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EFFETS DE LA RÉCOLTE FORESTIÈRE SUR 50% DE LA SUPERFICIE DE PETITS BASSINS VERSANTS SUR LES DÉBITS DE POINTE ET LA QUALITÉ DE L'EAU, FORÊT MONTMORENCY, QUÉBEC

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Résumé

La récolte forestière peut nuire à l'intégrité des écosystèmes aquatiques en modifiant le régime d'écoulement des cours d'eau ou en affectant leur qualité. Selon la revue de littérature, il est supposé que la récolte forestière jusqu'à 50 % de la superficie du bassin versant n'augmente pas les débits de pleins bords de plus de 50 %, seuil auquel la morphologie des cours d'eau et, par conséquent, les écosystèmes aquatiques sont réputés être affectés significativement. Il existe aussi un intérêt grandissant pour adopter une approche similaire en terme d'intensité de récolte afin d'assurer le maintien des bonnes caractéristiques physiques et chimiques du milieu aquatique. Qui plus est, nous avons émis l'hypothèse que les coupes localisées près du réseau hydrographique affectent plus fortement les débits de pointe et la qualité de l'eau que celles plus éloignées. La présente étude consiste à documenter les effets sur les débits de pointe et la qualité de l'eau de la récolte sur 50% de l'aire de petits bassins versants de moins de 50 ha, localisés à la forêt Montmorency, Québec. La récolte a été effectuée selon les pratiques d'aménagement respectueuses de la régénération préétablie et sur couvert de neige afin de réduire les perturbations du sol. Des lisières boisées de 20 m de large ont aussi été maintenues le long des ruisseaux permanents. Un bassin versant contigu a été conservé à son état naturel afin de servir comme témoin. Les effets de la récolte à court terme ont été mesurés durant deux saisons estivales suivant cinq années de calibrage. Des augmentations non significatives des débits de pointe ont été observées sur tous les bassins traités et les augmentations des débits de pleins bords ont toutes été maintenues sous le seuil de 50%. La récolte a toutefois provoqué de très faibles augmentations et diminutions significatives des températures estivales journalières maximales et minimales selon les bassins traités, alors que les variations diurnes ont significativement diminué. La plupart des autres modifications des caractéristiques physico-chimiques de l'eau sont faibles par rapport aux critères reconnus pour la protection de la vie aquatique. Les concentrations en NO_3^- et K^+ ont fortement augmenté, particulièrement durant le deuxième été après la récolte, alors que de faibles augmentations significatives ont été observées pour la conductivité et les concentrations en SO_4^{2-}, Mg^{2+} et Fe_{total}. Le pH a pour sa part diminué significativement mais faiblement tandis que la concentration en

PO_4^{3-} est demeurée relativement stable sur la plupart des bassins. Certaines observations suggèrent une augmentation de la concentration de sédiments en suspension, mais un manque de données d'avant récolte empêche de confirmer ce résultat. La proximité des aires de coupe du réseau hydrographique n'a pas affecté l'augmentation des débits de pointe ni les modifications des paramètres de qualité de l'eau. La présence d'un chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement n'a pas contribué à l'augmentation des sédiments en suspension.

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SECTION 1

Synthèse

1. Introduction

La pression anthropique s'accroissant depuis plus de 30 ans sur la forêt boréale et ses milieux humides (Schindler, 1998), il apparaît primordial d'améliorer l'évaluation des impacts des activités humaines sur ces ressources. La majorité des études nord-américaines portant sur l'hydrologie forestière ont été menées aux États-Unis et dans l'ouest canadien (Bosch et Hewlett, 1982; Guillemette et al., 2005; Hornbeck et al., 1993; Sahin et Hall, 1996; Stednick, 1996), dans des conditions hydrologiques, climatiques et biologiques propres à chaque région. Ce projet de recherche, en s'adressant à un cas particulier où la moitié de la superficie de bassins versants a été récoltée, vise à combler en partie la lacune des connaissances, tant au niveau de la quantité que de la qualité de l'eau, dans le cas de la forêt boréale québécoise (Coulombe et al., 2004). Les résultats de cette étude serviront à construire des outils de gestion pour l'aménagement du territoire et d'appui à la certification des opérations forestières. L'ajout de nouvelles connaissances permettra ainsi de mieux répondre aux exigences régionales et mondiales du développement durable, tant au niveau des ressources naturelles que de l'industrie.

2. Revue de littérature et problématique

Il existe une préoccupation largement répandue concernant les impacts réels, potentiels ou présumés des opérations forestières sur le régime d'écoulement de l'eau et sa qualité. La coupe d'arbres modifie généralement le bilan hydrique du bassin versant en diminuant l'évapotranspiration (i.e., l'eau évaporée suite à l'interception et la transpiration par la végétation), et en augmentant le contenu en eau du sol et le volume d'écoulement en ruisseaux (Harr et al., 1979; Ice, 1999; Jones et al., 2000; Wright et al., 1990). Les opérations forestières contribuent aussi à augmenter les débits de pointe lors d'événements pluvieux en diminuant le taux d'infiltration d'eau dans le sol compacté par la machinerie forestière et en accélérant la convergence du ruissellement dans les fossés et ornières (Beschta et al., 2000; Guillemette et al., 2005; Hornbeck et al., 1997; Jones et

Grant, 1996; Thomas et Mehagan, 1998; Wright et al., 1990). Sachant que l'augmentation des débits de pointe est généralement corrélée à la proportion de l'aire du bassin versant coupé (Beschta et al., 2000; Plamondon, 2004), une fraction suffisamment importante de la superficie du bassin doit tout de même être récoltée, soit plus de 10 à 15%, afin de constater des augmentations significatives des pointes (MacDonald et al., 1997). Il est reconnu que l'accroissement des débits de pointe est surtout important pour les faibles pointes (Beschta et al., 2000; Jones et Grant, 1996; MacDonald et al., 1997; Thomas et Mehagan, 1998), mais il a toutefois été récemment suggéré que les pointes importantes pouvant modifier la morphologie des cours d'eau pourraient aussi être augmentées (Beschta et al., 2000; Guillemette et al., 2005; MacDonald et al., 1997; Plamondon, 1993; Plamondon, 2004). Puisque le débit de pleins bords, caractérisé par une période de retour d'environ 1,5 à 2 ans, est responsable des caractéristiques morphologiques des cours d'eau (Dunne et Leopold, 1978), il a été rapporté dans la littérature qu'une augmentation de 50% de ce débit de pleins bords pourrait modifier significativement la morphologie des cours d'eau (Faustini, 2000; Ice, 1999) et affecter la faune aquatique (WFPB, 1997). Certaines études ont d'ailleurs démontré des effets néfastes sur les populations de poissons à la suite de la perturbation des frayères due à la coupe forestière (Rieman et McIntyre, 1993; Scrivener et Tripp, 1998). La revue de littérature suggèrent que le seuil d'augmentation de 50% du débit de pleins bords ne serait atteint que lorsque plus de 50% de la superficie du bassin versant est récoltée (Guillemette et al., 2005; Plamondon, 2004). L'hypothèse d'une augmentation plus importante des débits de pointe lorsque la récolte est effectuée près du réseau hydrographique a aussi été suggérée (Plamondon, 1993), puisque le ruissellement provenant des zones saturées le long des cours d'eau répond plus rapidement aux précipitations (Dunne et Leopold, 1978).

En ce qui concerne la qualité de l'eau, la récolte forestière peut accroître la température de l'eau en réduisant l'ombrage au ruisseau par l'enlèvement de la végétation riparienne (Beschta et Taylor, 1988; Brown et Krygier, 1971; Plamondon et al., 1982; Swift et Messer, 1971). L'intensification du rayonnement solaire atteignant le sol à la suite de la perte de la canopé peut aussi jouer un rôle déterminant en réchauffant l'eau contenue dans le sol qui s'écoule ensuite vers les cours d'eau (Hartman et Scrivener, 1990; Ice, 1999; St-Hilaire et al., 2000). Une augmentation importante de la température de l'eau peut être nuisible pour les poissons en causant une réduction de leur activité et de leur taux de reproduction (EPA, 1986), ou bénéfique en augmentant leur émergence et leur taux de croissance (Hartman et Scrivener, 1990). L'accroissement de la chaleur et de l'humidité du sol après la coupe peut aussi provoquer une augmentation dans les populations des micro-organismes (Martin et al., 2000), qui à leur tour peuvent augmenter la nitrification et donc la concentration en nitrates (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001). La production primaire peut ainsi être stimulée, ce qui peut être bénéfique ou nuisible pour la vie aquatique, selon la concentration en nitrates atteinte (EPA, 1986; MDDEP, 2006). Pour sa part, la concentration en phosphates demeure généralement relativement stable (Hartman et al., 1996; Seto, 2005; Swank et al., 2001) et ne contribue donc pas à la stimulation de la production primaire. L'intensification de la nitrification produisant de l'acide nitrique, de même que la décomposition des débris de coupe, peuvent provoquer une diminution du pH (Feller, 2005; Martin et al., 2000). La disponibilité de certains nutriments, la toxicité de certains polluants et les organismes aquatiques eux-mêmes sont sensibles aux variations du pH (EPA, 1986; Feller, 2005). La conductivité étant un bon indicateur de la concentration des solides en suspension (Plamondon, 1994), elle reflète la concentration des principaux ions présents dans l'eau (EPA, 1986). La récolte forestière, en augmentant la concentration de plusieurs ions tels que le potassium (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001), le magnésium (Jewett et al., 1995; Plamondon, 1994) ou le fer (Fe_{total}) (Plamondon, 1994), peut accroître la conductance électrique de l'eau (Plamondon, 1982). Certains de ces ions peuvent devenir nuisibles pour la faune aquatique à des concentrations importantes (EPA, 1986; Plamondon, 2002). Pour leur part, les sulphates diminuent généralement après la coupe, principalement à cause de l'intensification de la nitrification réduisant le pH du sol et accroissant ainsi sa capacité d'adsorption (Feller, 2005; Likens et al., 2002; Welsch et al., 2004).

Les opérations forestières peuvent finalement accélérer l'érosion en exposant le sol à nu (Croke et Hairsine, 2006), ce qui résulte généralement en une augmentation des sédiments en suspension dans l'eau (Beschta, 1978; Brown et Krygier, 1971). La construction de chemins est fréquemment considérée comme la principale source de sédiments (Brown et Krygier, 1971; Megahan et Kidd, 1972; Reid et Dunne, 1984). L'augmentation des sédiments en suspension peut être nuisible pour l'habitat aquatique en colmatant les frayères et en perturbant les invertébrés (Beschta, 1978; Ice, 1999), ou encore en réduisant la production primaire en restreignant la pénétration de la lumière dans la colonne d'eau (EPA, 1986).

Tous ces impacts de la récolte forestière sur la qualité de l'eau sont considérés comme étant généralement faibles en ce qui concerne les écosystèmes aquatiques (Plamondon, 1993). Les modifications des paramètres physico-chimiques dépassent rarement les critères reconnus pour la protection de la vie aquatique (EPA, 1986; MDDEP, 2006). Plusieurs études n'ont démontré aucune corrélation entre l'intensité de la coupe ou des opérations forestières et certaines caractéristiques de qualité de l'eau des ruisseaux (Degenhardt et Ice, 1996; Martin et al., 1981). Toutefois, il a été suggéré que les changements d'utilisation du territoire près des cours d'eau et sur les zones humides a plus d'effets sur les apports diffus de substances chimiques que les changements effectués sur les aires plus éloignées ou mieux drainées (Basnyat et al., 1999; Plamondon, 1993).

3. Objectifs et hypothèses

L'objectif de cette étude consiste à mesurer les changements des débits de pointe et des paramètres de qualité de l'eau en ruisseaux suivant la récolte sur 50% de l'aire de bassins versants situés dans la forêt boréale québécoise. L'étude s'attarde également à vérifier si les changements des caractéristiques physiques et chimiques de l'eau dépassent les critères reconnus pour la protection de la vie aquatique (EPA, 1986; MDDEP, 2006). L'hypothèse à valider est que la récolte jusqu'à 50% de la superficie du bassin limite les augmentations du débit de pleins bords sous 50%. L'étude vise aussi à déterminer si les coupes localisées près du réseau hydrographique affectent plus fortement les débits de pointe et la qualité de l'eau que celles plus éloignées et si la présence d'un chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement contribue à l'augmentation des sédiments en suspension.

4. Méthodes

Le site d'étude est localisé dans le bassin expérimental du ruisseau des Eaux-Volées (BEREV) de la forêt Montmorency, situé à 80 km au nord de la ville de Québec (Plamondon et Ouellet, 1980) (section 2, articles 1 et 2). La topographie y est caractérisée par un plateau d'altitude moyenne de 750 m et de collines pouvant atteindre 1000 m. Le sol est principalement de type podzol orthique ferro-humique (Plamondon et Naud, 1975) tandis que le sous-sol est formé de roches ignées et de dépôt glaciaire (Rochette, 1971). Le climat froid et humide, avec une température moyenne annuelle de 0,3°C et des précipitations moyennes annuelles de 1544 mm (Guillemette et al., 1998), classe le secteur dans le domaine bioclimatique de la sapinière à bouleau blanc (Robitaille et Saucier, 1998).

La méthode des bassins versants jumelés, traités et témoins, avec calage avant la récolte, a été utilisée afin d'isoler les changements provoqués par la coupe forestière, tant au niveau des débits de pointe que des paramètres de qualité de l'eau (Plamondon, 1993; Reinhart, 1967). La végétation a été récoltée sur trois bassins versants sur couvert de neige durant l'hiver 2004 selon les pratiques d'aménagement favorisant la protection de la régénération et des sols (section 2, articles 1 et 2). Des lisières boisées de 20 m de large ont été conservées le long des ruisseaux permanents. La récolte a été effectuée près ou loin du réseau hydrographique selon les bassins. Un bassin versant contigu a été maintenu à son état naturel afin de servir comme témoin.

Le débit a été mesuré à chaque heure durant cinq saisons estivales (de juin à octobre) avant la récolte (de 1999 à 2003) et durant les deux étés suivants (2004 et 2005) (section 2, article 1). De plus, le débit a été mesuré dans le cours d'eau drainant deux des trois bassins traités formant ainsi un bassin versant plus large en partie coupé près de son réseau hydrographique et en partie coupé loin de son réseau hydrographique. La température de l'eau a été mesurée à chaque heure durant les deux étés suivant la coupe, mais les mesures d'avant coupe ne sont disponibles que pour un seul été (2003) (section 2, article 2). Des échantillons d'eau ont été récoltés à une fréquence d'environ deux semaines durant les étés 2000 et 2002 avant la coupe et à intervalle de deux à trois fois par semaine durant les deux étés suivants la coupe. Similairement au débit, les échantillons ont été recueillis sur un bassin plus large incluant deux bassins dont la proximité de la coupe du réseau hydrographique est différente. Les échantillons ont aussi été récoltés à l'amont et à l'aval d'un chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement sur un des bassins coupés. Le pH, la conductivité, les sédiments en suspension et les concentrations en ions NO_3^-, PO_4^{3-}, K^+, Mg^{2+}, Fe_{total} et SO_4^{2-} ont été mesurées en laboratoire à partir des échantillons d'eau. Les mesures et la récolte d'échantillons ont toutes été effectuées à l'exutoire des bassins.

Diverses méthodes statistiques ont été employées afin de détecter les changements entre les mesures d'avant coupe et celles d'après coupe. Pour les débits de pointe, les droites de régression linéaire d'avant coupe entre les bassins traités et le bassin témoin ont été comparées à celles d'après coupe à l'aide du test de Z (section 2, article 1). Les températures mesurées après la coupe ont été comparées avec les températures prédites par la relation d'avant coupe entre les bassins traités et témoin à l'aide du test de *t* pairé et le test des rangs signés de Wilcoxon (section 2, article 2). Les caractéristiques chimiques de l'eau ont été analysées en soustrayant les valeurs mesurées sur les bassins traités d'avec celles mesurées aux mêmes dates sur le bassin témoin et en comparant les différences d'avant coupes d'avec celles d'après coupe à l'aide du test de *t* et du test de Wilcoxon-Mann-Whitney. Dû au manque de données d'avant coupe, les sédiments en suspension ont seulement été analysés selon leurs probabilités de non-dépassement des critères de qualité de l'eau pour la protection de la vie aquatique du Ministère du Développement Durable, de l'Environnement et des Parcs (MDDEP).

5. Résultats

Les différences non significatives entre les pentes et les ordonnées à l'origine des droites de régression d'avant et d'après coupe ont indiqué qu'il n'y avait pas d'augmentations moyennes significatives des débits de pointe après la coupe (section 2, article 1). De plus, les accroissements des débits de pleins bords n'ont, dans aucun cas, dépassé le seuil de 50%. Un seul événement pluvieux a provoqué des augmentations ponctuelles significatives des débits de pointe sur tous les bassins coupés. Les pointes individuelles y ont excédé la borne supérieure de la limite de confiance de 95% de la droite de régression des débits de pointe d'avant coupe.

Les températures journalières maximales et minimales de l'eau ont faiblement augmenté ou diminué significativement selon les bassins traités (section 2, article 2). Sur un des bassins, alors que les températures journalières maximales moyennes sont demeurées stables, les températures élevées ont augmenté significativement alors que les températures faibles ont diminué significativement. La même tendance a été observée sur le même bassin pour les températures journalières minimales. Dans aucun cas les changements n'ont excédé 1°C ou 17% des valeurs d'avant coupe. La variation diurne de la température (différence entre la température journalière minimale et maximale) a quant à elle diminué significativement, de près de 0,5°C ou 39%, sur la majorité des bassins traités. Les résultats des tests statistiques sont présentés dans la section 3: Annexes (tableaux A.1. à A.7).

Les concentrations en nitrates (NO_3^-) et en potassium (K^+) se sont fortement accrues, surtout durant le second été suivant la récolte forestière, avec des augmentations respectives atteignant 0,35 $mg{\bullet}l^{-1}$ et 0,63 $mg{\bullet}l^{-1}$, correspondant à 6089% et 283% des valeurs d'avant coupe (section 2, article 2). Les concentrations en magnésium (Mg^{2+}) et en fer (Fe_{total}) ont aussi surtout augmenté durant le deuxième été suivant la récolte, mais plus faiblement. Les augmentations significatives maximales ont été de 0,067 $mg{\bullet}l^{-1}$ et 0,139 $mg{\bullet}l^{-1}$ pour le magnésium et le fer respectivement, ce qui correspond à 19% et 71% des valeurs d'avant coupe. La conductivité a suivi la même tendance, avec des augmentations atteignant 4,2 $\mu S \cdot cm^{-1}$ ou 26% des valeurs d'avant coupe durant le deuxième été après la récolte. La concentration en sulphates (SO_4^{2-}) a aussi augmenté significativement sur certains des bassins traités, mais dans ce cas-ci, similairement pour les deux étés après la coupe. Les augmentations ont atteint 0,88 $mg \cdot l^{-1}$ ou 58% des valeurs d'avant coupe. La concentration en phosphates (PO_4^{3-}) est généralement demeurée relativement stable, mais quelques faibles augmentations ne dépassant pas 0,007 $mg \cdot l^{-1}$ ou 30% les valeurs d'avant coupe ont tout de même été observées. Le pH est pour sa part le seul paramètre ayant diminué significativement mais faiblement pour la plupart des bassins traités (ce qui suggère toutefois une augmentation de la concentration en ions H^+), sans décroître de plus de 0,4 unité ou 7% des valeurs d'avant coupe. Les résultats des tests statistiques sont présentés dans la section 3: annexes (tableaux A.8 à A.15).

La concentration des sédiments en suspension a excédé le critère de vie aquatique chronique du MDDEP de 4 à 8 fois après la récolte forestière selon les bassins traités, ce qui correspond à des probabilités de non-dépassement variant entre 0,91 et 0,95 (section 2, article 2). Dans le cas de la toxicité aiguë de la vie aquatique, le critère a été dépassé seulement une fois et sur un seul bassin, correspondant à une probabilité de non-dépassement de 0,99. Sur le bassin témoin, aucun critère n'a été dépassé.

6. Discussion

La majorité des études nord-américaines et mondiales ont observé des augmentations des débits de pointe (Beschta et al., 2000; Guillemette et al., 2005; Hornbeck et al., 1997; Jones et Grant, 1996; Thomas et Mehagan, 1998; Wright et al., 1990), et ce lorsqu'au minimum 10 à 15% du bassin versant était récolté (MacDonald et al., 1997). Les augmentations apparentes des débits de pointe mesurées suite à la récolte sur 50% de l'aire des bassins versants à la forêt Montmonrency ne sont pas significatives. Il est toutefois possible que la courte période d'après coupe (seulement 2 ans) par rapport à la plus longue période de calibrage (5 ans) n'ait pu affecter les résultats des tests statistiques. Afin d'obtenir des résultats plus fiables, et par souci de reproductibilité, un plus grand nombre de bassins versants coupés et témoins seraient aussi nécessaires. Similairement à la forêt Montmorency, des augmentations non significatives ont été observées sur d'autres sites d'étude, même lorsque plus de 15% du bassin versant était récolté (Moore et Scott, 2005; Troendle et al., 2001).

Bien que l'augmentation du débit de pleins bords n'ait pas excédé le seuil de 50%, celle-ci a atteint 49% sur un des bassins traités. Cette valeur limite suggère que la récolte n'aurait pu être plus intense et dépasser 50% de l'aire du bassin versant. Toutefois, étant donné que les augmentations moyennes des débits de pointe ne sont significatives pour aucun des bassins coupés, et donc de même pour les débits de pleins bords, la recommandation de Plamondon (2004) qui consiste à restreindre les aires de coupe sur 50% de la superficie des bassins versants est possiblement trop limitante du point de vue de la récolte forestière. Les bonnes pratiques d'aménagement, tel que pratiquées à la forêt Montmorency, doivent toutefois être respectées afin de protéger la régénération et les sols. Sur les aires sensibles, la coupe devrait donc être effectuée sur couvert de neige afin d'éviter le compactage du sol. Enfin, la lisière boisée de 20 m devrait être maintenue le long des cours d'eau permanents.

Les augmentations ponctuelles significatives des débits de pointe ayant eu lieu sur tous les bassins coupés ont été provoquées par la pluie particulièrement intense du 31 août 2005, correspondant à la queue de l'ouragan Katrina. Des augmentations individuelles entre 59% et 139% selon les bassins ont été observées. Les périodes de retour indiquent qu'en absence de récolte, tel que prédit par les droites de régression d'avant coupe, l'événement aurait causé des pointes approximant les débits de pleins bords sur tous les bassins. La pluie résultant de Katrina, dont l'intensité a dépassé 7 $mm \cdot h^{-1}$ durant les 15 heures qu'a duré l'événement et 15 $mm \cdot h^{-1}$ durant les 5 heures les plus intenses, combinée à la récolte forestière sur 50% de l'aire des bassins versants, est donc responsable du dépassement du seuil d'augmentation acceptable de 50% du débit de pleins bords. La faible dimension des bassins (< 50 ha), provoquant généralement une réponse rapide de l'écoulement en ruisseau suite aux précipitations, a aussi participé aux fortes augmentations observées. D'ailleurs, le bassin 7.5 ayant la plus petite taille (11,5 ha) a subi l'augmentation maximale de 139%. L'étroitesse de ce bassin, en réduisant le temps de résidence de l'eau de surface et souterraine, a aussi probablement joué un rôle non négligeable. L'événement pluvieux Katrina n'est toutefois pas le plus important ayant eu lieu durant les deux étés suivant la récolte. Quatre autres précipitations ont causé des débits de pointe plus importants sur le bassin témoin, mais les augmentations sur les bassins coupés ne se sont pas avérées ponctuellement significatives. Ce résultat est en accord avec la tendance observée par d'autres auteurs voulant que les pointes importantes réagissent moins à la récolte forestière que les pointes plus faibles (Beschta et al., 2000; Jones et Grant, 1996; MacDonald et al., 1997; Thomas et Mehagan, 1998) et qu'il existe d'autres facteurs tels que les caractéristiques des hyétogrammes, les conditions initiales d'humidité du sol, les pentes des différents versants, les essences forestières, les types de sols, pour ne nommer que ceux là, qui peuvent significativement affecter la réponse d'un bassin versant.

À la forêt Montmorency, une augmentation de 50% du débit de pleins bords correspond à un passage d'un débit ayant une période de retour de 1,5 ans à un débit ayant une période de retour de 2,6 ans. En conditions naturelles, le débit ayant une période de retour de 2,6 ans devrait se produire en moyenne 30,8 fois sur une révolution de coupe de 80 ans. Puisqu'un seul événement (queue de l'ouragan Katrina) a causé une augmentation ponctuelle significative de plus de 50% du débit de pleins bords sur tous les bassins de la forêt Montmorency, la fréquence des pointes ayant une période de retour de 2,6 ans aurait augmenté d'environ 3% (1 seul événement ayant augmenté le débit de pleins bords de plus de 50% / 30,8 fois sur une révolution de coupe de 80 ans X 100% des bassins). Par contre, ce calcul est effectué en se basant sur seulement deux années d'après coupe. La fréquence de la période de retour de 2,6 ans serait possiblement augmentée de manière plus importante en considérant que la coupe ait un effet à plus long terme sur les débits de pointe. D'ailleurs, Plamondon (2004) a estimé que, par rapport aux conditions naturelles, l'augmentation des pointes susceptibles de modifier la morphologie des cours d'eau se produit rarement plus de 3 fois durant les 10 années suivant la récolte forestière. De plus, lors d'une revue de littérature portant sur 50 études de partout dans le monde, Guillemette et al. (2005) ont constaté que seul le tiers des basins versants ayant subi la coupe sur plus de 50% de leur superficie ont expérimenté des augmentations de plus de 50% de leur débit de pleins bords. Par conséquent, similairement au cas de la forêt Montmorency, le risque d'augmenter le débit de plein bord de plus de 50% (ou d'accroître la fréquence des pointes ayant une période de retour de 2,6 ans) serait aussi accru d'environ 3% en considérant globalement un grand nombre de bassins versants (3 événements risquant d'augmenter le débit de plein bord de plus de 50% / 30,8 fois sur une révolution de coupe de 80 ans X 33% des bassins). Puisque plusieurs efforts sont entrepris afin de contrer les feux de forêts et les épidémies d'insectes, la récolte forestière pourrait en quelques sortes remplacer en partie les effets de ces perturbations naturelles sur les débits de pointe. De plus, les débits de pointe sont nécessaires pour nettoyer les frayères et en générer des nouvelles. Selon ces hypothèses, le risque de 3% d'augmenter le débits de pleins bords de plus de 50% à la forêt Montmorency ou sur un bassin versant d'un territoire donné est relativement faible.

Les augmentations les plus importantes de la température de l'eau ont eu lieu sur un bassin dont l'aire de coupe se situe loin du réseau hydrographique, sur la partie amont du bassin versant. Il a été suggéré que la récolte effectuée en amont des bassins peut avoir un effet sur la température de l'eau en aval puisque les petits cours d'eau de tête non permanents ou invisibles à la surface du sol ne sont généralement pas protégées par la lisière boisée (Bourque et Pomeroy, 2001). Les observations de terrain à la forêt Montmorency ont démontré que la végétation a effectivement été récoltée sur les zones éphémères de convergence du ruissellement et le long d'écoulements s'effectuant sous l'épaisse couche d'humus. Les faibles baisses de la température observées (de moins de 5%) sur un autre bassin versant sont par contre en désaccord avec les résultats généralement observés dans la littérature. Ce bassin est deux fois plus petit que la moyenne des bassins coupés et son écoulement devient intermittent lors d'épisodes secs au milieu de l'été. Durant ces périodes moins humides, l'eau fraîche souterraine constitue la majorité de l'écoulement des ruisseaux permanents (Alexander et Caissie, 2003), et donc de celui du bassin témoin. Les valeurs prédites à l'aide de la relation entre le bassin témoin et le bassin traité pourraient donc être biaisées et suggérer une apparente baisse de la température. D'autres variables externes à la récolte, tel que la manipulation des instruments de mesures, pourraient aussi avoir contribué à la faible baisse de température observée. L'augmentation des températures élevées et la diminution des températures faibles sur le troisième bassin suggère qu'après la coupe, le ruisseau a été réchauffé plus intensivement qu'avant la coupe durant les jours chauds et que la perte d'énergie a augmenté durant les jours frais due à la perte de la canopé. La tendance étant aussi bien remarquée pour les températures journalières maximales que minimales, le processus en cause agit donc autant le jour comme la nuit. D'autres processus comme l'évaporation, les échanges de chaleur sensible avec l'air et les échanges de chaleur par conduction avec le substrat ont pu influencer la température de l'eau (Johnson et Jones, 2000; Sinokrot and Stefan, 1993; Story et al., 2003).

