In extreme cases the inlet was completely closed, but with the aid of the water accumulated by the weirs it was possible to ensure the minimum discharge.

Phytoplankton

Great seasonal differences were registered concerning the succession of phytoplankton (Krisa *et al.*, 1995; 1996). They are correlated with a combination of different parameters such as radiation, temperature, nutrients, turbidity and the discharges of the river Danube. The level of phytoplankton biomass was low - normally less than 1 mg/l - in winter, whereas maxima of more than 30 mg/l (1993), 17 mg/l (1994) or 10 mg/l (1995) were observed in spring. It is striking that the maxima decline from year to year, a phenomenon that requires further investigation. During downstream transport in the MC taxonomic composition and biomass changed significantly. In general the mean biomass increased by about 58% (1994 and 1995) by natural growth. Additionally algae may be swept out of areas with lower current velocity (e. g. bays). At higher discharge levels the growth rate diminished because of the shorter retention time, the higher turbidity and the lower light intensity. The nutrient loading was too high to make any serious limitation seem likely.

Benthic Algae

The epilithon is dominated by a filamentous green alga - *Cladophora glomerata* (Krisa *et al.*, 1995; 1996). In 1993 the biomass in the MC was very high, but it decreased clearly in the following years. A significant correlation with nutrients (esp. phosphate) was not found. Maybe the lower concentration of suspended sediment and higher light transmission corresponding to it caused the high growth rates of 1993. Generally speaking, high current velocity and greater light intensity, as occur in shallow areas, seem to be of benefit to the growth of *Cladophora glomerata*. The biomass observed varies between a mean of 0.1 and 0.9 mg ashfree dry weight (AFDW)/cm² and a maximum of more than 7 mg AFDW/cm² at favourable locations. Diatomeae and *Spirogyra sp.*, another filamentous green alga, are very common in areas with fine sediment layer. High biomass was observed in summer at shallow locations with high temperatures and low or non-existent current velocity.

Macrophytes

The MC was colonized by macrophytes very quickly (Krisa *et al.*, 1995; 1996; Wychera and Janauer, 1995). Early colonization was carried out by *Chara sp.*, as has been observed in similar water systems (Wychera *et al.*, 1992). Only two years after the final filling, 13 submersed macrophytes, 3 pleustophytes and 5 floating hydrophytes were found. In 1995 the number of species decreased to 13. Still water species (e. g. *Hydrocharis morsus-ranae, Nymphea alba*) disappeared, obviously the current of the MC made a successful settlement impossible. Vice versa, rheophilic species such as *Ranunculus trichophyllos* or *R. fluitans* increased or occurred for the first time. The main reasons for the quick colonization are the distribution by water birds and the fact that the macrophytes can enter the channel directly from the Danube. Continuous changes in the range of species, both within one vegetation period as well as during the four years of observation suggest that the vegetation system is still unstable at present.

Current velocity measurements have shown that macrophytes both prefer locations with low velocity current conditions (thereby further reducing the velocity) and also settle in locations that were originally unfavourable (thereby creating new areas of low current velocity) (Kotek *et al.*, 1995).

Evertebrates

The evertebrate community in the upper stretch are clearly determined by the Danube (Gaviria *et al.*, 1995; Krisa *et al.*, 1995). The density of zooplankton in the retention basin is the same as in the Danube (1994: max. 400 individuals/l). The lower stretches are more isolated from the main river and here the density is higher, esp. in bays and areas of low current velocity (up to 13,000 ind./l). This fact indicates that the water does not remain in the retention basin for long enough.

Various stretches showed different patterns of distribution of zoobenthos. Where the water flows out of the retention basin, many filtering species were found. They profit from the increased supply of washed out animal and plant plankton. Some groups of animals had specific patterns of succession. The reduction of the number of filamentous green algae led to a fall in the number of epiphytic 'grazers', which were replaced by typical inhabitants of epilithic areas. There was a clear sequence of different sizes of microcrustaceans. Small types replaced the larger ones that dominated during the first two years. This phenomenon was probably caused by the diminished number of filamentous green algae and the increased population of fish needing nourishment. The more fine sediments were deposited, the more sand- mud- and detritus-eaters were to be found in the slow areas and bays of the channel. The density of benthal fauna was higher in the soft sediments (1994 max. approx. 120 ind./cm²) than in gravel (1994 max. approx. 30 ind./cm²).

Hyporheic Interstitial

Preliminary investigations showed the existence of meiofauna (Rotatoria, Copepoda, Oligochaeta, Chironomidae, Nematoda) and stygnobiont amphipodes in the gravel bed (min. thickness 60 cm) above the artificial sealing (Gaviria and Pospisil, 1995). More detailed investigations are planned.

Fishes and fish passes

In 1993 (one year after filling) 40 species of fish were found in the MC and the revitalised Rußbach (Schmutz *et al.*, 1994). The density of adult fish in the channel was low. In 1994 there were 4 further species. The density of individuals increased 2.5 - 4 fold during the same period. This unexpectedly high number of species is only possible because conditions in the Danube are quasi-natural (providing a diversity of species) both upstream and downstream of the MC-system. The fish entered the system from both ends: larvae drifted into the channel directly from the Danube, whereas adults and subadults entered via the mouth of the Rußbach. Counts showed that up to 15,000 larvae and young fish entered within 24 hours (mostly at night).

Fish passes were necessary to enable the fish to enter from below. After they had been adapted, the number of fish entering increased substantially. In 1993/94 there was a high density of fish in the weir pool. The reduction of this density in 1995 was due to the improved fish passes and the increased presence of piscivore species (Schmutz, 1995). This phenomenon was registered throughout the system in 1995, whereas the density of mass fish species fell. Hydraulic investigations of the fish passes showed that attracting current velocity was sufficient at flows of less than 4 m³/s and the resulting velocities in the main channel, whereas higher flows made the situation at the entrance of the fish pass the same as in the main channel (Bernhard *et al.*, 1995). Future research will show if a higher flow-rate in the bypass helps the fish to find the entrance, and so improve the performance of the fish passes.

DISCUSSION

The determining situation (physical environment)

The initial situation was not the same as with a conventional, newly created, flowing stretch of water. There is no natural source or upper stretch. Water is taken from the potamal of a big river - the Danube - and fed into a body of water of much smaller dimensions. Therefore the measurements of water quality and of some organisms (especially plankton) in the inlet area are identical to the measurements in the Danube. So, referring to the River Continuum

Concept, the MC can be called a discontinuity of the Danube. Compared to the river, the abiotic factors of the determining situation in the MC show some differences:

The temperature regime is completely different. Compared to the Danube the MC is a small and shallow body of water. The temperature periodics are therefore more marked, but the situation does not change within short stretches. Due to the high specific temperature of water the effects only become apparent at the end of the channel. The small volume of water in the channel means that its ratio of water to length of embankment is different to the Danube's. This fact (together with the existence of a riverside forest) should lead to a higher input of allochthonous nutrients; the temperature periodics should be slightly reduced by the shade. This has not yet happened, as both the body of water and the embankment vegetation are still new; but such a development can be expected within a few years.

Because of the morphological conditions, the low and not very variable discharge leads to a low rate of shifts of sediment and movement of gravel and coarse sand within the MC (Nachtnebel *et al.*, 1995). The retention basin causes a reduction of the amount of sediments in the MC. The average difference compared to the Danube is, however, rather small. But the channel management prevents the extremely high short-term concentration of sediments that can be caused by Danube floods from entering the MC. There are therefore no sediment dynamics or flood sediments. During the summer months a reduction of the spectral radial transmission was registered, as the water progressed along the MC. That would seem to contradict the measured reduction of sediments, but can be explained by a change in quality (more light-absorbing particles) (Eitzinger *et al.*, 1995). Detailed examinations of these facts have yet to be carried out. The river bed was made of gravel, sand and stones. Initially, there were therefore no areas of fine sediments, but they soon developed, encouraged by sedimentation in the retention basin and other areas of low velocity.

<u>Biotic factors</u>

Initial position

Most of the organisms in the MC entered from the Danube. Where the MC branches off from the river, the water quality is quite good (class II), and there are plenty of species and individuals present. Small organisms were carried in by the current (passively), and possibly also by water-birds (Krisa *et al*, 1996). Only fish entered actively, especially via the mouth of the Rußbach. This possibility is limited for small organisms. Only the benthic evertebrates in the lower stretches of the MC showed an influence of the Rußbach (Krisa *et al.*, 1995).

In the semiterrestrial/terrestrial area the situation is completely different, and can only be mentioned in passing here. Research proved that vertebrates and dragonflies initially entered from areas surrounding the MC (Cabela and Girolla, 1994; Gaviria *et al.*, 1996). There is no proof of the MC functioning as an 'ecological corridor', with species spreading along it from the woods near the mouth of the Rußbach.

Due to the direct link from the Danube, the aquatic area was populated fast, and densities comparable to the river's were soon reached. The fact that the organisms were in a position to become established, proves the quality of the quasi-natural design of the MC.

Changes observed along the course of the MC

Organisms that drift into the MC passively find conditions in it that are new to them. For phytoplankton the velocity of the current is of primary importance. The organisms need time to reproduce, and during that time they drift on; so it is only in the lower stretches of the MC that plankton communities different to the one in the Danube are to be found (Krisa *et al.*, 1996). Since 1994 an increased presence of phytoplankton has been registered in the lower stretches, probably due to the reduced turbidity and the depth compared to the Danube. The higher summer temperatures towards the end of the channel, relatively isolated areas with reduced flow (bays etc.) and other factors also encourage this development (Krisa *et al.*, 1995).

Benthic organisms are in a different position, as their way of living gave them the possibility to establish themselves in the upper reaches of the MC. Consequently these areas registered an increased presence of macrophytes. The free flowing potamal section of the Danube has only a few macrophytes (4 monocotyledonae and 1 dicotyledonae) (Kinzelbach, 1994). In the MC there is an increased rate of both species and individuals. In 1994 twelve species of water plants were already found in the retention basin (km 3.0 - 3.6) (Wychera and Janauer, 1995). Here again, reduced depth and turbidity are the causes. The above results showed the correlation with the current. Further research on the role of fine sediments in populating the MC are planned. Generally speaking, the higher presence of plant organisms (especially macrophytes) should lead to a higher rate of photosynthesis compared to the Danube. The colonization of the zoobenthos is more complicated. Large-scale and small-scale changes of the determining situation (physical environment) are of importance. Locally influenced conditions determine the pattern of colonization more than ones that change as the distance from the beginning of the channel increases.

Fish are highly mobile, and so they colonized the MC differently. Larvae and young fish drifted in passively like plankton, whereas adult fish entered from the lower end (Schmutz *et al.*, 1994). So, in the initial stages, there was a greater presence of adult fish in the lower stretches than in the upper ones. This effect was reinforced by the functionality of the fish passes.

Development over time

Conservative patterns of development over time (Klötzli, 1993) were found among macrophytes and fishes. No such convergence was found among other aquatic populations. The distribution and succession of zoobenthos depended on the circumstances. New choriotopes (such as fine sediments) led to the development of new communities. The development of the vegetation on the banks is expected to lead to further changes (fallen leaves, shade). The yearly reduction of the maxima of phytoplankton biomass that has been observed does not fit into any pattern. A precise analysis of the interaction between phytoplankton, zooplankton, fishes, macrophytes and benthic algae has yet to be carried out.

A comparison is even more difficult due to the fact that, strictly speaking, the colonization of the MC is not a primary succession (Bick, 1989). First of all, there were preliminary test-floodings in some sections of the channel before the final one, so some preliminary colonization also took place, and secondly the final flooding swept lots of organisms straight into the MC. The channel could rather be viewed as a new branch of the Danube than a new flowing body of water., and its succession as secondary or mixed (Klötzli, 1993). The ecosystem of the Danube is in a fairly good condition. When the water is entering the MC several factors of the different physical environment are influencing its biocoenoses. The abiotic factors either continuously change of their own accord (e. g. sediments), or are changed by organisms (e. g. macrophytes as sediment traps). The initial situation determines the extent to which individual organisms develop according to this pattern. The succession of plankton is obviously secondary, that of macrophytes and terrestrial organisms clearly primary.

Modells of succession

Vannote's River Continuum Concept (RCC) was originally developed for flowing waters not influenced by man and has been criticised by some authors (e. g. Schönborn, 1992). The RCC states that the river ecosystems are independent of time and have no succession, as they are in the state of flowing equilibrium. Without entering this discussion, it is possible to state that this certainly does not apply to the MC, as it is a newly created body of flowing water. Here a succession, defined as changes caused by irregular events (Bick, 1989), is definitely taking place. On the other hand, the RCC was not created for new flowing waters. No climax is to be expected as a final point of this development (Clements's theory of climax, Allred and Clements, 1949). Marghaleff's term of increasing maturity would seem more appropriate (Schönborn, 1992). Conell and Slayter (1977) extend the term climax and use it as a synonym for halted succession. At the same time, the authors admit there is no proof of a community of sexually reproducing individuals reaching a 'steady state equilibrium'. The problem is clearly explained by Pahl-Wostl (1995): there is a difference between ecosystems in a state of equilibrium and perpetually unstable ones. Stability can be either static - the important factors do not change with time - or dynamic - the factors vary periodically or chaotically around a so-called 'basin of attraction'. Such 'basin of attraction' can also change periodically or be a chaotic attractor.

The following points can be made about the development of flowing waters: frequent changes of the determining situation and local disturbances (which mean an unstable environment) lead to a dynamically stable ecosystem (Klötzli, 1993). There can not be any static equilibrium or final point. The development of a system's attractor' can be expected (Pahl-Wostl, 1995). The variability of this attractor is determined by changes of the abiotic factors and the retarded reaction of organisms during the development, i. e. changes of the biotic factors. The complete dependency of the MC on the Danube and the different possibilities of human management lead to a high variability. There can therefore be no linear development and no mechanically-linearly modelled forecast (Pahl-Wostl, 1995). But it should be possible to indicate attractors or 'fields of attractors'. This is one aim of the final analysis of this research program. Projects of renaturalisation seem to need general aims which allow natural and specifically local development more than detailed targets.

Quasi-natural and natural flowing water

The final question addresses the basic differences between the MC and natural flowing waters and the relevance of these statements. Future work will deal with these subjects and the possible use of the results of this research. One of the most important differences is the lack of discharge dynamics. A comparison makes that clear: above the mouth of the MC the Rußbach has an MQ of 0.2 - 0.3 m^3/s, a HQ10 of 11 m^3/s and a HQ100 of 26 m^3/s (the use of a retention basin prevents flooding below the mouth of the MC) (Errichtungsgesellschaft Marchfeldkanal, 1992). The necessary human management and the low variability of the discharge regime cause the complete absence of natural sediment dynamics. This should lead to consequences for the embankment vegetation and the balance of nutrients when the MC is further developed. It is not easy to prove that the lack of discharge dynamics has immediate negative effects on the biocoenoses (cf. the 'immediate disturbance theory'; Lampert and Sommer, 1993). There have been some experiments with higher rates of discharge, and more are planned (Muhar *et al.*, 1996).

Temperature (often regarded as the second most decisive factor in flowing waters after the current, e. g. Schönborn, 1992) is a further differentiating factor. The possible extent of yearly and daily dynamics in allow-lying, unshaded river is shown by data measured in the lower stretches of the MC. The more stable conditions in the upper stretches seem unnatural - but only because they are not caused by shade (like in streams in woodlands), but by the Danube.

The artificial sealing of the bed was expected to have immediate consequences for the hyporheic interstitial. It is possible to imagine the development of specific communities in the 60 cm gravel bed above the sealing, as this is a different biocoenosis to the ground water fauna (Schönborn, 1992). On the other hand continuous sedimentation could lead to the filling of the interstitial spaces and so to a natural sealing layer above the synthetic one. This often happens in low-land rivers. Natural sediment dynamics (with a high ratio of fine sediments in the bed) would help such a development towards a natural low-land river, but do not exist. We can only wait and see if the present rate of sedimentation produces a similar effect.

We can already see the importance of the diversity of structures in the water for the communities. Different widths and depths were included in the plans (Errichtungsgesellschaft Marchfeldkanal, 1992), but the bed was constructed rather uniformly, and there is not yet enough vegetation on the embankments, so, compared to a natural river, there are only a few different choriotopes. The local formation of the fine sediments was sufficient to lead to an increase in the number of benthic evertebrate species. Structures such as branches or large roots in the water are still missing. The number of different structures should increase with succession. The MC can therefore hardly be compared to a natural river with its much greater degree of maturity. The future development will be determined by upkeep (cutting embankment vegetation, removing branches etc. from the water). Careful timing and small-scale, local action are important. Human interference should be kept to a minimum possible, e. g. repairing damage or removing serious obstructions.

The striking difference is that the MC gets its water from a big river. Therefore it is highly improbable that it will ever develop into an independent natural low-land river ecosystem. It will always remain what it is - a special case.

