Forestry Impacts on Fish Habitat in the Northern Interior of British Columbia: A Compendium of Research from the Stuart-Takla Fish-Forestry Interaction Study

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ABSTRACT

MacIsaac, E.A. (ed.). 2003. Forestry impacts on fish habitat in the northern interior of British Columbia: A compendium of research from the Stuart-Takla Fish-Forestry Interaction Study. Can. Tech. Rep. Fish. Aquat. Sci. 2509: v + 266p.

Canadian federal and provincial government researchers and resource managers, in partnership with university researchers, the forest industry and local First Nations, initiated the Stuart-Takla Fish-Forestry Interaction Study in 1990. The study was designed as a long-term, multidisciplinary project with the goal of improving our knowledge of the effects of forest harvest activities on stream ecosystems and fish habitat in the north and central interior forests of British Columbia. Watersheds tributary to Takla Lake and Middle River were selected as the research sites because of their importance as sockeye salmon (*Oncorhynchus nerka*) spawning streams and the plans for future forest harvesting in some of the watersheds. Small watersheds were also included to investigate the effects of variable-retention riparian management and road crossings on small headwater streams. This compendium contains 17 papers that describe completed and on-going research and data collection, and it provides a record of the wide range of research activities conducted in the Stuart-Takla study watersheds.

RESUME

MacIsaac, E.A. (ed.). 2003. L'exploitation des forêts influe sur l'habitat de poisson dans l'intérieur septentrional de Colombie Britannique: UN abrégé de recherche de l'Etude d'Interaction de Poisson-Exploitation Des Forêts de Stuart-Takla. Can. Tech. Rep. Fish. Aquat. Sci. 2509: v + 266p.

Les chercheurs de gouvernement et les directeurs de ressource canadiens fédéraux et provinciaux, dans le partenariat avec les chercheurs d'université, l'industrie de forêt et locales les Premières Nations, inauguré l'Etude d'Interaction de Poisson-Exploitation Des Forêts de Stuart-Takla dans 1990. L'étude a été conçue comme un à long terme, le projet de multidisciplinary avec le but d'améliorer notre connaissance des effets d'activités de moisson de forêt sur les écosystèmes de ruisseau et l'habitat de poisson dans le nord et les forêts d'intérieur centrales de Colombie Britannique. Tributary de bassins au Lac de Takla et à la Rivière de Milieu a été choisi comme les sites de recherche à cause de leur importance comme leur saumon de sockeye (Oncorhynchus nerka) frayant des ruisseaux et les projets pour moissonner de forêt futur dans une partie des tournants. Les petits bassins ont été aussi inclus pour examiner les effets de direction de riparian de variable-rétention et les croisements de route sur les petits ruisseaux de source. Cet abrégé contient 17 papiers qui décrivent la recherche complété et de sur-aller et la réception de données, et il fournit un dossier de la gamme large d'activités de recherche dirigées dans les bassins d'étude de Stuart-Takla.

An Overview of the Stuart-Takla Fish-Forestry Interaction Project

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The Stuart-Takla Fish-Forestry Interaction Project (STFFIP) was born in 1990 from the recognition that our knowledge of the interactions between forest harvesting and the productive capacities of aquatic habitats was very limited for the interior forests of British Columbia (Macdonald et al 1992). At the time, forestry guidelines and regulations for the protection of fish in British Columbia were largely based on fish-forestry research projects conducted in coastal watersheds such as the Carnation Creek project (Hartman and Scrivener 1990) and others conducted in the coastal Pacific Northwest (Poulin 1984, Naiman et al 2000). This research significantly improved both our scientific knowledge and our management capability for mitigating the effects of logging activities on aquatic habitats. However, there are significant differences in climate, hydrology, geology, vegetation and aquatic species between coastal and interior watersheds that could affect the functional relationships between forestry and fisheries and their responses to logging disturbances. This uncertainty about the applicability of coastal forest management practices to interior forests was one of the primary driving forces behind the creation of the STFFIP.

The STFFIP was sited in the Stuart-Takla drainage in the northernmost watersheds of the Fraser River basin. Initially, four streams and their watersheds along Takla Lake and the Middle River were selected for the study (Bivouac, Forfar, Gluskie and O'Ne-ell creeks) because of their proximity to each other and similar physiographic characteristics. Later, Van Decar and Baptiste creeks and their watersheds were added to meet specific research objectives. For example, the Baptiste Creek upper watershed is an important site where studies of riparian management on small headwater streams are being conducted.

The Stuart-Takla streams are also the spawning grounds of the Early Stuart sockeye salmon (*Oncorhynchus nerka*), the earliest and longest migrating sockeye run in the Fraser River system. They are of great economic and cultural importance to the First Nations of the Fraser River and they are an important early sockeye run harvested by commercial ocean fisheries. The Stuart-Takla drainages were also areas of active forest development and harvesting, and there were concerns about the impacts of logging activities on these important sockeye salmon stocks. The STFFIP study site was thus an ideal location to improve our knowledge of the interactions between forestry and fish in BC interior watersheds.

From its inception, the STFFIP was designed to be a long-term, multidisciplinary study of the effects of current forest harvesting practices on the ecology of interior BC salmon and stream ecosystems. The site selection criteria, experimental and harvesting design, and the various research projects and project leaders of the STFFIP in the first 6 years of the project are described in detail by Macdonald and Herunter (1998). Two of the watersheds would be subject to forest harvesting (Gluskie and O'Ne-ell) and Forfar would remain an unharvested control for the duration of the project. The early years of the project were devoted to collecting baseline and pre-harvest data on the salmon, creeks and watersheds and studying natural physical, chemical and biological processes thought to be sensitive to forest harvesting effects. The STFFIP was designed to use spatial and temporal controls to separate forest harvesting effects from natural variations. These components of the study design and the early data collected are discussed in the proceedings of two workshops held early in the life of the project (Bernard et al 1994, Macdonald 1994).

To satisfy its multidisciplinary design, the STFFIP involved numerous researchers from various organizations including the Canadian Department of Fisheries and Oceans (DFO), BC Ministry of Forests, BC Ministry of Environment, Land and Parks, Environment Canada, Canadian Wildlife Service, University of North British Columbia, University of British Columbia and Simon Fraser University (Macdonald and Herunter 1998). DFO was the lead agency and Dr. Steve Macdonald provided overall liaison and coordination from the beginning of the project. DFO also provided field logistical support for many of the other researchers including some of the routine field sampling. The DFO Middle River camp, established and operated by Stock Assessment staff of DFO for the enumeration of Early and Late Stuart sockeye salmon escapements, was an important base of operations for many of the research activities that otherwise would have been difficult to complete in such a remote area of BC. DFO also established and maintained stream gauging stations on all of the main creeks and operated a weather station at the DFO camp.

The early involvement and collaboration of a forest industry partner was required for the success of the project to ensure harvesting and other forestry activities met the experimental designs and schedules of the investigators while reflecting current harvesting methods and forest management practices. Canadian Forest Products Limited (CanFor) was the forest licensee for the study watersheds and shared their harvesting plans and schedules, modifying them as required to meet specific research objectives. CanFor also provided invaluable project signage on the study streams, maps and geographic data, winter and summer road maintenance, and logistical support for field tours.

The early involvement of the local First Nations bands in the STFFIP was also an important element of the project. The study watersheds are part of the traditional territories of the Tl'azt'en Nation. Tl'azt'en band members, as well as members from other First Nations bands of the Carrier Sekani Tribal Council, were involved in many areas of the research. Many were employed in the DFO Stock Assessment enumeration programs for the Early Stuart sockeye salmon escapements and fry emigration on the study streams, projects which could not have been completed without their involvement. Thomas Alexis, a Tl'azt'en fisheries intern, was a full member of the DFO research team that provided field sampling and baseline data collection for many of the research projects.

This compendium provides a sampling of some of the STFFIP research based on papers given at a STFFIP workshop in Vancouver in 2000. Some projects are completed or ongoing while others are in the early stages of development. Not all research projects and types of data collected are described in this volume because the data hasn't been completely collected or analyzed or because publication in this compendium might jeopardize their ability to publish the data in a primary scientific journal. This compendium should only be considered a starting point for any future collations of the research conducted in the STFFIP. It is divided into three parts:

1. **Watershed, Stream Habitat and Salmon Studies**: papers describing some of the hydrologic, salmon and stream productivity research from the study watersheds and main-stem STFFIP creeks and watersheds
2. **Sediment and Bedload Studies**: papers describing research into the physical stream bed and sediment characteristics of the main-stem creeks
3. **Small Stream Studies**: papers describing research conducted on small fish-bearing and headwater streams in the STFFIP watersheds

Ultimately, the majority of the research conducted by the various investigators involved in the STFFIP is destined for publication in the primary and secondary scientific literature and readers are encouraged to search in the future for other publications by the authors in this volume. For example, some of the early research has been published in the proceedings of the Forest-Fish Conference: Land Management Practices Affecting Aquatic Ecosystems sponsored by the Canadian Forest Service in Calgary, Alberta in 1996. Macdonald and Herunter (1998) describe the design of the STFFIP while Heinonen (1998) and Beaudry (1998) discuss early results from hydrology and suspended sediment studies conducted in the watersheds. Petticrew (1998) looked at fine-grained sediment storage in the sockeye spawning streams while Gottesfeld and Mitchell (1998) illustrated bedload transport by sockeye redd digging and flood events. Scrivener and Macdonald (1998) discussed the intricate relationships between stream gravels, bedload movement, beavers and spawning salmon in the STFFIP streams. Cope and Macdonald (1998) looked at the intragravel environments for incubating salmon eggs and Macdonald et al (1998) described the impacts of changing stream temperatures on salmon incubation habitats. Tschaplinski (1998) described the distribution and habitat preferences of adult spawning sockeye salmon in the STFFIP streams while Hogan et al (1998) gave preliminary results from their channel morphology studies.

At this time, there has been limited forest harvesting in the study watersheds but the STFFIP has already significantly improved our knowledge of the natural processes that make interior watersheds and aquatic ecosystems different from coastal systems. STFFIP is also one of the first studies to begin looking at variable-retention riparian management on headwater streams in the upper Baptiste and Gluskie watersheds. Small streams receive little riparian buffer protection under the current BC Forest Practices Code and relatively little is known about the ecological roles of headwater streams in BC. Although the science is of penultimate importance, one of the great legacies of the STFFIP is the determination and effort of a diverse group of researchers and resource managers from university, industry, government agencies and the private sector to collaborate and address important resource management issues that affect all BC stakeholders

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PART I: WATERSHED, STREAM HABITAT AND SALMON STUDIES

Riparian Litterfall Inputs, Storage, and Processing in Undisturbed Forested Streams in the North-Central Interior of British Columbia

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INTRODUCTION

Streams in forested watersheds may receive considerable inputs of particulate organic matter as leaf litter and wood from the adjoining riparian vegetation (Cummins et al. 1983; Murphy and Meehan 1991; Benfield 1997). In very small forested streams, where shading by the riparian canopy limits instream primary production (Lamberti and Steinman 1997), allochthonous inputs from the riparian forest may dominate the organic matter budgets of the streams (Webster and Meyer 1997). As channel size increases, however, the relative importance of autochthonous primary production generally increases (Vannote et al. 1980; Conners and Naiman 1984). The stream-resident biota is believed to be adapted, both functionally and phenologically, to the dominant organic matter source (Vannote et al. 1980; Hawkins and Sedell 1981). Changes to particulate organic matter inputs that result from riparian logging can alter the composition and abundance of the stream biota (Carlson et al. 1990; Davies and Nelson 1994).

Organic matter sources are poorly defined for medium-size streams (bankfull widths, W_b, about 5 – 15 m) in north-central BC that are important spawning and rearing habitats for salmonid fishes such as bull trout (*Salvelinus confluentus*), rainbow trout (*Oncorhynchus mykiss*), and sockeye salmon (*O. nerka*). We are examining organic matter sources in undisturbed watersheds in north-central BC as part of the multi-agency, multi-disciplinary Stuart-Takla Fish-Forestry Interaction (STFFI) study. The STFFI is looking at the impacts of forest harvest on stream ecosystems and the effectiveness of riparian management regulations in preserving stream attributes. Our initial work is on largely undisturbed, old growth forests that will serve as reference systems for watersheds subject to normal forest harvest. Macdonald et al. (1992) further describe the project and its aims.

We report here the input, storage, and processing of allochthonous particulate organic matter in the lower reaches of four medium-size streams (W_b between 8.4 and 14.8 m) in the sub-boreal spruce biogeoclimatic zone of north-central BC. Our objectives were to characterize allochthonous particulate inputs and storage at each study reach, to describe the spatial pattern of inputs to the channels, and to relate allochthonous inputs to canopy closure at the collection site.

METHODS

Study site

Our study sites were three fourth-order watersheds (Bivouac, Gluskie, and Forfar creeks) at about 55° 02' N latitude by 125° 30' W longitude. The streams arise in the Hogem Range of the Omenica Mountains and flow NE into Takla Lake or its outflow, Middle River. The watersheds are physically similar (Hogan et al. 1998) and largely undisturbed except for a logging haul road that intersects the streams at bridge crossings located 500 to 1700 m upstream of the mouths. A fourth study watershed (Baptiste creek) has been partially logged. Watershed areas range from 38.4 km^2 (Forfar) to 74.7 km^2 (Baptiste), with main channel lengths of 16 to 19 km. Discharge and water temperature regimes are typical of northern Interior BC streams. Precipitation is low (about 50 $cm \cdot yr^{-1}$, Macdonald et al. 1992) and accumulates as snow between November and March. Discharge is dominated by the late spring-early summer snowmelt, and remains at low ("base flow") levels from mid July to late September. Water temperatures are 0-1°C from October to April, while summer temperatures vary between 8 and 13°C (Macdonald et al. 1997).

Undisturbed riparian areas along the lower, alluvial reaches of the streams are old-growth forests of hybrid white x Engelmann spruce (*Picea gluaca* x *engelmanni*) and subalpine fir (*Abies lasiocarpa*), with scattered black cottonwood (*Populus balsamifera* spp. *trichocarpa*) and a fringe of alders (*Alnus* spp.), willows (*Salix* spp.), red-osier dogwood (*Cornus stolonifera*), black twinberry (*Lonicera involucrata*), devil's club (*Oplopanax horridus*), *Rubus* spp., and other deciduous shrubs along the stream margins. Maximum tree sizes are about 110 cm dbh for conifers and 140 cm dbh for cottonwood.

We measured litter inputs and coarse particulate organic matter (CPOM, > 5 mm) storage in homogeneous reaches, each about 10 channel widths in length, that were stratified by channel morphology and planimetric form. Table 1 gives average reach characteristics measured from detailed surveys at monumented cross-sections. Reaches that are labeled with the same number (e.g., R1) have similar channel morphology, gradient, width, sediment texture, and pool-riffle characteristics (Hogan et al. 1997). All reaches contained large volumes of large woody debris (LWD). We determined average canopy closure in mid-July for each reach from spherical densiometer readings made at midchannel at intervals of one channel width along the reach.

Litter inputs

We collected direct inputs of allochthonous particulate organic matter in 0.10 m^2 plastic tubs between mid-July and the end of October in 1996 and 1997. There were 5 to 8 litter samplers per reach, depending on reach width and length. We affixed samplers to LWD within the active channel at measured distances from the nearest bank edge. We determined canopy closure at the sampler with a convex spherical densiometer. We emptied the samplers at approximately biweekly or monthly intervals depending on the amount of litterfall. All the material in a sampler was placed in a plastic freezer bag and frozen. In the laboratory, we sorted the thawed material into deciduous leaves, conifer needles, wood (twigs, bark), and “other” (seeds, buds, flowers, mosses, lichens) categories. Leaves and needles were further sorted to species for the dominant taxa. The sorted material was oven dried to constant weight at 60°C in preheated, preweighed aluminum pans, cooled in a desiccator, and weighed to the nearest mg. We determined ash-free dry weights (AFDW) by combusting 8 pre-dried, preweighed samples of each leaf species, wood, or “other” category for 4 h at 500°C, cooling the material in a desiccator, and re-weighing to the nearest mg. We used the mean ratio of AFDW to oven-dried weight (DW) for each category to estimate the AFDW for the remaining samples.

Benthic coarse particulate organic matter

We measured coarse benthic particulate organic matter (CBOM) in each study reach from monthly samples between June and October 1996. We sampled CBOM to a depth of 10 cm at 3 to 5 randomly chosen locations along the reach, excluding deep pools, using a 0.05 m^2 Hess sampler or (occasionally) using a 0.09 m^2 Surber sampler. We removed all macroinvertebrates, then collected the organic material greater than 475 μm with a Nitex® sieve, rinsed it with distilled water, and froze the 3-5 pooled samples from one reach and time in a plastic freezer bag. We later sorted the thawed CBOM sample into deciduous leaves, conifer needles, wood, small fragments of wood and leaves, fish remains, and “other” categories and determined DW as above. We applied the same correction factors as above to estimate AFDW.

We estimated the standing crop of stored large woody debris (LWD) for the Forfar and Gluskie reaches from unpublished estimates of average LWD volumes in late July 1991 to 1996 (D.L. Hogan, Research Branch, BC Ministry of Forests, 2204 Main Mall, University of British Columbia, Vancouver, B.C., V6T 1Z4). Hogan et al. (1998) describe how the volumes were obtained. We converted the volume estimates to biomass using mean densities of LWD (0.2258 $g \cdot cm^{-3}$) calculated from data for 200-year old *Picea-Abies* forests reported in Harmon et al. (1986, their Table 5) and then applying the above correction factors to obtain AFDW.

Instream processing rates

We measured the instream processing of alder and willow leaves in two reaches (B2 and G2) during September and October 1997. We collected leaves at the point of abscission by gently shaking small trees and collecting the leaves that fell onto plastic ground sheets. Leaves were pooled from many (> 30) trees. We held the leaves overnight, constructed artificial leaf packs by sewing leaves together with monofilament fishing line, and then anchored 15-18 artificial leaf packs (6 g fresh weight) to small bricks in riffle areas along each reach where leaves collected naturally. We recovered 3 leaf packs per reach at intervals from 2 to 42 days and froze the material in separate plastic bags. In the laboratory, we thawed the leaf packs, removed and enumerated benthic invertebrates, dried the leaves to constant weight at 60°C, and weighed them to the nearest mg. Stream temperatures were monitored hourly with thermistors located at the lower end of the reaches.

Statistical analyses

We examined relationships between litterfall and physical variables using Pearson product-moment correlation coefficients with Bonferroni-adjusted probabilities. We examined the variation in litterfall with bank distance and with canopy closure among streams by ANCOVA using $\log_{10}(x + 0.1)$-transformed data to linearize relationships and to improve homoscedasticity. We fitted functions to the dependence of litterfall on bank distance and on canopy closure by nonlinear least-squares estimation using the Gauss-Newton method. We examined time trends in stored CBOM among streams using repeated-measures ANOVA on $\log_{10}$-transformed data. We used SYSTAT for all statistical analyses.

RESULTS

Direct inputs of small organic debris (leaves, needles, twigs, bark, buds, flowers, seeds, mosses, and lichens) from the riparian forest to the active stream channel during July to October 1996 and 1997 varied considerably among collection sites (Fig. 1a). Total inputs ranged from 5.9 to 427 g AFDW · m^{-2} and generally declined with distance from the nearest bank. Total inputs were negatively correlated with bank distance (r = -0.480, N = 58, p < 0.001) and positively correlated with canopy closure (r = 0.774, N = 58, p < 0.001). The percentage distribution of litterfall among major categories (deciduous leaves, conifer needles, wood, “other” small organic litter) did not differ among the three geomorphic reach classes (Table 2, χ^2 = 5.96, df = 6, p = 0.43). About 80% of the material was leaffall from deciduous trees and shrubs, which entered between mid September and late October. Conifer needles accounted for about 11% of the inputs, while wood and “other” material each accounted for 4-5% of the litterfall.

Deciduous litterfall into Baptiste, Bivouac, Forfar, and Gluskie creeks at bank distances greater than or equal to 1 m (Fig. 1b) declined with distance from the nearest bank (ANCOVA, p < 0.001) at a common rate (p_{slopes} = 0.13). Deciduous litter inputs into the Bivouac Creek channel, adjusted for bank distance, were greater than those into Forfar or Gluskie creeks (Tukey hsd test, p < 0.05), which did not differ. Exponential functions with a common distance exponent provided reasonable fits to these data (r^2 = 0.45, N = 92), the least-squares fits being:

$$\text{deciduous leaffall (g AFDW} \cdot \text{m}^{-2}) = 152.8\ (\pm\ 18.3)\ e^{-0.49\ (\pm\ 0.09)\ \cdot\ \text{bank distance (m)}}$$

for Baptiste, Forfar, and Gluskie creeks, and

$$\text{deciduous leaffall (g AFDW} \cdot \text{m}^{-2}) = 325.3\ (\pm\ 50.4)\ e^{-0.49\ (\pm\ 0.09)\ \cdot\ \text{bank distance (m)}}$$

for Bivouac Creek beyond 1 m bank distance.

At bank distances less than 1 m in Bivouac Creek, the mean deciduous litterfall was 120.1 (SE = 35.2, N = 5) g AFDW · m^{2}.

The species composition of deciduous leaf inputs did not vary among the geomorphic reach classes (χ^2 = 7.41, df = 6, p = 0.28). Alder leaves accounted for about 50% of the total litter inputs, while willow leaves comprised another 17% (Table

2). Cottonwood leaves comprised about 5% of the total inputs, while leaves from a variety of deciduous trees and shrubs (birch, red-osier dogwood, black twinberry, devil's club, *Rubus* spp) comprised another 7%.

Conifer needle inputs ranged between 0.4 and 49.9 g AFDW · m^{-2} among collection sites (Fig. 1c), but did not vary with bank distance (ANCOVA, $p_{slopes} = 0.35$). Conifer inputs varied significantly among streams (ANOVA, $p = 0.005$), inputs to Gluskie Creek being lower than those to Baptiste or Forfar creeks (Tukey hsd test, $p < 0.05$). Geometric mean conifer inputs were 10.5 (±1.28) g AFDW · m^{-2} for Baptiste Ck., 5.5 (±1.19) for Bivouac Ck., 7.7 (±1.17) for Forfar Ck., and 3.8 (±1.20) for Gluskie Ck. The species composition of conifer needle inputs varied significantly among geomorphic reach types ($\chi^2 = 10.40$, $df = 4$, $p = 0.03$). Spruce needles were the most abundant conifer input at all geomorphic reaches (Table 2). Lodgepole pine (*Pinus contorta* var. *latifolia*) needles were abundant inputs to the upstream R3 reaches, while fir needle inputs were common in the R1 and R2 reaches.

Small woody debris inputs ranged between 0 and 228 g AFDW · m^{-2} among collection sites, however only one input exceeded 30 g AFDW · m^{-2} (Fig. 1d). We omit the 228 g AFDW · m^{-2} input from the following analyses because it is a statistical outlier (studentized residual = 9.45); we discuss its significance below. Wood inputs did not vary among streams (ANCOVA, $p = 0.11$) but did vary with distance from the nearest bank (ANCOVA, $p_{slopes} < 0.001$). Bank distance accounted for only a small proportion of the variance in wood inputs, however. The best exponential fit was:

$$\text{wood inputs (g AFDW} \cdot \text{m}^{-2}) = 12.7\ (\pm\ 3.9)\ e^{-1.01\ (\pm\ 0.35)\ \cdot\ \text{bank distance (m)}}$$
$$(r^2 = 0.12,\ N = 96).$$

Direct inputs of "other" small organic debris ranged between 0 and 55.3 g AFDW · m^{-2} among collection sites (Fig. 1e). Inputs varied with bank distance (ANCOVA, $p = 0.015$) but did not vary among streams (ANCOVA, $p = 0.068$). Bank distance accounted for only a small proportion of the variance in "other" inputs. The best exponential fit was:

$$\text{"other" inputs (g AFDW} \cdot \text{m}^{-2}) = 6.2\ (\pm\ 1.9)\ e^{-0.31\ (\pm\ 0.20)\ \cdot\ \text{bank distance (m)}}$$
$$(r^2 = 0.04,\ N = 97).$$

We estimated the average direct litter inputs to the study reaches by integrating the above equations over the mean bankfull channel width, summing over litter types, and dividing by channel width. Average direct litterfall inputs varied between 49 and 112 g AFDW · m^{-2} (Table 3) for the various study reaches.

Total litter inputs increased with canopy closure (Fig. 2, ANCOVA, $p_{slopes} < 0.001$) but did not vary among streams (ANCOVA, $p = 0.37$). The variation in total litter inputs with % canopy closure was best described by a power function, as:

$$\text{litter input (g AFDW} \cdot \text{m}^{-2}) = 0.760\ (\pm\ 0.61) \times (\%\ \text{canopy closure})^{1.207\ (\pm\ 0.19)}$$
$$(r^2 = 0.60,\ N = 58).$$

Willow leaf packs declined in dry weight from about 2.2 g to about 1.3 g over a 6 week period prior to freeze-up (Fig. 3a). Processing rates did not vary between the morphologically similar B2 and G2 reaches (ANCOVA, $p_{slopes} = 0.99$), nor did initial weights vary (ANCOVA, $p = 0.90$). The processing rate estimated directly from the best least-squares fit exponential function was 0.0141 (± 0.002) day^{-1}. Alder leaf packs declined in dry weight from about 4.2 g to about 2.5 g over the same period (Fig. 3b). Alder processing rates varied significantly between the B2 and G2 reaches (ANCOVA, $p = 0.049$), the best-fit exponents being 0.0129 (± 0.002) day^{-1} at B2 and 0.0084 (± 0.001) day^{-1} at G2. Stream temperatures declined from about 6.5°C to 2.0°C over the period. Average temperatures over the period were 4.0°C at B2 and 4.3°C at G2.

CBOM stored in the riffle and glide portions of the wetted channel varied considerably among sites and times (Fig. 4). CBOM generally increased through time between July and October as organic litter was sequestered in the channel (repeated measures ANOVA, $p_{time} = 0.017$). The time trend did not vary among streams ($p_{interaction} = 0.60$), nor did mean CBOM vary among streams (p = 0.24). Geometric mean CBOM levels increased from about 16 g AFDW · m^{-2} following a major freshet in late July to about 67 g AFDW · m^{-2} in late October (Fig. 4). Much of the CBOM was wood. Large particles of wood (branches, twigs, large bark fragments) averaged 36% of the total CBOM in each stream, however mixed small fragments of wood and leaves (often mostly wood) comprised another 34% on average. Large leaf fragments averaged 17%, while conifer needles accounted for about 8% of the CBOM. Fish fragments from decaying sockeye carcasses and other organic matter (moss, lichens, buds, seeds, etc.) comprised about 1% and 4% of the total CBOM in each stream over the monitoring period. LWD storage at the Gluskie and Forfar creek sites ranged from 682 to 2294 g AFDW · m^{-2}, while season average CBOM values at the same sites were 28 to 59 g AFDW · m^{-2} (Table 4).

DISCUSSION

Direct inputs of small organic debris from the riparian forest to the active channel of these undisturbed, medium-sized streams were dominated by a highly seasonal pulse of deciduous leaf litter. Although the surrounding old-growth forest was composed principally of large conifers, conifer needles accounted for only 11% of direct litter inputs during the July to October period while deciduous leaves accounted for 80%. A single deciduous species, alder, accounted for 50% of the total litterfall. Several other studies of direct litterfall to small streams in conifer-dominated boreal or montane forests have similarly shown deciduous leaves to comprise 50 to 80% of the inputs, with alder dominating (Conners and Naiman 1984; Hartman and Scrivener 1990; Richardson 1992). The composition of litterfall reflects the development of a distinct riparian vegetation community in close proximity to the channel, which differs from the forest overstory.

Inputs of deciduous leaves to the Takla-drainage streams were quite variable, depending on both bank distance and canopy closure. Deciduous litterfall declined with distance from the bank with an exponent of –0.49 (±0.09), which was similar to the –0.46 and –0.51 exponents found for annual litterfall into a similar-width (22 m) stream in an undisturbed boreal watershed (Muskrat River) in Quebec (Conners and Naiman 1984). The intercept value of 153 (±18) g AFDW · m^{-2} at the bank edge for Baptiste, Forfar, and Gluskie creeks was similar to those found for the Muskrat River (164 and

190 g AFDW · m^{-2}, Conners and Naiman 1984). It was also similar to deciduous inputs (132 to 175 g AFDW · m^{-2}) into small montane streams in southwestern BC (Richardson 1992). Measured litter inputs to small streams, however, encompass a wide range of values (0 to 843 g AFDW · m^{-2}, Benfield 1997) depending on the characteristics of the watershed. The much higher bank edge value for Bivouac Creek (325 ± 50 g AFDW · m^{-2}) possibly reflected local differences in the composition of the riparian forest along geomorphically similar reaches in different streams within the same geographic area. The dependence of litter inputs on local conditions was evident in the relationship between litterfall and canopy closure in our streams. Canopy closure provides a simply measured predictor of litter inputs to our streams.

Conifer needle inputs did not vary across the channel and averaged only 4 to 10 g AFDW · m^{-2}, a small proportion of the total litterfall. Other studies in undisturbed coniferous forests have generally noted higher areal inputs of conifer needles (Conners and Naiman 1984; Richardson 1992), but needle inputs nevertheless have generally been lower than deciduous inputs. Inputs of small woody debris and other small organic material varied with bank distance but comprised only a minor part of the total litterfall in our streams. Occasional outliers in our data indicate the irregular nature of wood inputs to the channel. Other studies have similarly noted high spatial and temporal variability in litter inputs (Conners and Naiman 1984).

A single process model accounts for the pattern of variation in litter inputs with distance from the bank for all the litter classes that we distinguished. An exponential decline model described the cross-channel variation in areal inputs of small woody debris, deciduous leaves, and other organic material (bud, flowers, lichens, etc.), however the exponents varied by litter type. The same model can describe conifer needle inputs by setting the exponent to zero. We suggest that the average horizontal dispersion of the various litter types is inversely proportional to their mass. The exponents for the various litter classes appear to rank order by the expected average mass of the litter particle: -1.01 ± 0.35 for twigs, -0.49 ± 0.09 for deciduous leaves, -0.31 ± 0.20 for buds, flower parts, and lichens, and 0.0 for conifer needles. This implies that the effective distance within the riparian area from which litter is recruited to the stream varies among litter classes. On average, the band of near-bank vegetation that contributes litter to the stream channel will be narrower for deciduous leaves than for conifer needles. Over 90% of the deciduous leaf litter that enters the stream as direct litterfall will be from a 5 m band along the bank edge.

Our litterfall estimates will be less than the true annual inputs because we did not measure lateral inputs over the bank and because we did not measure inputs throughout the entire year. Lateral inputs can vary widely depending on such factors as hillslope, vegetation composition, and flood frequency (Benfield 1997). The relative importance of lateral inputs compared to direct inputs decreases as stream width increases (Conners and Naiman 1984). For the 22-m wide Muskrat River, Quebec, lateral inputs were only 7% of the total litterfall (Conners and Naiman 1984). We expect unmeasured lateral inputs to be small because our sites had relatively wide channels (8 to 14 m), and because hillslopes were generally flat immediately adjacent to the channels. Our period of measurement should capture most of the annual direct litterfall. Numerous studies have shown that litterfall in temperate forests occurs primarily during the autumn (Benfield 1997; Richardson 1992) and that little direct litterfall occurs during the winter (Richardson 1992). Our data may under-estimate conifer needlefall and wood inputs, which may occur throughout the year. Both these components were small (5-10%) compared to the measured inputs of deciduous leaves, so that underestimation of these components may have only minor impacts on annual totals.

Processing rates for alder and willow leak packs in riffles during the autumn (-0.0084 to 0.014 day^{-1}) were similar to the range of rates reported for alder in small southwestern BC streams at the same time of year (Richardson 1992). It is difficult, however, to compare processing rates among studies because of differences in methods (Boulton and Boon 1991) which can greatly affect results. Because we used freshly fallen leaves (i.e., without oven drying) and arranged the leaves in artificial leaf packs, we likely avoided methodological artifacts (Cummins et al. 1980; Gessner 1991). Our measured rates place alder and willow in the "medium" group of leaf processing rates (Petersen and Cummins 1974), and imply that leaf detritus can persist for long periods after stream entry. About 160 to 270 days would be required to reduce the weight of leaves by 90%. Leaf litter may therefore be available as a (declining) food resource for detritivorous invertebrates throughout much of the year despite the limited period during which it enters the stream. Because detritivores in streams can be food limited (Richardson 1991), the persistence of leaf litter can influence benthic production.

CBOM storage in riffle and glide areas of the study reaches averaged about 20 to 60 g AFDW · m^{-2}. The values were similar to CBOM in small southwestern BC montane streams (Richardson 1991), but were generally low compared to other small streams in coniferous or boreal forests (Jones 1997). Our measurements may underestimate CBOM because we did not sample in LWD jams and deep pools. LWD jams are important retention sites for litter, but riffles generally retain more leaf litter than pools, which are erosional sites at high flows (Speaker et al. 1984). Average standing stocks of the leaf litter component of the CBOM at the various sites (0.5 to 4.1 g AFDW · m^{-2}) were also similar to those found in southwestern BC montane streams (2.8 and 4.7 g AFDW · m^{-2}, Richardson 1991). LWD accounted for 92 to 98% of the total BOM in our streams. The LWD standing stocks were similar to those in medium-large boreal forest streams (i.e., about 2.3 kg·m^{-2}, Naiman and Link 1997) but were considerably lower than in similar streams in coniferous forests in coastal Oregon (11.7-28.5 kg·m^{-2}, Triska et al. 1982).

Overall, the litterfall input, storage, and processing characteristics of the Takla study streams were similar to those of other undisturbed boreal forest streams. The input of labile CPOM is largely derived from a near-bank community of deciduous trees and shrubs, particularly alder. CBOM in the channel, however, is dominated by refractory LWD derived from the coniferous forest overstory. Measured processing rates indicate that deciduous leaf litter will be available as a food resource to benthic detritivores throughout much of the year. Direct inputs of leaf litter are extremely variable, but are predictable from percent canopy closure, which captures much of the variability in local vegetation composition and abundance.

Management Implications

1. Deciduous trees and shrubs in close proximity to the bank edge are the major sources of direct litterfall to these S2 streams, not the conifer overstory. Maintaining the species composition and abundance of the near bank riparian

vegetation will be important to maintain normal inputs of allochthonous CPOM to the stream.

2. Most of the deciduous leaf litter entering the channel at sites with low gradient hillslopes will originate within 5 m of the bank edge. Lateral inputs from uphill sources may be important at sites with steeper hillslopes. Conifer needle inputs will originate at much greater distances from the channel.

3. Instream processing rates for the dominant leaf litter sources, alder and willow, result in slowly declining standing crops of litter that will extend over most of the year. The life cycles of detritivorous insects will likely be synchronized to the time dynamics of the dominant allochthonous CPOM sources.

4. Over 90% of the CBOM stored in the channels is LWD derived from the conifer overstory.

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Table 1. Locations and biophysical characteristics of the study reaches. Channel morphology data for Forfar and Gluskie creeks are adapted from Hogan et al. (1997). UTM data (NAD83) are for the downstream boundary of the study reach.

	Baptiste Ck	Forfar Ck			Gluskie Ck			Bivouac Ck	
Characteristic/Reach	BAP2	F1	F2	F3	G1	G2	G3	B1	B2
UTM Northing	10U357711	10U342263	10U342026	10U341473	10U339394	10U339460	10U339472	10U335920	10U335584
UTM Easting	6085215	6102170	6101127	6100727	6103862	6103617	6102960	6106151	6105600
Reach type	R2	R1	R2	R3	R1	R2	R3	R1	R2
Bankfull width (m)	8.4	14.2	10.6	11.2	14.8	11.2	12.9	10.5	9.1
Length (m)	70	190	192	138	236	184	122	115	124
Gradient (%)	1.0	0.6	1.4	2.0	0.7	1.3	1.4	1.1	2.1
Bankfull depth (m)	0.66	1.03	1.35	1.06	1.30	1.21	1.54	0.68	0.65
Dominant substrates	cobble/gravel	gravel/sand	gravel/cobble	gravel/cobble	gravel/sand	gravel/cobble	gravel/cobble	gravel/sand	gravel/cobble
Pool length (%)	57	62	62	51	78	57	62	80	35
Riffle length (%)	19	0	5	28	0	25	15	4	24
Run length (%)	25	38	33	21	22	19	23	16	41
Number of log steps	0	5	5	0	3	3	0	4	2
Canopy closure (%)	53	31	34	21	20	31	37	53	56

Table 2. Percent composition of total litter inputs into the study reaches in Baptiste, Bivouac, Forfar, and Gluskie creeks during 1996 and 1997 by litter type and geomorphic reach class.

Litter type / Reach type	R1	R2	R3	Mean
Deciduous leaves:	80.18	82.46	79.45	80.71
Alder	49.98	53.98	50.59	51.52
Cottonwood	2.22	9.09	3.98	5.10
Willow	21.23	12.35	18.19	17.26
Other deciduous	6.75	7.04	6.69	6.83
Conifer needles:	9.76	7.55	14.47	10.60
Fir	2.29	2.85	0.85	2.00
Pine	0.00	0.03	5.97	2.00
Spruce	7.47	4.67	7.65	6.60
Wood (twigs, bark):	5.84	6.7	1.60	4.77
Other litter (buds, flowers, seeds, etc.):	4.22	3.12	4.48	3.94

Table 3. Mean inputs of riparian litter (g AFDW·m^{-2}) to the active stream channel at the

	Baptiste Ck	Forfar Ck			Gluskie Ck			Bivouac Ck	
Litter type / Reach	BAP2	F1	F2	F3	G1	G2	G3	B1	B2
Deciduous leaves	65.2	42.9	54.9	52.5	41.4	52.5	46.7	91.5	100.9
Conifer needles	10.5	7.7	7.7	7.7	3.8	3.8	3.8	5.5	5.5
Woody debris	3.0	1.8	2.4	2.2	1.7	2.2	1.9	2.4	2.7
Other small organic debris	3.5	2.5	3.1	3.0	2.5	3.0	2.7	3.1	3.4
Total inputs	82.2	55.0	68.1	65.5	49.4	61.6	55.2	102.4	112.4

Table 4. Mean storage of benthic organic matter (g AFDW·m^{-2}) in the riffle areas of the active stream channel at the study reaches. LWD is large woody debris; coarse benthic

	Baptiste Ck	Forfar Ck			Gluskie Ck			Bivouac Ck	
Material /	BAP2	F1	F2	F3	G1	G2	G3	B1	B2
CBOM	22.2	36.1	35.1	NA	59.2	27.9	NA	19.1	18.4
LWD	NA	1709	2294	1331	683	1178	701	NA	NA
Total storage	NA	1745	2329	NA	742	1206	NA	NA	NA

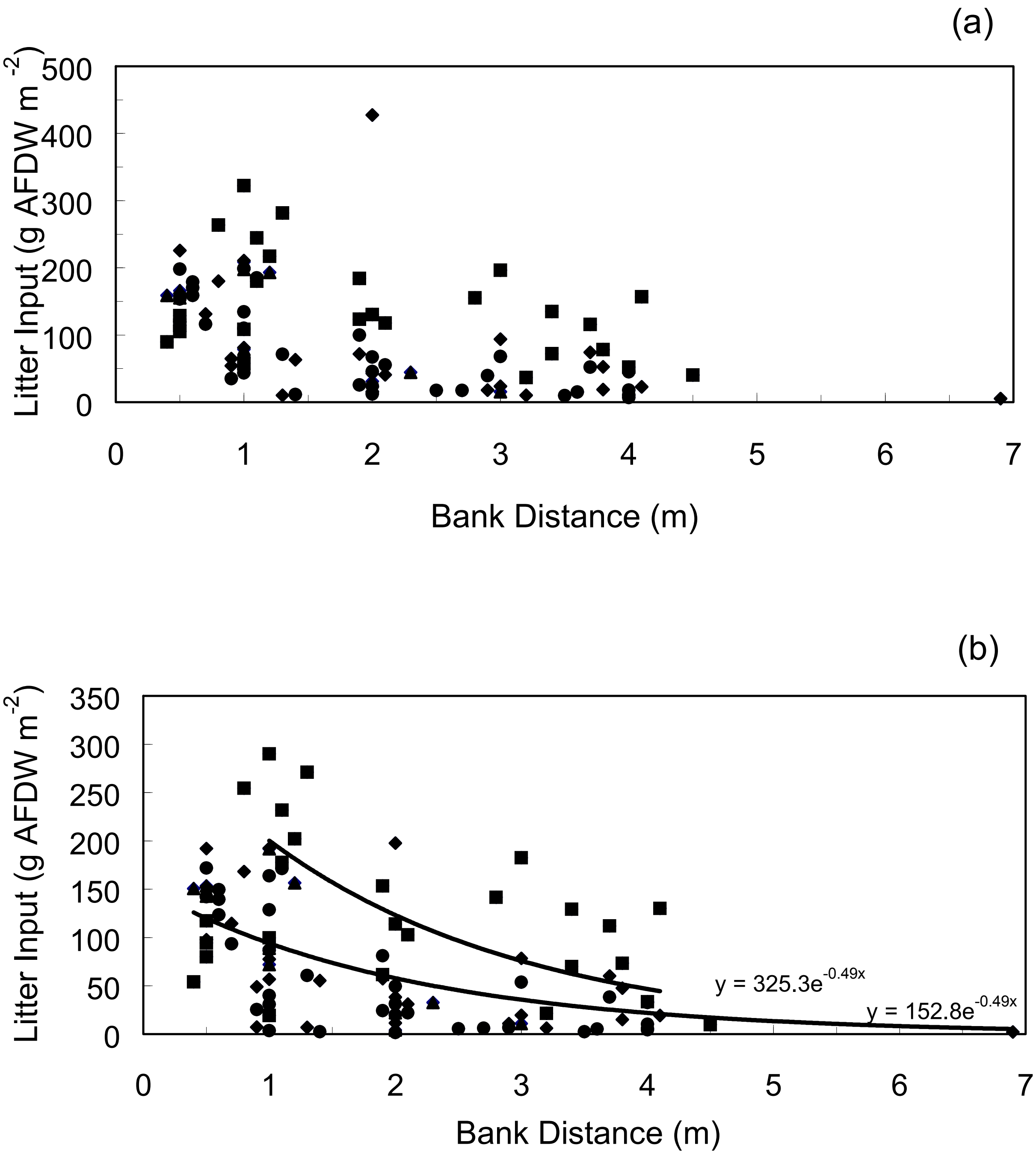


Fig. 1. Direct inputs of: (a) all small organic debris, and (b) deciduous leaf litter, to the active channel as a function of distance from the nearest bank. σ = Baptiste Ck., ■ = Bivouac Ck., ● = Forfar Ck., ◆ = Gluskie Ck. The solid lines in (b) are the nonlinear least-squares fits of a common exponential function to the data for 1996 and 1997 for Baptiste, Forfar, and Gluskie (lower line) and for Bivouac at distances >= 1 m (upper line.

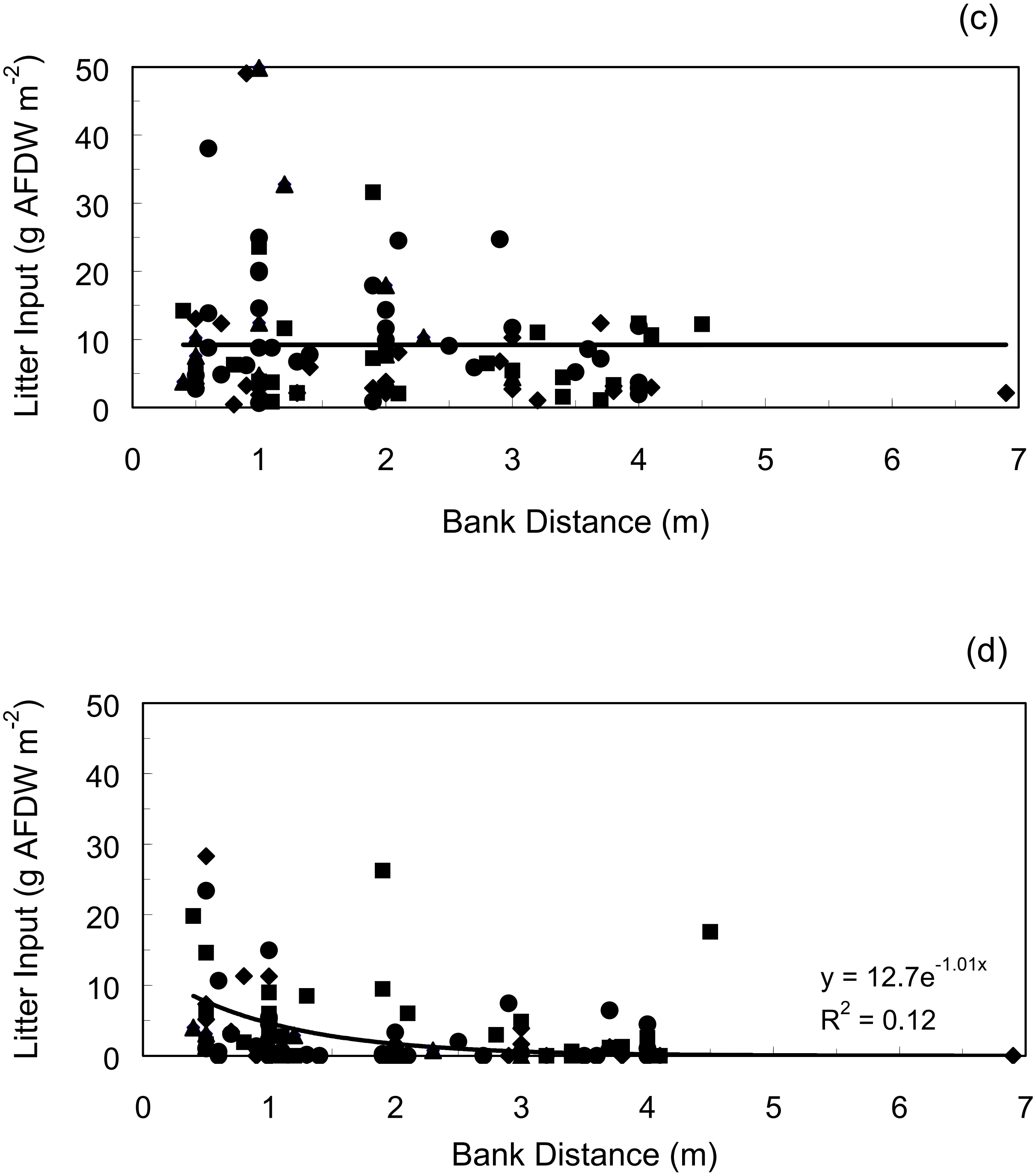


Fig. 1 (cont'd.). Direct inputs of: (c) conifer needles, and (d) small woody debris (bark and twigs), to the active channel as a function of distance from the nearest bank. σ = Baptiste Ck., ■ = Bivouac Ck., ● = Forfar Ck., ◆ = Gluskie Ck. The solid lines are the average input in (c) and the nonlinear least-squares fit of an exponential function in (d).

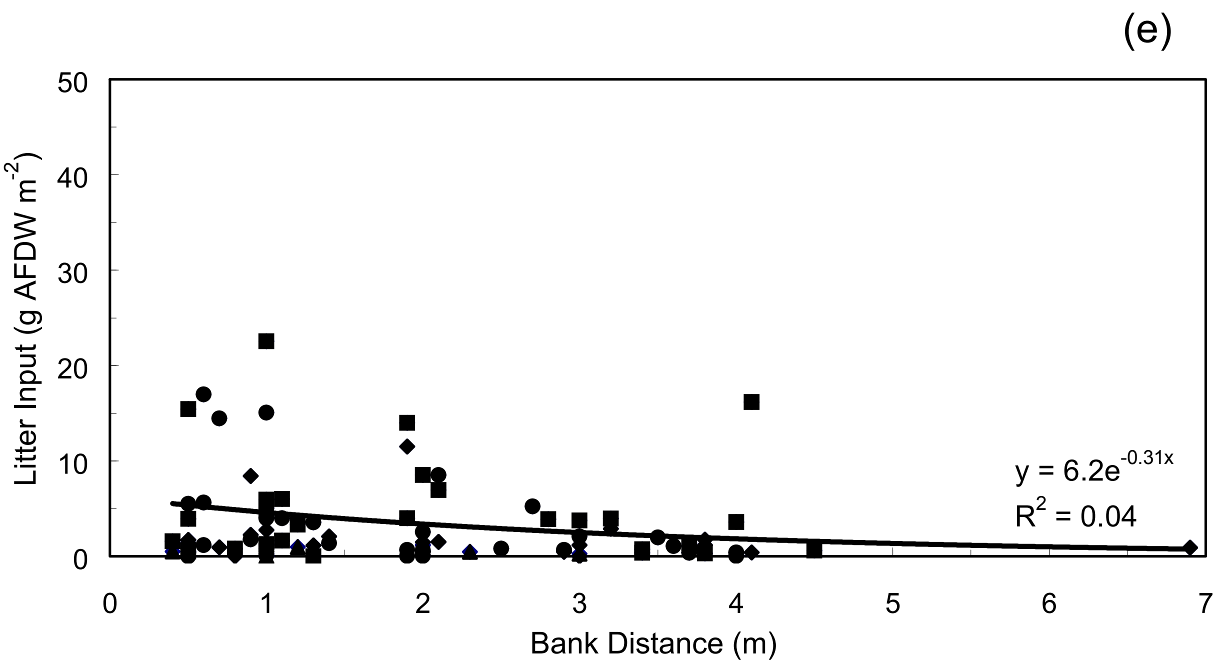


Fig. 1 (cont'd.). Direct inputs of (e) "other" organic debris (flowers, seeds, buds, mosses, lichens) to the active channel as a function of distance from the nearest bank. The solid line is the nonlinear least-squares fit of an exponential function to the data. σ = Baptiste Ck., ■ = Bivouac Ck., ● = Forfar Ck., ◆ = Gluskie Ck.