L'intensification de la nitrification suite à la coupe forestière, provoquée par l'accroissement des populations de micro-organismes dû à l'augmentation de la température et de l'humidité du sol (Martin et al., 2000), est responsable de l'augmentation fulgurante atteignant plus de 6000% de la concentration initiale en nitrates à la forêt Montmorency. Par contre, cette augmentation moyenne maximale, correspondant près de 0,4 $mg{\bullet}l^{-1}$, n'est pas extrême comparativement à une hausse moyenne de 1,3 $mg{\bullet}l^{-1}$ déjà observée dans le Charlevoix (Plamondon, 1994). D'après une revue de littérature incluant 14 études, l'accroissement de la concentration en nitrates peut même atteindre 3,7 $mg{\bullet}l^{-1}$ (Brown et Binkley, 1994). Le MDDEP (2006) stipule que la concentration doit excéder une valeur de 40 $mg{\bullet}l^{-1}$ pour observer des effets chroniques sur les organismes aquatiques, tandis que 200 $mg{\bullet}l^{-1}$ doit être atteint pour la toxicité aiguë. Avec une concentration mesurée maximale de 0,82 $mg{\bullet}l^{-1}$, ces critères sont loin d'avoir été dépassés à la forêt Montmorency.

Les quelques augmentations de la concentration des phosphates observées à la forêt Montmorency ont pu être causées par un ou plusieurs des processus suivants : la minéralisation accélérée de la matière organique, la diminution du pompage des nutriments par la végétation, l'augmentation du ruissellement et l'accroissement du taux d'érosion (Degenhardt et Ice, 1996). Ces faibles augmentations de 0,007 $mg{\bullet}l^{-1}$ et moins ont principalement eu lieu sur le bassin versant où la coupe a été effectuée près du réseau hydrographique et où des incursions des aires de coupe ont été tolérées à l'intérieur de la lisière boisée. Lorsque la lisière boisée était complètement récoltée, des augmentations beaucoup plus importantes, jusqu'à 0,08 $mg{\bullet}l^{-1}$, ont été mesurées dans le Charlevoix (Plamondon, 1994). La relative stabilité en phosphates sur les autres bassins coupés est donc probablement due au fait que les aires de coupes n'ont pas empiété sur la lisière boisée, mais aussi au fait que le contenu en calcium pouvant immobiliser le phosphore augmente généralement en profondeur dans les sols de la forêt boréale (Evans et al., 2000). De faibles diminutions de 0,04 $mg{\bullet}l^{-1}$ ont été rapportées ailleurs en forêt boréale québécoise lorsque la lisière boisée était maintenue (Plamondon, 1994). Aucune concentration limite en phosphates n'existe pour la protection de la vie aquatique, mais la concentration en phosphore total ne doit pas excéder 0,03 $mg{\bullet}l^{-1}$ pour les effets chroniques (MDDEP, 2006). Puisque les orthophosphates représentent normalement environ 40% du phosphore total dans les écosystèmes forestiers (Ice, 1999), le seuil pour les effets chroniques a été dépassé à plusieurs reprises à la forêt Montmorency, autant avant comme après la coupe et autant sur le bassin témoin que sur les bassins traités.

L'intensification de la nitrification a provoqué une hausse dans la production d'acide nitrique (HNO^{3-}) (Feller, 2005; Martin et al., 2000), phénomène responsable de la baisse du pH observée sur la plupart des bassins coupés à la forêt Montmorency. La décomposition des débris de coupe laissés sur place afin de protéger le sol du compactage a aussi contribué à cette diminution (Feller, 2005). La très faible hausse (de 0,2 unité de pH et moins) a été observée sur le bassin ayant enregistré les plus faibles augmentations des nitrates en valeurs absolues (en $mg{\bullet}l^{-1}$ et non en %). Des augmentations jusqu'à 0,5 unité de pH ont aussi déjà été observées ailleurs dans la forêt boréale québécoise (Plamondon, 1994; Seto, 2005). Les critères de qualité de l'eau pour la protection de la vie aquatique du MDDEP (2006) ne peuvent pas s'appliquer puisque le pH naturel des ruisseaux à l'étude excède les bornes inférieures des seuils recommandés.

Puisque le potassium est essentiel à la croissance des plantes et est absorbé par les arbres (Plamondon, 2002), l'enlèvement de la végétation peut provoquer l'augmentation de la concentration en potassium dans les cours d'eau (Ice, 1999; Rosén et al., 1996). Aussi, l'intensification de la nitrification produisant de l'acide nitrique (HNO^{3-}) peut causer la mobilisation du cation K^+ présent dans le sol qui, en remplaçant l'ion H^+ rattaché à l'anion NO^{3-}, tend à augmenter sa concentration dans l'eau du sol (Martin et al., 2000). Des hausses jusqu'à 1,5 mg•l^{-1} peuvent être observées (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001). L'augmentation maximale de 0,63 mg•l^{-1} observée à la forêt Montmorency est donc modérée, mais elle est quand même plus importante que les augmentations variant entre 0,04 et 0,3 mg•l^{-1} mesurées ailleurs dans la forêt boréale québécoise alors que la lisière boisée était maintenue (Plamondon, 1994). Des effets chroniques sur la faune aquatique peuvent être observés lorsque la concentration en potassium excède 5 mg•l^{-1}, alors que certaines espèces de poisson et d'invertébrées meurent à des concentrations respectives de 400 mg•l^{-1} et 700 mg•l^{-1} (Plamondon, 2002). La concentration maximale de 1,4 mg•l^{-1} mesurée après la coupe à la forêt Montmorency est très faible comparativement à ces concentrations extrêmes.

La concentration en magnésium peut aussi augmenter en remplaçant l'ion H^+ de l'acide nitrique produite par l'intensification de la nitrification (Martin et al., 2000). Dans Charlevoix et sur la Côte-Nord, la hausse du magnésium a varié entre 0,01 mg•l^{-1} et 0,2 mg•l^{-1} à la suite de la coupe avec lisière boisée (Plamondon, 1994). L'augmentation maximale de 0,067 mg•l^{-1} mesurée à la forêt Montmorency est donc modérée pour la forêt boréale québécoise. L'augmentation maximale de 0,531 mg•l^{-1} est nettement inférieure aux seuils associés aux effets chroniques (14 mg•l^{-1}) et à la toxicité aiguë (100-400 mg•l^{-1}) pour la faune aquatique (Plamondon, 2002).

L'intensification de la nitrification peut aussi être reliée à l'accroissement de la concentration en fer, puisque la baisse du pH de l'eau augmente la réduction du Fe^{3+} sous la forme plus soluble Fe^{2+}, de même que la libération du fer contenu dans les hydroxides insolubles (Feller, 2005). L'augmentation maximale de 0,139 mg•l^{-1} mesurée à la forêt Montmorency se rapproche des augmentations jusqu'à 0,1 mg•l^{-1} observées ailleurs dans la forêt boréale québécoise où la lisière boisée était conservée (Plamondon, 1994). Après la coupe, le critère québécois associé aux effets chroniques de 0,3 mg•l^{-1} (MDDEP, 2006) a été dépassé pour plus de 50% des échantillons d'eau des bassins traités, alors qu'il a été dépassé pour 40% des échantillons du bassin témoin. Le sol riche en fer (podzol ferro-humique) pourrait expliquer ces concentrations importantes. Le critère américain de 1,0 $mg \cdot l^{-1}$ pour les effets chroniques (EPA, 1986) n'a quant à lui été dépassé sur aucun des bassins.

Les augmentations significatives des sulphates observées à la forêt Montmorency sont contraires aux diminutions rapportées dans la littérature (Feller, 2005; Likens et al., 2002; Welsch et al., 2004). Les faibles baisses de pH mesurées à la forêt Montmorency pourraient avoir prévenu la hausse du taux d'adsorption des sulphates par les particules de sol. Aussi, durant le deuxième été après la coupe (2005), de nombreux feux de forêts ont eu lieu au Québec, accroissant ainsi le taux de déposition atmosphérique du SO_4^{2-}. En fait, durant les 10 dernières années (de 1996 à 2005), 38% de la superficie (1 007 794,5 ha) de forêt affectée a brûlé durant l'été 2005 (ce qui représente environ 380% de l'aire moyenne annuelle affectée) (SOPFEU, 2006). Le critère de 300 $mg \cdot l^{-1}$ pour la toxicité aiguë (MDDEP, 2006) est toutefois loin d'avoir été dépassé avec la concentration maximale de 3,4 $mg \cdot l^{-1}$ mesurée après la coupe à la forêt Montmorency.

La hausse importante des concentrations en nitrates et potassium, de même que les hausses plus faibles des concentrations en magnésium, fer et sulphates, sont directement responsables de l'augmentation de la conductivité observée à la forêt Montmorency. Les augmentations variant entre 0,5 et 4,2 $\mu S \cdot cm^{-1}$ sont similaires à celles variant entre 0 et 8 $\mu S \cdot cm^{-1}$ mesurées dans plusieurs régions de la forêt boréale québécoise alors que les lisières boisées étaient maintenues le long des cours d'eau (Plamondon et al., 1982). Lorsque la végétation riparienne était récoltée, des augmentations jusqu'à 70 $\mu S \cdot cm^{-1}$ ont toutefois été mesurées (Plamondon, 1994). Les augmentations modérées de la conductivité ne dépassant pas 26% les valeurs d'avant coupe à la forêt Montmorency peuvent être associées au maintien des lisières boisées conservées le long des cours d'eau permanents. Bien que les différentes espèces de poissons tolèrent des concentrations différentes en solides dissous (EPA, 1986), une conductivité de plus de 2000 $\mu S \cdot cm^{-1}$ peut leur être nuisible (Plamondon, 2002). Cette valeur est plus de 60 fois supérieure à la conductivité maximale de 31,6 $\mu S \cdot cm^{-1}$ mesurée à la forêt Montmorency.

Sans statistiques plus robustes, les résultats concernant les sédiments en suspension ne sont pas certains. Toutefois, le faible taux de dépassement des critères de qualité de l'eau suggère que la récolte forestière n'a pas causé d'augmentations importantes pouvant nuire à la vie aquatique. Par définition, le critère pour les effets chroniques doit être excédé sur une longue période de temps afin d'affecter les organismes aquatiques (MDDEP, 2006). Les dates auxquelles ce fut le cas sur les bassins coupés ne sont consécutives que sur une courte durée. Par exemple, sur un des bassins, les huit (8) dates pour lesquelles le critère a été dépassé le plus souvent sont les suivantes : les 16, 18, et 25 juin 2004, le 23 juillet 2004, les 13 et 27 août 2004 ainsi que le 26 septembre 2005. L'intervalle d'échantillonnage ayant été de deux à trois fois par semaine, plusieurs mesures sous le seuil des effets chroniques ont été prises entre ces dates. En ce qui concerne le seuil de toxicité aiguë, il ne doit être excédé que sur une courte période de temps pour affecter les organismes aquatiques (MDDEP, 2006). À la forêt Montmorency, il n'a été dépassé qu'une seule fois et sur un seul bassin coupé. La mesure de 30,5 $mg \bullet l^{-1}$ a été prise le 12 juillet 2004, entre deux événements pluvieux alors que sur les autres bassins, la concentration en sédiments en suspension n'avait pas dépassé 1,4 $mg \bullet l^{-1}$. Un effondrement des berges du ruisseau serait donc plus plausible que la coupe forestière elle-même afin d'expliquer ce dépassement du critère. Les données disponibles ne montrent pas l'effet de la présence d'un chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement sur les sédiments en suspension. Ces résultats sont corroborés par l'absence d'augmentation significative observée lors de la construction du chemin en 2000 (Beaudin, 2002). De plus, le dépassement du critère pour la toxicité aiguë du 12 juillet 2004 a eu lieu en amont du chemin et non en aval.

Étant donné que la récolte sur 50% de la superficie des bassins n'a pas provoqué d'augmentations moyennes significatives des débits de pointe sur aucun des bassins coupés, aucune influence ne peut être attribuée à la proximité des aires de coupe par rapport au réseau hydrographique. D'ailleurs, la plus faible augmentation de 18% du débit de pleins bords a été observée sur le bassin dont la récolte a été effectuée le plus près de son cours d'eau. En ce qui concerne la qualité de l'eau, seule la température a été affectée par la localisation des aires de coupe. Toutefois, ce n'est probablement pas le fait que l'aire de coupe se situait loin du réseau hydrographique permanent qui est responsable de l'intensification de l'augmentation des températures journalières maximales et minimales sur un des bassins traités, mais bien la position en amont sur le bassin puisque les zones éphémères de convergence et les écoulements sous la couche d'humus n'ont pas été protégés par les lisières boisées.

7. Conclusion

À la suite de la récolte sur 50% de petits bassins versants à la forêt Montmorency, les augmentations moyennes des débits de pleins bords sont demeurées sous 50%, seuil réputé pouvant modifier la morphologie des cours d'eau et ainsi affecter les écosystèmes aquatiques. Le risque de dépassement de ce seuil s'est accru de seulement 3% après la récolte forestière. Les températures journalières maximales et minimales ont faiblement augmenté ou diminué significativement selon les bassins traités, alors que la variation diurne a diminué. Les concentrations en nitrates et potassium ont fortement augmenté, surtout durant le deuxième été suivant la récolte. Les concentrations en magnésium et en fer, ainsi que la conductivité, ont suivi la même tendance, bien que les augmentations significatives observées aient été beaucoup plus faibles. La concentration en sulphates a pour sa part augmenté également durant les deux étés suivant la coupe, tandis que le pH a significativement diminué mais faiblement. La concentration en phosphates est quant à elle demeurée relativement stable sur la plupart des bassins. Un manque de données d'avant coupe ne permet pas de confirmer la hausse observée des sédiments en suspension.

L'intensité modérée de la récolte forestière effectuée sur couvert de neige selon les pratiques d'aménagement favorisant la protection de la régénération et des sols, de même que la conservation de la lisière boisée de 20 m le long des cours d'eau permanents, ont permis de restreindre l'augmentation des débits de pointe. Même si l'augmentation de 49% d'un débit de pleins bords sur un des bassins versants est à toute fin pratique à la limite du seuil acceptable de 50%, la recommandation de récolter 50% et moins des bassins versants est possiblement trop limitante du point de vue de la récolte forestière compte tenu de l'augmentation statistiquement non significative des débits de pointe et la faible hausse de 3% du risque d'augmenter le débit de pleins bords de plus de 50%. De plus, l'étude ayant été effectuée sur des bassins versants de faible superficie (< 50 ha) qui réagissent rapidement aux précipitations, la coupe sur 50% de bassins versants de plus grandes dimensions causerait probablement des augmentations plus faibles des débits de pointe. La relative faiblesse des modifications des caractéristiques physico-chimiques de l'eau par rapport aux critères reconnus pour la protection de la vie aquatique suggère aussi que la coupe sur 50% de la superficie du bassin semble respecter l'intégrité des écosystèmes aquatiques. L'effet de la proximité des aires de coupe du réseau hydrographique n'a pas été démontré, autant pour l'augmentation des débits de pointe que pour les modifications des paramètres de qualité de l'eau. La présence d'un chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement n'a pas non plus contribué à l'augmentation des sédiments en suspension.

Cette étude devrait être poursuivie afin de mesurer les effets à long terme de la coupe forestière. Le risque d'augmenter le débit de pleins bords de plus de 50% pourrait possiblement augmenter si d'autres événements similaires à Katrina se reproduisent. De plus, bien que ce type d'expérimentation est extrêmement contraignant tant au niveau physique que financier, il serait utile de reproduire un plus grand nombre de bassins versants témoins et traités, tout en conservant les mêmes méthodes sylvicoles. Les études à venir devraient tenter de relier l'impact de l'augmentation des débits de pointe sur la morphologie des cours d'eau, de même que les modifications des paramètres de qualité de l'eau sur les communautés d'organismes aquatiques.

8. Liste des références

Alexander, M.D. and Caissie, D., 2003. Variability and comparison of hyporheic water temperatures and seepage fluxes in a small Atlantic salmon stream. Ground Water, 41: 72-82.

Basnyat, P., Teeter, L.D., Flynn, K.M. et Lockaby, B.G., 1999. Relationships between landscape characteristics and nonpoint source pollution inputs to coastal estuaries. Environmental Management, 23: 539-549.

Beaudin, 2002. Peak and storm flow response to forest road construction, Mémoire de maîtrise, Université Laval, Quebec, 60 pp.

Beschta, R.L., 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research, 14: 1011-1016.

Beschta, R.L., Pyles, M.R., Skaugset, A.E. and Surfleet, C.G., 2000. Peakflow response to forest practices in the western cascades of Oregon, USA. Journal of Hydrology, 233: 102-120.

Beschta, R.L. and Taylor, R.L., 1988. Stream temperature increases and land use in a forested Oregon watershed. Water Resources Bulletin, 24: 19-25.

Bosch, J.M. et Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation change on water yield and evapotranspiration. Journal of Hydrology, 55: 3-23.

Bourque, C.P.-A. and Pomeroy, J.H., 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. Hydrology and Earth System Sciences, 5: 599-613.

Brown, G.W. and Krygier, J.T., 1971. Clear-cut logging and sediment production in the Oregon Coast Range. Water Resources Research, 7: 1189-1198.

Brown, T.C. and Binkley, D., 1994. Effect of management on water quality in north american forests. USDA Forest Service General Technical Report, RM-248, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Coulombe, G. et al., 2004. Rapport, Comission d'étude sur la gestion de la forêt publique québécoise, Décembre 2004.

Croke, J.C. and Hairsine, P.B., 2006. Sediment delivery in managed forests: a review. Environmental Reviews, 14: 59-87.

Degenhardt, D. et Ice, G., 1996. Forest management options to control excess nutrients for the Tualatin River, Oregon. Proceedings of the 1995 NCASI West Coast Regional Meeting, Special Report No. 96-04: 2-17.

Dunne, T. et Leopold, L.B., 1978. Water in environmental planning. Freeman, San Fransisco, 818 pp.

EPA, 1986. Quality criteria for water. Office of Water, Regulations and Standards, U.S. Environmental Protection Agency, Washington, D.C., 477 pp.

Evans, J.E., Prepas, E.E., Devito, K.J. and Kotak, B.G., 2000. Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 60-72.

Faustini, J.M., 2000. Stream channel response to peak flows in a fifth-order mountain watershed, PhD Thesis, Oregon State University, 339 pp.

Feller, M.C., 2005. Forest harvesting and streamwater inorganic chemistry in western North America: a review. Journal of the American Water Resources Association.

Guillemette, F., Plamondon, A.P. et Lévesque, D., 1998. Effet De la coupe sur le bilan hydrologique. Bassin expérimental du ruisseau des Eaux-Volées. Rapport du Ministère des Ressources Naturelles. Université Laval, 65 pp.

Guillemette, F., Plamondon, A.P., Prévost, M. et Lévesque, D., 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. Journal of Hydrology, 302: 137-153.

Harr, R.D., Frederiksen, R.L. et Rothacher, J., 1979. Changes in streamflow following timber harvest in southwestern Oregon. Water Resources Research, 15: 90-96.

Hartman, G.F. and Scrivener, J.C., 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin of Fisceries and Aquatic Science, 223: 148 p.

Hartman, G.F., Scrivener, J.C. et Miles, M.J., 1996. Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, et their implication for restoring fish habitat. Canadian Journal of Fisheries et Aquatic Science, 53(Suppl. 1): 237-251.

Hornbeck, J.W., Adams, M.B., Corbett, E.S. et Verry, E.S., Lynch, J.A., 1993. Long-term impacts of forest treatments on water yield: a summary for northeastern USA. Journal of Hydrology, 150: 323-344.

Hornbeck, J.W., Martin, C.W. et Eagar, C., 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Canadian Journal of Forest Research, 27: 2043-2052.

Ice, G., 1999. Streamflow and Water Quality: What Does the Science Show about Clearcutting in Western Oregon, Clearcutting in Western Oregon: What Does the Science Show? National Council for Air and Stream Improvement, Oregon State University, Corvallis, Oregon, pp. 20.

Jewett, K., Daugharty, D., Krause, H.H. et Arp., P.A., 1995. Watershed response to clearcutting: effects on soil solutions and stream water discharge in central New Brunswick. Canadian Journal of Soil Sciences, 75: 475-490.

Johnson, S.L. et Jones, J.A., 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 30-39.

Jones, J.A. et Grant, G.E., 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research, 32: 959-974.

Jones, J.A., Swanson, F.J., Wemple, B.C. et Snyder, K.U., 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology, 14: 76-85.

Likens, G.E. et al., 2002. The biogeochemistry of sulfur at Hubbard Brook. Biogeochemistry, 60: 235-316.

MacDonald, L.H., Wohl, E.E. et Madsen, S.W., 1997. Validation of water yield thresholds on the Kootenai National Forest. Department of Earth Resources, Colorado State University, Fort Collins, CO., 197 pp.

Martin, C.W., Hornbeck, J.W., Likens, G.E. et Buso, D.C., 2000. Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 19-29.

Martin, C.W., Noel, D.S. et Federer, C.A., 1981. The effect of forest clearcutting in New England on streamwater chemistry and biology, USDA Forest Service Northeastern Forest Experiment Station, Technical Completion Report, Project No. A-051-NH.

MDDEP, 2006. Critères de qualité de l'eau de surface au Québec. Ministère du Développement Durable, de l'Environnement et des Parcs, http://www.mddep.gouv.qc.ca/eau/criteres_eau/index.htm, visité le 21 juin 2006.

Megahan, W.F. and Kidd, W.J., 1972. Effect of logging and logging roads on erosion and sediment deposition from steep terrain. Journal of Forestry, 70: 136-141.

Moore, R.D. et Scott, R.D., 2005. Camp Creek revisited: streamflow changes following salvage harvesting in a medium-sized, snowmelt-dominated catchment. Canadian Water Resources Journal, 30(4): 331-344.

Plamondon, A.P., 1982. Augmentation de la concentration des sédiments en suspension suite à l'exploitation forestière et durée de l'effet. Canadian Journal of Forest Research, 12: 883-892.

Plamondon, A.P., 1993. Influence des coupes forestières sur le régime d'écoulement de l'eau et sa qualité. Revue de littérature. Ministère des Forêts du Québec. Rapport C-47, 179 pp.

Plamondon, A.P., 1994. Impacts of clearcutting on water quality with an emphasis on the boreal forest, Northern Research Basins Study (NRBS) Workshop. Government of Alberta and Faculty of Agriculture and Forestry, University of Alberta, Edmonton, pp. 30.

Plamondon, A.P., 2002. Manuel de météorologie et hydrologie forestières appliquées à l'aménagement des bassins versants. Centre de recherche en biologie forestière, Faculté de foresterie et géomatique, Université Laval, St-Foy.

Plamondon, A.P., 2004. La récolte forestière et les débits de pointe: État des connaissances sur la prévision des augmentations des pointes, le concept de l'aire équivalente de coupe acceptable et les taux régressifs des effets de la coupe sur les débits de pointe. Direction de l'Environnement forestier, Ministère des Ressources naturelles, Gouvernement du Québec, 236 pp.

Plamondon, A.P., Gonzalez, A. and Thomassin, Y., 1982. Effects of logging quality: comparison between two Quebec sites, Proceedings of the Canadian Hydrologic Symposium, Fredericton, New Brunswick, pp. 47-70.

Plamondon, A.P. et Naud, R.C., 1975. The watershed management program at the Montmorency experimental forest, Usefulness of an experimental forest research and teaching, Symposium, Forêt Montmorency. Faculté de foresterie, Université Laval, Québec, pp. 129-152.

Plamondon, A.P. et Ouellet, D.C., 1980. Partial clearcutting and streamflow regime of Ruisseau des Eaux-Volées experimental watershed. Proceedings of the Helsinki Symposium. IAHS, 130: 129-136.

Reid, L.M. and Dunne, T., 1984. Sediment production from forest road surfaces. Water Resources Research, 20: 1753-1761.

Reinhart, K.G., 1967. Watershed calibration methods. In: W.E. Sopper et H.W. Lull (Editors), Forest Hydrology. Pergamon Press, Oxford, pp. 715-721.

Rieman, B.E. et McIntyre, J.D., 1993. Demographic and habitat requirements for conservation of bull trout. USDA Forest Service, Intermountain Research Station, Gen. Tech. Rep. INT-302, 38 pp.

Robitaille, A. et Saucier, J.P., 1998. Paysages régionaux du Québec méridional. Publications du Québec, St-Foy, 213 pp.

Rochette, F.J., 1971. Hydrological study of Ruisseau des Eaux-Volées experimental basin. M.Sc. thesis, University of Western Ontario, 88 pp.

Rosén, K., Aroson, J.-A. and Erikson, H.M., 1996. Effects of clear-cutting on streamwater quality in forest catchments in central Sweden. Forest Ecology and Management, 83: 237-244.

Sahin, V. et Hall, M.J., 1996. The effects of afforestation and deforestation on water yields. Journal of Hydrology, 178: 293-309.

Schindler, D.W., 1998. A dim future for boreal waters and landscapes. Bioscience, 48: 157-167.

Scrivener, J.C. et Tripp, D.B., 1998. Changes in spawning gravel characteristics after forest harvesting in Queen Charlotte Island and Carnation Creek watersheds and the apparent impacts on incubating eggs. In: D.L. Hogan, P.J. Tschaplinski et S. Chatwin (Editors), Proceedings of the Carnation Creek and Queen Charlotte Island fish/forestry Workshop: Applying 20 years of coast research to management solutions. British-Columbia. Minstry of forest research program, Land Management Handbook 41, pp. 135-140.

Seto, M., 2005. Effets de l'exploitation forestière sur la qualité de l'eau en forêt boréale. Mémoire de maîtrise. Université Laval, pp. 97.

Sinokrot, B.A. and Stefan, H.G., 1993. Stream temperature dynamics: measurements and modeling. Water Resources Research, 29: 2299-2312.

SOPFEU, 2006. Société de protection des forêts contre le feu. http://www.sopfeu.qc.ca/html/francais/tour/stats_anneeCause.php, visité le 10 octobre 2006.

Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology, 176: 79-95.

St-Hilaire, A., Morin, G., El-Jabi, N. and Caissie, D., 2000. Water temperature modelling in a small forested stream: implication of forest canopy and soil temperature. Canadian Journal of Civil Engineering, 27: 1095-1108.

Story, A., Moore, R.D. and Macdonald, J.S., 2003. Stream temperatures in two shaded reaches below cutblocks and logging roads: downstream cooling linked to subsurface hydrology. Canadian Journal of Forest Research, 33: 1383-1396.

Swank, W.T., Vose, J.M. and Elliott, K.J., 2001. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. Forest Ecology and Management, 143: 163-178.

Swift, L.W. and Messer, J.B., 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. Journal of Soil and Water Conservation, 26: 111-116.

Thomas, R.B. et Mehagan, W.F., 1998. Peaf flow responses to clear-cutting and roads in small and large basins, western Oregon: a second opinion. Water Resources Research, 34: 3393-3403.

Troendle, C.A., Wilcox, M.S., Bevenger, G.S. et Porth, L.S., 2001. The Coon Creek Water Yield Augmentation Project: implementation of timber harvesting technology to increase streamflow. Forest Ecology and Management, 143: 179-187.

Welsch, D.L., Burns, D.A. and Murdoch, P.S., 2004. Processes affecting the response of sulfate concentrations to clearcutting in a northern hardwood forest, Catskill Mountains, New York, U.S.A. Biogeochemistry, 68: 337-354.

WFPB, 1997. Washington Forest Practices Board Manual: Standard Methodology for Conducting Watershed Analysis. Washington State Department of Natural Resources, Forest Practices Division.

Wright, K.A., Sendek, K.H., Rice, R.M. et Thomas, R.B., 1990. Logging effects on streamflow: storm runoff at Caspar Creek in Northwestern California. Water Resources Research, 26: 1657-1667.