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WETLANDS IN NIGERIA: CHARACTERISTICS AND AVAILABLE OPTIONS FOR THEIR UTILIZATION AND CONSERVATION

by

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ABSTRACT

Arising from inadequate technological development, and an unpredictable and severe climate in some cases, the use of wetlands for sustainable development in Nigeria is still beset with some setbacks. Partly for the stated reasons and partly on account of the uniqueness of environments that are yet to be fully understood, the full potentials of the wetlands remain largely unexplored and unexploited.

The two extremes of Nigeria (the North and the South) have wetlands that are as different in ecology as their climate. The northern wetlands reflect characteristics of the Sudano-Sahelian zone (SSZ) while the southern are distinctly deltaic. Although a few similarities do exist in their potentials, approach to their exploitation must consider environmental peculiarities that exist in each area.

At present emphasis is on developing the agricultural potentials of wetlands in the SSZ to supplement production from other areas. This is in a bid to attain adequate food supply in, and increase the revenue generating base of Nigeria through exportation of food and cash crops. These lofty goals, however, are still to be achieved. The deltaic wetlands currently provide the bulk of the mineral (oil) resources that give the country relative economic buoyancy. The pursuit of this is also being done to the complete exclusion of the identification and development of other natural resources. The region could be put to agriculture, recreation, games reserve and other tourism-enhancing uses to further boost foreign exchange earning of the country.

This paper presents the diversity of wetlands in Nigeria, their characteristics and the need for similarity in investigations irrespective of location. Such statutory requirements also provide a means of acquiring basic information and understanding that are essential in achieving effective conservation.

KEY-WORDS: Sudano-Sahelian Zone/Niger delta/Resources/Potential/ Characteristics/Environmental impact/Water/Oil/Agriculture/ Exploitation degradation/Options

INTRODUCTION

The text of the Ramsar Convention on the Conservation of Wetlands of International Importance signed in 1971 provides a clear picture of what constitutes wetlands. It defines wetlands as areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of maritime water, the depth of which at low tide does not exceed six metres. This view underscores the importance of maintaining hydrological and ecological balances in sustenance and conservation of wetlands. FAO (1976) estimated that wetlands account for 66 x 10^6 ha in the humid tropics, 9% of African's land area and 7% of Nigeria's 9.3 x 10^5 km^2 land area.

Wetlands in Nigeria have played significant roles in the socio-economic life of the people. This has been through the tapping of the immense natural resources that abound in those unique environments for food, recreation, transportation, shelter, and hydrocarbon, the last currently contributing at least three-fold the total from all other sources to the nation's foreign exchange earning.

Climate and ecology have combined to initiate and develop wetlands that are distinctly different in the northern and southern (coastal) fringes of Nigeria. As a result, while wetlands in the southern fringe have deltaic characteristics, those in the extreme north, reflect the harshness and general unpredictability of that environment. Figure 1 shows some wetlands, drainage system, storages and isohyets in Nigeria. Summary is presented on Table 1.

In terms of uses and development, considerable attention is given to the tapping of only the hydrocarbon resources of the Niger Delta wetlands to the complete neglect of their agricultural, recreational, educational and scientific potentials. The semi-arid wetlands of Nigeria in contrast are only exploited for agricultural production, marginally for recreation and only recently did oil prospecting initiated by the granting of oil prospecting licence to a few national and multi-national companies. The Baturia Wetlands Reserve that straddles Jigawa and Yobe States is an example of attempts at using northern wetlands for purposes other than agricultural in that part of the country.

Initially, little conscious effort was made to ensure wetland conservation while exploration and use were undertaken throughout the country. Global environmental awareness and the ensuing campaign to institutionalize multi-faceted sustainable and environment-friendly development, has induced government at the centre to create agencies that monitor activities that are related to exploration and production in the country. The foremost agency charged with enacting and

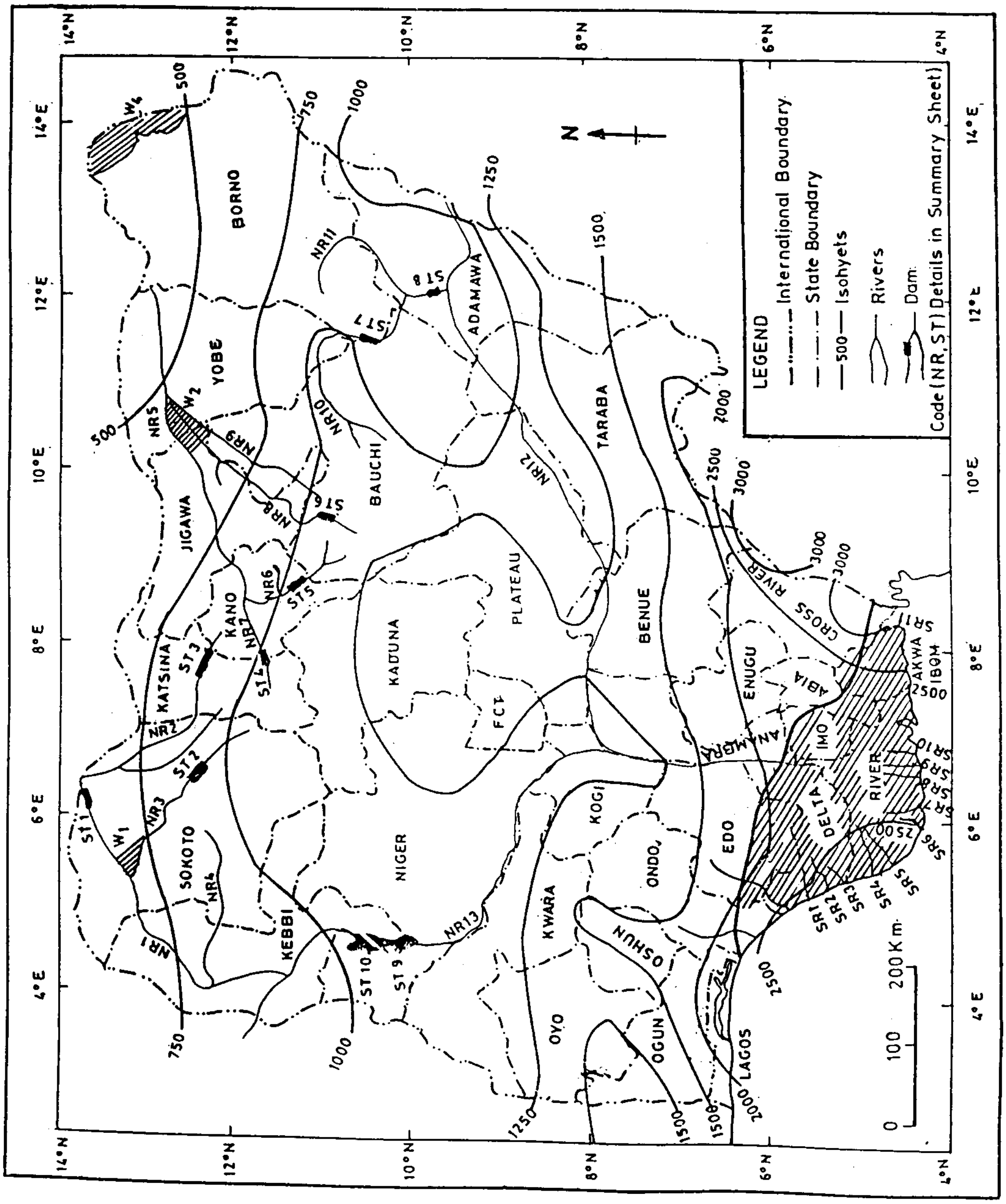

Figure 1: Map of Nigeria showing States, isohyets, some river systems and associated wetlands

enforcing compliance with environmental guidelines is the Federal Environmental Protection Agency (FEPA). Deriving partly from environmental sensivity, partly from relative importance in the nation's overall economy and partly from local, national and international perception of previous activities and their impact on the environment, more focus is on the Niger Delta wetlands than those elsewhere in the country.

Table 1: Summary Sheet for Figure 1

Rivers

Wetland		Sudano-Sahelian Zone		Niger Delta		Dams/Storages	
Code	**Name**	**Code**	**Name**	**Code**	**Name**	**Code**	**Name**
W1	Sokoto-Rima floodplain	NR1	Rima	SR1	Benin	ST1	Goronyo
W2	Hadejia-Nguru floodplain	NR2	Bunsari	SR2	Escarvos	ST2	Bakolori
W3	Deltatic floodplain	NR3	Sokoto	SR3	Forcados	ST3	Zobe
W4	Lake Chad	NR4	Zanfara	SR4	Amo	ST4	Challawa
		NR5	Hadejia	SR5	Dodo	ST5	Tiga
		NR6	Kano	SR6	Sangana	ST6	Kafinzaki
		NR7	Challawa	SR7	Brass	ST7	Dadin-Kowa
		NR8	Jama'are	SR8	Sombreiro	ST8	Kiri dan
		NR9	Misau	SR9	Bonny	ST9	Jebba
		NR10	Gongola	SR10	Imo	ST10	Kainji
		NR11	Hawai	SR11	Cross River		
		NR12	Benue				
		NR13	Niger				

This paper highlights the need to accord equal seriousness to identifying potential impacts of all projects, irrespective of location on the receiving wetland environment in particular as a way of achieving wetland conservation.

WETLANDS IN SEMI-ARID NIGEIRA

Wetlands in semi-arid Nigeria have been formed under severe climate. Average annual rainfall is between 500 and 1100 mm the entire total being received in 7-10 intense storms. Water balance in the region is in the deficit for 8 months of the year (Fig.2a). For every 100km, there is a decrease of >250 mm in total annual rainfall, while annual total evaporation has a reverse trend northeasterly. Two wetlands in the Sokoto-Rima and the Komadugu Yobe Basins in the North West and North East Nigeria, respectively, provide characteristics that typify wetlands in semi-arid Nigeria.

Sokoto-Rima Wetlands

The Sokoto-Rima floodplain is underlain by alluvium which is up to 1.5km wide in some places and is more than 20m thick. Topography is typically flat, subject to flooding and spotted with continuous, dry channels and level terraces. Soils are shallow and have developed on the micaceous Kurukuru and Gande deposits are fine and heavy textured.

Major vegetal cover comprises trees, sedges, herbs, creepers and floaters. These include *Metrogina inamis*, *Diasperus mesipiliformis*, *Typha spp* (reed-used for mats), *Cyperus diaformis*, *Sporobolus pyramidales*, *Hypoxy spp* and *Nymphia lotus*. These have economic medicinal and constructional importance in the lives of the local people.

Wetlands in Sokoto and Kebbi States are mainly put to agriculture (crop production and fisheries) and water supply for irrigation and domestic purposes. An estimated 614,000 ha of wetlands in the Sokoto/Kebbi States has significant potential for the development of shallow groundwater (<12m deep) for small scale irrigation. Over 60% of this area is on the floodplains of the Niger and Sokoto-Rima Rivers. Another 20% is on the floodplains of several medium-sized rivers which include Rivers Sokoto, Zamfara, Gagere, and Bunsuru that are underlain by sedimentary bedrock. The remaining 20% is on small-sized (100 to 3,000 ha) floodplains. Based on such potentials, the Sokoto-Rima wetlands are classified into major, intermediate and minor (Table 2). Yield varies between 50 to 200 lpm.

The productivity of these areas is four to five times that of cultivated upland because they have an extended growing season that lasts for several months after the cessation of the rains. The relatively more fertile floodplain soils are also put to recession farming with millet, sorghum, cowpea, onions being grown as sole or multiple crops. Even at the peak of the dry season, sugarcane still thrives int he intermediate and major fadamas.

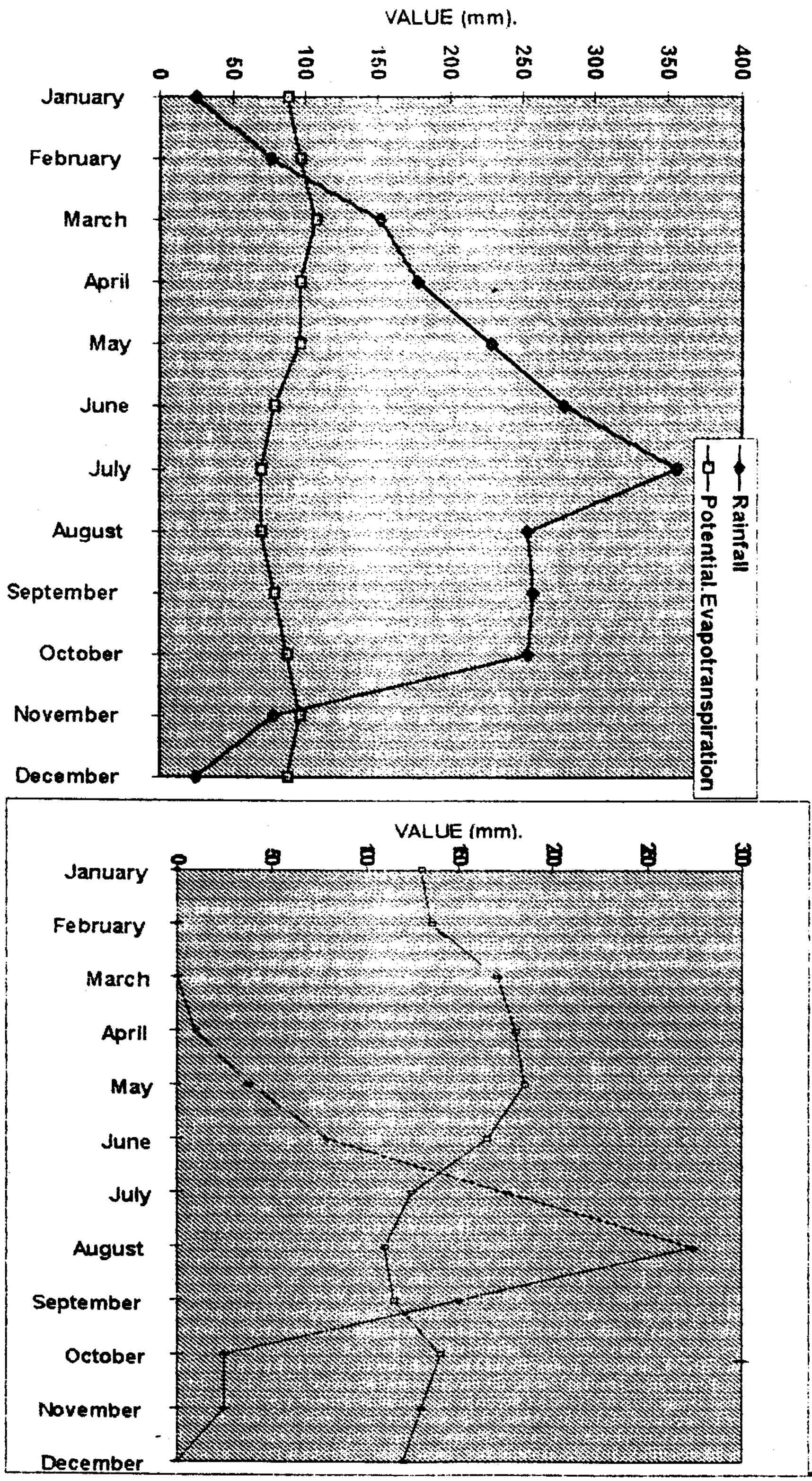

Fig. 2 : Water balance of (a) Niger Delta area, (b) Nigeria's Sudano-Sahelian zone.

Table 2: Groundwater Characteristics of Wetlands (Fadamas) in Sokoto/Kebbi States in Semi-Arid Zones of Nigeria

Class	Estimated Area (10^3 ha)	Location (Floodplain)	Geomorphology of Aquifers	Source of Recharge	Transmissibilities (m2/day)	Storage Coefficient
Major Wetland	178	Sokoto-Rima	Alluvial sands and fine gravels	Mainly surface water infiltration and some contribution from upward (capillary) flow from bedrock formations	200-3,000	1×10^{-2} to 1×10^{-4} (semi-confined)
Intermediate	52	Sokoto, Zamfara, Bunsuru, Gulbinka	Alluvial sands, gravels and bedrock sandstones	Upward flow from bedrock formation and infiltration	200-1,500	1×10^{-2} to 1×10^{-3}
Minor	50	Sokoto, Zamfara, Sheila Kondawa, Gulbinka	Alluvial sands, bedrock sandstones	Upward flow from bedrock formation and little infiltration	200-400	1×10^{-1} to 1×10^{-3} (unconfined to semi-confined)
Total	280					

Source: Oyebande and Balogun (1992).

The Hadejia-Nguru Wetlands

The Hadejia-Nguru wetlands are found within the floodplains of the diverse Kamadogu-Yobe river system that covers about 1.5×10^5 km^2. The climate is semi-arid and the wetlands have formed where the Hadejia and Jama'are rivers meet. This union has yielded an undefined secondary drainage system with multiple river channels that either permanently or seasonally experience flow.

The geological and geomorphological composition of the Hadejia-Nguru floodplain has been given by Adams et al (1993). The floodplain has weakly developed and halomorphic soils of deltaic alluvium. Clayey, alluvial non-leached ferruginous dune soils are to be found to the east and west.