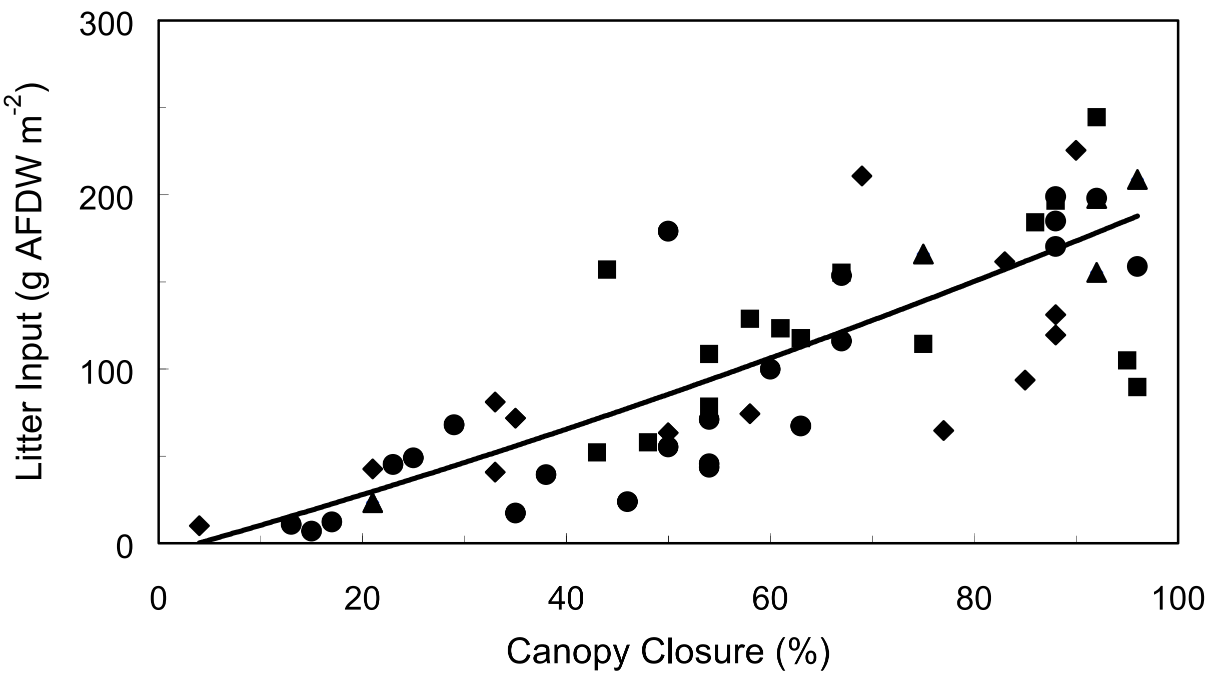
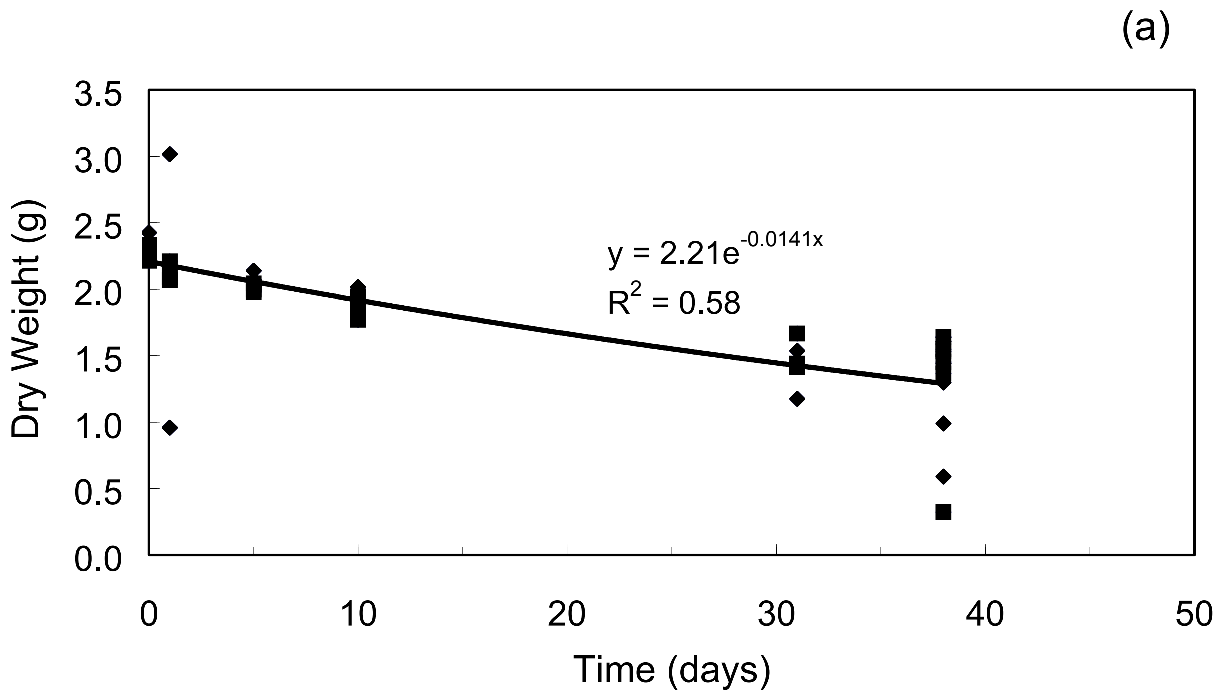


Fig. 2. The dependence of total litter inputs at a collection site upon the canopy closure measured at the site. σ = Baptiste Ck., ■ = Bivouac Ck., ● = Forfar Ck., ◆ = Gluskie Ck. The solid line is the nonlinear least-squares fit of a power function to the data.



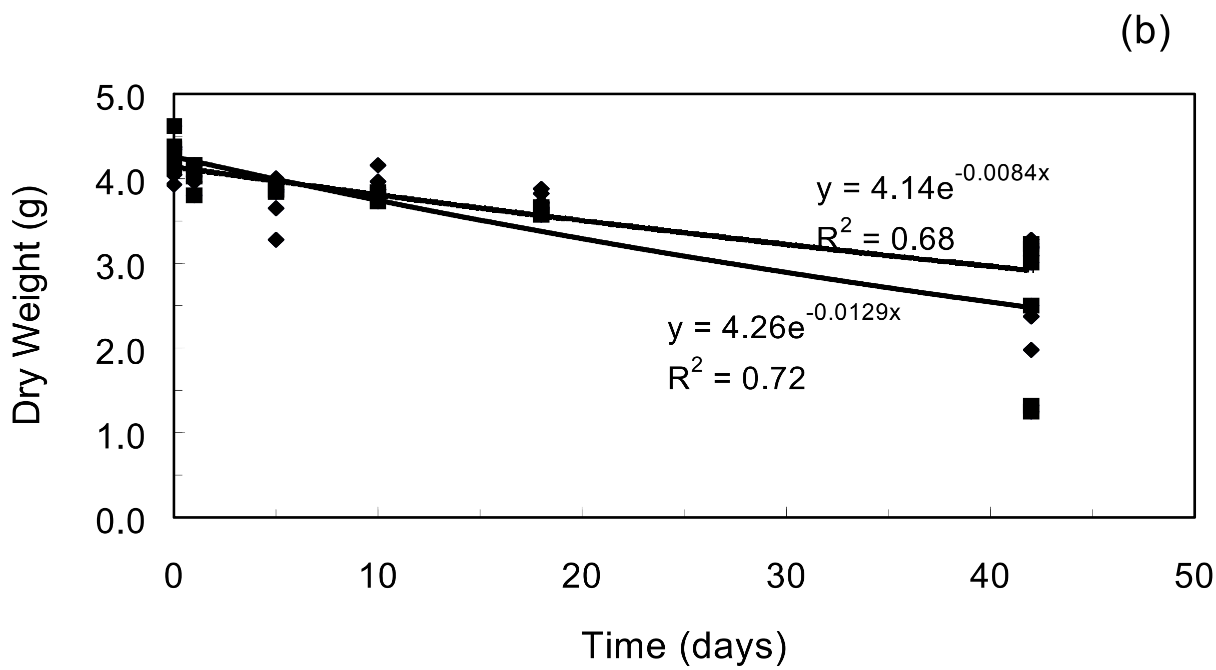


Fig. 3. Dry weights of: (a) artificial willow leaf packs and (b) alder leaf packs through time at the Bivouac R2 and Gluskie R2 reaches in late September and October 1997. ■ = Bivouac Ck., ◆ = Gluskie Ck. Lines are nonlinear least-squares fits of exponential functions to the data; refer to the text.

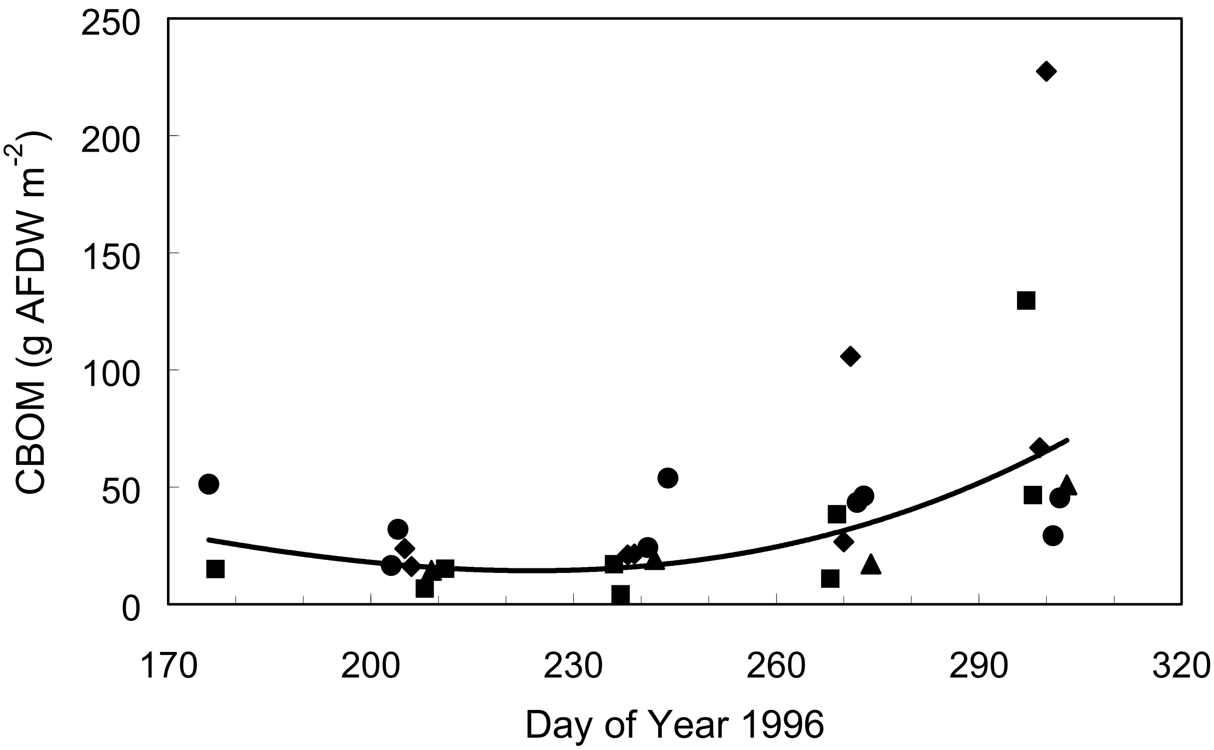


Fig. 4. Benthic coarse particulate organic matter (CBOM) stored in the wetted channel at different reaches from late June to late October 1996. σ = Baptiste Ck., ■ = Bivouac Ck., ● = Forfar Ck., ◆ = Gluskie Ck. The solid line is the nonlinear least-squares fit to the geometric means for the sampling periods.

Evaluation of a Periphyton Strip-Sampler for Monitoring Nutrient Effects from Fish and Forestry in Streams

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INTRODUCTION

The harvesting of trees and the harvesting of salmon are both human activities that can affect the levels of aquatic nutrients such as nitrogen and phosphorous in salmon spawning streams. Forest harvesting disturbs natural soil processes and reduces nutrient sequestering by vegetation in catchment basins, sometimes increasing nitrogen and phosphorous levels in surface runoff and groundwater inputs to streams (Feller et al. 2000). Salmon, in turn, enrich their natal streams with nitrogen, phosphorous and organic carbon from the decomposition of their carcasses after spawning, a nutrient input that fluctuates with natural and fisheries-induced annual variations in spawner abundance (Cederholm et al. 1999).

Historically many salmon spawning streams in British Columbia have experienced significant forest harvesting within their watersheds concurrent with long-term declines in the abundance of spawners largely due to fishing, habitat loss and reduced marine survival (Bradford and Irvine 2000). Although forestry activities could potentially increase nutrient inputs to salmon streams by increasing nutrient export from their watersheds, poor harvesting practices can also detrimentally affect the habitats and productivity of salmon stocks (Hartman et al 1996) and reduce subsequent nutrient inputs from carcasses. Thus forest harvesting and salmon may interact to determine annual and long-term changes in nutrient loadings to spawning streams. The levels of nitrogen and phosphorous in stream water often determine the level of productivity of nutrient-poor streams in BC (Johnston et al 1990, Mundie et al 1991) which can ultimately determine the abundance of stream macroinvertebrates and food for the growth and survival of juvenile salmon and other fish species rearing in the streams and downstream aquatic habitats (Bisson and Bilby 1998).

The Stuart-Takla Fish-Forestry Interaction Project (STFFIP) streams may be ideal for determining how forest harvesting and salmon interact and affect spawning stream productivity. Escapements of Early Stuart sockeye salmon (*Oncorhynchus nerka*) to the study streams exhibit large annual variations over their four-year cycle yet escapement levels tend to be similar among streams each year (Macdonald et al 1992, Ricker 1997). By monitoring changes in stream productivity in streams subject to forest harvesting (e.g. Gluskie Creek) relative to un-harvested control streams (e.g. Forfar Creek) over several years of varying salmon escapements and forest harvesting, the interacting effects of logging and salmon abundance can be studied. Upstream spatial differences in the abundance of spawners within streams can also be exploited to compare gradients in carcass nutrient inputs from spawners relative to nutrient effects from logging as forest harvesting progresses in the upper watersheds.

Periphyton is the attached algae that grow on the rock and wood substrates in streams and it is an important component of organic matter inputs to stream ecosystems (Lamberti 1996). Periphyton is also one of the most frequently used indicators of changes in the productivity and nutrient levels of aquatic ecosystems and there is a long history of using benthic algae as a biological indicator in fluvial ecosystems (Sheath et al. 1986). Periphyton monitoring could be a useful tool to assess how land-use and fisheries activities are affecting the productivity of salmon streams. Periphyton typically respond to increases in nitrogen and phosphorous inputs in low nutrient streams with higher biomass and rates of accrual so monitoring changes in periphyton as harvesting progresses in the STFFIP watersheds may reveal how salmon and forest harvesting are interacting and affecting the nutrient loads and productivity of the streams.

The purpose of this study was to develop and apply a periphyton monitoring protocol as a sensitive and easily measured bio-indicator of changes in nutrient loading and salmon stream productivity. A floating periphyton strip-sampler employing an artificial polyester substrate was chosen because it was simple and inexpensive to construct and use. Artificial growth substrates are frequently used for periphyton sampling to eliminate the high heterogeneity of attached algae found on natural substrates in streams (Perrin and Richardson 1997).We tested the protocol in streams of the STFFIP using reaches with spatial and temporal differences in salmon carcass abundances to determine whether changes in periphyton accrual were a good indicator of changes in stream productivity due to carcass nutrients.

METHODS

Study site

This study was conducted for three seasons from 1996 to 1998 in Gluskie and Forfar creeks, which drain two of the study watersheds of the STFFIP. The creeks are forested fourth-order streams that drain into Takla Lake and Middle River in the northern Fraser River drainage (near 55° 02’ N; 125° 30’ W) and Macdonald et al. (1992) and Hogan et al. (1997) describe them in detail. Bankfull widths are approximately 10-15m in the lower reaches. No forest harvesting has taken place in the Forfar watershed and only a small portion of the upper watershed of Gluskie Creek was harvested several years prior to this study.

Both creeks support sockeye salmon that spawn in the lower 2-5 km of their main stems beginning in early August. Fisheries and Oceans Canada operates salmon enumeration fences near the mouths of the creeks each season. There were large annual variations in spawner densities during the study with total fence counts ranging from 8300 to 11600 during 1996 and 1997 and an order of magnitude lower at 800 to 1000 in 1998. Differences in spawner densities among years, differences in the upstream abundance of spawners for different reaches within a creek, and the seasonal timing of periphyton sampling relative to the beginning of spawning in early August were used to designated periphyton stations as “High”, “Low”, and “No” salmon carcass inputs (Table 1). “No” salmon stations included periphyton stations upstream of the spawning areas or stations sampled in the weeks prior to spawners entering the streams. “Low” salmon stations included periphyton sampling conducted during the spawning year of 1998 when escapements were relatively low as well as stations located near the middle of the spawning beds during other years (e.g. station Mid Gluskie) when the majority of salmon spawned downstream of the station. “High” salmon stations were those immediately upstream of the counting fences during the two high escapement years as well as station Forfar 1260 which had extensive spawning immediately upstream of its location during 1996 and 1997.

Table 1. Year and location of periphyton sampling stations designated as "High", "Low" or "No" salmon which refer to the relative number of sockeye salmon spawning upstream of the sampling station and the presumed effects of upstream salmon carcasses at that station. Total sockeye counts through the fences are also given. Stations not sampled during a given year are designated “Not sampled”. Periphyton data from the months prior to spawning in early August are included in the “No” salmon data.

Station		1996	1997	1998
Gluskie sockeye fence count		8582	11557	812
Gluskie Station	Location			
Lower Gluskie	Upstream of fence	High	High	Low
Mid Gluskie	Upstream of bridge	Low	Low	No
Upper Gluskie	9 km upstream of fence	No	No	No
Forfar sockeye fence count		8381	10070	956
Forfar Station	Location			
Lower Forfar	Upstream of fence	High	Not sampled	Not sampled.
Forfar 1260	1260 m from mouth	High	High	Low

Periphyton sampler construction

We used a floating periphyton strip-sampler that used an artificial substrate for algal accrual because it was:

- a consistent and homogeneous colonization and growth substrate to eliminate the high variability associated with sampling natural periphyton substrates,

- simple and inexpensive to construct,
- easy to sample and inexpensive to analyze the samples to allow for high spatial and temporal replication of data,
- flexible for placement within a stream to sample sites with similar or different levels of shade and current conditions, and
- subject to low invertebrate grazing that might obscure periphyton responses to changes in stream nutrient conditions.

We modified the floating periphyton sampler design developed by Chessman and McCallum (1981) and used polyester strips as a growth substrate (Waite 1979). Mylar® D polyester film (0.2 mm thick) was chosen as the artificial substrate because it is relatively inert, strong, and it transmits >80% of visible light but negligible ultraviolet radiation (UVR) below 314 nanometers wavelength. Some UVR protection was thought necessary for a floating sampler that may be exposed to high solar radiance near the stream surface. The translucence of the film allowed sufficient photosynthetically-active light penetration for periphyton growth on both sides of the horizontally-oriented strips.

Each sampler was constructed from an oval PVC commercial fishing float (8.3 cm diameter, 13.7 cm length, 1.6 cm diameter center rope channel) that had a slit melted into it with a 3.8-cm wide, 0.6-cm thick piece of flat iron that had been heated with a propane torch. The slit intersected with the centre rope channel at mid-float, angling back at a 45° angle and used to hold one end of a polyester strip with the other end trailing in the current behind the float. The polyester film was roughened uniformly on both sides using a fine sanding block and 3-cm wide, 40-cm long strips were cut from the sheet. Strips were soaked overnight in distilled, de-ionized water and air dried. One end of each strip was folded several times around a silicone plastic washer, a hole was pierced through the wrapped film and washer hole, and one end of 1-m length of 2-mm diameter braided nylon cord was threaded through the hole and tied to secure the strip folded over the washer. The other end of the nylon cord was threaded through the melted slit in the oval float, the cord was threaded through the center rope channel of the float and then pulled through to wedge the end of the strip wrapped around the washer securely in the slit. A ¼-inch diameter, 2-inch long hole was drilled into the centre of the underside of the float at a angle similar to and in front of the melted slit and a ½-inch diameter, 2-inch long stainless steel lag bolt was screwed into the hole to provide weight to keep the bottom of the float in the water and the polyester strip submerged.

Samplers were placed in the streams by attaching a stainless steel swivel to the end of the nylon cord and tying the swivel to a line either anchored to a nylon sandbag filled with stream gravel or to an overhead rope tied above the stream from bank to bank. The sandbag anchor used in the first year of the study was found to make the sampler too vulnerable to debris collection on the nylon anchor line so only overhead ropes stretched above and across the stream were used to anchor the samplers in subsequent years. By using overhead anchors, the samplers could also be placed anywhere in the stream to avoid back eddies or to select the current velocities or shade levels the samplers were exposed to at each station. The polyester strips trailed behind each float oriented horizontally to and several centimeters below the stream surface.

Field sampling

Triplicate periphyton samplers were installed at different points across the streams at each station (Table 1) in early to mid July and sampled at 3-14 day intervals until late August. Sampling was more frequent in the first year to establish a high resolution time series to better define appropriate sampling intervals. Current velocities immediately downstream of each sampler were measured with a Marsh-McBirney Model 2000 flow meter after installation and at each subsequent sampling. Water temperature, conductivity and pH were measured in-stream at each sampling with a calibrated field meter and percent canopy closure was measured with a spherical canopy densiometer at each sampler.

Water for nutrient analyses was collected near each array of samplers every time they were sampled. Water was filtered using a clean 50-ml syringe and a 25-mm Acrodisc® glass fiber syringe filter. Each syringe filter was rinsed first with 20-mL deionized, distilled water and 20 mL of sample water before filtering the water sample. Filtered water was collected in acid-washed, 100-mL polypropylene bottles rinsed first with filtered sample and then filled to the shoulder and frozen in the field lab for later analysis of nitrate-nitrite (NO_3) and ammonium (NH_4). A clean, 20-mL Teflon-capped glass test tube was also rinsed with filtered sample and filled to the shoulder with filtered sample water and stored cold and in the dark for later analysis of total dissolved phosphorous (TDP).

Chlorophyll was used as the measure of periphyton biomass because it could be easily measured with high sensitivity and it is commonly used in periphyton studies. To sample the periphyton, a 1-cm wide piece was cut off the end of the trailing polyester strip, placed in a 20-mL glass vial and stored cold and in the dark until frozen in the field lab. On occasion, the last 0.5 cm of the strip was clipped off before taking the sample clip because some samplers placed in areas of high water velocity showed uneven periphyton growth on the trailing edge of the strip due to turbulence.

All periphyton chlorophyll and water chemistry analyses were conducted at the DFO Cultus Lake Salmon Research Laboratory. Methods of nitrate-nitrite, ammonium and total dissolved phosphorous analyses are given in Stephens and Brandstaetter (1983). Chlorophyll-*a* (Chl) was measured fluorometrically after extracting each sample clip in the vial in the dark for 24 hrs after adding 15 mL of 90% acetone (Stephens and Brandstaetter 1983). After extraction, each strip was cleaned, dried and weighed and the area of the sample clip (both sides) was calculated from an area/weight conversion factor for the polyester film. Chlorophyll density (µg Chl/cm^2) on the strip sample was calculated.

Periphyton accrual rates (µg Chl/cm^2/day) were calculated as changes in Chl density on the strips per day:

$$(\text{Chl}_{i+d} - \text{Chl}_{i}) / d$$

where Chl $_i$ is the initial Chl density (µg Chl/cm^2) and Chl $_{i+d}$ is the Chl density *d* days after the initial sampling.

RESULTS

Sampler performance

Each sampler could be constructed for less than $5 Can. and two people could easily construct 15 samplers per hour once the Mylar® strips were prepared. The samplers were light and easily transported to field sites. Deployment on the stream was simple when anchoring to a rope stretched across the stream but anchoring to a nylon sandbag filled with stream gravel was more time consuming and subjected the floating samplers to higher debris loadings because the submerged nylon line acted as a strainer in the stream. The across-stream anchor system also allowed the samplers to "float" over large wood debris and there was minimal fouling of the floats with leaves or other floating debris so this system was used exclusively after 1996. Overhead anchoring also made them resistant to disturbance by spawning salmon when installed in spawning areas and they could be more easily positioned within the stream to vary exposure to different current velocities and levels of shade.

Collecting and analyzing the periphyton Chl samples was straightforward in the field and the lab and a large number of samples could be collected and processed in a day depending on travel times between stations. The fluorometric Chl analysis was a very sensitive technique and small differences in periphyton Chl densities on the strips could be easily detected.

Periphyton colonization and accrual on the polyester strips was relatively uniform to the eye with the exception of some uneven growth on the trailing edge of the strips in high water flows due to turbulence and some tufts of filamentous green algae that colonized the strips at very low flow sites. These parts of the strip were removed or avoided when taking a sample. Although the polyester strips were a homogeneous substrate compared to natural stream substrates, there could sometimes be a large range in periphyton biomass among the triplicate samplers within a stream reach (2-3 fold) implying that small differences in the local growth environment could significantly influence colonization and accrual rates. For this reason, high sampler replication within a reach is recommended.

Invertebrates were rarely observed on the strips except for occasional black fly larvae (*Simuliidae sp.*) which attached to the underside of the polyester strips near the float shaded from direct light. Black fly larvae are filterfeeders and not scrapers that would graze periphyton from the strip (Wallace and Merritt 1980). In general, the floating samplers seemed to offer little opportunity for benthic invertebrates from the streambed or in the drift to settle on the strips and graze the periphyton.

Bears occasional exhibited an interest in the white PVC floats and a few samplers were disturbed or destroyed by them. Seasonal changes in water levels within the stream and occasional storm spates could also strand the occasional sampler or expose the strips to abrasion from rocks or logs so careful placement in the stream and inspection for damaged or abraded strips before sampling was required.

Periphyton responses to spawning salmon

Shade was low and light availability was high at the lower and middle reaches of Gluskie and Forfar creeks where canopy closure densities were <20% at all stream reaches sampled. There was more shade at the Upper Gluskie stream reach where canopy closure averaged 88%.

There was no apparent relationship between stream velocity and rates of periphyton accrual even though the samplers were exposed to a wide range of flow velocities between 0.01 to 0.8 m/sec (Fig. 1). However the abundance of spawning salmon clearly affected periphyton accrual rates with highest accrual >10 µg Chl/cm^2/day in stream reaches with “High” fish abundance and lowest by roughly an order of magnitude in reaches with “No” fish. Periphyton accrual rates at the “Low” fish reaches were intermediate.

During the two years of high salmon escapements in 1996 and 1997, there was a marked increase in periphyton accrual in the days after sockeye salmon began to move through the enumeration fences on both Gluskie and Forfar creeks (Fig. 2). However, there was little change in periphyton accrual during the low escapement year of 1998 when salmon abundance was roughly an order of magnitude lower.

In the weeks before salmon arrived on the spawning grounds, and at the Upper Gluskie reach upstream of a salmon migration barrier, levels of nutrients in Gluskie and Forfar creeks were low with NO_3 and NH_4 <15 µg-N/L each and TDP <7 µg-P/L. In Gluskie Creek, the increases in periphyton accrual that occurred after salmon spawner migration through the fences in 1996 and 1997 also coincided with significant increases in NH_4 and TDP levels in the creek water (Fig 3a). NH_4 and TDP levels showed little change in 1998 when escapements were low. Periphyton in Forfar Creek showed a similar pattern with high accrual coinciding with elevated nutrient levels during the high escapement years and no change in nutrients or periphyton accrual during the low escapement year (Fig 3b).

DISCUSSION

Our objective in this paper was to test a periphyton sampler and its ability to detect changes in periphyton accrual relative to seasonal, annual and spatial differences among stream and stream reaches in nutrient inputs from salmon carcasses. The floating strip samplers performed well in the STFFIP streams. They were easy to place and sample in different stream conditions with little fouling from floating debris or disturbance by spawning salmon. They were sampled frequently to determine changes in rates of periphyton chlorophyll accrual through the season as spawners moved into the streams.

The observed changes in periphyton accrual suggest that periphyton productivity in the STFFIP streams is nutrient limited and that annual and spatial variations in the abundance of spawning salmon and the nutrients they release will cause similar changes in the algal productivity of the streams. Rates of periphyton accrual increased markedly after spawning salmon moved into the streams and were highest where upstream spawners were most abundant. The increased periphyton accrual coincided with increases in nitrogen and phosphorous nutrient levels from carcasses in the streams. These variations in spawner-induced periphyton productivity may also induce similar variations in the abundance of benthic macroinvertebrates that utilize periphyton as a food source. Schuldt and Hershey (1995) found similar increases in nutrients and periphyton after salmon spawning in Lake Superior tributary streams and Wipfli et al

(1998) also found significant increases in biofilm and macroinvertebrate abundances in Alaskan streams due to salmon carcasses.

A variety of artificial substrate samplers have been developed to measure changes in the abundance and species composition of periphyton assemblages in various stream environments. Streams are highly spatially heterogeneous with large within-stream variations in water depth, flow, substrate type and light conditions that can affect periphyton growth. This patchiness in physical conditions causes periphyton patchiness and this high variability affects our ability to detect changes in periphyton assemblages on natural substrates (Townsend 1989). Artificial substrates are used to provide a consistent substrate for periphyton colonization and growth that can be placed in stream areas with similar physical conditions to reduce this patchiness. They are also easier to sample than natural substrates and are the method of choice for many monitoring programs.

We used a floating sampler in an effort to reduce grazing by benthic invertebrates on the periphyton. Invertebrate grazers can constrain periphyton accrual on natural and artificial substrates and obscure subtle responses of periphyton communities to changes in growth conditions in the stream (Rosemond et al. 2000). Various methods have been proposed for measuring periphyton biomass accrual in the absence of grazing pressure. Invertebrate exclusion screens, electric fields and slow-release insecticides have all been used to inhibit invertebrate grazing on artificial substrates, with mixed success (Feminella and Hawkins 1995). For this study, we used physical distance from the streambed to minimize the potential for colonization by benthic invertebrate grazers. Colonization by drifting invertebrates appeared to be limited to occasional *Simuliidae sp.* which are filterfeeders and generally not known to feed directly on periphyton films (Wallace and Merritt 1980).

Our strip-samplers used the periphyton that colonized the strips as a quantitative indicator of changes in stream nutrient conditions and were not intended to measure the dynamics of the natural periphyton communities (Cattaneo and Amireault 1992). The floating strips were almost exclusively colonized by benthic diatoms, suggesting that the flow and surface conditions of the strips may be selective for diatom films.

In this study, we used changes in periphyton accrual as an indicator of disturbances to stream nutrient regimes by salmon and logging in low nutrient interior streams. But forest harvesting can also change many other physical and chemical characteristics of streams which can affect periphyton productivity. The cumulative effects of these other changes in the stream environment also need to be considered when assessing the effects of logging on algal and stream productivity (Resh et al 1988). For example, natural periphyton assemblages require suitable substrates for colonization and low, stable water flows to minimize physical scour. Forest harvesting can introduce new sediment to the stream and reduce large woody debris inputs, affecting the quality of wood, cobble and other substrates suitable for periphyton colonization. Harvesting in the catchment of streams also often increases water yields, storm peak flows and summer base flows, increasing the streambed area for periphyton during summer but intensifying hydrologic scouring of attached algae during flood events (Wehr 1981). Removal of riparian vegetation during logging also reduces shade, increasing photosynthetic radiation for algal growth but also increasing ultraviolet and infrared radiation reaching the stream. High ultraviolet radiation can inhibit and alter periphyton communities and reduce grazing invertebrates (Bothwell et al. 1994) while infrared radiation increases stream temperatures (Brown 1983) affecting periphyton growth and species composition. Thus although periphyton can be a useful indicator that the nutrient regime of a stream has been disturbed, the overall effects on benthic algae and stream productivity are more complicated to assess.

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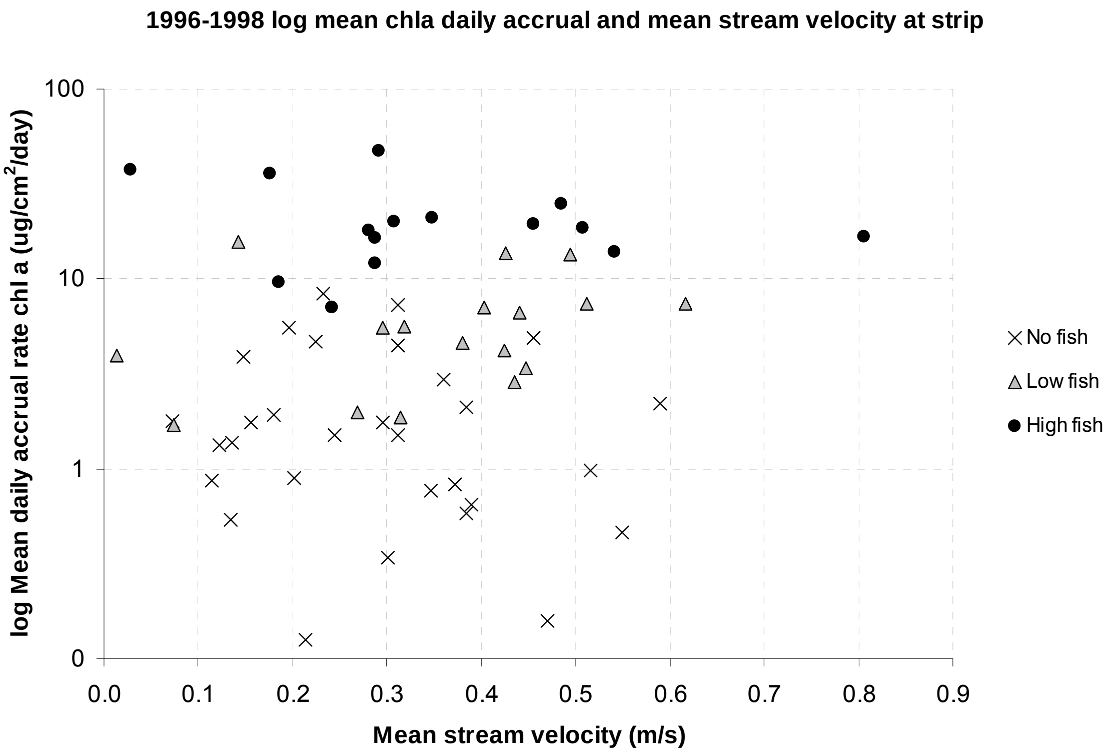

Figure 1: Log mean daily periphyton chl-*a* accrual and mean stream velocity measured at each periphyton sampler for 1996, 1997, and 1998. Growth rates are grouped into three categories; “No” fish (sites with no sockeye presence), “Low” fish (sites with low numbers of sockeye presence upstream), and “High” fish (sites with high fish densities upstream) as described in the text. Note log scale for periphyton accrual rates.

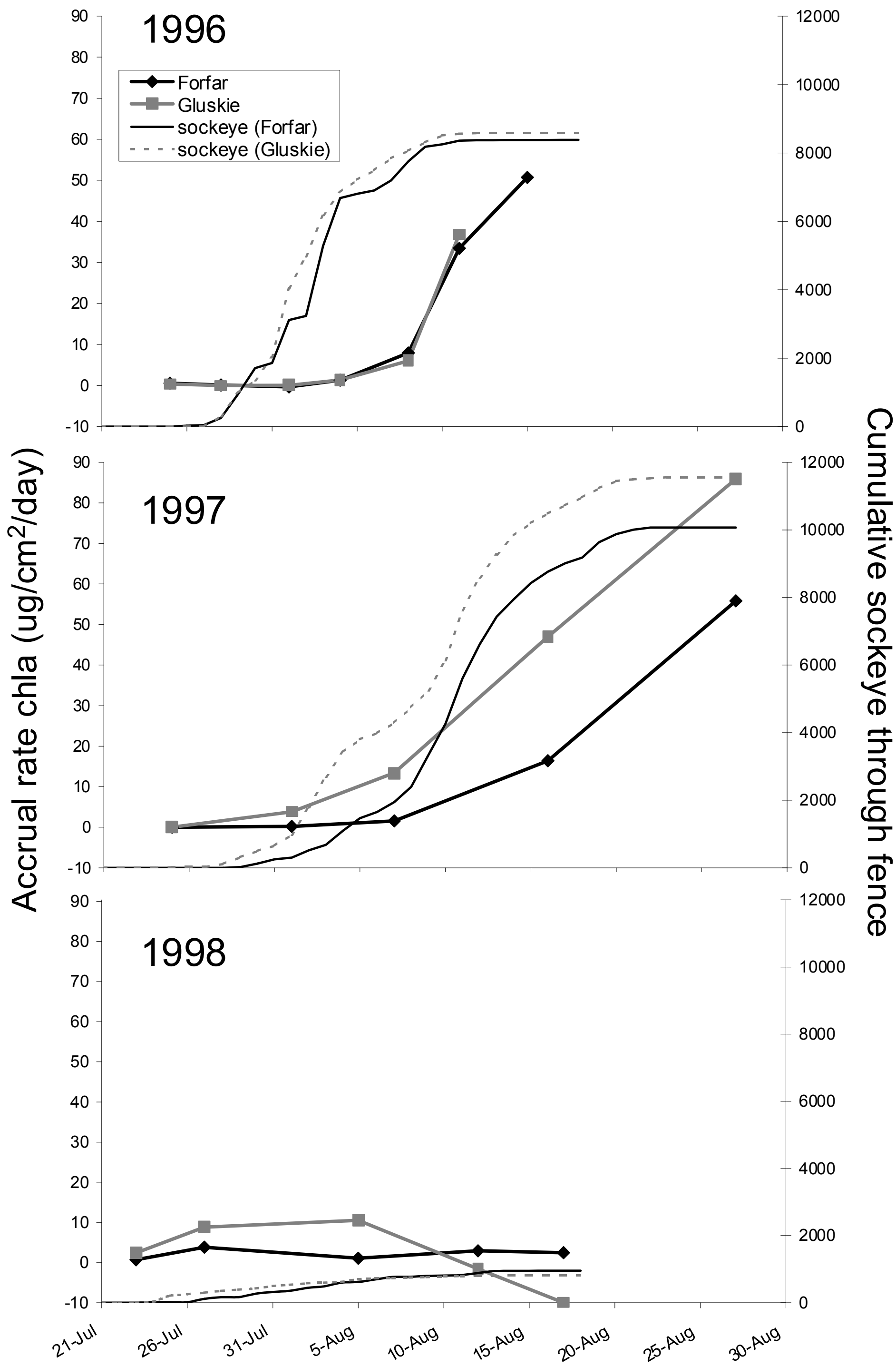

Figure 2: Lower Forfar and Lower Gluskie creeks periphyton daily accrual rate (µg chl-*a*/cm^2/day) and salmon abundance (measured as cumulative sockeye through a downstream enumeration fence) for 1996, 1997, and 1998.

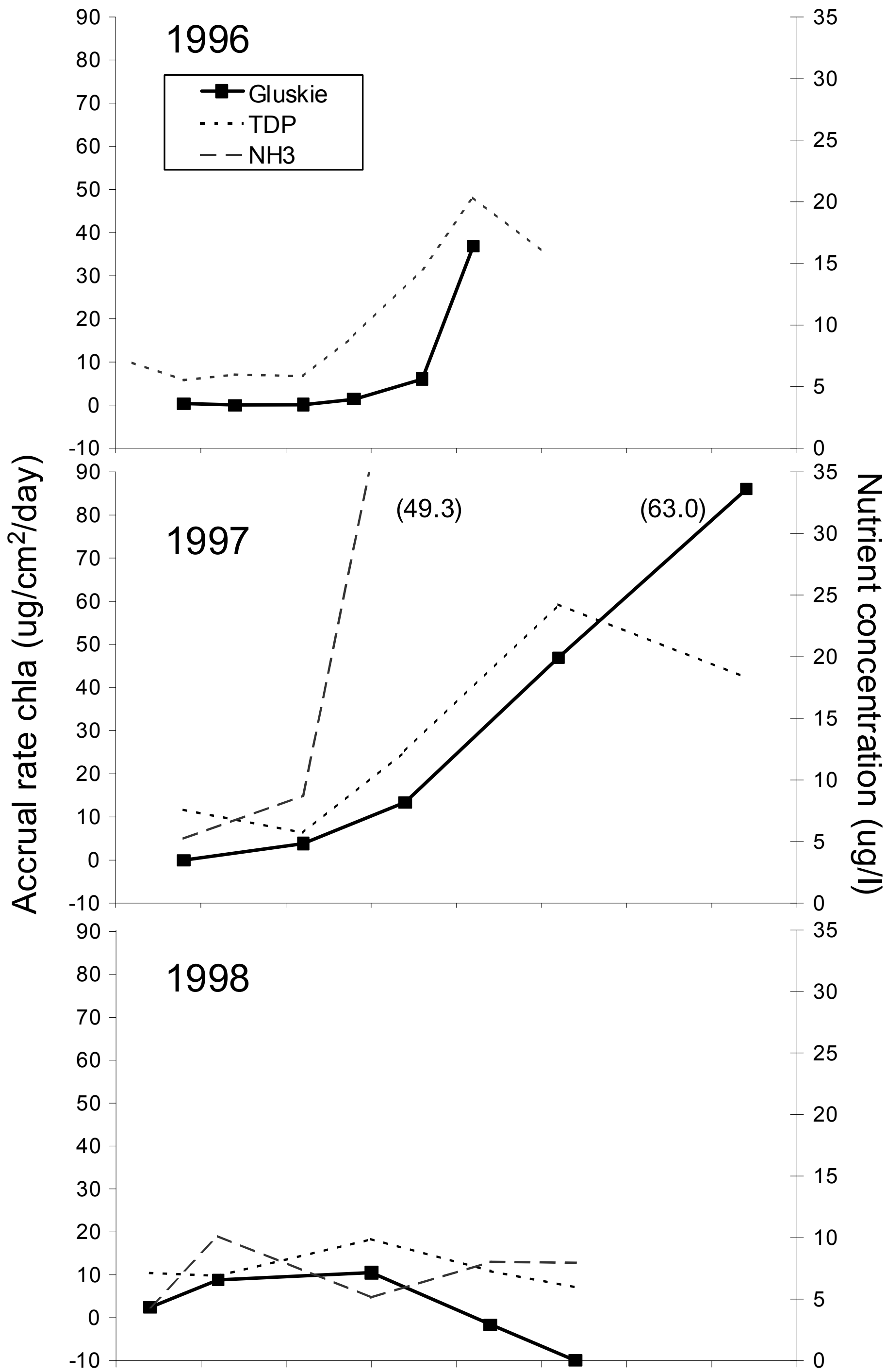

Figure 3a. Lower Gluskie Creek daily mean chlorophyll accrual rates (µg chl-*a*/cm^2/day) and stream ammonium (NH_4) and total dissolved phosphorous (TDP) levels (µg N or P/l) spanning the salmon spawning periods for 1996, 1997, and 1998.

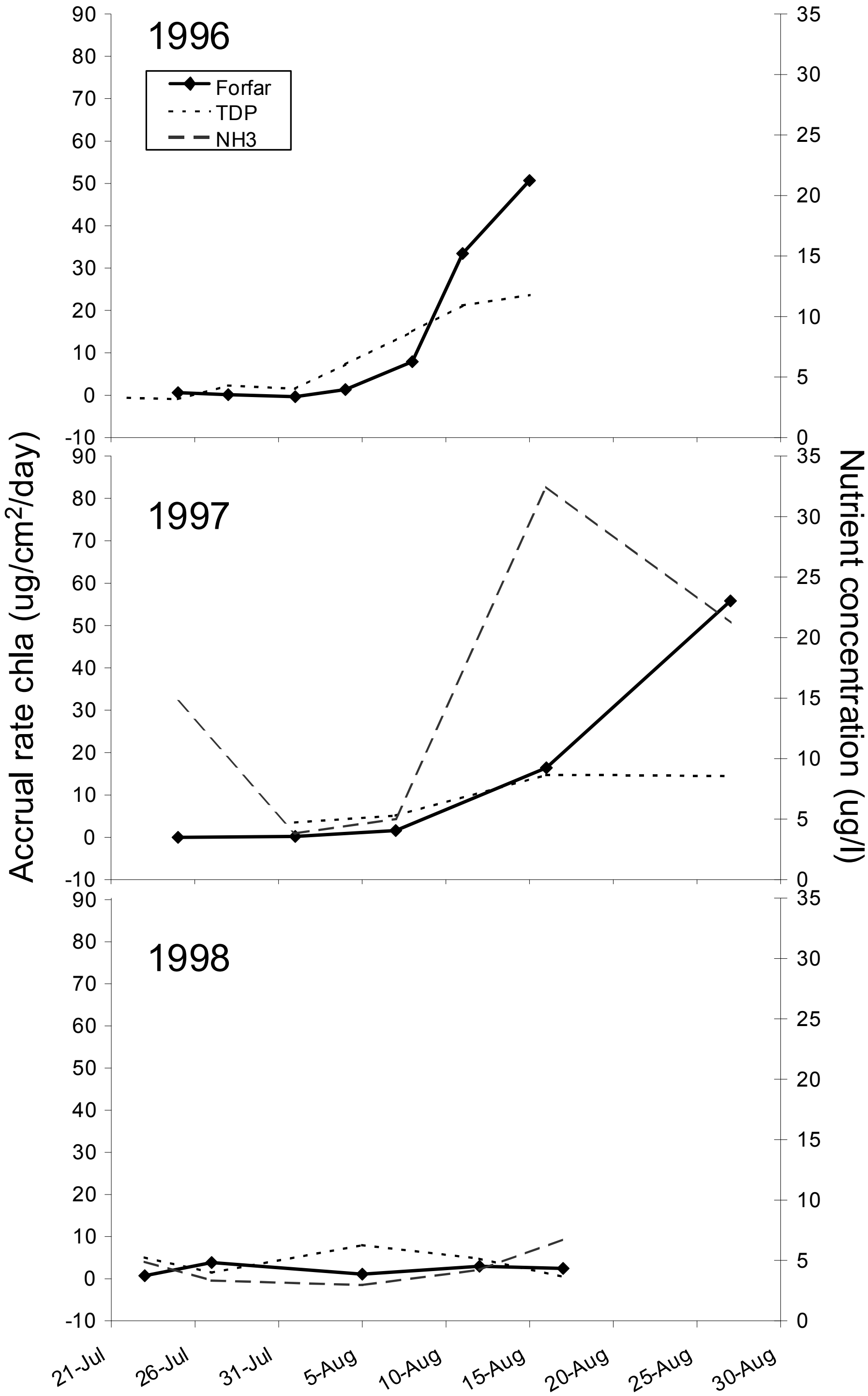

Figure 3b: Lower Forfar Creek daily mean chlorophyll accrual rates (µg chl-*a*/cm^2/day) and stream ammonium (NH_4) and total dissolved phosphorous (TDP) levels (µg N or P/l) spanning the salmon spawning periods for 1996, 1997, and 1998.

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Intergenerational Effects: The Relationship of Adult Sockeye Migration, Land-Use Impacts and Offspring Survival

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INTRODUCTION

Anadromous salmonids can experience extremes of environmental stress during their freshwater migration, natal stream spawning, and overwinter incubation periods. Climate-induced variations in seasonal river conditions during the upstream migration, such as high water temperatures or high discharge, can cause large numbers of salmon to die before spawning and reduce the number of eggs per female deposited at the spawning ground (Gilhousen 1990). The viability and quality of eggs that were deposited by females experiencing extreme en route conditions may be compromised (Donaldson 1990). Exposure to extreme environmental conditions in natal streams during incubation of the eggs could further reduce the survival of the limited number of eggs that were successfully deposited in years with difficult migration conditions (Taranger and Hansen 1993; Cope and Macdonald 1998). These intergenerational effects could have serious consequences for individual and population level fitness (Van Der Kraak and Pankhurst 1997).

Land-use and water management activities such as forestry, agriculture, urban development, and hydroelectric/flood management can mimic and compound the effects of extreme environmental conditions on migrating and spawning salmonids. Clearing of land by forest clearcutting and land development is known to affect the hydrology of watersheds and increase the temperature of downstream waters (Salo and Cundy 1987). Understanding how these environmental stresses affect the various physiological, reproductive and life cycle characteristics of salmonids is key to assessing and managing the impacts of human activity and environmental stress on fish populations.

Although the direct physiological effects of stress on salmonids is relatively well studied, very little is known about the cumulative and indirect effects of stress experienced during upstream migration and in the natal stream environment on the overall fitness of the next generation. These “intergenerational effects” may affect the long-term survival of salmon stocks. Previous research relating adult migration conditions and salmon fitness has been limited to estimates of in-river mortalities and incomplete spawning at a population level (Gilhousen 1990). However, it is not clear how the anadromous salmons’ reproductive system responds to environmental stressors such as high water velocity and high temperature during migration and spawning or how the subsequent generation is affected. Pacific salmon have a finite supply of energy to complete their freshwater migration and maturation of gonadal tissue. The allocation of this energy into gonads and migration is under physiological control, but when extreme environmental conditions are experienced during migration important trade-offs in the allocation of energy may result. One of the key adaptive physiological mechanisms may be the reallocation of this finite energy reserve between migration and reproduction. Clearly, any allocation of energy away from reproduction into the locomotory needs for migration could impact the quality and number of surviving offspring.

The evolution of energy allocation strategies between different physiological processes should support those strategies that maximize fitness. Semelparous Pacific salmon have a single opportunity to optimize the allocation of energy between reproduction and adult migration (Brett 1995). The energy available for reproduction is allocated between different forms of maternal investment, such as egg size, egg quality, and egg number (Quinn et al. 1995; Smith and Fretwell 1974). There is little known relating adult migration conditions and the potential maternal effect on egg size, quality, fecundity and subsequent offspring survival (Beacham and Murray 1993; Linley 1993). There has been little effort to relate an individual strategy, such as reducing egg size during migration, and the likelihood of successful propagation. To date, the differences in egg size and number in salmon have only been attributed to phenotypic plasticity in response to early growth history (Hutchings 1991; Jonsson et al. 1996), and incubation environments (Quinn 1995), and not to physiological responses to increased environmental stress.

In order to improve our understanding of how human activities and increased environmental stress affect the fitness of salmon populations through intergenerational effects, several specific questions needed further research.

- Can migrating salmon facultatively adjust the reallocation of a finite energy supply between migration and reproduction under a rigid semelparous strategy?
- How is this reproductive energy available for final maturation divided amongst contrasting reproductive traits such as egg size, egg number, and egg quality?
- What are the intergenerational implications associated with changes to these reproductive traits for subsequent offspring fitness?

STUDY DESIGN

Fraser River sockeye salmon stocks were selected for this study because of the high annual abundance (2 –10 million), variable migration distance (50km to 1200km), and similar life history strategies among stocks (majority 2 years freshwater residence and 2 years marine) (Forester 1968). Three reproductively isolated populations of Fraser River sockeye salmon (Weaver Creek, Gates Creek, and Early Stuart) with different migration distances (100 km, 400 km, 1100 km respectively) are being used to study the interaction between environmental stress caused by migration conditions and the reproductive physiology of salmon. Some of the Early Stuart stock group spawns in the tributary streams of the Stuart-Takla Fish-Forestry Interaction study area and their long migration approaches the energetic limits for sockeye salmon migration and exposes them to environmental extremes of water flow and temperature (Hinch 1995).

The overall study is composed of four main sub-experiments (in *italics*) that are briefly outlined below. Adult sockeye were sampled from the Early Stuart stock during their upstream migration to relate changes in *egg development* with specific migration conditions. Early Stuart sockeye were also collected near the beginning of their upstream migration and swam at different water velocities in the laboratory to simulate controlled differences in migration conditions (*Adult Swim Trials*). *Field fertilizations* were done to test for gamete quality and viability upon arrival at the spawning grounds for each of the three stocks. Maternal effects on egg characteristics, such as egg size, caused by adverse environmental conditions during adult migration may only become apparent under certain stressful egg *incubation environments*. Therefore, fertilized eggs from both field and laboratory experiments were also incubated under benign and adverse incubation conditions and monitored for survival. These experiments are designed to simulate the types of environmental changes in the incubation environment that may result from forest harvesting activity, such as alterations to the thermal regime (Macdonald et al. 1998).

METHODS AND RESULTS

Egg development during freshwater migration

Female sockeye from the Early Stuart stock were sampled along their upstream migration route to measure changes in different reproductive parameters; egg size, egg quality (protein, lipid, fatty acid), fecundity, reproductive hormones, stress indices, and body constituent analysis (protein, lipids). This will provide an estimate of energy allocation to reproduction and the timing of ovary development during different sections of freshwater migration and potentially highlight facultative adjustments to egg number, size or quality. For example, rainbow trout, *(O. mykiss*) can re-absorb eggs at late stages of development through atresia, reducing female fecundity (Donaldson 1990). Sockeye differentially use energy stores in the upstream migration, using fat reserves first followed by protein (Idler and Clemens 1959). This ability could extend to include differential remobilization and deposition of egg constituents - lipids, fatty acids, proteins- which could affect egg size and quality.

In 1999, Early Stuart sockeye were sampled at 8 locations during their upstream migration from Port Renfrew through Hell’s Gate and on to the spawning grounds at Kynoch Creek (aka O’Ne-ell creek). Blood plasma samples, histology samples, and carcasses were collected at each site and they are currently being processed. Preliminary information on changes to egg wet weight and egg diameter with migration distance have been analyzed (see Fig. 1). For Early Stuart sockeye, the majority of the egg mass is laid down during upstream freshwater migration. There was a two-fold increase in egg diameter and a four-fold increase in individual egg mass during the spawning migration. Comparisons between 1999, a very high water velocity year for the Fraser River migration, and other less stressful years will be performed to see if egg size was affected by adult migration.

Field fertilizations

Field fertilizations experiments were completed for each of the stocks in an attempt to correlate migration conditions, maternal health, and egg characteristics with progeny survival. Within each stock, eggs from individual females were fertilized with milt from individual males. Individual brood lines were established and survival documented to different development stages (see Fig. 2). Maternal condition will be assessed by proximate analysis of their carcasses and histology samples. The latter samples will be assessed for signs of disease and stress; these samples include sections of the kidney, head kidney, gills, pyloric caeca and liver. These parameters as well as female egg characteristics will be correlated against offspring survival to the different life stages. The overall survival of the eggs to the eyed staged for each of the stocks were high (88 % for Early Stuart, 86% Gates, 96% Weaver) despite the arduous 1999 migration conditions in the Fraser (Herunter et al. 2000). The two longer distance migrating stocks had lower survival and higher variability in egg survival than the short distance migrating Weaver stock.