SECTION 2

Articles

Article 1

Rainfall peakflow response to clearcutting 50% of three small watersheds in a boreal forest, Montmorency Forest, Québec

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Résumé

Selon la revue de littérature, il est suggéré que la morphologie des cours d'eau peut être modifiée significativement par une augmentation de 50% du débit de pleins bords et qu'une telle augmentation peut être évitée en coupant moins de 50% de l'aire du bassin versant. Il est aussi suggéré que la récolte forestière effectuée près du réseau hydrographique affecte plus fortement les débits de pointe que celle effectuée plus loin. Suite à cinq années de calibration, trois petits bassins versants (< 50 ha) localisés dans le Bassin Expérimental du Ruisseau des Eaux-Volées ont été récoltés sur 50% de leur superficie. La récolte a été effectuée selon les pratiques d'aménagement assurant la protection de la régénération et sur couvert de neige afin de prévenir les perturbations du sol. Un bassin contigu a été conservé à son état naturel afin de servir comme témoin. Durant les deux étés suivant la coupe, les régressions ont générées des augmentations non significatives des débits de pointe et des augmentations du débit de pleins bords demeurant sous le seuil de 50% pour tous les bassins. La proximité des parterres de coupe par rapport au réseau hydrographique n'a pas joué de rôle significatif dans l'accroissement des débits de pleins bords puisque l'augmentation la plus faible a été observée sur le bassin où la coupe a été effectuée le plus près de son ruisseau permanent. Un seul événement pluvieux particulièrement intense, correspondant à la queue de l'ouragan Katrina (31 août, 2005), a causé des augmentations significatives des débits de pointe au-dessus de la limite de confiance supérieure. Ces augmentations individuelles du débit de pleins bords étaient tous supérieures à 50%, indiquant ainsi que le seuil de récolte de 50% de la superficie des bassins versants n'est possiblement pas applicable sur les bassins de petite taille. Toutefois, l'événement en cause ayant une occurrence très rare et considérant une révolution de coupe de 80 ans, le risque d'augmentation de 50% des débits de pleins bords a été augmenté de seulement 3% à la forêt Montmorency.

Mots-clé

Débit de pointe; Coupe forestière; Forêt boréale; Aménagement du bassin versant; Débit de pleins bords.

Abstract

Based on the literature, it was previously suggested that stream morphology would be significantly modified by a 50 % increase of the bankfull discharge and that such increase could be avoided by clearcutting less than 50 % of the watershed area. It was also hypothesized that clearcutting near the stream network would have more effect on peak flow than harvesting a comparable area farther away from the stream network. Following a 5-year calibration period, three small watersheds (< 50 ha) within the "Ruisseau des Eaux-Volées" Experimental Watershed at Montmorency Forest were harvested over 50% of their areas. The regeneration was protected and logging was carried out during the snow cover season in order to prevent soil perturbations. A neighbouring control watershed was maintained undisturbed. During the two summers following logging, the regression lines yielded non-significant peakflow increases and bankfull discharge augmentations below the 50% threshold for all treated watersheds. The proximity of the cutover areas to the stream network did not play a significant role in increasing bankfull discharge since the smallest augmentation was recorded on the watershed where the cutover area was the closest to the stream network. Only one particularly intense rainstorm (Hurricane Katrina trail, August 31st 2005) significantly increased peakflows above the upper confidence limit of the pre-treatment regression. These individual bankfull increases were all above 50% indicating that the 50% cutting threshold may not be applicable on very small watersheds. However, the causing event has a very rare occurrence and considering a harvesting rotation every 80 years, the risk for peakflow to augment above 50% was increased by only 3% at Montmorency Forest.

Keywords

Peakflow; Clearcutting; Boreal Forest; Watershed management; Bankfull discharge

Introduction

Forest hydrology has been investigated extensively at many experimental sites in North America and across the World (Bosch and Hewlett, 1982; Guillemette et al., 2005; Hornbeck et al., 1993; Plamondon, 2004; Sahin and Hall, 1996; Stednick, 1996). In general, removal of vegetation diminishes interception, reduces transpiration and increases soil water content, while harvesting operations locally decrease infiltration rates due to soil compaction and may transmit more rapidly to the stream network water conveyed in road ditches and ruts (Harr et al., 1979; Ice, 1999; Jones et al., 2000; Wright et al., 1990). Increases in peakflows have frequently been attributed to these processes (Beschta et al., 2000; Caissie et al., 2002; Guillemette et al., 2005; Hornbeck et al., 1997; Jones and Grant, 1996; Thomas and Mehagan, 1998; Wright et al., 1990). The geomorphological and ecological impact of greater peakflows or greater frequency of large flows remain unclear to this day (Grant et al., 1999), but evidences show that it may have adverse effects on fish habitat, downstream sedimentation, and riparian conditions (MacDonald et al., 1997).

Augmentations in peakflows are highly variable among sites and are globally correlated with the proportion of the watershed area harvested (Beschta et al., 2000; Plamondon, 2004). The work of MacDonald et al. (1997) indicated that a minimum of 10% to 15% forest clearance is necessary for a peakflow change to be detected, irrespective of its generation by rainfall, snowmelt or rain on snow, while measurement techniques and statistical procedures failed to detect smaller changes. In general, small peakflows undergo greater increases than the large ones (Beschta et al., 2000; Jones and Grant, 1996; MacDonald et al., 1997; Thomas and Mehagan, 1998), although exceptions have been documented (Clary et al., 1974; Verry et al., 1983). Recent studies suggested that some increases may occur on peakflow large enough to affect stream morphology (Beschta et al., 2000; Guillemette et al., 2005; MacDonald et al., 1997; Plamondon, 2004). The most effective flow stage, frequent enough to transport the largest total amount of sediments over time, is the bankfull discharge which occurs when a stream channel is filled to the top of its banks (Wolman and Miller, 1960). It is generally accepted that this flow stage, characterized by a return period of 1.5 years ($Q1.5$), controls stream morphology (Dunne and Leopold, 1978). Recent analysis of 47 rivers in Ontario indicated a mean $Q1.6$, ranging from $Q1.5$ to $Q1.7$, for bankfull discharge (Annable, 1995).

Based on observations of 57 transects in four streams in Oregon, Faustini (2000) documented that stream bed modification occurred in 50% of the transects following a 50% increase in bankfull discharge. The author argues that this 50% increase, corresponding to an augmentation of a $Q1.5$ flow to a $Q3.2$ for two of the investigated streams and to a $Q4.4$ for the two others, can significantly modify portions of the stream bed. Other authors have argued that an increase of a $Q2$ flow to a $Q5$ can significantly modify stream morphology (Ice, 1999) and affect aquatic ecosystems (WFPB, 1997). For example, stream bed modification following forest harvesting can disturb salmon spawning sites and reduce the number of incubating eggs (Scrivener and Tripp, 1998; Tripp and Poulin, 1986). Also, large bedload sediment transport, scouring and aggradation of stream bed can be detrimental to bull trouts (Rieman and McIntyre, 1993). Therefore, the risk for temporary reduction of habitat quality during stream adaptation following logging must be taken into account in forest management, especially in watersheds where sensible organisms or species of great value are found.

Based on a 50 world-wide study review, Guillemette et al. (2005) indicated that a bankfull discharge augmentation of more than 50% was observed on 1/3 of the watersheds and at Montmorency forest, Quebec. All of the reviewed watersheds showing this result had their area harvested on more than 50%. Hence, to prevent a 50% increase of bankfull discharge in sensible watersheds, Plamondon (2004) recommended limiting the equivalent clearcut area to 50% of the watershed. This recommendation is compulsory for salmonid river watersheds greater than 100 km^2 in the Province of Quebec. Since saturated zones, primarily located along streams and convergence areas, respond quickly during rain events (Dunne and Leopold, 1978), harvesting in these areas may have a greater influence in peakflow increase than cutting further away from the streams. Plamondon (2004) also suggested distributing the cutovers wherever possible on the watersheds. Therefore, this study assessed the following hypotheses: (i) harvesting 50

% of watershed area maintains bankfull peakflow increases below 50 %; (ii) locating the 50 % cutover area harvested closer to the stream network rather than further away produces larger bankfull peak flow increases.

Site and methods

The studied watersheds are located within the 'Ruisseau des Eaux-Volées' Experimental Watershed (REVEW), in the Montmorency Forest, 80 km north of Quebec City (47°16'20''N, 71°09'40''W), Canada (Plamondon and Ouellet, 1980) (Fig. 1). The topography is characterised by a plateau with a mean altitude of 750 m and hills reaching 1000 m. The region lies within the Canadian Shield, more precisely in the Grenville Geological Province, and is composed of Precambrian charnockitic gneiss with vertically tilted fractures extending to a depth of 45 m (Rochette, 1971). The unconsolidated material is constituted of basal till more than 1-m thick on 80% of the area, extending from 0 m on summits to 18 m in valleys (Rochette, 1971). Ferro-Humic Podzol represents the most dominant soil type and is covered on average by an 8-cm humus layer (Plamondon and Naud, 1975). The first 30 cm of soil is highly permeable while the underlying till has a low permeability (Barry, 1984). According to the Köppen Classification System, the climate type is continental (*Dfc*) with a short growing season characterized by cool temperatures and elevated humidity. Mean annual and mean monthly temperatures for January and July are 0.3, -15 and 14.7°C, respectively, while mean annual precipitation, snow (in water equivalent), runoff and evapotranspiration are 1544, 465, 1085 and 459 mm respectively (Guillemette et al., 1998). Half of the precipitation occurs during the humid season (June to October) with mean monthly July and August precipitations greater than 150 mm (Plamondon et al., 1998). The bioclimatic domain is the balsam fir white birch ecosystem (Robitaille and Saucier, 1998). The stand is composed of more than 75% balsam fir (*Abies balsamea* (L.) Mill), 10% white birch (*Betula papyrifera* Marsh), 10% white spruce (*Picea glauca* (Moench.) Voss) and 3% black spruce (*Picea mariana* Mill) (Bélanger et al., 1991). Balsam fir stands generally yield from 75 to 200 $m^3 \cdot ha^{-1}$ of merchantable volume at 60 years of age (Guillemette et al., 2005).

The paired watershed approach with a calibration period prior to logging (Plamondon, 1993; Reinhart, 1967) was used to detect the impact of clearcutting 50% of watershed area on peakflows. The treated watersheds (7.2, 7.3, 7.5 and 7.7) nested in the REVEW and the neighbouring control watershed (0.2) (Table 1 and Fig. 1) were gauged from 1999 to 2005. The road constructed during summer 2000 covered 1.5 to 2.5% of the watersheds area and the measurement made above and below the road by Beaudin (2002) indicated no significant impact on peakflows. Harvesting operations were done during the winter season of 2004. The trees were delimbed and cut in logs with a tracked multifunctional harvester while the logs were carried out with a wheeled forwarder. The presence of one-meter thick snowpack and trimmed branches enabled a satisfactory soil protection where only rare and much localised rutting and soil compaction occurred. The regeneration was protected by locating the skid trails every 16 m and by sliding the trees in a straight line with the boom. Watershed 7.2 was logged near its hydrographic network in the lower part of the watershed, while watersheds 7.5 and 7.7 where logged in their upper parts. Forest buffers 20-m wide were left along the permanent streams, except for watershed 7.2 where logging of the riparian trees was partly allowed while the harvester remained at 15 m from the stream banks. Watershed 7.3, combining watersheds 7.2 and 7.5, was also gauged to evaluate the effect of forest harvesting partly near and partly far from the streams.

To measure stream flow, 90° V-notch weirs were built with aluminium plates and plywood and their upstream retention basins were lined with thick impermeable pool tarps, fibro-concrete plates and plywood. Water levels within the retention basin were measured every 15 min using a float and pulley system fixed on a potentiometer. Water level can normally be directly transformed to discharge with the Gourley and Grimp equation (Lencastre, 1996):

$$Q = 1.38H^{2.47} \tag{1}$$

where Q is discharge in $m^3 \cdot s^{-1}$ and H is water level in m over the bottom of the V. However, calibration measurements indicated that the following experimental relation was more representative of our design:

$$Q = 1.44H^{2.47} \tag{2}$$

Precipitations were also measured hourly with a tipping bucket rain gauge installed in the lower part of watershed 7.2 (Fig. 1) and at the Montmorency Forest meteorological station of Environment Canada located 4 km northeast of the experimental site.

In a former study, Guillemette et al. (2005) included in their analysis peakflows corresponding to storm events generating daily stream flows above 0.5 $m^3 \cdot s^{-1}$ at the outlet of the REVEW (917 ha). The same criterion was transformed in specific discharge (55 $l \cdot s^{-1} \cdot km^{-2}$) and applied to each small watersheds of the present study. Hence, a greater proportion of storm events than Guillemette et al. (2005) permitted enough observations for statistical analysis. Only rainfall generated peakflows without snow on the ground (normally from June 1st to October 31st) were considered since flows with a return period greater than two years are mostly generated by rainfall for watersheds smaller than 500 km^2 in southern Quebec (Rousselle et al., 1990). Linear regressions were used to assess the relationship of treated watershed peakflows with respect to control watershed peakflows for both pre- and post-treatment periods (Hewlett and Pienaar, 1973). Peakflows were log transformed (in natural logarithms) to meet the assumption of homoscedasticity and to improve the frequency distribution of the data along the interval of the regression (Beschta et al., 2000; Thomas and Mehagan, 1998; Wright et al., 1990). Pre- and post-treatment regression coefficients (slope and intercept) were compared using the Z-test (Kleinbaum and Kupper, 1978), calculated with the GLM procedure from SAS/STAT software (SAS Institute, 1989).

Bankfull discharges, with a return period of 1.5 years ($Q1.5$), were estimated for the studied watersheds by comparison with bankfull discharge for contiguous watershed 7A (122 ha) of the REVEW (Fig. 1). The 27-year data set (1967-1993) prior to logging watershed 7A was used to calculate the return period of the annual peakflow and to establish the corresponding bankfull discharge ($Q1.5 \pm 0.3$ years) at 0.66 m^3s^{-1} ($\pm$ 0.16 $m^3 \cdot s^{-1}$) or 541 $l \cdot s^{-1} \cdot km^{-2}$ (Table 2). This stream flow was effectively exceeded 16 times in 27 years on watershed 7A. ©HYFRAN, a hydrological frequency analysis software developed at INRS-ETE (Chair in Statistical Hydrology, 2002), was used to perform the calculations. The Log-Pearson type III model, the most appropriate model for certain

Québec regions (Hoang, 1978), provided the best fit (Fig. 2). A regression model was applied between peakflows of the pre-treatment period (1999-2003) for each studied watershed and watershed 7A (e.g., Fig. 3). Respective bankfull discharges for the control and treated watersheds were estimated using the regression lines based on bankfull discharge for watershed 7A (Table 3). The same method was used to estimate the related return period for peakflows other than bankfull discharge. Since the weirs measured only the flow within the stream, the smallest watershed does not exhibit the largest specific discharge.

Results

Peakflows of the treated and control watersheds were well correlated as shown with the R^2 of the regressions exceeding 0.85 (Table 4, Fig. 4). The slope of the regression lines did not differ significantly after treatment, indicating that increases in peakflows would be similar for flows of different magnitudes. The intercepts between pre- and post-treatment regression lines were not found to be statistically different indicating that there was no average peakflow increase. However, most of the post-treatment peakflows and the intercepts were found to be above the pre-treatment regression lines, showing a tendency for an increase caused by the timber harvest. Mean increases at bankfull discharges of 18%, 28%, 49% and 32% for treated watershed 7.2, 7.3 7.5 and 7.7 respectively (Table 5), were calculated as the ratio between pre- and post-treatment regression lines. Related $Q1.5$ increased to $Q1.9$, $Q2.2$, $Q2.6$ and $Q2.2$ for treated watershed 7.2, 7.3 7.5 and 7.7 respectively. A 50% increase in bankfull discharge would have corresponded to a $Q2.6$.

The upper 95% confidence limit of the individual, pre-treatment, peakflows (Fig. 4) indicated that increases larger than 52%, 62%, 82% and 55% at bankfull discharge were significant for treated watersheds 7.2, 7.3, 7.5 and 7.7 respectively (Table 6). Only one rainfall event, that generated peakflow augmentations greater or equal than the 50% bankfull discharge increase, was found above the 95% upper confidence limit for all treated watersheds. The Hurricane Katrina trail, that occurred on August 31st 2005, produced, for treated watersheds 7.2, 7.3, 7.5 and 7.7, respectively, peakflows 59%, 75%, 139% and 92% greater than those in the absence of treatment as predicted by the pre-treatment regression lines (Table 7). For the treated watersheds 7.2, 7.3, 7.5 and 7.7, the associated pre-treatment *Q*1.4, *Q*1.4, *Q*1.6 and *Q*1.6 increased to *Q*2.6, *Q*3.6, *Q*11.3 and *Q*5.1, respectively. These peakflow increases from watersheds 7.2, 7.5 and 7.7 were inversely proportional to their areas. Watershed 7.3 which combines watersheds 7.2 and 7.5 shows peak flow increases between the 7.2 and 7.5 values. Another important recorded peakflow was greater than the upper confidence limit on watershed 7.2 only and corresponded to an increase of 58%, representing a change of a *Q*1.6 (larger than bankfull) to a *Q*3.1.

Discussion

After treatment, the regression lines indicated that peakflow increases at bankfull level remained below the 50% threshold for all treated watersheds. The augmentation was 49% on the smallest watershed 7.5 while it remained below 32 % on the others. The small size of watershed 7.5 (11.5 ha) and the small distance between the watershed limits and the stream in its lower part may explain the stronger response of the watershed to rainfall events as compared to the others. When watershed 7.5 is combined with watershed 7.2 to form watershed 7.3, the mean increase of 36% of bankfull discharge lies between the 7.2 and 7.5 response.

In the absence of treatment, the tail of Hurricane Katrina (August 31st 2005) would have generated peakflows in the same order of magnitude of bankfull discharges (between *Q*1.3 and *Q*1.6 on the different watersheds). For all treated watersheds, the rainstorm produced significant bankfull peakflows increases greater than 50%. The 139 and 92% increases of the two smallest watersheds are much above the 82 and 55% increases to be significant. The very intense Katrina rainfall event, 7.33 $mm \cdot h^{-1}$ for 15 hours with a maximum of 15 $mm \cdot h^{-1}$ for 5 hours, may explain the peakflow response to this event. The results indicate that the proposed 50% clearcut to limit bankfull peakflow increase under 50 % may not be applicable on very small watersheds. However, this type of event has a very small occurrence. During the post-treatment period, four (4) larger peakflows than Katrina were recorded on the control watershed but the associated increases on the treated watersheds were not significant (below the upper confidence limit). The results with the exception of Katrina support the tendency observed elsewhere showing that large peakflows tend to produce smaller stream flow changes than smaller ones (Beschta et al., 2000; Jones and Grant, 1996; MacDonald et al., 1997; Thomas and Mehagan, 1998).

At Montmorency Forest, a 50% bankfull discharge augmentation corresponds to the rise of a *Q*1.5 flow to a *Q*2.6. In natural conditions, a *Q*2.6 would occur on average 30.8 times in a typical harvest rotation period of 80 years. Since only one recorded event (Hurricane Katrina trail) significantly increased bankfull discharge of more than 50% on all watersheds at Montmorency Forest, the frequency for the *Q*2.6 to occur was increased by about 3% (1 recorded event exceeding the 50% bankfull discharge augmentation / 30.8 times in a 80 years rotation period X 100% of the watersheds). This calculation is however only based on two post-treatment years but the probability to have another Katrina type of storm within the first 10 years after harvesting (regeneration < than 3 m high) is very small. The frequency of the *Q*2.6 to occur would be augmented more importantly considering that the forest harvest may have a longer impact. From a literature review Plamondon (2004) estimated that the 50 % peakflow increases are usually observed less than three times during the 10 years following logging. Furthermore out of the 50 world-wide watershed studies with cutover areas exceeding 50% reviewed by Guillemette et al. (2005), only one third (1/3) did experience bankfull discharge augmentations of more than 50%. Consequently, considering a large number of watersheds during a long period the risk for augmenting bankfull discharge of more than 50% (or the frequency increase of the occurrence of the *Q*2.6) would also have increased by about 3% (3 recorded events exceeding the 50% bankfull discharge augmentation / 30.8 times in a 80 years rotation period X 33% of the watersheds).

Since many efforts are deployed to counteract forest fires and insect epidemics, forest harvesting may to some extent replace these natural disturbances that can affect watershed hydrology. Moreover, some peakflows are necessary for scouring spawning sites and generate new ones. Based on these considerations, the 3% risk increase observed at Montmorency Forest is acceptable. Furthermore, this small risk seems to decrease with watershed size and since the increases in peakflows from the regression line were not statistically significant, the 50% clearcut threshold recommended by Plamondon (2004) should be maintained. Perhaps, it may be too stringent for slightly larger watersheds of the boreal forest were regeneration and soils are well protected and a forest buffer is maintained along perennial streams.

Without statisticaly significant increases in peakflows, no effect could be attributed to the different cutover locations within the watersheds. Moreover, the harvest near the stream network of watershed 7.2 produced the smallest, but non-significant effect (18% bankfull discharge increase). The recommendation of Plamondon (2004) distributing the cutovers far from the stream network is not validated and, perhaps, it may be too conservative for small watersheds of the boreal forest if regeneration and soils are protected and a forest buffer is maintained along perennial streams. It also indicates that patch cutting can be distributed over the watershed including the buffer zone when needed for forest or wildlife management.

Conclusion

Mean bankfull discharge augmentations remained below 50% on all watersheds of Montmorency Forest harvested on 50% of their area. Flows exceeding this threshold are believed to have the potential to significantly modify stream morphology and affect aquatic ecosystems. The risk for the 50% bankfull discharge augmentation did increase by about 3% for the two summers following logging and was attributed to a rare event. Since the risk of large peakflow augmentation is small, especially considering that logging replaces some effects of natural disturbances, a 50% forest clearance could be allowed on small watersheds of the boreal forest. Moreover, since the proximity of the cutovers to the stream networks did not influence the increase in peakflows, tree stands removal does not have to be distributed at different distances from the streams as long as a 20 m wide forest buffer is maintained along perennial streams. Patch cutting of the riparian trees can even be aloud if needed for forest and wildlife management. However, good management practices such as the protection of advance regeneration and soils must be respected.

This study was performed over the two years following forest harvesting and focused on the effect on peakflows on a short term period only. There is a need to pursue this project to evaluate the modifications over many more years, although the impact should gradually be damped with time as new vegetation growth takes place. It would also be useful to repeat the number of control and treated watersheds undergoing the same type of treatment in order to obtain more robust results. For future watershed studies it would be interesting to repeat the 50 % clearcutting to assess our findings in other types of topography, climate, forest, and more importantly watershed size. In fact, one study including two treated watersheds is being carried out north of Baie-Comeau, in the North-Coast region of Quebec. Finally, extended studies on the morphological impact of peakflow changes are necessary, as well as their effects on aquatic ecosystems.

The 50% cutover-area criterion may be adequate in boreal forests of Quebec according to bankfull discharge, but it might not be the same for water quality. The proximity to the drainage network of the clearcut may have greater effects on physical and chemical characteristics of water. These hypotheses must be addressed in order to validate the 50% threshold. A joint study on water quality for the same watersheds nested in the REVEW is actually in process. Results will be addressed in a future publication.

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References

Annable, W.K., 1995. Morphological relationships of rural watercourses in southwestern Ontario for use in natural channel designs, University of Guelph.

Barry, R., 1984. Prédiction de la teneur en eau du sol minéral dans la sapinière laurentienne de la forêt Montmorency. Mémoire de maîtrise, Université Laval, 100 p.

Beaudin, 2002. Peak and storm flow response to forest road construction, Mémoire de maîtrise, Université Laval, Quebec, 60 pp.

Bélanger, L., Bertrand, L., Bouliane, P. and Lussier, L.J., 1991. Plan général d'aménagement de la forêt Montmorency. Université Laval, 216 pp.

Beschta, R.L., Pyles, M.R., Skaugset, A.E. and Surfleet, C.G., 2000. Peakflow response to forest practices in the western cascades of Oregon, USA. Journal of Hydrology, 233: 102-120.

Bosch, J.M. and Hewlett, J.D., 1982. A review of catchment experiments to determine the effect of vegetation change on water yield and evapotranspiration. Journal of Hydrology, 55: 3-23.

Caissie, D., Jolicoeur, S., Bouchard, M. and Poncet, E., 2002. Comparison of streamflow between pre and post timber harvesting in Catamaran Brook (Canada). Journal of Hydrology, 258: 232-248.

Chair in Statistical Hydrology, 2002. HYFRAN: hydrological frenquency analysis, version 1.1. INRS-ETE, Université du Québec, Québec.

Clary, W.P., Baker, B., O'Connell, P.F., Johnsen, T.N. and Campbell, R.E., 1974. Effects of Pinyon-Juniper removal on natural resources products and uses in Arizona. USDA Forest Service Research Paper RM-128, 28 pp.

Dunne, T. and Leopold, L.B., 1978. Water in environmental planning. Freeman, San Fransisco, 818 pp.

Faustini, J.M., 2000. Stream channel response to peak flows in a fifth-order mountain watershed, PhD Thesis, Oregon State University, 339 pp.

Grant, G.E., Megahan, W.F. and Thomas, R.B., 1999. A re-evaluation of peak flows: Do forest roads and harvesting cause floods? 1999 NCASI West Coast Regional

Meeting, National Council of the Paper Industry for Air and Stream Improvement, Portland, OR.

Guillemette, F., Plamondon, A.P. and Lévesque, D., 1998. Effet De la coupe sur le bilan hydrologique. Bassin expérimental du ruisseau des Eaux-Volées. Rapport du Ministère des Ressources Naturelles. Université Laval, 65 p.

Guillemette, F., Plamondon, A.P., Prévost, M. and Lévesque, D., 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. Journal of Hydrology, 302: 137-153.

Harr, R.D., Frederiksen, R.L. and Rothacher, J., 1979. Changes in streamflow following timber harvest in southwestern Oregon. Water Resources Research, 15: 90-96.

Hewlett, J.D. and Pienaar, L., 1973. Design and analyses of the catchment experiment. In: E.D. White (Editor), Proceedings Symposium on the use of small watersheds in determining the effets of forest land use on water quality, pp. 88-106.

Hoang, V.D., 1978. Meilleur ajustement statistique aux débits journaliers de crue et d'étiage des rivières du Québec. Ministère des richesses naturelles, Direction générale des eaux, Service de l'hydrométrie, Québec, 28 pp.

Hornbeck, J.W., Adams, M.B., Corbett, E.S. and Verry, E.S., Lynch, J.A., 1993. Long-term impacts of forest treatments on water yield: a summary for northeastern USA. Journal of Hydrology, 150: 323-344.

Hornbeck, J.W., Martin, C.W. and Eagar, C., 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Canadian Journal of Forest Research, 27: 2043-2052.

Ice, G., 1999. Streamflow and Water Quality: What Does the Science Show about Clearcutting in Western Oregon, National Council for Air and Stream Improvement, Oregon State University, Corvallis, Oregon, pp. 20.

Jones, J.A. and Grant, G.E., 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research, 32: 959-974.

Jones, J.A., Swanson, F.J., Wemple, B.C. and Snyder, K.U., 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology, 14: 76-85.

Kleinbaum, D.G. and Kupper, L.L., 1978. Applied regression analysis and other multivariate methods. Wadsworth Publishing Company Inc., Belmont, 486 pp.

Lencastre, A., 1996. Hydraulique générale. Éditions Eyrolles, Paris, 633 pp.

MacDonald, L.H., Wohl, E.E. and Madsen, S.W., 1997. Validation of water yield thresholds on the Kootenai National Forest. Department of Earth Resources, Colorado State University, Fort Collins, CO., 197 pp.

Plamondon, A.P., 1993. Influence des coupes forestières sur le régime d'écoulement de l'eau et sa qualité. Revue de littérature. Ministère des Forêts du Québec. Rapport C-47, 179 pp.

Plamondon, A.P., 2004. La récolte forestière et les débits de pointe: État des connaissances sur la prévision des augmentations des pointes, le concept de l'aire équivalente de coupe acceptable et les taux régressifs des effets de la coupe sur les débits de pointe. Direction de l'Environnement forestier, Ministère des Ressources naturelles, Gouvernement du Québec, 236 pp.

Plamondon, A.P., Lévesque, D., Ma, Y. and Prévost, M., 1998. Long-term effects of forest mosaic management on storm and snowmelt flow, Quebec. Proceedings of the British Hydrological Society International Conference, Exeter, July 1998: Hydrology in a changing environment, 1: 503-515.

Plamondon, A.P. and Naud, R.C., 1975. The watershed management program at the Montmorency experimental forest, Usefulness of an experimental forest research and teaching, Symposium, Forêt Montmorency. Faculté de foresterie, Université Laval, Québec, pp. 129-152.