Vegetation consists of a variety of tree, scrub and grass communities that are essentially identified with three locales of the wetlands (Table 3).

Uses to which the Hadejia-Nguru wetlands and their resources have been put are mainly wet season rice cultivation, flood recession agriculture, dry season irrigation, artisanal fisheries, graze-land for livestock, small-scale lumbering for fuelwood, and fodder for horses.

Status of Semi-Arid Wetlands

Wetlands in semi-arid Nigeria reflect impacts of both natural and artificial origins. Many attempts have been made to use the wetlands without endangering the environment. Among organisations that have attempted developing the wetlands are the different northern Nigeria based River Basing Development Authorities (RBDA's), Wetland Conservation projects, Federal and State Ministries and Parastatals charged with agricultural, water resources, and rural development responsibilities, and international organisations. Efforts so far, are still to have the desired impact.

Over the years, wetlands in the north, in addition to being used mainly for survival by individual farmers and fishermen without a coordinated institutional advisory and supervisory contribution, have also taken hard knocks from climate. Although the actions of the individual farmers essentially wreak localized impact, the aggregate impact can be enormous. This is further heightened by the severe consequences of current climate variations and change on wetlands in that region. While the human impact can relatively be easily mitigated against and ameliorated, current climatic trends give rise to a lot of concern about their use and conservation. A few examples suffice to underscore the need to correct the misuse of these naturally unique environments.

The essential role of water in the sustenance of wetland characteristics is not in doubt. Also incontrovertible is the fact that the only significant source of water in this context is rainfall that replenishes flow in the rivers on whose floodplains the wetlands have formed.

Table 3: Vegetation Types in Hadejia-Nguru Wetlands

Locale	Vegetation Type
Hadejia-Katagum floodplain	_Guiera_ shrub savanna with remnants of _Acacia senegal_ and _Combretum glutinosum_ on dunes; grasslands on which _Schizachyrium exile_ thrive; swamp grassland has _Vetiverial_, _Andropogon gayanus_ and _Echinochloa sp_ while savanna tree species _Acacia_ and _Balanites_ are also found
Wazagal Plain	Woodland comprises largely of _Acacia_ and _Balanites_ with _Guiera_ or _Ziziphus spp_ of scrub in areas relatively better drained; _Andansonia_ may exist on cultivated areas, parkland.
Nguru Plain	Wooded sandy plains and dunes have _Guiera_ and _Acacia albida_ while grasslands comprise _Schizachyrium exile_ and _Aristida spp_; Spill plains have combinations of _Acacia_/_Balanites_ or _Ziziphus_/_Acacia_ with _Vetiverial_/_Andropogon spp_ on alluvial grasslands. Only _Echinochloa spp_ inhabit swamp grasslands.

The effect of past droughts on surface and groundwater availability in the north has been reported (e.g. Oyebande and Balogun, 1995; Shamonda, 1990). The 20-year (1969/70 - 1989/90) declining trend in the total annual runoff yield into Kainji Lake one of many hydrological evidences and can be linearly represented by:

In $Y = 0.04x + 3.7$, $(r = -0.69)$

where y is time in years and x total annual runoff into the lake.

The RBDA's are charged with some functions which include construction and maintenance of dams and other storages that ensure adequacy of river flow at all times. Oyebande and Balogun (1995) reported that more than 30.34 x 10^9m^3 of water is stored in 162 dams built for irrigation, water supply, hydropower, fisheries, navigation, recreation and flood control. These have had various impacts on existing wetlands, some of which are:

* the construction of Tiga dam on a tributary of the River Hadejia in the 1970's worsened low rainfall effects of the last 2 decades resulting in both vertical and horizontal shrinkage of the wetland;
* while flow at Gashua for 1964-1973 averaged 1386 x 10^6m^3, post-Tiga dam flow averaged 792 x 10^6m^3, indicating a 23% Tiga dam effect in addition to another 23% drought effect on flow;
* statistics also show that between 1970 and 1987, there was reduction in magnitude and duration of floods, contraction of area inundated and widespread shortage of water in the lower parts of the Yobe basin;
* in the Hadejia-Nguru wetlands, Hollis et al (1993), reported that since 1974, area inundated is only 66% of that of the 1950's. For example, area inundated was 700, 910, 962 and 552 km2, in 1987, 1990, 1991, and 1992, respectively.

Table 4: Evaporation Losses from Dam Reservoirs in Semi-Arid and Arid Zones of Nigeria

Dam (1)	Zone	Active Capacity (10^6m^3) (2)	Evaporation Losses (10^6m^3) (3)	Loss Capacity Ratio (%) (4)=(3)/(2)
Jibiya	Sahel	121	36.4	30
Zobe	"	170	54.0	31
Bakolori	"	403	96.0	24
Goronyo	"	933	280.0	30
Kontangora	Sudan	200	39.0	20
Kiri	"	325	132.0	41
Challawa Gorge	Sahel	904	120.0	25
Tiga	Sudan/Sahel	1845	214.0	12
Kafin Zaki	Sudan	2500	300.0	12

Table 4 also presents the tremendous losses of water to evaporation from storages in the semi-arid and arid environments in Nigeria. The Bakolori and Goronyo dams that should play significant roles in the sustenance of wetlands in north west Nigeria lose 24 and 30%, respectively, of their storage capacities.

WETLANDS IN NIGER DELTA, NIGERIA

Location

The Niger Delta covers about 25,000 km^2 and encompasses Rivers, Delta, Akwa-Ibom and Cross River States. It also extends 200 nautical miles into the sea.

Climate

The climate is influenced by the Intertropical Discontinuity (ITD). Number of raindays on the average is about 200 annually. This gives rise to a moist monsoon climate for a large part of the year. Total annual rainfall is about 2600 mm. Relative humidity is 85% on the average throughout the year. Minimum and maximum daily temperatures are about 27 and 30°C, respectively, indicative of a low diurnal fluctuation. As a result, evapotranspiration is about 1040 mm annually and combines with rainfall to give a surplus water budget for most of the year as shown on Figure 2b.

Wetland Types and Uses

The influence of climate and other environmental factors have led to the formation of two basic wetland types; mineral and organic. While the mineral wetlands are formed in areas inland, the organic wetlands are brackish and are found at locations influenced by tides. The floodplains of the numerous creeks and creeklets criss-crossing the Niger delta act as receptors of sediments and diverse chemical substances of even more diverse concentrations.

This has enhanced the roles of (i) a permanently highly reducing environment, (ii) a sufficient supply of sulfur, and (iii) high organic matter content as factors that have influenced the conversion of an appreciable fraction of Niger Delta wetlands into potential and full-blown acid sulfate soils. Various workers (e.g. Anderson, 1967; Ojanuga and Lekwa, 1984; Ojanuga et al, 1984; van Breemen, 1980; and Balogun and Oyebande, 1994) have given the characteristics of these special wetlands in detail. They are given special attention in this paper because of their high areal coverage and risk of acidity intensification and general irreversible modification that could result from oil-related activities that are prevalent in the Niger Delta.

Niger Delta wetlands have been intensively and extensively studied. Such studies of recent are pre-requisites for oil exploration projects as demanded by FEPA. Little use is made of the wetlands for agricultural purposes.

Operations and all forms of developmental activities in the oil industry in Nigeria are supposedly regulated by specific laws, guidelines and standards. Examples of such include:
*the Petroleum Act of 1969 which empowers the Minister of Petroleum Resources to promulgate regulations for pollution prevention;
*the 1991 Environmental Guideline and Standards for the Petroleum Industry which makes the Environmental Impact Assessment (EIA) study and its report not only mandatory;
*the 1992 Federal Environmental Protection Agency Decree that requires that all new major development activities must be preceded be an EIA report.

Current Status of Niger Delta Wetlands

The many studies undertaken to generate data and information for EIA reports have immensely contributed to characterizing the Niger Delta wetlands. Table 5 shows that virtually all impactable components of the environment have been studied in as many locations as there are existing and/or proposed oil prospecting and mining activities.

Similarities exist in the environmental components of the Niger Delta wetlands. Studies undertaken along the floodplains of Rivers Benin, Escravos, Bonny, Brass, Forcados, and associated creeks and creeklets reveal the generalized characteristics of the environment given on Table 6.

Oil companies that are involved in oil exploration activities within the Niger Delta wetlands include Shell Petroleum Development Company, Chevron, Agip, Texaco, Mobil, Nigerian National Petroleum Corporation, Elf and Ashland. A few indigenous companies are also currently proposing going into oil ventures that include petroleum refinery.

Both beneficial and detrimental impacts of oil-related activities have registered on the receiving wetlands. Beneficial impacts have more or less been on socio-economic components. These include opening up and tarring of primary and secondary roads, provision of educational and health facilities as well as employment of unskilled, semi-skilled and skilled indigenes, often on temporary basis during site preparation and facility construction phases. Such community assistance projects are usually project-linked.

Wetland exploitation in the Niger Delta also have deleterious physiological and psychological effects particularly on the physical environment. For example, the construction of the Utorogu Gas Plant required the sand-filling of the generally low terrain that resulted in intensified flooding in some areas. The exacerbated waterlogging modified micro-hydrology with resultant adverse effect on vegetation particularly rubber trees. Other negative impacts associated with oil production include:

* localized increase in temperature from gas flaring;
* emission of pollutants (NOx, SOx, CO_2, CO);
* scorching of vegetation by spilled oils, drilling muds and other effluents surface water;
* loss of wildlife habitat and species to land clearing and noise;
* disturbance of fish spawning grounds;

Table 5: Impactable Components of the Environment and Associated Impact Indicators Investigated in the Niger Delta

Components of Environment	Impact Indicators
Climate	Humidity, temperature
Air Quality	Particulates NOx, SOx, CO_2, THC
Water Quality	Solids (Ds, Ss), turbidity, toxicity, eutrophication
Hydrology	Drainage/discharge, hydrologic balance, sedimentation, shoreline erosion
Hydrogeology	Groundwater level and quality
Soil/Land Use	Erosion, subsidence, farming, hunting, recreation
Ecology	Diversity and abundance of aquatic and terrestrial flora and fauna
Fisheries	Productivity, diversity and abundance fish kill
Archaeology	Cultural relics, sites
Noise & Vibration	Day-time disturbance, hearing loss, communication interference
Socio-economics	Population, income, settlement pattern, health, safety and security
Wildlife & Forestry	Conservation areas, habitats, sensitive areas.

Table 6: Generalized Niger Delta Characteristics

Environmental Component	Description
Soil	General acid (pH 4.5-6.5), loamy texture, permeable, low to medium fertility, high organic matter (>8%) wide and C/N ratio, high sulphur, toxic levels of micronutrients (Fe, Zn, Mn) as a result of prolonged waterlogging, low heavy metal contrations (Ni, V, etc), low total hydrocarbons
Water Quality	Surface water is slightly acidic to alkaline (pH 5.3-8.5), dissolved oxygen is 4.0-6.0 mg/l, high salinity (10-30%), groundwater is slightly acidic to neutral (pH 6.5-7.2), total solids, hardness, calcium, nitrate, iron, magnesium and high sulphate and sodium. Total hydrocarbon is low in both (1.0-10.0 mg/l).
Air Quality	CO_2 (3.0-25 mg/m^3); NOx (0.1-5.0 mg/m^3), SOx (0.0-0.5 mg/m^3), NH_3 (0.1-0.5 mg/m^3), VOC (0.0-0.01 mg/m^3), heavy metals are 0.0-3.0 ppm; O_3 is below detectable level
Vegetation	Rhizophora spp; Dabalgia spp. rainforest species (Dalium guineense, Icacina trichanta, Elaeis guineensis, Mangifera indica, Ananas comosus, Chromoleana odorata, among several others
Fish	Chrysichthys spp, Pseudotolithus spp, Heterotis spp, Gymnacus niloticus, Synodontis spp, Clarias spp
Land Use	Settlement, agriculture (crop production, fishery), water bodies, forestry
Geology	The region is plains-dominated with extensive basal rocks of loam, sandstones, shale and clayey alluvial materials. Geologic structure belongs to Pliocene-Pleistocene and Oligocene-Miocene periods around the Delta State. Around Edo State is the Tertiary Benin Formation. The coastal Plain Deposits around Akwa Ibom and Cross River States are a mosaic of marine, deltaic estuarine, lagoonal and fluvo-lacustrine material.
Socio-economy	Population density is generally low, houses are of concrete, red earth or laterite. Occupation is mainly fishing, farming, hunting and trading in diminishing order of importance. Others are local gin distilling, teaching and craftsmanship. Annual earnings are about N15,000 on the average. Infrastructural facilities (roads, potable water, electricity) are mainly provided by oil companies with some from Oil Mineral Producing Area Development Commission and State governments. Governance is traditional and usually comprises Elders Council, and Youth Council both integrated to form the Executive Council that is responsible for the day-to-day administration of the town or village. Religion is predominantly Christianity and traditional.
Wildlife	Mammals include: Thryonomys swinderianna I, Potamochoerus porcus, Tragelaphus scriptus, Sylvicapra grimma, Dendohyrax arborea, Viverra civetta, Funisciurus pyrrhopus, Cercopithecus noma, Mungus mungo. Birds found include Neophron monochus, Mulvus migrans, Gypohierax angolensis, Streptopelia senegalensis, Turtuc tympanistria, Ciccaba woodfordi, Macrodipteryx longipennis, Turdus pelios, Prinia subflava, Passer griseus, Pycronotus barbatus, and Merops albecollis.

* probable mutation of aquatic life forms (e.g. crayfish) resulting in diminished sweetness;
* occasional blow-out of oil wells;
* vibration-induced cracking of house walls;
* coastal/bank erosion; and
* acid rain from emissions from manufacturing, oil and other energy-related industrial activities which although are individually insignificant, become devastating when pooled.

Although these issues are real, some exaggerations are introduced in a few cases either to whip up sentiment or to set the stage for demand for compensation from the oil companies that are seen to be financially very solvent. In general, the pollution of the Niger Delta wetlands by oil production accounts for 60% degradation (NEWSWEEK, 1995).

The foregoing highlights the dire consequences of wetland utilization in deltaic environment where social, economic and political strife alongside climate variations and change increase the susceptibility of the environment to degradation.

CONCLUSIONS AND RECOMMENDATIONS

Wetlands in the Sudano-Sahelian zone and the Niger Delta areas of Nigeria reflect the effects of climate (temperature, precipitation, evapotranspiration), parent material (mineralogy, texture), organisms (vegetation, animals, microbiology), topography (slope, altitude), and time. These have given them distinct characteristics that are still to be fully studied and understood.

The uses to which the wetland are currently put are informed more by survival than sustainable and environment-friendly developmental needs. This is apparent in the fact that different organisations seem genuinely responsible for their use and development. Efforts at SSZ wetlands development have been made by the RBDAs (e.g. Sokoto-Rima, Hadejia-Nguru, and Chad Basin Development Authorities) while FEPA is at the forefront of attempts at ensuring sustenance of wetland quality in the Niger Delta. Statutorily, FEPA has the entire nation as its jurisdiction. Political and economic considerations as well as perceived low level of susceptibility to degradation are factors that are responsible for FEPA's relative non-interest in SSZ wetlands utilization.

Conservation of SSZ wetlands is inhibited by natural and human factors. The impact of the former is worsened by water management measures that:

* are designed using only wet period data;
* are designed without proper evaluation of environmental impact.

Degradation of Niger Delta wetlands arises from petroleum prospecting which has significantly rendered natural contributions obscured. The supervisory and policing role of FEPA is yet to have desired impact, probably because it has just come into the mainstream of environmental monitoring.

For a national wetland use that recognises conservation as the only option of sustainable development, the lapses identified must be corrected. This can be through:

* studies of all wetlands irrespective of where located must be as comprehensive as statutorily required;
* institutionalization of environmental monitoring programme during which sensitive quality indices of the environment are routinely studied for mitigation and/or prompt amelioration of adverse impacts;
* the statutory requirement that environmental impact assessment report of proposed project must be approved by government (based on its environmental consequences) prior to project execution must be strictly enforced;
* strengthening of FEPA through adequate provision of human and material resources for effective national coverage.
* adequate funding of non-governmental-organisations (NGOs) that have proved to be reliable `watchdogs´ for wetland conservation by Nigerian governments as well as international organisations. The contribution of the Nigerian Conservation Foundation in this context is particularly commendable.

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ENVIRONMENTAL IMPACT ASSESSMENT METHODOLOGY BASED ON A RELIABILITY KNOWLEDGE MANAGEMENT

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ABSTRACT

The rehabilitation of hydroelectric stations built in urban areas at the beginning of the century is one of the current concerns of their administrators. The activities conducted during the repair work make up a potential source of repercussions on the natural environment.

In order to optimize the evaluation process of these impacts, it is necessary to assimilate the environmental concerns to every stage of the project. The difficulty in bringing together all the experts possessing the required knowledge complicates the smooth progress of the process. A computerized decision-making help system, that gathers the knowledge stemming from various expertise, makes up a very interesting tool.