Adult swim trials

By swimming fish in a controlled environment we can manipulate the amount of energy used for migration and see how the energy left for reproduction is distributed amongst egg number, size and quality (Brett 1995). In July 1999, 40 Early Stuart sockeye were captured at Franklin Rock, the beginning of the Fraser Canyon 175km upstream from the mouth, and transferred to the Cultus Lake Laboratory. The fish were forced to swim under controlled migration regimes until final maturation. Twenty fish, 14 females and 7 males, swam against a water velocity of 0.5 m/s for 28 days, covering a ground distance of 1200 km. If these fish had not been captured they would have had 900 km left to swim taking 19 days and this translates into an average ground speed of .55 m/s (Hinch pers. comm.). However, when you factor in the velocity of water and use the swim speed values, the total distance swam from Franklin Rock to the spawning grounds is approximately 1400 km on an average year (Hinch pers. comm.). The remaining 20 fish, 14 females and 7 males, were held in a similar tank with no directional flow for the same period. It is unlikely that maximal swimming and maximal egg development can occur simultaneously during upstream migration. Chinook salmon reduce blood flow to the intestinal system during periods of maximal swimming (Thorarensen et al. 1993). This type of arterial control of blood flow could allow blood to be shunted away from the ovaries and towards the locomotory muscles, with the potential to negatively affecting egg size and quality.

Doing full crosses of all surviving females with all surviving males created individual brood lines for each pairing. Progeny survivals were monitored to different development stages (eyed, hatch, alevin, and fry) to evaluate maternal effects from different migration conditions. Survival to the eyed stage does not appear to be related to treatment effect. The overall survival to eyed stage from the swim trial (45%) was half that of the Early Stuart field trails (88%) despite using similar dry fertilization techniques (Herunter et al. 2000). Part of these differences was due to poor water quality at Cultus Lake Laboratory, where fungus growing on the incubating eggs caused high mortalities. The overall confinement stress may have masked any real differences between the two treatments reflected in the highly variable and equivocal results. Alternatively, it is also possible that the fish did not swim against a strong enough current to elicit a treatment response. Changes to the overall body condition are being examined by looking at histology samples and body constituent analysis. In addition, blood plasma samples were collected at the time of spawning to determine acute levels of stress and differences in the reproductive endocrinology of exercised and non-exercised fish.

Endocrine control of maturation may provide a useful tool in assessing migrational stress on salmonid reproduction, particular in the regulatory control of egg size, number, and quality (Donaldson 1990; Pottinger et al. 1999). Acute and chronic stress applied to maturing adults can affect a salmons' reproductive physiology (Campbell et al. 1992; Pottinger et al. 1999). Under controlled conditions, stress applied during gonad maturation has a negative effect on egg size and on progeny survival (Campbell 1992). Therefore, circulating plasma levels of androgens, estrogens, cortisol, and vitellogenin will be collected from field and laboratory trials (Estay et al. 1998). Hormones levels will also be sampled in developing offspring to determine the relative contribution from maternally derived origin and any possible connection to offspring viability (Stratholt et al. 1997).

Incubation environment & fitness

Maternally derived qualities, such as egg size and egg quality have been used as surrogates for fitness, but very little empirical evidence exists of a consistent relationship to survival in salmonids (Hutchings 1991; Quinn et al 1995). Moreover, trade-offs involving egg number, size, and quality may only become apparent under certain environmental conditions experienced by developing embryos and juveniles (Hutchings 1991; Quinn et al 1995). Fertilized eggs from the field and lab experiments will be incubated under both optimal and sub-optimal conditions. In 1999 work had began on incubating Early Stuart eggs at different thermal regimes to determine the rate of egg development to different critical life stages. This information will be coupled with individual variation in development rate at a single thermal regime. The idea is to relate a maternally derived characteristic, such as egg size, to development rate (Brannon 1987; Linley 1988) and then determine how a change in thermal regime will affect a potential intergenerational effect (egg size) on time to critical life stages. For example, in a year when females deposit smaller eggs because of harsh en route conditions, these eggs may also face high incubation temperatures increasing growth rate to a point where yolk reserves are utilized prior to optimal out-migration timing (Macdonald et al. 1998).

MANAGEMENT IMPLICATIONS

Fisheries management

Intergenerational effects associated with spawning migration would have a direct impact on stock management decisions. Any change in average fecundity has a direct impact on the subsequent density dependent mortality of offspring. Currently the best number available for estimating brood year size is the estimated egg deposition. This number is calculated by multiplying the average female fecundity by the estimated number of effective females (adjusted for egg retention). There is no correction or adjustment made for changes in overall egg viability. If we assume poor egg quality in years of harsh in-river migration, optimal spawning and incubation conditions may be necessary to ensure juvenile survival. It is anticipated that the present research would make a major contribution to generating a correction factor. If river conditions are expected to be bad or perturbations to incubation environments have occurred then managers could change escapement targets to better reflect these changes. Therefore any information relating progeny fitness associated with an intergenerational effect would aid in stock management.

Habitat management

Habitat managers would benefit from knowing how these intergenerational changes in egg characteristics interact with the natal stream environment. Changes in the natal stream environment caused by forest harvesting have the potential to interact with these maternal effects in a myriad of ways. Forest harvesting has the potential to alter stream environments through changes in sediment inputs and thermal regime (Chapman 1988; Macdonald et al. 1998).

Certain land use activities are known to decrease spawning ground quality (Salo and Cundy 1987). Sockeye have adapted their egg size to match local incubation environments, with finer gravels favouring smaller egg size (Quinn et al. 1995). Smaller eggs have a greater surface to volume ratio increasing the survival chances in lower intergravel dissolved oxygen conditions that are associated with finer sediments (Quinn et al.1995). Increases in suspended sediments and the concommittant increase in deposition of fines associated with forest harvesting can affect the water quality of the incubation environment by reducing gravel permeability and decreasing intergravel dissolved oxygen (Chapman 1988). Stressed females that produce smaller eggs may have higher survival in gravels with high concentrations of fines but this benefit may be negated by the alteration of the thermal regime and its effect on juvenile salmon growth.

Changes in stream temperatures, caused by land use activities, will affect juvenile sockeye development rate (Brannon 1987; Macdonald et al.1998). Any alteration to this development rate either through a maternal effect on egg size or a forestry related change in thermal regime could affect offspring survival. Early Stuart sockeye spawn during the period of highest annual stream temperatures (Andersen 2000). Currently these temperatures are within the optimum range for spawning in these predominantly intact watersheds. The majority of the thermal units required for development are accrued during the first two months following egg deposition (Cope and Macdonald 1998). Any change in spawning date could have serious consequences for egg development rates and emergence timing. In 1999, sockeye arrived at Takla Lake later than average, which was attributed to the high discharge in the Fraser River (Macdonald pers. comm.). This late arrival delayed spawning, limiting the amount of thermal units available for egg development prior to winter. Developing embryos collected from some of the Early Stuart streams in early December 1999, when stream temperatures were near $0^{o}C$, had not hatched which is unusual for these streams (DFO unpublished data). The thermal regime for forest harvested watersheds in the winter suggests that stream temperatures may be colder earlier in the late-fall, early-winter period. This would make late hatching eggs more susceptible to *in situ* freezing of the intergravel substrate. These changes in development rate due to temperature changes are confounded by another potential maternal effect, egg size. Temperature and egg size both effect the rate of salmonid embryo development (Brannon 1987; Linley 1988). Egg development rates have evolved to maximize survival opportunities at hatching and fry emergence (Brannon 1987; Macdonald et al. 1998).

PROJECT SUMMARY

The aim of this research is to demonstrate how the reproductive system of sockeye salmon responds to adult migrational stress, how this migration stress affects their progeny, and how land use activities and intergenerational effects interact with changes in the natal streams associated with land use activities. We will measure reproductive traits to determine if adult migration conditions affect adult reproductive physiology and progeny survival. We can then gain a better understanding of the possible plasticity in the physiological mechanisms available to salmonids to adjust egg size, fecundity, and egg quality during spawning migrations. The potentially facultative decisions involving changes in reproductive investment have implications in both short term - individual offspring survival- and long term - population adaptation and survival. These implications are not in isolation; any intergenerational change in maternal investment will be tested *in situ* against an altered incubation environment. Our understanding of the biological implications of forestry harvesting on salmonid survival will be aided by examining the interaction between maternally derived egg characteristics and natal stream environments.

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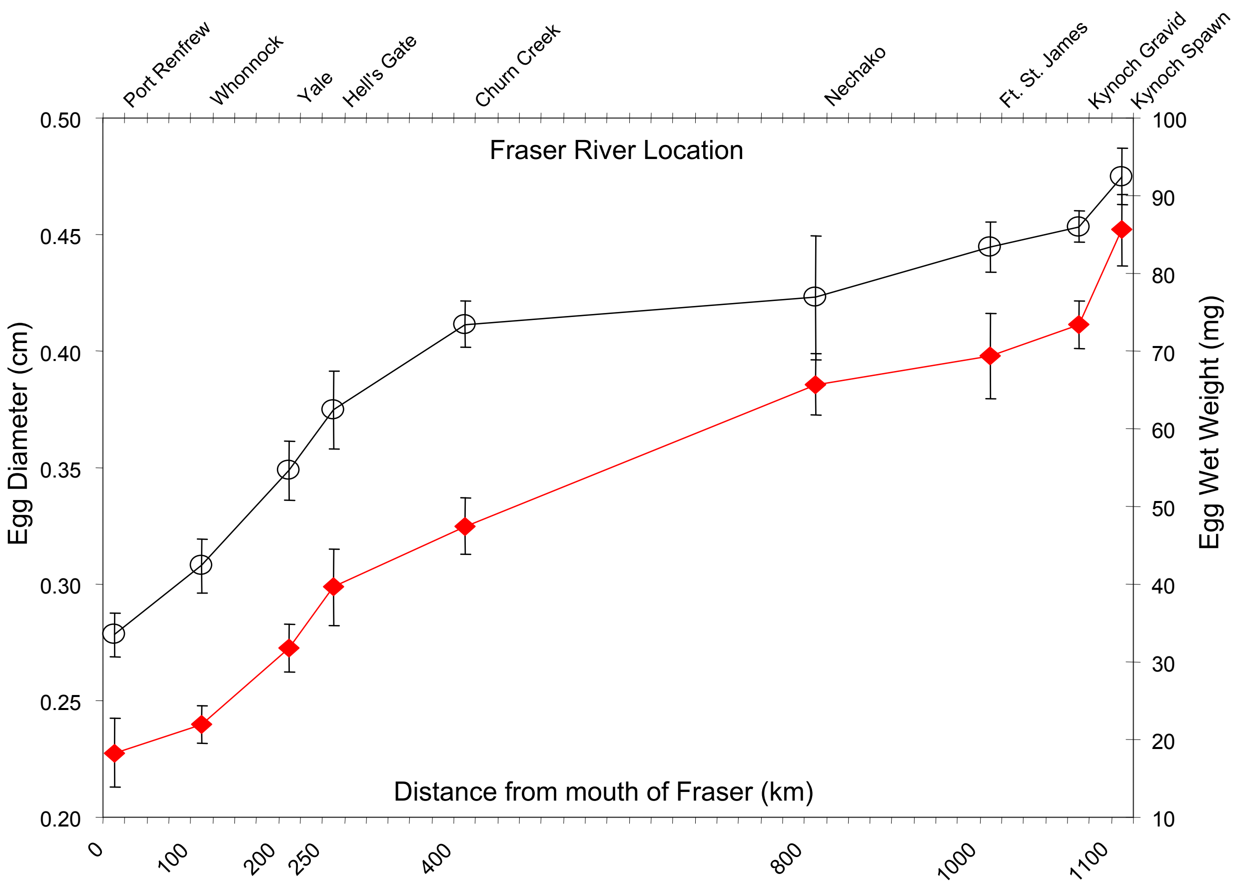

Fig. 1: Egg measurements from 10 females per sample location along the Fraser River. Bars represent 95% Confidence Limits

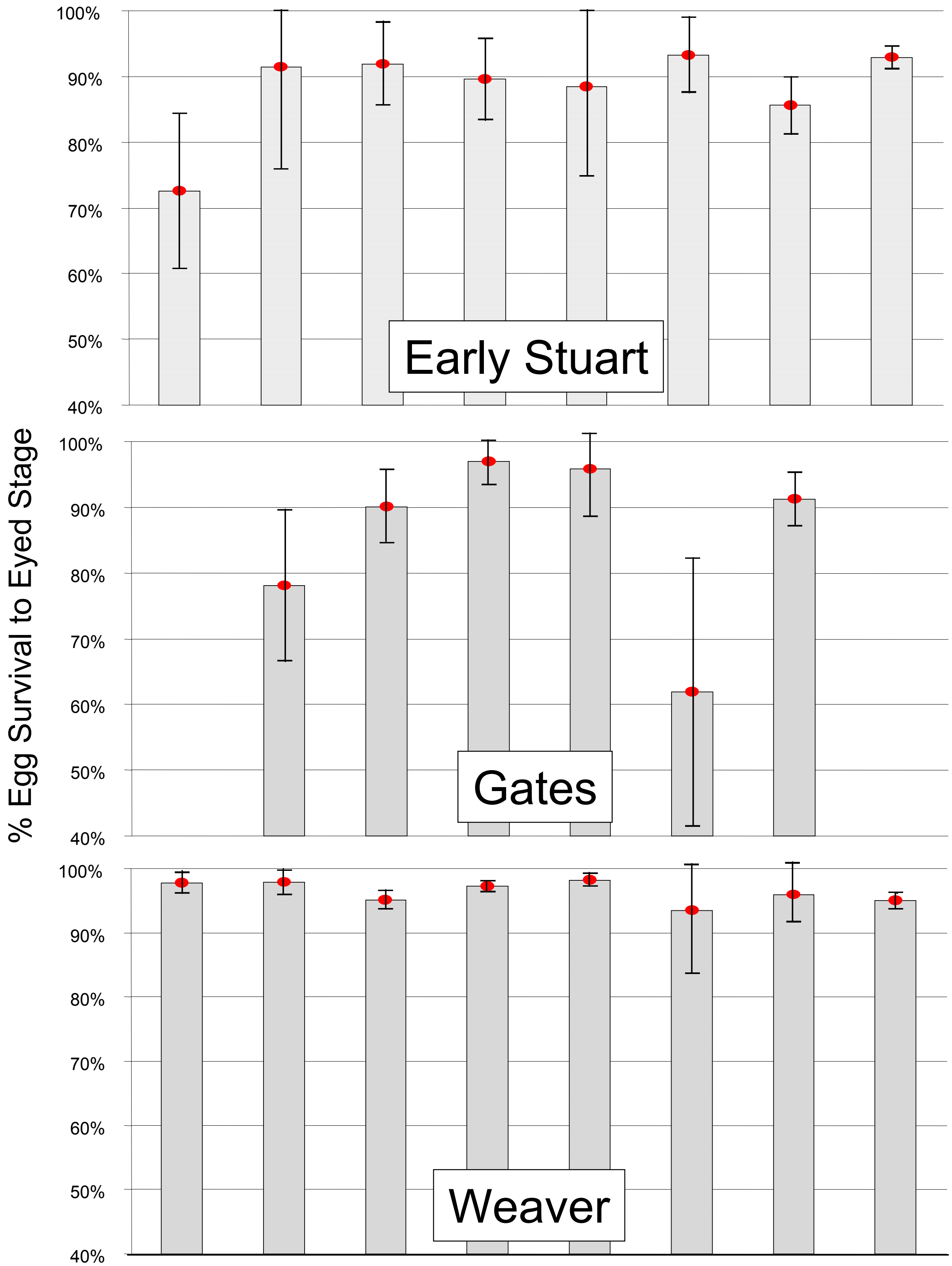

Fig. 2: Egg survival to eyed staged for individual females (Bar) from each stock. Lines represent 95% C.I., n=3 for Gates and Early Stuart and n=6 for Weaver.

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Snow Survey Measurements in the Stuart-Takla Project

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INTRODUCTION

Early-Stuart sockeye salmon spawn in late July and early August and their eggs hatch in late autumn. Alevins over-winter in the stream gravel and emerge in the spring as fry, then move downstream to their lake rearing habitats. Snowmelt has an important influence on the behaviour of early-Stuart sockeye salmon fry, by providing a hydrologic cue in their natal streams which affects emergence timing from the gravel and subsequent emigration to the lakes in which rearing takes place. Macdonald et al. (1998) reported that under undisturbed conditions the majority of fry emigrated from the Takla study streams on the rising limbs of each of the snow-melt generated water discharge events, over a period ranging from 15 days to greater than 1 month in late April and May.

Alteration of patterns of snow accumulation and melt by clearcut logging a portion of the natural forest cover in a watershed may to lead hydrologic changes that are ultimately reflected in changes in flows in the salmon bearing streams. Modification of the flow regime may, in turn, result in shifts in sockeye fry emergence and emigration timing which could be detrimental to the sockeye stock. For example, earlier emigration due to earlier snowmelt may result in fry arriving in the rearing habitat prior to zooplankton, the primary food source, being available in sufficient quantity for survival. Another outcome of changes in snow melt patterns could be the extension of emigration over a longer period, thereby increasing the risk of predation by piscivorous resident fish.

An assessment of the nature and magnitude of logging-induced changes to natural snow accumulation and melt patterns is a key part of understanding the interaction between logging activities and sockeye salmon populations. An overview of the broader snow hydrology component of the Stuart-Takla Fish-Forestry Interaction Project has been provided by Heinonen (1998). Selected snow survey measurements made to date are reported on and briefly discussed here.

METHODS

Forfar Snow Courses ~ Topographic Distribution of Snow

Snow courses were established in 1993 at three elevations (“upper” at 1410 m, “middle” at 1150 m, and “lower” at 735 m) in the Forfar Creek watershed. These snow courses were installed in forested locations according to BC Ministry of Environment Lands and Parks snow survey network standards (Water Inventory Section 1982). These sites do not attempt to measure natural variability in snow properties in forests, rather they provide an index of snow accumulation and ablation in very small forest openings, much less than one tree height in diameter. Their purpose is to assist in developing an understanding of the topographic and temporal relationships between basic snow properties in the study watersheds. The data will also form an important component of planned hydrologic modelling. Measurements are made by provincial snow survey staff at five locations per snow course at the end of February, March, and April each year. If snow pack conditions are high a set of measurements may also be done at the end of May.

Clearcut versus Forest Snow Comparisons

In the autumn of 1993 four 50-meter long snow courses were established in a new 51-hectare clearcut and another four in the adjacent natural forest. Their purpose is to quantify differences in fundamental snow properties between forested and clearcut areas in the Takla Lake region. These snow courses are in a small un-named watershed directly tributary to the Middle River between Forfar Creek and Gluskie Creek watersheds. Snow water equivalent (SWE), depth, and density are measured using a standard snow tube. Observations indicate that the surface water flow in this lower elevation watershed is ephemeral, limited to the periods influenced by snowmelt, and also resulting from some rainfall events. The site elevation is approximately 710 to 720 m, roughly 20 to 30 m above Takla Lake, and the topography is gently sloping and undulating.

Each snow course consists of 10 measurement locations spaced 5 meters apart along a line, for a total of 40 samples per treatment. Measurements commenced in March 1994 and have since been made up to 3 times per year, generally in the late February to late April peak of snow accumulation period. Initially sample locations were "paced-off" between two end markers, then in 1996 a new sampling method after Teti (1995, personal communication) was implemented. This method consists of spacing 10 markers at 5 meter intervals along the lines formerly paced-off; using a random number from 1 to 12, the sample is taken at a point 1 meter out from the marker as if it lay at that "hour" on the face of a clock. This greatly reduces site selection bias; particularly in the forest where otherwise the tendency is to choose "easier" sampling spots. If the sample location thus selected corresponds to a tree, a value of zero is recorded for all snow measures, so the ground surface area along the snow course occupied by tree stems is reflected in the data. The clearcut snow courses are all greater than 50 m from the forest/clearcut boundary, and the forest snow courses are greater than 30 m from the boundary to avoid edge effects.

RESULTS AND DISCUSSION

Topographic Distribution of Snow

The Forfar Creek snow course data from 1993 to 1999 are summarized in Figure 1. The graph shows that over the seven years of measurement on average SWE increased progressively during the late winter period at all sites except the lower elevation site. The lower site had net loss of water from the snow pack by the end of April both on average, as well as in each individual year. It should be noted that a 2.5%

decline in SWE was recorded at the high site in one of the six years at the end of April, while at the middle elevation site similar small declines occurred in three of the six years.

The average rates of SWE change with elevation and are summarized in Figure 2. The range of values is similar to that reported by Toews and Gluns (1986) in south-eastern B.C. The more than doubling of the rate of SWE change at the end of April in the lower to mid-elevation zone is a reflection of the net ablation conditions at the lower elevation site combined with continued accumulation or minimal ablation at the mid-elevation site.

Clearcut versus Forest Snow Differences

The differences between the clearcut and adjacent forest in SWE and snow depth for each snow course date are shown in Figures 3 and 4. With just one exception, the only dates on which greater SWE was recorded in the forest were in late April and early May after all the snow had melted from the clearcut, including along the snow courses.

Figure 5 shows spring snow pack differences between the low elevation (710 m) forest and adjacent clearcut area over the 1994 to 1999 period. These data do not include the late April and early May measurement dates when the snow had already ablated from the clearcut. The late February and March measurements are indicative of conditions near the peak of snow accumulation. These snow course measurements over six years reveal that there was on average 51% more water in the clearcut snow pack and that it was 37% deeper and had a 10% greater density than the forest. The SWE difference was larger than the 37% higher SWE in clearcuts reported by Toews and Gluns (1986) in south eastern B.C., and the 11 to 43% range of differences reported by Winkler (1999) in south-central B.C.

Toews and Gluns (1986) found the range of SWE increase with elevation for clearcuts was nearly double that of forested slopes. If this pattern also occurs in the Takla watersheds, it could be highly significant for sockeye fry. Late April and early May hydrograph peaks in the fry bearing reaches are likely most strongly influenced by snowmelt from lower and middle elevation zones in the Takla study watersheds. Changes in the timing of melt at low and middle elevations due to logging is of particular interest to fry migration, despite the fact that its SWE is less than the higher elevation zone. In the higher elevations the bulk of the snowpack melts later, once the fry have already emigrated from their natal streams.

An additional issue is the “de-synchronization” of melt between the low elevation forested and clearcut areas after logging. If this results in a reduction in hydrograph peaks in late April, when the low elevation melt appears to be dominant, fry emigration in the early part of the melt season may be delayed by suppression of the hydrologic cues initiating fry emergence from the gravel. This scenario could be beneficial to salmon populations, since concentration of emigration later in the melt season would tend to reduce predation and be timed more favourably for food supply in rearing habitats. Lack of a clear understanding of the nature of the hydrologic cues and biological and other factors complicates the analysis.

Watersheds with clearcuts distributed over a greater range of elevations may experience less hydrologic impact on fry emergence and emigration behaviour as changes in melt patterns offset one another. Hydrologic modelling and data collection focused on the topographic rates of SWE change between the natural forest and clearcuts will lead to an improved understanding of these relationships.

Hydrologic recovery of clearcuts occurs when the planted trees grow to a point where they begin to behave hydrologically in the same manner as the natural forest. The length of time until hydrologic recovery of clearcut forests is not precisely known for the Takla area but can safely be assumed to be several decades. A conservative logging strategy, both in ECA and in the topographic distribution of clearcutting in a watershed, should aim to avoid any adverse hydrologic impacts on fish-bearing streams. A slow progression of logging in an individual watershed is more likely to achieve this by keeping the ECA low. The question of interest is how low an ECA is required to avoid hydrologic impacts of the type described above.

SUMMARY

Selected snow survey measurements made in the Takla Lake area were reported on. The SWE at peak of snow accumulation has been 51% higher on average in the low elevation clearcut relative to the adjacent forest. This is a reflection of the 10% greater snow density and 37% deeper snow pack in the clearcut. The hydrologic implications of clearcut logging in the study watersheds will depend on the ECA eventually reached, as well as the topographic distribution of the logged areas, among a range of factors. Lower and mid-elevation zones appear to be primary contributors to late April hydrograph peaks under natural forest conditions, and may contribute to an even greater degree when the forest cover is removed. Additional work on hydrologic modelling and the topographic rate of increase in SWE are suggested.

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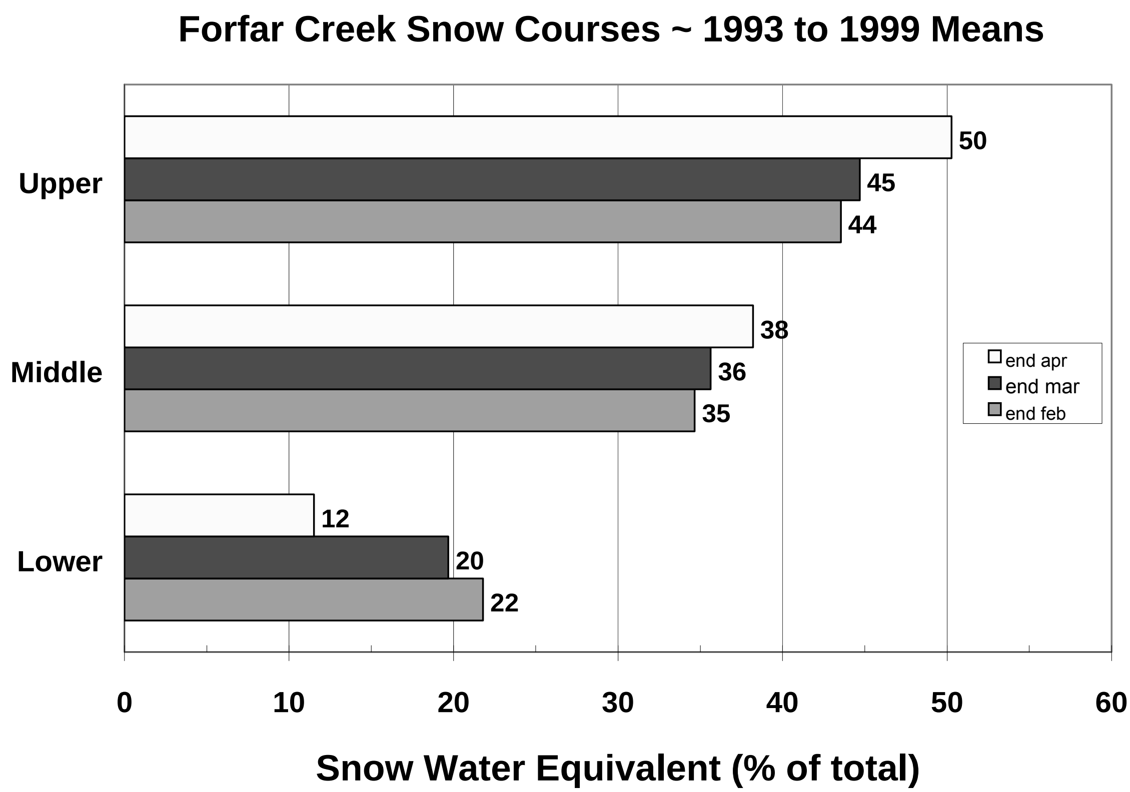

Fig. 1. 1993 to 1999 late winter mean snow water equivalent at three elevations.

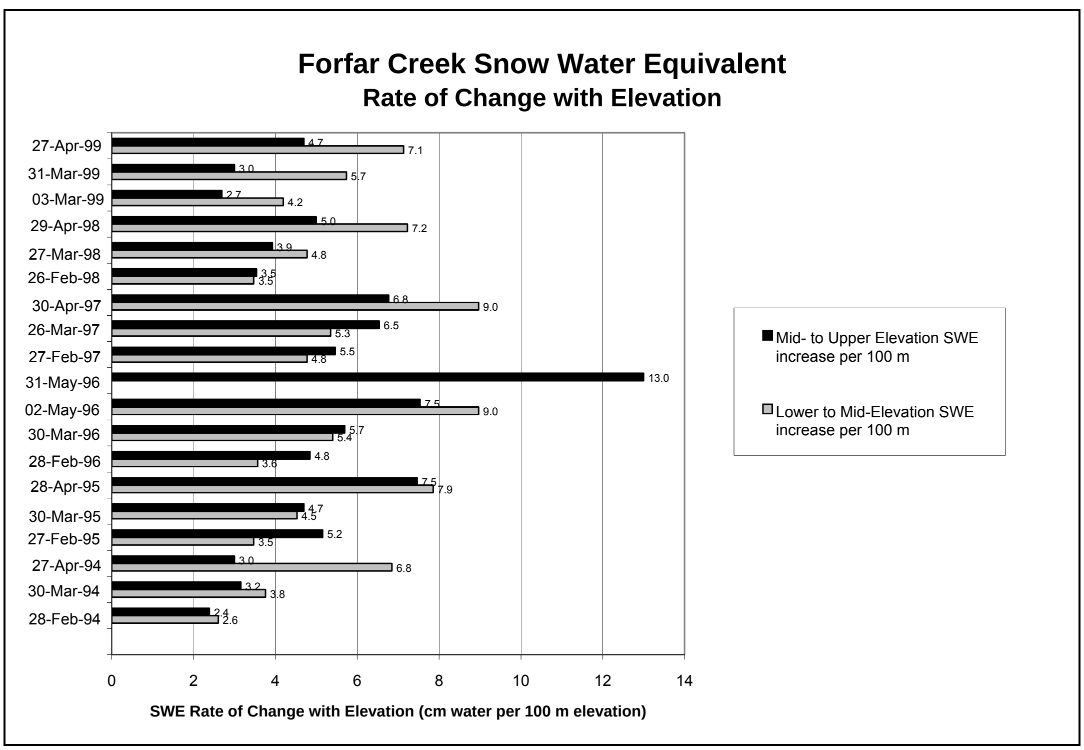

Fig. 2. Rate of late winter snow water equivalent change with elevation.

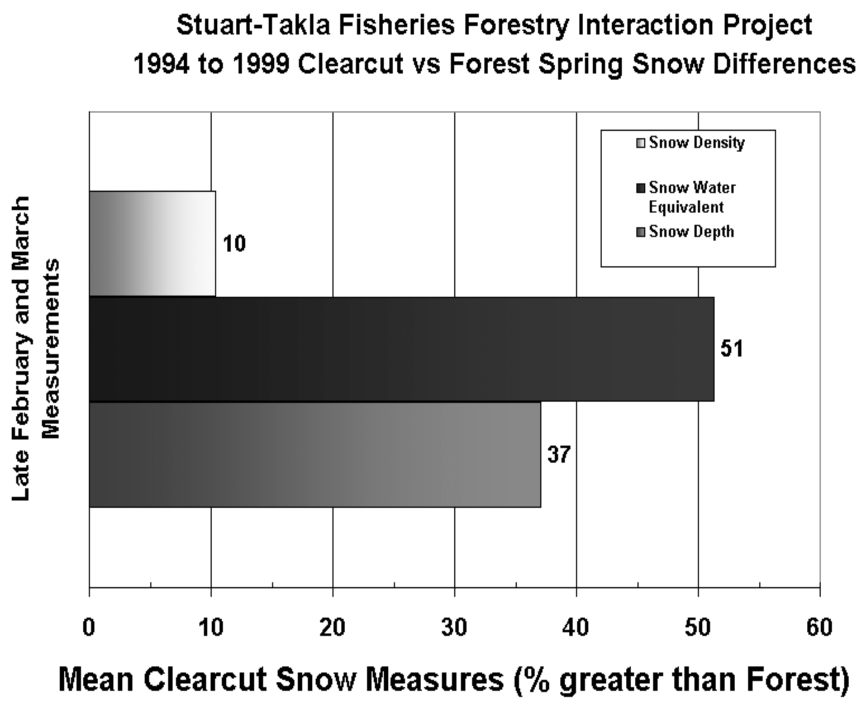

Fig. 3. 1994 to 1999 clearcut versus forest mean differences in snow depth, density, and water equivalent.

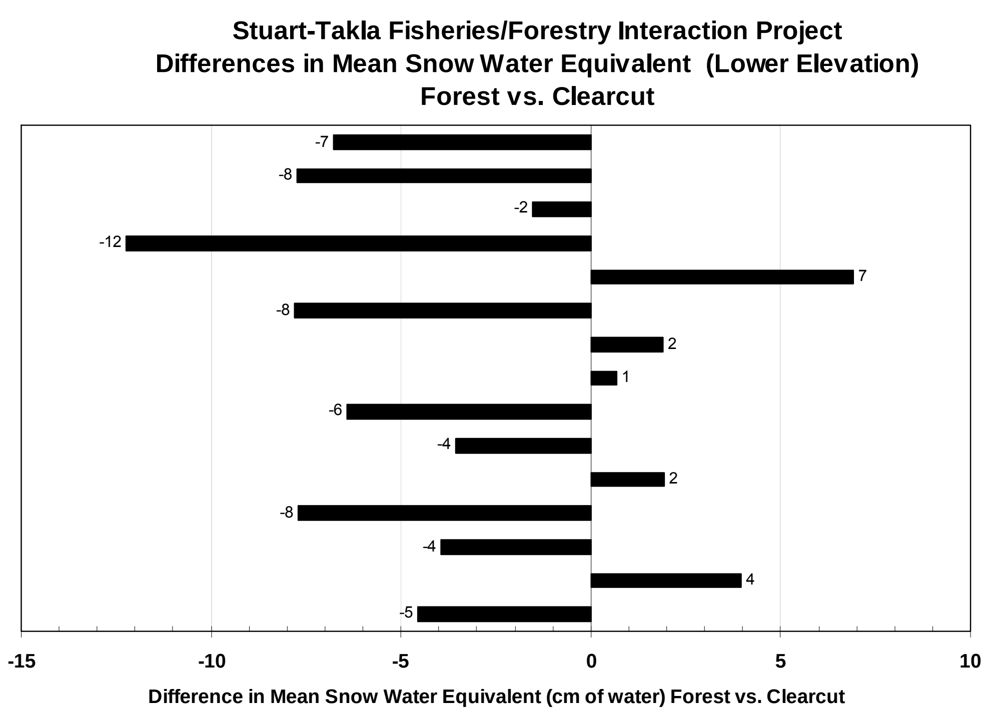

Fig. 4. Forest versus clearcut differences in mean snow water equivalent on individual sampling dates.

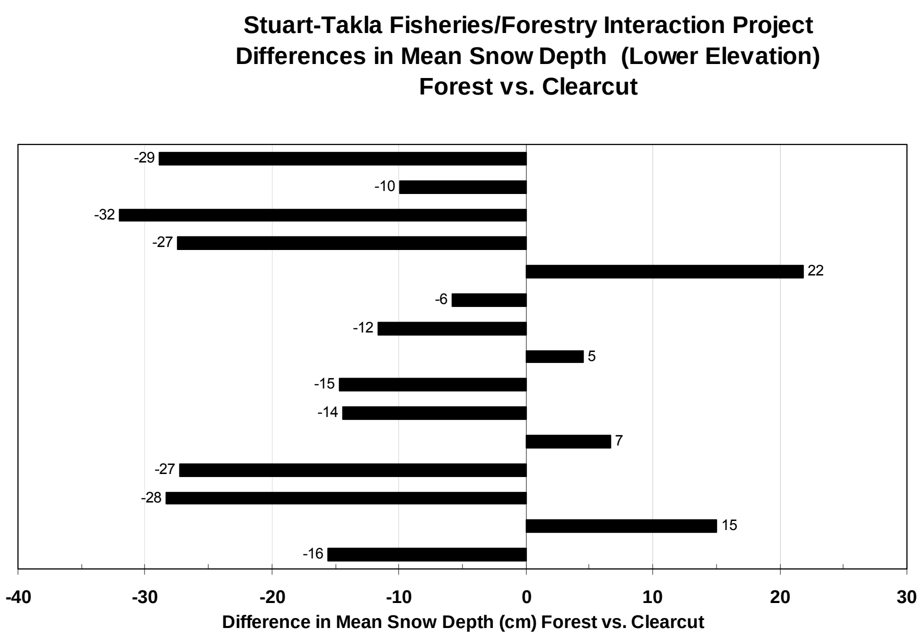

Fig. 5. Forest versus clearcut differences in mean snow depth on individual sampling dates.

PART II: SEDIMENT AND BEDLOAD STUDIES

Spawning Sockeye Salmon Contributions to Sediment Transport in Forfar Creek

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INTRODUCTION

Major sediment transport within streams is typically associated with high discharge events such as spring freshet and storms. Transport of bed material occurs when the hydrodynamic forces (tractive forces) acting upon an individual grain exceed the critical threshold maintaining that grain in place (Milner et al. 1981). Particles are carried downstream in the water column as suspended load, or saltate along the streambed as bedload. With increasing discharge, increasingly larger particles are moved and particles transported by saltation can become entrained into suspension (Knighton 1984; NCASI 1999). However, processes other than high discharge events may also contribute sediment transport mechanisms within streams.

There is growing recognition that salmonids are important contributors to their physical environment. For example, adult salmon returning to their natal streams to spawn and eventually die, playing a key role in nutrient supply to freshwater and terrestrial environments (Cederholme et al. 2000). Spawning salmonids also contribute to sediment transport processes. During spawning, female salmonids excavate pits in streambeds into which they deposit their eggs, the nests (redds) are then covered with gravel. Fine sediments are removed from redds by the digging action of spawning salmonids and are likely transported as suspended load (Kondolf et al. 1993; McCart 1969). Bedload movement facilitated by spawning salmonids has been documented by Gottesfeld (1998). He demonstrated, in a low gradient system, that spawning salmonids moved 50 mm clasts the same average distance as flood events (about 4 m). He estimated spawning salmonids can account for up to 11% of the total annual bedload transport of a stream. Although spawning salmonids may play an important role in bedload transport the relative amount and fate of smaller grained sediments (< 50 mm) displaced by salmonid redd digging has not been studied in detail.

As a component of the Stuart-Takla Fish-Forestry Interaction Project (see Macdonald 1994 and Macdonald and Herunter 1998) bedload data were collected over several years in a salmon spawning stream. This paper presents a preliminary analysis of a portion of those data. The analysis estimates the relative influence of biological and hydrological factors on sediment (<50 mm in diameter) transport in a gravel-bed stream. We used three events to represent different conditions in which sediment transport could occur: high discharge with no spawning activity (spring flood), low discharge with spawning activity, and low discharge with no spawning activity (winter). Total accumulation, accumulation rate, and composition of sediment moved were analysed for each of the three events.

MATERIALS AND METHODS

Data Collection

The bedload data used for analysis were collected from Forfar Creek, a tributary to Middle River with a catchment area of 37.4 km^2, at the northern extent of the Fraser River drainage basin. A sediment trap sampling station was established on a low-gradient riffle-glide section 200 m upstream from the creek mouth on July 19, 1991. At the sampling station, the creek was about 7 m bankfull width. Forfar Creek is typical of interior streams, which are hydrologically dominated by a freshet event in the spring (during snowmelt). Early Stuart sockeye spawn in the creek in July and August, with average female escapements past the DFO counting fence of 3400 (range 1189 to 6781) for the years 1991 to 1997 (unpublished DFO data).

Five 20-L plastic buckets lined with polyethylene bags were placed at 0.75 m intervals in a transect across the stream. Sampling slots of 0.10 x 0.05 m (July 19, 1991 to June 27, 1993) and 0.07 x 0.07 m (June 27, 1993 to September 1997) were cut into the bucket lids. The samplers were dug vertically into the gravel so that the top of the sampler was level with the streambed. This type of sampler was designed to trap particles saltating down the stream-bed.

Two types of data collection began in October of 1991 and continued through late September of 1997. First, the distance between accumulated bed materials and the sampler lid was recorded several times each month from April through September of each year to determine accumulation rates of trapped materials. Second, physical samples of trapped materials were taken after most flood, spawning, and winter events to determine grain size distribution (composition). The sampling buckets were reset for the next event by placing a new sampling bag in the sampler and replacing the lid.

DATA ANALYSIS

Sediment Accumulation

We converted distance measurements taken after each event for all samplers into depth of accumulated materials by subtracting the recorded field measurement from the depth of an empty sampler (0.36 m). The timing and duration of flood and winter events were calculated from hydrographs (unpublished DFO data), while spawning event duration was taken from sampler deployment date and field observations. Total accumulation and accumulation rate (total accumulation/event duration) were calculated for available samples in the sampling transect, and then averaged for the stream cross-section. Thus, average total accumulation and accumulation rate were obtained for each event each year.

One-way analysis of variance (ANOVA) tests (Minitab©) were performed on discharge, total accumulation, and accumulation rate data comparing flood, spawning, and winter events. Shapiro Wilk W tests (Minitab©) indicated that accumulation rate data was not normally distributed. Levene tests (Minitab©) indicated that the discharge, total accumulation, and accumulation rate data did not have equal variances. Discharge and total accumulation data was log transformed and accumulation rate data was natural log transformed. The transformed data met all ANOVA assumptions. The results of ANOVA on both the non-transformed and transformed data were the same in all cases; therefore, we only present the results of the non-transformed data.

Sediment Composition

All bedload samples were dry sieved and weighed at the DFO West Vancouver Laboratory; specific procedures are outlined in Scrivener and Macdonald (1998). The following sieve sizes were used in the dry weight analysis: 0.0 (pan), 0.075 mm, 0.3 mm, 1.18 mm, 2.36 mm, 9.5, and 25 mm. All samples were truncated at 50 mm, as this was the smallest trap opening for the samplers from 1991 to 1993. Percent composition was determined for each sample (size class weight/ total sample weight), then an average was developed for the whole sampling transect for each event. Data analysis was performed by plotting the average percent weight of each size class, pooling the events between years. Some exploratory compositional data analysis was performed using simple correspondence analysis (Minitab©) as a potential method to incorporate sample variability (Greenacre 1984; Lebart et al. 1984) which traditional methods such as cumulative percent graphs do not.

RESULTS AND DISCUSSION

Sediment Accumulation

Average event duration over the six years of data collection was 33, 20, and 213 days for flood, spawning, and winter, respectively (Table 1). Flood events occurred between early May and late June, spawning events occurred between mid-July and mid-August, and winter events occurred from late September to late April. Average discharge for each event was 2.94, 0.54, and 0.59 m^3/s for flood, spawning and winter events, respectively. An ANOVA of the discharge data indicated that the flood event had a significantly higher discharge ($F=57.72$, $p=<0.0001$, Tukey-Kramer Honestly Significant Difference [HSD] test) than spawning and winter discharges, and that there was no significant difference between spawning and winter discharges (Fig. 1a).

Average total accumulation was 26, 23, and 3 cm for flood, spawning and winter events, respectively (Table 1). ANOVA results indicated a significant difference in average accumulation ($F=10.28$, $p=0.0015$). An HSD test showed that flood and spawning events were similar while the winter event was different (Fig. 1b). Average accumulation rates were 1.02, 1.14, and 0.02 cm/day for flood, spawning and winter events, respectively. We detected a significant difference in mean accumulation rate of sediments between events ($F = 5.92$, $p = 0.0128$). An HSD test indicated that there was no significant difference between flood and spawning accumulation rates but the winter rate was significantly smaller (Fig. 1c). During flood and spawning events in 1993 and 1995, all samplers were saturated; therefore, total accumulation and accumulation rate for these four events underestimate the actual values.

Winter sediment accumulation within the trap samplers was relatively small, while accumulations were the same for flood and spawning events even though these two events had very different discharges. This indicates that sediment transport is not dependent solely on high discharge and that spawning salmon are an important component of sediment transport within Forfar Creek. In their summary, Sheridan et al. (1984) suggest that fine sediments are mobilized from the streambed by flood events and that spawning salmon also liberate fine sediments from the streambed. Gottesfeld (1998) calculated that, at a location near this study site, spawning sockeye accounted for 48% (3 year average) of the annual transport distance of clasts greater than 50 mm in size. Our results suggest that spawning sockeye salmon accounted for 44% of total accumulation within the sediment samplers during the three events measured. These results indicate that spawning events play a significant role in sediment transport. However, the contribution of spawning fish to bedload transport is likely only relevant to interior streams as coastal streams tend to have flood events on a frequency that would overwhelm fish driven transport events.

Sediment Composition

The average grain size distribution of the sediment within the samplers was similar for the hydrologically driven events (flood and winter) and relatively different for the spawning event (Fig. 2). Flood and winter events were dominated by the 0.3 – 1.18 mm size class, whereas the spawning samples were dominated by the 9.5 – 25 mm size class.

The correspondence analysis indicated similar results, but showed separation of all three events. Flood and winter events were more similar than spawning as indicated by their proximity to each other in Figure 3. In general, flood events were most associated with grain sizes from 0.0 to 0.75 and 0.3 to 1.18 mm, spawning with grain sizes 9.5 to 50 mm, and winter with grain sizes 0.75 to 0.3 mm. The 1993 spawning event (longest duration and frequent rain storms throughout), 1995 flood event (longest duration) and 1992 winter event (shortest duration) profiles were not consistent with the average composition for those events.

The lack of fine sediments (< 2.36 mm) sampled within the spawning event and the similarity between the winter and flood sample compositions were not expected results. We anticipated that flood and spawning events, given their similar accumulation rates, would have similar sediment composition. There are several possible explanations for these differences including sampler efficiency differences during different discharges and the possibility that spawning fish were actively selecting redd sites that only contained coarse material. Another probable explanation is that during spawning there were limited fine particles (< 2.36 mm) available for transport. Beaudry (1998), in his work on the same creek, suggested that fine material is supply-limited in this system. During flood (snowmelt) events he measured high suspended sediments on the rising limb of the hydrograph but measured significantly less suspended sediment during the falling limb or on subsequent discharge peaks. His results indicate that fine sediments may not have been available for transport during spawning.

Kondolf et al. (1993) suggest that excavated materials during redd building are differentially transported such that large gravels are locally re-deposited and finer materials are carried downstream. Cheong et al. (1995), using grab samples in Forfar Creek (1992 and 1993), found slight increases in suspended sediment concentration during spawning events, but they also suggest that fine sediments are in limited supply within this system. Correspondence analysis indicates that the 1993 spawning event was anomalous in that a finer sediment composition was present in the sediment traps than that found in spawning events of other years. Frequent summer storms during the 1993 spawning event may have liberated fine sediments from supply areas (upstream storage areas) thereby increasing fine sediment concentration in both suspended sediment and bedload trap samples. In subsequent years Beaudry (1998), using continuous recording equipment and Optical Back-scatter Sensors, did not detect significant increases in suspended sediments during the spawning event.

During spawning events fine sediments (< 50 mm) may not experience sufficient discharge (tractive force) to entrain them as bed or suspended load, however they may be re-worked within the streambed. The redd building process may liberate fine sediments which then settle immediately on the streambed surface; these re-worked sediments may then be available for transport during post spawning events (winter and flood). This interrelationship between biological and physical events is also supported by Gottesfeld (1998) who suggests that both spawning and flood events increase the efficiency of transport with each process contributing to the other through the “loosening” of bed materials.

In summary, sediment transport during spawning events was significantly greater than expected due to discharge conditions alone, and the composition of sediment sampled during spawning events differed from those obtained during winter and flood events. The correspondence analysis shows promise as an analytical tool for sediment composition analysis. It provides relational associations which can simplify complex data analysis.

Management Implications

High fine sediment concentrations within redds have adverse effects on salmonid egg and alevin survival (reviewed by Chapman 1988). Egg survival is positively related to permeability of the gravel matrix within redds, which is inversely related to percent fine composition (MacNeil and Ahnell 1964, Everest et al. 1987). From a fisheries management perspective, any change in egg-to-fry survival rates are important, and understanding more about the dynamics of fine sediment transport within streams may be necessary for assessing the effects of increased sediment input on salmonid production.

As resource management moves toward an ecosystem based approach, managers will need to consider the dynamics and influence of major components of an ecosystem. The National Council for Air and Stream Improvement recently published a bulletin on cumulative effects of sediments in watersheds without a single mention of salmonids affecting fine sediment concentrations (NCASI 1999). One application for the results presented in this paper, specific to fisheries management, is to take into account a spawner facilitated cleaning or re-working effect when setting escapement targets.

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Year	Event	Composition Sample Taken	Start Date	End Date	Event Duration (days)	Discharge (m^3/s)	Sediment Accumulation (cm)	Accumulation Rate (cm/day)
1992	Flood	,	14-May-92	17-Jun-92	34	2.70	13	0.39
1993	Flood	,	7-May-93	19-May-93	12*	3.49	31	2.57
1994	Flood	,	29-Apr-94	31-May-94	32	2.10	33	1.02
1995	Flood	,	27-Apr-95	7-Jun-95	41*	2.26	36	0.88
1996	Flood	,	21-May-96	25-Jun-96	35	3.25	14	0.39
1997	Flood		12-May-97	11-Jun-97	30	3.84	27	0.90
Average					**33**	**2.94**	**26**	**1.02**
1992	Spawning		22-Jul-92	24-Aug-92	33	0.19	21	0.63
1993	Spawning	,	5-Jul-93	12-Aug-93	38*	0.79	36	0.95
1994	Spawning	,	16-Jul-94	11-Aug-94	26	0.49	29	1.10
1995	Spawning	,	26-Jul-95	10-Aug-95	15*	0.30	36	2.40
1996	Spawning	,	1-Aug-96	12-Aug-96	11	0.99	16	1.42
1997	Spawning		31-Jul-97	10-Aug-97	10	0.48	4	0.35
Average					**20**	**0.54**	**23**	**1.14**
1992	Winter	,	6-Oct-91	26-Apr-92	203	0.65	3	0.02
1993	Winter	,	22-Sep-92	17-Apr-93	207	0.53	2	0.01
1994	Winter		16-Sep-93	29-Apr-94	225	0.58	1	0.00
1995	Winter	,	28-Sep-94	27-Apr-95	211	0.67	7	0.03
1996	Winter	,	27-Sep-95	27-Apr-96	213	0.68	2	0.01
1997	Winter		24-Sep-96	1-May-97	219	0.43	5	0.02
Average					**213**	**0.59**	**3**	**0.02**

Table 1. Bedload trap sampler data from Forfar Creek, B.C. (1992-1997) by event (flood, spawning, winter) including event duration, average discharge, total sediment accumulation, and sediment accumulation rate during the event. “*” denotes that samplers were saturated (completely full) when sampled. “,” denotes if a sample was collected for compositional (grain size) analysis.

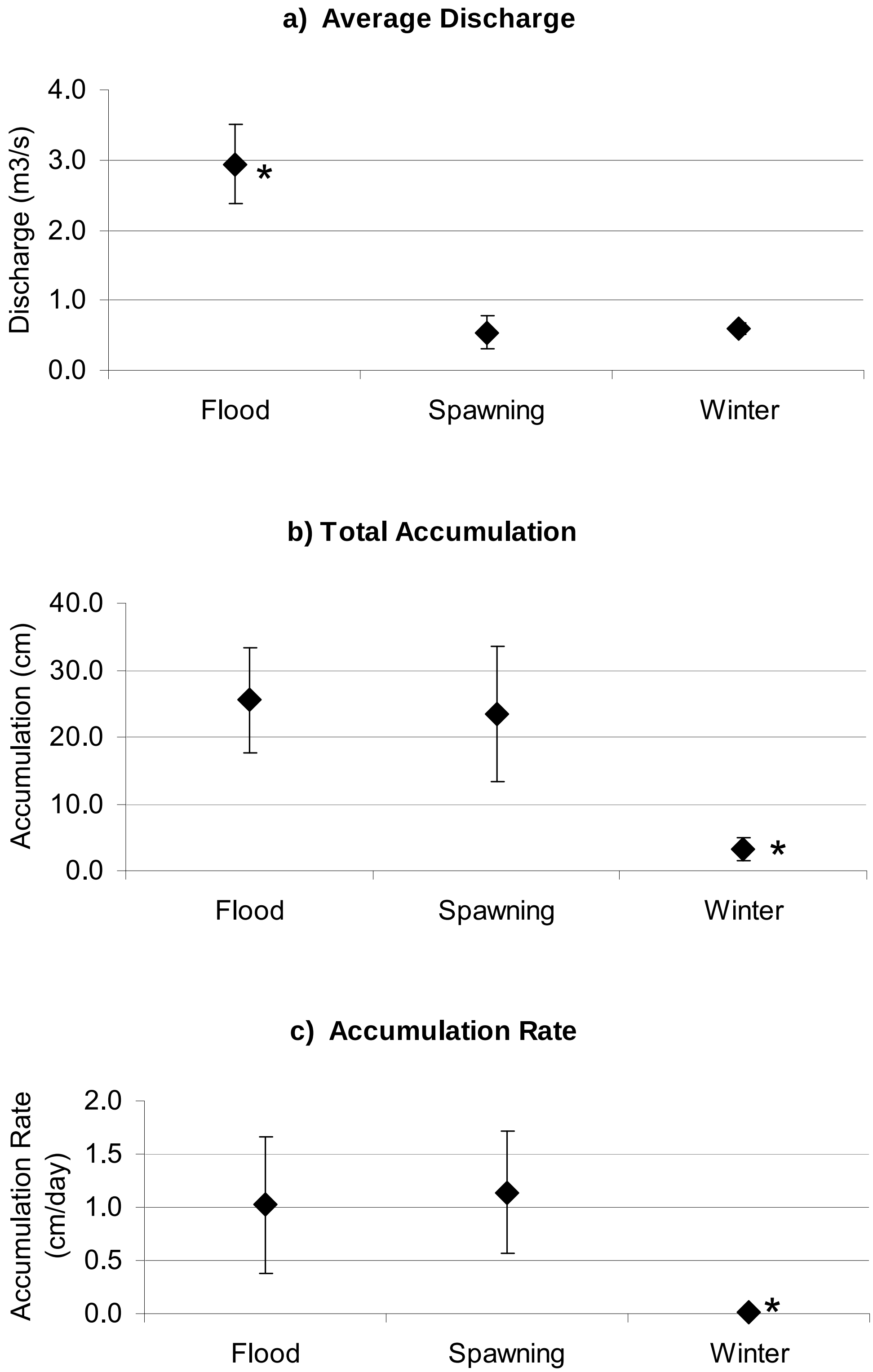

Fig. 1. Average creek discharge (a), total sediment trap accumulation (b), and sediment trap accumulation rate (c) during three events, flood, spawning and winter, on Forfar Creek, 1992 – 1997. Diamonds are means; error bars show 95% confidence interval. Significantly different means ($p<.05$) are indicated with a "*".