Plamondon, A.P. and Ouellet, D.C., 1980. Partial clearcutting and streamflow regime of Ruisseau des Eaux-Volées experimental watershed. Proceedings of the Helsinki Symposium. IAHS, 130: 129-136.

Reinhart, K.G., 1967. Watershed calibration methods. In: W.E. Sopper and H.W. Lull (Editors), Forest Hydrology. Pergamon Press, Oxford, pp. 715-721.

Rieman, B.E. and McIntyre, J.D., 1993. Demographic and habitat requirements for conservation of bull trout. USDA Forest Service, Intermountain Research Station, Gen. Tech. Rep. INT-302, 38 pp.

Robitaille, A. and Saucier, J.P., 1998. Paysages régionaux du Québec méridional. Publications du Québec, St-Foy, 213 pp.

Rochette, F.J., 1971. Hydrological study of Ruisseau des Eaux-Volées experimental basin. M.Sc. thesis, University of Western Ontario, 88 pp.

Rousselle, J., Watt, W.E., Lathem, K.W., Neill, C.R. and Richards, T.L., 1990. Hydrologie des crues au Canada - Guide de planification et de conception. Conseil National de Recherches Canada, Ottawa, 227 pp.

Sahin, V. and Hall, M.J., 1996. The effects of afforestation and deforestation on water yields. Journal of Hydrology, 178: 293-309.

SAS Institute, 1989. SAS/STAT User's Guide, Vol. 1, ACECLUS-FREQ. SAS Institute, Cary, NC, 943 pp.

Scrivener, J.C. and Tripp, D.B., 1998. Changes in spawning gravel characteristics after forest harvesting in Queen Charlotte Island and Carnation Creek watersheds and the apparent impacts on incubating eggs. In: D.L. Hogan, P.J. Tschaplinski and S. Chatwin (Editors), Proceedings of the Carnation Creek and Queen Charlotte Island fish/forestry Workshop: Applying 20 years of coast research to management solutions. British-Columbia. Minstry of forest research program, Land Management Handbook 41, pp. 135-140.

Stednick, J.D., 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology, 176: 79-95.

Thomas, R.B. and Mehagan, W.F., 1998. Peaf flow responses to clear-cutting and roads in small and large basins, western Oregon: a second opinion. Water Resources Research, 34: 3393-3403.

Tripp, D.B. and Poulin, V.A., 1986. The effects of logging and mass wasting on salmonid spawning habitat in streams on the Queen Charlotte Islands, Land Management Report 50. British-Columbia Ministry of Forests and Lands, Land Management Report 50, 19 pp.

Verry, E.S., Lewis, J.R. and Brooks, K.N., 1983. Aspen clearcutting increases snowmelt and storm flow peaks in north central Minnesota. Water Resources Bulletin, 19(59-67).

WFPB, 1997. Washington Forest Practices Board Manual: Standard Methodology for Conducting Watershed Analysis. Washington State Department of Natural Resources, Forest Practices Division.

Wolman, M.G. and Miller, J.P., 1960. Magnitude and frequency of forces in geomorphic processes. Journal of Geology, 68: 54-74.

Wright, K.A., Sendek, K.H., Rice, R.M. and Thomas, R.B., 1990. Logging effects on streamflow: storm runoff at Caspar Creek in Northwestern California. Water Resources Research, 26: 1657-1667.

Table 1

Watersheds characteristics and treatments

	Watersheds				
Characteristics	0.2*	7.2	7.3	7.5	7.7
Watershed area (ha)	27.5	36	50	11.5	24
Average watershed slope (%)	12.6	13	11.7	9.7	11.2
Stream width at watershed outlet (m)	1.05	0.95	1	0.5	0.9
Average stream slope (%)	15	11	10.3	9.3	7.3
Treatments					
Proportion cut (%) within 50 m from the stream	0	70	50	20	40
Proportion cut (%) within 100 m from the stream	0	60	50	35	45
Proportion cut (%) within 150 m from the stream	0	55	50	40	55
Proportion cut (%) over the whole watershed	0	50	50	50	50

*Control watershed

Table 2

Return period (*T*) of maximum annual peakflow for watershed 7A from 1967 to 1993 and associated discharge calculated with the log-Pearson III model in ©HYFRAN

T	Discharge ($m^3 \cdot s^{-1}$)	Standard deviation	95% Confidence limits	
50	2.97	0.82	1.36	4.57
20	2.28	0.48	1.35	3.22
10	1.82	0.30	1.23	2.41
5	1.39	0.19	1.02	1.76
3	1.09	0.14	0.83	1.36
2	0.84	0.10	0.65	1.04
1.50	0.66*	0.08	0.50	0.82
1.43	0.63	0.08	0.48	0.77
1.25	0.53	0.07	0.40	0.65
1.11	0.41	0.06	0.30	0.53
1.05	0.34	0.06	0.23	0.45
1.02	0.27	0.06	0.16	0.39
1.01	0.24	0.06	0.11	0.36

*Estimated bankfull discharge

Table 3

Regression coefficients for pre-treatment peakflows between watershed 7A and control and treated watersheds and approximative bankfull discharge (*Qbf*)

Watershed	*n*	Slope	Intercept	R^2	*Qbf* ($l \cdot s^{-1} \cdot km^{-2}$)
0.2*	42	0.99	0.13	0.79	577
7.2	42	0.97	0.26	0.89	581
7.3	32	0.95	0.26	0.90	512
7.5	23	0.94	0.05	0.60	388
7.7	42	0.95	0.42	0.84	601

*Control watershed

Table 4

Regression coefficients and *p*-values according to the *Z*-test for peakflows between control and treated watersheds from the pre-treatment (1999 to 2003) and post-treatment periods (2004 and 2005)

Paired watersheds	*n*	Slope	*p*-value	Intercept	*p*-value	R^2
0.2 vs. 7.2						
Pre-treatment	40	0.88		0.64		0.91
Post-treatment	30	0.87	0.826	0.89	0.363	0.97
0.2 vs. 7.3						
Pre-treatment	30	0.85		0.69		0.88
Post-treatment	29	0.89	0.631	0.74	0.881	0.97
0.2 vs. 7.5						
Pre-treatment	21	0.95		-0.09		0.85
Post-treatment	29	1.00	0.638	0.05	0.726	0.96
0.2 vs. 7.7						
Pre-treatment	40	0.91		0.59		0.91
Post-treatment	29	0.98	0.254	0.38	0.529	0.94

Table 5

Mean bankfull discharge (*Qbf*) increase and related return period (*T*) change for treated watersheds

Watershed	Pre-treatment		Post-treatment		Increase (%)
	Qbf ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	*Qbf* ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	
7.2	581	1.5	684	1.9	18
7.3	512	1.5	654	2.2	28
7.5	388	1.5	577	2.6	49
7.7	601	1.5	795	2.2	32

Table 6

Corresponding increase from the pre-treatment regression line (*RL*) of the upper 95% confidence limit (*UCL*) at bankfull discharge and related return period (*T*) change for treated watersheds

Watershed	Pre-treatment		Post-treatment		Increase (%)
	RL ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	*UCL* ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	
7.2	581	1.5	880	2.6	52
7.3	512	1.5	837	3.1	62
7.5	388	1.5	706	3.6	82
7.7	601	1.5	934	3.1	55

Table 7

Increase of individual post-treatment peakflows (*Qpk*) greater than the upper 95% confidence limit and greater than bankfull discharge and related return period (*T*) change for treated watersheds

Watershed	Pre-treatment		Post-treatment		Increase (%)
	Qpk ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	*Qpk* ($l \cdot s^{-1} \cdot km^{-2}$)	*T*	
7.2	619	1.6	976	3.1	58
7.2	556	1.4	883*	2.6	59
7.3	501	1.4	875*	3.6	75
7.5	437	1.6	1045*	11.3	139
7.7	629	1.6	1209*	5.1	92

*Same peakflow event for all watersheds (Hurricane Katrina trail, August 31st 2005)

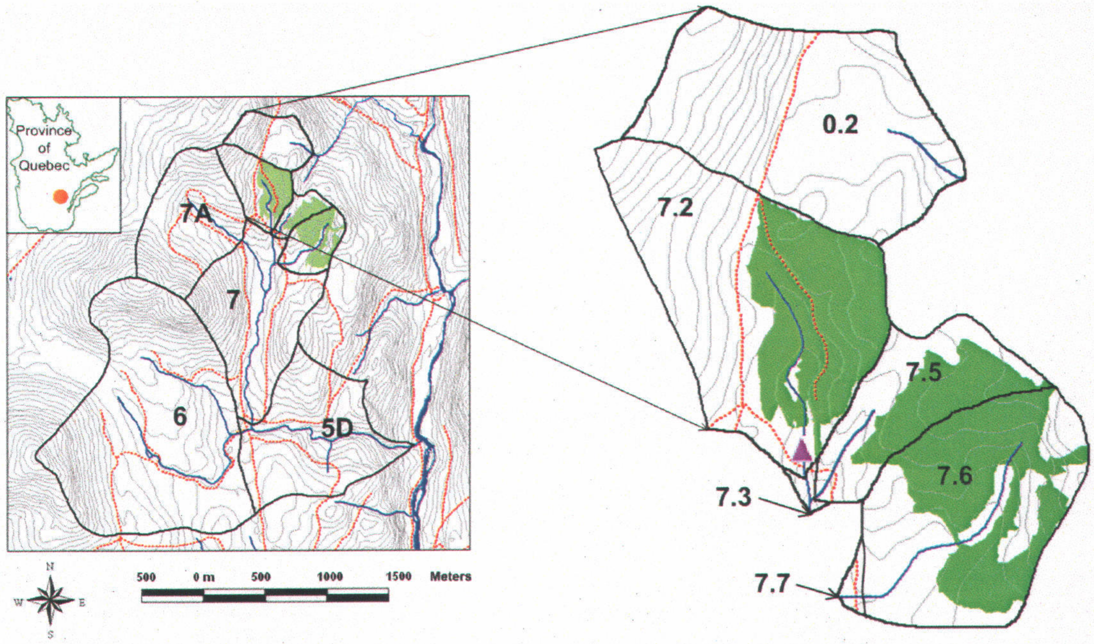

Fig. 1. Control (0.2) and treated watersheds (7.2, 7.3, 7.5 and 7.7) within the Ruisseau des Eaux-Volées Experimental Watershed (REVEW), Quebec, with clearcut areas ( ), streams (——), forest roads (········), 10-m elevation contours (——) and rain gauge location (▲). Watershed 7A is representative of long-term, local, hydrological behaviour of the REVEW.

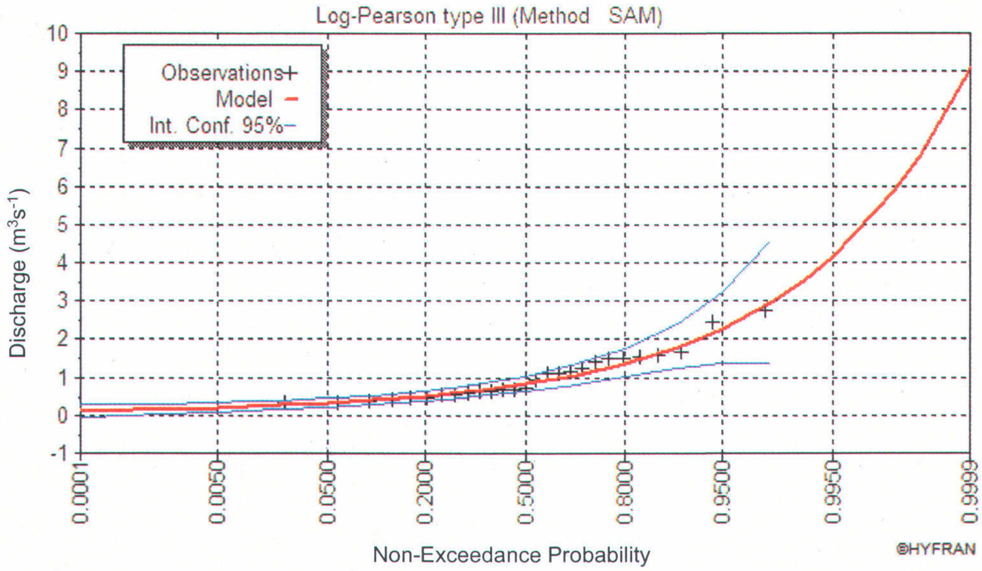

Fig. 2. Non-exceedance probability for maximum annual discharge of watershed 7A as calculated by ©HYFRAN Software with the Log Pearson type III model on Normal/Cunnane paper (return period in years = 1 / (1 - non-exceedance probability)).

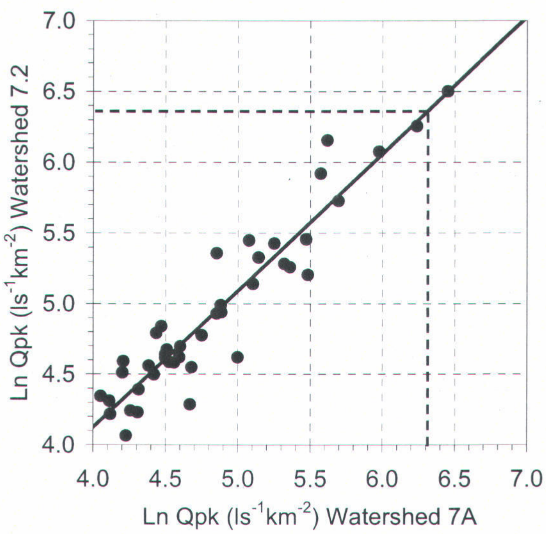

Fig. 3. Regression line for pre-treatment peakflows (Qpk) exceeding 55 $ls^{-1}km^{-2}$ on watersheds 7A and 7.2 (●; ———) and bankfull discharge for each watersheds (-------). Regression equation : $y = 0.97x + 0.26$; $R^2 = 0.89$; $n = 42$.

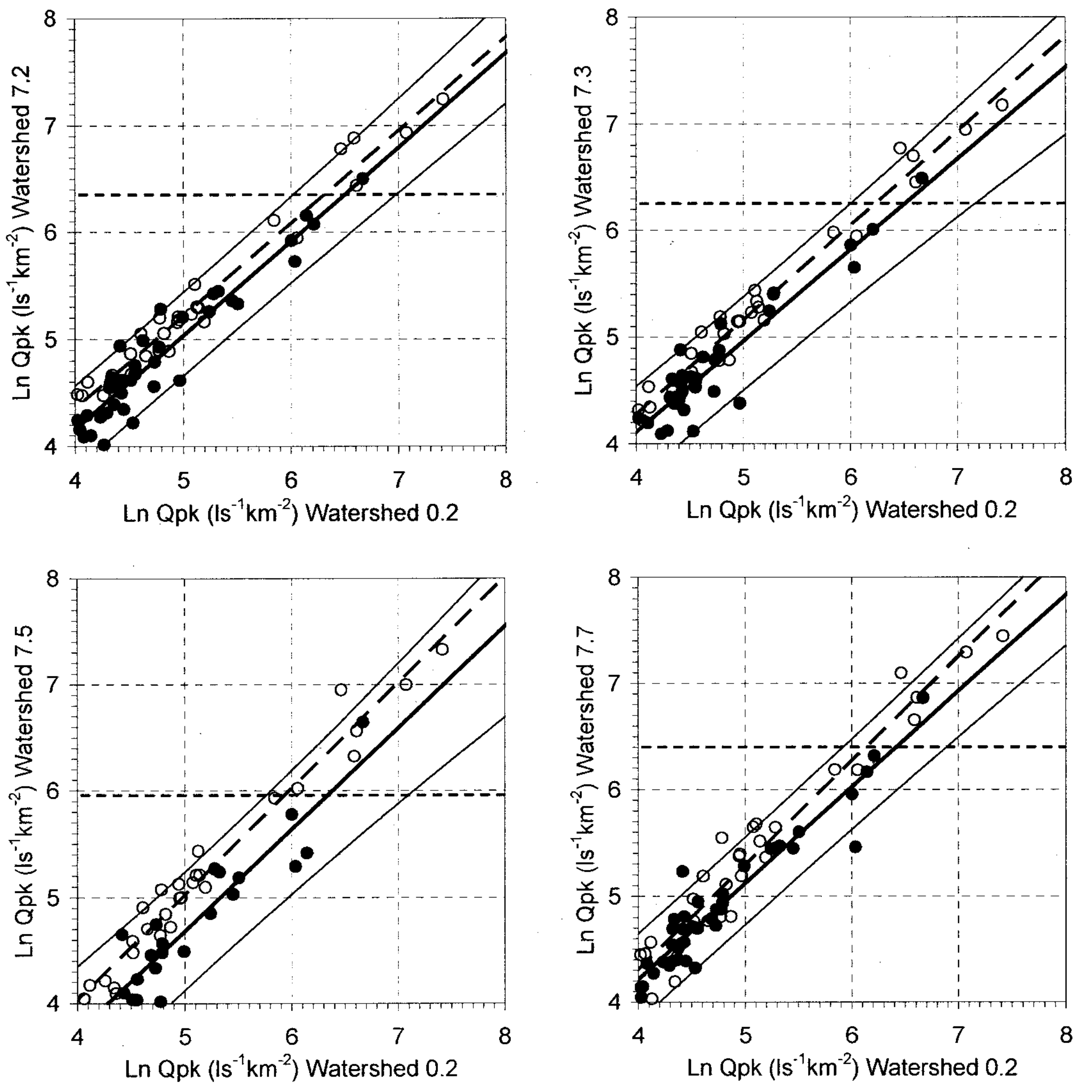

Fig. 4. Regression lines for peakflows (Qpk) exceeding 55 $ls^{-1}km^{-2}$ on the control and treated watersheds for the pre-treatment (●; ———) and post-treatment (○; — — —) periods with the 95% confidence limits (———) and pre-treatment bankfull discharge (-------) for the treated watersheds. Watershed 7.2 was treated near its hydrological network; watershed 7.3 was treated near and far its hydrological network; watershed 7.5 and 7.7 were treated far from their hydrological network.

Article 2

Stream water quality response following clearcutting 50% of watershed area at Montmorency Forest, in the boreal zone of Quebec

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Résumé

Des paramètres de qualité de l'eau ont été mesurés durant plusieurs saisons estivales avant et après la récolte forestière sur 50% de l'aire de trois petits bassins versants (< 50 ha) contenus dans le Bassin Expérimental du Ruisseau des Eaux-Volées, à la forêt Montmorency, Québec. Les bassins ont été coupés selon les pratiques d'aménagement permettant la protection de la régénération et sur couvert de neige afin de prévenir les perturbations du sol. Une lisière boisée de 20 m de large a été maintenue le long des ruisseaux permanents. Sur un des bassins (7.2), la récolte a été effectuée près du réseau hydrographique et la coupe a été en partie tolérée à l'intérieur de la lisière boisée. La récolte a été effectuée loin du réseau hydrographique pour les deux autres bassins (7.5 et 7.7). Des observations ont aussi été effectuées sur un autre bassin en aval (7.3) collectant l'écoulement des bassins 7.2 et 7.5 et sur la portion amont d'un chemin forestier (7.6) du bassin 7.7. Un bassin contigu (0.2) a été conservé à son état naturel afin de servir comme témoin. La plupart des modifications des caractéristiques physiques et chimiques de l'eau étudiées se sont avérées faibles en ce qui a trait aux critères de qualité de l'eau reconnus pour la protection de la vie aquatique. De faibles augmentations sur certains bassins et de faibles diminutions sur d'autres ont été observées pour les températures journalières maximales et minimales alors que les variations diurnes ont diminué partout. Des hausses importantes ont eu lieu pour les concentrations en nitrates et en potassium alors que des augmentations moins importantes ont été mesurées pour la conductivité et les concentrations en magnésium et en fer. Le pH a quant à lui diminué tandis que la concentration en phosphates est demeurée relativement stable sur la plupart des bassins. Contrairement aux résultats rapportés dans la littérature, la concentration en sulphates a augmenté sur certains bassins, entre autres à cause de nombreux feux de forêt ayant eu lieu sur le territoire québécois durant le second été suivant le traitement. La concentration des sédiments en suspension a probablement augmenté, mais un manque de données d'avant coupe ne permet pas de confirmer ce résultat. La proximité des parcelles de coupe par rapport au réseau hydrographique, de même que les intrusions tolérées de la coupe dans la lisière boisées n'ont pas affecté les changements des paramètres qualité de l'eau, excepté pour les phosphates et les sédiments en suspension. La présence d'un

chemin forestier et ponceau stabilisé avec une toile géotextile et empierrement n'a pas contribué à la hausse des sédiments en suspension.

Mots-clé

Qualité de l'eau; Chimie de l'eau; Coupe forestière; Forêt boréale; Aménagement du bassin versant

Abstract

Water quality parameters were measured during the summer seasons before and after clearcutting 50% of the boreal forest of three small watersheds (< 50 ha) nested in the 'Ruisseau des Eaux-Volées' Experimental Watershed, at Montmorency Forest, Quebec. Harvesting was carried out applying the guidelines to protect the advance growth and in the presence of a snow cover to prevent soil perturbations. Forest buffers 20-m wide were maintained along perennial streams. For one of the treated watersheds (7.2), the harvest was made near the hydrographic network and logging was partially allowed within the riparian zone. For the two other watersheds (7.5 and 7.7), the harvest was made farther from the hydrographic network. Observations were also made for another downstream watershed (7.3) collecting the flow from watersheds 7.2 and 7.5 and for the uproad portion (7.6) of watershed 7.7. A contiguous control watershed (0.2) was maintained undisturbed. Most of the modifications in the water quality parameters were small and met water quality standards for the protection of aquatic life. Small augmentations on some basins and small diminutions on others were noticed for the summer daily maximum and minimum stream temperatures while diurnal variations mostly declined for all. Large increases in NO_3^- and K^+ concentrations occurred, especially during the second summer after logging, while smaller increases were observed for specific conductance, Mg^{2+} and Fe_{total} concentrations. The pH mainly declined while PO_4^{3-} concentration remained relatively stable on the majority of the watersheds. For the two summers after clearcutting, contrary to the results reported in the literature, SO_4^{2-} concentration significantly increased for some watersheds, partly because of widespread forest fires that occurred in Quebec during the second post-treatment summer. Suspended sediment concentration seemed to have increased, but there were not enough pre-logging data to statistically confirm this result. The proximity of the cutover to the stream network and logging within the riparian zone did not produce different responses in water quality, except maybe for temperature, PO_4^{3-} and suspended sediment concentration. The forest road and a culvert with geotextile did not increase the suspended sediment concentrations.

Keywords

Water quality; Water chemistry; Clearcutting; Boreal Forest; Watershed Management

Introduction

Canopy removal, by exposing the forest floor and stream surface to direct solar radiation, can augment soil and water temperatures (Beschta and Taylor, 1988; Brown and Krygier, 1970; Hartman and Scrivener, 1990; Plamondon, 1982; St-Hilaire et al., 2000; Swift and Messer, 1971). Diminution of rain interception and plant transpiration can increase soil moisture and water yield (Harr et al., 1979; Ice, 1999; Jones et al., 2000; Wright et al., 1990). Along with soil moisture increase, the logging operations may also locally reduce soil infiltration, modify flow pathways and increase storm and peak flows (Beschta et al., 2000; Caissie et al., 2002; Guillemette et al., 2005; Hornbeck et al., 1997; Jones and Grant, 1996; Thomas and Mehagan, 1998; Wright et al., 1990). Combination of soil heat and moisture increases can result in rises of micro-organism abundance that in turn enhance the rate of nitrification and nitric acid (HNO^{3-}) production, thus increasing soil and water acidity (Feller, 2005; Martin et al., 2000). Soil processes such as ion exchange capacities, adsorption reactions and oxydo-reduction reactions are strongly affected by changes in pH and thus contribute to modify the chemical composition of water in contact with soil particles (Feller, 2005). Aquatic processes occurring within the stream, such as many ion exchanges, reduction reactions and microbial transformations, are also highly dependant on pH (Feller, 2005). Combination of pH drop and soil temperature increase can enhance weathering rates and thus the liberation of soluble cations (Feller, 2005). Water temperature increase alone can affect the growth, development, movement, distribution and metabolism of aquatic organisms (Brown and Krygier, 1970; Feller, 1981; Johnson, 2004), and influence the rate of nutrient uptake within stream water. Stream discharge increase can augment the dilution potential of certain chemical substances (Caissie et al., 1996; Feller, 2005), as well as it can enhance large suspended sediment concentrations that are often correlated with important flow events (Brown and Krygier, 1971).

Forest harvesting is thus responsible for a variety of changes in stream water quality. The impact of logging is often considered to be weak for the forested ecosystem and regeneration. For example, nutrient export rate increase of less than 17% of the pre-treatment levels were measured in New Brunswick (Jewett et al., 1995). Many studies have demonstrated that careful logging may not modify water quality parameters beyond the drinking standards (Plamondon, 1994), although aquatic organisms such as macro-invertebrates (Haggerty et al., 2004; Martin et al., 2000) and fishes (Hartman et al., 1996; Latterell et al., 2003) may be negatively or positively affected. It has also been suggested that landuse change near streams has more effects on physico-chemical characteristics of stream water than management over the rest of the watershed (Basnyat et al., 1999; Plamondon, 1993).

To verify whether the modifications of water quality parameters could exceed the criteria for the protection of aquatic life (EPA, 1986; MDDEP, 2006), this study focused on monitoring water physico-chemical changes following careful logging of 50% of watershed area in the boreal forest of Quebec. Moreover, it addressed whether cutover located near the stream network affects more water quality than cutover located farther away, as well as whether the presence of an already well constructed forest road and culvert had an influence on suspended sediment.

The next section reviews the importance, for the aquatic ecosystem, of each water quality parameter measured in this study. The general tendencies of the impact of forest harvesting have been summarized in Table 1. It is followed with the Method section which provides a description of the study site, timber harvest treatment, field sampling, laboratory measurements, and analytical methods. The results are then detailed for each parameter and followed by a discussion based on the criterion usually recognised for the protection of aquatic life. The conclusion addresses the hypothesis at the root of this study.

Background

Temperature

Stream temperature is a determinant factor in aquatic ecosystems controlling many processes such as the dissolution of oxygen and other gases; growth, development, movement and distribution of aquatic organisms; and rates of metabolism and organic matter decomposition (Brown and Krygier, 1970; Feller, 1981; Ice, 1999; Johnson, 2004). Modification of stream water temperature can alter existing aquatic community such as changing the dominance of various phytoplankton groups or reducing the number and distribution of benthic organisms (EPA, 1986). Moreover, important permanent temperature increases can be detrimental to fishes and cause a reduction in activity and reproduction, and even mortality for fry and adults (EPA, 1986). Toxicity also generally increases with increased temperature while organisms subjected to stress from toxic materials are less tolerant to temperature extremes (EPA, 1986). Modifications in the daily fluctuation of temperature can affect existing community structure and geographic distribution of species that in turn, can influence the reproductive cycles of aquatic organisms and other life factors (EPA, 1986). Inversely, temperature increase can also be beneficial to aquatic organisms by stimulating primary production in streams where water temperature always remain below 15°C (Plamondon, 1994). Enhancement in fish growth, through rapid embryonic development and emergence of salmonid fry has already been observed in British Columbia (Hartman and Scrivener, 1990).

It is widely recognized that forest harvesting, through the removal of riparian vegetation, can increase stream summer temperatures due to additional direct solar radiation reaching the water surface (Beschta and Taylor, 1988; Brown and Krygier, 1971; Plamondon et al., 1982; Swift and Messer, 1971). Maintenance of a forest buffer along water courses was originally thought to effectively prevent stream temperature increases (Brown and Krygier, 1970; Feller, 1981; Swift and Messer, 1971), but more recent studies with total or partial retention of the riparian vegetation have observed small but statistically significant augmentations (Bourque and Pomeroy, 2001; Macdonald et al., 2003; Seto, 2005). Soil water flow warmed by the heating of mineral soil by direct solar radiation is probably the main cause for these rises (Hartman and Scrivener, 1990; Ice, 1999; St-Hilaire et al., 2000). Temporal patterns of stream temperature can also be modified following logging, with a shift of annual maximum temperature to earlier in the summer season and earlier during the day (Johnson and Jones, 2000).