To provide a realistic analysis of the studied situation, such a system must simulate the reasoning done by the experts and take into account the restraints to which they are confronted. They must, among other things, compose with incomplete and more or less reliable data. A decision-making help system must therefore consider the uncertainty inherent to the data used and include management mechanisms for this uncertainty.

We therefore propose a prototype of an expert system capable, in reproducing the reasoning mechanisms of experts, to identify the effects of the repair work of small hydroelectric stations and to proceed to a first evaluation of these repercussions by considering the quality of the information used and obtained.

KEY-WORDS: hydroelectric stations - rehabilitation - environmental impacts - evaluation - expertise integration - decision-making help - computer system - knowledge quality management

INTRODUCTION

A project for developing a computerized decision-making help system was conducted in 1993 in collaboration with Hydro-Quebec. The objective of this system was to proceed to the identification and a first evaluation of the apprehended environmental repercussions during the repair work of the hydroelectric stations. During this project, the required knowledge in the various expertise fields related to the repair projects were identified and structured by studying the work method and reasoning mechanisms of experts (Podesto, 1993).

The matrix approach was retained, for its knowledge structure, because of its use at Hydro-Quebec. It consists of connecting the planned activities and their potential impacts inside bidimensional tables. The repercussions are characterized by their degree of importance. These matrices are regularly evaluated and revised by the experts. For the development of the system, thirteen matrices were built, according to the levels of work, then computerized.

The structure of the environmental matrices allows a global and quick vision of the stakes involved in a project by presenting the impacts identified in a simple and concise manner. However, it does not allow, during the use of the software developed, to visualize the network of intermediate links woven by the expert's reasoning by presenting only the results, in other words the final repercussions on the components of the environment (Robert and Podesto, 1993).

Furthermore, once the matrices are integrated to the computer system, it is difficult to bring about alterations; we must then evaluate the repercussions generated by the elements to be added, establish new matrices that include this additional information and integrate them to the computer structure. The matrix approach is therefore not transparent and progressive enough to efficiently meet the demands of the environmental evaluation. A new method based on a more flexible knowledge structure and allowing to consider the notions of quality of knowledge was therefore developed.

NEW APPROACH

The environmental matrices only deal with the ends of the reasoning followed by the experts during the evaluation of environmental impacts of projects, such as the identification of data and the evaluation of final repercussions. The main objective of this new method of knowledge structure is therefore to reproduce the entire reasoning mechanism. It can then be adapted to situations different from the ones considered at the time of development, for example the addition of unplanned work.

The first reasoning step involves the characterization of the site to be studied. The available data is then identified and its degree of certainty is estimated. Once this characterization is done, the experts evaluate the potential effects, according to the planned activities, by applying laws representing the phenomena affecting the environment's components. The selection of these laws is done according to the availability of data.

During this process, the experts must deal with varying reliability and precision data and laws producing results of equally varying quality. They must therefore constantly adapt their analysis to the condition of the knowledge they possess. The management of the quality of information for that matter makes up one of the main elements of an

Evaluation Of The Quality Of Data And Laws

The data identified during the first step represents, with more or less accuracy, in other words in a more or less reliable manner, the technical and environmental components tied to the evaluation of impacts of projects. Before using them, it is important to know their quality from which depends, in part, the quality of results provided by the laws. We therefore proceed to an estimate of their reliability. During this evaluation, the use of qualitative mechanisms, coming from the expert's experience is recommended instead of probabilistical models.

In a similar manner, the laws selected describe the physical or biological phenomena, originating from the repair activities, to various degrees of precision that express, in fact, the reliability attributed to them. This reliability of laws exercising an influence on the quality of information drawn from their application, it is necessary to evaluate it and consider it, just like the reliability of data. An evaluation scale was set up to evaluate the reliability of data and laws. It is a qualitative scale, on which the degrees are expressed with linguistic labels such as Very High, High, Medium or Low. The degree of reliability given to a data or a law measures the accuracy with which it represents reality. The reliability of laws is a constant characteristic for each law and it is integrated to the knowledge bases. The data's reliability is rather determined, by the user or by the system, according to the knowledge we have of the components it represents.

Evaluation Of The Quality Of Results

After applying a law, we must proceed to the evaluation of the quality of the result obtained. In fact, due to the uncertainty inherent to the data and the vagueness of laws, the responses provided are themselves more or less certain. The process consists in determining the influence of the laws' and data's reliability on the quality of responses.

A data's influence on a law's response is what we call the sensitivity of the law to the data. It expresses the measure according to which the response is affected by a variation of the data. By transposing this definition to the relation between the reliability of data and the reliability of response, we obtain the following principle: the reliability of data influences the reliability of response proportionally to the sensitivity of the law to this data.

Therefore, the more sensitive a law is to a data, the more the influence of the data in the law is important and the higher its reliability must be. The application of this concept to the context of evaluation of environmental repercussions enables to determine the quality of knowledge that we must possess on the repercussions according to the vulnerability of environmental components they affect. An evaluation scale of the laws' sensitivity has also been defined. This scale is made up of three degrees of sensitivity, that is High, Medium and Low.

The evaluation of the responses' reliability starts from the laws's reliability. In fact, if we admit that a response cannot be more reliable than the law that provided it, the degree of reliability of the law represents therefore the maximum degree that a response may reach. This maximum reliability is reached if all the law's data is also of maximum reliability. If the reliability of certain data is inferior, the reliability of the response is also diminished proportionally to the law's sensitivity to the data involved. The response's reliability is therefore determined by the reduction of the reliability, with respect to the one of the law, due to the uncertainty on the data.

expert's reasoning, no matter what situation he is confronted to. The new system therefore proposes an original methodology for the management of knowledge quality of which the fundamental principles are inspired by the approach taken by the experts. It consists of pragmatic concepts, for the most part qualitative, relying on good sense more than numerical methods.

Management Methodology Of Knowledge Quality

Data Identification

The first step, in the evaluation process of environmental impacts, consists in identifying the available data characterizing the work to be done and the site where the work will take place. This data concerns the technical components, that is the activities to be conducted and the equipment and installations to be rehabilitated as well as the environmental components which include the biophysical properties of water, the hydraulic system, the waterway's bed, the animal and plant communities, their habitats, etc., likely to be affected by the interventions. Obviously the higher the number and quality of this data, the easier and more reliable the identification and evaluation of the repercussions. A data base containing all the information thus acquired is then established. The data being integrated can come from cartographical tools, previous studies on the site, evaluations of similar projects or the user.

Selection Of Laws From An Expertise Field

To evaluate the effect of the work on the environment, we must then define the laws that put in touch the technical components and the environmental components. These laws that describe the phenomena triggered by the interventions come from distinct expertise fields even if interrelated. They are integrated to the system, according to these expertise fields, in different knowledge base that can evolve independently one from the other.

In order to avoid selecting a large number of laws of which some are not really pertinent to the study, first the system establishes, from the characterization of the work and the site, the expertise fields linked to the components running the risk of being more solicited. The laws are then chosen within these expertise fields according to the available data.

Only the laws where all the data is known may be applied. This principle is inspired by the endorsement theory developed by Cohen (1986) according to which the verification of the necessary conditions of application allow to endorse or not an event (Robert, 1989). The laws selected and applied by the system are therefore those from which the conditions of application, that is the knowledge data they contain, are enforced. So, if the data that is available is rare or uncertain, the high precision laws cannot be used. This can be the case at the beginning of a study when we do not possess all the necessary information. To evaluate a phenomenon, we must then be able to count on laws of varying precision that are selected at various stages of the study according to the knowledge's condition.

Example Of Application Of The Management Methodology Of Knowledge Quality

From the characterization of the hydraulic system and the waterway's bed sediments, we can determine if these sediments will be set in suspension after dragging by comparing the shear velocity of the flow, U_*, and the settling velocity of the particles, w_s. The characteristics to consider are:

- mean diameter of sediments: D_{50} (m);
- specific gravity of sediments: s (dimensionless);
- water depth: d (m);
- riverbed slope: S_r (dimensionless);
- gravitational acceleration: g (m/s^2).

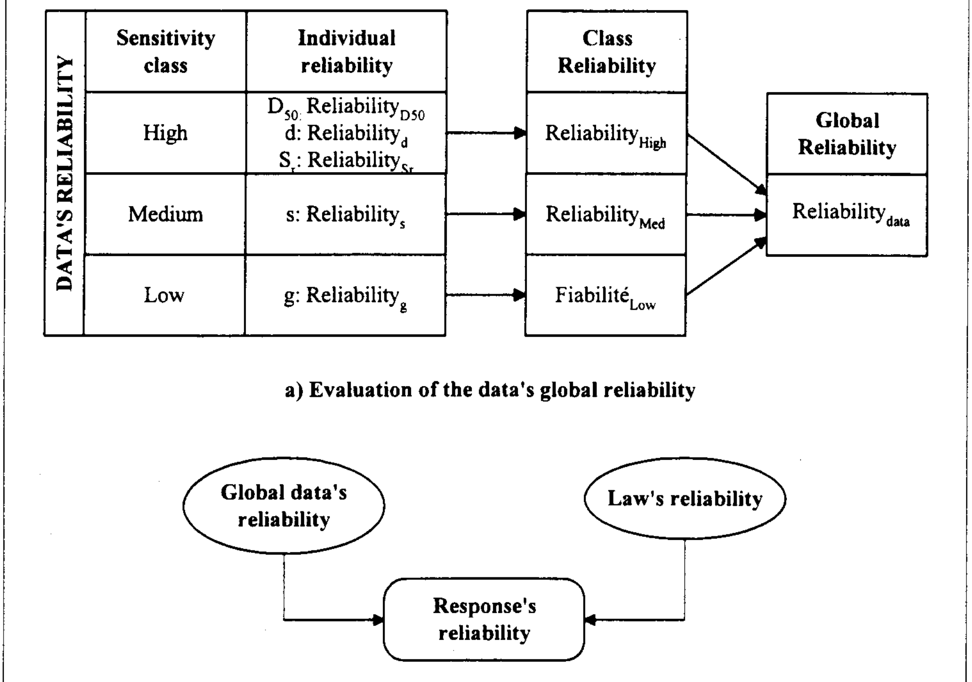
The law to use is structured as follows: knowing that $U_* = \sqrt{g\,d\,S_r}$

and $w_s = 1{,}1[(s-1)\,g\,D_{50}]^{1/2}$,

we compare U_* and w_s: if $U_* > w_s$, then: the sediments are set in suspension;

if $U_* < w_s$, then: the sediments settle on the riverbed.

The response's reliability is then evaluated in function of the law and data's reliability and sensitivity. Figure 1 illustrates the management methodology of knowledge reliability.



a) Evaluation of the data's global reliability

b) Evaluation of the response's reliability

Figure 1: Management methodology of knowledge reliability

CONCLUSION

By its innovative reasoning structure, the new developed approach proposes a more complete management of knowledge. The progressive acquisition of information with the progress of studies enables to apply more and more precise laws and thus increase the reliability of identified potential repercussions.

Furthermore, by integrating to the data base the intermediate results obtained, it is possible to proceed to the cumulative evaluation of impacts, essential to the global analysis of the effects of the work on the different components of the environment.

Finally, the identification of the main repercussions of a project at the preliminary stage enables to direct the sampling campaigns and to plan, very early, changes and mitigation measures to the actions that have strong effects on the environment.

The report presented at the Symposium will show, with the help of concrete examples of application, the possibilities of the system.

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NATURE-FRIENDLY DESIGN AND MAINTENANCE OF WATERCOURSES. A WAY TO PROMOTE ECOLOGICAL ENGINEERING IN THE MINISTRY OF THE FLEMISH COMMUNITY.

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ABSTRACT

Since 1990 the study group 'Ecological Engineering' has been active within the ministry of the Flemish Community (Flanders, Belgium). The main goal is to promote the use of ecological sound techniques to design, plan, execute and manage infrastructure works.

All levels of the public sector, but also the engineering offices and the general constructors must constantly be informed and sensitized. In 1994 a manual 'Ecological Engineering: design and maintenance of watercourses' was published in Dutch. At present more than 2900 copies have been distributed, most of them to civil and cultural engineers. The possibilities of the theoretical background are tried out in the field in a number of selected pilot projects. The English translation of the manual will be completed later this year.

KEY WORDS: Ecological engineering/ Infrastructure work/ Watercourse/ Sensitization/ Manual/ Pilot project

INTRODUCTION

Belgium is a country with a federal structure and is divided into three regions and three linguistic communities. Flanders is one of those three regions and is inhabited by the Dutch speaking community of Belgium.

Every region has a large autonomy on personal and territorial issues. In 1990 the department of the Environment and Infrastructure was founded within the ministry of the Flemish Community. It was for the first time in the Belgian history that the environment sector merged with the infrastructure sector into one department.

ACTUAL SITUATION OF NATURE AND INFRASTRUCTURE IN FLANDERS

Flanders is a densely populated region situated in the heart of Europe: more than 400 inhabitants/km^2. It is a very urbanized environment and economic activities such as industry, agriculture and recreation are still increasing. These activities demand a drastic management of navigable and non-navigable watercourses. Flanders has one of the densest water carriage systems in Europe (Eurostat, 1992). The density of navigable watercourses is nearly 50m/km^2.

Unfortunately, in this kind of system nature is put into a tight corner. In Flanders only 0.7 % of the total area is protected as nature preserve (Kelchtermans, 1990; Kuijken, 1994). Infrastructure works have several negative impacts on nature, such as:
- occupation of space;
- habitat loss;
- disturbance;
- animal victims;
- habitat fragmentation.

For those reasons protection and development of nature outside the nature preserves is a very important issue. The Council of Europe has declared 1995 the second European Nature Conservation Year (25 years after the first one) with special attention given to conservation of nature outside the protected areas. An important way to achieve this goal is the promotion of ecologically friendly methods for constructing and managing public infrastructure works. Eventually, the need for an integration of nature in the infrastructure can no longer be denied.

SHORT HISTORY

Over the last 20 years, the decision makers paid insufficient attention to environmental policy. However, even with a little extra manpower and the proper attitude, attemps can be made to plan, to build and to maintain public works in a more nature-friendly way. This approach requires a permanent discussion and collaboration between ecologists and engineers.

Some six years ago, the opportunity of such a collaboration was created with the foundation of the department of the Environment and Infrastructure. Both existing sectors of environment and of infrastructure had a self-willed and an independent way of working. From the start, it was clear that both sectors had different backgrounds, interests and approaches in solving certain problems. Nevertheless, they should try to apply a multidisciplinary and integrated way of thinking and working.

How could these sectors be stimulated to cooperate? Should every engineer become an ecologist? Or should every ecologist become an engineer? The head of the department of the Environment and Infrastructure decided to create a forum where the two groups can meet, each group with its own background and knowledge. The study group 'Ecological Engineering' was established with this aim in September 1990. There the two sides can explain their specific problems and can try to find solutions to these problems.

OBJECTIVES AND WORKING METHODS 'ECOLOGICAL ENGINEERING'

The main goal of the study group is to reduce the negative ecological impacts of many of Flanders' existing and future infrastructure works (Cherretté, 1993). This means to design, to plan, to execute and to manage public infrastructure works in a more environmentally friendly way. These public works include design as well as maintenance works on roads and watercourses. One of the major focuses of the study group is the general promotion of several rather simple ecological techniques. Within the study group an interdisciplinary approach by ecologists and engineers is essential.

There are many possibilities to conserve or even develop nature during the execution of public works. It should be noted however that nature itself can play an important role within the technical concept of a public work.

All the members of this study group are public servants: the services responsible for environment and public works are represented. Everybody works voluntarily. The involvement of all levels of the administration is necessary. The chairman is the secretary-general of the department of the Environment and Infrastructure. In addition to public servants, engineering offices and general constructors are the target public. A constant sensitization of this target public is necessary.

From the start, the study group tried to realize its goals in a practical way: firstly by the publication of manuals on ecological engineering; secondly by the implementation of ecological engineering in pilot projects.

ACTUAL SITUATION

Manuals 'ecological engineering'

The first manual 'Ecological Engineering': design and maintenance of watercourses' (Claus en Janssens, 1994) deals with the integration of hydraulic infrastructure in the natural environment. This manual is a practical ecological specification for public works on watercourses.

The first part of the manual is a kind of technical guideline. In the introduction you can read why ecological engineering is important to obtain an integral water management. Here you can also find a summary of the theoretical background of ecological engineering and the specific characteristics and functions of watercourses are described. On the index cards, a lot of specific information is given about the techniques to achieve an ecologically friendly way of maintaining watercourses. These index cards are divided into three groups: cross section, linear profile and water quality. At the end of every text or index card, there are cross-references to other index cards and to the recommended literature.