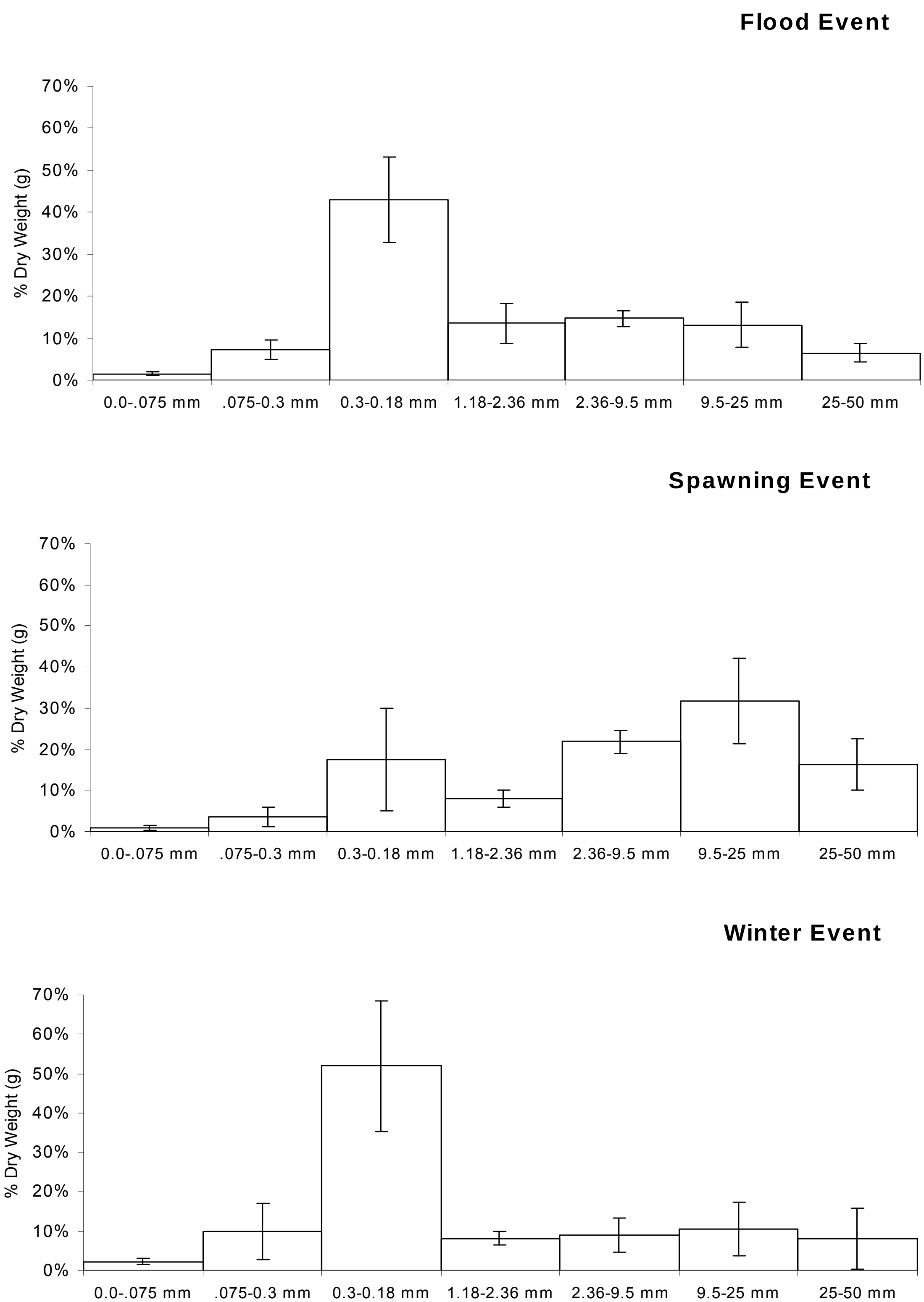

Fig. 2. Percent dry weight of each grain size class of material accumulated within sediment traps during spawning, flood, and winter events of Forfar Creek (1992 – 1997). Error bars show 95% confidence intervals.

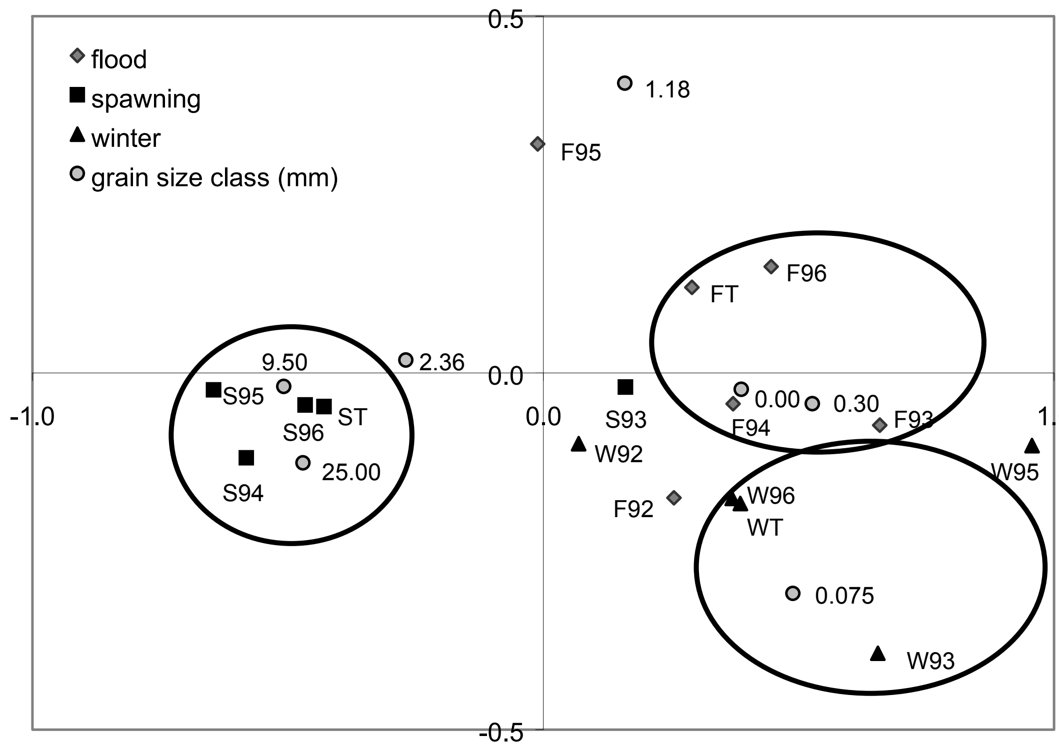

Fig. 3. Correspondence analysis diagram of event and grain size class. Circles denote clusters of events and grain size classes that show similarity. Events are denoted by year, T = average of all years. Grain size number denotes the lowest size in the class.

Application of Multi-Element Geochemical Methods to Trace Sediment from Forestry Activities: A Summary

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INTRODUCTION

Land disturbance associated with logging activities has the potential to alter sources and rates of sediment supply to streams both directly (erosion of stream channels and banks, and gullying from landings and roads) and indirectly (by increasing landslide activity and causing changes in hydrology). Increased sedimentation may be harmful to benthic invertebrates and spawning salmon, by creating anoxic conditions or burying salmonid eggs (Reiser 1998). In order to regulate and mitigate these effects, it would useful to have the ability to determine how much sediment is contributed by different sources, and to be able to track in-channel movement and storage of sediments from the different sources (Christie and Fletcher, *in prep*).

Multi-element stream sediment geochemical methods have previously been used to recognize and track contaminants from mining, industrial and urban sources (e.g., Ódor et. al. 1998). However, these are situations in which the contamination signal is usually much greater than the natural geochemical background. Newly eroded natural material, derived from similar parent material to the sediments already in the stream, is likely to have a subtler geochemical signature. Nevertheless, insofar as fluvial processes modify the texture, and hence mineralogy and geochemistry of material entering and moving along a stream channel, it seems likely that there will be geochemical differences between natural materials entering a stream channel and the composition of sediments already within the channel. Evidence for this can be found in Sirinawin et al (1987), Hou and Fletcher (1996), Fletcher and Loh (1997), and Fletcher and Muda (1999). This study therefore examines if changes in stream sediment geochemistry can be used to identify and track inputs of materials into stream channels as a result of forestry activities.

METHODS

Sample collection and identification of sediment sources

Fieldwork was undertaken in the Baptiste Creek drainage basin in August 1996 and August 1997; logging activities occurred from October 1996 to January 1997. Six small streams were sampled (Fig. 1), four of which were to be logged and two that were to remain unlogged as controls. In 1997 the same sites were re-sampled together with additional sites required to replace sites sacrificed to road construction. In both years, sampling involved field screening to obtain ~2 kg of -11 mm sediment from small, active gravel bars. The +11 mm and -11 mm fractions were weighed in the field, the width of the stream measured, and the site and stream bed photographed. Duplicate samples were taken at approximately twenty percent of sites to study in-site variability. Thirteen bulk (20 kg) samples were also collected for detailed size distribution-mineralogy studies (Christie and Fletcher 1999).

Six new sources of sediment were identified in 1997. These all developed as a result of land disturbance by road building and construction of stream crossings (Fig. 1). Culverts were installed at stream crossings and local fill material, till and bedrock was used to construct the road base. Ditches were constructed along roads and cross ditches were installed to collect sediment generated from runoff from the road surface and from fill slopes. From each “new” source, six to nine samples were collected of the material used to construct the crossings and from gullies draining from the road into the stream.

Sample preparation and analysis

Sample preparation for multi-element geochemical analysis involved wet sieving into five size fractions. The -212 μm size fraction was then submitted to a commercial laboratory for multi-element ICP analysis after a total decomposition with $HClO_4$-HF-HNO_3 and a strong acid attack with *aqua regia (*HCl-HNO_3).

Quality control

Analytical precision was monitored by random insertion of analytical and field duplicates. Also, roughly 24% of samples from 1996 were randomly selected for reanalysis in 1997. CANMET lake and stream sediment analytical standards (Lynch 1990) were inserted in analytical batches to monitor accuracy.

For most elements, accuracy is within +/- 20% of accepted values for both the *aqua regia* and total digestions. Laboratory precision, evaluated from scatterplots and Thompson-Howarth control graphs, is also within +/- 20% (at the 95% confidence level) for most elements with both decompositions in 1996 and 1997 (e.g., *aqua regia* in Table 1). Exceptions are elements such as Pb that are at or near their analytical detection limit. For samples re-analyzed over the two-year period laboratory precision decreases to ~ +/- 25%.

RESULTS

Stream sediments

Work in 1996 shows that prior to logging each stream had a distinct multi-element geochemical signature related to drainage basin geology (Fig. 2). Year-to-year variations in these trends were evaluated by re-sampling in 1997, the 1996 control sites undisturbed by forestry activities (Column 7, Table 1). The variability (= within-site year-to-year variability + analytical variability between and within years + random variability) from 1996 to 1997 is typically about ± 20-30% (Table 1) and is only slightly greater than the total variation for combined within-site and laboratory variability in a single year. The natural, multi-element geochemical signature of undisturbed stream sediments thus remained reasonably constant between the two years and changes associated with new sediment sources, described below, are much greater than the natural variability of the undisturbed streams (Christie and Fletcher, in prep.)

New sediment sources and their effects on sediment composition

No overall effect of logging on stream sediment geochemistry was apparent within or downstream of the cut-blocks - probably because unlogged buffer zones along the stream channels prevent the development of new sediment sources. However, construction of stream crossings and roadside ditches created new sources of sediment supply to the streams. Comparison of sediments and new after-logging sediment sources clearly shows that there are differences in geochemical composition between them (Table 2). These differences are also shown in profiles of element concentrations along streams - e.g., stream B4 (Figs. 3 and 4) and stream B3 (Fig. 5). In each of these profiles both the sediment source and the 1997 sediments downstream from the source have higher concentrations of Ni, Co and Mg than the pre-disturbance 1996 sediments. Concentrations of Ni, Co and Mg increase by as much as 110 ppm, 6 ppm and 1%, respectively for approximately 120m on stream B4, and 50 ppm, 20 ppm and 0.5%, respectively, for approximately 170m on B3. On stream B1, where the road crossing the stream was constructed five years earlier, the profile shows an increase in Ca, K and Mg concentrations in sediments over 700 m below the source (Fig. 6), with the peak concentration shifting downstream from 1996 to 1997 (Christie and Fletcher, in prep.)

In addition to geochemical differences between sediment and the new sediment sources, there are also significant differences in texture between them. Sources were found to contain up to thirteen times more fines (-0.212 μm) than the sediments (Table 2). Despite this difference the stream profiles show that there are no significant changes in abundance of fine sediment, downstream from the sediment sources, from 1996 to 1997 (Figs. 4,5 and 6) (Christie and Fletcher, in prep.).

Although the new sediment sources do not have elevated concentrations of zinc, there are strongly anomalous concentrations of zinc (>200 ppm) below new sources of sediment (Figs. 7a & b). Examination of the 1996 data shows a similar increase in zinc concentrations below roads constructed across streams B1, B2, B3 and B4 in 1992. The most likely source of these anomalies is zinc abraded from the surface of newly installed culverts during energetic transport of coarse bedload by flood events (or possibly from zinc being dissolved from the surface of the galvanized culvert and then precipitated or absorbed on sediments below the culvert) (Christie and Fletcher 1999).

DISCUSSION

The sediment inputs associated with road construction were found to create local, but distinct, increases in concentrations of elements in sediments below the source (Figs. 3,4,5, and 6). These geochemical changes occur even though the parent material (i.e., local glacial till) for the undisturbed sediments is probably geochemically the same or very similar to the new sediments sources. This occurs because the texture and mineralogy, and hence the geochemistry of the undisturbed stream sediment will have evolved and diverged from that of its parent material as fluvial processes eliminate fine sediment from the stream bed and concentrate heavy minerals (Sirinawin et al

1987; Hou and Fletcher 1996; Fletcher and Loh, 1997; Christie 1998; Fletcher and Muda 1999, Christie and Fletcher, in prep.).

In this study, input of material from new sediment sources increases element concentrations in the stream sediments downstream from the source because relatively unweathered glacial till, with an abundance of fine material and relatively high element concentrations, is being added to the sediments. The opposite effect (i.e., dilution of sediment values downstream from sediment sources) was reported by Fletcher and Muda (1999) in rain forest streams in Malaysia and attributed to the input of strongly leached tropical regolith into the stream by mass wasting events. More complex relations between sediment geochemistry and input of material from landslide activity were reported in southern British Columbia by Hou and Fletcher (1996). These studies have shown that geochemical methods can be used to trace “new” sediment to streams; however, the resulting patterns will vary in different geological environments and in response to different magnitude of disturbances

The potential for an element to act as sediment tracer can be assessed by assuming that the greater the compositional difference between the stream sediments and the new source, the greater the possibility of the new source changing the composition of the sediments. Elements which might be useful tracers can thus be selected in two ways: (i) from the geochemical contrast (ratio) between the concentration of the element within a sediment source and stream sediments taken above the source (Table 3) and, (ii) by testing mean values of the sediment and source for significant differences (Table 4). In this study, these tests both indicate that, for at least one of these streams, the elements Ba, Ca, Cr, Fe, Mn, Ni, P, Sr, and Ti are all potential tracers. The elements that showed the clearest influence of the influx of new material on sediment composition (i.e. the elements in Figs. 3, 4, and 5) were all identified by one or both of these tests in the data from the individual streams.

After establishing that sediment can be traced in-channel, dilution equations (Peart and Walling 1986; Hou and Fletcher 1996) can be used to estimate the quantity of material derived from the new sources of sediment and incorporated in the –212 μm size fraction at varying distances downstream:

$$P_s = 100 \times (C_2 - C_1) / (C_{ss} - C_1) \qquad \text{(For } C_1 < C_{ss}\text{)}$$
$$P_s = 100 \times \{1 - (C_2 - C_{ss}) / (C_1 - C_{ss}) \qquad \text{(For } C_1 > C_{ss}\text{)}$$

Where:

P_s = Percentage of sediment from sediment source
C_{ss} = Concentration representative of sediment source (mean or maximum).
C_2 = Concentration at a site downstream of the sediment source in 1997
C_1 = Concentration of the stream at the same site as C_2 before Introduction of sediment.

For example, concentrations of Mg and Ni increase in sediments from stream B4 downstream from Source 4 (Fig. 4). Based on calculations for both elements the amount of material added to the sediments varies from ~ roughly 50% close to the source to <13% 117 m downstream (Table 5). Similarly, on B3 dilution calculations using Co, Mg and Ni indicates that ~30% of new sediment 80 m downstream from Source 5 decreases to ~2% at 190 m.

Although the dispersion trains and dilution calculations clearly indicate that material from the new sediment sources has been incorporated into the stream sediment there are no visible or measured changes in sediment texture commensurate with the geochemical changes immediately downstream from the sediment sources (Figs. 4, 5 and 6). A sediment exchange process must therefore take place whereby sediment from the new sources is exchanged for sediment already stored within the stream bed. There are at least two mechanisms for trapping fines in gravel bed streams: infiltration as fine suspended sediment infiltrates the bed (Beschta and Jackson 1979); and deposition and trapping of fines as the gravel framework is re-established on declining flood peaks. With infiltration, source sediment in suspension is unlikely to replace older sediment already trapped. However, during flood events, such as the spring freshet when the framework of the bed is disturbed, fines are remobilized and removed from the streambed. Fine sediment is then re-deposited and becomes trapped in voids in the bed as flood peaks decline (Day and Fletcher 1991). Bank erosion is also most prevalent during high precipitation and flood discharge events. Sediment from new sources is therefore most likely to account for a large percentage of the sediment deposited in the streambed during the declining flows that follow flood events.

With respect to fisheries issues, it is important to note that although the geochemical fingerprint clearly shows that sediment from the new sources has been incorporated in the bed, there has been no corresponding increase in the proportion of fines (-212 μm) in the bed. Creation of new sediment sources need not, therefore, result in further infilling of the voids in gravel bed streams. Also, the 700 m displacement of the geochemical peaks on stream B1, where the road crossing the stream was constructed five years earlier, and the further downstream shift of these peaks from 1996 to 1997 suggests that the increased sediment supply as a result of culvert installation stops, or at least decreases, within a year of construction.

In this study sediment samples were collected before and after forestry disturbance. However, for routine purposes sampling a suspected source together with sediments upstream and downstream from the source should be sufficient to verify the source, track movement of new sediment along the stream channel, and provide insights into fluvial processes. The applicability of this technique to trace sediment, without the necessity for a pre-disturbance database, and to detect inputs of sediment where there are no associated changes in sediment texture, should make it particularly useful for environmental monitoring and regulation agencies dealing with disturbance after-the-fact.

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

Results show that new sediment sources, in this case related to construction of logging roads and installation of culverts, can systematically change the multi-element geochemical fingerprint of stream sediments. This occurs even though the original stream sediment and the new sediment source are derived from the same or similar parent materials. The geochemical fingerprint of new sediment sources can therefore be used to track movement of sediment from the sources and to estimate the proportion of new sediment incorporated in the stream bed. Geochemical monitoring over several years can also provide information on ongoing supply or changes in sediment supply with time.

ACKNOWLEDGEMENTS

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Table 1. Summary of field sampling and analytical (aqua regia digestion) precision in 1996 and 1997 estimated by the method of Thompson and Howarth at the 95 % confidence level. (From Christie and Fletcher, in prep.).

Element	Detection limit	Precision (%)					
		1996		1997		96-97**	96-97***
		Field	Lab	Field	Lab	Field	Lab
Al	0.01%	25	15	25	20	20	25
As	2 ppm	>50	>50	>50	>50	>50	30
Ba	10 ppm	25	15	25	25	25	25
Ca	0.01%	25	15	25	30	25	25
Co	1 ppm	20	10	20	20	20	15
Cr	1 ppm	20	15	20	15	25	25
Cu	1 ppm	>50	15	>50	20	>50	25
Fe	0.01%	15	10	15	15	15	10
K	0.10%	30	15	35	30	30	30
Mg	0.01%	15	10	15	20	20	15
Mn	5 ppm	50	10	35	20	40	15
Ni	1 ppm	25	15	20	20	25	25
P	10 ppm	15	10	15	10	20	10
Pb	2 ppm	>50	>50	>50	>50		
Sr	1 ppm	25	20	35	35	30	30
Ti	0.01%	30	30	40	30	35	30
V	1 ppm	25	20	25	20	30	30
Zn	2 ppm	30	10	25	10	25	15

** = Precision estimated for 1996 fieldsites resampled in 1997
*** = Precision for 1996 samples reanalyzed in 1997

Table 2. Average geochemistry from aqua regia digestions and grain size distribution of stream sediments from B1 to B6 and sediment sources SS1 to SS6. Geochemical results for stream sediments and sources for 1997 data. (From Christie and Fletcher, in prep.).

		Stream B2		Stream B3		Stream B4		Stream B5			
Element		Sediment	SS6	Sediment	SS5	Sediment	SS4	Sediment	SS1	SS2	SS3
		Elemental Concentrations									
		n=6	n=11	n=6	n=7	n=6	n=4	n=12	n=7	n=6	n=8
Al	%	2.01	1.72	1.75	2.00	1.64	1.82	2.32	2.12	1.69	2.12
Fe	%	3.54	3.22	3.30	3.67	3.39	3.61	3.56	3.66	3.27	3.66
Mg	%	1.41	1.21	1.13	1.48	1.11	1.72	1.01	0.93	0.94	0.93
As	ppm	64	13	12	17	9	12	8	23	8	23
Ba	ppm	152	109	127	127	112	128	171	130	110	130
Co	ppm	20	18	20	21	18	20	17	16	15	16
Cr	ppm	148	113	111	130	106	139	74	69	64	69
Cu	ppm	98	61	90	58	68	76	143	80	67	80
Mn	ppm	3010	673	1589	887	1313	750	2295	981	706	981
Ni	ppm	323	112	121	138	105	166	87	68	59	68
Sr	ppm	47	35	33	37	37	38	79	46	40	7
V	ppm	57	56	61	61	59	59	56	61	58	61
Zn	ppm	109	86	99	92	116	100	195	153	82	153
Size Fraction		Grain Size distribution data									
		Percentage of sediment finer (%)									
(mm)		n=19	n=11	n=26	n=7	n=18	n=4	n=20	n=7	n=6	n=8
> 11	%	17.2	8.4	25.2	37.1	29.5	13.5	32.6	0.0	0.0	0.0
2 - 11	%	48.0	34.7	40.8	21.3	38.1	29.2	43.3	33.8	33.3	22.3
2 - 0.85	%	13.6	7.7	11.6	7.7	14.4	11.0	11.8	7.1	8.5	9.3
0.85 - 0.425	%	8.1	5.5	8.8	6.1	8.2	9.0	6.6	5.6	8.4	10.5
0.425 - 0.212	%	5.9	4.9	5.7	4.8	4.7	7.3	2.9	5.8	9.5	12.6
< 0.212	%	7.3	38.9	7.9	23.0	5.0	30.1	2.9	47.7	40.2	45.3

Table 3. Geochemical contrast: ratio of the mean concentration of an element within a sediment source to the mean concentration in sediments upstream from the source. Minus 212 micron fraction and aqua regia digestion. Only ratios <= 0.9 and >= 1.1 are shown. (From Christie and Fletcher, in prep.).

	Ratio			
	SS6 / B2	SS5 / B3	SS4 / B4	SS1 / B5
Aqua Regia				
Al	0.9	1.14	1.11	
As	**0.2**	**1.42**	**1.29**	**3.38**
Ba	**0.7**		1.14	0.83
Ca	**0.7**	0.89		**0.51**
Co	0.9		1.12	
Cr	**0.8**	1.17	**1.31**	
Fe	0.9	1.11		
K	1.1	**1.26**	1.16	
Mg	0.9	1.31	**1.55**	
Mn	**0.2**	**0.56**	**0.57**	**0.36**
Ni	**0.3**	1.14	**1.58**	
P	0.9	1.15	**1.28**	0.84
Sr	**0.7**	1.11		**2.05**
Ti	**1.2**	1.14		**0.60**
V				**1.51**
Zn	**0.8**		0.86	1.12
C	**0.3**	**0.39**	**0.72**	1.18

Table 4. T-tests comparison of element concentrations in sediment sources versus sediments upstream of sources. Minus 212 micron fraction and aqua-regia digestion. Only values significant at the 0.05 level are shown. (From Christie and Fletcher, in prep.).

	Stream			
	SS6 - B2	SS5 - B3	SS4 - B4	SS1 - B5-1
Aqua Regia				
Al	3.3			
As	5.39			
Ba	5.9		-2.55	
Ca	7.39			4.05
Co				
Cr	2.58	-2.41		
Fe	2.35			
K				
Mg		-2.94		
Mn	6.5	3.57	2.58	
Ni	5.13			
P	2.42	-3.38	-2.66	
Sr	4.06			2.91
Ti	-2.7			-3.59
Zn	5.62			
C	8.71			3.1

Table 5. Percentage of new sediment downstream from source SS4 on stream B4 based on Mg and Ni concentrations, and from source SS5 on stream B3 based on Co, Mg, and Ni concentrations. The mean amount of sediment is calculated using the mean concentration of the sediment source, while the minimum amount of sediment contributed is calculated using the maximum concentration of the sediment source. (From Christie and Fletcher, in prep.).

	New sediment (%) downstream from Source SS4.			
Distance (m):	13	43	87	117
Mean amount of sediment contributed				
Mg	49	46	23	9
Ni	54	35	25	13
Minimum amount of sediment contributed				
Mg	15	16	8	3
Ni	19	11	9	5

	New sediment (%) downstream from Source SS5.			
Distance (m):	82	106	167	191
Minimum amount of sediment contributed				
Co	37	37	16	5
Mg	29	11	21	3
Ni	29	19	8	-1

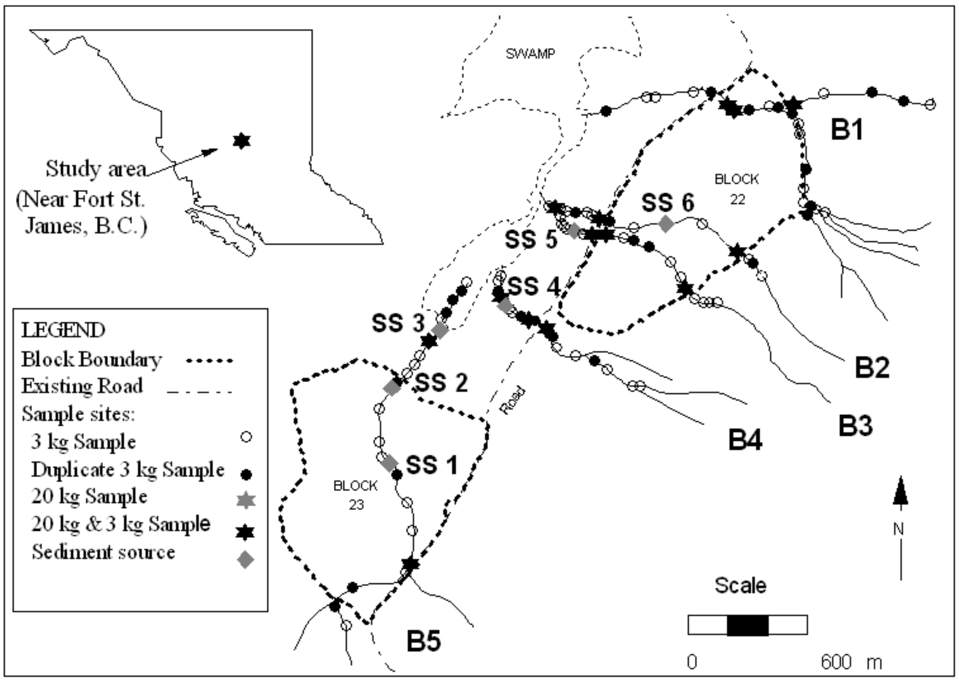

Fig. 1. Location map, sample locations (1997), streams B1 to B5 (B6 is approximately 1 km off the map to the northeast) and sediment sources SS1 to SS6. From Christie and Fletcher,1999.

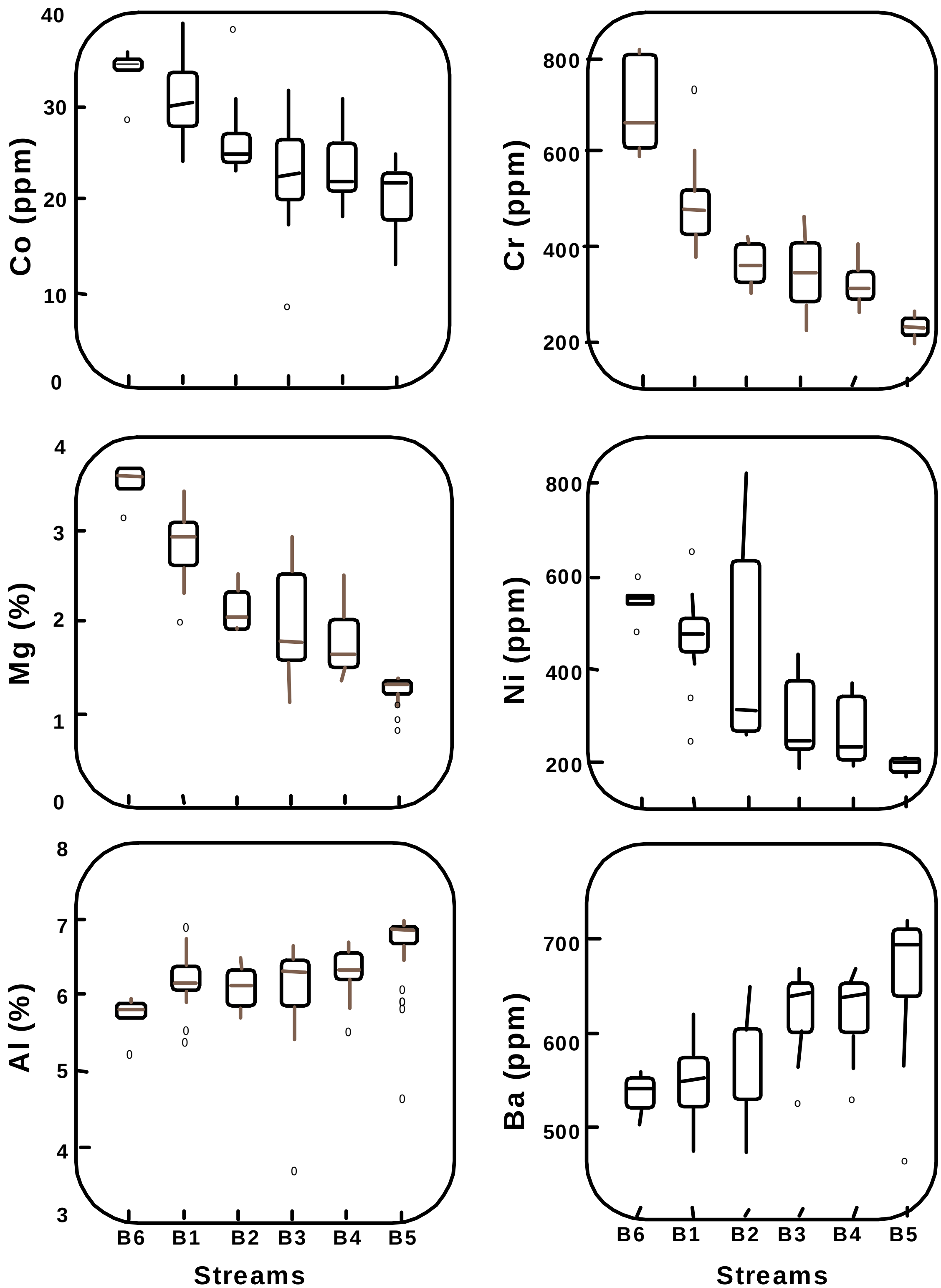

Fig. 2. Geochemical trends showing the decrease of Co, Cr, Mg and Ni from stream B6 in the northeast to stream B5 in the southwest and the corresponding increase in Al and Ba.

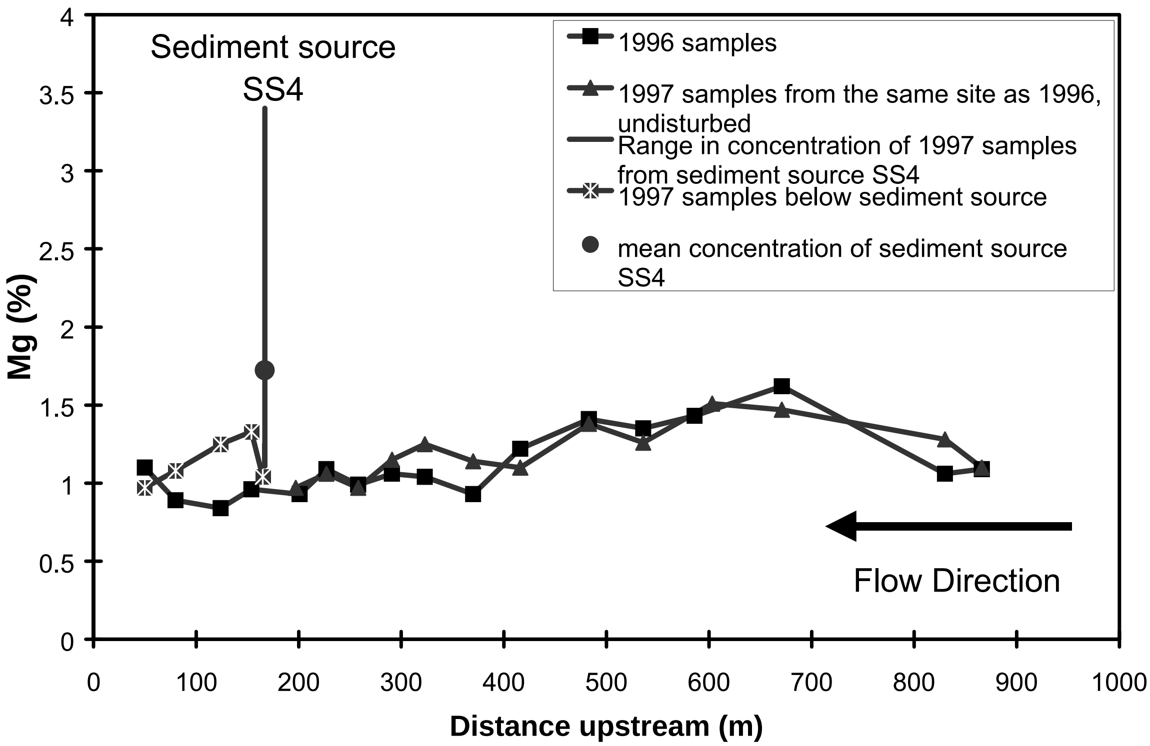

Fig. 3: Distribution of Mg along stream B4 in 1996 and 1997 in relation to sediment source SS4. The vertical line indicates the position of sediment source SS4 and its range and mean element concentration. From Christie and Fletcher (1999).

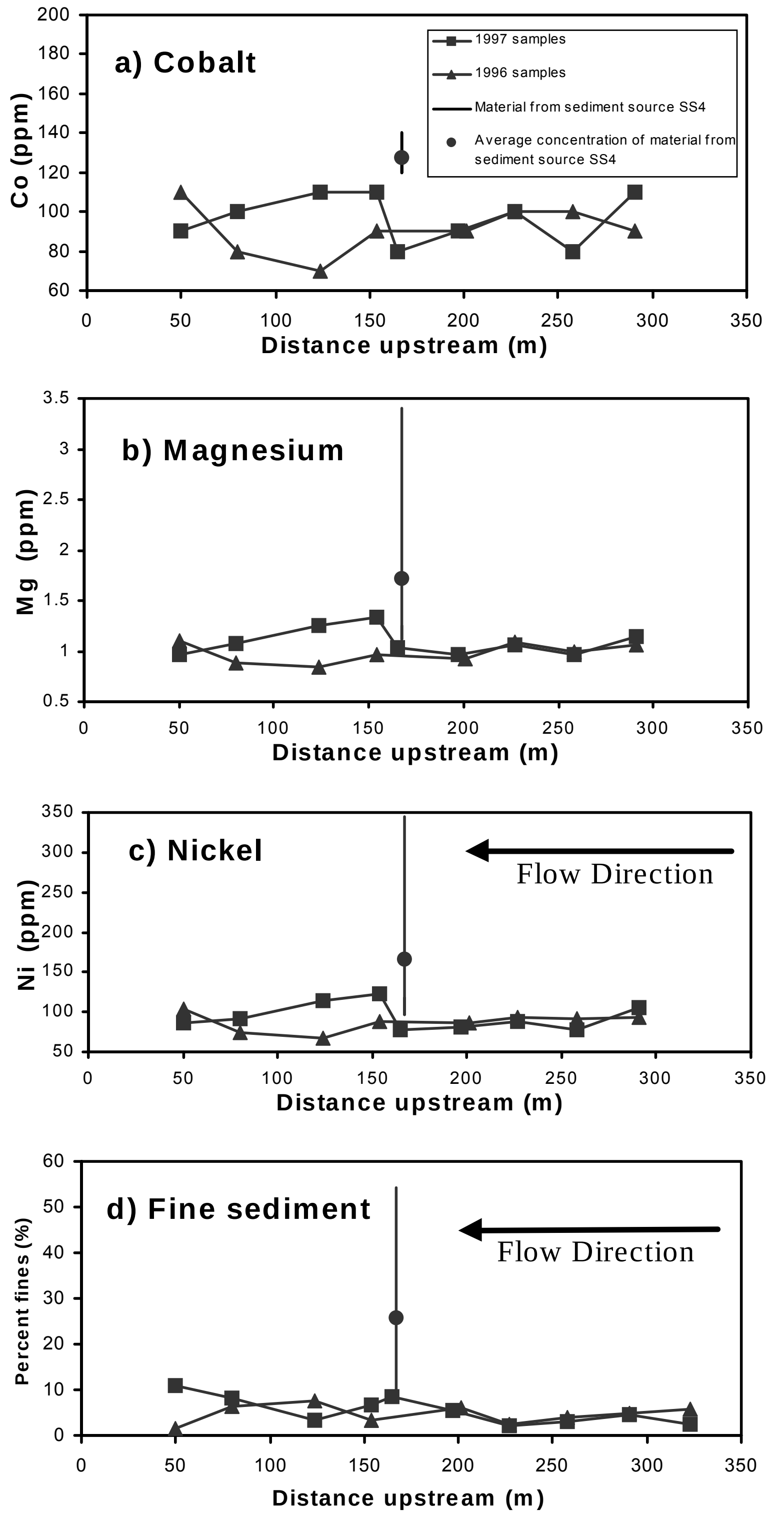

Fig. 4. Distribution of Co, Mg and Ni (all *aqua regia* digestion) and –0.212 mm sediment along stream B4 in 1996 and 1997 in the vicinity of sediment source SS4. The vertical line indicates the position of SS4 and its range and mean composition.

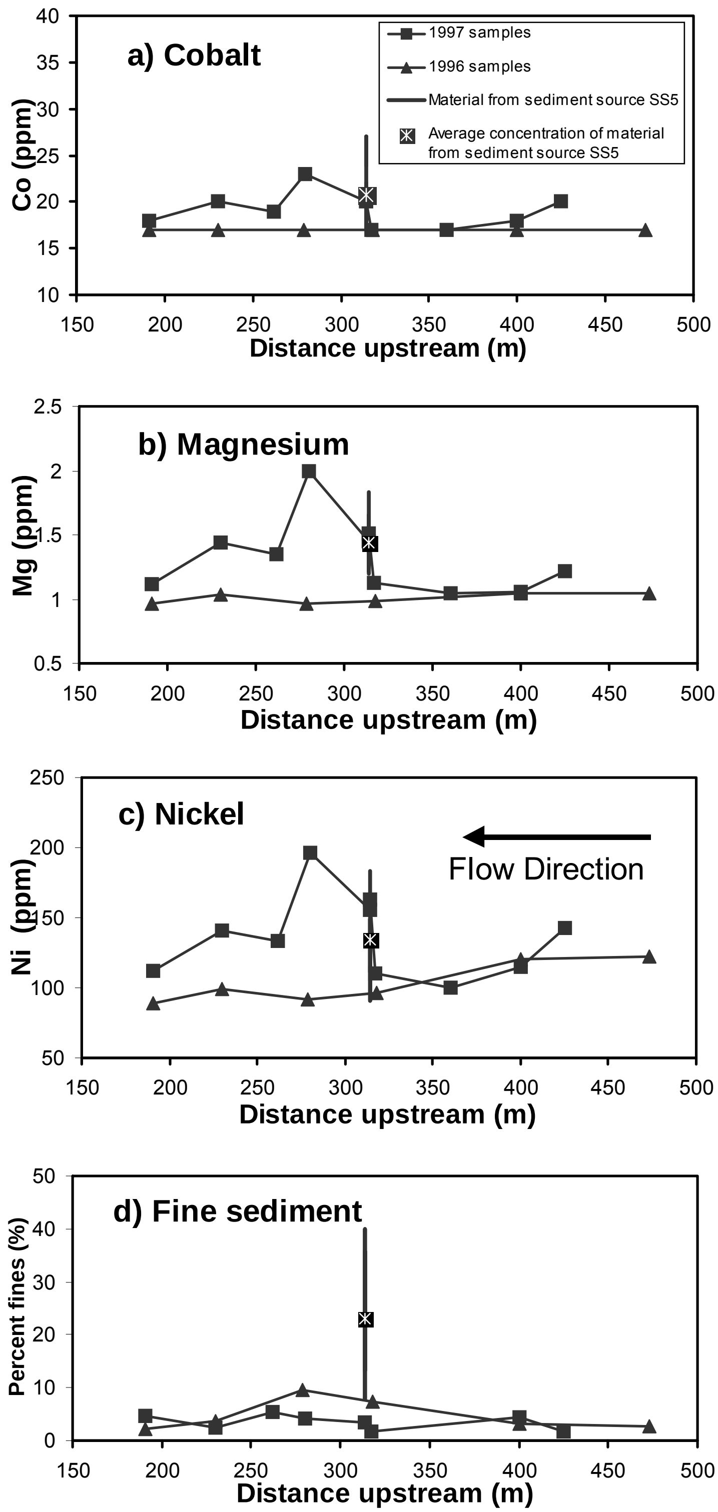

Fig. 5. Distribution of Co, Mg and Ni (all *aqua regia* digestion) and –0.212 mm sediment along stream B3 in 1996 and 1997 in the vicinity of sediment source SS5. The vertical line indicates the position of SS5 and its range and mean composition.

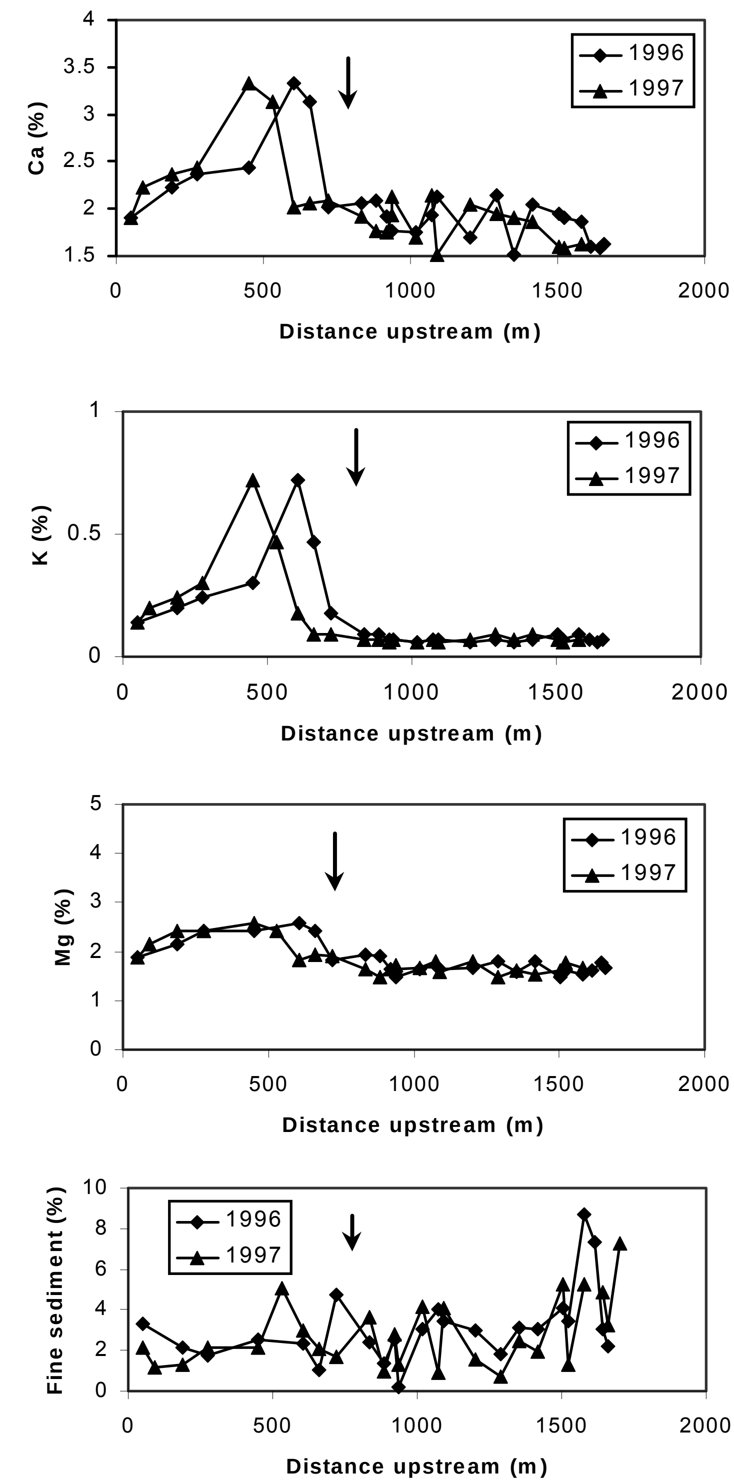

Fig. 6. Apparent shift in geochemical patterns for Ca, K and Mg (all *aqua regia* digestion) from 1996 to 1997 in stream B1. Arrow indicates the position of the stream crossing constructed in 1992.

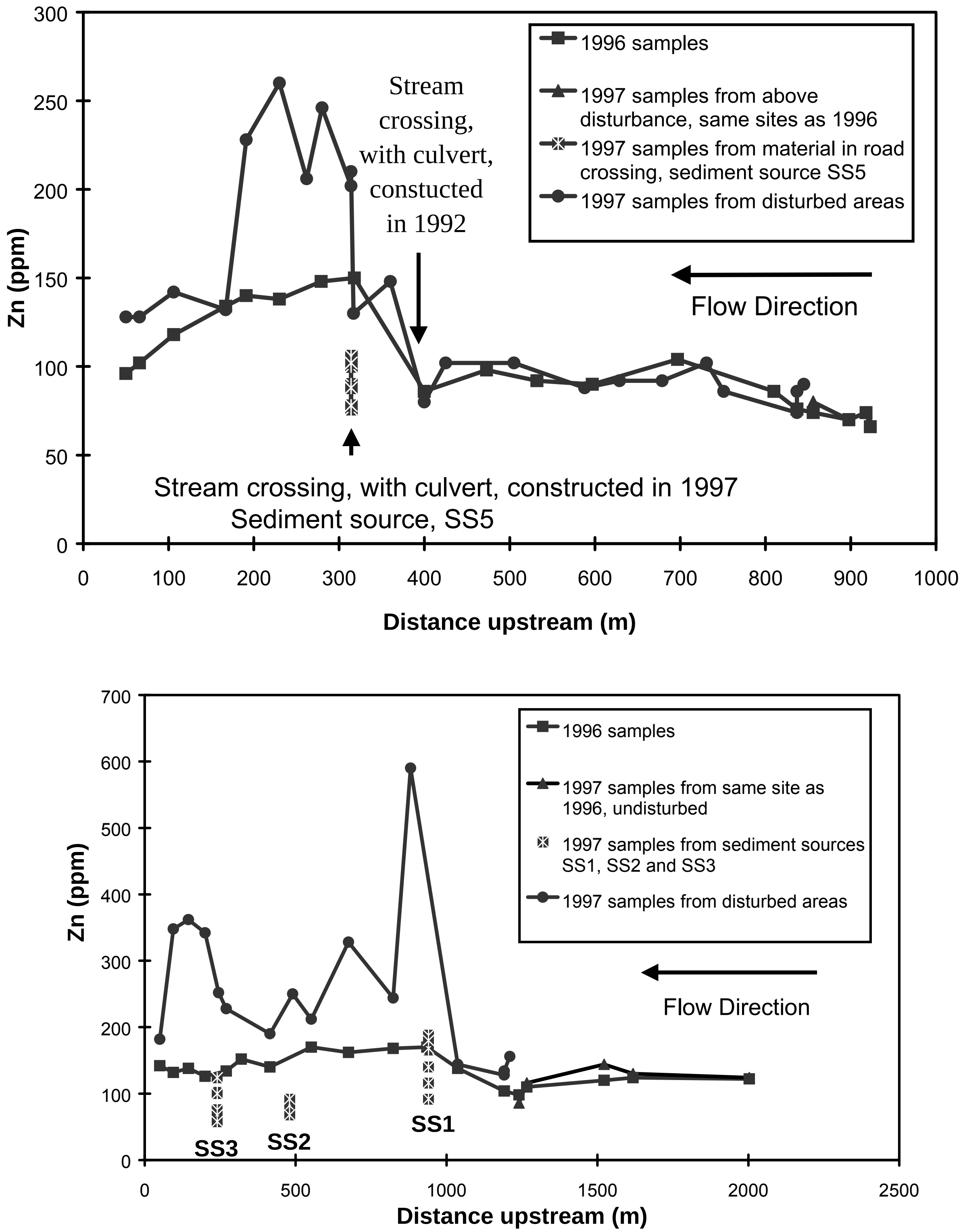

Fig. 7. (a) Distribution of Zn along stream B5 in 1996 and 1997 in relation to SS1, SS2, and SS3. (b) Distribution of Zn along stream B3 in 1996 and 1997 in relation to stream crossings and culverts installed in 1992 and in 1997 at SS5 (Christie and Fletcher,1999).

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Variations in Bedload Transport during a Spring Flood: O'ne-Ell Creek, Stuart Takla Experimental Watersheds

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INTRODUCTION

Water discharge in the Stuart -Takla creeks is strongly dominated by spring snow melt flooding in late May or early June, when approximately six months of stored precipitation is delivered over the course of a few weeks. During the high flows of the nival flood, coarse sediment is entrained and moved along the creek bed. This event constitutes the bulk of annual bedload movement; however, some bed material is also moved by sockeye spawning activities and occasional brief summer storm flows.

Bed load consists of particles that are transported by rolling, sliding or bouncing along the stream bed. In the relatively high gradient mountain streams of the Stuart – Takla region, bedload consists of material coarser than 1 mm. In peak flow events, boulders over 20 cm in length may be transported. This bedload material makes up the bulk of sediment in motion and forms the channel morphology. Bedload transport is a critical component of channel development and equilibrium, yet it is probably the most difficult component of the sediment budget to observe directly and quantify.

The present study was directed towards obtaining a continuous record of sediment flux across a stream cross section during the spring flood event. We used the UNBC Bedload Movement Detector (BMD), which was installed across the width of O'Ne-ell Creek (also known as Kynoch Creek), in the fall of 1997. The device has recorded detailed information on the timing and quantity of clasts passing through a riffle reach over the course of two nival floods (1998 and 1999). Because of poor salmon runs in these two years we have not yet recorded the bedload transport activity of spawning salmon.