Nutrients concentration (NO_3^- and PO_4^{3-})

Inorganic NO_3^- is the most soluble and mobile form of N and, thus, the most important form of N found in stream water of forested ecosystems (Vitousek et al., 1979). Organic N are also found in great amounts, but they are not in solution since they are bounded to sediments and litter (Ice, 1999). Many studies measured important increases in stream water NO_3^- concentrations following clearcutting (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001). Nitrification can be enhanced by canopy removal since light, heat and moisture increases may promote a rise in soil micro-organism abundance (Martin et al., 2000). Removing the vegetation also temporarily diminishes the uptake of N, thus increasing its concentration in stream water (Ice, 1999; Rosén et al., 1996). NO_3^- increase in aquatic system stimulates primary production, which can be considered beneficial to some forested ecosystems (Plamondon, 2002). Important increases in NO_3^- concentration can, however, be detrimental to aquatic life (EPA, 1986; MDDEP, 2006), but such concentrations are rarely attained in natural ecosystems.

Weathering, wet deposition and dry deposition correspond to the major sources of P to forested watersheds (Ice, 1999). Plants uptake P and transform it into organic P, which is in turn transformed into soluble orthophosphates ($H_2PO_4^-$, HPO_4^{2-} and PO_4^{3-}) through the decomposition of organic matter by bacteria (Degenhardt and Ice, 1996). Orthophosphates can directly influence biological activities within aquatic ecosystems due to uptake by algae and retransformation into organic P (Degenhardt and Ice, 1996). P increase in water can stimulate excessive or nuisance growth of algae and other aquatic plants which in turn influence the chemistry of the water and can lead to eutrophication (EPA, 1986). Little to no response in PO_4^{3-} concentration are usually noticed following forest harvesting (Hartman et al., 1996; Seto, 2005; Swank et al., 2001). However, small increases are sometimes observed due to accelerated mineralization of organic matter, reduced plant uptake, increased runoff caused by the decrease in evapotranspiration, and increased erosion rates (Degenhardt and Ice, 1996). As for N, removal of the vegetation reduces the uptake of P, thus promoting an increased concentration in stream water (Ice, 1999; Rosén et al., 1996).

pH

The H+ ion has the greatest influence on other chemical substances since many chemicals increase in concentration as soil pH decrease: H+ can displace many important cations in soils, such as Al^{3+}, K^{+}, Na^{+}, Mg^{2+} Ca^{2+}, Fe^{2+} and other trace metals, which in turn bound with anions such as NO_3^-, SO_4^{2-} and to some extend, Cl^-, and enrich the soil solution (Feller, 2005). Increase in water pH also augments the solubility of many ions, such as PO_4^{3-} and trace metals compounds (EPA, 1986; Feller, 2005), thus influencing the availability of nutrients and other chemical complexes to aquatic organisms. Moreover, aquatic organisms, including fishes, are sensitive to pH variation and the toxicity of several common pollutants is markedly affected by pH changes (EPA, 1986). Nitrification producing nitric acid (HNO^{3-}) is responsible for the pH decrease often observed following clearcutting (Feller, 2005; Martin et al., 2000). The release of organic acids from the decomposition of woody debris can also play the same role (Feller, 2005).

Major cations concentration (K^+, Mg^{2+} and Fe_{total})

Important concentrations in K^+, Mg^{2+} and Fe_{total} can be detrimental to aquatic organisms (EPA, 1986; Plamondon, 2002). Increases in K^+ and Mg^{2+} are often observed (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001) and rises in Fe_{total} have already been measured in the boreal forest following clearcutting (Seto, 2005). Forest harvesting affects different processes that are responsible for these augmentations. Since K^+ is essential for plant growth and is uptaken as a nutrient (Plamondon, 2002), the removal of the vegetation can cause an increased K^+ concentration (Ice, 1999; Rosén et al., 1996). Increased acidity and temperature of water flowing through the rock fissures accelerate weathering and thus the release of soluble cations such as K^+ and Mg^{2+} (Feller, 2005). Nitrification enhancement which produces nitric acid (HNO_3) can cause the mobilization of Mg^{2+} and K^+ cations by replacing H^+ ion bounded with NO_3^- and increase their amount released to the soil solution (Martin et al., 2000). Nitrification enhancement is also responsible for diminution in stream water pH, that in turn promotes the release of Fe^{3+} ions forming insoluble hydroxides, as well as the reduction of Fe^{3+} into the more soluble form Fe^{2+} (Feller, 2005). Increases in acidity can also result in rises of Mg^{2+} and K^+ and some other trace metals concentrations (Hall et al., 1980).

Sulphate concentration (SO_4^{2-})

In forested ecosystems, atmospheric deposition is believed to be the primary source of the highly soluble anion SO_4^{2-} (Feller, 2005; Likens et al., 2002), which is generally the dominant anion deposited from the atmosphere (Welsch et al., 2004). SO_4^{2-} deposition contributes to acidification of surface water that can result in a degradation of the aquatic ecosystem (Likens et al., 2002; Welsch et al., 2004). Weathering can also be a major source of S, especially in watersheds underlaid with S-rich rocks such as gypsum (Likens et al., 2002). SO_4^{2-} is essential for all forms of life and is uptaken by the vegetation, until it is released back to the soil solution through litterfall (Likens et al., 2002). Most studies show diminutions of SO_4^{2-} in stream water following forest harvesting (Feller, 2005; Likens et al., 2002; Welsch et al., 2004). Two main processes are believed to be responsible for this decrease: first, dry deposition may diminish because of the removal of the vegetation that directly intercepts SO_4^{2-} as a dry deposit; and second, soil pH decrease due to intensified nitrification may lead to an increased SO_4^{2-} soil adsorption capacity (Feller, 2005; Welsch et al., 2004). Dilution of SO_4^{2-} in stream water caused by increased runoff and changes in decomposition rates resulting from greater microbial retention or reduced S mineralization rates may also play non-negligible roles in SO_4^{2-} concentration diminution (Feller, 2005; Welsch et al., 2004).

Specific conductance

Specific conductance constitute a good indicator of water quality since a direct relationship exists between conductivity and dissolved solids concentration (EPA, 1986; Plamondon, 1994). Different fish species are tolerant to different ranges of dissolved solids concentration (EPA, 1986). Since the concentration of principal inorganic anions (CO_3^{2-}, Cl^-, SO_4^{2-} and NO_3^-) and cations (Na^+, K^+, Ca^{2+} and Mg^{2+}) dissolved in water (EPA, 1986) are mostly increased from forest harvesting, the specific conductance should augment.

Suspended sediment concentration

Contaminants are believed to be generally associated with fine-grained sediments (Yuzyk et al., 1992). Sediments can affect water quality in delivering nutrients and sorbed chemicals to downstream aquatic environments (Novotny and Chesters, 1989). Increased sedimentation can have a detrimental impact on stream habitat by clogging fish spawning beds, disturbing invertebrate populations and affecting primary productivity (Beschta, 1978; Ice, 1999). Moreover, it can restrain light penetration through the water column, thus decreasing primary production and food for fish (EPA, 1986). Logging and road construction typically accelerate soil erosion by exposing areas of bare ground to raindrop splash, surface runoff and wind (Croke and Hairsine, 2006), resulting generally in elevated sediment concentrations in streams (Beschta, 1978; Brown and Krygier, 1971). The importance of these processes is often related to the intensity of disturbances and rain events, producing a great variability in the magnitude of response (Croke and Hairsine, 2006). Road building is often considered to be the major source of sediments (Brown and Krygier, 1971; Megahan and Kidd, 1972; Reid and Dunne, 1984), but the effect can be highly variable since only a few road segments largely participate in sediment production (Luce and Black, 1999). Stream flow and sediment load increases due to forest harvesting operations can in turn increase the input of sediment to streams by altering stream banks (Lisle, 1989). The supply of sediments to the stream after logging usually declines rapidly with the re-vegetation (Plamondon 1982), but the effect on fish spawning beds may persist for several years since the fine material deposited may be retained within the streambed gravel (Brown and Krygier, 1971).

Methods

Location

The study took place in the 'Ruisseau des Eaux-Volées' Experimental Watershed (REVEW), located in the Montmorency Forest, 80 km north of Quebec City (47°16'20''N, 71°09'40''W), Canada (Plamondon and Ouellet, 1980) (Fig. 1). The topography is characterised by a plateau with a mean altitude of 750 m and hills reaching 1000 m. The region lies within the Canadian Shield, more precisely in the Grenville Geological Province, and is composed of Precambrian charnockitic gneiss with vertically tilted fractures extending to a depth of 45 m (Rochette, 1971). The unconsolidated material is constituted of basal till more than 1-m thick on 80% of the area, extending from 0 m on summits to 18 m in valleys (Rochette, 1971). Ferro-Humic Podzol represents the most dominant soil type and is covered on average by a 8-cm humus layer (Plamondon and Naud, 1975). The first 30 cm of soil is highly permeable while the underlying till has a low permeability (Barry, 1984). According to the Köppen Classification System, the climate type is continental (*Dfc*) with a short growing season characterized by cool temperatures and elevated humidity. Mean annual and mean monthly temperatures for January and July are 0.3, -15 and 14.7°C, respectively, while mean annual precipitation, snow (in water equivalent), runoff and evapotranspiration are 1544, 465, 1085 and 459 mm, respectively (Guillemette et al., 1998). Half of the precipitation occurs during the humid season (June to October) with mean monthly July and August precipitations greater than 150 mm (Plamondon et al., 1998). The bioclimatic domain is the balsam fir white birch ecosystem (Robitaille and Saucier, 1998). The stand is composed of more than 75% balsam fir (*Abies balsamea* (L.) Mill), 10% white birch (*Betula papyrifera* Marsh), 10% white spruce (*Picea glauca* (Moench.) Voss) and 3% black spruce (*Picea mariana* Mill) (Bélanger et al., 1991). Balsam fir stands generally yield from 75 to 200 $m^3 \cdot ha^{-1}$ of merchantable volume at 60 years old (Guillemette et al., 2005).

Timber harvest treatment

Watersheds 7.2, 7.3, 7.5, 7.6 and 7.7, which are nested within the REVEW, were clearcut over 50% of their area during the 2004 winter season (Table 2, Fig. 1). A road was constructed during summer 2000 and covered 1.5 to 2.5% of watersheds area (Beaudin, 2002). The trees were delimbed onsite and cut in logs with a tracked multifunctional harvester while the logs were carried out with a wheeled forwarder. The presence of one-meter thick snowpack and trimmed branches enabled a satisfactory soil protection where only rare and much localised rutting and soil compaction occurred. The regeneration was protected by locating the skid trails every 16 m and by sliding the trees in a straight line with the boom. Watershed 7.2 was logged near its hydrographic network, in the downstream parts of the watershed and partly within the 20-m wide forest buffer along the perennial stream. The harvests on watersheds 7.5 and 7.7 remained farther away from their hydrographic networks, in the upstream parts of the watersheds, without logging within the riparian zones. Watershed 7.3, including sub-watersheds 7.2 and 7.5, was assessed in order to evaluate the combined effect of forest harvesting partly near and partly far from the streams. Watershed 7.6, the uproad portion of watershed 7.7, was used along with watershed 7.3 to differentiate the influence of the road on water quality, especially on suspended sediments. Watershed 0.2, contiguous to the treated watersheds, was maintained undisturbed and constituted the control watershed.

Field sampling and laboratory analysis

90° V-notch weirs located at the outlet of each watershed were built in 1999 in order to measure stream flow for the combined studies on peakflow response following road construction (Beaudin, 2002) and clearcutting (Tremblay et al., 2007). Hourly water temperatures (±0.01°C) were extracted from HOBO Water Temp Pro thermometers located under rocks on the streambed a few meters upstream of the weirs on watersheds 0.2, 7.2, 7.5 and 7.6. Stream temperature was measured during summer months without snow on the ground (June to October), in 2003 before treatment, and in 2004 and 2005 after treatment. At the same sites, water samples were collected manually with 500-ml, high-density, polyethylene (HDPE) bottles during summer 2000 for watersheds 0.2, 7.3 and 7.6, and summers 2002, 2004 and 2005 for all six watersheds (0.2, 7.2, 7.3, 7.5, 7.6 and 7.7). During the pre-treatment period (2000 and 2002), samples were collected at two-week intervals, resulting, on average, in ten (10) samples for each watershed and each season. During the post-treatment period, sampling was much more intensive, two to three times a week, resulting in about 50 samples for each watershed and each season. Water samples were analysed in laboratory for pH, specific conductance, suspended sediment concentrations, NO_3^-, PO_4^{3-}, K^+, Mg^{2+}, Fe_{total} and SO_4^{2-}. pH was determined at a ± 0.2-unit precision with a Hanna Instruments electrode (model HI-1332-B) and a Cole-Parmer amplifier (model C-2707-00). Specific conductance was measured at a ±2-μScm^{-1} precision with a Cole-Parmer conductivity meter (model C-01481-62) with temperature compensation to 25°C. Suspended sediment concentrations were obtained by filtering the 500-ml water samples with 0.45-µm Millipore (model HAW P04700) filters and by weighting their content at a ± 0.1-mg precision after 48 hours of oven drying at 65°C. Thereafter, NO_3^- and PO_4^{3-} were measured by flow injection analysis using QuickChem Automated Ion Analyzer (method 12-107-04-1-A and 12-115-01-1-A, Lachat

Instruments) with detection limits of 0.02 and 0.005 mg•l^{-1}, respectively. K^+, Mg^{2+}, Fe_{total} and SO_4^{2-} concentrations were analysed with a Perkin-Elmer Ion Coupled Plasma Emission Spectrophotometer (model P40) with detection limits of 0.005, 0.002, 0.0008 and 0.005 mg•l^{-1}, respectively.

Analytical methods

Stream water temperatures were analysed both graphically and statistically. The warmest discrete days and the warmest seven (7)-day periods (largest mean of the daily maximums for 7 consecutive days) on the treated watersheds for each summer were first compared to the control watershed. Mean hourly stream temperatures of the treated watersheds were also graphically compared to the control, as well as annual means of daily maximum, minimum and diurnal variation between the pre- and post-treatment periods. Since sufficient pre-treatment data were available to perform a linear regression between the control and treated watersheds, post-treatment temperatures on the control watershed were used to predict the temperatures that should be observed on the treated watersheds if no treatment had occurred. Actual, measured, post-treatment temperatures were then compared to these predicted temperatures with the paired *t*-test or the Wilcoxon signed-rank test (levels of significance of $p<0.05$ and $p<0.10$), depending on the validation of the assumption of homoscedasticity according to the Fisher-Snedecor test.

The chemical parameters were also analysed both graphically and statistically. The means for the control and treated watersheds were first illustrated graphically for each summer in order to detect obvious changes. Then, non-exceedance probability curves were computed for each studied watershed and for the pre- and post-treatment periods. Derived from these curves, the 1st, 2nd, and 3rd quartiles (25%, 50% and 75%) for each watershed and period were calculated and compared to detect tendencies in the chemical modifications of stream water. Finally, to eliminate the effect of climate and other external variables, measurements on the treated watersheds were subtracted from those of the corresponding date on the control watershed. The differences from the pre-treatment summers were grouped into one sample for each treated watershed in order to have sufficient data for statistical analysis. They were then compared separately to the differences of each post-treatment summer with the *t*-test or the Wilcoxon-Mann-Whitney test (levels of significance of $p<0.05$ and $p<0.10$), depending on the validation of the assumption of homoscedasticity according to the Fisher-Snedecor test.

Suspended sediment concentrations were more problematic to analyse due to their greater temporal and spatial variability. From an analysis of world streams, it was found that variability in suspended sediment concentrations generally increase with decreasing watershed area or stream discharge (Finlayson and McMahon, 1988). Moreover, only a few important storms may play a large role on annual sediment yields (Brown and Krygier, 1971). Where great variability may be found, longer calibration periods are required to characterize changes following landuse disturbance and longer sediment records are needed to reduce the larger error on annual means (Olive and Rieger, 1992). Based on several Canadian rivers, it was reported that mean sediments concentrations were sufficiently precise for engineering and environmental applications after at least 10 years of record (Day, 1988). In small headwater watersheds in the UK, at least 500 samples, and maybe 1000, were found to be required for the evaluation of precise enough suspended sediment concentration means (Johnson, 1992). Finally, more than one control watersheds would be required for more accurate comparison since a great variability is sometimes observed in the response of neighbouring watersheds to the same storm (Brown and Krygier, 1971).

Because insufficient sampling for suspended sediments was made before logging (a few measures were done during the 2002 summer only), traditional methods for the detection of changes following landuse disturbance could not be applied. Therefore, the chosen substituting method constituted of analysing post-treatment measurements only in terms of non-exceedance probabilities according to some thresholds for aquatic organism disturbance. Criteria for the protection of aquatic life used by the Ministry of Sustainable Development, Environment and Parks of the Province of Quebec (MDDEP, 2006) were selected for this analysis. The first criterion, chronic effect, is the maximum concentration of a substance at which aquatic organisms can be exposed all life long without undergoing adverse effects. The second criterion, acute toxicity, is the maximum concentration of a substance at which aquatic organisms can be exposed on a short period of time without being gravely affected. For suspended sediments, natural concentrations in streams must not be exceeded by more than 5 $mg{\bullet}l^{-1}$ and 25 $mg{\bullet}l^{-1}$ in order to respect the first and second criteria, respectively. Monitoring made for this study (less than 100 samples for each watershed during the whole studied period) was not sufficient to calculate natural concentration means with acceptable precision according to Johnson's (1992) findings (at least 500 samples are needed). Moreover, these measurements were almost all collected after logging, thus not necessarily reflecting natural concentrations. Therefore, the 325 measures of suspended sediments from the neighbouring and untreated watershed 6 comprised in the REVEW (Fig. 1) were used to calculate a natural concentration of 1 $mg{\bullet}l^{-1}$ for the area. The median was preferred for this estimation because of the data asymmetric distribution (the mean was biased with a few very large values). The criteria for the protection of aquatic life were then estimated at 6 $mg{\bullet}l^{-1}$ for chronic effect and 26 $mg{\bullet}l^{-1}$ for acute toxicity.

Results

Temperature

For all watersheds, including the control, the warmest days for stream temperature occurred during the second post-treatment summer (2005) (Table 3). The largest daily maximum stream temperature of 13.5°C for the whole studied period was reached for treated watershed 7.2, while for treated watersheds 7.5 and 7.6 and for control watershed 0.2, the maximum recorded temperatures were 13.1°C, 12.7°C and 12.8°C, respectively. The warmest 7-day period for stream temperatures (largest mean of the daily maximums for 7 consecutive days) also occurred during the second post-treatment summer, except for watershed 7.5, where it occurred during the pre-treatment summer (2003). A maximum value of 12.0°C for the whole studied period was observed for control watershed 0.2, while they were of 11.5°C, 11.9°C and 11.8°C for treated watersheds 7.2, 7.5 and 7.6, respectively. The dates at which the warmest discrete days or the warmest seven (7) consecutive days occurred did differ between watersheds but in both cases, no effect can be attributed to forest harvesting.

Graphical analysis of mean hourly stream temperatures on control watershed 0.2 suggested that the temperatures during summer 2005 were naturally about 1°C warmer than those of 2003 and 2004 (Fig. 2). Also, as shown for summer 2003, daily maximum stream temperature naturally occurred 2 to 3 hours later during the afternoon on the control watershed than on the treated ones. As compared to the control watershed, forest harvesting seemed to have caused an increase of the mean hourly stream temperatures of about 1°C during the summer following logging watershed 7.2 and of about 1.5°C during the second summer. On watershed 7.5, diminutions of about 1°C and 0.5°C for the first and second post-treatment summers, respectively, were suggested, while on watershed 7.6, augmentations near 0.5°C were noted for both summers after harvesting.

During the summer before treatment, mean daily maximum temperature on the treated watershed 7.2 was about the same as that of the control watershed, but in 2004, it was about 0.5°C warmer than that of the control, while in 2005, it was again very close to that of the control (Fig. 3a). These observations suggest that forest harvesting is responsible for a small augmentation in daily maximum stream temperatures on watershed 7.2 for the first post-treatment summer only. On watershed 7.5, a slight augmentation of about 0.5°C was noticed in 2004 while a reduction of at least 0.5 °C was observed in 2005. For watershed 7.6, the mean daily maximum temperatures rise due to the forest harvest reached about 2°C and 1°C in 2004 and 2005, respectively. On watershed 7.2, forest harvesting resulted in an augmentation of a few tenth of a degree of the mean daily minimum temperatures during the first post-treatment summer and a reduction of a few tenth of a degree during the second post-treatment summer (Fig. 3b). The same tendency was observed for watershed 7.5. For watershed 7.6, however, logging considerably increased mean daily minimum temperatures by nearly 2°C in 2004 and by about 1°C in 2005. The control watershed showed that, compared to 2003, diurnal stream temperature variations were naturally reduced by 0.1 to 0.2°C in 2004 and naturally augmented by a few tenth of a degree in 2005 (Fig. 3c). The same tendency was noticed for watershed 7.2. For watersheds 7.5 and 7.6, diminutions of nearly 0.5°C were observed for the two post-treatment summers.

The post-treatment measured temperatures compared to the predicted ones (from the regression between the pre-treatment temperatures of the treated and control watersheds) revealed that daily maximum stream temperature for the summer season underwent significant, but small, augmentations on watershed 7.6 only and for the two post-treatment summers (less than 1.0°C corresponding to a less than 10% increase) (Table 4). No modification was observed for watershed 7.2 for the first post-treatment summer while a very small significant decrease of 0.1°C occurred during the second summer. Significant diminutions of 0.4 and 0.3°C for 2004 and 2005 were measured, respectively, for watershed 7.5, but they did not represent more than 4% decreases. When grouping together the daily maximum stream temperatures that exceeded 10°C, significant increases of less than 1°C were still observed for watershed 7.6 for the two post-treatment summers, as well as significant decreases of less than 0.5°C on watershed 7.5. However, for watershed 7.2, significant augmentations of 1.0°C and 0.8°C for 2004 and 2005 were found, respectively, but they did not represent more than 10% increases. When grouping together the daily maximum stream temperatures smaller than 10°C, the same tendency occurred for watersheds 7.5 and 7.6 while the opposite was observed on watershed 7.2 with significant diminutions of 0.2°C and 0.5°C for 2004 and 2005 respectively.

Regarding the daily minimum stream temperatures, no significant change occurred for the first post-treatment summer for watersheds 7.2 and 7.5. During the next summer, a very small significant diminution of 0.1°C occurred for watershed 7.2 while a very small significant augmentation of 0.1°C was observed for watershed 7.5. On watershed 7.6, significant increases of 0.8°C and 1.0°C were noticed for 2004 and 2005, respectively. When grouping together the daily minimum stream temperatures that exceeded 8°C, the same tendency was observed for watersheds 7.5 and 7.6, but significant augmentations of 0.5°C and 0.2°C occurred on watershed 7.2 for the 2004 and 2005 summers, respectively. When grouping together the daily minimum stream temperatures smaller than 8°C, the same tendency was again observed for watersheds 7.5 and 7.6, except that the rises were slightly greater (up to 17% in 2004 for watershed 7.6). In the case of watershed 7.2, the opposite occurred with diminutions of 0.2°C and 0.5°C for 2004 and 2005 respectively.

For diurnal variations, no change was noticed for watershed 7.2 while diminutions occurred for watersheds 7.5 and 7.6. These diminutions are all small in terms of absolute temperature (up to 0.5°C for watershed 7.5 in 2005), but large in terms of percentage (up to 39% for the same watershed and summer).

Nutrients concentration (NO_3^- and PO_4^{3-})

NO_3^- concentrations remained stable for the control watershed throughout the studied period (Fig. 3d; Table 5). For the treated watersheds, important augmentations were mostly noticed during the second post-treatment summer. The non-exceedance probability curves also suggested augmentations in NO_3^- concentration following clearcutting, with most of the treated watershed curves substantially moving to the right (except for watershed 7.5) and the control watershed curve remaining stable (Fig. 4). The graphics revealed that the increases of the 1st quartiles of watersheds 7.2, 7.3, 7.6 and 7.7 varying between 0.01 and 0.05 $mg{\bullet}l^{-1}$ were all very weak compared to the 3rd quartiles varying between 0.23 and 0.37 $mg{\bullet}l^{-1}$, suggesting very weak increases for the small NO_3^- concentrations and great augmentations for the large ones. Statistical analysis demonstrated that, during the first post-treatment summer, watersheds 7.2 and 7.3 did not experience any significant modification, while watersheds 7.5, 7.6 and 7.7 underwent small increases of 0.01, 0.05 and 0.04 $mg{\bullet}l^{-1}$, respectively, corresponding to substantial relative changes varying between 451 and 756% (Table 4). During the second post-treatment season, NO_3^- concentrations increased for all five watersheds, with augmentations of 0.37, 0.31, 0.06, 0.36 and 0.35 $mg{\bullet}l^{-1}$ on watersheds 7.2, 7.3, 7.5, 7.6 and 7.7, respectively, representing increases in terms of percentage ranging from 258 to 6089%.

When compared to the control watershed, PO_4^{3-} concentration naturally and importantly decreased after the pre-treatment summer and remained relatively stable for the two post-treatment summers (Fig. 3e; Table 5). The same tendency was observed on the treated watersheds, except that all phosphate concentrations seemed to increase slightly during the second post-treatment summer. All the non-exceedance probability curves moved to the left from the pre-treatment to the post-treatment summers (not shown), also suggesting a natural decrease throughout the studied period. However, with this graphical method, it was not possible to determine whether the decrease was more important for the treated watersheds than for the control. The same conclusion was drawn from the quartile analysis, since for the same quartile, phosphate concentrations were all reduced by the same magnitude between the control and treated watersheds. Only small augmentations of 0.004 and 0.006 $mg \cdot l^{-1}$ for watershed 7.2 in 2004 and 2005, respectively, and of 0.007 $mg \cdot l^{-1}$ on watershed 7.6 in 2005 were found to be statistically significant (Table 4). These augmentations did not exceed more than 30% the mean phosphate concentrations measured before logging.

pH

Stream water pH naturally decreased from summer 2000 to summer 2005, but for treated watersheds 7.2, 7.3, 7.6 and 7.7, the diminutions during post-treatment summers were more important than on control watershed 0.2, suggesting reductions in stream water pH due to forest harvesting (Fig. 3f; Table 5). For treated watershed 7.5, the decrease was similar to that of the control watershed. The 1st, 2nd and 3rd quartiles of the control watershed were all reduced between the pre- and post-treatment summers, but not by more than 0.3 pH unit. Except for watershed 7.5, most quartiles of the treated watersheds underwent more important reduction than the control. In the cases of watersheds 7.2 and 7.3, the reduction was the greatest for the 1st quartiles and the weakest for the 3rd. Therefore, following logging, pH decreased more for the smaller pH values than for the larger ones. Inversely, on watersheds 7.6 and 7.7, the reduction was the greatest for the 3rd quartiles and the weakest for the 1st quartiles, demonstrating that pH decreased more for the larger pH than for the smaller ones. The small diminutions of stream water pH on the treated watersheds were statistically significant for the two post-treatment summers (Table 4). Decreases of 0.4, 0.2, 0.3 and 0.2 pH unit were observed during 2004 on watersheds 7.2, 7.3, 7.6 and 7.7, respectively, while diminutions of 0.2, 0.1, 0.2 and 0.2 unit occurred in 2005. These drops correspond to small decreases, varying between 1 and 7%. Inversely, pH significantly increased by 0.2 and 0.1 on watershed 7.5 for the two post-treatment summers respectively. These 3 and 2 % rises are also considered small.

Major cations concentration (K^+, Mg^{2+} and Fe_{total})

As revealed from the control watershed, a slight decrease in K^+ concentration naturally occurred during the two post-treatment summers (Fig. 3g; Table 5). Inversely, increases were noticed for the treated watersheds especially for the second summer. The increases observed for the treated watersheds were the weakest for the 1st quartile and the strongest for the 3rd quartile, except on watershed 7.2. Therefore, forest harvesting did result in a more important potassium augmentation for the larger concentrations. Statistical analysis corroborated the above observations, with significant rises of 0.34, 0.27, 0.11, 0.20 and 0.14 mg•l^{-1} of K in 2004 for watersheds 7.2, 7.3, 7.5, 7.6 and 7.7, respectively, corresponding to relative augmentations of up to 125% (Table 4). In 2005, the significant increases for these watersheds were respectively of 0.39, 0.34, 0.42, 0.63, and 0.58 mg•l^{-1} representing rises of up to 283%.