The second part of the manual gives useful background information. Watercourses are important natural elements in the environment. Pictures of some animals and plants are included to illustrate the ecological importance of the watercourse to the engineers. Because in Flanders many authorities are responsible for the design and maintenance of watercourses, we have also included several maps on which the distribution of the geografical competence of these authorities (with addresses) is given. There is also a small compilation of the legislation and administrative regulations on the design and maintenance of watercourses. Most of the recommended literature is in Dutch and easy to obtain for the users in Flanders. Finally, there is the glossary which gives an explanation of technical and ecological terms, that are often used in an inaccurate way. The manual is a loose-leaf system, so new index cards can be added easily and old ones can be updated to include for example more advanced techniques or to adjust other issues when necessary.

The first manual was presented to the public during a workshop in Brugge (Belgium) in March 1994. The large attendance of this workshop (more than 600 participants) proved the importance and the interest of the public in ecological engineering.

At this very moment, at least 2850 copies of the manual have been distributed in the Flemish administration and sold to local authorities, constructors and engineering offices. Some 100 copies were ordered in the Netherlands. We received positive comments from the cultural and civil engineers. They consider the manual a practical and useful guideline with techniques which might provide an additional ecological function to their work.

Pilot projects

At first, 17 pilot projects were selected to visualize and test the advantages and disadvantages of ecological engineering of watercourses by practical experience (Figure 1). In these pilot projects the development of more diversity in nature was one of the limiting conditions for the execution of the civil work. However, we try to integrate environmental considerations into the planned, ongoing and existing works.

An interesting example of such a pilot project is the execution of environmentally attractive embankments along a number of canals (Elskens et al., 1992; Cherretté et al., 1994). Instead of traditional hardened slopes (in concrete or gabions), a hardened front defense with a splash berm is used against water-erosion. The splash berm is converted into a wet zone and the slope is hardened only slightly if at all (Figure 2).

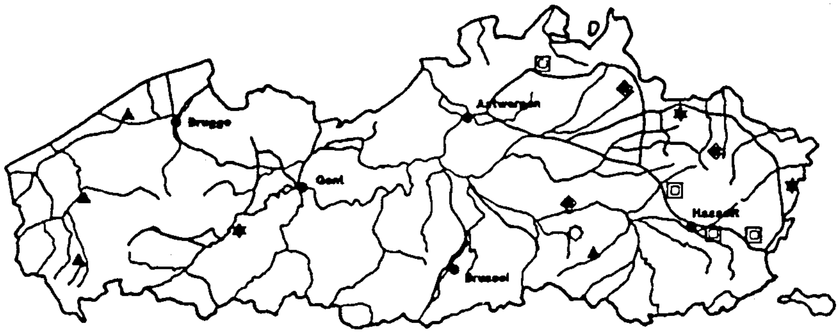

Figure 1: Existing pilot projects in the surrounding of watercourses

- ✦ environmentally friendly embankment
- ⊡ spawning place
- ◆ integrated water management project
- ○ wastewater treatment plant
- ▲ nature development

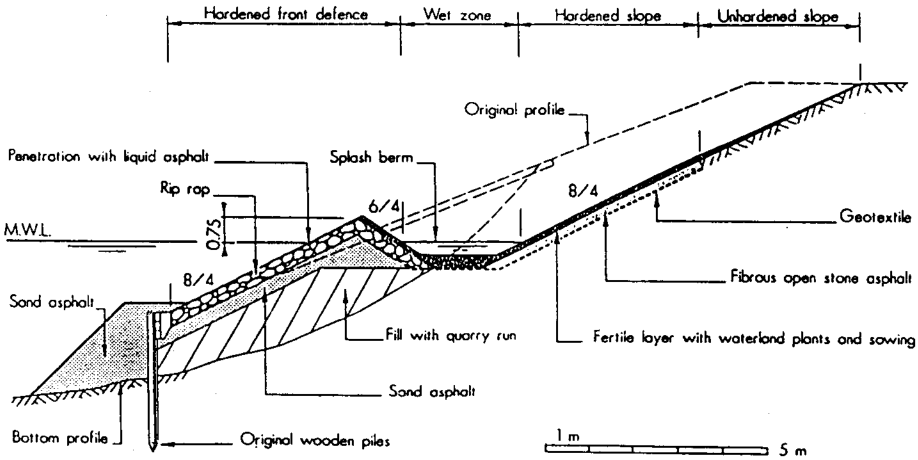

Figure 2: Environmentally attractive embankment (After Cherretté et al, 1994)

Flood-control dams, sluices and milldams compartmentalize watercourses and hinder the upstream migration of certain fish species. This problem can be solved by the construction of fish ladders. In several places in Flanders fish ladders are being installed (Coeck et al., 1991).

An important part of Flanders is flat land below sea level. Precipitation must be pumped up and discharged into the sea. Most of the pumping is done during rainy nights in autumn. Exactly during these nights, adult eels (*Anguilla anguilla*) start their catadromous migration towards the Sargasso Sea. This migration is initiated by a strong water current, which will be intensified by the pumping installations. Most of the pumped up eels are seriously injured or killed (Germonpré et al., 1994). The replacement of the pumps by Archimedes' screws or the installation of light, sound or electric avoidance systems will reduce the negative effect on the eels. At least one pumping station in Flanders has been adapted in that way.

The steep or even vertical slopes of canals prevent some animals from getting out of the water and many of them drown. The installation of special exit ladders for animals is effective and reduces the number of drowned animals (Van Haaften, 1984; Bekker, 1990). In this way further habitat fragmentation can be restricted. Many of these 'getting out' steps are constructed in certain regions rich in game.

PROSPECTS

Constant sensitization

It is obvious that a permanent sensitization is a major task of the study group. To achieve its goals, the study group tried to proceed pragmatically. The strategy used to reach the infrastructure sector is to a great extent indirect. The recognition of specific problems and possible solutions by the public servants responsible for the design and maintenance of watercourses is of crucial importance. It is at least as important to stimulate these servants to convince their colleagues to use nature-friendly techniques whenever possible.

Recently, a lot of interested regional public servants participated in three information afternoons on ecological engineering in and around watercourses. Examples of existing projects with possibilities and restrictions were given by engineers. Soon, at least five information sessions will be organized on the same subject and all local authorities will be invited. Once again, the civil and cultural engineers will be invited to discuss the advantages and disadvantages of ecological engineering. In addition lectures for engineers on ecological applications in hydraulic engineering will be organized.

New pilot projects

To support this indirect sensitization new pilot projects are badly needed to promote the technical and financial possibilities of ecological engineering. Obviously the existing pilot projects must be followed-up and evaluated constantly.

It is not always easy to convince civil and cultural engineers to integrate nature in infrastructure works. Some engineers hesitate because they are not convinced of the durability of the nature-friendly solutions. An often used comment is the cost-raising effect of those solutions.

However, a lot of engineers are already working in an integrated way. They know the intrinsic quality and possibilities of nature to solve certain hydraulic problems. For other problems technical solutions are needed, but an infrastructure work can be executed in such a way that nature development becomes an important part of it. Usually, nature-friendly solutions demand more space and sometimes cause extra costs. The increased value of nature is a compensation for those supplementary costs.

Cooperative engineers are asked to suggest new pilot projects to illustrate the advantages and disadvantages of ecological engineering of watercourses to their colleagues.

Specifications and administrative regulations

Engineers are used to work with technical specifications. Therefore, the manual 'Ecological Engineering': design and maintenance of watercourses' (Claus en Janssens, 1994) has an analog structure and classification as the most common specifications.

An important step to promote ecological engineering within the public sector is the adaptation of the technical specifications with the incorporation of more nature-friendly techniques. The replacement of the old standard techniques by more ecological sound ones is an interesting evolution. Still, these new techniques remain standard solutions. The major problem is the need of more specificity in the use of ecological engineering in hydraulic structures. Ecological engineering demands a lot of inventiveness and sufficient creativity of the engineer involved.

Administrative regulations for the use of ecological engineering within the department of the Environment and Infrastructure of the ministry of the Flemish Community, are proposed in circular letters. Each time the services responsible for public works design, plan, execute or manage an infrastructure work, they are requested to consult the services responsible for environment to obtain ecological advice.

CONCLUSION

Of course, the manual 'Ecological Engineering: design and maintenance of watercourses' will be used for a further sensitization of the engineers. However the ecologists have yet to discover the manual. So far, the nature conservation societies have not given any comments on the manual. This is regretful, because we had to try to convince the ecologists to cooperate with the engineers to support the use of the vademecum along with other specifications. This will be a difficult part of the sensitization. In Flanders, there is no tradition of good contacts between nature conservation societies and the authorities and executors of public works. Nevertheless, the vademecum must become a common ground to bring both sides closer together. Only by this constant interaction, a broad consensus on the design and maintenance of public works can be attained.

Unfortunately, the manual is available only in Dutch and is not easily accessible for non-native speakers. Despite this handicap the vademecum received a lot of interest during the 28th International Navigation

Congress (PIANC) in Sevilla (Spain) in May 1994. We also presented the vademecum during the workshop 'Ecological Engineering for Ecosystem Restoration' of the International Ecological Engineering Society (IEES) in Zeist (The Netherlands) in November 1994. Finally the study group was selected as one of the ten success stories on the conference 'Towards a new development approach' in Brussel (Belgium) in November 1994, organized at the occasion of the retirement of Mr. J. Delors, chairman of the European Commission.

We therefore think that an English translation of the manual 'Ecological Engineering: design and maintenance of watercourses' will be most useful. The translation project has started earlier this year. With some minor adaptations, most of the ideas of ecological engineering for hydraulic works can be used outside Flanders. Of course, problems and solutions, inherent to the Flemish situation, must be replaced.

Finally we think that ecological engineering can only be successful if a permanent cooperation exists between engineers and ecologists not only with regard to research and teaching, but also during the actual execution of works. An interdisciplinary study group, like the one presented, can support such a cooperation.

ACKNOWLEDGMENTS

We would like to thank Mr. R. De Paepe, Mr. F. Desmyter and Mr. M. Cherretté for their indispensable stimuli to put the study group into action, for emphasizing the importance of ecological engineering during the design and maintenance of public works and for giving us a lot of opportunities to promote the activities and the achievements of the study group.

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Formation of the International Aquatic Modeling Group

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ABSTRACT

The International Aquatic Modeling Group (IAMG) was formed by a group or research scientists and engineers to facilitate a focused and open forum for the exchange of applied research centered around instream flow assessment methodologies. The IAMG has an overall main objective to develop, test, and validate assessment methods used to quantify the interactions between instream flora and fauna and alterations in either stream structure or flow patterns. The group aims to meet the increasing demands for improved operational tools for instream flow assessments using a broad based multi-disciplinary approach that focuses on an ecological based framework. This document summarizes the collective input of the existing Scientific Committee of the IAMG on setting the short-term and longer term research objectives. Members of the Scientific Committee and their corresponding affiliations are noted at the end of the document.

BACKGROUND

The desire for quantitative tools which can assess the impact of artificial influences on aquatic ecosystems in a consistent and reproducable manner has increased over the past two decades. At present, many of the existing tools or methodologies have come under criticism, especially in their applicability to meet more holistic ecosystem management goals. There is a growing recognition that such an approach to the management of aquatic ecosystems will require the development and/or integration of tools which utilise physical, chemical and biological process driven methods. Although there are a large number of individual organisations and researchers examining topics directly related to this issue, there does not exist a framework for a focused, integrated and timely exchange of current research progress and output, or for the development of strategic, multi-national, co-operative research programmes. The IAMG is intended to provide this framework, capitalising on existing individual and networked research programmes, through the focusing of the particular expertise and strengths of these programmes within a clear set of research areas. This will assist in the development of the best achievable assessment methodologies for applied use in aquatic ecosystem management. The IAMG has formed a series of sub-groups to provide a focus within specific research areas which represent key elements of an integrated multi-disciplinary assessment framework. The current empahsis of the IAMG integrated research priorities are highlighted below.

IAMG SUB-GROUPS

The existing structure of the sub-groups within the IAMG is shown in Figure 1 with the associated contact person listed for each group. It is intended that where appropriate, Sub-Groups will be broken down into more specific focus groups (i.e. Biological and Life History) in order to provide a more organized focus of research and scholarly exchange. This is already occurring within the existing Sub-Group structures as evidenced by the breadth of research topics discussed below. The existing Sub-Group (and focus group) structure is intended to be a flexible paradigm that will shift as continued interest expands the participation by other scientist and engineers.

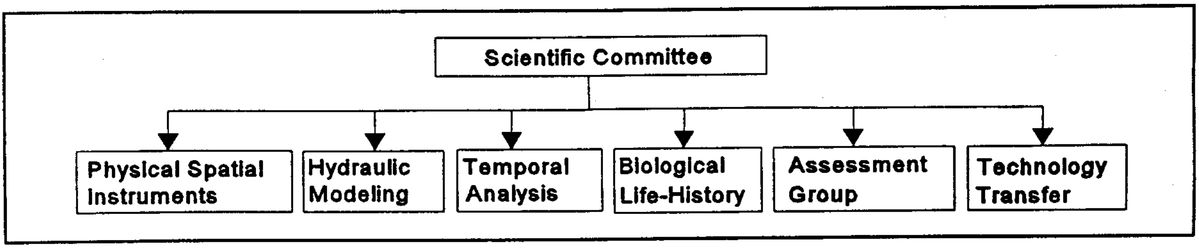

Figure 1: IMG Subgroups

RESEARCH FOCUS AREAS

Characterization of the Spatial Domain

This effort is intended to focus research at the applied level which can improve more efficient field measurement strategies suitable for better delineation of aquatic habitat characteristics and allow direct linkage of data to 1, 2, or 3-dimensional hydraulic modeling. Evaluation of field collection strategies will also focus on data acquistion over spatial domains that are amendable for use in utilization of landscape ecology metrics applied to the aquatic domain. The initial goal of this research is the evaluation of the effacicey of different sampling strategies such as stratified random, systematic, systematic irregular, etc, to provide spatially accurate representations of the aquatic environment over different spatial scales. The evaluation of these different sampling strategies is intended to lead to the development of a strategic sampling framework for various river types and assessment objectives which consider both the requirements of biological and physical modeling and provide for the comparability of results. This research is also intended to generate a database of high quality field samples from a variety of river systems which will be continuously overviewed, classified and provided to other subgroups for model evaluation purposes.

Instrumentation

This area of research will focus on the evaluation of new instumentation technologies suitable for highly accurate cost effective spatial mapping within river systems. This currently involves the evaluation of differentially corrected global positioning systems, depth pressure transducers, hydro-acoustic arrays for bottom profiling, and integration of other advanced survey techniques using total stations with automatic target search, and remote sensing systems. The short term goal of this effort is the development of cost-benefit criteria for application of these technologies to meet specific data quality objectives given specific spatial domains. A longer term goal is the development of a systematic sampling framework that incorporates spatial sampling strategies given data requirement needs for specific types of hydraulic, habitat, and other assessment models.

Measurement Scale Transferability

This research is intended to evaluate appropriate techniques which permit the characterization of aquatic ecosystems over a variety of spatial scales, while maintaining meaningful representations of the spatial requirements of target organisms. Research will initially focus on addressing the accuracy of spatial measurement scales at the micro-habitat, macro-habitat, and reach level characterizations. This will also include the evaluation of mapping methods and the transferability of results at the micro-habiat level to meso- and macro-scales. The potential use GIS analysis of spatial interactions, sensitivity studies, and generalization of procedures are also consider important venues within this area of research. The results of these activities are intented to provide guidelines for the development of sampling strategies and is closely related to work within the Instrumentation Sub-Group.

Temporal Dynamics

This area of research will examine the functional relationships between the temporal dynamics of physical habitat fluctuations and biological responses with the aim of determining the limiting conditions for aquatic populations in rivers. The main biological reference species to be examined are fish and macro-invertebrates, although macrophyte species will also be examined since their growth is affected by physical habitat and may also lead to changes in aquatic habitat for other species. An important aspect of this work will be the field validation of existing and newly developed model results. This will require the selection of river study sites where long-term records of biological and physical data have been collected. The group will also examine methods of expressing changes in aquatic habitat with time and develop guidelines to aid the interpretation of time dependant model results.

River Scale Morphology

The scale of the impact of artificial influences on river systems may vary from a few kilometers to several hundred kilometers of river. As a result, there is a need for a consistant, repeatable method of defining river habitats, at the meso- or macro-habitat scale, to allow the extrapolation of aquatic habitat model output from localized studies to larger spatial areas of the river. However, there are important temporal aspects which also need to be addressed. For example, research needs to focus on the temporal changes in the distribution of habitats within rivers in response to seasonal flow patterns as well as to longer term trends associated with changes in anthropogenic flow alterations. This research will initially focus on the evaluation of methods that can be used to predict changes in the micro- or meso-scale habitat characteristics both on an annual basis as well as under the influence of longer term changes associated with anthropogenic induced flow alterations. This research is intended to link geofluvial processes modeling at the meso-habitat scale to delineate expected trends at the reach scale level.