The BMD was designed collaboratively in the Physics and Natural Resource Science Departments at the University of Northern British Columbia. It consists of 82 magnetic sensors mounted within a hollow aluminum beam placed across the channel width and buried flush with the streambed surface. During flood events, voltage signals from the sensors are collected by a data acquisition system and recorded by a personal computer. The resultant data records from the device show the passage of naturally magnetic stream bed material over the active width of the cross section at a very high temporal resolution. These data provide a unique insight into the dynamics of coarse sediment transport.

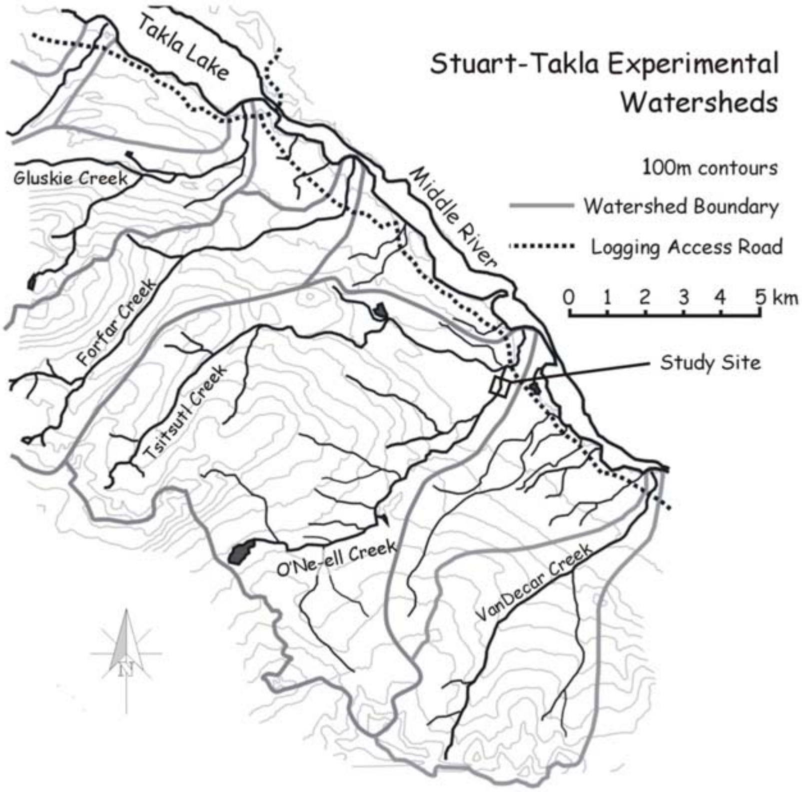

Fig. 1. Location of the bedload movement detector in the O'Ne-ell Creek watershed.

The project site is approximately 2 km upstream of the mouth of O’Ne-ell Creek at Middle River (Fig. 1). The present work discusses the design and function of the BMD, and some of the patterns of transport observed over the course of the 1999 nival event.

THE NATURE OF BEDLOAD TRANSPORT

There are numerous factors that affect the rates of coarse sediment transfer. Rates of transport are influenced by the water discharge, the slope and surface texture of the streambed, as well as the development of bedforms and bars, and interactions with obstacles such as coarse woody debris. The motion of individual grains tends to occur as a series of short intermittent bursts (Reid and Frostick 1994). As shear stress over the bed increases during a flood, more material is set in motion, and grains begin to move in clusters, propelled by inter-granular collisions and turbulent forces within a channel. As material becomes fully mobilized, transport becomes unsteady, moving in ‘pulses’, rather than as a steady discharge (Gomez et al. 1989).

The mobility of sediment in a gravel bed river is related to the particle size distribution of the bed material and their packing arrangement (Ashworth and Ferguson 1989; Wilcock and Southard 1989). The relative abundance of coarser or finer grains will dictate the overall mobility of bed material. Stream beds in larger stream systems may develop an armour layer, after low and intermediate magnitude flows selectively transport finer material, leaving a residue of well-sorted, coarse immobile particles

(Andrews and Parker 1987). Armoured stream beds are characterized by surface material with grain diameters up to two or three times those of the underlying material. Subtle rearrangement of the particles increases the interlock between neighbouring grains (Parker et al. 1982; Andrews 1983). The gravel in these stream sections will often tend to resist motion until the shear force of the flow is able to entrain the larger surface material, and the armour layer is broken. This usually requires an extreme flood. Once the packing arrangement is disturbed, all sizes become available for transport.

Although creeks in the Stuart -Takla watersheds may exhibit some characteristics of armoured stream beds, they do not entirely fit the general model described above. Imbrication and obstacle clast structures (Reid and Frostick 1984) do occur in the creek, but the effect is not as pronounced as in other, larger systems. The pattern of downstream fining in the creeks indicates that there is some size-selective entrainment at work. Studies by Gottesfeld (1998; 2000, see this volume) and Poirier (2000, *in preparation*) have shown that a substantial percentage of the bed surface is disturbed during annual flood events. It has been suggested that the regular disruption of the bed material by spawning salmon may contribute to an ‘overloose’ gravel fabric (Gottesfeld, personal communication). Disturbed material will generally have a lower threshold of motion.

Bedload transport varies in character depending on the flood stage and the nature of the bed material. Some authors divide up bedload transport in alluvial streams into two phases (Emmett 1976; Jackson and Beschta 1982). The first phase is the mobilization of smaller, loose material that is freely available for transport. Sand and fines are selectively carried from bars, channel margins, and pools. Gravel-sized particles, especially smaller sized ones, rotate out of the pockets in which they lie, and roll or bounce downstream until they settle into a new position. Andrews (1984), points out that a significant portion of material may be transported during this first, marginal phase.

With higher flows, a second phase of transport begins: the armour layer is disrupted. As the whole bed becomes mobile, the size distribution of the bed load coarsens. The velocities in pool sections exceed those at the riffles, and material is carried from one riffle to the next. Significant changes in bed morphology may occur during this period. It requires exceptional flow conditions to reach this stage of transport and large volumes may be moved.

Subsequent studies and field observations have shown that there are numerous degrees and variations of this two phase concept (Ashworth and Ferguson 1989) depending on the size distribution of the bed material and sediment availability. With somewhat less control from armouring, a more varied supply of sediment over time, and the regular disruption of gravels by spawning activity, O’Ne-ell Creek presents a unique river type that has not been well documented in bedload studies.

MEASURING BEDLOAD TRANSPORT

There are numerous methodologies presently available for the measurement of bedload transport in gravel bed rivers. Sampling strategies may be defined as ‘direct’, meaning samples are taken during a flood event or ‘indirect whereby measurements are taken after one or several flood events have occurred and the flood transport is reconstructed.

The standard apparatus for direct measurement are portable net or basket samplers, such as the pressure difference Arnhem, VUV or Helley-Smith samplers (see Hubbell 1964; Helley and Smith 1971; Gomez 1997). The advantages of these samplers are their portability and the instantaneous recovery of material from the flow so that the grain size distribution can be observed. The trapping efficiency is considered to be 90 to 100 percent for particle sizes from 0.5 to 32 mm. The main disadvantages are that most of the finer material passes through the 0.25 mm standard mesh, and that temporal and cross-channel variation in rates of bedload discharge are not very well represented.

Sophisticated sonar, acoustic and seismic equipment have all been applied to the task of quantifying bedload movement (Dinehart 1989; Williams 1989). More elaborate - usually permanent - installations have been developed, such as vortex tube traps (Klingeman and Milhous 1970; Tacconi and Billi 1987), pressure-pillow slot samplers (Reid et al. 1980; Kuhnle 1992), and *in situ* magnetic detection devices. Custer et al. (1986, 1987) and Spieker and Ergenzinger (1990) installed a device at Squaw Creek, Montana consisting of 5 sets of paired multiple choke-coil units mounted in a beam set lengthwise across the stream. This device is able to detect naturally magnetic pebbles and cobbles >32 mm up to a rate of two per second.

The most common methodologies for indirect measurement consist of tagged sediment recovery (Laronne and Carson 1976; Butler 1977), scour chains (Laronne et al. 1994; Hassan 1990) and channel cross-section measurements. These methods only provide an estimate of the *net* transport volume, since several cycles of scour and deposition may have occurred throughout the flood event.

Bucket and pit traps have also been used to measure the amount of bedload moving over a section of streambed. Various combinations of all of the above methodologies have provided much insight into the problems of accurately determining the conditions of entrainment, the rates and distances of transport, and exploring the relation between water and sediment discharge.

THE STUDY SITE

The O'Ne-ell Creek watershed (Fig. 2) encompasses about 68 km^2. The BMD site is located above the Tsitsutl Creek confluence where the drainage area is about 38.5 km^2. The watershed upstream of the detector site remains undisturbed, although there is a short section of road, and future plans for logging a stand one kilometer from the site. An abundant supply of large woody debris has created a forced pool-riffle morphology (Montgomery et al. 1995) for most of the creek within the lower reaches of the watershed. The detector site is on a relatively straight transport zone of the creek, between two sediment storage areas, and has a gradient of about 1.3 percent.

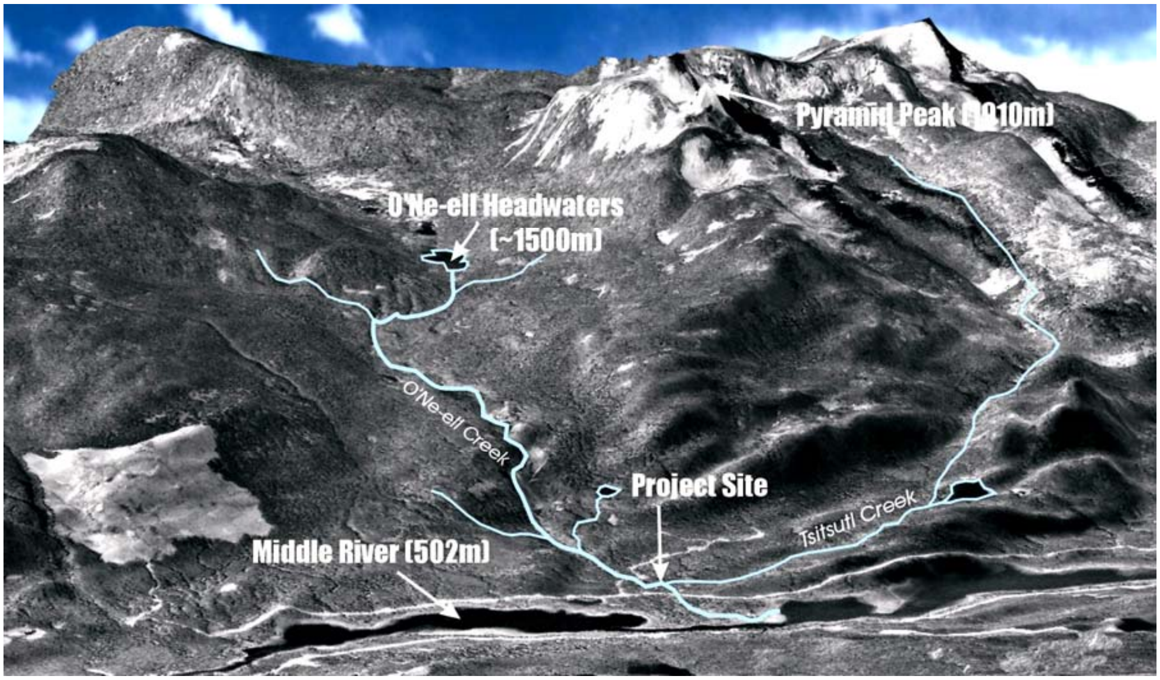

Fig. 2. Digital Terrain Model of the O'Ne-ell Creek Watershed (1987 Air Photos).

Sediment finer than 1 mm makes up 1% to 10% of the streambed at the study site. Bedload ranges from coarse sand to small boulders with diameters up to 30 cm. A dozen samples of the mobile portion (upper 15 cm) of the streambed were taken with a freeze-core sampler, and four bulk samples were taken on 1 m^2 plots (to a depth of 20 cm). The D_{50} is about 42 mm and the D_{90} about 128 mm. Sediments are predominantly derived from tills, fluvioglacial deposits, and ultramafic bedrock (Ryder 1994).

THE UNBC BEDLOAD MOVEMENT DETECTOR

The UNBC Bedload Movement Detector (BMD) functions by means of an array of 82 highly responsive magnetic sensors that are able to detect the natural magnetism of clasts passing over their sensing surface. The dimensions of a BMD sensor are similar to those of a hockey puck – 3 cm high with an 8 cm diameter – and it consists of a copper coil seated within a torus-shaped magnet. An outer steel casing confines the field of the sensor. The casing is filled with a durable epoxy resin that waterproofs the components and holds them in place. As any ferrous object passes through the sensor’s magnetic field, an electromotive force is induced in the coil. Signals are on the order of a few microvolts to several hundred millivolts. The vertical range of the sensors is approximately 5 cm.

Extensive sampling and laboratory tests have determined that the sensors are able to detect at least 30% of the bed material in O’Ne-ell Creek. It is expected that a better data acquisition system would substantially improve the sensitivity of the system. However, since the device is already detecting up to 3 x 10^5 particles per hour during peak flood discharge, the device already provides sufficiently daunting processing problems. Clasts that yield strong signals include a variety of lithologies: volcanic rocks, granodioritic to gabbroic plutonic rocks, serpentinized ultramafic rocks, metasediments and metavolcanics.

The sensors are potted along the length of a hollow aluminum beam (8.4m x 40cm x 10cm), spaced such that there is a 10 cm span between the center points of each sensor. A ¼" aluminum cover plate is bolted to the beam, shielding the sensors. The device is buried in the channel, and set flush with the streambed surface so that there is negligible interference with the near-bed fluid stream (Fig. 3). The device can be raised or lowered on threaded rods, set into the frame mounts, to accommodate aggradation or scour within the channel. Electrical cabling from the sensors is sent to a data collection shed via a polyethylene tube (Fig. 4).

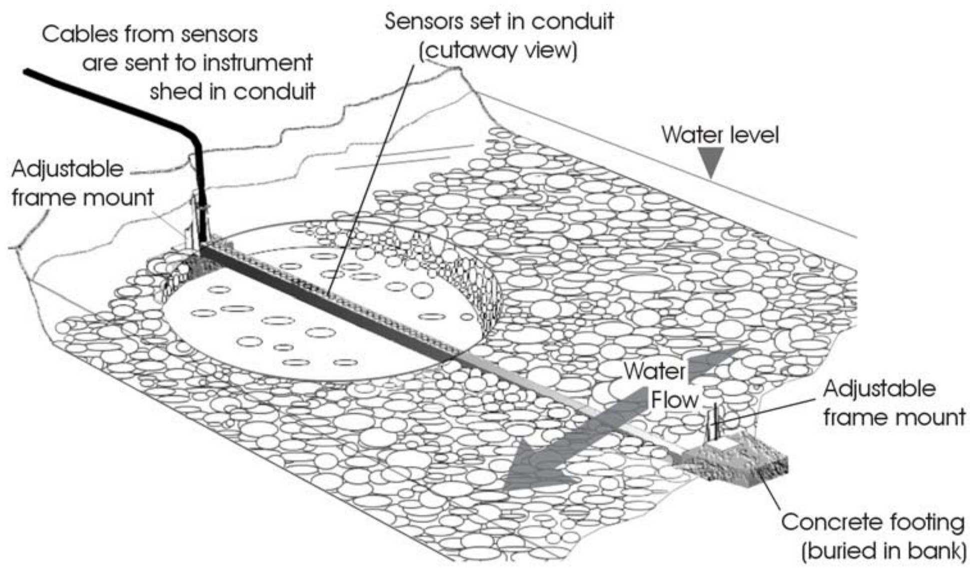

Fig. 3. Schematic diagram of the BMD installed in the creek bed.

Two telecommunications batteries store power generated by three solar panels at the site. This provides enough electricity to run a data acquisition system, a laptop computer, a Jaz™ drive, temperature and stage logger, and lights. A 5500 W generator is on site, in order to charge the battery array when necessary.

The data acquisition system consists of 6, 16-channel analog-to-digital signal-conditioning boxes that amplify the microvolt signals from the sensors and convert them into a digital value. A 4 kHz low-pass filter ensures that high frequency distortion is minimized. The data from the 82 channels is streamed to disk at sampling rates ranging up to100 Hz, requiring over 120 megabytes per hour of storage space.

The data have been analyzed using a number of signal processing tools. Digital filters are used for noise reduction and removing undesirable frequency content; spectral analyses can be applied on several time scales to determine the wavelength of signals and patterns of transport. Arithmetic routines are employed to normalize the data, sum "counts" of events over individual sensors, and then display the information in a graphical format.

Fig. 4. Photo of the study site in early August. The instrument shed is located on the east side of the creek.

DATA COLLECTION

Data from the 1999 nival flood event was collected at a rate of 100 Hz. At this rate of acquisition, a 1 mm sand grain passing at 1 m/s would register a sine wave defined by up to 7 data points. At lesser rates of data acquisition, the signal resolution of passing particles becomes poorer, and the size becomes difficult to accurately ascertain.

As a clast passes over the device, a signal similar to Figure 5 is recorded. The signal amplitude, or intensity, is primarily a function of the magnetic properties of the clast. The periodic length of the signal is determined by the size and velocity of the particle. The positive and negative excursions of the signal are related to the orientation of the sensor's magnetic field.

A particle's size and velocity determine the period of the signal. The period of a signal produced by a pebble will be shorter than that of a cobble. If the velocity of different size particles can be modeled successfully, then the size of passing particles can be determined.

The amplitude of the voltage signal from a passing clast is an imprecise estimation of the clast volume, since the magnetic response to the clast lithologies present in O'Ne-ell Creek varies over three orders of magnitude. For instance, a pebble of magnetite may produce similar voltage intensities as a sedimentary stone 50 times its mass.

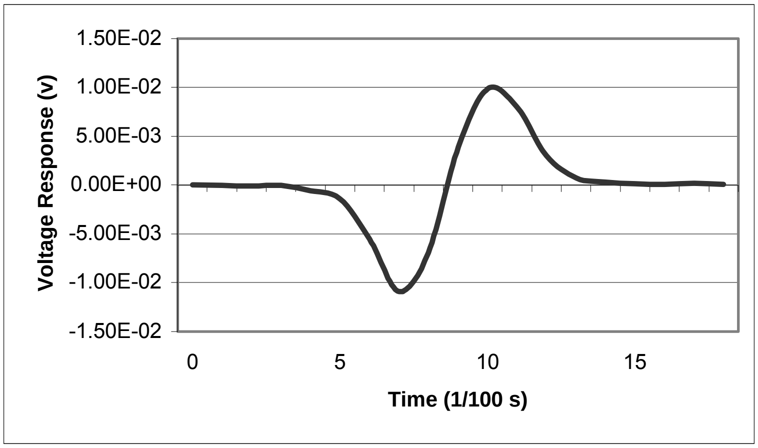

Fig. 5. Voltage signal x(*t*), from the passage of a clast over a sensor.

There are a number of issues to be addressed with regards to the passage of sediment over the sensors. Passing particles will not always pass the full width of the circular sensing face, and thus a shorter signal will result. Larger particles may trigger more than one sensor as they pass, registering two or more signals of varying length. Lastly, more than one grain may pass over a sensor in the same instant, making it difficult to discern the actual periods of the particles. These situations require sophisticated signal processing algorithms that have not yet been perfected.

The simplest method of depicting the relative amount of bedload transport activity across the channel throughout a flood event is to show the total amount of voltage recorded at each sensor versus time. The voltage readings from the BMD are taken to be a reasonable proxy for the rate of sediment flux. It is emphasized that although this is not entirely accurate, it is reasonable for illustrative purposes. If it is assumed that the distribution of magnetic particles varies uniformly over time, then statistically these summations are a good approximation of the bedload activity.

Reading the BMD Record

Much of the initial data analysis has consisted of summing the many hours and days of data into 30 second voltage sums for each sensor (Fig. 6). Sensor 1 is on the west bank and sensor 82 on the east. Charts of the total transport activity over time are then plotted onto graphs similar to Figure 7. Obvious spatial and temporal trends in transport activity then become evident, and the active periods of transport can be identified and further analyzed.

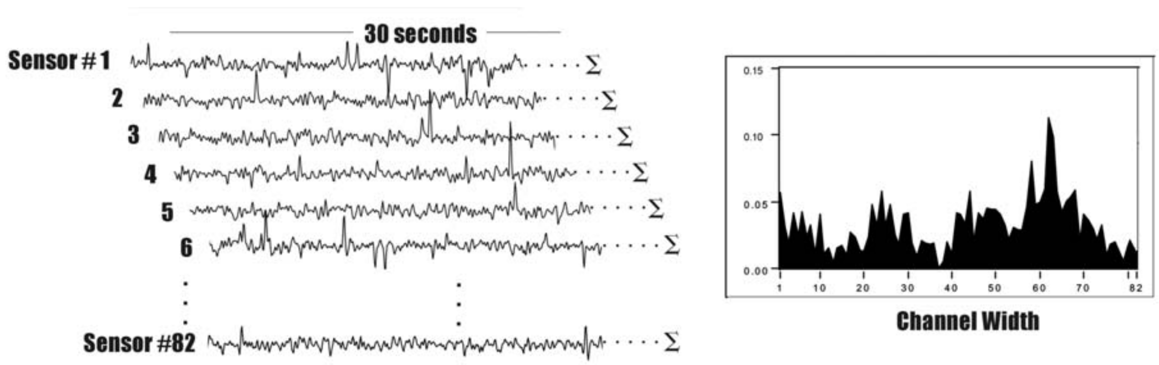

Fig. 6. The BMD record consists of summing the voltage activity in absolute (positive) units for each channel in 30 second intervals. The resultant sums show the rate of activity across the channel width in the chart on the right.

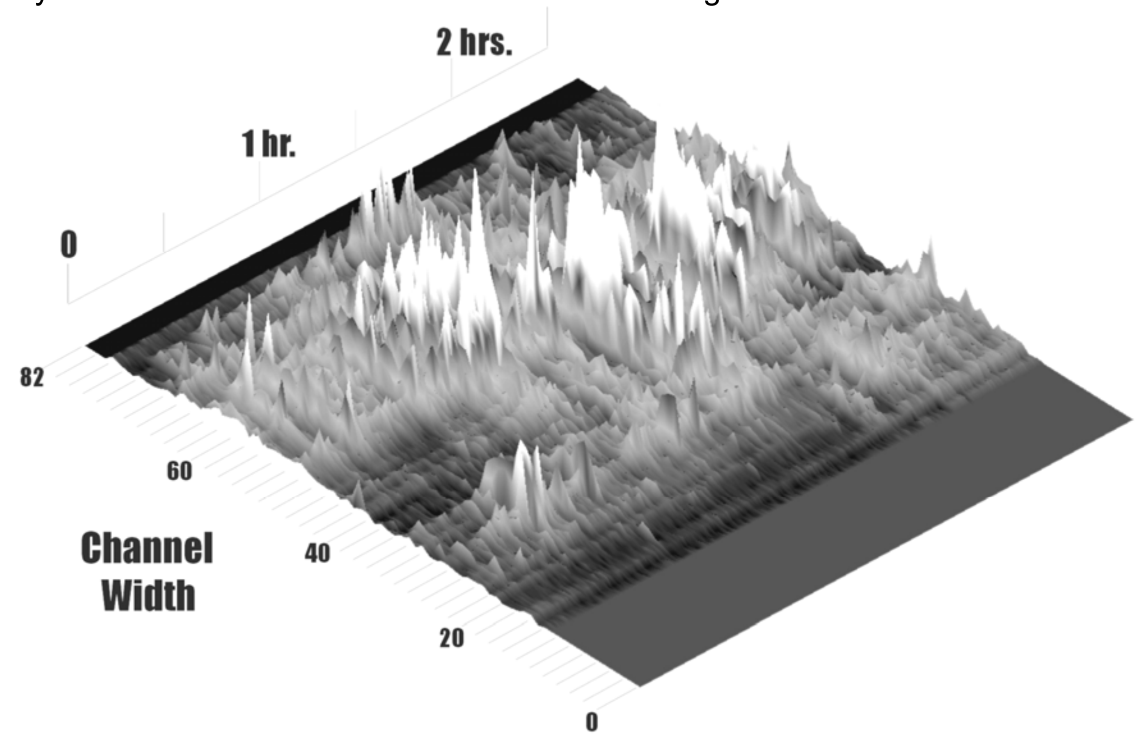

Fig. 7. The sums for each 30-second interval are then plotted over the hours and days of the flood event. Here a 2.5 hour section of data from 20:00 to 22:30 on June 11 is shown.

The 1999 Nival Flood Event

In the first season of collecting data (1998) the nival flood was smaller than average and was several weeks late (FOC 2000). A brief but energetic flood was recorded between May 24 and 25 (Tunnicliffe et al. 1999). The creek swelled to bankfull overnight, and then subsided quickly. This flood wave mobilized a large portion of the bed for twenty-four hours. Transport activity then dwindled again to sparse, random movement. The data was sampled at 30 Hz, which appeared to be the minimal sampling rate for capturing the passage of clasts.

In 1999 a larger than average snow pack was present at the beginning of the snowmelt season. Cool weather caused a delay in the spring melt, and when the nival flooding did finally come, it was drawn out for over 2 weeks. Data was recorded continuously during this period. A wide range of flow conditions was observed over the course of the flood, enabling us to identify a number of distinct patterns or phases of sediment transport.

Figure 9 shows a graph of the stage level, bedload discharge, and suspended sediment discharge from June 9 to June 18, 1999. The strong diurnal signal is due to the higher altitude snowmelt events in the afternoon that create a flood wave, passing by the study site some 6 to 8 hours later. The suspended sediment index rises with discharge levels most days, although there is a subdued response on the night of the 13th, when the availability of fine sediment appears to decline following a period of prolonged bedload activity.

Discharge through the study section reached a peak of 8.6 m^3/s on June 15. Instantaneous velocities in the creek were measured daily, and ranged up to 2.6 m/s. The channel cross section at peak discharge is shown in Figure 8. The depth and velocity measurements were taken roughly 30 cm downstream of the BMD. The relatively flat cross section of this spot was present before installation of the BMD.

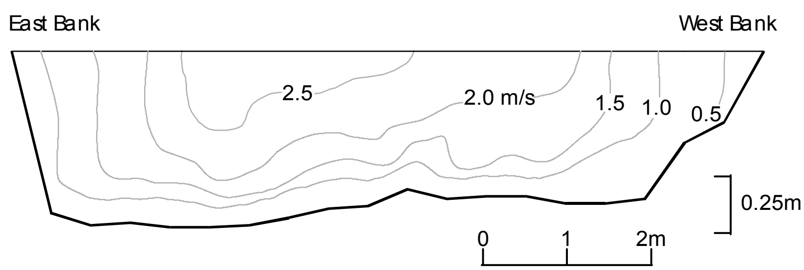

Fig. 8. BMD Site Cross Section: contours based on maximum stage measurements in 1999 show the variation in velocity across the channel width. The vertical dimension is exaggerated threefold.

Figure 10 shows the 8 days of bedload activity throughout the 1999 nival event. Gaps in the record are due to periods of maintenance and technical problems. Some portions of the record have been removed due to excessive noise levels. Most of the record on June 16 was missed after a disk drive failure.

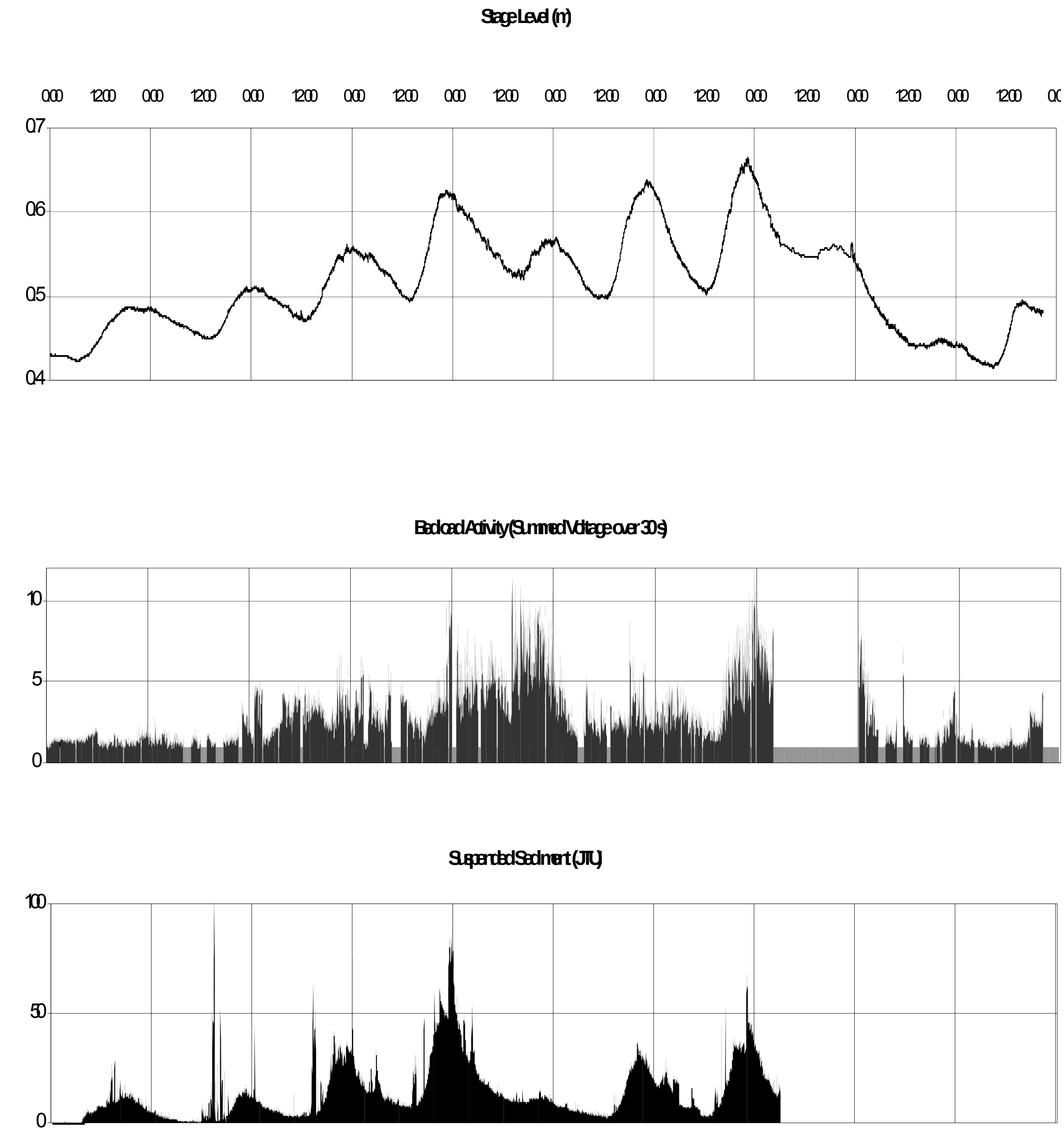

Fig. 9. Stage, bedload activity and suspended sediment readings from the 1999 flood.

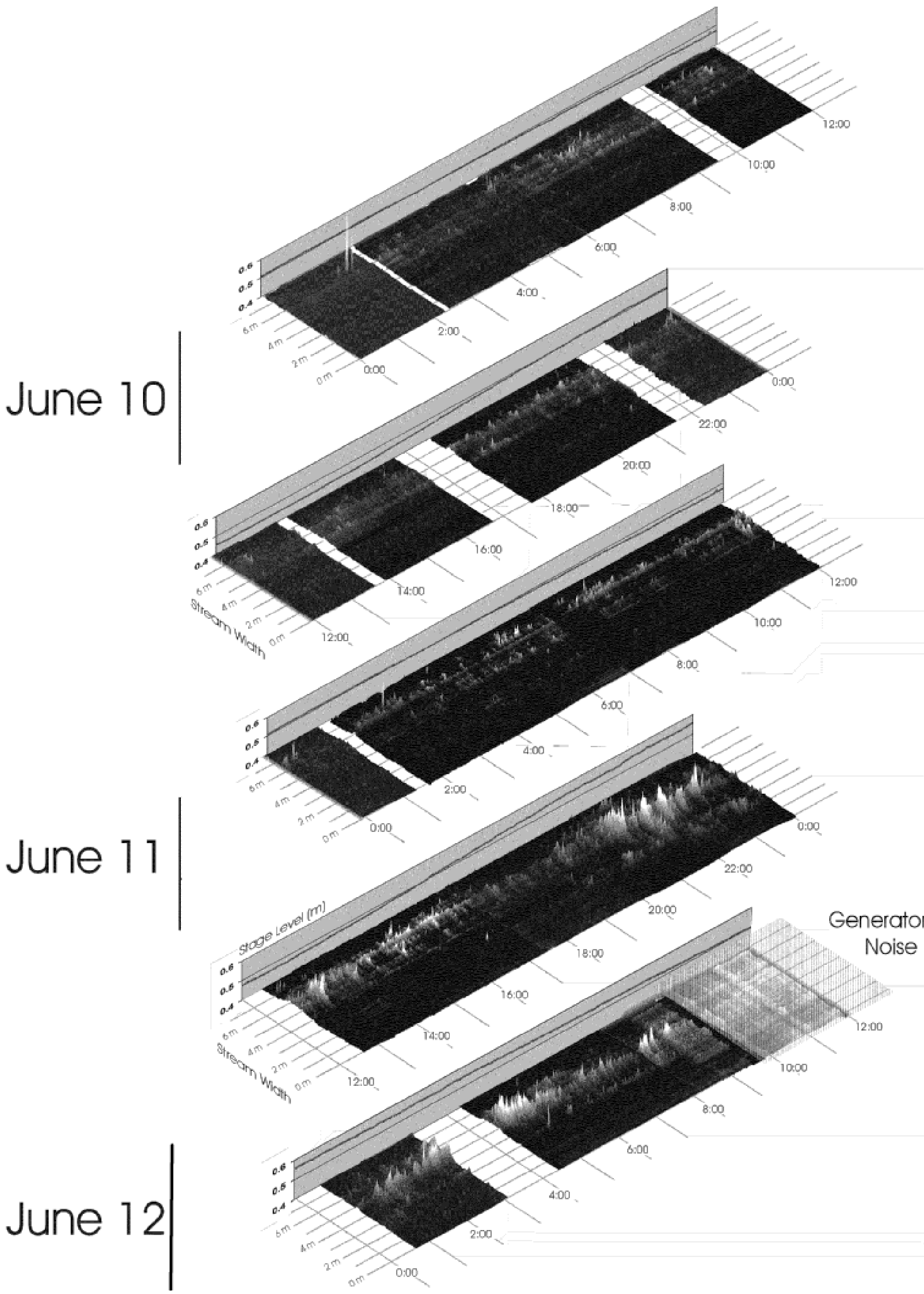

Fig. 10. The BMD record from June 10 to June 17. The channel width is measured from the west side of the channel to the east. File lengths vary from one half hour to four hours. Gaps between the files are from maintenance and technical problems. Most of the record on June 16 was missed after a disk drive failure.

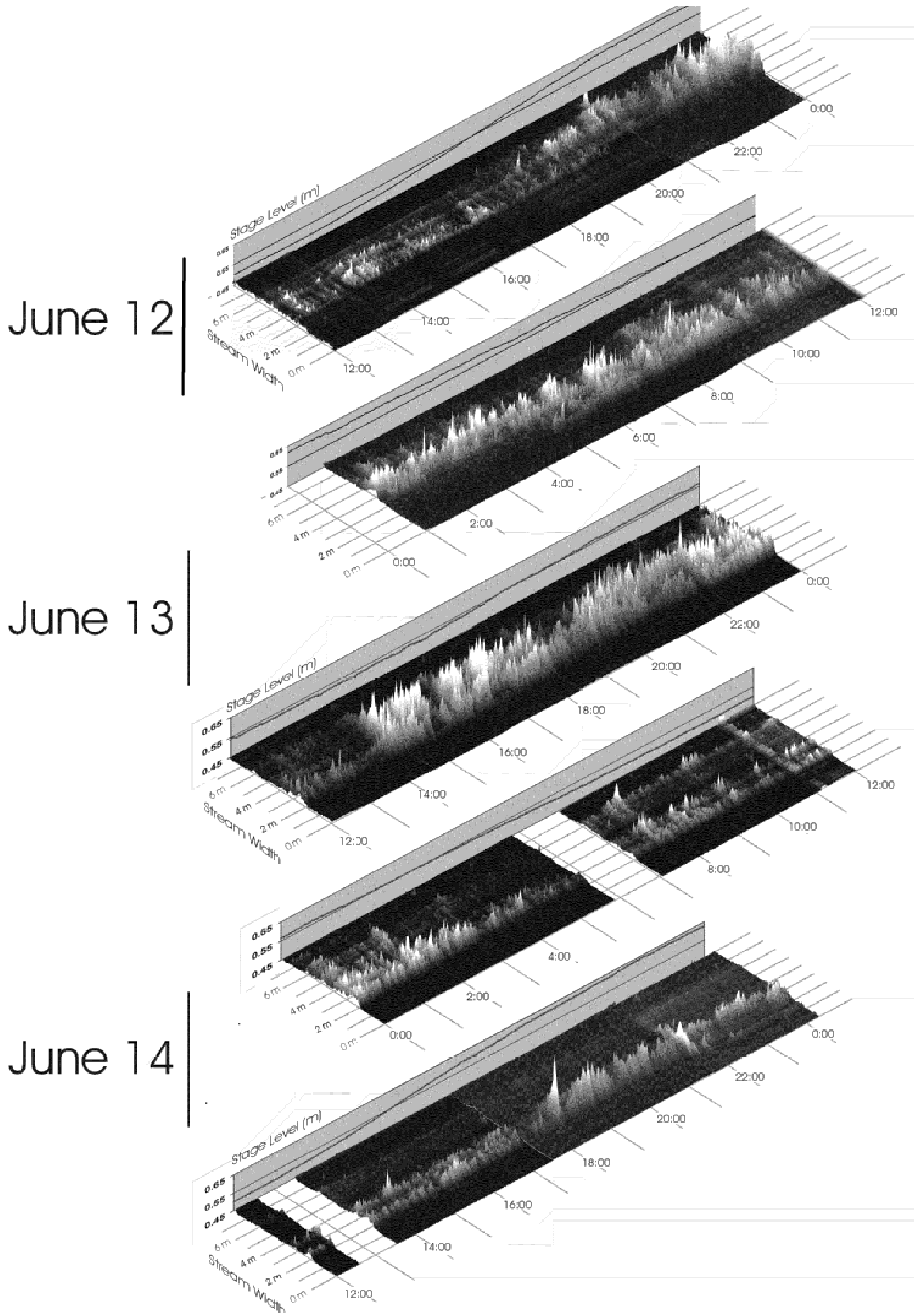

Fig. 10. Continued.

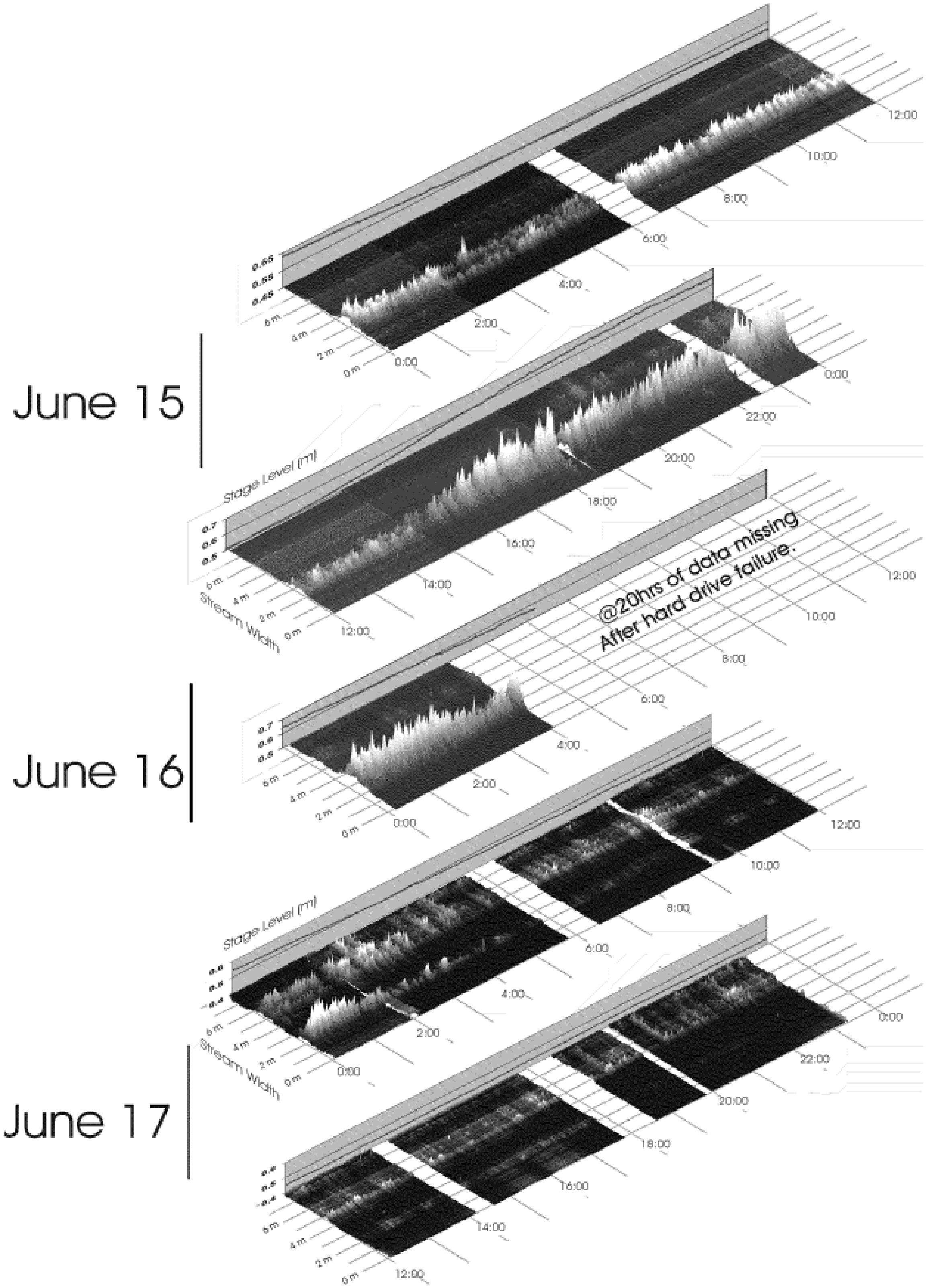

Fig. 10. Continued.

Marginal Transport

From May 23 to June10, the creek discharge was close to the threshold of competence. Only small amounts of organic debris, clusters of fine grains and pebbles were transported. For a few evenings between May 30 and June 5, the stage rose to a point where particles in the pebble range were transported, but it is mostly sand-sized particles that are evident in the record. Transport was generally sporadic, and when the discharge and turbulence increased, particles showed a tendency to pass as clusters.

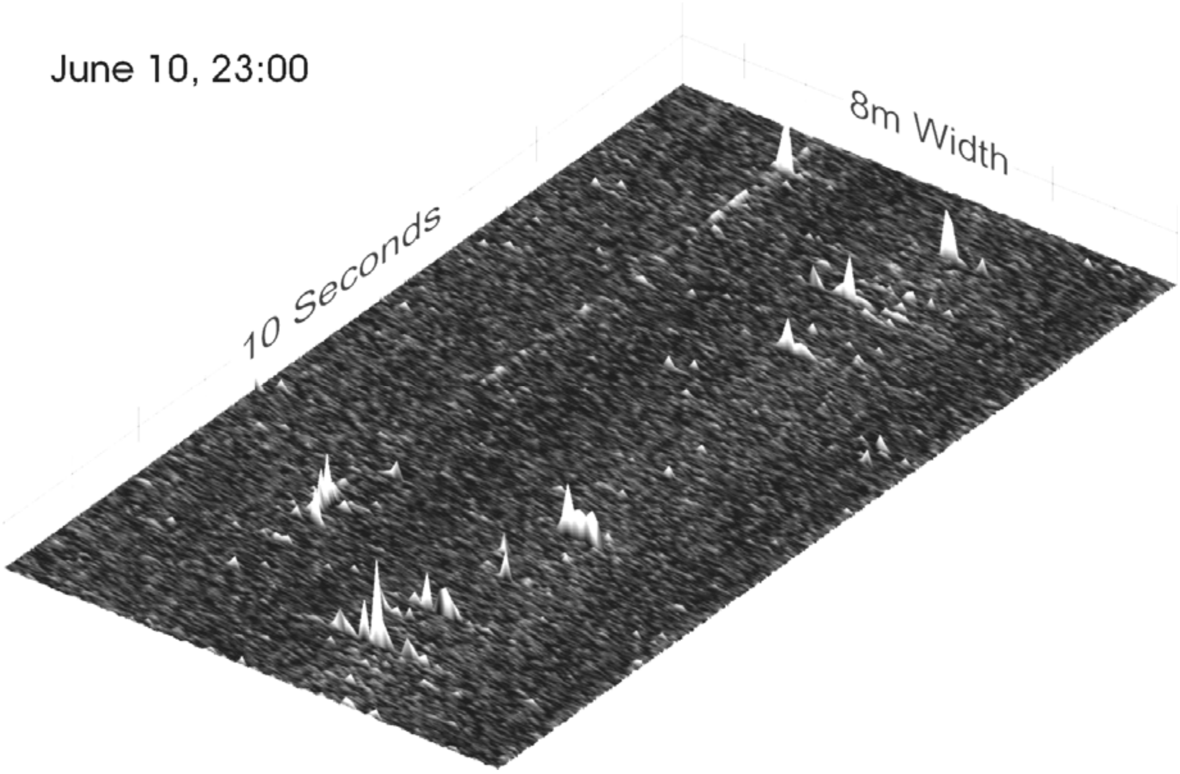

Fig. 11. Sand-sized material in transport

Figure 11 shows a sample of the record during this early phase of transport. The majority of the transport activity is on the east side of the channel where flows were slightly faster. The narrow band of activity on the west part of the channel is sandy easily transported material from a bar immediately upstream of the BMD.

On June 9, a heavy rain fell for most of the morning. This contributed to the snowmelt flow, which was probably accelerated by substantial melt in the higher reaches of the basin. The stage rose to a point where larger clasts up to 2 cm were transported over the device. Two days later, temperatures rose to 25ºC, and material began passing in greater volumes.

Clustered Transport

As increasing numbers of particles are picked up and carried along the bed, fluid turbulence and particle interaction cause grains to cluster together. There are many points in the record where it becomes difficult to distinguish among several particles passing at once.

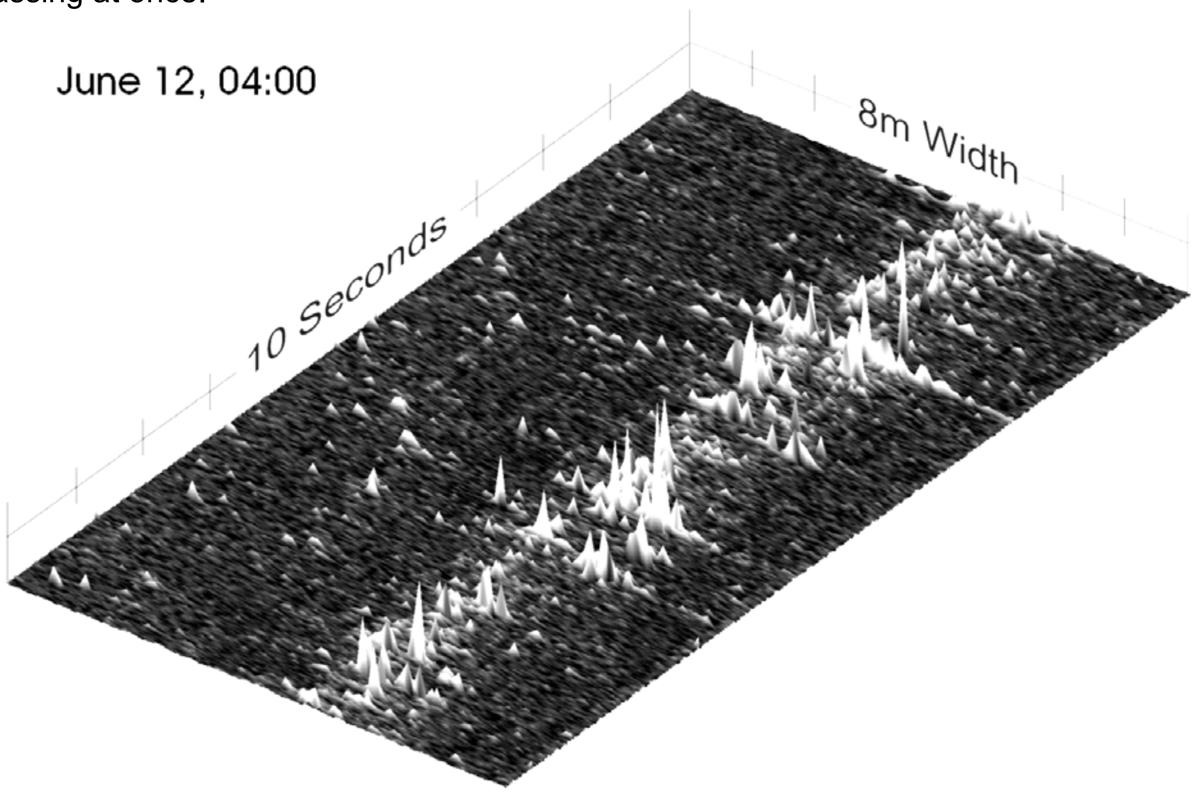

Fig. 12. Clustered transport

Figure 12 shows an example of transport during clustered transport when material is moving faster, with particles ranging up to 2-3 cm diameter. Some of the bundles of sand and pebbles wash over the device in swaths up to 3 m wide.

Continuous Transport

As the stage rose during the afternoon and evening of June 12, increasingly intense waves of sediment passed over the device. Inter-granular collisions, and the sheer volume of material in motion, blur the distinction of clusters passing. At times, the material seems to pass in steady sheets. The activity continues until about mid-day on June 13, when the amount of material in transit subsides for a few hours and then resumes and reaches a maximum on the evening of June 13 (Fig. 13). There is clearly an abundant supply of material entrained by the high flow conditions and the bedload discharge continues even as the stage level drops during the mid-day period of June 13.

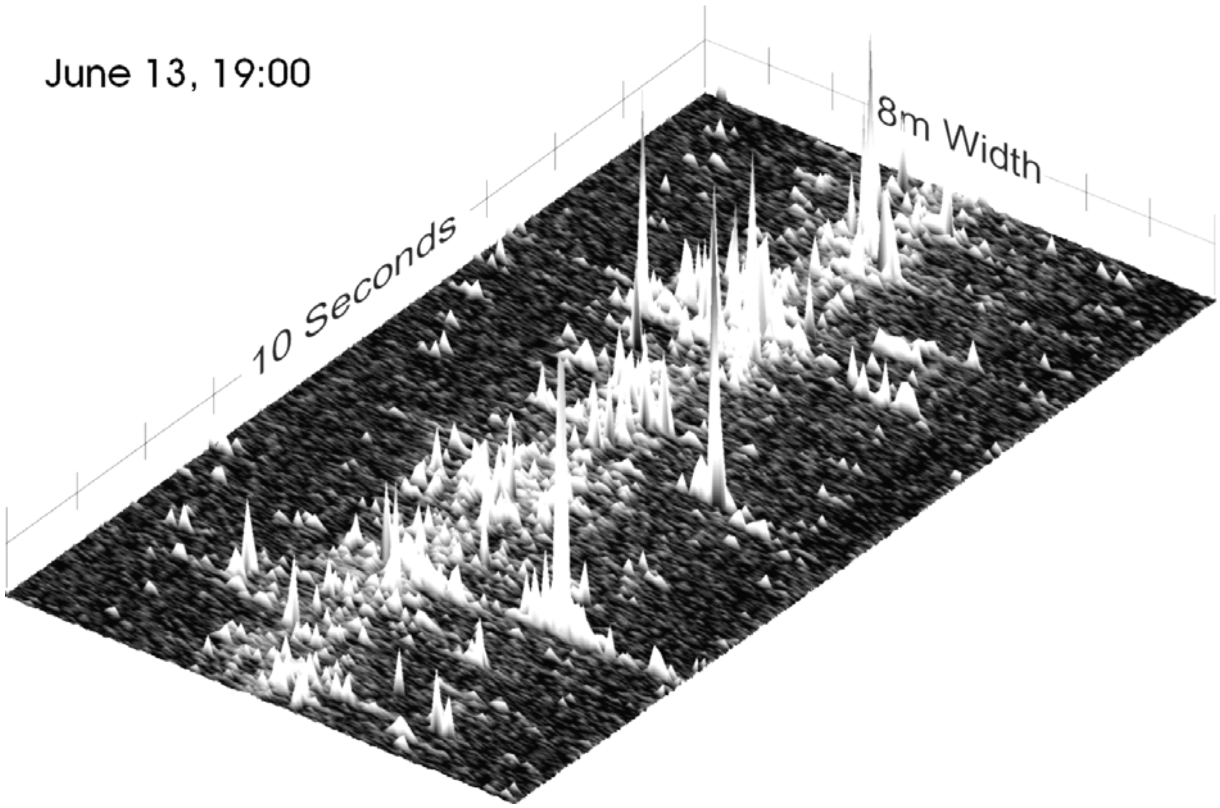

Fig. 13. Transport at the height of the flood event.