For all the treated watersheds during the first post-treatment summer, small decreases were observed in Mg^{2+} concentration, while important increases were noticed for the second post-treatment summer, except for watershed 7.5 (Fig. 3h; Table 5). However, the quartile analysis did not show any tendency for the treated watersheds, with augmentations in some cases and diminutions in others. The modifications were very small and the reductions of Mg^{2+} concentration in 2004 were not found to be statistically significant for any treated watersheds (Table 4). Inversely, in 2005, the augmentations of 0.067, 0.044, 0.022 and 0.039 mg•l^{-1} were all significant for watersheds 7.2, 7.3 7.6 and 7.7, respectively, corresponding to relative increases varying between 2% and 19%. No significant change was noticed for watershed 7.5.

Based on the control watershed, Fe_{total} concentration seemed to naturally increase during the post-treatment period, especially during the second summer (2005) (Fig. 3i; Table 5). Increases due to logging are noticed for 2005 only and for the treated watersheds 7.5, 7.6 and 7.7. For watersheds 7.2 and 7.3, the increases of the 1st and 2nd quartiles were weaker than that of the control watershed while the increases of the 3rd quartiles were larger. For the other treated watersheds, the increases of all quartiles were greater than for the control. Moreover, the increases of the 3rd quartiles were larger, showing that forest harvesting was responsible for more important Fe_{total} augmentations for the large concentrations. Statistical analysis revealed that Fe_{total} concentration was maintained to pre-treatment levels during the two post-treatment summers on watershed 7.2, during the second post-treatment summer on watershed 7.3 and during the first post-treatment summer on watersheds 7.5 and 7.6 (Table 4). The 2004 increase on watershed 7.3 was 0.056 $mg{\bullet}l^{-1}$, representing a 71% augmentation. On watersheds 7.5 and 7.6, the increases in 2005 were of 0.122 and 0.066 $mg{\bullet}l^{-1}$, respectively, corresponding to rises of 55% and 23%. On watershed 7.7, total iron concentration augmented to 0.058 and 0.139 $mg{\bullet}l^{-1}$ in 2004 and 2005, respectively, which represents increases of 30% and 71% when compared to the level measured during the pre-treatment period.

Sulphate concentration

SO_4^{2-} concentrations had significantly augmented during the two post-treatment summers on watersheds 7.2, 7.3 and 7.5 (Fig. 3j; Table 4 and 5). The augmentations on these three watersheds were, respectively, of 0.88, 0.88 and 0.13 $mg{\bullet}l^{-1}$ in 2004 and of 0.82, 0.83 and 0.07 $mg{\bullet}l^{-1}$ in 2005. Rises varying between 54% and 58% represent the increases observed for watersheds 7.2 and 7.3, while rises of only 6% and 3% correspond to the increases for watershed 7.5 in 2004 and 2005, respectively. Watershed 7.6 underwent increases of 0.85 $mg{\bullet}l^{-1}$ during the first post-treatment summer only, representing a 54% rise. For watershed 7.7, the SO_4^{2-} concentrations decreases were not significant for the two summers following logging. Also, watersheds 7.2, 7.3 and 7.6 did experience smaller decreases in their quartiles than the control, suggesting increases in sulphate concentration due to logging.

Specific conductance

Specific conductance was naturally greater during post-treatment summers than the pre-treatment ones, but the augmentations were much more important for the treated watersheds than for the control, especially for the second post-treatment summer (Fig. 3k; Table 5). The quartiles revealed that the specific conductance augmentations were in part due to forest harvesting since increases greater than the control were observed for almost all quartiles of the treated watersheds. On watersheds 7.5, 7.6 and 7.7, the increases were the weakest for the 1st quartile while they were the strongest for the 3rd quartile. Forest harvesting was responsible for more important augmentations in the case of the larger specific conductances. For watersheds 7.2 and 7.3, no clear tendency was observed in relation to the magnitude of the conductance.

Statistical analysis revealed that specific conductance mostly increased during the second post-treatment summer on the treated watersheds (Table 4). For the first summer following logging, significant increases of 1.0 $\mu S \cdot cm^{-1}$ and 0.5 $\mu S \cdot cm^{-1}$ were only observed on watersheds 7.5 and 7.7, respectively, representing small rises of 6% and 3%. In 2005, all treated watersheds, except watershed 7.3, experienced significant augmentations of 2.2, 3.2, 4.2 and 4.1 $\mu S \cdot cm^{-1}$ on watersheds 7.2, 7.5, 7.6 and 7.7, respectively, corresponding to large increases of 12%, 20%, 26% and 26%.

Suspended sediment concentration

Suspended sediments include organic and inorganic particulate matter that was filtered out of the water using a 0.45 μm filter. Without the pre-treatment non-exceedance probability curves, the post-treatment curves cannot be used to determine whether logging caused modifications in suspended sediment concentration at Montmorency Forest (Fig. 5). However, they can reveal by what magnitude the criteria for the protection of aquatic life (MDDEP, 2006) were exceeded. They demonstrated clearly that during the post-treatment summers, the control watershed never exceeded the criteria while all treated watersheds did for at least one criterion, suggesting that logging may have resulted in small increases in suspended sediment concentrations. The criterion for chronic effect was exceeded four (4) times for watersheds 7.5 and 7.6, five (5) times for watershed 7.3, six (6) times for watershed 7.7 and eight (8) times for watershed 7.2, which corresponded to non-exceedance probabilities varying between 0.91 and 0.95 (Table 6). The acute toxicity criterion was exceeded only once and on watershed 7.6 only, which corresponded to a non-exceedance probability of 0.99.

Discussion

Temperature

It has been reported lately that logging in the upstream parts of watersheds can have a statistically significant effect on stream water temperature downstream because of the non-maintenance of a forest buffer along upstream water courses which are not permanent or not visible at the surface (Bourque and Pomeroy, 2001). This may explain the most important, but nevertheless small increases in daily maximum and minimum temperatures noticed for watershed 7.6 (up to 17% the pre-treatment temperature), where the harvest was made in the upper sections, far from the permanent hydrographic network. Based on field observations, the steeper topography in the upper parts of the watershed showed evidences of surface convergence zones that may canalize runoff only during important rain storms. These ephemeral channels were not protected with buffer strips. Moreover, upstream water courses invisible at the surface were often formed below the soil humus layer and again, no riparian vegetation was maintained.

The unexpected decreases in maximum and minimum temperatures for watershed 7.5 were contrary to the general findings reported in the literature. However, the diminutions were small, not exceeding more than 5% the pre-treatment temperatures. The stream of the small watershed 7.5 became intermittent during short periods of time during the pre- and post-treatment summers. Intermittent periods are associated with droughts, which in turn are associated with colder groundwater inputs (during summer only) constituting the principal component of the discharge (Alexander and Caissie, 2003) before streams become dry. The unimportant decreases in stream temperature of watershed 7.5 could be attributed to a bias in the relationships with the control watershed. Moreover, external factors rather than forest harvesting, such as a change in the location of the thermometer on the stream bed or other field and instrument manipulations. Although, the same is true for the small augmentations described above.

Watershed 7.2, which was clearcut near the hydrographic network in the downstream parts of the watershed, did experience augmentations of the higher temperatures and diminutions of the smaller ones. However, the modifications were small, with increases reaching 10% the pre-treatment temperatures and decreases not exceeding 7%. This tendency suggested that stream water was heated more intensively than before the cut during warm days and the losses of energy increased during colder days due to canopy removal. The same tendency was observed for the daily maximum and minimum temperatures, suggesting that this phenomenon was occurring during both day and night. Other processes besides heating from direct solar radiation can influence stream water temperature, such as evaporation and sensible heat exchange with air (Sinokrot and Stefan, 1993) and energy exchange from conduction with alluvial substrates and hyporheic exchange (Johnson and Jones, 2000; Story et al., 2003).

The criterion for the protection of aquatic life stipulates that temperature changes should not result in the displacement or modification of aquatic populations, as well as it should not disturb localized sensitive zones such as spawning beds (MDDEP, 2006). No thresholds are defined in absolute degrees Celsius. Since the changes observed at Montmorency Forest are very small (not exceeding 1°C), it can be assess that disturbing thresholds for aquatic organisms were not attained.

Nutrients concentration (NO_3^- and PO_4^{3-})

Nitrification intensification was responsible for mean NO_3^- concentration rises; reaching more than 6000% pre-treatment values at Montmorency forest (Martin et al., 2000). However, this augmentation, corresponding to a near 0.4 mg•l^{-1} mean increase, was not so extreme for the boreal forest of Quebec where a mean augmentation of 0.8 mg•l^{-1} has already been observed in Charlevoix (Plamondon, 1994). The NO_3^- criterion for the protection of aquatic life (MDDEP, 2006) stipulates that more than 40 mg•l^{-1} of NO_3^- are needed for aquatic organisms to undergo chronic effects while 200 mg•l^{-1} must be reached for acute toxicity. From a 14-study review, it was found that maximum NO_3^- concentration increase in stream water can reach 3.7 mg•l^{-1} following forest harvesting (Brown and Binkley, 1994). In Charlevoix, it even reached 4.03 mg•l^{-1} for a buffered stream (Plamondon, 1994). These values are far from being reached at Montmorency Forest, where a maximum concentration of 0.82 mg•l^{-1} was measured during the second summer after logging.

In this study and from a 10 year study on watersed 7A of the BEREV it was observed that the maximum increase of NO_3^- concentration occurs during the second year after logging (Marquis 2006, personal communication). This has also been observed elsewhere following forest harvesting (Jewett et al., 1995; Swank et al., 2001). As proposed by Jewett et al. (1995), this lag may be caused by an increased availability of readily metabolized carbohydrates such as wood debris that could contribute to an initial weak soil NO_3^- concentration due to increased microbial N immobilization.

For the two summers following logging, the small annual mean augmentation in PO_4^{3-} of less than 0.007 $mg•l^{-1}$ only occurred on watershed 7.2, where the cutover was located near the stream network and logging was partially allowed within the forest buffer. From a 17-study review, about half of the watersheds did not experience any change in PO_4^{3-} concentration, while the mean changes varied between -0.004 and 0.04 $mg•l^{-1}$ on the others (Brown and Binkley, 1994). The relative efficiency of the forest buffers at Montmorency Forest may have restrained PO_4^{3-} augmentation such as elsewhere in the boreal forest of Quebec where increases up to 0.08 $mg•l^{-1}$ have been observed in streams without buffers (Plamondon, 1994). The fact that Ca^+, that immobilizes P, generally increases in content with depth in the boreal forest (Evans et al., 2000) may also explain this relative stability.

No limit in the concentration of PO_4^{3-} exists for the protection of aquatic life, but total P must not exceed 0.03 $mg•l^{-1}$ according to the chronic effect criterion (MDDEP, 2006). Since orthophosphates usually represent about 40% of total P in forested ecosystem (Ice, 1999), the chronic effect criterion was probably exceeded before and after the harvest for several water samples collected on the control and treated watersheds at Montmorency Forest.

pH

Nitrification intensification producing nitric acid (HNO^{3-}) contributed to the small pH decreases observed on most of the treated watersheds at Montmorency Forest (Feller, 2005; Martin et al., 2000). Acidification of stream water actually occurred with important increases in NO_3^- concentration. The release of organic acids from the decomposition of trimmed branches left in place during the harvesting operations has also contributed to reduce stream pH (Feller, 2005). Small augmentations of 0.2 or less pH unit have occurred for watershed 7.5 only, from which the smallest NO_3^- augmentation in terms of absolute values (in mg•l^{-1}, not in %) were measured. Significant increases of up to 0.5 pH unit have also been observed elsewhere in the boreal forest of Quebec (Plamondon, 1994; Seto, 2005). The criterion for the protection of aquatic life cannot be applied here since natural pH for stream water at Montmorency Forest is below the lower limit stipulated by the MDDEP (2006).

Major cations concentration (K^+, Mg^{2+} and Fe_{total})

The augmentations in K^+ generally do not exceed 1.5 mg•l^{-1} following logging (Jewett et al., 1995; Martin et al., 2000; Rosén et al., 1996; Swank et al., 2001). The maximum augmentation of 0.63 mg•l^{-1} measured at Montmorency Forest was thus moderate, but still more important than the increase range of 0.04 to 0.3 mg•l^{-1} observed elsewhere in the boreal forest of Quebec when the riparian vegetation was maintained (Plamondon, 1994). Without forest buffers, an increase reaching more than 3 mg•l^{-1} was measured in Haute-Mauricie, Quebec (Plamondon, 1994). Chronic effects on aquatic fauna occur when K^+ concentration exceeds 5 mg•l^{-1} (Plamondon, 2002). When 400 mg•l^{-1} is attained, certain fishes can die while mortality in invertebrates occurs at concentrations exceeding 700 mg•l^{-1} (Plamondon, 2002). The maximum concentration measured after the clearcuts at Montmorency Forest was only 1.4 mg•l^{-1}, a concentration considered too small to have adverse effects on aquatic life.

The maximum average augmentation in Mg^{2+} of 0.067 mg•l^{-1} measured at Montmorency Forest is moderate compared to what has been observed elsewhere in the boreal forest of Quebec, where concentration rises ranged from 0.01 to 0.2 (Plamondon, 1994). The maximum concentration of 0.531 mg•l^{-1} measured after logging was much smaller than the criterion associated for chronic effect (14 mg•l^{-1}) and for acute toxicity (100-400 mg•l^{-1}) on aquatic fauna (Plamondon, 2002).

The maximum augmentation in Fe_{total} of 0.139 mg•l^{-1} measured at Montmorency Forest was similar to the increases that did not exceed 0.1 mg•l^{-1} elsewhere in the boreal forest of Quebec where the riparian vegetation was protected (Plamondon, 1994). The Quebec chronic effect criterion of 0.3 mg•l^{-1} for the protection of aquatic organisms (MDDEP, 2006) was exceeded for more than 50% of the samples on the treated watersheds, while it was exceeded for about 40% of the samples from the control watershed. The iron-rich soils (ferro-humic podzols) of the studied watersheds (Plamondon and Naud, 1975) could explain these great exceedance frequencies. However, the 1.0 mg•l^{-1} American criterion was never exceeded (EPA, 1986).

Sulphate concentration (SO_4^{2-})

At Montmorency Forest, the SO_4^{2-} increase observed on most of the treated watersheds is contrary to the general findings reported in the literature (Feller, 2005; Likens et al., 2002; Welsch et al., 2004). It can be hypothesized that SO_4^{2-} soil adsorption capacity increase, the main process normally causing reduction in SO_4^{2-} (Feller, 2005; Welsch et al., 2004), may have been prevented by the pH reduction that was quite small. Moreover, enhanced atmopheric deposition of SO_4^{2-} coming from unusually important forest fires in Quebec could also be responsible for the increases observed during the second post-treatment summer only. In fact, during the 10 last years (1996 to 2005), 38% of the 1007794.5 ha area burned occurred during the summer of 2005 (representing about 380% of the mean annual affected area) (SOPFEU, 2006). The acute toxicity criterion of 300 mg•l^{-1} (MDDEP, 2006) was by far never exceeded since the largest measurement recorded for the whole studied period was 3.4 mg•l^{-1}.

Specific conductance

The important increases in NO_3^- and K^+ concentration, as well as the smaller augmentations in Mg^{2+}, Fe_{total} and SO_4^{2-}, were directly responsible for the specific conductance increases measured in the treated watersheds of the Montmorency Forest. The augmentations, ranging from 0.5 to 4.2 µS•cm^{-1}, were similar in magnitude to the increases measured elsewhere in the boreal forest of Quebec (i.e., ranging from 0 to 8 µS•cm^{-1}) where forest buffers were maintained (Plamondon et al., 1982). Where the riparian vegetation had been removed, augmentations of up to 70 $\mu S \bullet cm^{-1}$ were observed (Plamondon, 1994). The moderate increases in specific conductance (not exceeding 26%) were probably associated with the maintenance of forest buffers along permanent streams at Montmorency Forest. Specific conductance of more than 2000 $\mu S \bullet cm^{-1}$ can be detrimental for fresh water fishes (Plamondon, 2002). This value is more than 60 times larger than the maximum of 31.6 $\mu S \bullet cm^{-1}$ measured the second summer after clearcutting at Montmorency Forest.

Suspended sediment concentration

The great non-exceedance probabilities of the criterion for the protection of aquatic life (MDDEP, 2006) suggest that logging did not cause important augmentation in suspended sediment concentration that could have impacted the aquatic ecosystem at Montmorency Forest. The few times the chronic effect criterion was exceeded after treatment is not of a great concern since, by definition, the criterion must be exceeded over a long period of time in order to affect adversely the aquatic organisms. The dates at which it occurred did not show long successive periods, meaning that the recorded events were of short duration. For example, the eight (8) dates at which the criterion was exceeded for watershed 7.2 were, in 2004, June 16, 18 and 25, July 23, August 13 and 27, and in 2005, September 26. Several measurements made between those dates (sampling was performed two to three times a week) showed concentrations smaller than the criterion. The case of the acute toxicity criterion may be of greater concern since, by definition; the criterion must be exceeded over a short period of time to affect aquatic organisms. However, it was exceeded only once and for only one watershed. The July 12 concentration of 30.5 $mg \bullet l^{-1}$ for watershed 7.6 was measured between two peak events when stream flow was moderate. For the other watersheds, suspended sediment concentrations were all small, not exceeding 1.4 $mg \bullet l^{-1}$. It was therefore not possible to link this event for watershed 7.6 to forest harvesting. For example, a stream bank collapse may have been the cause for this large concentration. Forest buffer strips along streams have prevented large increases in suspended sediment concentrations, as found elsewhere in the boreal zone of Quebec (Plamondon, 1982; Seto, 2005). The short distance of the cutover from the stream and the log removal into the riparian zone may be responsible for the smallest non-exceedance probability on watershed 7.2.

Conclusion

Clearcutting 50% of watershed area at Montmorency caused small augmentations in summer daily maximum and minimum stream temperatures on one watershed and small diminutions on another. On the third watershed, small augmentations as well as small diminutions occurred. For diurnal temperature variations, it mostly diminished for all three treated watersheds on which temperature was recorded. Large increases in NO_3^- and K^+ concentrations were observed on all five treated watersheds on which samples were collected, while smaller increases were noticed for specific conductance, Mg^{2+} and Fe_{total} concentrations on four out of the five watersheds. Partly because of important forest fires that occurred during the second post-treatment summer, SO_4^{2-} concentration also significantly increased on four out of the five watersheds. While pH declined on all watersheds, PO_4^{3-} concentration remained relatively stable on three watersheds and increased very weakly on the two others. Suspended sediment concentration seemed to have increased for all five watersheds, but the results are not certain because of the lack of sufficient pre-treatment data.

The results obtained in this study are generally in concordance with those found in the literature (table 1). SO_4^{2-} concentration was the main exception partly because of the enhancement of SO_4^{2-} atmospheric deposition caused by the great forest fires that occurred in Quebec during the second post-treatment summer. Small diminutions in stream temperature measured on some watersheds are also opposite to the literature, but greater losses of energy at night and during cooler days may have more than compensated the warming effects of sunny days. The small increases in PO_4^{3-} concentration observed on two watersheds are not surprising since some augmentations have already been reported elsewhere.

Based on numerous studies, careful logging may not modify water quality parameters beyond the drinking standards (Plamondon, 1994), although aquatic organisms can be affected (Haggerty et al., 2004; Hartman et al., 1996; Latterell et al., 2003; Martin et al., 2000). However, following the logging treatment at Montmorency Forest, temperature, specific conductance, NO_3^-, K^+, Mg^{2+}, and SO_4^{2-} concentrations were very far to exceed their criteria for the protection of aquatic life recognized in Quebec (MDDEP, 2006) and in the U.S. (EPA, 1986). The PO_4^{3-} and Fe_{total} chronic effect criterion may have been exceeded at several times, but it occurred before and after the treatment, as well as on the control and the treated watersheds. Also, natural pH of Montmorency Forest was already below the criteria. Finally, suspended sediment concentration criteria was exceed a few times after treatment, but the lack of pre-treatment data did not permit to conclude that it was due to forest harvesting. Even if other studies have failed to demonstrate a relationship between clearcut intensity and water quality modifications (Degenhardt and Ice, 1996; Martin et al., 1981) the moderate forest harvesting intensity (50% of watershed area clearcut) at Montmorency Forest may explain the small modifications in relation to water quality standards and protection of aquatic life. The respect of good management practices, such as maintenance of a forest buffer along perennial streams, forest operations on snow cover and trimmed branches for the protection of soils, and evenly spaced skid trails for the protection of forest regeneration, may have also contributed to the weak changes in water quality.

The results of this study did not clearly demonstrate that water quality is more influenced by landuse disturbance near streams than throughout the watershed (Basnyat et al., 1999; Plamondon, 1993), except maybe for the PO_4^{3-} and suspended sediment concentrations increases on watershed 7.2 which was clearcut close to its hydrographic network. Nevertheless, these exceptions may be directly related to the allowance of log removal within the riparian zone rather than the distance of the clearcut from the stream. Moreover, the PO_4^{3-} concentration increase is very small and the suspended sediments results are not certain since the pre-treatment data set is not large enough. The results of this study did however clearly demonstrate that the presence of the forest road did not play a role in the augmentation of suspended sediment concentration, nor as it did during its construction (Beaudin, 2002).

The present study was executed in parallel with another study trying to validate the hypothesis that clearcutting 50% of watershed area does not increase bankfull discharge more than 50%, a threshold at which stream morphology may be significantly modified and consequently affect aquatic organisms (Tremblay et al., 2006). This study determined that the 50% treatment area criterion may be sufficient in terms of bankfull discharge augmentation. According to the results presented in the present paper, the same conclusion can be suggested for water quality.

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References

Alexander, M.D. and Caissie, D., 2003. Variability and comparison of hyporheic water temperatures and seepage fluxes in a small Atlantic salmon stream. Ground Water, 41: 72-82.

Barry, R., 1984. Prédiction de la teneur en eau du sol minéral dans la sapinière laurentienne de la forêt Montmorency. Mémoire de maîtrise, Université Laval, 100 p.

Basnyat, P., Teeter, L.D., Flynn, K.M. and Lockaby, B.G., 1999. Relationships between landscape characteristics and nonpoint source pollution inputs to coastal estuaries. Environmental Management, 23: 539-549.

Beaudin, 2002. Peak and storm flow response to forest road construction, Mémoire de maîtrise, Université Laval, Quebec, 60 pp.

Bélanger, L., Bertrand, L., Bouliane, P. and Lussier, L.J., 1991. Plan général d'aménagement de la forêt Montmorency. Université Laval, 216 pp.

Beschta, R.L., 1978. Long-term patterns of sediment production following road construction and logging in the Oregon Coast Range. Water Resources Research, 14: 1011-1016.

Beschta, R.L., Pyles, M.R., Skaugset, A.E. and Surfleet, C.G., 2000. Peakflow response to forest practices in the western cascades of Oregon, USA. Journal of Hydrology, 233: 102-120.

Beschta, R.L. and Taylor, R.L., 1988. Stream temperature increases and land use in a forested Oregon watershed. Water Resources Bulletin, 24: 19-25.

Bourque, C.P.-A. and Pomeroy, J.H., 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: an inter-catchment, multiple-year comparison. Hydrology and Earth System Sciences, 5: 599-613.

Brown, G.W. and Krygier, J.T., 1970. Effects of clear-cutting on stream temperature. Water Resources Research, 6: 1133-1139.

Brown, G.W. and Krygier, J.T., 1971. Clear-cut logging and sediment production in the Oregon Coast Range. Water Resources Research, 7: 1189-1198.

Brown, T.C. and Binkley, D., 1994. Effect of management on water quality in north american forests. USDA Forest Service General Technical Report, RM-248, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado.

Caissie, D., Jolicoeur, S., Bouchard, M. and Poncet, E., 2002. Comparison of streamflow between pre and post timber harvesting in Catamaran Brook (Canada). Journal of Hydrology, 258: 232-248.

Caissie, D., Pollock, T.L. and Cunjak, R.A., 1996. Variation in stream water chemistry and hydrograph separation in a small drainage basin. Journal of Hydrology, 178: 137-157.

Croke, J.C. and Hairsine, P.B., 2006. Sediment delivery in managed forests: a review. Environmental Reviews, 14: 59-87.

Day, T.J., 1988. Evaluation of long term sediment records for selected Canadian rivers. Sediment Budgets, IAHS Publication, 174: 189-195.

Degenhardt, D. and Ice, G., 1996. Forest management options to control excess nutrients for the Tualatin River, Oregon. Proceedings of the 1995 NCASI West Coast Regional Meeting, Special Report No. 96-04: 2-17.

EPA, 1986. Quality criteria for water. Office of Water, Regulations and Standards, U.S. Environmental Protection Agency, Washington, D.C., 477 p.

Evans, J.E., Prepas, E.E., Devito, K.J. and Kotak, B.G., 2000. Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the Boreal Plain. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 60-72.

Feller, M.C., 1981. Effects of clearcutting and slashnburning on stream temperature in southwestern British Columbia. Water Resources Bulletin, 17: 863-867.

Feller, M.C., 2005. Forest harvesting and streamwater inorganic chemistry in western North America: a review. Journal of the American Water Resources Association.

Finlayson, B.L. and McMahon, T.A., 1988. Australia vs the World: a comparative analysis of streamflow characteristics. In: R.F. Warner (Editor), Fluvial Geomorphology of Australia. Academic Press, Sidney, pp. 17-40.

Guillemette, F., Plamondon, A.P. and Lévesque, D., 1998. Effet De la coupe sur le bilan hydrologique. Bassin expérimental du ruisseau des Eaux-Volées. Rapport du Ministère des Ressources Naturelles. Université Laval, 65 p.

Guillemette, F., Plamondon, A.P., Prévost, M. and Lévesque, D., 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. Journal of Hydrology, 302: 137-153.

Haggerty, S.M., Batzer, D.P. and Jackson, C.R., 2004. Macroinvertebrate response to logging in coastal headwater streams of Washington, U.S.A. Canadian Journal of Fisheries and Aquatic Science, 61: 529-537.

Hall, R.O., Likens, G.E., Fiance, S.B. and Hendrey, G.R., 1980. Experimental acidification of a stream in the Hubbard Brook Experimental Forest, New Hampshire. Ecology, 61: 976-989.

Harr, R.D., Frederiksen, R.L. and Rothacher, J., 1979. Changes in streamflow following timber harvest in southwestern Oregon. Water Resources Research, 15: 90-96.

Hartman, G.F. and Scrivener, J.C., 1990. Impacts of forestry practices on a coastal stream ecosystem, Carnation Creek, British Columbia. Canadian Bulletin of Fisceries and Aquatic Science, 223: 148 p.

Hartman, G.F., Scrivener, J.C. and Miles, M.J., 1996. Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, and their implication for restoring fish habitat. Canadian Journal of Fisheries and Aquatic Science, 53(Suppl. 1): 237-251.

Hornbeck, J.W., Martin, C.W. and Eagar, C., 1997. Summary of water yield experiments at Hubbard Brook Experimental Forest, New Hampshire. Canadian Journal of Forest Research, 27: 2043-2052.

Ice, G., 1999. Streamflow and Water Quality: What Does the Science Show about Clearcutting in Western Oregon, National Council for Air and Stream Improvement, Oregon State University, Corvallis, Oregon, pp. 20.

Jewett, K., Daugharty, D., Krause, H.H. and Arp., P.A., 1995. Watershed response to clearcutting: effects on soil solutions and stream water discharge in central New Brunswick. Canadian Journal of Soil Sciences, 75: 475-490.

Johnson, R.C., 1992. Towards the design of a strategy for sampling suspended sediments in small headwater catchements, Erosion and Sediment Transport Monitoring Programmes in River Basins. IAHS, Proceedings of the Oslo Symposium, pp. 225-232.

Johnson, S.L., 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Science, 61: 913-923.

Johnson, S.L. and Jones, J.A., 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 30-39.

Jones, J.A. and Grant, G.E., 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resources Research, 32: 959-974.

Jones, J.A., Swanson, F.J., Wemple, B.C. and Snyder, K.U., 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology, 14: 76-85.

Latterell, J.J., Naiman, R.J., Fransen, B.R. and Bisson, P.A., 2003. Physical constraints on trout (Oncorhynchus spp.) distribution in the Cascade Mountains: a comparison of logged and unlogged streams. Canadian Journal of Fisheries and Aquatic Science, 60: 1007-1017.

Likens, G.E. et al., 2002. The biogeochemistry of sulfur at Hubbard Brook. Biogeochemistry, 60: 235-316.