Hydraulic Modeling

A large number of hydraulic models are available for use in modeling aquatic habitat, but very few of them are currently being used. The potential of alternative modeling approaches such as two and three-dimensional solutions to the Navier-Stokes equations, characterising the near-bed-flow environment, and statistical hydraulic models have been discussed but as yet not achieved wide application in studies of aquatic habitat. The evaluation and development of these innovative approaches for use as operational tools is of special interest to the Hydraulic Modeling Sub-Group. One area of research focus is to improve the evaluation of aquatic habitat by developing, validating and making widely available suitable, reliable and efficient hydraulic modeling techniques. The first main objective is to link the description of physical variables required to evaluate the habitat of aquatic biota which include the physical properties of relevance such as velocity and depth, the scale and location at which these properties are important and the required characterisation of any spatial and temporal variations of these properties. In principle a model of the interaction between species habitat selection and the physical stream environment should be identified prior to selecting appropriate hydraulic models. A short term goal will be to provide a representative list of suitable hydraulic models using different modeling approaches that could be used in an aquatic modeling study. This effort will include a summary description of the models, including field data requirements, calibration facilities, required computer performance, type of results, validation possibilities and a reference list.

Another important area of research emphasis will be the field validation of these modeling approaches for aquatic habitat studies. Scientifically approved methods for validation of the hydraulic model performance must be ensured or established. The uncertainty in the predictions of the physical habitat characteristics and the minimum data collection and modeling effort required to apply them under the range of circumstance encountered in operational studies will be determined and evaluated. This effort will also consider a focus on the need for modeling on larger spatial and temporal scales. This will consider modeling of dynamic changes like sediment transport and deposition, ice formation and other dynamic physical processes in streams. In order to understand and manage the whole aquatic ecosystem, dynamic hydraulic models must be used over a long time-scale giving the opportunity of studying both short-term and long-term variations in the physical habitat and effects on the ecosystem. The final objective of this research area is to develop a set of guidelines to be used by aquatic modelers for selection of the most appropriate hydraulic model for their particular application. Where hydraulic models are not available to predict certain habitat characteristics in an operational mode, the group will initiate research into the development of an appropriate hydraulic modeling approach.

<u>Biological life-history</u>

This area of research emphasis represents one of the more critical elements for the IAMG. The need for more fundamental research on the life history requirements of target species, population dynamics, and community interactions has been clearly articulated within the literature. The primary goal of this reasearch effort is the development of biological metrics which can be used within an applied assessment framework. In particular, the evaluation of landscape ecology metrics applied to the aquatic environment will be considered since these metrics may be ideally suited for examination of the broader spatial scales at the river reach and watershed levels. This effort will provide for the evaluation of these types of metrics and a framework for their application and interpretation at the operational level of instream flow analyses. A second area of research emphasis will involve the evaluation of analytical techniques to quantify the ontogenetic shifts in flow dependant response variables of aquatic species that can be linked to existing individual based models as well as more classical habtitat modeling approaches. Issues such as development, testing and transferability of suitability curves will be considered. An important short term goal for this effort will involve the development of a listing of available individual, population, community, and habitat based models suitable for use in applied instream flow applications. This effort will include a summary description of the models, including field data requirements, calibration facilities, required computer performance, type of results, validation possibilities and a reference list. Finally, research efforts will focus on comparitive evaluations of these respective methods/models using existing data sets and where possible, design and implementation of integrated research projects with this specific goal in mind.

<u>Assessment Methods</u>

The focus of this research is to integrate the work products from the other sub-groups into a systematic framework for applied impact assessments in water resource systems. At present, the application of analytical tools in assessment of impacts lacks a clear and consistent framework for interpretation, as well as on-going issues as to the validity of the model output to predict or correlate changes in the response variable(s) of interest (e.g. fish populations over time). The emphasis of this group will rely heavily upon field validation studies using existing data for systems where the emerging tools from other sub-groups can be hind-cast to evaluate their ability to predict the observed changes in these systems. In addition, this effort will focus on the development of a protocol or guidelines which clearly articulates the steps in the application and integration of the component assessment tools and their consistent interpretation in operational instream flow studies.

Technology Transfer

The IAMG has a strong commitment to facilitate the transfer of data, study results, and available modeling tools. As part of this commitment, the Institute for Natural Systems Engineering (INSE) at Utah State University has agreed to establish several support services for the IAMG. First a world wide web (WWW) home page is being established which will list the on-going research elements underway within the IAMG, participating agencies and investigators, and help for contacting individual collaborating investigators/agencies. The WWW page will be updated on a frequent basis to keep exchange of on-going research focus and results current. In addition, the INSE will host an anonymous FTP site as a depository of reports, software, etc. to facilitate scientific exchange of data, information, and modeling tools for use by interested parties. Finally, the INSE will also host a Bulletin Board Service (BBS) forum on instream flow assessment methods where questions, answers and debates on methods and study results can be carried out via the Internet. The BBS will serve as a convenient forum for more focused interchange of ideas and comments within the active research community on instream flow assessment methods.

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NATIONAL WILDLIFE AREAS : ESSENTIAL HABITATS ON THE ST. LAWRENCE

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ABSTRACT

In 1996, the Québec network of national wildlife areas comprises eight sites along the St. Lawrence from lac Saint-François to the Îles de la Madeleine and protects more than 6000 ha. This network is of part of a much larger one comprizing 49 sites across Canada.

The inclusion of three of these wildlife areas on the Ramsar Convention list acknowledges their international importance as wetlands essential to wildlife.

The present communication will make you travel along the St. Lawrence to discover those sites, from the large marshes of Lac St. François to the sand dunes of Pointe de l'Est in the Gulf of St. Lawrence.

Kew-words : wildlife/habitats/St. Lawrence/wetlands/network/marshes/islands/birds/waterfowl.

INTRODUCTION

Aware of the importance of protecting essential wildlife habitats, the Canadian Wildlife Service set up a program in 1969 to acquire strategic habitats along the St. Lawrence.

The acquired properties, made up in large part of wetlands, form a network of national wildlife areas representing diversified and highly productive environments.

LAC SAINT-FRANÇOIS

Lac Saint-François National Wildlife Area is located on the south shore of the St Lawrence River 50 km upstream of Salaberry-de-Valleyfield.

Extending over 1,347 ha, the wildlife area is located in the climatic zone of the predominantly deciduous sugar maple-basswood forest area. Other species present include beech, white ash, eastern hemlock, black ash, the rare rock elm and poplars. In swampy sites grow larch, red maple, silver maple, alder and willow.

As far as the eye can see, marshland etched with ponds and channels blends into Lake Saint-François. This is a paradise for botanists, who will be thrilled by the diversity of vegetation and the forty or so rare plants found here, such as the very tall milkweed that grows nowhere else in Québec but on Mount Saint-Hilaire. Ginseng, a vulnerable species, is also found in the wildlife area's maple-beech woods.

On and along the great expanses of marsh and swamp, the natural and man-made ponds, the streams and the lake, waterfowl are present in spectacular numbers during the migration, breeding and moulting periods. The woods, transition zones and open spaces provide favourable conditions for an exceptional diversity of terrestrial fauna year-round : the annotated species list for the wildlife area contains over 220 entries.

At the height of migration, thousands of dabbling ducks stop over for anywhere from a few days to a few weeks. The greater snow goose, brant, red-breasted merganser, white-winged scoter, canvasback, oldsquaw and ruddy duck are visitors of particular interest, while the northern pintail, mallard, black duck and Canada goose are among the more familiar transients. Nesting nearby, the great egret feeds in the marshes of the wildlife area.

At least 110 species of birds nest in lac Saint-François National Wildlife Area. The least bittern, Canada goose, redhead, gad-wall, ring-necked duck, lesser scaup, common goldeneye, pileated woodpecker, red-headed woodpecker and indigo bunting are among the most spectacular breeding species. At this location, the sedge wren establishes one of its largest colonies in Québec. Its habitat, the sedge marsh, like the habitats of many other birds, depends for its future on the protection afforded the site by its wildlife area status.

One of thirteen duck species that breed in the wildlife area, the wood duck with its colourful plumage is a delight to bird-watchers. The marshes developed in the wildlife area by Ducks Unlimited are highly productive in the breeding period and extensively used in the migration season by many species of ducks.

ILES DE LA PAIX

Îles de la Paix National Wildlife Area (120 ha), a string of marsh-bordered islands, some 20 km Southwest of Montreal serves as a quiet haven for local waterfowl, since the banks of Lake Saint-Louis are mainly developed, urbanized and industrialized. The clear need to protect these island habitats fully justifies their national wildlife area status.

Most of the islands dip in the centre, forming a basin and favouring a wetland plain to marsh plant succession where the inexorable laws of a varied plant and animal world are interwoven. Around the islands stretches a zone of more than a thousand hectares where the water depth does not exceed one metre except in a few navigable channels. This is the realm of some twenty emergent plants such as sweetflag, discoid beggar-ticks (a rare species), horsetails and cattails.

On the water's surface, yellow pond-lilies and tuberous water-lilies create a perfectly executed pointillist effect. Under the water, large vegetation beds made up of pondweed, myriophyllum and waterweed harbour abundant populations if invertebrates that serve as a base for the diets of fish young and ducklings.

The woodlands here are generally highly diversified, although in some locations red maple is predominant. Depending on soil conditions, the cortege of tree and shrub growth is made up of silver maple, gray birch, white and red ash, basswood, white elm, common buttonbush, winterberry, staghorn sumac, plums and viburnum.

The alluvial woods harbour a jewel of the plant kingdom, the green dragon, recorded as a vulnerable species on the list prepared by the Committee of the Status of Endangered Wildlife in Canada (1988). Visiting the islands' various habitats, the knowledgeable observer will discover close to fifteen rare plant species.

The Îles de la Paix island chain represents on of the five to-ranked areas in Lake Saint-Louis for springtime waterfowl density. More than 5,000 ducks stop here to replenish their energy reserves from early April to mid-May, when they begin setting out for breeding grounds.

The concentration of dabbling ducks nests of the islands is high, and one can observe a very singular form of nesting behaviours attributable to the springtime water level peaks. Visitors will be amazed to find nests of black ducks, mallards, blue-and green-winged teals and northern pintails perched in tree forks, sometimes as high as two metres in the air. Other species of note that breed in the islands include the northern shoveler, the American widgeon and the colourful wood duck.

ILES DE CONTRECOEUR

The 28 alluvial islands of Îles de Contrecoeur National Wildlife Area cover a land area of 200 ha and are located a short distance off the banks of the municipality of Contrecoeur approximately 35 km Northeast of Montréal.

An important component of a habitat favourable to waterfowl breeding, reed canary grass sometimes covers entire islands and form elegant groupings that undulate in the breeze. Vast beds emergent vegetation form in ways determined by environmental conditions and competition among their components. Here it is cattails that advance inexorably and displace other less aggressive species such as broad-fruited bur-reed, river and American bulrushes and even arrowhead.

The vast emergent and submerged beds serve as a larder for thousands of young and adult duck and other waterfowl during the breeding and migration periods.

The Contrecoeur island chain is a prime site for the birds of the Montréal region. In its grassy meadows, waterfowl find a suitable habitat for breeding. Of all the island chains from Montréal to Sorel, this is undoubtedly the most important for waterfowl reproduction and thus is fully entitled to the protection conferred upon it by its national wildlife area status. The site's significance extends beyond the regional level; it is unhesitatingly recognized as being of provincial importance because of the uncommon abundance of species such as the gadwall.

The northern pintail and American widgeon also breed here in great numbers. Waterfowl aside, a variety of bird species nest in the area. Other species present include the red-winged blackbird, the ever-so-long-billed common snipe, the spotted sandpiper that moves to the rhythm of music that it alone can hear, the American bittern, the common moorhen, the black term and the Virginia rail. All of these species rear their young in the islands.

Birds of prey like the northern harrier and short-eared owl tirelessly over the site. The objects of their search, whether birds or mammals, are fearfully aware of their silhouettes and their intentions.

Visitors will also notice the constant comings and goings of the ring-billed gulls over the four colonies in the archipelago that support more than 5,000 nests.

Protected from human disturbance by their isolation, the islands serve as a migratory staging area for species that breed farther north, such as the Canada goose. This majestic bird is found in large groups during the spring migration along with many common goldeneyes.

CAP TOURMENTE

Cap Tourmente National Wildlife Area is located on the north shore of the St Lawrence River about 50 km downstream from Québec City. The 2,391-ha site extends immediately to the south and Northeast of the villages of Saint-Joachim and Cap-Tourmente.

Nowadays, it serves as a staging area more than for 400,000 greater snow geese on their twice-yearly journeys between the Atlantic Coast and the Far North. The population was only about 3,000 birds at the beginning of the century, and Cap Tourmente National Wildlife Area was created to ensure the survival of the species. During the fall migration, the entire world population of greater now geese stops off in the vicinity, where it feeds on American bulrush rhizomes in the marsh and grain in the fields.

From the vast expanses of American bulrush on the tidal marsh to the abundant vegetation of the coastal marsh and the groves, cultivated fields and man-made waterfowl ponds of the coastal plain, the visitor has to travel only a short distance to see habitats that are varied and teeming with life.

The characteristic forest cover of the immense glaciation-ravaged mass of granite called Cap Tourmente is made up of maple, black spruce, birch and beech stands.

In the fall, Cap Tourmente is the most important sector in the entire St Lawrence corridor for dabbling ducks. In addition to waterfowl, many species such as warblers and other songbirds are concentrated in the spring and fall along the narrow corridor formed by the river and the Cap Tourmente escarpment. A birders' outing at these times of the years may be graced with sightings of rare or unexpected species such as the Ross', greater white-fronted, pink-footed, bean or barnacle goose, ruff, boreal owl, western kingbird and Townsend's solitaire.

Of the 250 species which have been sighted in the wildlife area, about one hundred are recognized as local breeders and eight are on the list of endangered birds : the common barn-owl, Cooper's hawk eastern bluebird, great gray owl, least bittern, loggerhead shrike, peregrine falcon and red shouldered hawk. Warblers, the gems of the birds world, are represented by 18 breeding species. Among these, the most familiar are the black-throated green warbler, ovenbird, yellow-rumped warbler, common yellowthroat and incomparable American redstart.

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LA BAIE DE L'ISLE-VERTE

Located on the south shore 30 km Northeast of the municipality of Rivière-du-Loup in the estuary portion of the St Lawrence River, Baie de L'Isle-Verte National Wildlife Area takes in 646 ha.

Its mains feature is a vast marsh dotted with ice extraction pans created by the action of the retreating ice in the spring. This is the site of the largest remaining spartina marsh in southern Québec.

The tidal marsh that lies on either side of the cove or little bay of L'Isle-Verte is the last vestige of the great spartina marshes of the St Lawrence Estuary. The ice extraction pans that give the marsh its unusual appearance produce vast quantities of living organisms and serve as rearing sites for the broods of black ducks that are omnipresent in season because of the very favourable survival conditions they find there.

Moving up the natural slope from the tangled kelp beds of the extreme low tideline, the visitor will encounter seaweed beds followed by the spartina-dominated tidal marsh. Then comes the coastal marsh, which has a highly diversified flora but is still ruled in some spots by the alders that once characterized this stretch of shoreline.

The only forest vegetation communities, which are coniferous, occupy the rocky heights of the wildlife area. From these vantage points, where the dominant species are black spruce, jack pine, balsam fir and white birch, unfold landscapes if unrivalled serenity.

Over 130 bird species have been sighted locally and several of them, including the peregrine falcon, king rail, Cooper's hawk and loggerhead shrike, are on the 1988 list prepared by the Committee on the Status of Endangered Wildlife in Canada.

The number of migratory birds at L'Isle-Verte in the spring is estimated at 35,000, compared with 10,000 in the fall. Of this number, 27,000 are greater snow geese and Canada geese in the main, accompanied by black ducks, green-winged teals, northern pintails, common eiders, scoters, gulls and double-crested cormorants. Anatidae make up the largest group of migrants in the fall, when black ducks number over 4,000.

LAC SAINT-FRANÇOIS

Lac Saint-François National Wildlife Area is located on the south shore of the St Lawrence River 50 km upstream of Salaberry-de-Valleyfield.

Extending over 1,347 ha, the wildlife area is located in the climatic zone of the predominantly deciduous sugar maple-basswood forest area. Other species present include beech, white ash, eastern hemlock, black ash, the rare rock elm and poplars. In swampy sites grow larch, red maple, silver maple, alder and willow.

As far as the eye can see, marshland etched with ponds and channels blends into Lake Saint-François. This is a paradise for botanists, who will be thrilled by the diversity of vegetation and the forty or so rare plants found here, such as the very tall milkweed that grows nowhere else in Québec but on Mount Saint-Hilaire. Ginseng, a vulnerable species, is also found in the wildlife area's maple-beech woods.

On and along the great expanses of marsh and swamp, the natural and man-made ponds, the streams and the lake, waterfowl are present in spectacular numbers during the migration, breeding and moulting periods. The woods, transition zones and open spaces provide favourable conditions for an exceptional diversity of terrestrial fauna year-round : the annotated species list for the wildlife area contains over 220 entries.