With the ample availability of material for transport, the clustering effect becomes less pronounced, and we observe the onset of pulses lasting for 10-20 minutes. In the brief intervals between pulses, the transport rate drops almost to zero.

Sand Street Transport

The final piece of the record (June 14^{th} to June 17^{th}) is dominated by the passage of sediment stringers, originating from a collapsed section of the bank approximately 65 m upstream (Fig. 14). A shift in a logjam diverted flow against the bank and caused a large failure, sending roughly 5 m^3 of sandy material downstream. The transport is confined to one or two narrow transport zones, 1-2 meters wide, which migrate laterally about one meter. The position of the transport "streets' is determined by newly developed minor bars upstream of the device and probably individual small boulders. The recorded signals indicate that the stringers were comprised mostly of sand and gravel material.

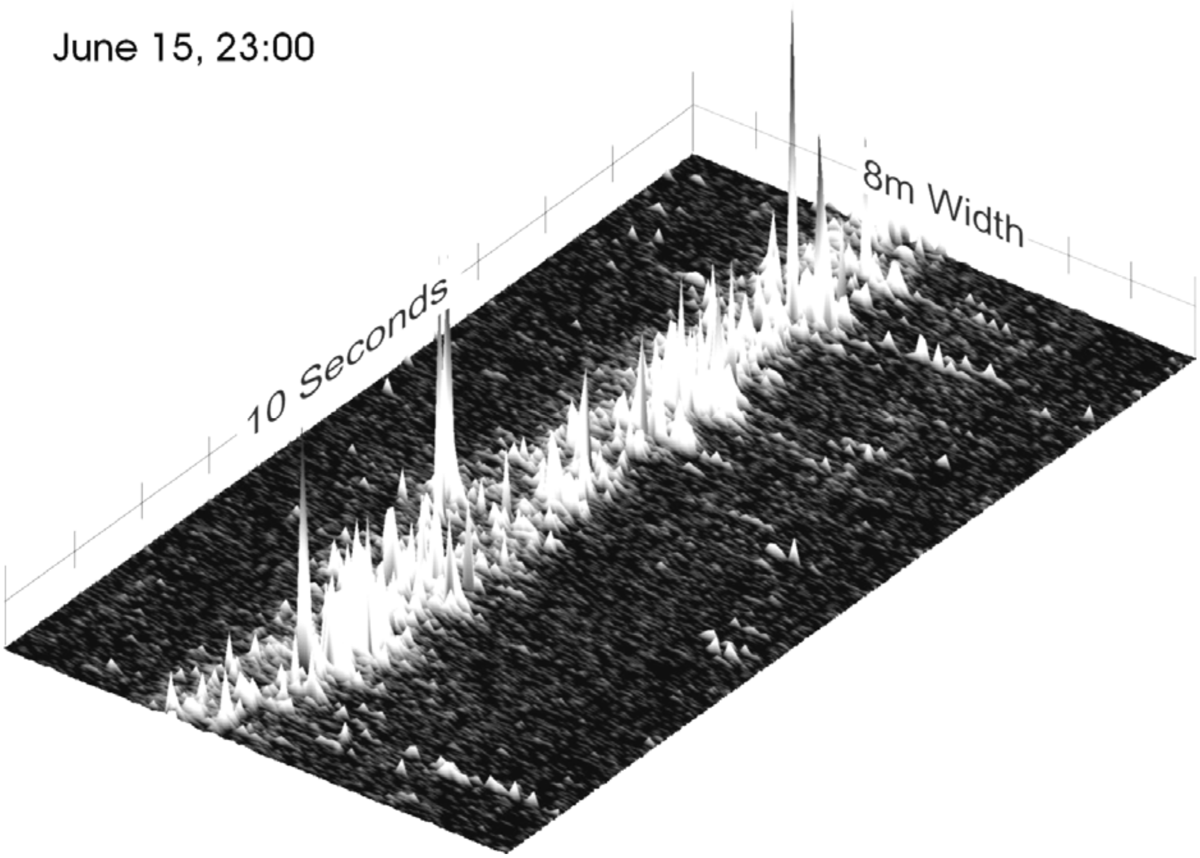

Fig. 14. Steady discharge of sand and gravel along the bed.

Observations of Pulses in Bedload Transport

The BMD has generally proven to be an effective instrument for recording the passage of sediment through a river reach. The pulsating nature of sediment transport is clearly illustrated on several scales. The record clearly shows that as a sufficiently quantity of coarse sediment is set in motion, transport becomes congested and sediment travels in waves, or pulses. Flume studies have shown that pulses with very regular periodicity arise as a matter of course in heterogeneous sediments (Kuhnle and Southard 1988; Hoey and Sutherland 1991). In natural gravel-bed rivers, with increasing complexity of bedforms, turbulence and sediment supply, the process becomes more chaotic. This phenomenon has long been understood as an integral part of bedload transport, however it has rarely been quantified at this resolution.

Figure 15 shows a typical section of the record from the peak of bedload movement on June 13. The upper portion of the graph shows the spatial variation in the transport activity. The active transport zone is mostly confined to a 1 m lane, although sporadic activity can be seen along the whole length of the device. The graph below indicates the summed voltage record from a single sensor located 3.4 m from the east bank. The pulsations seem to occur every 10-20 minutes, although this varies throughout the flood.

It is interesting to note that the locus of transport in the channel moved substantially throughout the event, drifting to within 2 m of the left bank (see June 13, 00:00) to within 2 m of the right bank (June 14, 00:00).

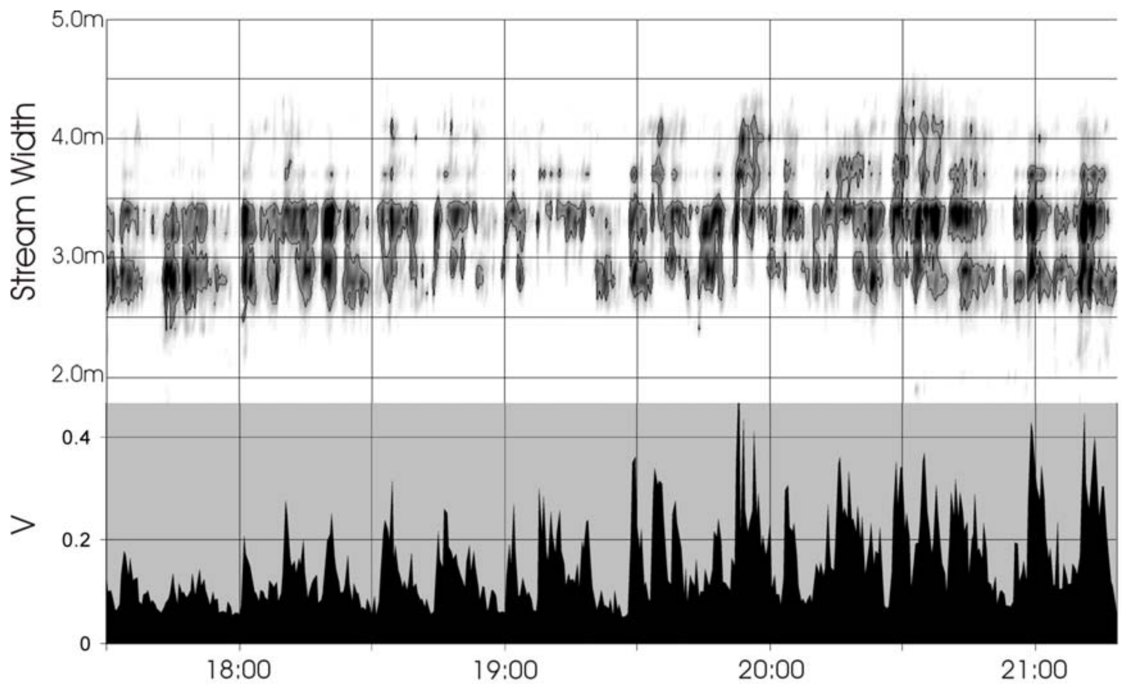

Fig. 15. The upper portion of the graph shows activity across a 3 m width of the channel. Measurements are from the east bank. The lower portion of the graph shows summed voltage activity across a single sensor located 3.4 m from the east bank.

In O’Ne-ell Creek we observe pulses on several time scales. The ‘pulses’ observed with the BMD at O’Ne-ell Creek are generally less than 24 hours; smaller than the larger multi-event cycles of degradation and aggradation known as ‘slugs’ or ‘translational waves’ which exist on a timescale of months or years (Nicholas et al. 1995).

Firstly, there are daily pulses observed, as sediment moves in response to flood discharge. Sediment movement gains intensity in the late afternoon and peaks just before midnight. Secondly, as the volume of sediment in transit reaches a threshold, the transport becomes unsteady and moves in pulses of 10-20 minutes. Finally, at the instantaneous time scale, sediment is observed moving in clustered forms, rather than as discrete particles.

Periodic pulsations on the order of 5 minutes to 2 hours have been widely reported, but the reasons are poorly understood. Numerous studies have related the pattern to the migration of distinct bedforms such as dunes or sheets of sediment (Kuhnle and Southard 1988; Whiting et al. 1988; Gomez et al. 1989), although such coherent forms are not often observed in coarse gravel systems such as O’Ne-ell Creek.

A likely hypothesis is that part of the variation in sediment discharge rate is due to changes in the rate of the sediment supply during the flood event. Supply may vary in response to the scour and fill cycles that take place upstream. As material is transferred from one reach to the next, traffic jams undoubtedly occur. The influence of large woody debris, bed armouring and bar development undoubtedly plays a role as well. The collapse of the stream bank section noted above effectively illustrates a typical source of a sediment pulse.

CONCLUSIONS

The UNBC Bedload Movement Detector has provided a unique glimpse into the streambed dynamics at O'Ne-ell Creek in the Stuart-Takla Fish-Forestry Interaction watersheds. Using an array of highly sensitive magnetic sensors to detect the passage of the naturally magnetic streambed material, instantaneous observation of bedload transport is possible. To date the device has only captured two nival flood events, however we hope to apply the device towards measuring transport rates during summer or fall floods, as well as during salmon spawning activity at O'Ne-ell Creek. It is expected that as computer processing power and storage capabilities develop, it will eventually become feasible to leave an automated recording station at the site.

Results from the first recorded floods have shown that bedload discharge is a highly variable phenomenon, strongly related to sediment supply, bed conditions and water discharge. At least four distinct phases of transport are discernible from the 1999 flood, although we presume that with different bed material composition and changing hydraulic conditions, other phases would manifest. The 1999 nival flood was unusual in that it was so long and that the peak discharge was relatively low. The 1998 flood had a short period but with significantly higher flow. Larger floods (> 10 year return interval) will have a larger transporting capacity with greater, and more stochastic, sedimentary inputs. We might then expect to see roughly similar patterns of transport, with coarser material, higher rates of flux and greater variability in the transport record.

Substantial calibration work remains to be done, so that we may be better able to estimate the size of particles in transport. We anticipate that with further development the BMD will provide important insights into the relationships between bedload transport, river morphology, sediment supply, and river habitat.

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Morphology and Composition of Suspended and Gravel-Stored Sediments in Salmon Bearing Streams of the Takla Region of British Columbia

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INTRODUCTION

The impact of fine sediment accumulation in gravels used as spawning habitat for salmonids has been well documented (Chapman 1988; Platts et al. 1989; Kondolf et al. 1993). The definition of fine sediment varies amongst studies with sands from 0.1 up to 4 mm being identified as problematic for egg survival in a review by Chapman (1988). These materials settle out of the water column quickly and fill up the pore spaces between gravels, reducing water flow around the eggs. Silts and clays, or those inorganic sediments smaller than 63 μm, have also been of interest in aquatic habitat quality studies as they too can fill gravel pore spaces. Delivery of these fine sediment particles into flowing water would presumably result in their advection out of the system, as they should remain in suspension due to their slow settling rates. But silts and clays have the ability to flocculate or aggregate which modifies their transport and settling behaviour (Eisma 1993). This process of flocculation involves the attachment of small individual particles to form larger, generally more porous composite structures with more complex edge shapes, which act to modify their settling behaviour. A number of conditions have been documented to facilitate this process and includes high conductivity, high suspended sediment concentrations and/or the presence of biological breakdown products, which act somewhat like a glue holding particles together (Muschenheim et al. 1989; Liss et al. 1996). The increase in settling rates observed due to aggregation or flocculation (Droppo et al. 1998) means that the small constituent particles that exhibit very slow fall velocities as individual particles are now capable of settling out in flowing waters. The aggregated inorganic fine sediment will be stored in or on the gravels along with the associated organic matter that is incorporated in the floc. This results not only in a physical filling of gravel pore space but also in an increased sediment oxygen demand which can be deleterious to fish egg survival.

This paper presents the results of work undertaken in two experimental watersheds in the Stuart-Takla region of British Columbia. It is one aspect of a larger, multi-disciplinary study investigating the impact of forest harvesting activities on the hydrologic, geomorphic and thermal regimes of rivers and their influences on the abundance, quality and distribution of fish habitat. More detail on the design of the Stuart-Takla Fish-Forestry Interaction study (STFFIS) is provided in Macdonald et al. (1992) and Macdonald and Herunter (1998).

The pre-harvest evaluation of the gravel beds in the Takla fish-bearing streams indicates that fines (defined in this paper as sediment < 63 μm) make up only a very small proportion of the material below 50 mm. Herunter (pers. comm.) determined that salmon redds (gravel mounds that are created by the fish digging and cleaning the gravel bed to deposit eggs) in Forfar Creek had less than 1%, by weight, of the sediment in the size class smaller than 75 μm. While the relative abundance of fine sediments is currently low, what is of interest is the structure of this material and its composition before activities in the watershed potentially increase fine sediment delivery. The size, fall velocities and composition of the composite particles generated from fine inorganic sediment will regulate its movement and storage as well as its impact on the gravel conditions.

Earlier work on the structure of suspended and gravel-stored sediment indicated that in these biologically active headwater streams the fines were well flocculated (Petticrew 1996, 1998). The aggregates or flocs exhibited maximum sizes 7 (suspended) to 14 (gravel-stored) times greater than the maximum size of the constituent inorganic material comprising the composite structures (Petticrew 1996). Following these findings research has continued in these streams to: a) determine the structure and fall velocities of the suspended and gravel-stored sediment, b) investigate seasonal differences in sediment structure and fall velocities and c) evaluate the implications of these results on forest management approaches.

METHODS

Study Sites

Two of the watersheds from the STFFIS are included in this study. O'Ne-ell and Forfar Creeks have watershed areas of 75 and 42 km^2 respectively. Both streams empty into the Middle River in a region that was once covered by a post-glacial lake. Therefore the lower 2-3 km of each stream lies on a low sloping (< 2%) glacio-lacustrine deposit, 2-3 m thick (Ryder 1995), which is composed of fine silts and sands (Petticrew 1996). Currently the lower reaches of the river exhibit gravel sizes favourable for spawning and support an extensive sockeye salmon (*Oncorhynchus nerka*) stock (Macdonald et al. 1992). Sampling of suspended sediments and water occurred in the lower portion of the river either at the bridge location where the main road crosses the stream, or downstream of this, in the reaches with active spawning. The bridge crosses O'Ne-ell Creek at a distance of 1700 m upstream of its mouth while Forfar Creek is bridged at a distance of approximately1650 m.

Results are presented from two sampling periods. Sediment was collected in late August of 1996 following a period of intensive spawning in Forfar and O'Ne-ell Creeks. Springmelt sediments were collected from O'Ne-ell Creek in late May 1997. At the time of sampling the only significant anthropogenic disturbance which could potentially alter sediment delivery to the streams was the road crossing and its associated ditches that expose the glacio-lacustrine deposit.

Field Sampling

To obtain samples of both the in-stream and gravel-stored sediments during the post-spawn period in 1996, the channel bed in the region of fish redds was artificially disturbed. The top 0.04 - 0.06 m of gravel were loosened with a force similar to that of spawning salmon (Petticrew 1996). This took place approximately 3-5 m upstream of the collection point in order to obtain a sample of resuspended fine sediments stored in the gravel matrix, but to give sufficient travel distance to allow the resettling of heavier sand particles.

The suspended sediment was collected from Forfar and O'Ne-ell Creeks using a settling chamber. A rectangular plexiglass box (1.5 x 0.14 x 0.06 m) with two removable end caps was built to hold approximately 13 L of water. A scale was mounted on the outside back wall of the settling chamber using white adhesive paper that aided in photographing and sizing particles. The settling chamber was aligned into the water flow such that water and suspended sediment passed through it. When a sample was required the ends were capped and the box carried in a horizontal position to the side of the creek, where it was placed vertically in front of a 35 mm single lens reflex (SLR) camera. After a period of several minutes, during which fluid turbulence decayed, a series of timed photographs were taken. Pairs of sequential images were then examined to identify individual flocs and estimate their particle size and the distance travelled. The lower resolution using this technique was approximately 150 μm (Petticrew and Droppo 2000). The observed fall velocities of the identified particles were calculated assuming Stokes' settling conditions, and then they were plotted against both the calculated equivalent spherical diameter (ESD) and the long axis of the particles. ESD is the diameter of a sphere with the same area as the measured particle.

In the spring of 1997 the settling chamber was used to collect suspended sediment samples from the snowmelt flood events in O'Ne-ell Creek. At this time the box was lowered and returned to the bridge platform using a winch system. The box was filled and capped by persons standing in the stream. The photographic system employed in the field at this time was a video capture system. A black and white digital camera (a charged-coupled device - CCD), with a resolution of 512 by 512 pixels, was connected to a personal computer running Empix Imaging's Northern Exposure software. This field setup allowed an automated image grabbing system, which recorded the current time (accurate to 10^{-2} s) on each image. A run of 45 images could be grabbed in just over a minute and a half. The resultant images had square pixel dimensions of 55 μm $\pm$10 μm. The images were then analysed via a custom-developed (Biickert 1999) settling rate measurement application.

Measurements of particle size and settling velocity for both the SLR and video imaging method allowed for the derivation of particle Reynold's numbers as well as particle density using the equations presented in Namer and Ganczarczyk (1993).

Discharge data was obtained from Fisheries and Oceans Canada who maintain the gauging stations upstream of the bridge sites at each of the streams.

RESULTS

Fall velocities of particles moving in the water column and released from storage in the gravels during the post-spawn period of August 1996 are shown in Figure 1. Two distinct types of composite particles were identified through visual inspection of the SLR photographs. Compact particles, here called aggregates, were observed as well as fluffy, porous particles, which are termed flocs in this paper. A proportion of the photographed particles were noted to be a combination of the two structural types (floc-like and aggregate-like) while five of the 70 measured particles appeared to fit neither category and were classified as undefined. The compact aggregates had long axis dimensions between 400-1900 μm and exhibited fall velocities between 4-15 mm/s. Alternately, the floc particles exhibited sizes between 400-2300 μm and fell at rates less than 6 mm/s.

Suspended sediment was collected during the springmelt flows of 1997 on O'Ne-ell Creek. Sampling on May 28 and 30 represented the rising limb of a rainstorm in the lower portion of O'Ne-ell Creek, but a rain-on-snow event at higher elevations. The first flush of snowmelt waters from the lower elevations in the watershed had occurred in mid-May, previous to the sampling period (Fig. 2). Fall velocities of the sampled suspended sediment indicated that only one population of particles was collected. Particle long axes for the total springmelt sediment population sampled measured between 119 and 712 µm, but only three of the 280 particles identified by the video capture technique were observed to have high fall velocities (ie. > 6 mm/s) similar to those of the compact aggregates from post-spawn periods (Fig. 3).

Densities of the particles observed in the settling box were calculated and plotted against the particle long axis (Figs. 4a and b). The springmelt particles exhibit higher densities (maximum = 1.76 g/cm^3) and tend to have lower numbers of larger particles (only 3 % exceed 500 µm) whereas the post spawn samples have much lower densities (all particles are less than 1.08 g/cm^3) but have larger sized particles (84 % exceed 500 µm). In Figure 4b only the visually certain flocced and compact particles are represented (n = 32 and 17 respectively). Note that the compact particles are always slightly denser than flocced composite particles of the same size during this post-spawn period (Fig. 4b).

DISCUSSION

Following the period of active spawning the composite particles moving in the water column and/or those released from gravel storage are found to consist of two quite different morphologies. Compact aggregates appeared less porous and had less complex edge structures than the fluffy, composite floc particles. Some of the compact aggregates exhibited settling rates that exceed that of fine sands, which with a diameter of 100 µm and a density of 2.65 g /cm3, settle at 9 mm/s. An individual silt-sized particle of 50 µm, which would represent one of the larger sizes of constituent particles comprising a floc, with a mineral density as before of 2.65 g/cm^3 has a calculated Stokes' settling velocity of 2.25 mm/s. The settling rates observed for the majority of the post-spawn composite particles exceed this value increasing the probability that they will be retained in the system.

Particles with the higher fall velocities (> 6 mm/s) were not prevalent in the springmelt samples. The bulk of sediment sampled (99%) at this time of year had fall velocities similar to the floc structures of post-spawn period. The video imaging system did not allow a visual characterization due to the pixilation of the images, but if many compact aggregates were collected in these samples their fall velocities at this time of year are lower.

The calculated densities (Fig. 4a and b) indicate higher maximum values for springmelt samples and lower maximum values for the post-spawn period. Approximately 60% of the springmelt samples have densities exceeding 1.08 g/cm^3, or the maximum density observed in the post-spawn samples. But note that the other 40% of lower density (<1.08 g/cm^3) are comprised of particles over the full size range (151-712 µm). It is interesting that all of the particles in both the springmelt and post-spawn periods that are larger than 400 µm are low density. In the springmelt samples (Fig. 4a) there are also particles smaller than 400 µm, with low density. These less dense, small flocs would be composed of higher concentrations of organic matter, relative to the denser particles of the same size, as density in this size range is controlled more by composition than by size or shape.

All of the flocs larger than 400 μm are determined to be low density (<1.08 g/cm^3) but they exhibit fall velocities over the range of 0.1 to 15 mm/s. This wide variation would be expected to be a function of their differing composition, porosity and shape. Extracellular polymeric substances, which act to bind together inorganic particles, encourage large, porous floc structures which have fast settling rates (Droppo et al. 1998). Very large flocs are not observed in the high velocity flows of the spring melt (60 to 140 cm/s) in the Takla streams or in southern Ontario springmelt flows (Droppo et al. 1998), whereas the lower base flows of post-spawn (25-30 cm/s) are associated with floc structures exceeding 1000 μm. Low shear velocities and an abundance of biological breakdown products encourage the growth of large instream flocs during low base flow periods. The more compact, composite structures that are released from storage in the gravels are both large and fast settling. These aggregates consistently have densities greater than the suspended sediment flocs of the same size in the post-spawn (Fig. 4b) indicating a difference in composition and structure. These particles may be the result of the processing of organic matter within the gravels by benthic organisms reworking the stored or passing sediment, producing fecal pellets lower in organic content. Also, gravel storage of the fluffier, porous flocs, over a period of days or weeks, would result in the mineralization of organic matter as well as the de-watering of the flocs, resulting in compact, slightly denser particles. The source of these compact structures is not likely to be the aggregated clay materials eroded from the exposed glaciolacustrine deposits, as they would exhibit higher densities. It is most likely that these particles are generated in-situ, from the processing of stored or passing flocs by the local invertebrate and bacterial populations.

As mentioned previously the availability of abundant organic breakdown products during the post-spawn period would encourage the growth of large porous flocs, which would increase their settling rates and availability to the organisms at the gravel-water interface. The 1996 returning sockeye numbers in Forfar and O'Ne-eil Creeks were 7,161 and 9,484 respectively, representing a significant biomass of decaying fish in these lower reaches of the river. This seasonal pulse of biological breakdown products would encourage the growth of algae and bacteria, both potentially good sources of organic substances that facilitate floc formation. Currently the limiting factor for extensive floc formation, at this time of year, could be seen to be the availability of fine inorganic material.

The sources of suspended materials collected in the springmelt flow would include detritus and inorganic sediments from the floodplain, channel bank sediments and road ditch drainage in the lower snow-free portion of the watershed. At the upper elevations the overwintering channel-stored material would also be a source. As the first flush of the springmelt was not sampled in 1997 the compact aggregates that would have been released from overwinter gravel storage in the lower spawning reaches were not observed in the samples. Biofilms that exist on the overwintering channel bed would likely have been removed from the downstream reaches in the first flush, but as the hydrograph peak sampled was associated with the snowmelt from higher elevations in the watershed the low density composite particles may represent the upstream channel-stored sediment and biofilms delivered at this time. Droppo et al. (1998) noted significant differences in settling rates of composite particles of the same size on the rising and falling limb of the springmelt flows in a southeastern Ontario stream. Densities were noted to be higher on the falling limb reflecting a depletion of organic matter sources for the sediment structures over the course of the high flow event. The springmelt particles that exhibit densities exceeding those of post-spawn (1.08 g/cm^3) are all smaller than 400 µm. These smaller, denser, composite particles are presumably composed of greater proportions of inorganic sediment that would be delivered from channel banks, floodplains and ditch runoff.

Note that no composite particles greater than 712 microns are noted in the springmelt samples (Fig. 3). The fact that no large, low-density, fast settling particles are delivered from the watershed indicates that the post-spawn composite particles are likely generated in-situ. If these large low-density aggregates, noted to be stored in the gravels after spawning, were actually delivered from the banks, ditches or floodplain they should also be represented in the springmelt samples. While larger low-density particles delivered from the watershed could get broken into smaller flocs in the higher flows it is likely that this would also occur if they were delivered to the stream in high summer stormflows. If they were aggregates of predominantly inorganic material from banks or ditch exposures they would also exhibit much greater densities. It would seem then that the composite, compact, low-density particles observed in post-spawn period are being generated in-situ.

Management Implications

The introduction of fine sediment during periods of high biological activity such as post-spawn results in in-stream flocculation, generating composite particles up to 14 times as large as the maximum size of the constituent particles. Resultant changes in the fall velocities of the fine sediment and its associated organic matter, means that some of this material is delivered to the gravels where it is stored. It appears that during this period of storage an alteration of particle structure occurs resulting in more compact aggregates. While these particles are not very dense, their fall velocity is increased, as it is regulated by the large size and compact shape. These fast settling particles, generated from the processing of flocs derived from fine inorganic sediments in association with an influx of organic matter from salmon decay, will tend to settle back quickly to the channel bed during low flows even after they are resuspended by fish cleaning their redds.

This implies that the stored aggregates will move downstream through the system much like the sands, being resuspended, carried a short distance with the flow, settled out and potentially resuspended later again. These gravel-stored fines are not moving quickly through the system, as they require some disturbance of the gravels to entrain them into the water column. Therefore if we introduce more fines, especially during periods of high biological activity such as fish die-off, we risk storing large amounts in the gravels. This is problematic as they act to smother the eggs, but as well since their organic matter is being processed in the gravels, an increased biological oxygen demand would be exerted, potentially depriving the incubating fish eggs of required oxygen.

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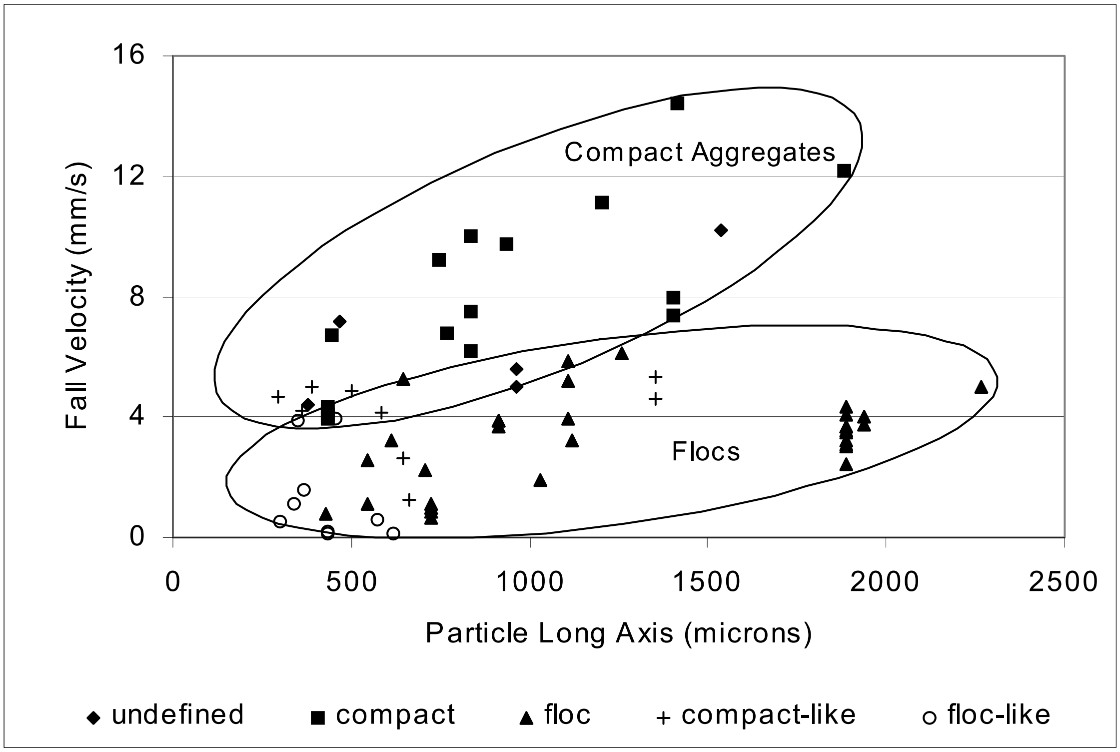

Fig. 1. Fall velocities of suspended and gravel stored particles from Forfar and O'Ne-ell Creeks. The fall velocities were estimated using a field based settling chamber in August, 1996 following intensive spawning.

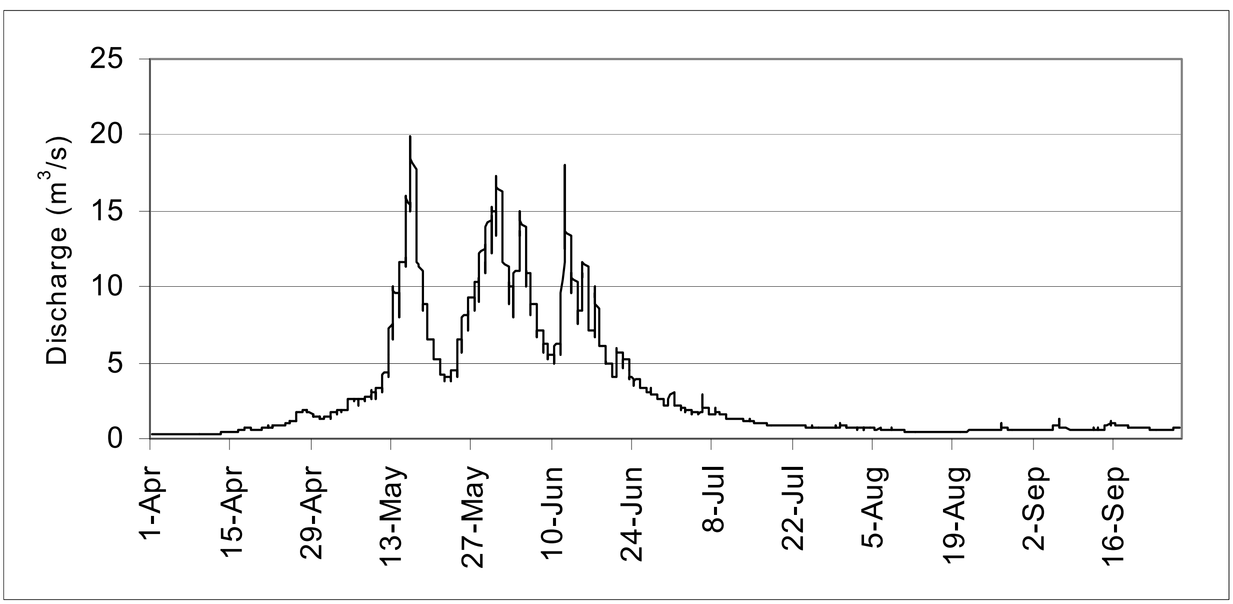

Fig. 2. Hydrograph for O'Ne-ell Creek, 1997. Measured at the bridge crossing, approximately 1700 m upstream of the stream mouth.

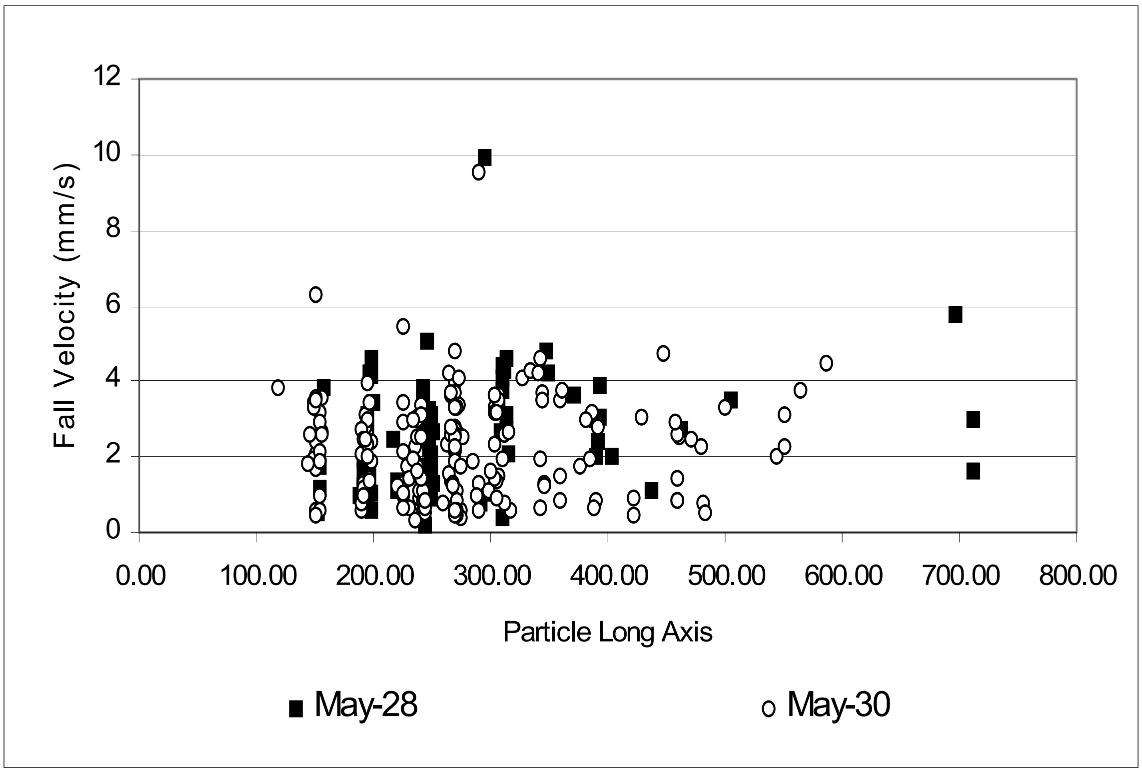

Fig. 3. Fall velocities of particles sampled on the rising limb of a springmelt storm in O'Ne-ell Creek, May 1997.

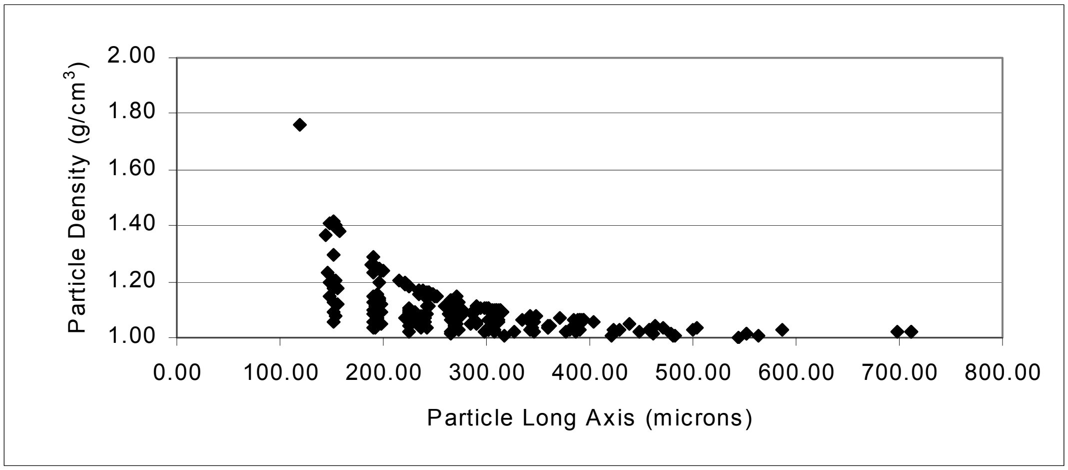

Fig. 4a. Calculated particle densities of springmelt particles in O’Ne-ell Creek.

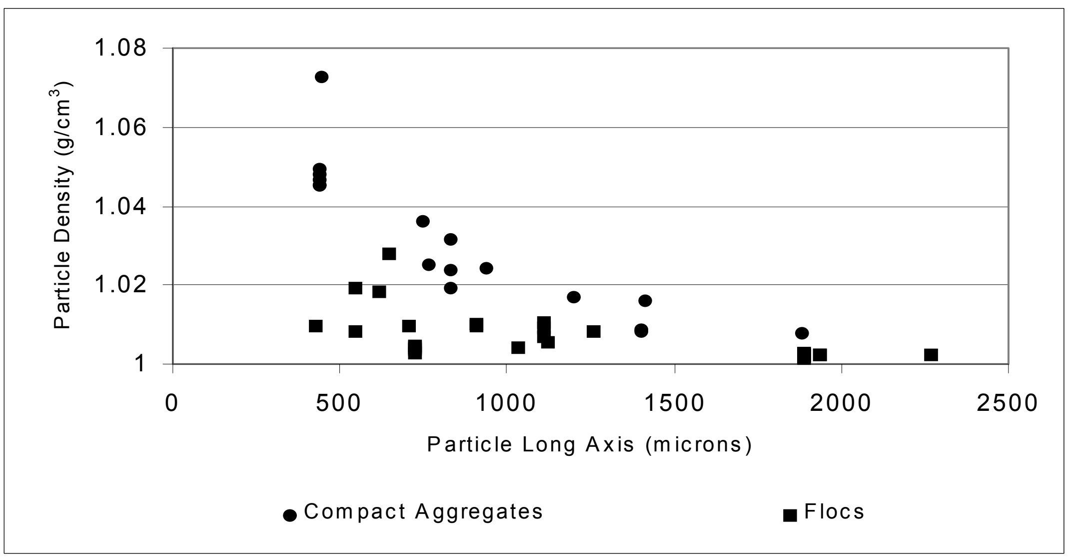

Fig. 4b. Calculated densities of suspended and gravel stored particles sampled in the post-spawning period in August 1996 from O’Ne-ell and Forfar Creeks.

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Effects of Suspended Sediment on Salmon Egg Fertilization

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INTRODUCTION

Many studies have examined the effects of suspended sediments in streams on adult and juvenile Pacific salmon, *Oncorhynchus spp.* (reviewed by Newcombe and Jensen 1996), eggs during incubation, the subsequent alevin development (reviewed by Chapman 1988), and the physical alterations to fish habitat (Anderson et al. 1996). Conspicuously lacking are studies on the potential interference of suspended sediments on the spawning and fertilization process. Salmon are oviparous; gametes combine outside the body cavity, making them susceptible to external environmental influences that might impede egg-sperm contact. Acute concentrations of suspended sediments in streams during spawning could potentially affect fertilization success. Negative effects on egg fertilization success rates would influence salmon populations by depressing recruitment rates to the next generation.

Recent declines in many stocks of salmon caused by overfishing, habitat destruction, and poor marine survival may leave the remaining populations highly sensitive to further environmental stresses or habitat impacts (Bradford and Irvine 2000). The freshwater spawning, incubation, and rearing habitats of salmon are particularly sensitive to land-use impacts which can alter sediment loading to fish streams and rivers. Anthropogenic influences such as placer mining (Birtwell 1999), road building and wood harvesting (Slaney et al. 1977; Beschta 1978) are all risk factors for sudden, high sediment inputs into lotic systems. Naturally unstable surficial soil formations (e.g., unstable clay banks), and catastrophic events (e.g., volcanic eruptions; Martin et al. 1984) can also deliver significant amounts of sediment into streams.

There could be different mechanisms that are involved in fertilization-sediment interactions. Suspended sediment may reduce and interfere with sperm motility, preventing contact with the egg micropyle. Fertilization could also be impaired by particles binding to the egg surface, excluding sperm-egg contact.

The purpose of this project is to address the lack of research on egg fertilization-suspended sediment interactions by testing the effect of suspended sediment on salmonid fertilization success. This was done by simulating the fertilization process in a controlled flow flume using different concentrations of suspended particles. The concentration of suspended sediment at which negative effects occur may differ with sediment type, water conditions, and salmon stock or species. Experiments utilized rainbow trout (*Oncorhynchus mykiss*) and sockeye salmon (*O. nerka*) gametes, and involved incubations in laboratory and natural settings.

METHODS

To date two preliminary experiments have been conducted using two different salmon species and employing two slightly different methodologies. Sediment used was standardized across experiments by using commercial clay derived from natural illite, ball clay, and kaolinite deposits (Plainsman clay: H550). This product allowed consistency in sediment type and suspended sediment concentrations between different experiments. Silt and clay particles are defined as those less than 0.062 mm (Wentworth 1922), and the experimental clay had 88% particles (by weight) less than 0.053 mm. Other suspended sediment and salmon studies used sediments collected both from the Fraser River directly (Servizi and Martens 1987, 1991) and anthropogenically produced sediments (Lake and Hinch 1999). In this study determination of total suspended sediment concentrations was done following the procedures from Standard Methods (1995).

Experimental results were analysed with JMP IN statistical analysis software (version 3.2.1; SAS Institute Inc.) using one-way analysis of variance (ANOVA) to test the hypothesis that the mean fertilization success rate was the same for all treatment and control groups.

Laboratory experiments at Cultus Lake: Experiment One

A water flume was constructed at the Fisheries and Oceans Research Laboratory at Cultus Lake to conduct fertilization experiments under different levels of suspended sediment. The apparatus consisted of two 320-L header tanks each drained by 5 cm diameter ABS pipe into a 20 cm wide x 15 cm deep x 150 cm long plexiglas flume. Valves at the outlet of each tank and the inlet of the flume were used to adjust water input. One header tank was filled with clear water and the other with water and 8.2 kg of sediment. Sediment was pre-suspended in eight 20 L buckets (filled with 15 L water) each with approximately 1 kg sediment. A 2.5 cm diameter air hose attached to the bottom of the tank allowed air from a compressor to bubble through, agitating the water and maintaining suspension of all but the largest particles.

Rainbow trout gametes were obtained from the Pennask Lake Provincial egg collection station and transported to the Cultus Laboratory. Sperm was stripped onsite at the Pennask station from males by anterior to posterior abdominal massage. Sperm from thirty-two males was collected into separate ‘whirlpack’ plastic bags to allow for independent testing of viability before pooling. A total of sixteen females were stripped into metal bowls, also by anterior to posterior abdominal massage. Eggs from four females were pooled in four groups into clean plastic containers. Care was taken at all steps to ensure no water contaminated the gametes, and the gametes were stored in a cool, insulated container. Experiments were completed within 24 hours of gamete collection.

Sperm was checked for motility using a compound microscope at 125x magnification. A drop of sperm was placed on a microscope slide with a drop of water to activate the sperm. A slide cover was added and the sperm was examined for actively motile cells under the microscope. Each of the sperm samples from the thirty-two males was examined and only one particular sample did not show cell activation, possibly due to earlier activation from water contamination. Sperm that demonstrated good motility was pooled (n=31 samples), but the small amount of sperm collected limited the volume used in each trial to 0.10 mL. A completely randomized design (CRD) was used; however, only sixteen trials were run due to time constraints (4 replicates of control and ‘high’ treatments, 5 replicates of ‘low’, and 3 replicates of ‘medium’ treatments). The four treatment concentrations of suspended sediment were achieved by adjusting each tank outlet value according to Table 1. One control dry fertilization, swirling milt with eggs in a weigh boat, was done after all trials were complete.

Water flow in the flume was started (velocity = 0.22 $m \cdot s^{-1}$) and pre-counted batches of 125 eggs were placed into a submerged tray in the center of the flume where flow was laminar. Two 1.0 mL syringes each containing 0.05 mL sperm were used to distribute the milt over the eggs in the flume. Syringes were placed in a holder spanning the flume so that the distance between eggs and needle tip was consistently 0.5 cm for all trials. The plunger of each syringe was simultaneously drawn up to the 0.5 mL mark with flume water then immediately pushed down at a steady rate, discharging the sperm into the water flowing over the eggs. The eggs remained in the water for 30 seconds post-milt discharge before removal. A grab sample of water from the flume was taken for suspended sediment concentration analysis. Eggs from each completed fertilization trial were placed directly into incubation baskets and arranged in a Heath tray in the same CRD order. The eggs were incubated undisturbed until eyed stage (Velsen 1980), then removed and counted. Fertilization success was determined as the percentage of live eyed eggs to total eggs.

Field experiment on the Stuart Takla system: Experiment two

A portable flume system allowed fertilization tests to be carried out over two days in the field on sockeye salmon gametes from the Early Stuart stock. The flume system was a 10.2 cm diameter ABS pipe (approximately one meter long) cut in half lengthwise creating a trough. A circulating pump drew water from one end and released it back at the top. A plastic straw bundle at the inlet end of the pipe straightened the flow, reduced turbulence, and helped ensure an even distribution of suspended sediment.

Gametes were collected from sockeye spawning in Kynoch creek, a tributary to Middle River immediately south of Takla Lake. Male sperm and female eggs were stripped by anterior to posterior abdominal massage as in the first experiment. Sperm was pooled from three males into screw-cap plastic jars, and each pool was examined for motility before use in fertilization trials. Pooled sperm was checked for motility using a compound microscope at 125x magnification at least 20 minutes after collection. This ensured that water contaminated samples would be detected by lack of motility, and could be excluded from the experiment. Eggs were pooled from three or four females and stored in clean plastic containers. All gametes were stored in an insulated cooler and care was taken to prevent water contamination.

Different sediment concentrations in the flume were attained by mixing 15 L water in each of three buckets containing 30 g, 75 g, and 300 g of sediment, corresponding respectively to ‘low’, ‘medium’, and ‘high’ treatments. An additional bucket with clear creek water was used as a control. Buckets were placed in a shallow part of the stream to keep the water the same temperature as the creek water. Before removal of treatment water for each trial, the specific treatment bucket was stirred vigorously to re-suspend any settled sediment.

All trials, including controls, were treated with the exact same procedure. Each trial used 3 L of water from one of the four buckets. The order of the trials was determined according to a randomized complete block (RCB) design, randomly ordering all four treatments into each block. Treatment water was poured into the flume and the water circulation motor started. While the water circulated (velocity = 0.12 $m \cdot s^{-1}$) and the suspended sediment concentration equilibrated, eggs were carefully weighed (21 g corresponding to approximately 200 eggs) from one of the egg dishes. Pooled sperm was drawn into three 1.0 mL syringes (0.5 mL each) and fitted into a syringe holder. The holder ensured consistent distribution of sperm over the eggs for each trial. Before fertilization, a grab sample of water was taken from the flume for suspended sediment concentration analysis. Eggs were gently transferred to a brass wire tray and placed into the flume water, and the syringe holder placed over the flume with the syringe needle immediately upstream of the egg tray. The plunger of each syringe was simultaneously drawn up to the 1.0 mL mark with flume water and immediately pushed down at a steady rate, discharging the sperm in front of the egg tray so the flow of water carried the sperm through and over the eggs.

After milt discharge, eggs were left in the flume for 40 seconds, removed, and rinsed in clean water and placed in a 500 ml jar for water hardening. At the end of each trial the flume was emptied and rinsed with creek water to remove any remaining sediment in the system. Eggs were allowed to water harden for a minimum of 30 minutes before being counted into egg incubation capsules (see Scrivener 1988) in batches of 50 eggs per capsule. Each trial of 200 eggs was divided into four capsules. Capsules were randomly buried in an artificial redd in the stream at redd depth, 20-25 cm into the gravels (Macdonald 1994) until eyed stage (Velsen 1980). All capsules were removed at the same time, after a minimum of 38 days incubation. Fertilization success was determined by counting eggs as per experiment one, with each trial fertilization success rate being the mean success of all four capsules. Control dry fertilizations (n= 6 on day one, n= 5 on day 2) were performed under optimum conditions (swirling milt with eggs) throughout the procedure to track any possible decline in gamete fertilizability. Forty trials were run over two days, yielding ten replicates of each treatment.

All surplus eggs were fertilized and buried in the stream gravels in man-made redds. A two inch pipe was inserted into the streambed, and gravel built up around the pipe base. Fertilized eggs were poured down the pipe, and remained in the gravel when the pipe was removed.

RESULTS AND DISCUSSION

Laboratory study

Suspended sediment led to significantly lower fertilization rates in experiment one with rainbow trout, but no difference was detectable with sockeye in the second experiment.

Overall mean fertilization success rate for controls of the fixed flume laboratory experiment at the Cultus Lake was 16.6 ± 8.6% (mean ± 95% C.I.). One control trial had egg opacity evident in almost all eggs just a few hours post-fertilization, and 0% survival at the eyed stage. This indicated an extreme shock that was not apparent in other trials with or without sediments (no other trials had 100% mortality), and so it was discarded from the analysis.

The low fertilization success for the control trials was attributed in part to several factors; the small volumes of sperm used with each trial, aging gametes (there was 15 hours between collection and the start of fertilization testing), and excessive egg handling and agitation during the experimental procedure causing mechanical shock mortality (Jensen and Alderdice 1989). Some shock mortality was evident and observed in all trials within a few hours post-fertilization, turning eggs opaque. The dry control fertilization yielded a fertilization success of 72%, and when compared to the later field study (dry control fertilization = 90% $\pm$ 2.5%), shows that the gametes were combining at less than optimal rates.

After initial mixing and particle settling, the concentration in the header tank averaged 11.3 $\pm$ 1.0 $g{\cdot}L^{-1}$. A consistent and precise suspended sediment concentration was maintained for each treatment. The mean concentration in the flume trials is shown in Table 1. Fertilization success results were grouped by treatment (control, low, medium, and high) and an analysis of variance applied. There were statistically significant (ANOVA, $F=5.65$, $p=0.0136$) differences among the mean fertilization rates (Figure 1). A Tukey-Kramer Honest Significant Difference (HSD) test demonstrated that control trial fertilization success was significantly different compared with the sediment trials.

Table 2. Header tank valve positions and resulting suspended sediment concentration ($g{\cdot}L^{-1}$) for each treatment in experiment one.

Treatment	Tank 1 - clear (% valve opening)	Tank 2 – sediment (% valve opening)	Suspended sediment concentration, $g{\cdot}L^{-1}$ (mean $\pm$ 95% C.I.)
Control	100	0	0.7 $\pm$ 1.3
Low	75	25	2.9 $\pm$ 0.7
Medium	25	75	6.3 $\pm$ 0.6
High	0	100	9.3 $\pm$ 1.5

Field studies

Sediment did not detectably affect fertilization success in the field experiment. The mean trial control fertilization rates in the second set of experiments (30.6 $\pm$ 7.5%) were slightly higher than those that occur during natural sockeye spawning (~20% egg to fry survival rates; DFO unpublished data) in Kynoch creek.