Lisle, T.E., 1989. Sediment transport and resulting deposition spawning gravels, north coastal California. Water Resources Research, 25: 1303-1319.

Luce, C.H. and Black, T.A., 1999. Sediment production from forest roads in western Oregon. Water Resources Research, 35: 2561-2570.

Macdonald, J.S., MacIsaac, E.A. and Herunter, H.E., 2003. The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia. Canadian Journal of Forest Research, 33: 1371-1382.

Martin, C.W., Hornbeck, J.W., Likens, G.E. and Buso, D.C., 2000. Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. Canadian Journal of Fisheries and Aquatic Science, 57(Suppl. 2): 19-29.

Martin, C.W., Noel, D.S. and Federer, C.A., 1981. The effect of forest clearcutting in New England on streamwater chemistry and biology, USDA Forest Service Northeastern Forest Experiment Station, Technical Completion Report, Project No. A-051-NH.

MDDEP, 2006. Critères de qualité de l'eau de surface au Québec. Ministère du Développement Durable, de l'Environnement et des Parcs, http://www.mddep.gouv.qc.ca/eau/criteres_eau/index.htm, Visited on June 21 2006.

Megahan, W.F. and Kidd, W.J., 1972. Effect of logging and logging roads on erosion and sediment deposition from steep terrain. Journal of Forestry, 70: 136-141.

Novotny, V. and Chesters, G., 1989. Delivery of sediment and polluants from nonpoint sources: a water quality perspective. Journal of Soil and Water Conservation, 44: 568-576.

Olive, L.J. and Rieger, W.A., 1992. Stream suspended sediment transport monitoring - why, how and what is being measured? Erosion and Sediment Tranport Monitoring Programmes in River Basins. IAHS, Proccedings of the Oslo Symposium, pp. 245-254.

Plamondon, A.P., 1982. Augmentation de la concentration des sédiments en suspension suite à l'exploitation forestière et durée de l'effet. Canadian Journal of Forest Research, 12: 883-892.

Plamondon, A.P., 1993. Influence des coupes forestières sur le régime d'écoulement de l'eau et sa qualité. Revue de littérature. Ministère des Forêts du Québec. Rapport C-47, 179 pp.

Plamondon, A.P., 1994. Impacts of clearcutting on water quality with an emphasis on the boreal forest, Northern Research Basins Study (NRBS) Workshop. Government of Alberta and Faculty of Agriculture and Forestry, University of Alberta, Edmonton, pp. 30.

Plamondon, A.P., 2002. Manuel de météorologie et hydrologie forestières appliquées à l'aménagement des bassins versants. Centre de recherche en biologie forestière, Faculté de foresterie et géomatique, Université Laval, St-Foy.

Plamondon, A.P., Gonzalez, A. and Thomassin, Y., 1982. Effects of logging quality: comparison between two Quebec sites, Proceedings of the Canadian Hydrologic Symposium, Fredericton, New Brunswick, pp. 47-70.

Plamondon, A.P., Lévesque, D., Ma, Y. and Prévost, M., 1998. Long-term effects of forest mosaic management on storm and snowmelt flow, Quebec. Proceedings of the British Hydrological Society International Conference, Exeter, July 1998: Hydrology in a changing environment, 1: 503-515.

Plamondon, A.P. and Naud, R.C., 1975. The watershed management program at the Montmorency experimental forest, Usefulness of an experimental forest research and teaching, Symposium, Forêt Montmorency. Faculté de foresterie, Université Laval, Québec, pp. 129-152.

Plamondon, A.P. and Ouellet, D.C., 1980. Partial clearcutting and streamflow regime of Ruisseau des Eaux-Volées experimental watershed. Proceedings of the Helsinki Symposium. IAHS, 130: 129-136.

Reid, L.M. and Dunne, T., 1984. Sediment production from forest road surfaces. Water Resources Research, 20: 1753-1761.

Robitaille, A. and Saucier, J.P., 1998. Paysages régionaux du Québec méridional. Publications du Québec, St-Foy, 213 pp.

Rochette, F.J., 1971. Hydrological study of Ruisseau des Eaux-Volées experimental basin. M.Sc. thesis, University of Western Ontario, 88 pp.

Rosén, K., Aroson, J.-A. and Erikson, H.M., 1996. Effects of clear-cutting on streamwater quality in forest catchments in central Sweden. Forest Ecology and Management, 83: 237-244.

Seto, M., 2005. Effets de l'exploitation forestière sur la qualité de l'eau en forêt boréale. Mémoire de maîtrise. Université Laval, pp. 97.

Sinokrot, B.A. and Stefan, H.G., 1993. Stream temperature dynamics: measurements and modeling. Water Resources Research, 29: 2299-2312.

SOPFEU, 2006. Société de protection des forêts contre le feu. http://www.sopfeu.qc.ca/html/francais/tour/stats_anneeCause.php, visited on October 10 2006.

St-Hilaire, A., Morin, G., El-Jabi, N. and Caissie, D., 2000. Water temperature modelling in a small forested stream: implication of forest canopy and soil temperature. Canadian Journal of Civil Engineering, 27: 1095-1108.

Story, A., Moore, R.D. and Macdonald, J.S., 2003. Stream temperatures in two shaded reaches below cutblocks and logging roads: downstream cooling linked to subsurface hydrology. Canadian Journal of Forest Research, 33: 1383-1396.

Swank, W.T., Vose, J.M. and Elliott, K.J., 2001. Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. Forest Ecology and Management, 143: 163-178.

Swift, L.W. and Messer, J.B., 1971. Forest cuttings raise temperatures of small streams in the southern Appalachians. Journal of Soil and Water Conservation, 26: 111-116.

Thomas, R.B. and Mehagan, W.F., 1998. Peaf flow responses to clear-cutting and roads in small and large basins, western Oregon: a second opinion. Water Resources Research, 34: 3393-3403.

Tremblay, Y., Rousseau, A.N., Plamondon, A.P. and Lévesque, D., 2007. Rainfall peakflow response of clearcutting 50% of three small watersheds in a boreal forest, Montmorency Forest, Québec. Not published.

Vitousek, P.M. et al., 1979. Nitrate losses from disturbed ecosystems. Science, 204: 469-474.

Welsch, D.L., Burns, D.A. and Murdoch, P.S., 2004. Processes affecting the response of sulfate concentrations to clearcutting in a northern hardwood forest, Catskill Mountains, New York, U.S.A. Biogeochemistry, 68: 337-354.

Wright, K.A., Sendek, K.H., Rice, R.M. and Thomas, R.B., 1990. Logging effects on streamflow: storm runoff at Caspar Creek in Northwestern California. Water Resources Research, 26: 1657-1667.

Yuzyk, T.R., Gummer, W.D. and Churchland, L.M., 1992. A comparison of methods used to measure suspended sediments in Canada's federal monitoring programs, Erosion and Sediment Tranport Monitoring Programmes in River Basins. IAHS, Proceedings of the Oslo Symposium, pp. 289-297.

Table 1
General changes in water quality parameters in response to forest harvesting

Water quality parameter	Stream temperature	pH	Specific conductance	Nitrates	Phosphates	Potassium	Magnesium	Iron	Sulphates	Suspended sediments
Direction of change	↑	↓	↑	↑	–	↑	↑	↑	↓	↑

Increase: ↑ ; decrease: ↓ ; no change: –

Table 2
Watersheds characteristics and treatments

	Watersheds					
Characteristics	0.2*	7.2	7.3	7.5	7.6	7.7
Watershed area (ha)	27.5	36	50	11.5	22	24
Average watershed slope (%)	12.6	13	11.7	9.7	11.2	11.2
Stream width at watershed outlet (m)	1.05	0.95	1	0.5	0.85	0.9
Average stream slope (%)	15	11	10.3	9.3	7.3	7.3
Treatments						
Proportion cut (%) within 50 m from the stream	0	70	50	20	40	40
Proportion cut (%) within 100 m from the stream	0	60	50	35	45	45
Proportion cut (%) within 150 m from the stream	0	55	50	40	55	55
Proportion cut (%) over the whole watershed	0	50	50	50	50	50

*Control watershed

Table 3

Warmest punctual days and 7-day periods according to the daily maximum stream temperatures on the control and treated watersheds during the pre-treatment (2003) and post-treatment summers (2004 and 2005)

Watershed	Year	Warmest punctual day	T (°C)	Warmest 7-day period	Mean T (°C)
0.2	2003	August 22	12.5	August 17 to 23	11.4
	2004	August 1	11.1	July 29 to August 4	10.8
	2005	July 19	12.8	July 17 to 23	12.0
7.2	2003	August 17	11.6	August 6 to 12	10.9
	2004	July 23	12.3	July 17 to 23	10.8
	2005	July 1	13.5	July 1 to 7	11.5
7.5	2003	August 22	12.9	August 17 to 23	11.9
	2004	July 23	12.3	July 29 to August 4	10.8
	2005	September 1	13.1	July 17 to 23	11.6
7.6	2003	August 22	11.1	August 17 to 23	10.5
	2004	July 23	10.5	July 29 to August 4	10.4
	2005	September 1	12.7	July 14 to 20	11.8

Table 4
Logging effects on the means of stream water quality parameters for the treated watersheds

Parameter	Year	Watershed 7.2		Watershed 7.3		Watershed 7.5		Watershed 7.6		Watershed 7.7	
		Effect	(%)	Effect	(%)	Effect	(%)	Effect	(%)	Effect	(%)
T-max	2004	0.0	0	-	-	** -0.4	-4	** 0.6	7	-	-
(°C)	2005	** -0.1	-1	-	-	** -0.3	-3	** 0.8	9	-	-
T-max > 10°	2004	** 1.0	10	-	-	** -0.4	-4	** 0.6	6	-	-
(°C)	2005	** 0.8	8	-	-	** -0.3	-2	** 0.9	9	-	-
T-max < 10°	2004	** -0.2	-3	-	-	** -0.3	-3	** 0.7	8	-	-
(°C)	2005	** -0.5	-5	-	-	** -0.4	-5	** 0.7	9	-	-
T-min	2004	0.1	1	-	-	0.0	0	** 0.8	10	-	-
(°C)	2005	** -0.1	-2	-	-	** 0.1	1	** 1.0	12	-	-
T-min > 8°	2004	* 0.5	6	-	-	** -0.1	-1	** 0.7	9	-	-
(°C)	2005	** 0.2	2	-	-	** 0.1	1	** 1.0	12	-	-
T-min < 8°	2004	** -0.2	-4	-	-	** 0.2	3	** 1.0	17	-	-
(°C)	2005	** -0.5	-7	-	-	0.1	1	** 0.9	15	-	-
T-diurnal var.	2004	0.0	1	-	-	** -0.3	-28	** -0.1	-15	-	-
(°C)	2005	0.0	3	-	-	** -0.5	-39	** -0.2	-21	-	-
pH	2004	** -0.4	-7	** -0.2	-4	* 0.2	3	** -0.3	-5	** -0.2	-4
	2005	** -0.2	-4	** -0.1	-1	* 0.1	2	** -0.2	-4	** -0.2	-4
Conduc.	2004	-0.5	-3	-2.3	-12	** 1.0	6	0.5	3	* 0.5	3
($\mu S \cdot cm^{-1}$)	2005	** 2.2	12	0.2	1	** 3.2	20	** 4.2	26	** 4.1	26
NO_3^-	2004	0.04	32	0.03	23	** 0.01	451	** 0.05	761	** 0.04	756
($mg \cdot l^{-1}$)	2005	** 0.37	282	** 0.31	258	** 0.06	2810	** 0.36	5727	** 0.35	6089
PO_4^{3-}	2004	** 0.004	20	0.002	8	-0.001	-5	-0.003	-9	0.002	9
($mg \cdot l^{-1}$)	2005	* 0.006	30	0.005	20	0.002	8	0.002	6	** 0.007	29
K^+	2004	** 0.34	125	** 0.27	93	* 0.11	45	* 0.20	91	** 0.14	53
($mg \cdot l^{-1}$)	2005	** 0.39	145	** 0.34	114	** 0.42	175	** 0.63	283	** 0.58	214
Mg^{2+}	2004	-0.009	-3	-0.022	-7	-0.009	-4	-0.046	-15	-0.026	-9
($mg \cdot l^{-1}$)	2005	** 0.067	19	* 0.044	13	0.004	2	** 0.022	7	* 0.039	14
Fe_{total}	2004	0.037	53	* 0.056	71	0.027	12	-0.024	-9	** 0.058	30
($mg \cdot l^{-1}$)	2005	-0.004	-6	0.018	23	** 0.122	55	** 0.066	23	** 0.139	71
SO_4^{2-}	2004	** 0.88	58	** 0.88	58	** 0.13	6	* 0.85	54	-0.08	-3
($mg \cdot l^{-1}$)	2005	** 0.82	54	** 0.83	54	** 0.07	3	0.73	46	-0.19	-7

* Statistically significant effect at p<0.10
** Statistically significant effect at p<0.05

Table 5
1st, 2nd and 3rd quartiles of stream water quality parameters for the control and treated watersheds during pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers

Parameter	Quartile	Treatment	Watershed					
			0.2	7.2	7.3	7.5	7.6	7.7
pH	1st (25%)	Pre-	5.0	6.1	5.8	4.9	5.1	5.1
		Post-	4.8	5.4	5.3	4.7	4.8	4.8
		Difference	-0.2	-0.7	-0.5	-0.2	-0.3	-0.3
	2nd (50%)	Pre-	5.2	6.3	5.9	4.9	5.5	5.6
		Post-	4.9	5.7	5.6	4.8	5.1	5.1
		Difference	-0.3	-0.6	-0.3	-0.1	-0.5	-0.5
	3rd (75%)	Pre-	5.3	6.5	6.2	4.9	5.8	6.0
		Post-	5.1	6.1	6.0	4.9	5.2	5.4
		Difference	-0.3	-0.5	-0.2	-0.1	-0.5	-0.6
Conduc. ($\mu S \cdot cm^{-1}$)	1st (25%)	Pre-	13.5	16.7	16.9	15.3	15.5	14.7
		Post-	14.6	19.1	18.8	17.6	18.0	17.5
		Difference	1.1	2.4	2.0	2.3	2.5	2.8
	2nd (50%)	Pre-	14.0	17.6	18.1	16.0	16.3	15.9
		Post-	15.5	20.7	20.0	18.7	19.2	18.8
		Difference	1.6	3.2	1.9	2.7	2.9	2.9
	3rd (75%)	Pre-	15.2	18.9	20.9	16.5	17.1	16.5
		Post-	16.6	21.9	21.0	20.8	21.4	21.3
		Difference	1.4	3.0	0.1	4.3	4.3	4.8
NO_3^- ($mg \cdot l^{-1}$)	1st (25%)	Pre-	0.00	0.10	0.09	0.00	0.01	0.00
		Post-	0.01	0.16	0.14	0.01	0.02	0.01
		Difference	0.01	0.05	0.05	0.01	0.01	0.01
	2nd (50%)	Pre-	0.01	0.14	0.11	0.00	0.01	0.01
		Post-	0.01	0.38	0.32	0.01	0.12	0.11
		Difference	0.00	0.24	0.21	0.01	0.11	0.11
	3rd (75%)	Pre-	0.01	0.18	0.18	0.00	0.08	0.01
		Post-	0.02	0.48	0.41	0.01	0.39	0.38
		Difference	0.01	0.30	0.23	0.01	0.31	0.37
PO_4^{3-} ($mg \cdot l^{-1}$)	1st (25%)	Pre-	0.019	0.012	0.012	0.018	0.027	0.012
		Post-	0.003	0.005	0.003	0.006	0.008	0.007
		Difference	-0.017	-0.007	-0.010	-0.012	-0.019	-0.005
	2nd (50%)	Pre-	0.023	0.017	0.023	0.032	0.032	0.023
		Post-	0.006	0.007	0.007	0.008	0.010	0.010
		Difference	-0.017	-0.010	-0.016	-0.024	-0.022	-0.013
	3rd (75%)	Pre-	0.032	0.034	0.034	0.034	0.035	0.035
		Post-	0.008	0.008	0.009	0.010	0.014	0.012
		Difference	-0.024	-0.026	-0.025	-0.024	-0.021	-0.023

Table 5 (cont'd)
1st, 2nd and 3rd quartiles of stream water quality parameters for the control and treated watersheds during pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers

Parameter	Quartile	Treatment	Watershed					
			0.2	7.2	7.3	7.5	7.6	7.7
K^+	1st (25%)	Pre-	0.13	0.21	0.25	0.18	0.20	0.15
($mg \cdot l^{-1}$)		Post-	0.08	0.44	0.41	0.17	0.32	0.30
		Difference	-0.05	0.23	0.16	-0.01	0.12	0.15
	2nd (50%)	Pre-	0.17	0.25	0.30	0.22	0.20	0.24
		Post-	0.10	0.53	0.50	0.42	0.61	0.58
		Difference	-0.07	0.28	0.20	0.20	0.41	0.34
	3rd (75%)	Pre-	0.21	0.34	0.33	0.27	0.21	0.29
		Post-	0.14	0.60	0.60	0.58	0.78	0.77
		Difference	-0.07	0.26	0.27	0.32	0.58	0.48
Mg^{2+}	1st (25%)	Pre-	0.20	0.31	0.30	0.19	0.30	0.26
($mg \cdot l^{-1}$)		Post-	0.20	0.30	0.29	0.19	0.22	0.23
		Difference	0.00	-0.01	-0.01	0.00	-0.07	-0.03
	2nd (50%)	Pre-	0.22	0.34	0.32	0.23	0.30	0.29
		Post-	0.21	0.35	0.32	0.21	0.27	0.27
		Difference	-0.01	0.01	0.00	-0.03	-0.03	-0.02
	3rd (75%)	Pre-	0.25	0.38	0.40	0.26	0.31	0.32
		Post-	0.22	0.41	0.38	0.22	0.30	0.31
		Difference	-0.02	0.04	-0.02	-0.04	0.00	-0.01
Fe_{total}	1st (25%)	Pre-	0.17	0.04	0.05	0.15	0.18	0.12
($mg \cdot l^{-1}$)		Post-	0.22	0.08	0.08	0.24	0.25	0.25
		Difference	0.06	0.04	0.03	0.10	0.07	0.13
	2nd (50%)	Pre-	0.24	0.07	0.06	0.18	0.20	0.15
		Post-	0.29	0.11	0.14	0.31	0.34	0.32
		Difference	0.05	0.04	0.08	0.13	0.14	0.17
	3rd (75%)	Pre-	0.29	0.08	0.07	0.25	0.30	0.20
		Post-	0.37	0.20	0.23	0.40	0.44	0.43
		Difference	0.08	0.12	0.16	0.16	0.15	0.23
SO_4^{2-}	1st (25%)	Pre-	1.73	1.15	1.00	2.20	1.00	2.43
($mg \cdot l^{-1}$)		Post-	0.78	0.82	0.83	0.93	0.79	0.78
		Difference	-0.95	-0.33	-0.17	-1.28	-0.21	-1.65
	2nd (50%)	Pre-	2.59	1.28	1.55	2.45	1.00	2.70
		Post-	0.84	0.86	0.87	0.98	0.85	0.87
		Difference	-1.75	-0.42	-0.68	-1.46	-0.15	-1.83
	3rd (75%)	Pre-	3.10	1.65	1.61	2.48	2.30	2.92
		Post-	0.88	0.92	0.93	1.04	0.96	0.96
		Difference	-2.21	-0.72	-0.68	-1.43	-1.33	-1.96

Table 6

Non-exceedance probabilities and number of times the suspended sediment concentration (mg•l^{-1}) criteria for the protection of aquatic life (MDDEP, 2006) was exceeded on the control and treated watersheds during post-treatment summers only (2004-2005)

Watershed	Chronic effect (> 6 mg•l-1)		Acute toxicity (> 26 mg•l-1)	
	Nbr of times	Non-exceedance probabiliy	Nbr of times	Non-exceedance probabiliy
0.2	0	1	0	1
7.2	8	0.91	0	1
7.3	5	0.94	0	1
7.5	4	0.94	0	1
7.6	4	0.95	1	0.99
7.7	6	0.93	0	1

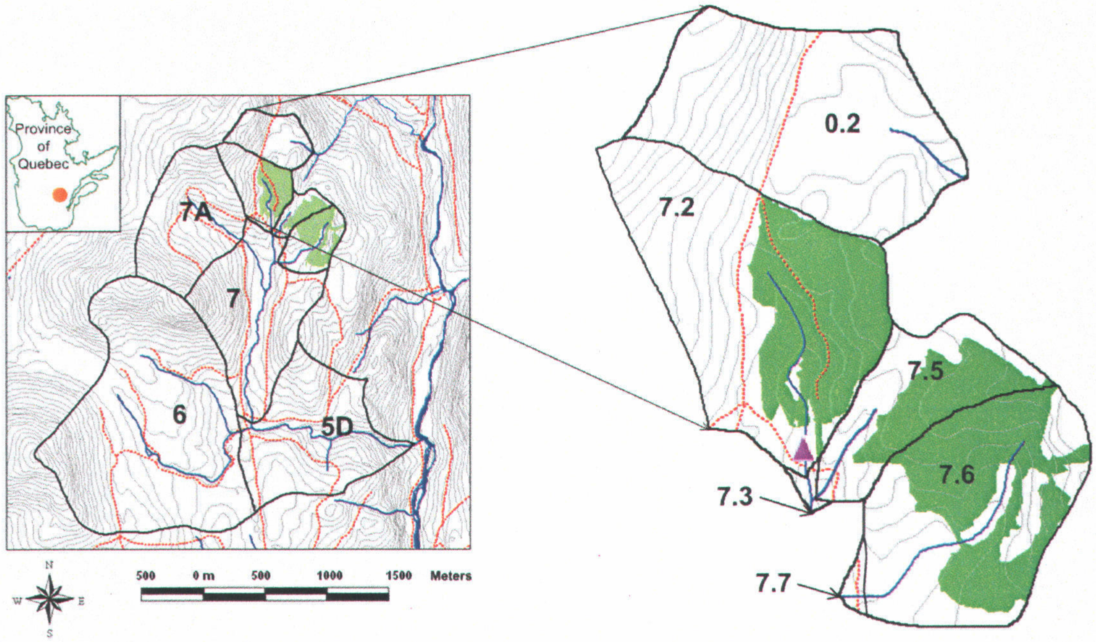

Figure 1. Control (0.2) and treated watersheds (7.2, 7.3, 7.5, 7.6 and 7.7) within the Ruisseau des Eaux-Volées Experimental Watershed (REVEW), Quebec, with clearcut areas ( ), streams (——), forest roads (········) and 10-m elevation contours (——).

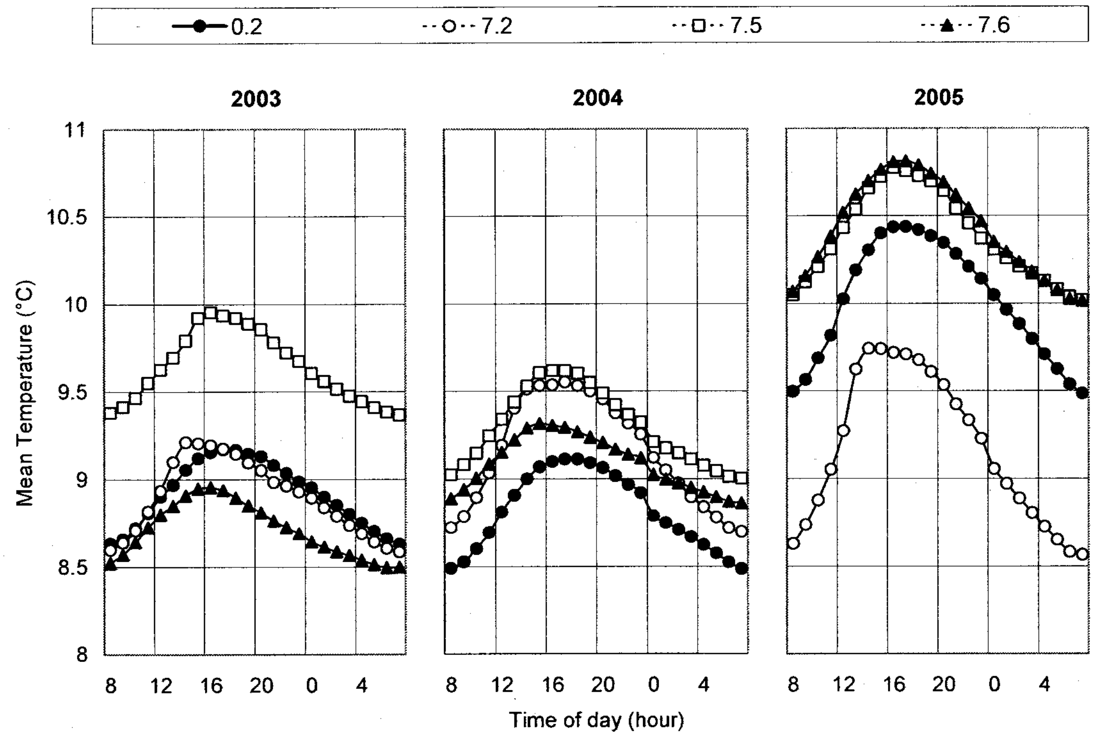

Figure 2. Mean hourly stream temperature (°C) for the control and treated watersheds during pre-treatment (2003) and post-treatment (2004-2005) summers.

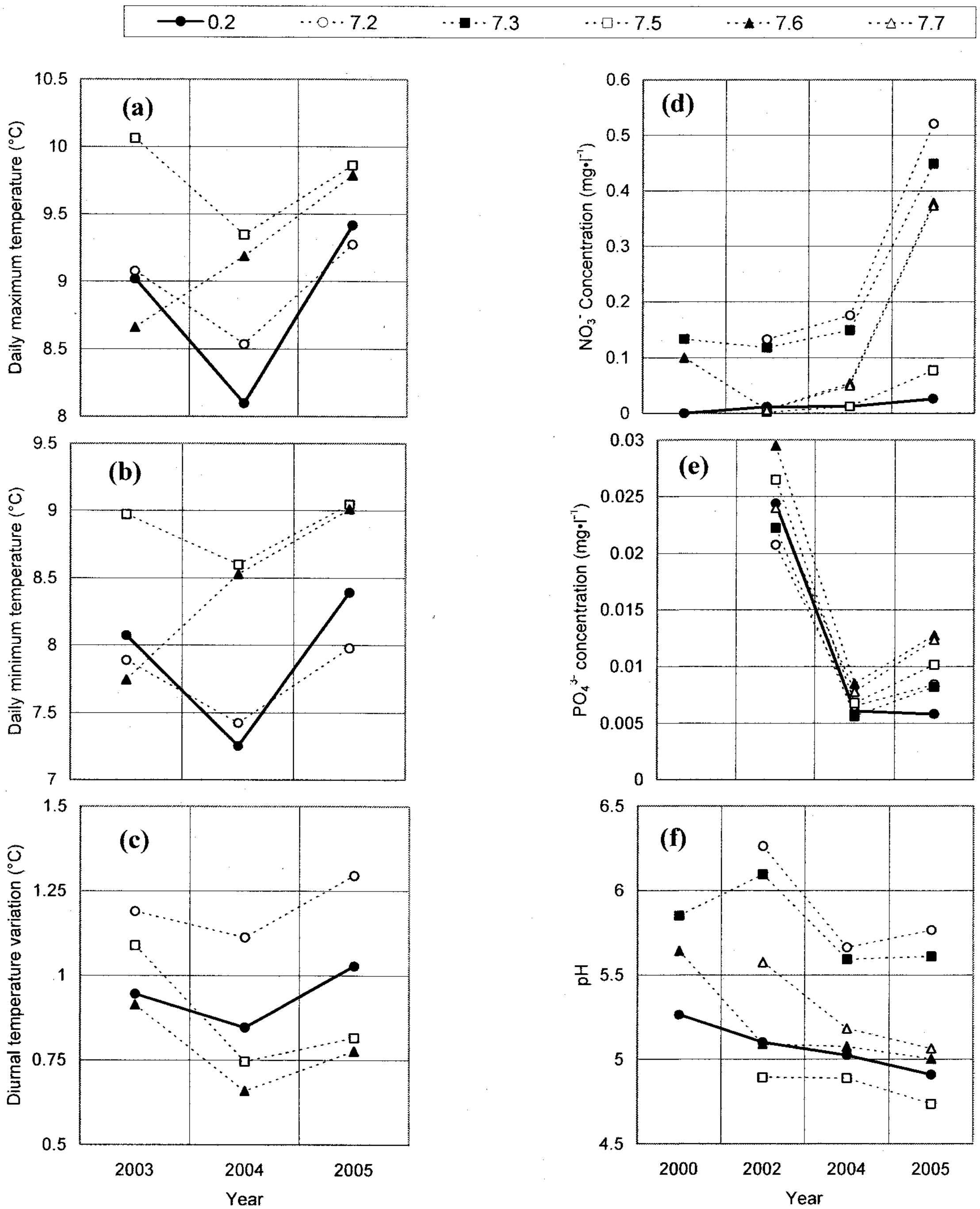

Figure 3. Mean water quality parameters for the control and treated watersheds during pre-treatment (2000-2003) and post-treatment (2004-2005) summers: (a) daily maximum temperature, (b) daily minimum temperature, (c) diurnal temperature variation, (d) NO_3^- concentration, (e) PO_4^{3-} concentration, (f) pH. (For clarity, error bars are not shown).