At the height of migration, thousands of dabbling ducks stop over for anywhere from a few days to a few weeks. The greater snow goose, brant, red-breasted merganser, white-winged scoter, canvasback, oldsquaw and ruddy duck are visitors of particular interest, while the northern pintail, mallard, black duck and Canada goose are among the more familiar transients. Nesting nearby, the great egret feeds in the marshes of the wildlife area.

At least 110 species of birds nest in lac Saint-François National Wildlife Area. The least bittern, Canada goose, redhead, gad-wall, ring-necked duck, lesser scaup, common goldeneye, pileated woodpecker, red-headed woodpecker and indigo bunting are among the most spectacular breeding species. At this location, the sedge wren establishes one of its largest colonies in Québec. Its habitat, the sedge marsh, like the habitats of many other birds, depends for its future on the protection afforded the site by its wildlife area status.

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More than 60 bird species breed in the wildlife area. The commonest are the budding bobolink, red-winged blackbird, common grackle and opportunistic brown-headed cowbird. The site is of vital importance for the sharp-tailed sparrow, which breeds very locally in Québec.

Within the St Lawrence Estuary, the wildlife area is one of the main breeding grounds for the black duck, which occupies the upper reaches of the marsh and the edges of brushlands and streams. But the rearing period is when the site shows all of its true importance - the marsh provides shelter for large numbers of broods from the vicinity. The fifteen kilometres or shoreline in the wildlife area enable more than 500 ducklings to reach fledging age. The common eider, a diving duck typical of the St Lawrence Estuary, also uses the marshes of L'Isle-Verte for rearing its young.

POINTE-AU-PERE

Pointe-au-Père National Wildlife Area, located 5 km east of Rimouski, covers approximately 23 ha. It is the least extensive national wildlife area in Québec but the most important in terms of the protection it affords thousands of shorebirds.

The local spartina marshes broken up by ice extraction pans are similar to those at L'Isle-Verte. Scanning from north to south, the observer first sees a rocky notch followed by a small spartina zone.

The Pointe-au-Père marsh is a highly attractive site for waterfowl in both the migration and breeding periods. Over 110 bird species are on the list of recorded sightings in the wildlife area and over fifteen breed here. Among exceptional sightings we find the eared grebe (a first in the province), great egret, little egret, snowy egret, tricolored heron, greater white-fronted goose and Smith's longspur as well as many other species such as the Forster's tern (recorded here for the first time east of Québec City).

The wildlife area is part of an increasingly limited series of staging areas along the shorebird migration corridors. Flocks of least sandpipers sometimes over 500 strong are present in the month of May. The site is also visited by the spectacularly plumed ruff on occasion and by 500 to 1,000 short-billed dowitchers.

The great blue heron feeds in the marshes along with the black-crowed night-heron and large groups of brants and Canada geese. The northern harrier flies over the marshland in search of prey; other raptors - including the sharp-shinned, Cooper's, broad-winged and red-tailed hawks and golden eagle - patrol the shores in spring.

Fall visitors include red-throated and common loons, large numbers of Canada geese, greater snow geese (in recent years) and the merlin and peregrine falcon (currently the subject of national efforts to augment its endangered population). Shorebirds are numerous in this season : at times up to 900 ruddy turnstones frequent the tidal marsh, which accommodates large groups of sanderlings and semipalmated sandpipers as well as the white-rumped sandpiper, dunlin and rare stilt sandpiper.

During the migration period, the site also hosts the oldsquaw, the common merganser and, on occasion, the ring-necked duck, redhead, king eider and Eurasian widgeon. Common species include the herring gull and ring-billed gull.

For the common eider, the coastal marsh at Pointe-au-Père is one of six sites considered exceptional in the whole of the middle and marine portions of the St Lawrence Estuary. While the species does not actually breed in large numbers at the site, the marsh is of fundamental importance for the survival of the local breeding population, primarily as a feeding and rearing area.

Dabbling ducks and their broods in particular (black duck, mallard, northern pintail and green-winged teal) busily seek out insect larvae trapped in the ice extraction pans. The common snipe, the killdeer that constantly cries out its name, the red-winged blackbird and the savannah sparrow also breed here.

ILES DE L'ESTUAIRE

Îles de l'estuaire National Wildlife Area extends over a distance of more than 120 km between the municipalities of Kamouraska and Bic. From west to east, it takes in Île de la Providence, Île Brulée, Rocher de l'Ouest, Rocher de l'Est, most of Grande Île, a small part of Long Pèlerin, Île aux Fraises and its reefs, much of Pot du Phare (Îles du Pot à l'Eau-de-Vie), Île Blanche and the largest part of Île Bicquette. Its total area is over 646 ha.

The islands are mainly covered with coniferous boreal forest dominated by white spruce and balsam fir. The kelp, rockweed and spartina marshes of some islands develop according to the height of tides and the depth of the water. Immense mud flats mainly on the south side of the islands are exposed at low tide.

About one hundred bird species, over half confirmed local breeders, are entered on the islands' particular list of ornithological sightings. We should start by mentioning that major fall concentrations of seaducks - sometimes numbering up to 50,000 and in large part formed of scoters and oldsquaws - are the greatest in the entire St Lawrence system for this species group. Also in the fall, over 5,000 dabbling ducks can be found in the vicinity of Île aux Fraises, Île Blanche and the Îles du Pot à l'Eau-de-Vie. For the black duck, this is one of the best sides along the St Lawrence between Montréal and Rivière-du-Loup.

In the forests during the breeding period, the commonest species are the white-throated sparrow, yellow-rumped and magnolia warblers, Swainson's thrush, winter wren and black-capped and boreal chickadees. The fox sparrow, a species with a boreal range, is particularly fond of sparse, low forest and brushland.

Birds that breed in colonies, such as the common eider, razorbill, black guillemot, great blue heron, black crowned night-heron and gulls, are especially dependent upon the protection given to the site by its national wildlife area status.

Together, the Îles Pèlerins, and Îles du Pot à l'Eau-de-Vie harbour the largest colonies of razorbills and black guillemots in the St Lawrence Estuary. The black-legged kittiwake colony of Long Pèlerin is the most westerly in Québec, while the razorbill colony, which contains more than 1,000 nests, is the largest in the estuary. Over 15,000 pairs of common eiders - from which we get eiderdown, the unique and highly prized insulating material - breed on the islands of the estuary. In terms of numbers, Île Bicquette ranks first in the St Lawrence Estuary and Gulf, accounting for nearly 9,000 pairs.

The region is also frequented by the beluga, a small white whale whose St Lawrence population is endangered and most sightings occur in the vicinity of Île aux Lièvres and Île aux Fraises. The south-western tip of Île aux Lièvres and the reefs of Île aux Fraises are believed by some people to be a beluga birthing and rearing area.

POINTE DE L'EST

As a vestige of the sole ecosystem of its kind in Québec, Pointe de l'Est National Wildlife Area is a major natural component of the Îles de la Madeleine in the southern part of the Gulf of St Lawrence.

The 996-ha site is composed primarily of sand, the area has been colonized to a large extent by marine vegetation. The plant that receives top billing in the sandy habitat is non other than beachgrass which, through its well-developed root system, is able to hold dunes in place, This truly unique landscape is characterized by the crowberry moor, the stunted forests of spruce and fir, the spartina, samphire and sedge of the saltwater ponds, the yellow pond-lily of the freshwater ponds, and the sphagnum moss of the marshes with their procession of heath and carnivorous plants.

For many shorebirds, the Îles de la Madeleine are an essential stopping point in their twice-yearly migrations, The Pointe de l'Est site, while it is important for waterfowl breeding, is first and foremost a major staging area. It is visited by many birds that are rare or at the edge of their range, including the stilt sand-piper, ruff, eared grebe, northern wheatear and Townsend's solitaire.

In late August, the Étang de l'Est plays host to daily gatherings of 300 to 500 black ducks, 100 to 200 blue-winged teals, greater scaups, common goldeneyes and red-breasted mergansers. The shallow waters of the outlying ponds and salt marshes mainly attract black-bellied plovers, semipalmated and white-rumped sandpipers and greater yellowlegs. The rough-legged hawk, American kestrel and snowy owl (Québec's bird emblem) fly over the dune habitat while they are migrating.

During the fall migration period, some shorebirds show a preference for specific habitats. For example, the lesser yellowlegs, short-billed dowitcher, least sandpiper, lesser golden-plover and pectoral sandpiper are most at home on the salt meadows. The whimbrel frequents the crowberry moor.

The presence of the piping plover on the sandy shores of Pointe de l'Est during the breeding period is one of the main justifications for protecting the habitats, since this bird is on the list of endangered species. The chance to see the horned grebe, a rarity, on the Étang de l'Est in the nesting season will no doubt thrill many visitors.

A sharp-eyed observer can also spot white-throated sparrows and dark-eyed juncos hiding under the forest cover, not to mention the cedar waxwings attracted by the ripe delights of the crowberry moors. Most abundant in the wildlife area are the savannah sparrows that feed in the more open dune habitat.

The islands and islets of the Étang de l'Est provide a space for the least sandpiper and gulls to breed. Colonies of herring and great black-backed gulls and common and arctic terns cans be found around the edges of the pond. Within the wildlife area breed mainly the black duck, northern pintail, red-breasted merganser and greater scaup. For the last species, this breeding site is an isolated station in eastern Canada.

DISCUSSION

The Canadian Wildlife Service continues to consolidate the network by acquiring new properties or by negotiating agreements with other federal departments. In the next two years, the Canadian Wildlife Service plans to create two new Wildlife Areas, one in the Varennes area and the other on in the Lake St.Pierre archipelago. Those territories will add about 1 000 hectares to the network.

WETLANDS CONSERVATION ALONG THE ST. LAWRENCE RIVER : AN EXEMPLARY PARTNERSHIP

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ABSTRACT

Over the past 40 years there has been unprecedented economic growth and prosperity in North America. One cost of this unchecked growth has been the loss of vast areas of natural habitat to residential, industrial and agricultural expansion. Wetlands are among the areas most critically affected by this economic development. As wildlife habitat, they are essential to the survival of a wide variety of native plant and animal species. They also play a broad ecological role in the environmental well-being of this continent. They buffer flooding and reduce soil erosion; they are natural reservoirs capable of preserving, purifying and receiving precious water supplies; and they play a crucial role in the early detection of potentially dangerous ecological imbalances.

Despite their many values functions, there is an urgent need to protect wetlands because they are being lost at alarming rates. Governments at all levels, and a growing number of private organizations and corporations, are reacting with increasing sensitivity to the need to protect these invaluable natural systems. The existence in Quebec of programs such as St. Lawrence Vision 2000, the North American Waterfowl Management Plan and the Fish Habitat Restoration Fund highlight a growing determination of many partners to preserve and enhance wetland habitats of the St. Lawrence River.

Key-words : Wetlands/habitats/conservation/partnership/St.Lawrence/fish/biodiversity/waterfowl.

INTRODUCTION

The present communication aims at emphasizing the goals of each of those programs and presenting the exemplary results of those partnerships for the protection, restoration and enhancement of wetlands of the St. Lawrence River.

ST. LAWRENCE VISION 2000

With the goal of pursuing depollution efforts undertaken in 1988 as part the St. Lawrence Action Plan, governments of Canada and Quebec have agreed to implement the St. Lawrence Vision 2000 action plan, the second 5-year phase of the SLAP. With a budget of $191 million, this new federal-provincial agreement aims at enhancing conservation, protection, depollution activities and recovering the ways that the St. Lawrence River and some of its tributaries are utilized.

By signing the St. Lawrence Vision 2000 agreement, which will expire on March 31, 1998, the two levels of governments intend to intensify measures for preventing pollution and conserving river ecosystems by favouring a more global approach towards managing the environment, and by ensuring the active participation of the community and the full range of partners involved. This broader approach entails seven areas of intervention : biodiversity, agriculture, community involvement, assistance for decision making, health, protection and restoration. Although the focus of programs and activities is still the St. Lawrence River itself, St. Lawrence Vision 2000 allows for interventions on seven tributaries, namely : the Assomption, Boyer, Chaudière, Richelieu, Saguenay, Saint-Maurice and Yamaska rivers.

The biodiversity component of St. Lawrence Vision 2000 is designed to implement the main provisions of the United Nations Convention on Biological Diversity. Focus is being placed on living species and ecosystems from the dual points of view of conservation and sustainable use. In addition, particular attention is being paid to threatened or vulnerable species whose numbers are low or declining as well as the habitats of these species. Finally, an effort is also being made to identify factors affecting biological diversity as well as remedial or mitigative measures.

The long-term objective of this component of St. Lawrence Vision 2000 is to preserve and maintain the biological diversity of the St Lawrence River ecosystem and this component seeks primarily to conserve 7,000 hectares of habitat.

RESULTS SLV 2000 - HABITAT PROTECTION (MARCH 31, 1996)

Sites	Protected area (ha)
Lac Saint-François	56,78
Îles Avelle-Wight-et-Hiam	62,28
Pointes Hébert et Goyette	11,92
Ruisseau St-Jean	46,28
Île de Grâce	212,78
Île Lapierre	47,90
Îles Millette et Stranham	13,70

Sites	Protected area (ha) (continue)
Baie Lavallière	15,04
Île Soyez	13,50
Île au Cochon	39,50
Île Saint-Jean	0,68
Léon-Provancher	40,00
Pointe-Platon	60,00
Baie de Saint-Augustin	23,00
Grands-Ormes	705,00
St-André-de-Kamouraska	15,01
Battures de l'Île-aux-Lièvres	287,17
Isle-Verte	19,55
Grand-Lac-Salé	2 350,00
Pointe-Heath	995,00
Pointe de l'Est	306,97
Cumulative Total	**5 322,06**

THE NORTH AMERICAN WATERFOWL MANAGEMENT PLAN

Governments at all levels, and a growing number of private organizations and corporations, are reacting with increasing sensitivity to the need to protect wetlands. Its was the need to preserve and enhance wetland habitat around the world that led to the signing, in 1986, of the North American Waterfowl Management Plan (NAWMP). This historic agreement commits Canada, the United States and Mexico to a long-term program to help increase waterfowl populations and preserve the habitats on which their survival depends. Under NAWMP a series of Joint Ventures have been put in place to protect and manage important wetland habitat in a particular region for the benefit of waterfowl and other wetland wildlife.

A total of 14 Joint Ventures have already been established. One of these is the Eastern Habitat Joint Venture (EHJV). Its aim is to protect and enhance those wetlands in eastern Canada that contribute significantly to the waterfowl and other migratory birds of the Atlantic Flyway and, to a lesser extent, to the Mississippi Flyway.

The southern area of eastern Canada contains 65 percent of the nation's human population, which results in some of the most stressed wetland ecosystems in the country. Unfortunately, these wetlands are also generally the most productive.

Most North America waterfowl species occur in eastern Canada at some time of the year. Whether for breeding, feeding, or staging, the availability of wetland habitat is critical to their survival.

Simply stated, the purpose of the Eastern Habitat Joint Venture is to secure the waterfowl resources of eastern Canada by maintaining and enhancing the abundance and quality of wetlands. This is being achieved through two programs - an intensive program and an extensive program.

The goal of the intensive program is to secure, enhance, or restore over 240,000 hectares of important wetland in priority areas.

The extensive program focuses on a much larger area (about 1.55 million hectares) through a mix of public education, conservation incentives to private landowners, and inter-governmental agreements to develop constructive land-use policies and make wetland values an integral component in the sustainable development of the landscape.
Over a 15-year period (1989-2004), the Eastern Habitat Joint Venture aims to conserve, approximately 1.8 million hectares of wetlands in eastern Canada. Complementary initiatives by federal and provincial agencies and non-government organizations such as Wildlife Habitat Canada and Ducks Unlimited Canada are expected to add significantly to the total wetland area that will benefit from improved stewardship during that time.

Within each EHJV program area the special biological features of the wetland is the deciding factor in designing projects to maximize the benefits to all species. The overriding objective is the enhancement of productivity, biodiversity and ecological integrity of the wetlands.

Quebec's EHJV program focuses on habitat securement and wetland restoration and enhancement along the St. Lawrence and Ottawa Rivers and is supported by an agricultural stewardship program. The Quebec Wildlife Foundation, local non governmental organizations and municipalities form an efficient partnership for the implementation of the EHJV program in Quebec.