Some procedural changes improved gamete survival in experiment two. All trials were done streamside; this allowed a considerable reduction in the time between gamete removal and the start of fertilization tests (less than 2 hours in experiment two versus more than 15 hours in experiment one). Control dry fertilizations (90% $\pm$ 2.5%) were used to track declining gamete fertilizability throughout the procedure on either day, and showed no detectable decay in fertilization rate. Eggs were handled with more care before and after fertilization, as Jensen and Alderdice (1989) and the first experiment showed that unactivated eggs are very susceptible to mechanical shock mortality. The counting of eggs for each trial was done by weight, reducing handling, and virtually eliminating mechanical shock mortality. An egg water-hardening step was also added. Counting eggs into incubation capsules after water hardening reduced mortality due to post-fertilization handling stress. This was evident by the absence of opaque eggs any time post-fertilization, and by higher mean fertilization success rates than experiment one (30.6 ± 7.5% versus 16.6 ± 8.6%).

The ‘high’ concentration of 6.6 $g{\cdot}L^{-1}$ of suspended sediments tested was lower than desired (20 $g{\cdot}L^{-1}$) due to particle settling in the mixing buckets. This concentration may have been too low for direct interference in sperm-egg contact given the high sperm to egg ratio. However, the suspended sediment concentrations achieved in the flume between different treatments were very consistent and precise (0.04 ± 0.02 $g{\cdot}L^{-1}$, 0.80 ± 0.05 $g{\cdot}L^{-1}$, 1.83 ± 0.09 $g{\cdot}L^{-1}$, 6.62 ± 0.23 $g{\cdot}L^{-1}$, corresponding to control, low, medium, and high treatments respectively).

Trial results were pooled by treatment over the two days, and the results (see Figure 2) showed no detectable effect of suspended sediment on egg fertilization success (ANOVA, $F= 0.1738$, $p= 0.9134$). There was considerable variation in fertilization success, especially in the ‘high’ sediment treatments (2.5% to 59.9%). Sockeye males yielded as much as 20 ml of sperm each, so sperm volume was not a limiting factor as in the previous experiment. The inconclusive results could stem from the fact that the volume of sperm per trial (1.5 ml versus 0.10 ml in experiment one) was very high, potentially overwhelming any sediment interference with egg fertilization.

CONCLUSIONS

Literature comparisons show that the mean upper level (‘high’ treatment) of suspended sediment used in experiment two (6.6 $g{\cdot}L^{-1}$) was lower than that naturally found in some highly turbid rivers which can have peaks of 14 to 15 $g{\cdot}L^{-1}$ (Gurnell and Clark 1987). Rivers such as the Red Deer and Peace can have maximum daily suspended sediment concentrations of 11 to 12 $g{\cdot}L^{-1}$ (Anonymous 1980; in Lake and Hinch 1999). Greater concentrations of suspended particles than in experiment two should be tested in future work.

Sperm volume differed considerably between experiments and the high sperm volume in experiment two likely overwhelmed any sediment-fertilization interactions. The required salmonid sperm to egg ratio for fertilization shows that the amount of sperm used in experiment two was ten times greater than that used in experiment one, and one thousand times greater than that needed for fertilization under optimal conditions (Billard et al. 1974; Moccia and Munkittrick 1987). The interaction between sperm volume and suspended sediment concentration will be explored in additional experiments. Ongoing research will include difference salmon species.

Based on the experience gained from these pioneer laboratory and field experiments, the fertilization protocol will be refined to eliminate possible confounding effects of aging gametes, mechanical shock to eggs, and will utilize higher mean concentrations of suspended particles (30 $g{\cdot}L^{-1}$). Consistent and careful gamete handling techniques were critical in providing consistency between trials and experiments. Reducing gamete age at fertilization, prevention of water contamination (ensuring sperm activation and motility), and reduction of mechanical shock mortality to eggs proved to be important factors in the first two experiments.

The mechanisms leading to interference by suspended sediments in salmonid fertilization are likely dependent on a complex interaction between the biological characteristics of the spawning event and gametes, and the physical characteristics of the redd, water, and suspended sediments. The experimental methodologies and results presented in this report are a first attempt to quantify and explore what factors are key considerations and what inherent difficulties are involved in investigating these interactions. Considering the paucity of information and research on this subject coupled with ever increasing anthropogenic encroachment into salmonid habitats with activities that can cause acute increases in suspended sediment (e.g. forestry and road building), research regarding suspended sediment-salmonid fertilization interactions may be important to future management of the species.

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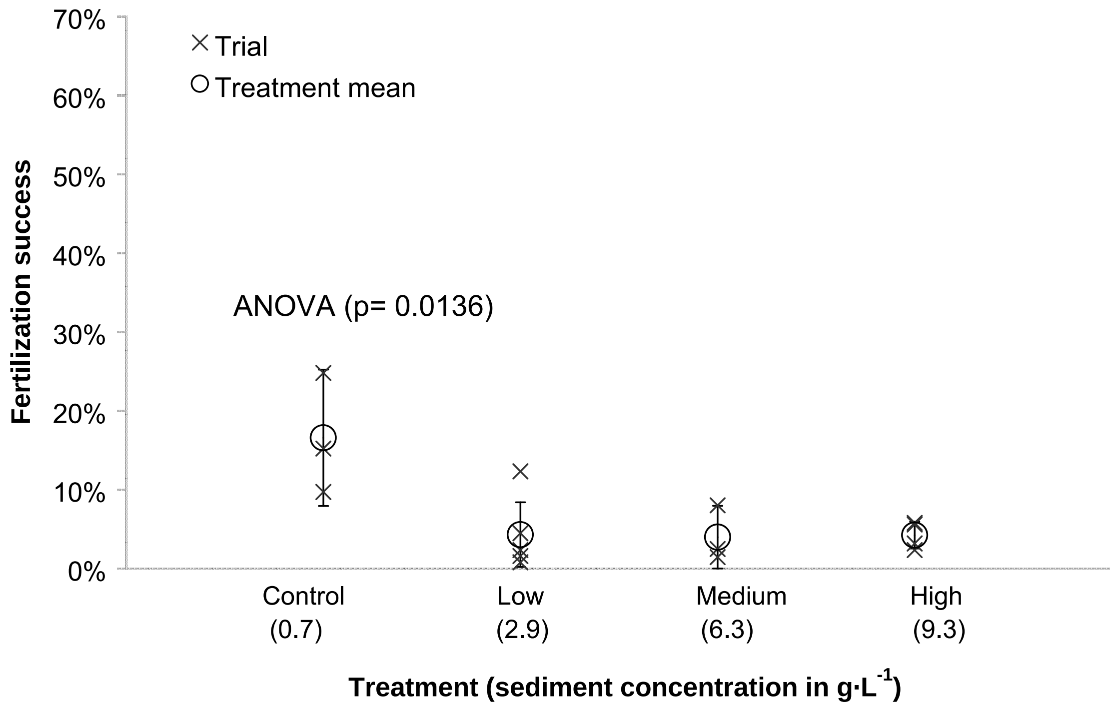

Fig. 1. Experiment one fertilization success rates by treatment with 95% confidence intervals.

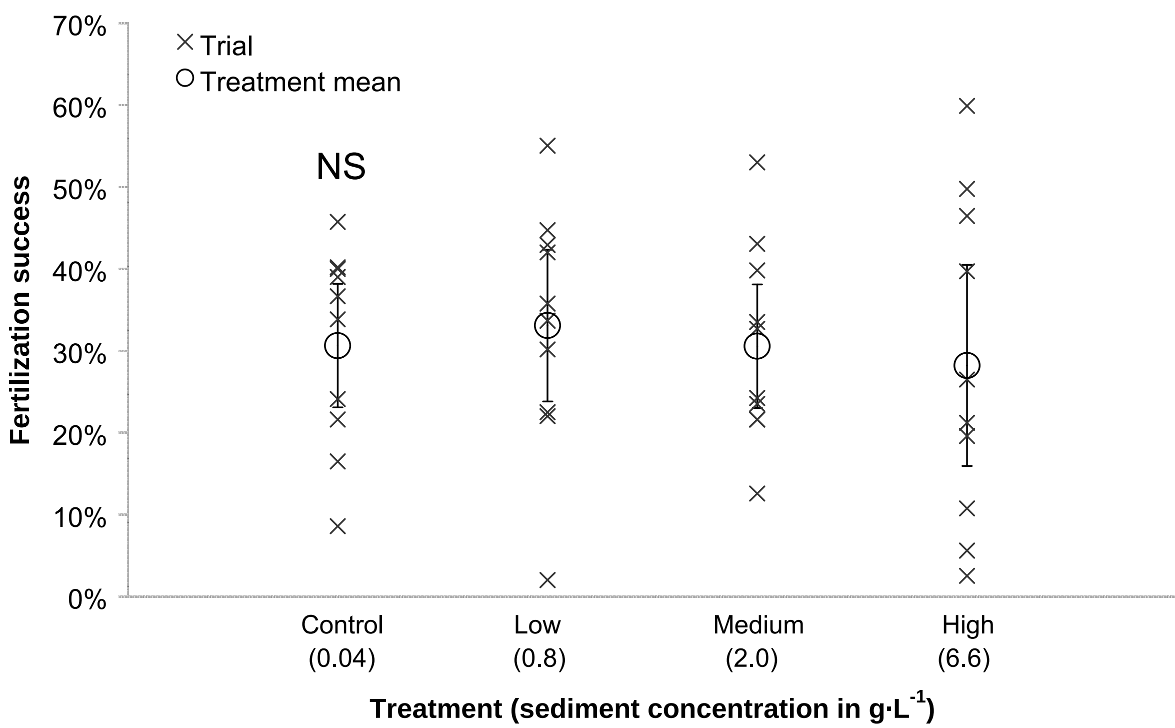

Fig. 2. Experiment two fertilization success rates by treatment with 95% confidence intervals.

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PART III: SMALL STREAM STUDIES

Effects of Riparian Management Strategies on the Hydrology of Small Streams in the Takla Region of British Columbia - Fourth Year Results

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INTRODUCTION

Riparian forests play a vital role in the landscape by protecting water quantity and quality, increasing landscape connectivity and providing plant and wildlife habitat (Brosofske et al. 1997; Gilliam 1994; Gregory et al. 1991). The riparian area has been defined in a number of ways by various individuals (Gregory et al. 1987; Naiman and Decamps 1997; Murphy and Meehan 1991). The British Columbia Forest Practices Code (FPC) offers this definition: "Riparian areas occur next to the banks of streams, lakes and wetlands and include both the area dominated by continuous high moisture content and the adjacent upland vegetation that exerts an influence on it. Riparian ecosystems contain many of the highest value non-timber resources in the natural forest (B.C. Ministry of Forests 1995). The riparian zone may include vegetation outside the zone that is directly influenced by hydrologic conditions if it contributes organic matter (e.g. leaves, wood, dissolved materials) to the floodplain or channel, or if it influences the physical regime of the channel by shading or intercepting sediment flow (Brosofske et al. 1997; Gregory et. al. 1991).

The Forest Practices Code of British Columbia legislates the width of the Riparian Management Area (RMA) that must be left along stream channels on public land. The RMA is comprised of a reserve zone (RRZ) and a management zone (RMZ). As a general rule, no forest harvesting is permitted within the RRZ and forest harvesting within the RMZ must be done in such a way as to maintain the integrity of the RRZ. The width of the RMA, and its two components (RRZ and RMZ), is based on the width of the stream channel, measured at bankfull height. The smallest fish bearing stream channel has been given the label of "S4" and has a width less than 1.5 m. The smallest non-fish bearing stream channel has been labelled "S6" and has a width of less than 3.0 m. For both S4 and S6 streams the legislation states that no riparian reserve zone is required, but it designates the width of the RMZ as 30 m and 20 m respectively. The recommended best management practice (which is not a legislated requirement) for S4 streams is to retain all trees within 10 m of the streambank on windfirm sites (B.C. Ministry of Forests 1995). Where these practices cannot be achieved, due to moderate or high windthrow hazard, a series of alternate practices are recommended which essentially amount to maintaining as many windfirm trees as possible. For S6 streams the recommended practices are to retain sufficient streamside vegetation in order to maintain important wildlife habitat values, streambank and channel stability, and streambank shading. The application of these guidelines is flexible and subject to various interpretations and consequently has proved to be controversial. The effectiveness of these guidelines, for protecting aquatic ecosystems, has been questioned. This project was initiated in response to the lack of knowledge about these ecosystems and the concerns expressed about the hydrologic response to forest harvesting in the vicinity of S4 and S6 streams in the central interior of BC.

METHODS

Experimental Design

The experimental design is based on the classical paired watershed approach (Bates and Henry 1928). For this experiment, three small, first order, adjacent drainage basins were chosen in the headwaters of the Batiste watershed. Two of the drainage basins were harvested with different riparian treatments and the third basin has remained unlogged to serve as the experimental control. The two experimental treatments consist of harvesting about 40% of the watershed with the following riparian treatments.

1) An “aggressive” riparian harvest leaving a limited number of trees adjacent to the stream channel.

2) A “conservative” riparian harvest, retaining all trees within a distance of a least 10 m of the streambank (Fig. 1).

These sites were instrumented in the fall of 1995. Pre-harvesting data were collected until early winter of 1996. The sites were harvested in January 1997. Post – harvesting data have been collected at these sites since that time. The initial results, describing the effects of the first year after harvest are presented in Beaudry (1998a). This report summarizes all of the results to date, including the second and third years after harvest (i.e. spring 1998 and 1999).

Study Sites

This study was conducted in three small headwater tributaries of Baptiste Creek, located near the north end of Trembleur Lake in the Fort St. James Forest District. Present day landforms and surficial materials are largely a result of the last Pleistocene glaciation, known as the Fraser Glaciation. Terrain throughout most of the watershed is gently to moderately sloping, and hummocky or rolling in detail. Terrain is generally bedrock-controlled, including hummocks and glacial lineations (Collett and Ryder 1997). The three study creeks are identified as B3, B4 and B5. The physical characteristics of each of these three watersheds are provided in Table 1. Information on forest types, logging activity and experimental treatments for each of the watersheds is provided in Table 2.

Table 1. Physical characteristics of the three study watersheds.

	Size (ha)	Aspect	Elevation range (m)	Total channel length (m)	Average channel gradient (%)	Average channel width (m)	Surficial materials[1]	Dominant Texture	Surface erosion potential
B3	42.5	NW	980 – 1340	1373	26.2	0.60	Basal till	Silty-sand	M to H
B4	48	NW	980 – 1340	1200	30	0.85	Basal till	Silty-sand	M to H
B5	150	N	980 - 1300	2093	6.7	1.38	Basal till	Silty sand	M

[1] Typically, the basil till consists of a fine grained matrix of sand (about 45%), silt (about 37%) and clay (about 18%) surrounding and supporting clasts of a variety of sizes, shapes and rock types. It is highly consolidated, and commonly highly cohesive, making it one of the strongest surficial materials.

Table 2. Forest types and logging activity for the three study watersheds.

	Dominant tree species[1]	Average crown closure (%)	Avg. dominant tree height (m)	% of watershed clearcut harvested	Length of channel harvested (m)	Channel gradient within cutblock (%)	Riparian treatment[2]
B3	Bl, Sx	46-65	28.5 - 37.5	38	900	30	Conservative
B4	Bl, Sx, Pl	56-75	19.5 - 28.4	Unlogged	Unlogged	Unlogged	Unlogged
B5	Bl, Sx, Pl	36-56	28.5 - 37.4	40	1060	4.8	Aggressive

[1] Bl= *Abies lasiocarpa*, Sx = *Picea glauca x engelmannii*, Pl = *Pinus contorta*
[2] In the aggressive treatment all merchantable timber was removed within the RMA. Merchantable timber is defined as diameter breast height (DBH) >15cm for Pl and >20cm for Sx and Bl. In the conservative treatment only timber greater than 30 cm DBH was removed and full retention in difficult access or steep gully areas for a distance of 30 m from the stream edge.

Instrumentation

At each of the three watersheds, a 9 inch Parshall flume (Parshall 1936) was installed at the mouth of the stream, immediately below the cutblock. Also, a 6 inch Parshall flume was installed on B5 and B3 just above the upper edge of their respective cutblocks. Stage height in the flume was measured using a 1 m capacitative water depth probe manufactured by Unidata America. The concentration of suspended sediments in the water column, passing through the flume, was measured using a combination of an optical turbidity probe (OBS3, manufactured by D & A Instrument Company) and a pump water sampler (Sigma 800SL automatic liquid sampler, manufactured by American Sigma). Measurements of stage height and turbidity were made every 10 seconds, averaged over a 15 minute period and recorded on an electronic data logger (Starlogger model 6004-1, manufactured by Unidata America). The electric pump sampler and the collection of water samples were electronically triggered by the turbidity readings. Water samples were collected only during pre-set levels of turbidity, allowing intensive sampling during periods of high turbidity and limited (and unnecessary) sampling during periods of clear flows. The water samples, once analysed, were used to establish a relationship between the concentration of suspended solids and the turbidity readings obtained from the turbidity probes. Air temperature and rainfall intensities were measured at the site and recorded by a data-logger.

During the snow melt period the monitoring sites were visited about once a week. Access to the sites was usually by helicopter from Fort St. James. During these site visits a snow survey was completed in a reference site located next to the rainfall gauge. Also at this time, the instruments were maintained, the water samples were collected from the pump sampler and labeled, the sampler was reset, and a variety of visual observations were made and recorded in the field book. In 1997 and 1998 we walked the length of the harvested creeks, during every site visit, in an attempt to locate any point sources of sediment entering the creeks. Spot measurements of turbidity were made at approximately every 100 m, and above and below every crossing, using a handheld turbidity probe.

Data Analysis

Stream discharge, in each of the Parshall flumes, was computed using the average 15 minute stage height and the following equations:

$$\text{for 9 inch flume: discharge } (m^3/sec) = 3.07(S * 3.281)^{1.53} * 0.0283 \qquad (1)$$

$$\text{for 6 inch flume: discharge } (m^3/sec) = 2.06(S * 3.281)^{1.58} * 0.0283 \qquad (2)$$

where S equals the 15 minute average stage height (m) measured at 1/3 of the distance between the top of the flume and the start of the hydraulic jump (Parshall 1936). The 15 minute averages are obtained from 90 measurements (i.e. one measurement every 10 seconds).

The relationship between the raw OBS-3 turbidity values recorded by the data logger and the concentration of suspended sediments (ppm) obtained from the water samples was obtained using a simple linear regression. The best fit model, for all flows, yielded the following regression equation:

$$TSS = 0.44\,(OBS) + 1.25 \qquad R^2 = 0.64 \qquad S.E. = 4.17 \quad n=32 \qquad (3)$$

where TSS is the concentration of total suspended solids (ppm), and OBS is the signal (mv) sent by the turbidity probe to the datalogger. Equation 3 was used for all sites to transform the raw data provided by the OBS turbidity probe to values of concentration of suspended sediments. This method for calibrating the OBS signal has been successful in other projects (Beaudry 1998b; Beaudry and Sloat 1999).

The 15 minute data, for both discharge and TSS, were averaged over a 24 hour period to obtain average daily values during the snowmelt period. A regression analysis was performed using the 1996 daily data to generate calibration models to describe the pre-harvest relationships between the control stream (B4) and the two treatment streams (B3 and B5). This was done for both stream discharge and TSS. The pre-harvest calibration models were then used with the 1997, 1998 and 1999 B4 stream data to obtain “predicted” estimates for B3 and B5. These “predicted” values, which are simulations of unharvested conditions for B3 and B5, were compared with the measured values of discharge and TSS obtained from the harvested watersheds. This was done to assess the effect of forest harvesting activities on stream discharge and TSS. Deviations from the calibration models were considered to be statistically significant and attributed to the harvesting if they exceeded 95% prediction intervals about the calibration models (Hornbeck et al. 1993). The 95% prediction intervals for each individual estimate of discharge or TSS were calculated using the method described in Helsel and Hirsch (1992). The data from the flumes located at the top of B3 and B5 streams were not used in this analysis. This was because the flows at the top of B3 were almost always too low to be measured accurately by our instrumentation and the site at the top of B5 experienced some serious flume leakage problems in both 1996 and 1997.

RESULTS AND DISCUSSIONS

Snowpack Conditions

The rates of spring snowmelt at the study site are presented in Figure 2. During the four years in which snow surveys have been performed in the Baptiste watershed, the smallest snowpack was registered in 1998, while the deepest snowpack was in 1997. For all years, except 1998, the snow disappeared from the reference site at the end of May. In 1998, the snow was gone by the first week of May.

Discharge and Suspended Sediments during the Pre-harvest Period

The snowmelt hydrograph and sedi-graph for 1996 are presented in Figures 3 and 4. These are the pre-harvest data used to calibrate the treatment watersheds to the control watershed and develop a pre-harvest regression model. The model parameters are provided in Table 3. For both discharge and suspended sediments, the two treatment watersheds are very well correlated to the control watershed, thus providing excellent pre-treatment models. Prior to harvesting, the peak snowmelt discharge in B5 is about 2.2 times greater than the peak discharge in the control watershed. For B3, the peak discharge is about a third of the peak discharge in the control stream (Table 3). Concentrations of suspended sediments, during the pre-harvest snowmelt period, are almost identical for all three streams (Fig. 4).

Table 3. Regression equations and statistical parameters for 1996 (pre-harvest period).

Regressions	Equation	R^2	n	Lower 95% Confidence Interval	Upper 95% Confidence Interval	P-Value Slope
Discharge B3 vs. B4	y = 0.328 (x) + 0.0055	.92	25	0.2857	0.3719	$8.00\ E^{-14}$
Discharge B5 vs. B4	y = 2.213 (x) + 0.0133	.98	25	2.0807	2.3452	$4.04\ E^{-05}$
TSS B3 vs. B4	y = 1.079 (x) – 1.33	.84	25	0.9176	1.1322	$1.77\ E^{-10}$
TSS B5 vs. B4	y = 1.021 (x) – 0.48	.85	25	0.8390	1.2031	$4.30\ E^{-10}$

Regression Analysis - Stream Discharge

Linear regression models were developed to describe the relationships between the stream discharge for both treatment streams (B3 and B5) and the control stream (B4) for each of the four years of measurements. The pre-harvest models are described in Table 3 and the parameters and statistics for the three post harvest models are provided in Tables 4 and 5. The graphical representations of these relationships are provided in Figure 5.

For both B3 and B5 watersheds, there was no significant increase in stream discharge for the first year after harvest (i.e. 1997). In 1998, there were large increases in snowmelt peak discharge at both B3 and B5 (41% and 78% respectively). However, the increase was statistically significant only for B5. In 1998, the snowpack was relatively shallow and the duration of snowmelt runoff was short, consequently the sample size (i.e. the number of days of snowmelt runoff) was small. This resulted in very wide confidence intervals for the regression model and consequently a weaker model. This partially explains the lack of statistical significance despite the large increase in stream discharge.

In 1999, there was a statistically significant increase in peak stream discharge for both B3 and B5. The increases were approximately 52% and 75% for B3 and B5 respectively. These increases are similar to those registered for 1998 and suggest that the removal of about 40% of the forest cover causes a significant increase in peak stream discharge for small headwater watersheds that are dominated by snowmelt runoff.

Table 4. Regression equations and statistical parameters for B3 vs. B4 stream discharge.

YEAR	Discharge regression equation	R^2	nn	Lower 95% Confidence Interval	Upper 95% Confidence Interval	P-Value Slope
1997	B3 = 0.285*(B4) + 0.0081	0.89	33	0.2476	0.3218	$2.82\ E^{-16}$
1998	B3 = 0.514*(B4) – 0.00062	0.70	13	0.2918	0.7359	$3.48\ E^{-4}$
1999	B3 = 0.536*(B4) +0.00239	0.80	43	0.4525	0.6202	$4.80\ E^{-16}$

Table 5. Regression equations and statistical parameters for B5 vs. B4 stream discharge.

YEAR	Discharge regression equation	R^2	nn	Lower 95% Confidence Interval	Upper 95% Confidence Interval	P-Value Slope
1997	B5 = 1.960*(B4) + 0.039	0.80	28	1.567	2.348	$1.11\ E^{-10}$
1998	B5 = 4.219*(B4) – 0.019	0.82	13	2.8933	5.5438	$2.25\ E^{-5}$
1999	B5 = 3.875*(B4) + 0.0231	0.79	38	3.1998	4.5496	$9.16\ E^{-14}$

Regression Analysis – Total Suspended Sediment (TSS)

Similarly to stream discharge, linear regression models were developed to describe the relationships between the concentration of suspended sediments for both treatment streams (B3 and B5) and the control stream (B4), for each of the four years of measurements. The pre-harvest models are described in Table 3 and the parameters and statistics for the three post harvest models are provided in Tables 6 and 7. The graphical representations of these relationships are provided in Figure 6.

For B3 watershed, there was a significant increase of 21% in the peak concentration of suspended sediments for the first year after harvest (i.e. 1997). In 1998, there was no significant increase and in 1999 the results suggest that there was actually a decrease in TSS, relative to the unharvested model. It is unknown why the TSS in B3 watershed would actually be lower in the third year after harvest, compared to the unharvested model.

For B5 watershed, there was a large increase in TSS during the first 5 days of the 1997 snowmelt runoff (114%). For the remainder of the snowmelt period the concentration of suspended sediments was very similar to the pre-harvest model (Fig. 8). Because the large increase only lasted for a few days, the statistical relationship between B5 and B4 TSS for the entire 1997 snowmelt period is poor (R^2= 0.58, Table 7). The relationship improves substantially, and is statistically significant, when only the first six days of the snowmelt period are considered (i.e. the period when a large increase was detected).

For both 1998 and 1999, there were significant increases in the concentration of suspended sediments during the peak snowmelt runoff in B5 watershed (93% and 67% respectively). From the three years of post-harvest data collected to date it appears that the effects of the treatments on increases in suspended sediment concentrations in B5 watershed is decreasing annually.

Table 6. Regression equations and statistical parameters for B3 vs. B4 TSS.

YEAR	TSS Regression equation	R^2	nn	Lower 95% Confidence Interval	Upper 95% Confidence Interval	P-Value Slope
1997	B3 = 1.40*(B4) + 0.0081	0.92	27	1.2322	1.5686	2.46 E $^{-15}$
1998	B3 = 1.120*(B4) - 0.49	0.93	13	0.9176	1.3216	9.83 E $^{-08}$
1999	B3 = 0.567*(B4) - 1.163	0.85	11	0.3874	0.7468	5.43 E $^{-05}$

Table 7. Regression equations and statistical parameters for B5 vs. B4 TSS.

YEAR	TSS Regression equation	R^2	nn	Lower 95% Confidence Interval	Upper 95% Confidence Interval	P-Value Slope
1997	B5 = 1.269*(B4) + 0.82	0.58	28	0.8212	1.7171	4.38 E $^{-06}$
1997 (peak only)	B5 = 2.147*(B4) - 0.0075	0.91	6	1.210	3.084	3.13 E $^{-03}$
1998	B5 = 2.323*(B4) - 11.23	0.81	13	1.5746	3.0718	2.84 E $^{-04}$
1999	B5 = 1.685*(B4) - 0.222	0.90	11	1.2667	2.1031	7.70 E $^{-06}$

Effects of the Forest Harvesting Treatments on Stream Discharge

In Figure 7 the measured streamflow hydrograph is compared with the predicted values for each of the two treatment watersheds, for each of the three years post-harvest. The “predicted” values represent the expected streamflow that would have occurred if there had been no forest harvesting. These were calculated using the pre-harvest regression models (Table 3) and the streamflow values measured at B4 (i.e. the control) for each of the three post-harvest years.

In 1997, there was an insignificant difference between the predicted streamflows and the measured streamflows for both B3 and B5. This lack of treatment effect was attributed to two main factors. 1) Since harvesting was not completed until the end of

January 1997, the effect of increasing snow accumulation in the cutblock was limited by the presence of a forest canopy for part of the snow accumulation period (i.e. November to January). 2) The mulching and insulation effect caused by the logging slash left on the snowpack. This deep insulating layer, which covered the snowpack over the entire cutblock, would have considerably reduced the effects of canopy removal on increased snowmelt rates. This no doubt had a significant mitigating effect on increases in spring runoff during the first year after harvest.

In the spring of 1998 there was no mulching effect and without this insulation layer, the spring peak flows of 1998 were significantly larger than the predicted discharges, and this for both treatment watersheds. These results provide an excellent example of the effects of forest removal on changes to snowmelt peakflows. The reduction in forest cover results in both an increase in the amount of snow accumulation in the clearcut and an increase in the amount of energy available for melting snow, which in theory is the cause of the increased peakflows. The additional energy comes from an increase in direct solar radiation and increases in both advected and sensible heat to the snowpack (Beaudry and Golding 1987).

The 1999 snowmelt period was quite different from the preceding three years as it began slowly and was extended for a lengthy period of time. The snowmelt discharge began in mid-April and proceeded at a slow rate until mid-May, at which time discharges increased but did not reach high volumes. Peak discharges in 1999 were the lowest of the four years measured (Fig. 7). Despite the “atypical” snowmelt pattern, the measured peak discharges for 1999 were greater than the predicted discharges for both treatment watersheds. Consequently, the treatment effect (i.e. increased snowmelt peakflows) continues to be detectable.

The increase in peak discharge is greater in B5 watershed than it is in B3 watershed for both 1998 and 1999. This is attributed to two factors. The primary factor is associated with the differences in the slope and aspect of the two watersheds. Although both watersheds are essentially north facing, the shallower slope of B5 (7% for B5 vs. 26% for B3) results in a greater exposure to direct solar radiation and consequently more energy for melting snow. In B3 watershed it appears that the steeper northern slope causes a small delay in peak snowmelt discharge, compared to B5, and a smaller difference with the unharvested scenario of B3 (Fig. 7).

A secondary factor could be attributed to the differences in riparian treatments between B3 and B5. The riparian areas of a watershed are often considered to be an area of disproportionately high importance to peakflow runoff. According to the variable source area concept, the stream surface, channel banks and surrounding riparian areas and zones of restricted conductivity (e.g. road surfaces at stream crossings) are the main source of peakflow runoff (Hewlitt and Hibbert 1967). Thus, it is reasonable to speculate that, in addition to the clearcut effect, the substantial reduction in forest cover in the riparian area of B5 mainstem was a significant contributing factor to the large increase in early snowmelt runoff.

Effects of the Forest Harvesting Treatments on the Concentration of Suspended Sediments

In Figure 8 the measured concentration of suspended sediments (sedigraph) is compared with the predicted values for each of the two treatment watersheds, for each of the three years post-harvest. The “predicted” values represent the expected concentration of sediment that would have occurred if there had been no forest harvesting. These were calculated using the pre-harvest regression models (Table 3) and the TSS values measured at B4 (i.e. the control) for each of the three post-harvest years.

In 1997, there was a statistically significant, but small, increase in the average daily concentration of suspended sediment at B3, during the spring peak flows. In 1998, there was no statistical difference (ρ=0.05) between the measured and predicted values. These results are intuitively reasonable since the harvesting activities in B3 watershed (i.e. the conservative treatment) did not create any significant sediment sources that were linked directly to the stream channel. All of the harvesting operations were located well away from the active channel, and the channel was protected by a large riparian buffer. Only one stream crossing was built above the monitoring site, and this was a skid crossing built with logs. This kind of crossing construction usually does not require any excavation, either for the installation or the removal, consequently minimal soil disturbance occurs. This absence of soil disturbance near the stream channel is reflected in the results, i.e. no significant change in sediment concentrations after forest harvesting for B3 watershed.

In B5 there was a significant increase in TSS for both 1997 and 1998. The spot measurements of turbidity, taken along the length of the creek, clearly identified the lower stream crossing of B5 as the major source of sediment responsible for the measured increases at the mouth of the creek. In 1997 the increase in suspended sediments was attributed to a large landing located immediately above the lower B5 crossing. The snow on this landing melted early in the season and generated a large amount of surface runoff that transported sediment down the road surface and into B5 creek. No other significant point sources of sediment were found along the entire stream channel. The aggressive riparian treatment removed most of the forest cover along the stream channel, but the harvesting and yarding equipment did not infringe upon a regulated 5-m machine free buffer zone on either side of the stream channel. The machine free zone appears to have been adequate to protect the stream channel from direct physical disturbance of the bank and prevent transport of sediment from the cutblock into the stream channel.

In 1998, there was also a statistically significant increase at B5, however this time it was much larger than in 1997, with a maximum increase of 94%. We attribute this large increase in the concentration of suspended sediments to the removal (i.e. de-activation) of the stream crossing, located about 200 m above the B5 monitoring station. The removal of the culvert required substantial excavation and soil disturbance. The culvert was removed in the fall of 1997. Steep, unstable cutbanks of approximately 3 meters in height were left on either side of the channel (Fig. 9). No erosion or sediment control measures were put in place to control the delivery of fine sediment to the stream channel. The movement of fine material from these cutbanks into the stream channel resulted in the high sediment concentrations observed in the spring of 1998.

In 1999, there was only a small increase in the concentration of suspended sediments at the mouth of B5. At B3 there was no increase, and the data actually suggest that there was a small decrease in suspended sediments. These results could be interpreted as meaning that the increases in suspended sediments, measured in

1997 and 1998, were short lived and are slowly disappearing as the fluvial system and the access trails stabilize following the harvesting disturbances.

CONCLUSIONS AND MANAGEMENT RECOMMENDATIONS

Stream Discharge

The results of this project suggest that, for areas where the annual hydrograph is dominated by spring runoff, harvesting about 40% of the watershed causes a substantial increase in peak flows of small headwater streams in the vicinity. Although increases were not documented for the first year after harvest, there were clear increases for the second and third years after harvest for both B3 and B5 watersheds. The lack of an increase in 1997 was attributed to the mulching effect created by the thick insulating layer of logging slash deposited on the snowpack during the winter harvesting operations.

The peakflow increases were greater in B5 watershed than in B3 watershed. This was attributed to the differences in slopes of the two watersheds, where B5 is exposed to more direct solar radiation during the spring snowmelt period. This causes the snow accumulated in the cutblock to melt faster and consequently increases peak discharges.

Small headwater channels can be important habitat for certain species of fish and can often be a source of domestic water for homes, farms and small communities. The protection of these aquatic resources may depend on maintaining peak flows within their natural ranges. When planning harvesting activities within headwater basins, it is important to identify the downstream resources that may potentially be impacted from increased peak flows.

Impacts may range from accelerated stream channel erosion to creating unfavorable flow regimes for certain life stages of different fish species to the disturbance of water intake facilities. If the resource is sensitive to increases in peak flows, then the harvesting plan may need to be designed to minimize impacts to an acceptable level. Possible mitigative measures to minimize increases in peak flows may include:

1) Smaller percent harvest within the watershed.

2) Reduction in the number of stream crossings. These are usually sources of accelerated water input into the stream channel from road ditches and road running surfaces.

3) Extra precautions and more conservative practices in the riparian zone. This will ensure that the stream banks maintain maximum stability and resistance to increased peak flows.

Suspended Sediments

In B3 watershed, the effect of the forest harvesting on increased concentrations of suspended sediments was small (21%) and short lived, being detectable only the first year after harvest. The riparian treatment along this stream was more conservative than in B5 and full retention was maintained in many areas for a distance of up to 30 m along each side of the stream. B3 had only one stream crossing above the flume which did not accumulate any substantial amount of ditch runoff (compared to three crossings for B5). This conservative type of riparian treatment appears to have minimized the increases in suspended sediments to the stream channel. However, in 1999 there was a large amount of blowdown that occurred in the riparian reserve left along the B3 stream channel. This severe blowdown may cause increases in sediment during the 2000 spring runoff. Conclusions about the effects of this blowdown cannot be made until the analysis of the 2000 data and the documentation to changes in channel morphology are complete.

In B5 watershed, the effect of the forest harvesting on increased concentrations of suspended sediments has been relatively large and has been clearly detectable for all three years after the treatments. The observed increases were attributed, almost exclusively, to the lower stream crossing and the roads and trails that lead to it. In 1997, the landing and the road above the stream crossing generated a lot of sediment that was transported directly into the stream channel. In the summer of 1997 the road was de-activated, but not adequately stabilized. During the spring of 1998 this crossing was identified as a major contributor to the increased sediment concentrations (Fig. 9). In 1999 it appeared that the logging related sediment sources were slowly stabilizing and thus generating less sediment to the stream channel.

Increases in the delivery of fine sediments to small headwater streams may cause negative impacts to fish and their habitats and to industrial, recreational and domestic water users. Possible mitigative measures to minimize increases in the delivery of sediments to stream channels could include:

1) Minimize stream channel crossings.

2) Control drainage and erosion in the cutblock so that sediment is not transported to the stream channel.

3) Ensure that de-activated stream crossings are properly stabilized and that appropriate erosion and sediment control practices are implemented.

4) Minimize (or preferably eliminate) soil disturbances near the edges of the stream channel (e.g. avoid mounding treatments).

5) In wetter ecosystems minimize harvesting of trees within 10 m of the stream channel. The harvest of most or all of the trees within the riparian management area will result in increased soil moisture and greater susceptibility to blowdown. If possible, harvest the single trees that are located immediately adjacent to the stream channel. This will avoid accelerated bank disturbance if there is a blowdown event.

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Figure 1: Riparian treatments at the Baptiste study site. The buffer strip in the foreground is the conservative treatment (B3 watershed), the buffer strip in the far cutblock is the aggressive treatment (B5 watershed) and the unlogged section between the two cutblocks (upper side of road) is the control watershed (B4 watershed). The photo looks west.

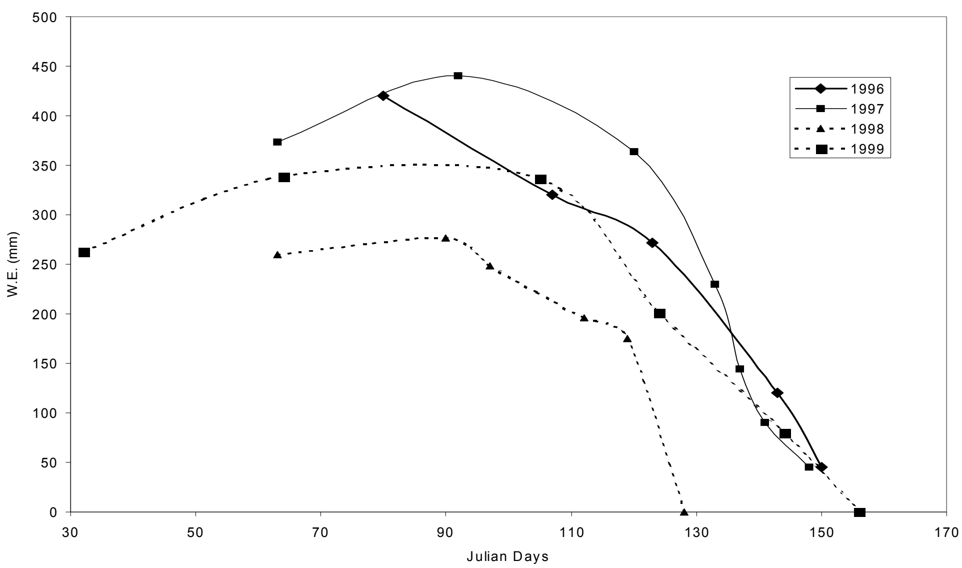

Figure 2: Changes in water equivalency (w.e.) of the snowpack during the springs of 1996, 1997, 1998 and 1999.

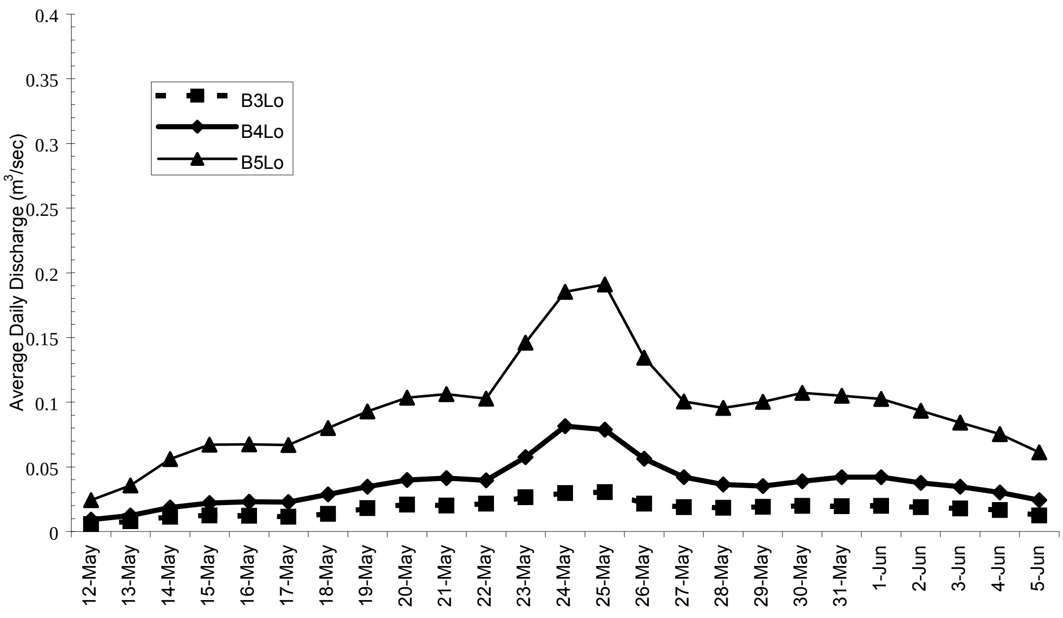

Fig. 3. Average daily stream discharge during the 1996 snowmelt period (pre-harvest period).

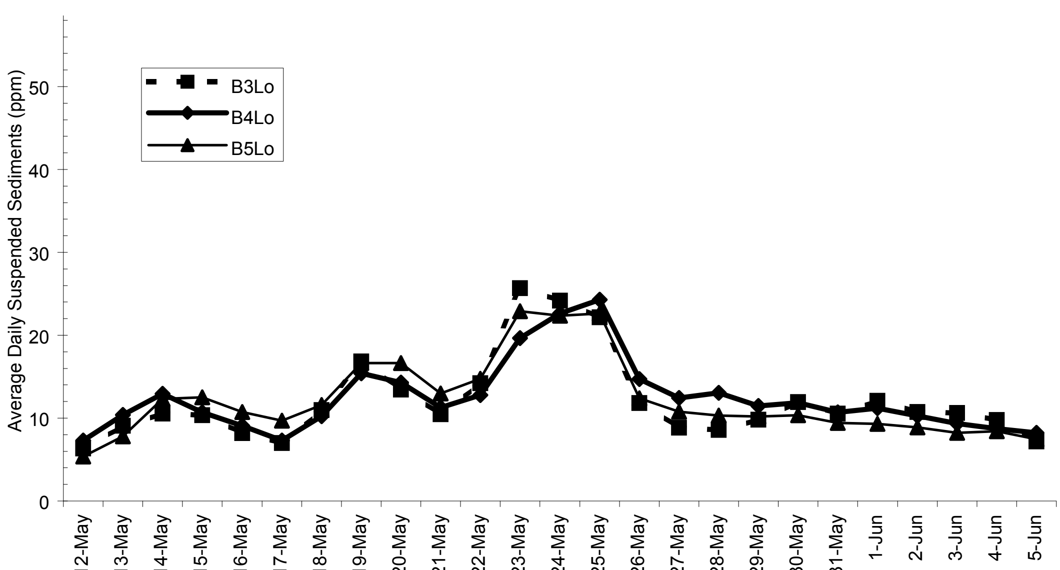

Fig. 4. Average daily concentration of suspended sediments during the 1996 snowmelt period (pre-harvest period).

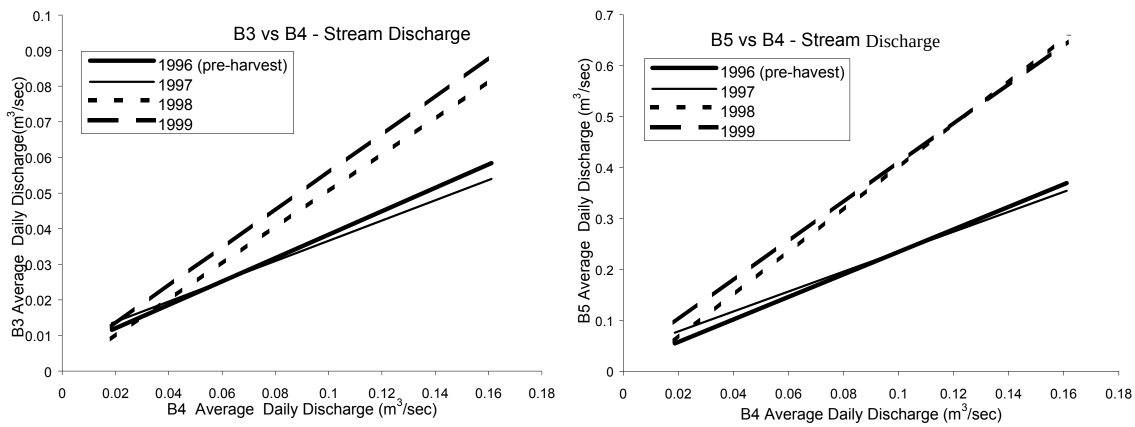

Fig. 5. Comparison of pre and post harvest regression lines for B3 vs. B4 and B5 vs. B4 - stream discharge.

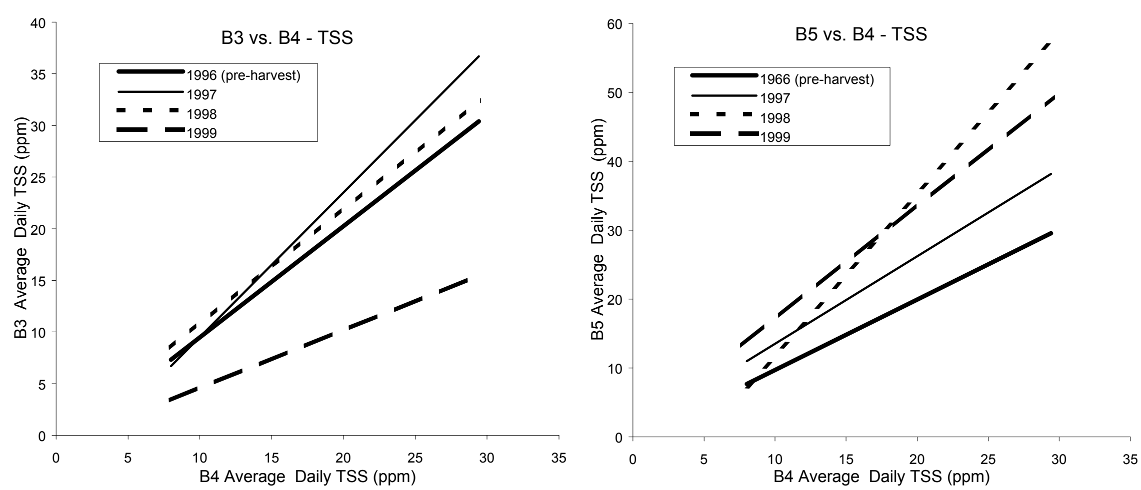

Fig. 6. Comparison of pre and post harvest regression lines for B3 vs. B4 and B5 vs. B4 - total suspended sediment.

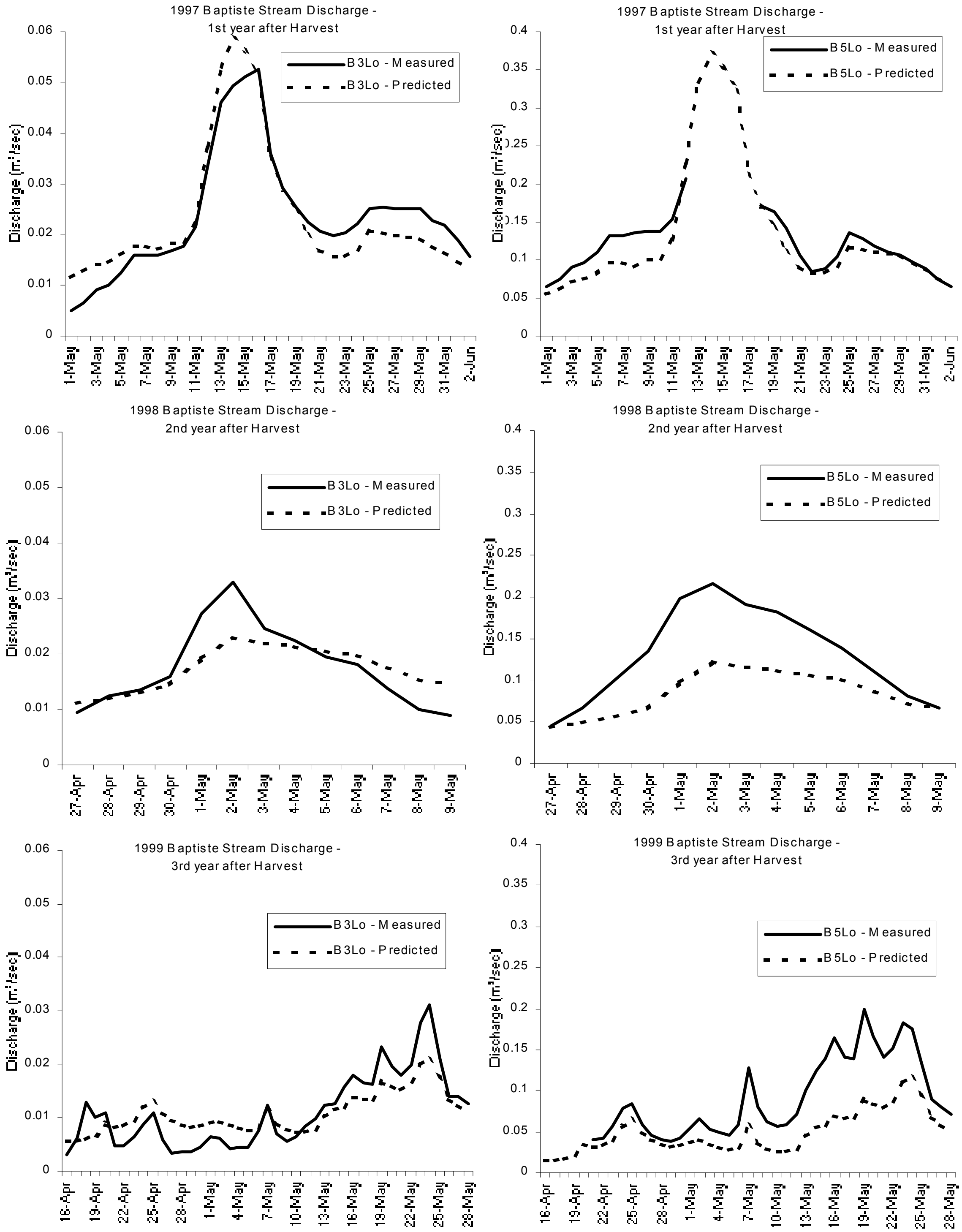

Fig. 7. Measured and predicted daily average snowmelt discharge for the two treatment watersheds for each of the three years post-harvest (i.e. 1997, 1998, 1999).