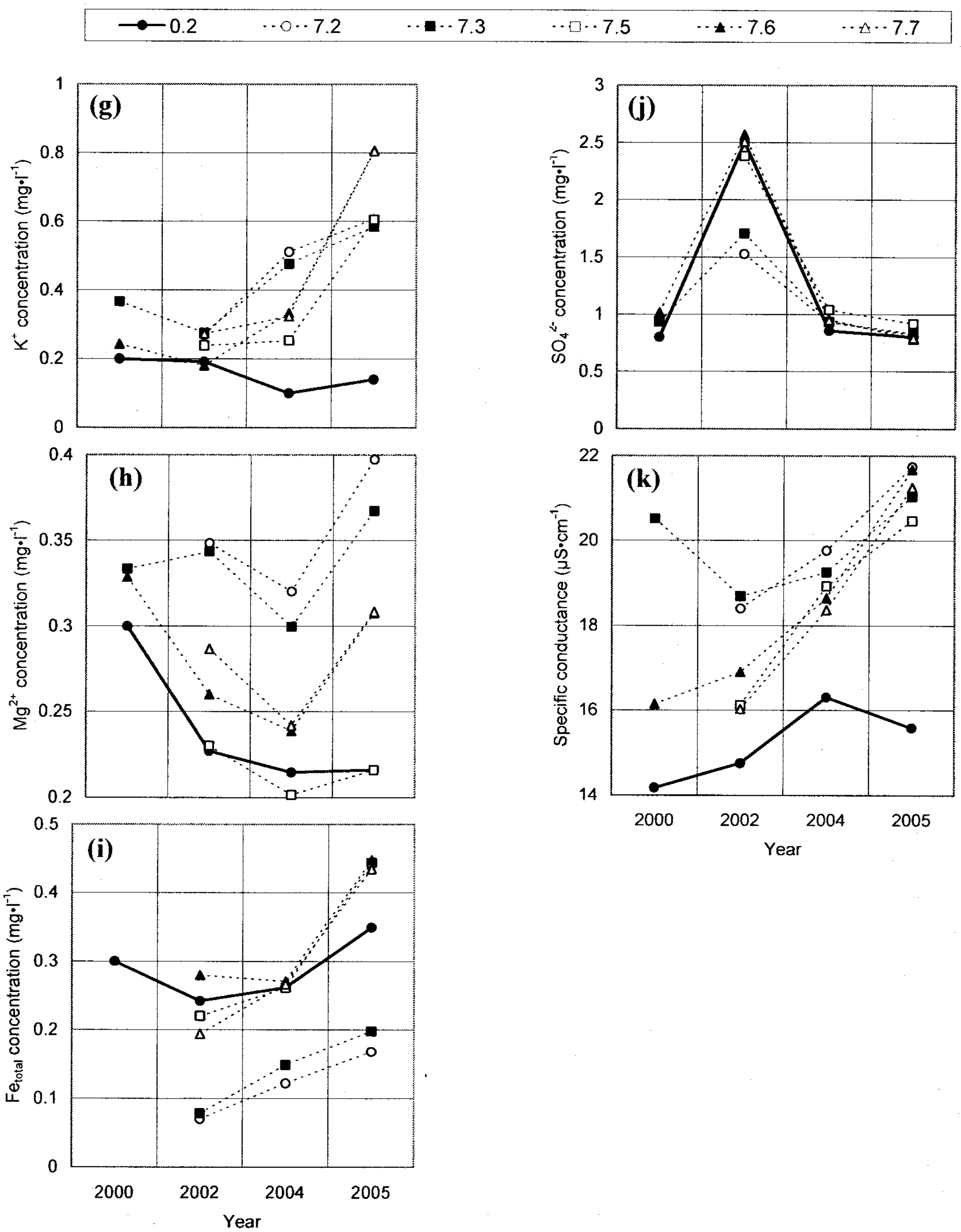

Figure 3 (cont'd). Mean water quality parameters for the control and treated watersheds during pre-treatment (2000-2003) and post-treatment (2004-2005) summers: (g) K^+ concentration, (h) Mg^{2+} concentration, (i) Fe_{total} concentration, (j) SO_4^{2-} concentration, and (k) specific conductance. (For clarity, error bars are not shown).

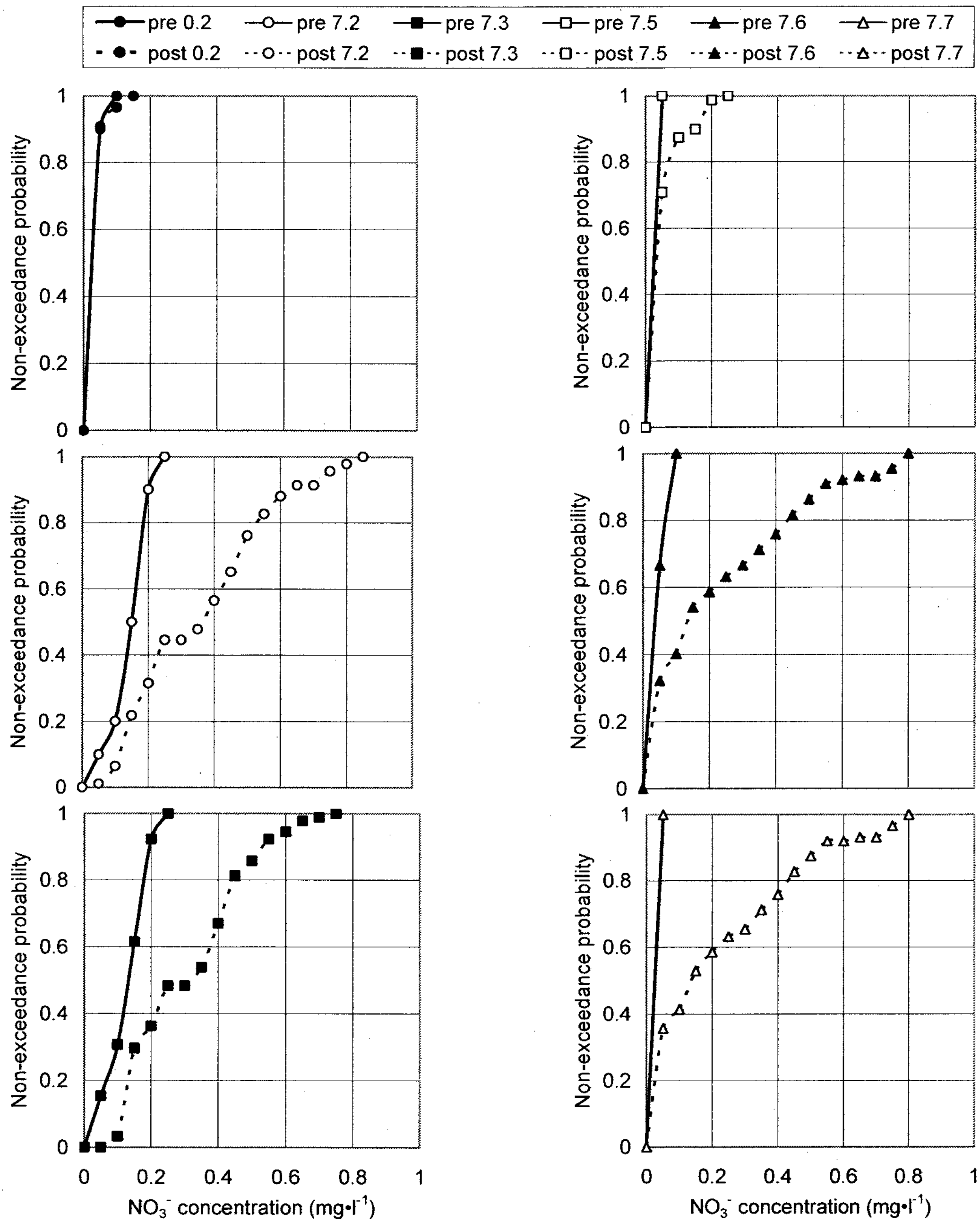

Figure 4. Non-exceedance probability for stream water NO_3^- concentration ($mg \cdot l^{-1}$) of the control and treated watersheds during pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers.

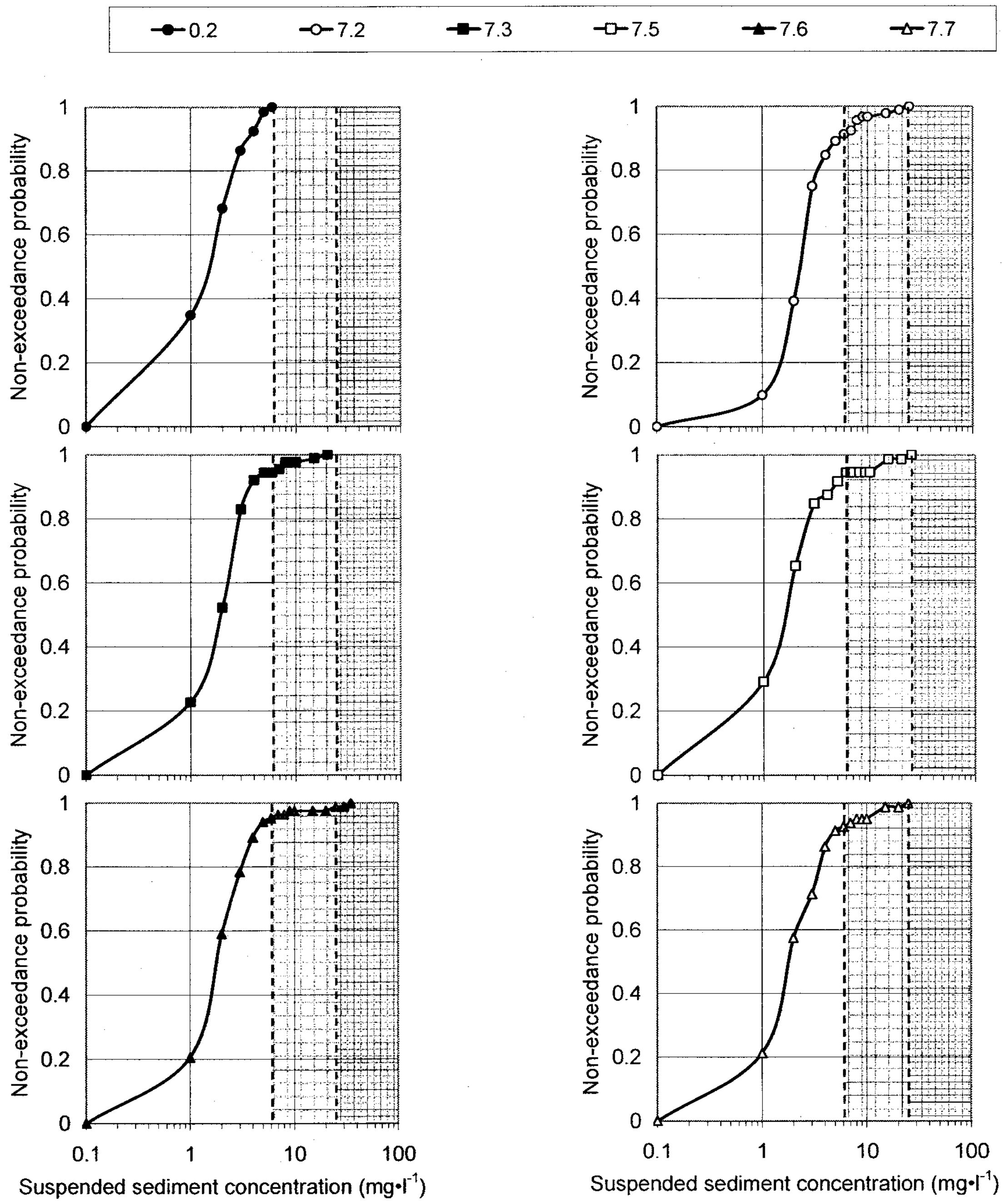

Figure 5. Non-exceedance probability for suspended sediment concentration (mg•l^{-1}) of the control and treated watersheds during post-treatment summers (2004-2005) and criteria for the protection of aquatic life (cronic effect ▦ acute toxicity ⊞). (For example, 20% of the samples did not exceed 1 mg•l^{-1} on watershed 7.6).

SECTION 3

Annexes

Tableau A.1. Moyennes des températures journalières maximales (°C) prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily maximum stream water temperature (°C) of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	8.5	8.5	0.0	0	0.000	0.669	0.795
	2005	9.3	9.3	-0.1	-1	0.000	0.527	0.059
7.5	2004	9.7	9.3	-0.4	-4	0.000	0.000	0.000
	2005	10.2	9.9	-0.3	-3	0.000	0.000	0.000
7.6	2004	8.5	9.2	0.6	7	0.293	0.000	0.000
	2005	9.0	9.8	0.8	9	0.441	0.000	0.000

Tableau A.2. Moyennes des températures journalières maximales de plus de 10°C prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily maximum stream water temperature larger than 10°C of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	9.7	10.7	1.0	10	0.769	0.000	0.000
	2005	10.3	11.1	0.8	8	0.089	0.000	0.000
7.5	2004	10.9	10.5	-0.4	-4	0.580	0.000	0.000
	2005	11.3	11.0	-0.3	-2	0.358	0.000	0.000
7.6	2004	9.9	10.5	0.6	6	0.897	0.000	0.001
	2005	10.1	11.0	0.9	9	0.784	0.000	0.000

Tableau A.3. Moyennes des températures journalières maximales de moins de 10°C prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily maximum stream water temperature smaller than 10°C of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	8.2	7.9	-0.2	-3	0.002	0.009	0.006
	2005	8.9	8.4	-0.5	-5	0.114	0.000	0.000
7.5	2004	9.2	8.8	-0.3	-3	0.465	0.000	0.000
	2005	8.7	8.3	-0.4	-5	0.737	0.000	0.000
7.6	2004	8.3	8.9	0.7	8	0.225	0.000	0.000
	2005	7.7	8.4	0.7	9	0.768	0.000	0.000

Tableau A.4. Moyennes des températures journalières minimales (°C) prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily minimum stream water temperature (°C) of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	7.3	7.4	0.1	1	0.007	0.105	0.139
	2005	8.1	8.0	-0.1	-2	0.068	0.049	0.029
7.5	2004	8.6	8.6	0.0	0	0.007	0.716	0.453
	2005	9.0	9.0	0.1	1	0.068	0.017	0.012
7.6	2004	7.8	8.5	0.8	10	0.152	0.000	0.000
	2005	8.0	9.0	1.0	12	0.783	0.000	0.000

Tableau A.5. Moyennes des températures journalières minimales de plus de 8°C prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily minimum stream water temperature larger than 8°C of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	8.4	8.9	0.5	6	0.932	0.000	0.000
	2005	8.9	9.1	0.2	2	0.255	0.093	0.112
7.5	2004	9.3	9.2	-0.1	-1	0.039	0.005	0.002
	2005	9.6	9.7	0.1	1	0.036	0.035	0.039
7.6	2004	8.3	9.0	0.7	9	0.006	0.000	0.000
	2005	8.7	9.7	1.0	12	0.722	0.000	0.000

Tableau A.6. Moyennes des températures journalières minimales de moins de 8°C prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean daily minimum stream water temperature smaller than 8°C of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	6.3	6.0	-0.2	-4	0.085	0.006	0.016
	2005	7.2	6.8	-0.5	-7	0.817	0.000	0.000
7.5	2004	6.9	7.1	0.2	3	0.431	0.001	0.002
	2005	6.7	6.8	0.1	1	0.802	0.210	0.087
7.6	2004	5.7	6.7	1.0	17	0.568	0.000	0.000
	2005	5.8	6.7	0.9	15	0.844	0.000	0.000

Tableau A.7. Moyennes des variations diurnes (°C) prédites et mesurées durant les étés d'après coupe (2004-2005) pour les bassins versants traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, du t pairé et des rangs signés de Wilcoxon.

Predicted and measured mean diurnal stream water temperature variation (°C) of the treated watersheds for the post-treatment (2004-2005) summers, logging effects and p-values according to the Fisher-Snedecor test, the paired t-test and the Wilcoxon signed-rank test

Watershed	Year	Predicted	Measured	Effect	(%)	p-Fischer	p-Student	p-Wilcoxon
7.2	2004	1.1	1.1	0.0	1	0.027	0.795	0.542
	2005	1.3	1.3	0.0	3	0.000	0.489	0.142
7.5	2004	1.0	0.7	-0.3	-28	0.027	0.000	0.000
	2005	1.3	0.8	-0.5	-39	0.000	0.000	0.000
7.6	2004	0.8	0.7	-0.1	-15	0.039	0.002	0.000
	2005	1.0	0.8	-0.2	-21	0.224	0.000	0.000

Tableau A.8. Moyennes des concentrations en NO_3^- (mg•l^{-1}) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water NO_3^- concentrations (mgl•$^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect (%)		*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	0.01	0.13	0.12					
	2004	0.01	0.18	0.16	0.04	32	0.164	0.177	0.383
	2005	0.03	0.52	0.49	0.37	282	0.018	0.000	0.000
7.3	2000 and 2002	0.01	0.12	0.11					
	2004	0.01	0.15	0.14	0.03	23	0.478	0.267	0.459
	2005	0.03	0.45	0.42	0.31	258	0.065	0.000	0.000
7.5	2000 and 2002	0.01	0.00	-0.01					
	2004	0.01	0.01	0.00	0.01	451	0.072	0.012	0.000
	2005	0.03	0.08	0.05	0.06	2810	0.000	0.000	0.008
7.6	2000 and 2002	0.01	0.01	-0.01					
	2004	0.01	0.05	0.04	0.05	761	0.000	0.008	0.001
	2005	0.03	0.38	0.35	0.36	5727	0.000	0.000	0.001
7.7	2000 and 2002	0.01	0.01	-0.01					
	2004	0.01	0.05	0.04	0.04	756	0.000	0.011	0.001
	2005	0.03	0.37	0.35	0.35	6089	0.000	0.000	0.000

Tableau A.9. Moyennes des concentrations en PO_4^{3-} ($mg \bullet l^{-1}$) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water PO_4^{3-} concentrations ($mgl \bullet^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	0.024	0.021	-0.004					
	2004	0.006	0.007	0.000	0.004	20	0.000	0.369	0.049
	2005	0.006	0.008	0.003	0.006	30	0.006	0.226	0.061
7.3	2000 and 2002	0.024	0.022	-0.002					
	2004	0.006	0.006	0.000	0.002	8	0.000	0.660	0.648
	2005	0.006	0.008	0.002	0.005	20	0.000	0.341	0.228
7.5	2000 and 2002	0.024	0.027	0.002					
	2004	0.006	0.007	0.001	-0.001	-5	0.000	0.523	0.577
	2005	0.006	0.010	0.004	0.002	8	0.000	0.267	0.297
7.6	2000 and 2002	0.024	0.030	0.005					
	2004	0.006	0.008	0.002	-0.003	-9	0.000	0.454	0.130
	2005	0.006	0.013	0.007	0.002	6	0.000	0.299	0.116
7.7	2000 and 2002	0.024	0.024	0.000					
	2004	0.006	0.008	0.002	0.002	9	0.000	0.667	0.627
	2005	0.006	0.012	0.007	0.007	29	0.000	0.184	0.009

Tableau A.10. Moyennes du pH durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water pH of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	*p*-Wilcoxon
7.2	2000 and 2002	5.2	6.3	1.1					
	2004	5.0	5.7	0.6	-0.4	-7	0.031	0.000	0.001
	2005	4.9	5.8	0.9	-0.2	-4	0.304	0.007	0.010
7.3	2000 and 2002	5.2	6.0	0.8					
	2004	5.0	5.6	0.6	-0.2	-4	0.099	0.001	0.004
	2005	4.9	5.6	0.7	-0.1	-1	0.302	0.012	0.023
7.5	2000 and 2002	5.2	4.9	-0.3					
	2004	5.0	4.9	-0.1	0.2	3	0.000	0.415	0.097
	2005	4.9	4.7	-0.2	0.1	2	0.865	0.077	0.065
7.6	2000 and 2002	5.2	5.5	0.3					
	2004	5.0	5.1	0.0	-0.3	-5	0.178	0.049	0.106
	2005	4.9	5.0	0.1	-0.2	-4	0.132	0.044	0.018
7.7	2000 and 2002	5.2	5.6	0.4					
	2004	5.0	5.2	0.2	-0.2	-4	0.569	0.050	0.066
	2005	4.9	5.1	0.2	-0.2	-4	0.009	0.025	0.007

Tableau A.11. Moyennes des concentrations en K^+ ($mg \bullet l^{-1}$) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water K^+ concentrations ($mgl \bullet^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	0.19	0.27	0.08					
	2004	0.10	0.51	0.41	0.34	125	0.116	0.000	0.000
	2005	0.14	0.61	0.47	0.39	145	0.128	0.000	0.000
7.3	2000 and 2002	0.19	0.30	0.10					
	2004	0.10	0.48	0.38	0.27	92	0.008	0.000	0.000
	2005	0.14	0.58	0.44	0.34	114	0.090	0.000	0.000
7.5	2000 and 2002	0.19	0.24	0.05					
	2004	0.10	0.25	0.15	0.11	45	0.361	0.076	0.161
	2005	0.14	0.60	0.46	0.42	175	0.275	0.000	0.000
7.6	2000 and 2002	0.19	0.22	0.03					
	2004	0.10	0.33	0.23	0.20	91	0.940	0.074	0.012
	2005	0.14	0.81	0.67	0.63	283	0.891	0.010	0.004
7.7	2000 and 2002	0.19	0.27	0.08					
	2004	0.10	0.32	0.23	0.14	53	0.505	0.045	0.006
	2005	0.14	0.80	0.66	0.58	214	0.441	0.000	0.000

Tableau A.12. Moyennes des concentrations en Mg^{2+} ($mg \bullet l^{-1}$) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water Mg^{2+} concentrations ($mgl \bullet^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	0.23	0.35	0.11					
	2004	0.21	0.32	0.11	-0.009	-3	0.004	0.626	0.394
	2005	0.22	0.40	0.18	0.067	19	0.004	0.086	0.048
7.3	2000 and 2002	0.23	0.34	0.11					
	2004	0.21	0.30	0.09	-0.022	-7	0.006	0.351	0.352
	2005	0.22	0.37	0.15	0.044	13	0.018	0.101	0.060
7.5	2000 and 2002	0.23	0.23	0.00					
	2004	0.21	0.20	-0.01	-0.009	-4	0.000	0.347	0.861
	2005	0.22	0.22	0.00	0.004	2	0.068	0.809	0.506
7.6	2000 and 2002	0.23	0.30	0.07					
	2004	0.21	0.24	0.02	-0.046	-15	0.012	0.930	0.900
	2005	0.22	0.31	0.09	0.022	7	0.023	0.083	0.008
7.7	2000 and 2002	0.23	0.29	0.05					
	2004	0.21	0.24	0.03	-0.026	-9	0.000	0.127	0.106
	2005	0.22	0.31	0.09	0.039	14	0.000	0.125	0.086

Tableau A.13. Moyennes des concentrations en Fe_{total} ($mg \bullet l^{-1}$) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water Fe_{total} concentrations ($mgl \bullet^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	0.25	0.07	-0.18					
	2004	0.26	0.12	-0.14	0.037	53	0.259	0.340	0.427
	2005	0.35	0.17	-0.18	-0.004	-6	0.225	0.704	0.651
7.3	2000 and 2002	0.25	0.08	-0.17					
	2004	0.26	0.15	-0.11	0.056	71	0.217	0.040	0.059
	2005	0.35	0.20	-0.15	0.018	23	0.007	0.788	0.903
7.5	2000 and 2002	0.25	0.22	-0.03					
	2004	0.26	0.26	0.00	0.027	12	0.511	0.151	0.127
	2005	0.35	0.44	0.09	0.122	55	0.329	0.001	0.000
7.6	2000 and 2002	0.25	0.28	0.03					
	2004	0.26	0.27	0.01	-0.024	-9	0.939	0.657	0.642
	2005	0.35	0.45	0.10	0.066	23	0.842	0.040	0.013
7.7	2000 and 2002	0.25	0.19	-0.05					
	2004	0.26	0.27	0.00	0.058	30	0.618	0.043	0.041
	2005	0.35	0.43	0.09	0.139	71	0.360	0.000	0.000

Tableau A.14. Moyennes des concentrations en SO_4^{2-} ($mg \bullet l^{-1}$) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water SO_4^{2-} concentrations ($mgl \bullet ^{-1}$) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	p-Wilcoxon
7.2	2000 and 2002	2.33	1.53	-0.81					
	2004	0.86	0.93	0.07	0.88	58	0.000	0.019	0.005
	2005	0.80	0.82	0.02	0.82	54	0.000	0.023	0.013
7.3	2000 and 2002	2.33	1.53	-0.80					
	2004	0.86	0.94	0.08	0.88	58	0.000	0.016	0.013
	2005	0.80	0.83	0.03	0.83	54	0.000	0.022	0.025
7.5	2000 and 2002	2.33	2.38	0.05					
	2004	0.86	1.04	0.18	0.13	6	0.000	0.047	0.022
	2005	0.80	0.92	0.12	0.07	3	0.000	0.068	0.033
7.6	2000 and 2002	2.33	1.58	-0.75					
	2004	0.86	0.96	0.10	0.85	54	0.000	0.181	0.096
	2005	0.80	0.78	-0.02	0.73	46	0.000	0.405	0.205
7.7	2000 and 2002	2.33	2.51	0.18					
	2004	0.86	0.95	0.09	-0.08	-3	0.000	0.672	0.554
	2005	0.80	0.79	-0.01	-0.19	-7	0.000	0.832	0.155

Tableau A.15. Moyennes de la conductance spécifique (μS•cm^{-1}) durant les étés d'avant (2000 et 2002) et d'après coupe (2004-2005) pour les bassins versants témoin et traités, les effets de la coupe forestière et les p-valeurs selon les tests de Fisher-Snedecor, de t et de Wilcoxon-Mann-Whitney.

Mean stream water specific conductance (μS•cm^{-1}) of the control and treated watersheds for the pre-treatment (2000 and 2002) and post-treatment (2004-2005) summers, logging effects and *p*-values according to the Fisher-Snedecor test, the *t*-test and the Wilcoxon-Mann-Whitney test

Watershed	Year	Control	Treated	Difference	Effect	(%)	*p*-Fisher	*p*-Student	*p*-Wilcoxon
7.2	2000 and 2002	14.5	18.4	3.9					
	2004	16.3	19.8	3.5	-0.5	-3	0.081	0.843	0.900
	2005	15.6	21.7	6.2	2.2	12	0.001	0.165	0.038
7.3	2000 and 2002	14.5	19.7	5.2					
	2004	16.3	19.3	3.0	-2.3	-12	0.000	0.144	0.132
	2005	15.6	21.0	5.5	0.2	1	0.000	0.961	0.393
7.5	2000 and 2002	14.5	16.1	1.7					
	2004	16.3	18.9	2.6	1.0	6	0.073	0.002	0.005
	2005	15.6	20.5	4.9	3.2	20	0.174	0.000	0.000
7.6	2000 and 2002	14.5	16.3	1.9					
	2004	16.3	18.6	2.3	0.5	3	0.063	0.243	0.164
	2005	15.6	21.7	6.1	4.2	26	0.292	0.000	0.000
7.7	2000 and 2002	14.5	16.0	1.6					
	2004	16.3	18.4	2.1	0.5	3	0.152	0.090	0.064
	2005	15.6	21.3	5.7	4.1	26	0.160	0.000	0.000