RESULTS - EHJV
HABITATS PROTECTION AND ENHANCEMENT (1987-1995)

Project	Area Protection (ha)	Cost Protection $	Area Enhancement (ha)	Cost Enhancement $
Baie-du-Febvre Nicolet-Sud	404	492 000	215	925 000
Saint-Fulgence	63	291 000	-	-
Île du Milieu	94	217 000	-	-
Marais Desrochers	119	83 000	95	168 000
Lac McLaurin	342	1 022 000	-	-
Longue-Pointe	57	126 000	93	137 000
Rivière Marguerite	40	3 000	-	-
Saint-Barthélemy	334	856 000	-	-
Île Saint-Bernard	206	53 000	-	-
Commune de Baie-du-Febvre	326	50 000	62	500 000
Île Dupas	700	-	153	790 000
Îles aux Alouettes	11	-	6	14 000
Marais de Beauharnois	200	-	296	415 000
Îles du Pot à l'Eau de Vie	58	-	45	106 000
Île aux Fraises	7	-	7	85 000
Isle-Verte	59	-	114	145 000
Îles Les Pélerins	234	-	234	18 000
Îles aux Pommes	60	-	26	70 000
St-Gédéon	104	251 000	-	-

RESULTS - EHJV
HABITATS PROTECTION AND ENHANCEMENT (1987-1995) (continue)

Project	Area Protection (ha)	Cost Protection $	Area Enhancement (ha)	Cost Enhancement $
Baie Lavallière	14	67 000	-	-
Île Marie	45	86 000	45	47 000
Île Bouchard	43	26 000	43	15 000
Thurso	-	-	106	29 000
Varennes	-	-	233	106 000
Oie Blanche	-	-	20	68 000
Cumulative total	**3 520**	**3 623 000**	**1 793**	**3 638 000**

FISH HABITAT RESTORATION FUND (FRHAP)

The fish habitat restoration fund was established further to the fine levied against Tioxide Canada Inc. in May 1993 for unlawfully discharging toxic effluent into the St. Lawrence River. At the time, the court of Quebec ordered the company to pay a fine of $4 million, of which $3 million represented compensation for damage to fish and fish habitat. This ruling was unprecedented in Canadian jurisprudence for destruction of wildlife habitats.

In keeping with the court's decision, a steering committee composed of representatives of Environment Canada, Fisheries and Oceans Canada, the Quebec Department of the Environment and Wildlife, and the Quebec Wildlife Foundation, prepared a five-year management plan encompassing some 19 projects conforming to existing priorities for preserving and rehabilitating fish in the St. Lawrence.

To ensure that the fund's investments have a multiplier effect, the projects are carried out in partnership with conservation associations and other local organizations concerned. As proponents, they participate in managing and monitoring the work and are called on to identify potential sources of complementary funding, since the total amount required for the entire list of selected projects exceeds the sum received.

The fish habitat restoration fund is being used primarily for conservation and enhancement projects. These include protecting and restoring spawning grounds, re-establishing the free movement of fish, for example after spring thaw, developing techniques for reintroducing species that are at risk, acquiring and developing habitats such as marshes and islands crucial to the survival of the fishery resource, and restoring habitats to their natural state.
The area covered by the projects extends from the Lac Saint-Pierre, where Tioxide Canada's discharges had a particularly severe impact, and on upstream to Beauharnois. FRHAP is mandated to implement the 19 projects in the management plan approved by the court of Quebec. Consequently, no other projects are eligible at the present time. The list of projects in 1994-1995 is the following :

FISH HABITAT RESTORATION FUND

Programming - April 1, 1994 to March 31, 1995

Project	Type of project	Location
1. Ile Dupas (Phase 1)	Restoration of spawning grounds	Tracy-Sorel region
2. St-Eugène	Restoration of spawning grounds	Lac St-Pierre
3. Baie Lavallière	Restoration of spawning grounds	Tracy-Sorel region
4. Ile de Grâce, Ile aux Corbeaux	Habitat restoration	Tracy-Sorel region
5. Ruisseau St-Jean (Phase 1)	Restoration of spawning grounds	Montréal
6. Restoration of copper redhorse population and habitat	Restoration of species at risk	Rivière Richelieu
7. Islands in the Berthier-Sorel archipelago	Conservation	Tracy-Sorel region
8. St-Barthélemy	Conservation	Tracy-Sorel region
9. Ruisseau St-Jean	Conservation	Montréal
10. Pointes Hébert et Goyette	Conservation	Montréal
11. Rivière-aux-Pins	Conservation	Montréal
12. Fishway at Beauharnois dam	Feasibility study	Beauharnois
13. Baie du Febvre/Nicolet Sud (Phase 1)	Habitat restoration	Lac St-Pierre

IMPACT ANALYSIS AND RESTORATION PLANNING USING THE RIVERINE COMMUNITY HABITAT ASSESSMENT AND RESTORATION CONCEPT (RCHARC)

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Abstract

A new approach, the Riverine Community Habitat Assessment and Restoration Concept (RCHARC), is used for impact analysis and restoration planning for the main stem Missouri River. The RCHARC contrasts depth and velocity distributions at specific time intervals from a reference stream against distributions from different operational or channel alternatives in a target stream. The more nearly depth and velocity distributions for an alternative resemble distributions from the reference stream, the more highly that alternative is ranked. For this application, the reference system was the historical (preproject) Missouri River as defined by HEC-II analysis using preproject cross sections and preproject mean monthly flows. Reference monthly depth and velocity distributions are contrasted to monthly distributions associated with specific reservoir release alternatives routed through the existing channel. The results of the analysis allowed the Corps of Engineers to rank operational alternatives by their ability to provide hydraulic habitat most likely to support native warmwater fishes. The community-level perspective and whole-river descriptions used in the approach facilitate assessments of ecosystem integrity of target systems.

KEY-WORDS: Instream Flow Needs / RCHARC / Missouri River / Habitat Analysis / Physical Habitat Modeling / Stream Impact Assessment / Flow Requirements / Maintenance Flows / Fish Community

INTRODUCTION

Population growth and continued economic development are partially founded on continued development of water resources for power production, water supply, flood control, navigation, and other benefits. Streams and rivers of many regions of the United States are viewed as a preferred source of water supply to promote economic development and support population growth (Petts 1984). Unmanaged stream regulation and diversion have the potential to severely impact environmental quality and fish and wildlife resources. Increased water resources development has intensified the conflict between economic benefits of stream diversion and regulation and the need to protect and maintain the integrity of lotic ecosystems (Petts 1984).

Impact assessment methodologies that are quantifiable, repeatable, accepted, and defensible are required to mediate conflicts between water resources development and natural resource preservation (Hobbs et al. 1989). The U. S. Army Corps of Engineers, Missouri River Division (MRD), controls, maintains, and conserves water resources to provide for flood control, navigation, irrigation, power generation, recreation, water quality, water supply, and fish and wildlife protection and enhancement by regulating the releases from dams on the main stem Missouri River. Economic development in the basin, plus changing demographic, social and land use patterns, is placing increased demands on the natural resources in the system. Operation of the main stem Missouri River system of dams impacts several different natural resource categories including inpool reservoir fisheries, wetlands, wildlife, and downstream fish resources. Efforts by MRD to continue fostering economic development in the basin while simultaneously protecting environmental resources require predictive tools that can be used to balance the developmental and environmental needs of the region. We present an application of the RCHARC to the tailwater of Gavin's Point Dam to assess the effects of different releases on preregulation habitat for warmwater fishes. However, the RCHARC can also be used to provide a framework for habitat restoration of the Missouri River or to describe the long-term effects of river regulation on physical habitat.

The RCHARC links broad depth and velocity patterns described by frequency distributions (similar to Hogan and Church 1989) to community response by building on the observation that different species of fishes seem to prefer different parts of depth or velocity gradients (Bain et al. 1991). Some fishes prefer shallow water having low velocity, others may select deep, faster velocity areas, whereas the remainder of the community may prefer deep, slow water or shallow, fast areas. Figure 1 illustrates the conceptual relationship between a depth gradient and habitat requirements of a hypothetical group of species represented by an ordinated set of habitat suitability curves (represented as species A, B, C, etc.). The right abscissa represents the habitat value from 0.0 to 1.0 for depth for each species. The left abscissa represents the percent distribution of each depth increment. The relative value of the habitat for each species can be determined by how much of the frequency distribution falls within its suitability curve, or restated, the composition of this hypothetical group of species is determined by the depth distribution. Thus, the composition of the fish community will be determined by long-term patterns of depth and velocity frequency distributions, all other factors being equal. Changes in the frequency distribution of depth and velocity will result in associated changes in the warmwater fish community. For example, a shift in the frequency distribution of depth that reduces the amount of shallow water will favor species that inhabit deeper water.

METHODS AND RESULTS

Application of the RCHARC to the regulated Missouri River required four steps:

<u>Step one.</u> A comparison reference was selected against which the project alternatives could be contrasted. The reference was considered to provide ideal habitat conditions, both in terms of channel configuration and seasonally varying flow

pattern (in this case, defined by mean monthly flows), for the aquatic community in the project river system. The reference was selected in coordination with state and Federal resource agencies. For the Missouri River application, the reference standard was the preproject Missouri River as recreated using hydraulic simulation (HEC-II) based on historical cross-section data archived by the Omaha District of the Corps of Engineers. However, for other applications of the concept, the reference could be a nearby river system, reaches of the river upstream or downstream of the project and not impacted by the project, or the project river reach but evaluated in a "without project" condition.

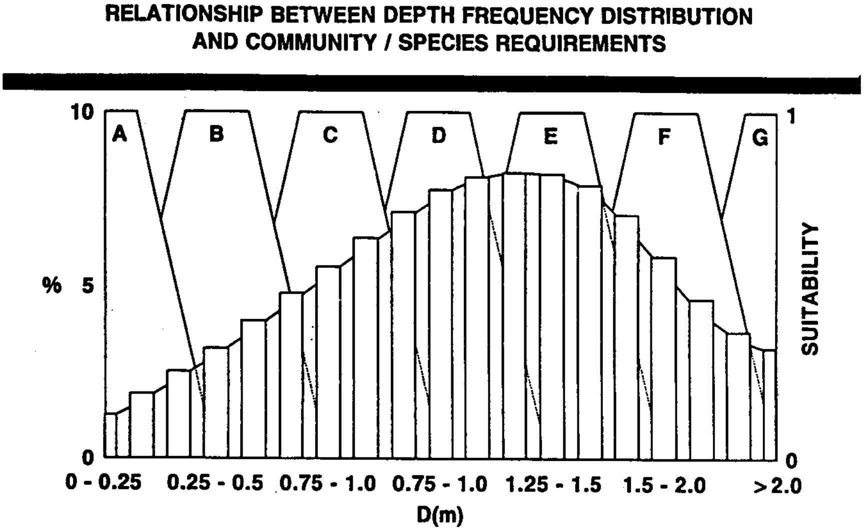

Figure 1. Conceptual relationship between a depth gradient and habitat requirements of a hypothetical group of species represented by an ordinated set of habitat suitability curves (represented as species A, B, C, etc.).

Step two. Hydrologic and hydraulic features of the reference standard having fish habitat value were described and summarized as an annual series of monthly depth or velocity frequency distributions (e.g., Figure 2). Standard methods of stream gaging and hydraulic simulation were used to describe cell-by-cell depths and velocities (Milhous et al. 1991) for each monthly flow. Eight transects combined into four channel categories - wide, narrow, transitional, and divided - were utilized to describe preproject habitat conditions in the Missouri River. The results from each channel category were expanded by a weighting factor reflecting the relative proportion of the entire study reach that each category represented.

Step three. A similar approach was used to describe hydrologic and hydraulic features of the project alternatives. Figure 2 presents a comparison of depth distributions for the narrow channel category for the preproject (solid line), existing operation in the present channel (dotted line), and depth distribution most nearly like the preproject depth distribution that can be achieved through operational control at the dam (dashed line). Project and preproject distributions are based on August median flows.The eight transects used to characterize the preproject Missouri River were resurveyed in 1992 under high and low flows to characterize the project operational alternatives. Velocity and depth distributions at intermediate flows were determined using methods described in Step two.

Step four. The habitat value of each project alternative was determined by the similarity of its depth or velocity distributions to the distributions of the reference system on a monthly basis. Table 1 presents correlation coefficients that relate depth distributions at incremental project discharges. (does not include overbank flows) to the preproject depth distributions for the narrow channel category of the Gavin's Point tailwater for a median water year. Maximum correlation coefficients in each column (bold) generally follow the preproject hydrograph. The effect of a particular operations plan can be determined by summing (or integrating in some other way) the correlation coefficients. Optionally, the coefficients can be adjusted by a weighting factor based on the similarity of the project channel topwidth to the preproject channel topwidth. This optional weighting factor assures that habitat quantity (topwidth) is considered as well as habitat quality (depth/velocity distributions). The more similar an alternative was to the reference stream, the higher that alternative was ranked. We employed Pearson product-moment correlation analysis (SAS Institute1988) of the velocity distributions to determine

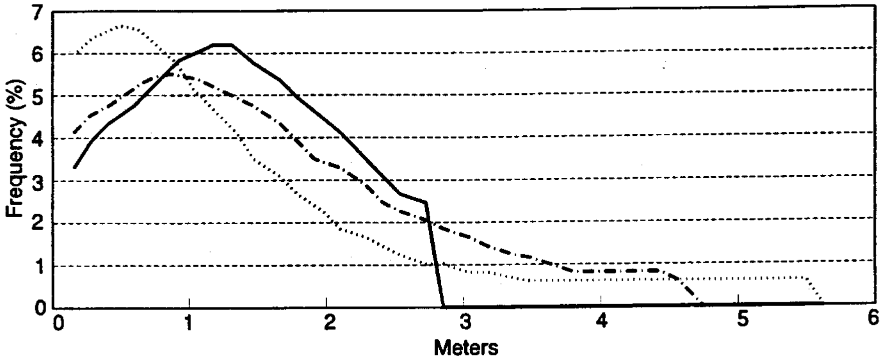

Figure 2. Comparison of depth distributions for the narrow channel category.

Table 1. Correlation coefficients that relate depth distributions at incremental project discharges.

Q	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC
6,000	0.9	0.7	-0.2	-0.2	-0.4	-0.4	-0.3	0.6	0.7	0.7	0.7	**0.9**
8,000	0.9	0.8	-0.2	-0.1	-0.3	-0.3	-0.2	0.7	0.8	0.8	0.8	0.9
10,000	**0.9**	0.9	-0.1	-0.1	-0.3	-0.3	-0.1	0.8	0.9	0.8	0.9	0.9
12,000	0.9	0.9	-0.0	0.1	-0.2	-0.2	-0.0	0.9	0.9	0.9	0.9	0.9
14,000	0.8	0.0	0.1	0.2	-0.1	-0.1	0.1	0.9	1.0	1.0	1.0	0.7
16,000	0.8	**1.0**	0.2	0.3	0.0	-0.1	0.2	**1.0**	**1.0**	**1.0**	**1.0**	0.7
20,000	0.8	0.8	-0.0	0.1	-0.2	-0.3	-0.1	0.7	0.8	0.7	0.8	0.8
24,000	0.8	0.7	0.0	0.1	-0.2	-0.3	-0.0	0.7	0.7	0.7	0.7	0.8
28,000	0.8	0.8	0.1	0.2	-0.1	-0.2	0.1	0.7	0.8	0.8	0.8	0.8
32,000	0.7	0.8	0.3	0.4	0.1	-0.0	0.3	0.8	0.8	0.8	0.8	0.7
36,000	0.7	0.8	0.5	0.6	0.3	0.2	0.5	0.9	0.8	0.8	0.8	0.6
40,000	0.5	0.7	**0.7**	**0.8**	**0.5**	**0.4**	**0.7**	0.8	0.7	0.7	0.7	0.4
46,000	0.5	0.5	0.3	0.3	0.1	0.1	0.3	0.5	0.5	0.5	0.5	0.5
50,000	0.5	0.6	0.4	0.5	0.3	0.2	0.4	0.6	0.6	0.6	0.6	0.4

similarity between the reference and target systems. We tested other measures of similarity but the ranking of alternatives did not change. For the Missouri River application, total impact of each operational alternative was determined by summing the velocity correlation coefficients (depth coefficients provided similar patterns) over the 93-year hydrologic period of record available for the analysis. We presently employ bivariate descriptions of depth and velocity and compare them using the Canberra Coefficient.

DISCUSSION

Use of correlation coefficients or other similarity coefficients to describe the similarity between the reference and project alternatives allows complex patterns of habitat to be described as single numbers. This simplification facilitates incorporation of the analysis results into water resources management decisions. Correlation coefficients that range from 1.0 (perfect correlation) to -1.0 (inverse correlation) can be easily rescaled. By rescaling the correlation coefficients between 0.0 to 1.0, the effects of reservoir operation on fish habitat can be expressed as either a "penalty function" or "value function". For penalty functions, the correlation coefficients are scaled so that the highest correlation coefficients have the smallest penalty. For value functions, the highest correlation coefficients are scaled to have the highest value. Use of penalty functions allows the results of the RCHARC analysis to be considered in reservoir optimization studies. Use of value functions allows for easy assessment of each operational/channel alternative against other beneficial uses of reservoir storage also couched as value functions. Value/penalty functions can be presented separately for different reaches or an entire river system.

The RCHARC analysis facilitates description of impact and identification of operational and structural mitigation (Figure 2). Impact is defined as differences between project and preproject depth/velocity distributions. Maximum operational mitigation is defined as differences between existing operation distributions and the operation that provides depth distributions most nearly like the preproject depth distributions. Potential restoration (channel modification) is defined as differences between optimum operation and preproject conditions. Long-term changes in the hydraulic environment between the preproject Missouri River and project Missouri River can be described with monthly plots of depth and velocity frequency distributions at median flows or other flows having biological importance.

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