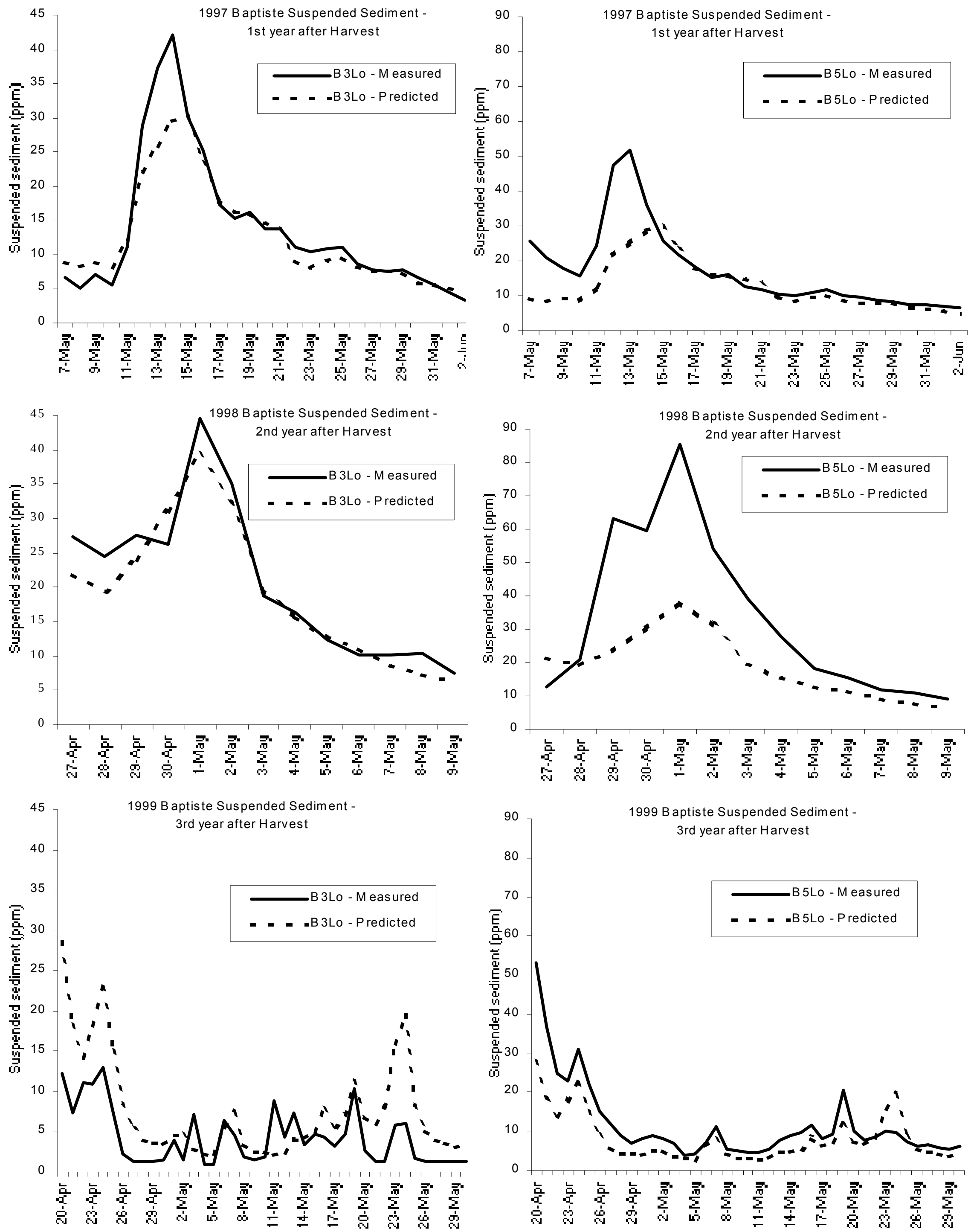

Fig. 8. Measured and predicted daily average concentrations of suspended sediments for the two treatment watersheds for each of the three years post-harvest (i.e. 1997, 1998, 1999).

Figure 9: View of the steep, unstable cutbanks left after the removal of the culvert. This site is located approximately 200m upstream of the B5 monitoring site.

Temperatures in Aquatic Habitats: The Impacts of Forest Harvesting in the Interior of B.C.

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INTRODUCTION

From previous studies that were conducted in coastal and interior settings, we can predict that summer stream temperatures will rise in association with streamside clearcut forest harvesting (Brown and Krygier 1970; Brownlee et al. 1988). Stream surface exposure to sunlight is cited as the most important mechanism and maintenance of shade the most obvious preventative measure. However, other forest harvesting impacts such as changes to groundwater regimes and alteration of winter stream temperatures have been documented (Ringler and Hall 1975; Hartman and Scrivener 1990) but are poorly understood, particularly in northern interior watersheds. Groundwater influences may act to maintain both summer and winter flows and offset increased energy input at stream surfaces in areas where riparian canopy has been lost.

The protective influence of riparian management zones as prescribed in the Forest Practices Code of British Columbia (FPC of BC), are as yet unproven, particularly in small tributary systems where frequently only non-commercial streamside vegetation remains after forest harvesting (S4, S6 streams; Anonymous 1995). Also of concern is the effectiveness of riparian strategies that incorporate variable retention harvesting at maintaining natural stream temperature regimes. Retention of 50 - 75% of the natural pre-harvest shade has not always allowed temperature objectives to be met (Rashin and Graber 1992). Streamside vegetation is prone to windthrow, particularly when left in narrow strips perpendicular to the prevailing wind and/or in narrow valleys that are geographically situated in the same direction as the prevailing winds (Moore 1977; Stathers et al. 1994). Windthrow along the edge of streams may negate the benefits sought through riparian management, by adding additional sediment and woody debris to the stream (Salo and Cederholm 1981; Mitchell 1998).

There are biological consequences to elevated stream temperatures at all stages of the salmonid life cycle, and small interior streams in BC are used by many life stages throughout the year including during spawning, incubation and rearing (Macdonald et al. 1992; Macdonald et al. 1998; Hickey et al. 1997). Temperature impacts to headwater streams that are not inhabited by fish may result in alteration of habitat through downstream mechanisms. As poikliotherms, fish are highly sensitive to temperature (Neill et al. 1972) and can detect changes between 0.05 and 0.1^{o}C (Murray 1971). While temperatures of 24^{o}C can be lethal to salmonids due to acute thermal shock (Servizi and Jensen 1977), chronic exposures to temperatures > 15 ^{o}C may also result in stress and premature death (Williams 1973). High temperatures can increase the susceptibility of salmonids to disease (Groberg et al. 1978) and decrease the interval between infection and death (Sanders et al. 1978). Elevated temperatures within a fish's tolerance levels may influence growth and life history events in a manner that has effects at the population level. Life history models of coastal and interior salmonid populations demonstrated increased growth, earlier migration and increased population size after forestry related changes to winter and spring stream temperature (Holtby 1988; Thedinga et al. 1989; Macdonald et al. 1998), but demonstrated no response to substantial and prolonged increases in summer temperatures.

This paper provides a summary of preliminary investigations of groundwater and stream temperature regimes in harvested and unharvested watersheds in the northern interior of BC, with reference to the influence of a variety of riparian prescriptions. Of particular importance is the documentation of impacts to stream temperatures during cooler months of the year when the loss of streamside vegetation allows increased radiation loss to the atmosphere. Natural stream temperature variability is also described as a preliminary step towards identifying and quantifying the impacts of forest harvesting.

METHODS

This study was undertaken as a component of the Stuart-Takla Fish-Forestry Project that examines ecosystem processes and forestry impacts in five watersheds at the northern extent of the Fraser River drainage, BC. The overall goals of the project have been described by Macdonald et al. (1992), Macdonald (1994) and Macdonald and Herunter (1998). Stream temperature and groundwater measurements have occurred in all of the Takla experimental watersheds, in some cases since 1990. As many of these watersheds are largely unharvested, these measurements provide a description of baseline inter-annual temperature variability. The early run of Stuart River sockeye salmon (*Oncorhynchus nerka*) spawn in the Takla tributaries during early August when annual stream temperatures are at their warmest and water levels are at their lowest summer levels (Macdonald et al. 1992). They complete a 1150 km migration up the Fraser River system to reach their spawning grounds (Burgner 1991).

This report provides a description of the annual thermographs of several lower order tributaries in the Baptiste watershed where data collection began in 1995 and harvesting occurred in the winter of 1996 following a variety of riparian treatment prescriptions (streams, treatments and design are fully described in other papers in this compendium). For comparative purposes a thermograph from Kynoch Creek is also provided to represent a higher order system typical of natal sockeye streams in this geographical area. Stream temperatures were recorded on an hourly basis using Vemco ™ temperature dataloggers at sites above and below cutblocks on B3 and B5 and on B4, the unharvested control. Temperature dataloggers were also placed in 2 m metal wells in several locations in each watershed to record annual groundwater temperatures. The temperature loggers have a precision of $\pm 0.2^{o}$C. Daily mean, maximum and minimum temperatures were computed from the hourly values. A weather station located near the mouth of Middle River within the Takla experimental area provided daily temperature information.

RESULTS

The annual pattern of stream temperatures in the Takla research watersheds are typical of records collected from other northern interior watersheds including the Nechako River (Blachut and Faulkner 1988) and tributaries to Slim Creek (Brownlee et al. 1988, Choromanski et al. 1993). Since 1995, summer daily average water temperatures in a sockeye spawning stream within the Takla area (Kynoch Creek) were between 8.0 and 14.3^{o} C, peaking in late July or early August (Fig. 1). Diurnal ranges were 1.0-4.2^{o} C (Fig. 2). Before riparian harvesting, tributaries to these systems were several degrees cooler in the summer than the higher order systems into which they flow (maximum daily average of 11.5^{o}C, B3, B4, B5, Figure 1), and they had lower diurnal temperature ranges (< 2.5^{o}C, Fig. 2). Annual maximum air temperature and water temperatures throughout the watersheds occur in late July or early August in coincidence with peak sockeye salmon spawning (Macdonald et al. 2000). They remain warm through August and a portion of September, but decline rapidly to below 1^{o}C by late October. Stream temperatures usually remain below 1^{o}C from the third week of October to the second week of April and show little diurnal variability (<0.5^{o}C). During the period of spring snow-melt water temperatures rise to approximately 5^{o}C by late May, when most of the sockeye salmon fry have emigrated from the streams (Macdonald et al. 1992). Groundwater temperatures within the tributary riparian zones reach summer peaks a few weeks later and are generally cooler in the summer and warmer in the winter than the adjacent stream (Fig. 3).

Harvesting impacts to Baptiste tributary temperatures were clearly demonstrated during the summer months, while mid-winter stream temperatures were not influenced by harvesting. In the summers following harvesting, stream temperatures rose up to 3.1^{o}C and showed greater diurnal range relative to the control, B4 (Fig. 4). The magnitude of the impact was greatest during the warmest period in the summer (July/early August) and was dependent on riparian treatment. Initially, variable retention methods (B3) moderated temperature impact to less than half that observed when all merchantable trees were removed from the riparian strip (B5). However, in the years following the harvest relative summer temperatures rose, likely as a result of riparian windthrow within the B3 watershed. Trees were also lost to wind-storms along portions of the forestry access roads. While not yet quantified, portions of B3 that were initially shaded by the riparian retention may have been more exposed to direct solar radiation by the summer of 1998.

Changes to groundwater temperature regimes were variable with relative increases of 1.5^{o}C in B3 during the first year following forest harvesting, and apparent decreases in subsequent years (Fig. 4). While relative groundwater temperatures in B5 remained constant the first year after harvesting, they rose up to 1.5^{o}C in the following years. Cumulative summer daily average air temperatures decreased slightly over the years following forest harvesting (Table 1).

Table 1. Cumulative summer daily average air temperatures (^{o}C) (June 2 to September 8) from the DFO Middle River Camp weather station.

Year	1995	1996	1997	1998	1999
Cumulative Air Temperature	1288	Harvesting**>** 1161	1104	1005	1007

DISCUSSION

A plethora of information from many geographical areas emphasizes the value of riparian vegetation for the moderation of summer stream temperatures (reviewed in Anderson 1973 and Beschta et al. 1987). Stream exposure to the direct effects of solar radiation is the dominant mechanism involved in the increased temperatures, while air temperature and energy loss from the stream due to evaporation play minor roles (Brown 1970 and Brown and Krygier 1970). Geomorphic features also bear consideration; as streams become deeper (Adams and Sullivan 1989) and wider (Chen et al. 1998), riparian vegetation has less influence on moderating temperatures. Small first order streams are frequently narrow enough to be completely shaded by riparian vegetation and are generally cooler in the summer than the second and third order streams into which they flow. Elevation and gradient may also contribute to this relationship, as do groundwater regimes. Direct groundwater input of cool water in the summer and warm water in the winter (relative to surface water temperatures) is the primary (and immediate) source of flow for many first order streams. The moderating influence of groundwater has been recognized by several authors in both coastal and interior settings (e.g. Hartman and Leahy 1983; Bilby 1984; Brownlee et al. 1988; Macdonald 1994). These studies measured water temperatures beneath the streambed and didn't differentiate between the influence of upwelling groundwater originating from upslope areas, and hyporheic exchange within the streambed. Shepherd et al. (1986) and Moore et al. (this compendium) have recognized the need to make this differentiation in an effort to improve management strategies.

Following harvesting, summer temperatures and diurnal temperature variation in first order streams will rise, but may not exceed similar measurements from larger, unharvested systems downstream and may remain within the tolerance limits of local biota. This suggests that some first order stream temperatures may be less susceptible to riparian harvesting than larger systems depending on the proximity to groundwater inputs. It also serves to emphasize the importance of maintaining natural groundwater temperatures and hydrologic patterns when preparing harvesting plans (Hetherington 1987). If post-harvesting stream temperature and diurnal variation never exceed the tolerance limits of the resident aquatic biota, the maintenance of streamside shade may not be a necessary riparian prescription for thermal management. In high latitude and/or high elevation locations fish may actually benefit from warmer water temperatures that result from riparian removal (e.g. S.E. Alaska - Thedinga et al. 1989; Vancouver Island - Holtby 1988). However, modest changes in thermal regimes may alter migration timing or cohort structure of fish populations, with unknown consequences (Macdonald et al.

1998). In some situations benefits from warmer temperatures may be tempered by reductions in large organic debris, destabilization of stream-banks, or increased levels of ultraviolet radiation associated with riparian loss (Clare and Bothwell, this compendium). If our management strategy depends on groundwater to moderate forestry-induced temperature impacts, we are assuming that groundwater dynamics are not altered as a result of forest harvesting. Hewlett and Fortson (1982) suggest that shallow groundwater may warm after clearcutting and Hartman and Scrivener (1990) demonstrated a rise in groundwater levels after logging in Carnation Creek on Vancouver Island. While analysis is not yet complete, preliminary results suggest that harvesting may have had a modest impact on groundwater temperatures in the Baptiste watershed (Fig. 4). Road construction and the interruption of sub-surface flow patterns, may also contribute to warmer stream-flow (Herunter et al. - this compendium).

Buffer strips (riparian reserves) have long been considered an effective means to regulate summer stream temperatures in forested watersheds (Swift and Baker 1973; Beschta et al. 1987), but buffer width by itself may not be a good predictor of degree of temperature protection. The amount of riparian timber retained and ultimately the amount of canopy that shades the stream from the influence of direct-beam solar radiation are the factors pertinent to reserve zone effectiveness. Retention of 50% to 75% of pre-harvest shade was not entirely effective at meeting temperature objectives in Washington State (Rashin and Graber 1992), and partial retention was effective for just one year before wind damaged the buffer strip in the Baptiste watershed. Intact and stable buffers of widths of 30 m or more may be sufficient in coastal regions of the Pacific Northwest (Beschta et al. 1987). Riparian width requirements vary among jurisdictions and may be prescribed for purposes beyond the management of aquatic resources (reviewed by Young 2000). Ultimately riparian buffers most effective at maintaining natural summer stream temperatures will be those that provide the greatest canopy densities on the side of the stream that faces towards the summer solar radiation (measured as angular canopy density - Steinblums et al. 1984; Teti pers. comm.).

Forest harvesting impacts to stream temperatures during winter are poorly understood particularly in northern interior locations. Meehan et al. (1969) reports little change to winter stream temperatures due to harvesting in a coastal Alaskan stream. In Carnation Creek, a coastal stream in southern BC, winter stream temperatures were generally warmer (< 1.0^{o}C) after harvesting; which is thought to have positive consequences to the development and subsequent rearing of juvenile coho salmon (Hartman and Scrivener 1990). At Carnation Creek, winter temperatures are less likely to be influenced directly by solar radiation (due to shorter days and reduced solar angles) and more likely influenced by changes to groundwater temperatures and levels (Shepherd et al. 1986; Hartman and Scrivener 1990). In the Baptiste watershed there was no clear effect of riparian harvesting on winter stream temperatures, but an effect was noted by Macdonald et al. (1998) in slightly larger watersheds in the same biogeoclimatic zone (Gates Creek). In northern interior environments, cold winter air temperatures, in conjunction with the loss of the insulating qualities associated with riparian vegetation, may promote radiant and/or conductive loss of energy from the stream to the atmosphere at a greater rate in systems where riparian canopy has been lost regardless of the direction of solar radiation (Anderson 1973; Beschta et al. 1987; Macdonald et al. 1998). Therefore, overall canopy density rather than angular canopy density may provide a better annual predictor of stream temperature protection in northern interior systems particularly if snow accumulations are insufficient to cover and insulate streams from temperature loss. In the interior this seasonal reversal in the effect of forestry activities may have consequences unique to northern aquatic biota such as reduced production periods, reduced water levels and habitat area, and earlier ice cover and possible increased anchor ice formation (Macdonald et al. 1998).

MANAGEMENT IMPLICATIONS

Riparian prescriptions for first order streams (S4 and S6) in B.C. are less strict than for larger second and third order systems (Anon 1995) requiring less, and frequently no commercial timber retention. In some situations this management strategy may have scientific merit as it applies to temperature protection. Summer and winter temperatures in many small streams in the Takla area are moderated by the influence of groundwater. Following streamside harvesting, summer temperatures and diurnal temperature variation in first order streams will rise, but may not exceed similar measurements from larger, unharvested systems downstream and may remain within the tolerance limits of local biota. Modest increases in stream temperature can be managed for and may even have minor biological benefits. However, caution is required as these benefits may be overshadowed by less beneficial impacts in the autumn and winter. Stream temperature reductions upon re-entry to forested areas have also been documented and are generally attributed to the inflow and mixing of cool tributary streams and groundwater (Hall and Lantz 1969; Swift and Baker 1973; McGurk 1989; Macdonald et al. 1998; Moore et al. - this compendium). This suggests that forested temperature recovery zones should be included in forestry planning processes that incorporate riparian harvesting. Wind-firm riparian buffer strips may remain the most cautious approach available to resource managers, but other management approaches deserve consideration.

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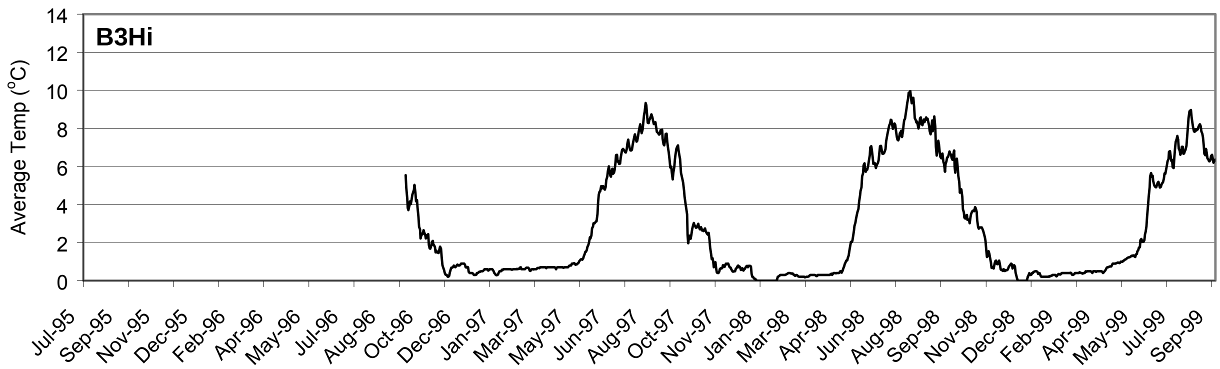
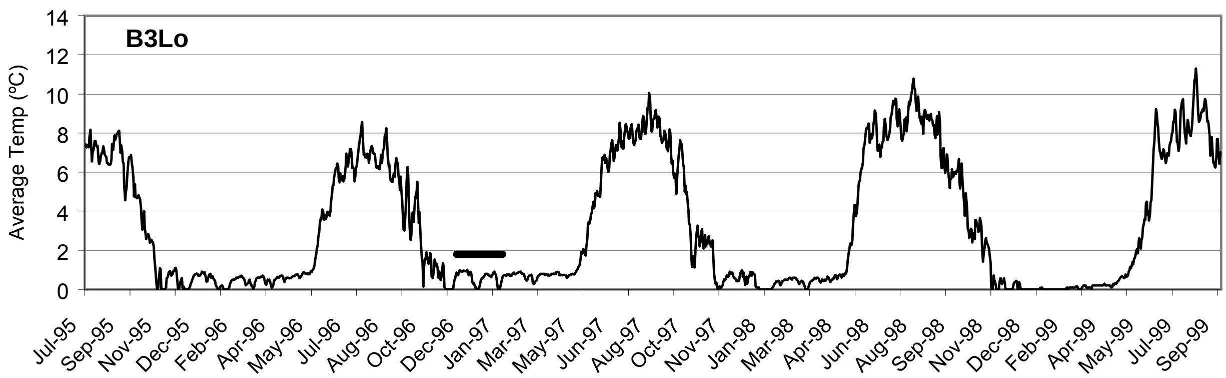




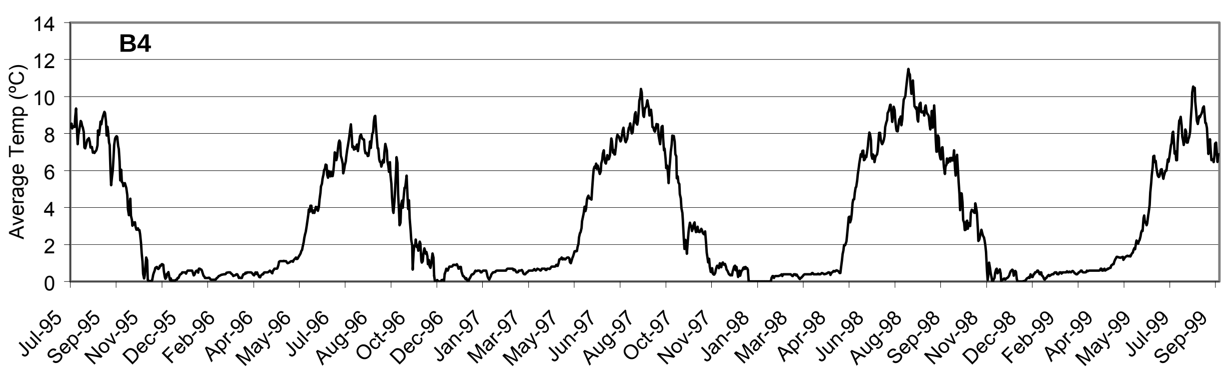
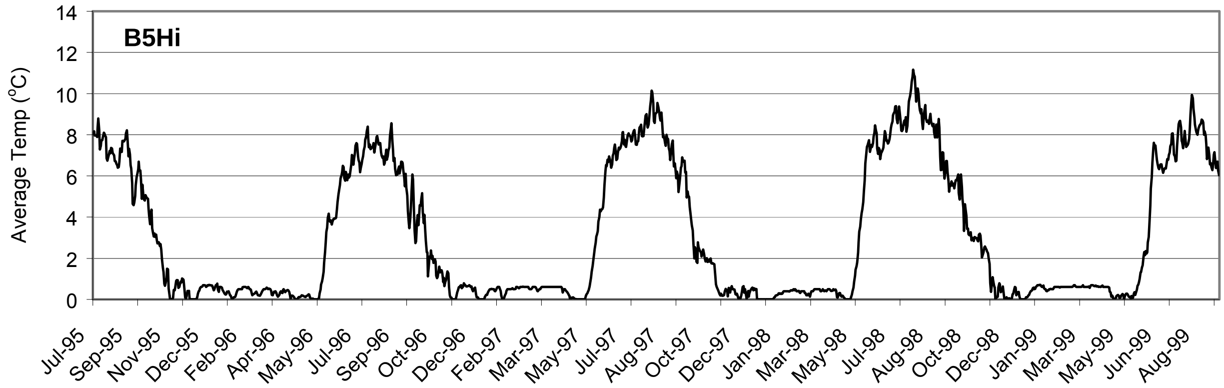
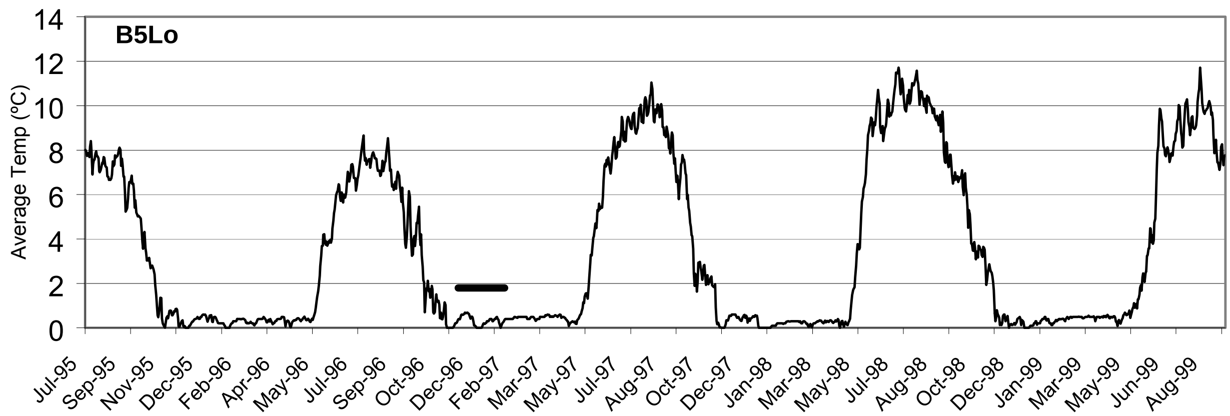


Fig. 1. Daily average thermographs of the Baptiste study creeks and one salmon spawning creek in the Takla area (Kynoch Creek). B3Lo and B5Lo were located at the downstream edge of cut blocks, B3Hi, B5Hi, and B4 represent spatial controls. Solid horizontal lines on B3Lo and B5Lo show when forest harvesting occurred.





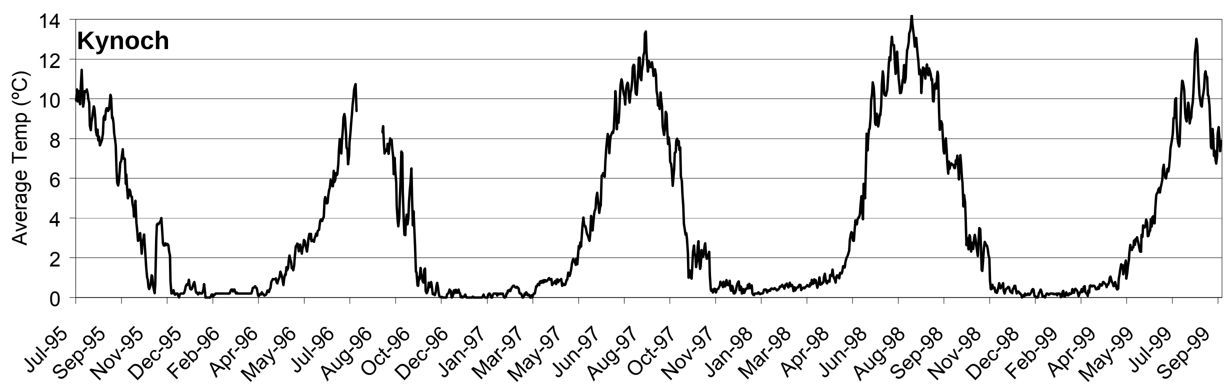


Fig. 1. Cont'd. Daily average thermographs of the Baptiste study creeks and one salmon spawning creek in the Takla area (Kynoch Creek). B3Lo and B5Lo were located at the downstream edge of cut blocks, B3Hi, B5Hi, and B4 represent spatial controls. Solid horizontal lines on B3Lo and B5Lo show when forest harvesting occurred.

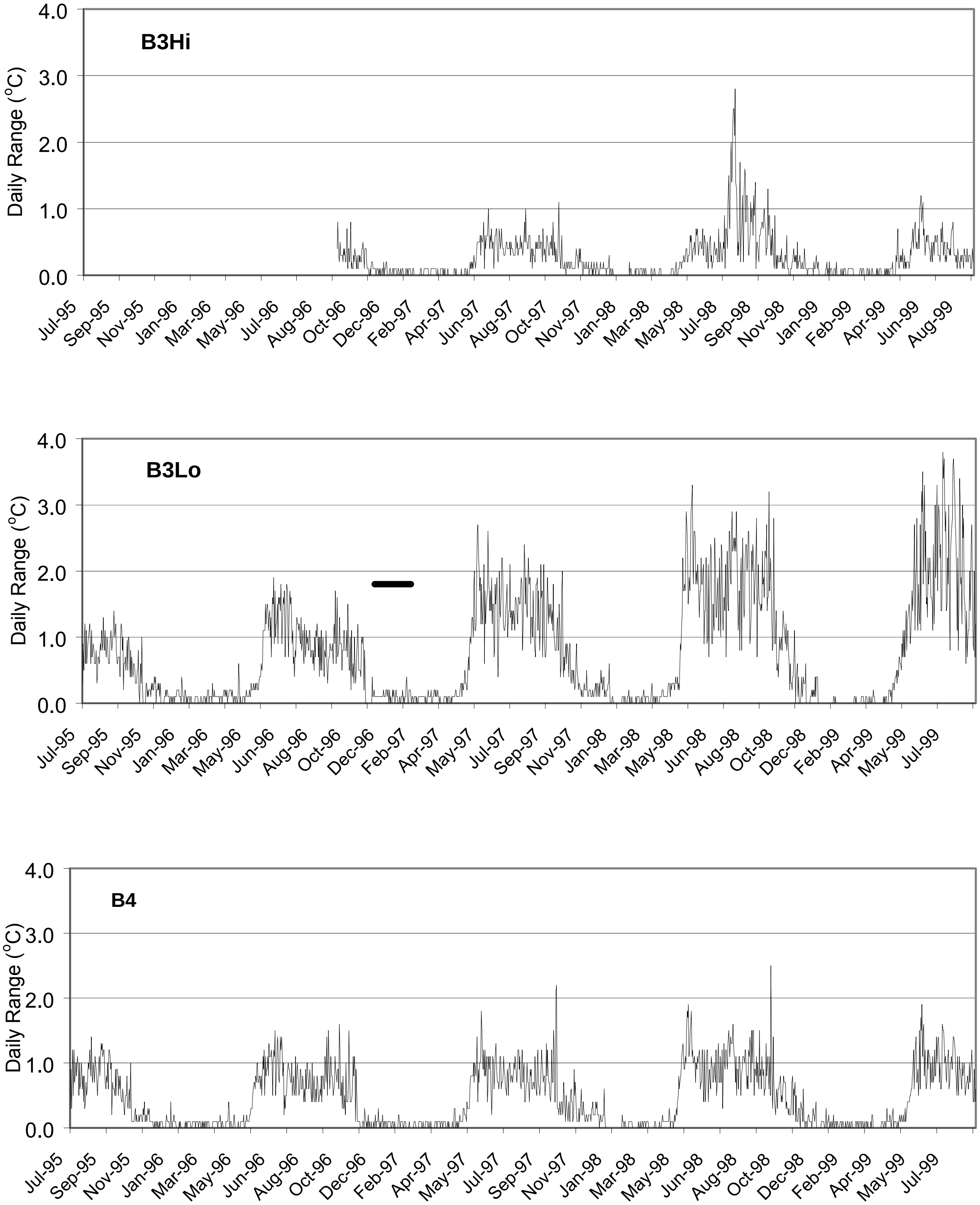


Fig. 2. Daily range of the Baptiste study creeks and one salmon spawning creek in the Takla area (Kynoch Creek). B3Lo and B5Lo were located at the downstream edge of cut blocks, B3Hi, B5Hi, and B4 represent spatial controls. Solid horizontal lines on B3Lo and B5Lo show when forest harvesting occurrred.

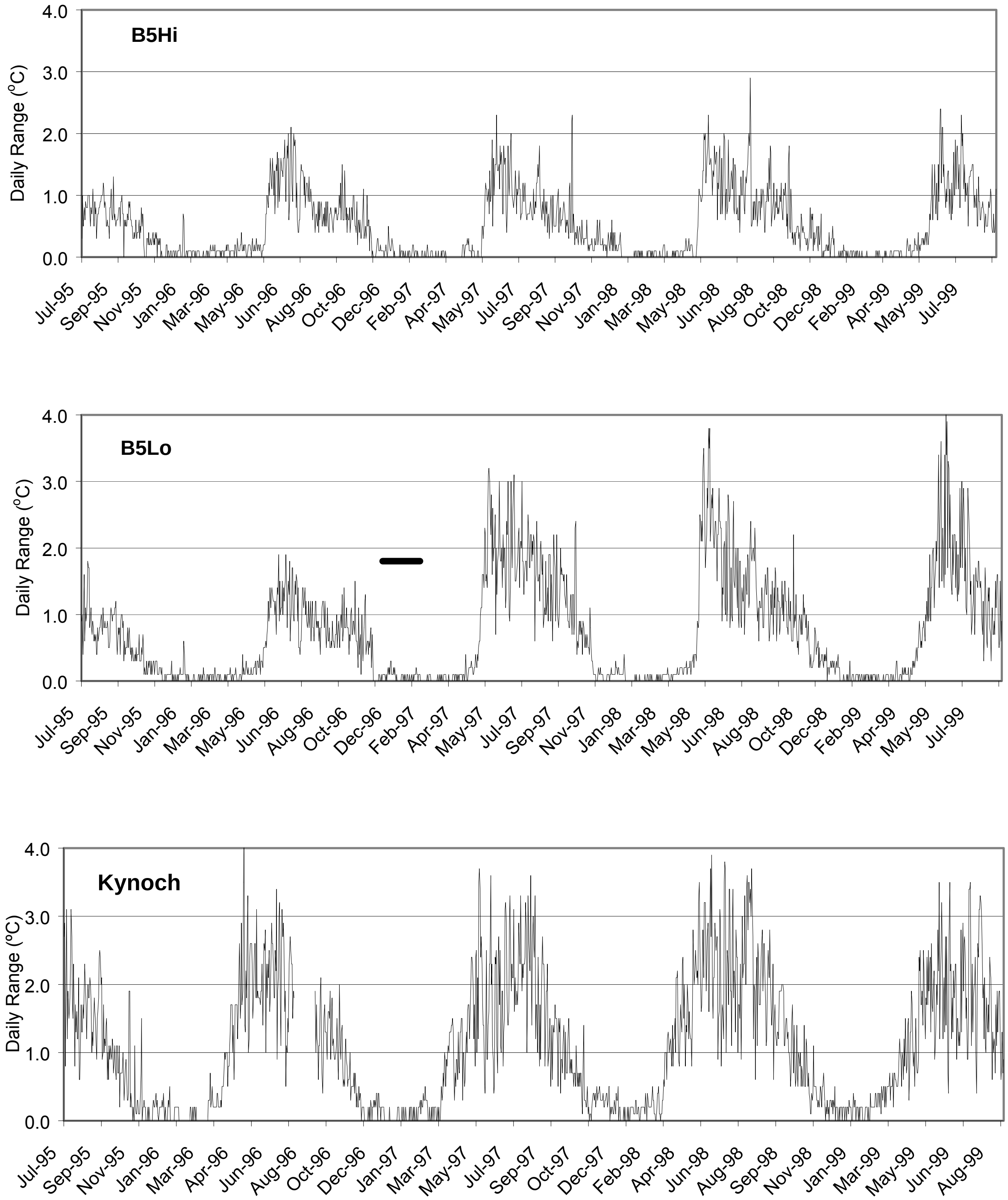


Fig. 2. Cont'd. Daily range of the Baptiste study creeks and one salmon spawning creek in the Takla area (Kynoch Creek). B3Lo and B5Lo were located at the downstream edge of cut blocks, B3Hi, B5Hi, and B4 represent spatial controls. Solid horizontal lines on B3Lo and B5Lo show when forest harvesting occurred.

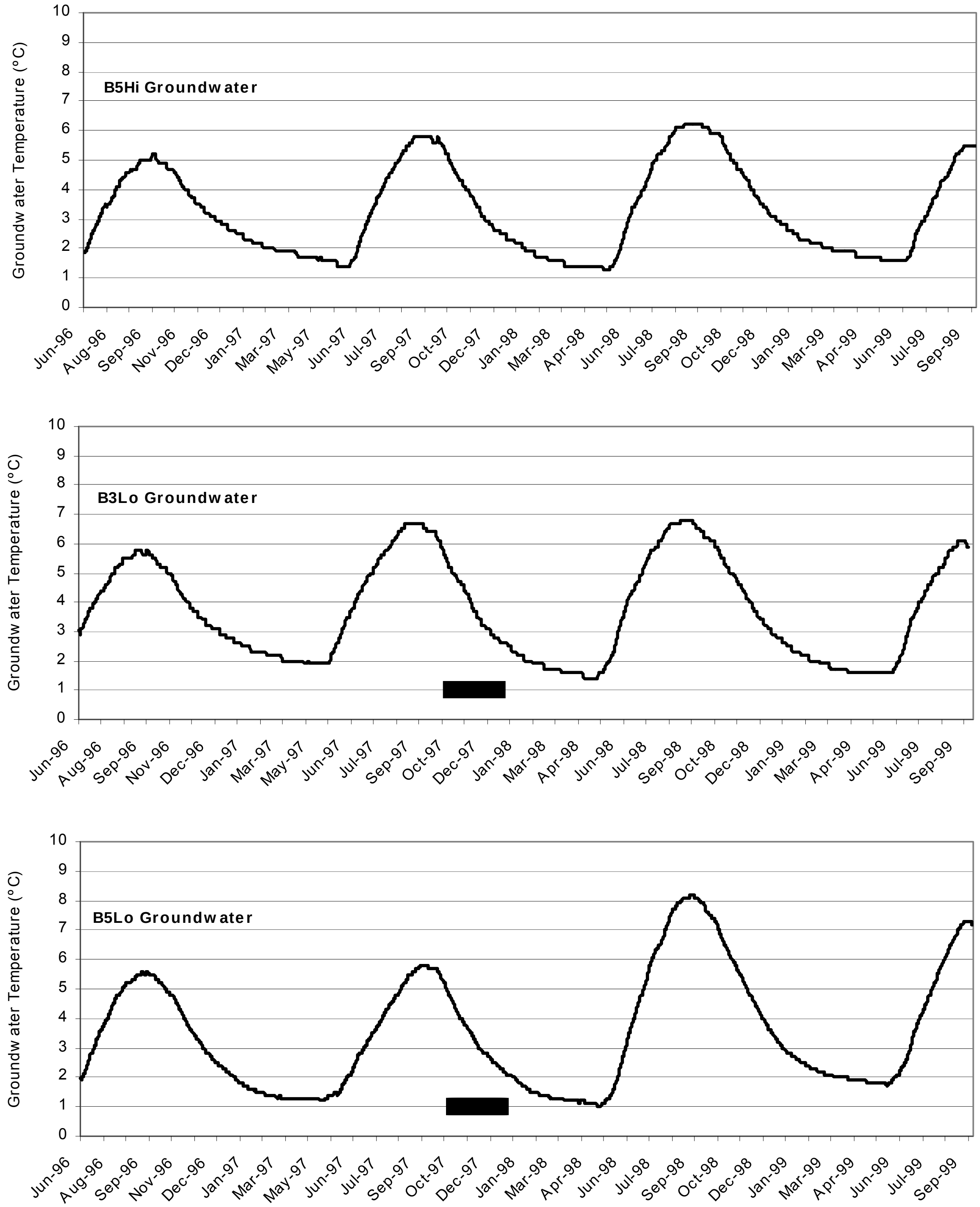


Fig. 3. Daily average riparian groundwater thermographs of the Baptiste study creeks. B3Lo and B5Lo were located at the downstream edge of cut blocks, B5Hi represents the spatial control. Solid horizontal lines on B3Lo and B5Lo show when forest harvesting occurred.

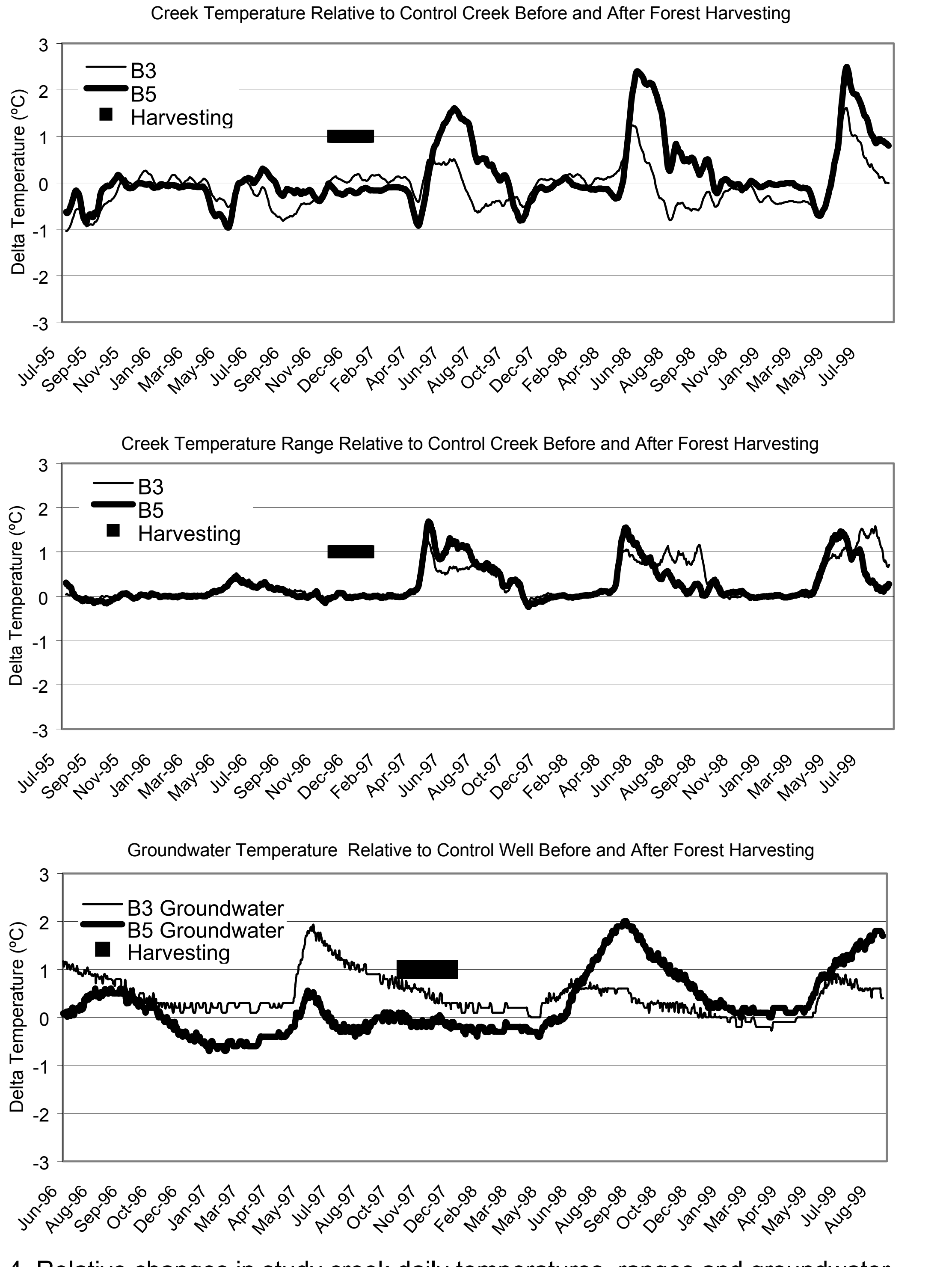


Fig. 4. Relative changes in study creek daily temperatures, ranges and groundwater temperatures before and after forest harvesting. The difference between the experimental creeks and spatial controls (B4-creek temperature; B5Hi – groundwater temperature) is plotted. Fifteen day running averages were used to smooth the data for graphing clarity. Solid horizontal lines show when forest harvesting occurred.

Downstream Thermal Recovery of Headwater Streams below Cutblocks and Logging Roads

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INTRODUCTION

Removal of the vegetation surrounding a stream alters the energy exchanges across the stream surface. During summer days, the main effect is to increase inputs of solar radiation, which normally results in a warming of the streamwater as compared to the undisturbed situation. This warming effect is widely considered to be a critical stressor to aquatic organisms, particularly to cold water species such as salmonids (bull trout and salmon) (e.g., Beschta et al. 1987). The warming of streams as they flow through clearcuts has been documented in terms of both the magnitudes of warming and the energy balance (Brown and Krygier 1970). Quantitative models have been developed to predict the warming effect of vegetation removal and to aid in the design of riparian buffers (Chen et al. 1998). Land-use management prescriptions depend heavily on riparian buffers and management zones to achieve aquatic protection goals (e.g., B.C. Forest Service and B.C. Environment 1995).

What happens to a heated stream after it flows back under a forest canopy has been relatively unexamined. Brown (1985) argued on the basis of energy balance considerations and limited empirical data that a stream would likely continue to warm during daylight hours, albeit at a lower rate than in the cutblock. There are, however, some data that indicate that cooling can occur (e.g., Macdonald et al. 1996; McGurk 1989). The most likely mechanisms for such downstream cooling would be interactions between the stream water and cooler subsurface water in the riparian zone, and/or heat conduction into the bed.

As part of the Stuart-Takla Fish-Forestry Interaction Project, the Department of Fisheries and Oceans conducted a study of thermal effects of forest practices. The treatments included clearcut logging, with a range of riparian buffers, and road crossings. In addition, stream temperatures were recorded at locations downstream of the forest operations, after the streams had flowed out of the cutblock into undisturbed forest, in order to document the occurrence and magnitude of downstream thermal recovery. This note presents some initial analyses of the magnitude of downstream thermal recovery, as well as some pilot studies designed to explore the working hypothesis that the recovery was caused by inflow of cooler groundwater in the recovery reaches.

MATERIALS AND METHODS

Stream temperatures were recorded on an hourly basis using Vemco ™ temperature loggers at a number of sites in the Baptiste drainage. The temperature loggers have a precision of ±0.2°C. Daily mean, maximum and minimum temperatures were computed from the hourly values. This report focuses on a comparison of temperatures on streams B3, B4 and B5 at the lower edges of the road right-of-way (i.e., where the stream flows back under the forest canopy), and at locations in the forest, 170 m below the road crossing.

In addition to the continuous temperature monitoring, field visits were made July 8-9 and August 11-12, 1999, to conduct pilot studies on the effect of groundwater on stream cooling below the road cuts. To examine variations in discharge along the stream, salt (NaCl) solution was injected at a constant rate using a Mariotte bottle (Stream Solute Workshop 1990). Resulting increases in salt concentration were determined by measuring the change in electrical conductivity (EC) using a WTW™ conductivity probe and meter. The WTW instrument automatically corrects electrical conductivities to a standard temperature of 25°C using the DIN 19 266 nonlinear calibration.

When conducted in a short, well-mixed reach, measurements of discharge based on constant-rate salt injection can have a precision of ±5% or better (Johnstone 1988). An important influence on the precision of streamflow measurements using salt dilution is the increase in stream EC relative to background EC and the resolution of the conductivity meter. In the pilot studies, relatively dilute injection solutions (approximately 20 g/L) and low injection rates (approximately 0.5 mL/s) were used. Because the resulting increases in EC were only about 3 to 6% of stream background, the precision of the flow measurements was on the order of ±10%.

Constant-rate salt injection can be used to determine locations of net groundwater input by injecting until steady-state concentrations are achieved along a reach of stream. Locations of net groundwater input are indicated by decreases in streamwater EC.

In addition to the salt-dilution trials, measurements were made of stream temperature and background electrical conductivity using the WTW conductivity meter, as well as subsurface temperatures in the stream bed and banks at a depth of about 30 cm with a Kane-May KM330 digital thermometer (precision ± 0.2°C).

RESULTS

Stream temperature monitoring

In B3 during summer, stream reentry into the forest resulted in cooling up to 2°C in terms of mean daily temperature and over 4°C for maximum daily temperature (Fig. 1). The magnitude of differences in minimum temperatures was generally lower than for mean or maximum temperatures, and tended to be in the opposite direction. That is, during summer, minimum temperatures tended to be higher at the downstream location within the forest.

Downstream cooling at B5 was more marked than for B3, with differences in mean daily temperature ranging up to about -3.7°C, compared to -2°C for B3 (Fig. 2). More notably, downstream cooling was apparent in the daily minimum temperatures, with differences frequently exceeding -2°C.

The thermal behaviour at B4 differed from that at B3 and B5 (Fig. 3). The downstream location was usually warmer than the road crossing. Relatively brief periods of downstream cooling occurred in spring and autumn, but the magnitude was of the same order as the precision of the temperature loggers.

The influence of reentry into the forest on stream temperature varied substantially within each season at all streams, probably reflecting the short-term influence of weather changes. In addition, the timing and magnitude of downstream cooling at B3 and B5 varied amongst years. This interannual variability probably results from the combined effects of differences in winter snow accumulation and summer weather patterns. For example, a delay in stream warming in the 1999 spring-summer season was associated with a heavy snowpack that persisted into the summer, due to cool weather conditions through June.

Pilot studies of groundwater influences

On July 8, 1999, streamflow in B5 decreased from 7.2 L/s at the lower flume, located above the road cut, to 4.7 L/s below the road near the lower standpipe (point E in Fig. 4). This difference is greater than could be explained by measurement error. Although no discharge measurements were made below point E, it appeared visually that flow decreased around the bend between E and D, then increased again. This observation was confirmed by observations on August 11, when the channel was dry for a length of approximately 20 m between points E and D. The discharge at the B5 flume on that date was 3.1 L/s.

Stream temperature and electrical conductivity were measured along B5 on August 11, 1999, between about 12:00 and 1:00 p.m., when air temperature was about 10°C (Fig. 5). The marked drop in stream temperature between points E and D coincided with the loss and subsequent gain of flow in that section of stream. Subsurface temperatures in the bed and bank were slightly greater than water temperature at sites I to E. At site D, subsurface temperatures were about equal to or lower than stream temperature. At site C, subsurface temperature differed by 2.3°C from the left to right banks, over a distance of about 1 m.

Electrical conductivity dropped significantly between points E and D (Fig. 5). This observation suggests that the recovery of flow below the dry section of channel was not comprised of the same water that flowed further upstream, but was at least partly derived from native groundwater originating as local recharge, rather than from advected streamflow.

On August 12, 1999, the measured discharge below the B2/B3 confluence was 3.3 L/s, and the background electrical conductivity was 388 μS/cm along the entire reach. Electrical conductivity measurements during a salt-injection trial indicate a dilution between 25 and 35 m downstream of the confluence, coinciding with a slight drop in temperature (Fig. 6). This dilution and drop in temperature likely reflect a localized input of cooler near-channel groundwater.

DISCUSSION

The observed rates of downstream cooling in B3 and B5 greatly exceed rates previously reported below cutblocks. For example, Macdonald et al. (1996) reported partial temperature recovery 500 m below several cutblocks during summer on small streams in the interior of B.C. McGurk (1989) reported partial recovery 130 m below a clearing in California, and suggested that 1.6 km would be required for complete recovery to pre-harvest temperatures. Stream cooling in B3 and B5 occurred even under conditions when the stream would be expected to experience a net gain of radiant energy from the forest canopy, assuming canopy temperature ≈ air temperature (Black et al. 1991). This observation indicates that the cooling was most likely related to interactions between streamwater and cooler subsurface water in the riparian zone. The three streams varied in regard to the magnitude of downstream cooling. In fact, B4 exhibited a dominant trend to downstream warming rather than cooling. This tendency might reflect the fact that B4 was a control stream with no forest harvesting and had the lowest temperatures at the Below Road station. The only temperature impacts on B4 would be due to the main road and the low “winter” road, which at B4 was a minimal impact design with a right-of-way width of 20 m.

It is hypothesized that the thermal differences between B3 and B5 at least in part reflect differences in the nature and magnitude of interactions between stream water and groundwater in the recovery reaches. Further detailed field study is required to test this hypothesis.

Observations at B5 indicated that, contrary to our initial expectations, discharge did not monotonically increase downstream through accrual of groundwater. In fact, upstream of the lowest temperature logger, the stream experienced a loss of flow, followed by a flow increase. It therefore appears that the temperature recovery resulted from a gain of cooler subsurface water. This subsurface water was likely a mixture of native groundwater (i.e., resulting from local recharge) and advected streamwater from higher in the catchment that had infiltrated the bed and banks. That is, the stream cooling resulted from a mechanism more akin to hyporheic exchange between the stream and riparian zones, rather than a one-directional input of groundwater.

Observations during the salt injection trial at B2/B3 indicated that the location of stream cooling coincided with the location of net groundwater input. However, the stream had not been warmed dramatically through the road cut due to the generally cool, overcast weather that prevailed during the August field trip, resulting in a modest thermal contrast between the stream and groundwater.

These pilot studies have helped us refine hypotheses regarding the role of subsurface water in downstream cooling below land-use activities. Further studies are planned to examine in more detail the hydrologic and thermal processes associated with interactions between surface and subsurface water in the riparian zones of the Baptiste study streams.

Management Implications

This study indicates that headwater streams that have been heated by flowing through clearcuts and road crossings can recover after flowing back under a forest canopy. Taking these results at face value, it would appear that downstream thermal impacts of small streams on larger streams could be mitigated by leaving an adequate unharvested recovery zone. However, the cooling appears to be dominated by interactions between streamwater and near-stream groundwater, which are difficult to predict on the basis of readily available information, such as surface topography. Therefore, caution must be exercised in using these results for forest management without further verification of the results on other headwater stream systems.

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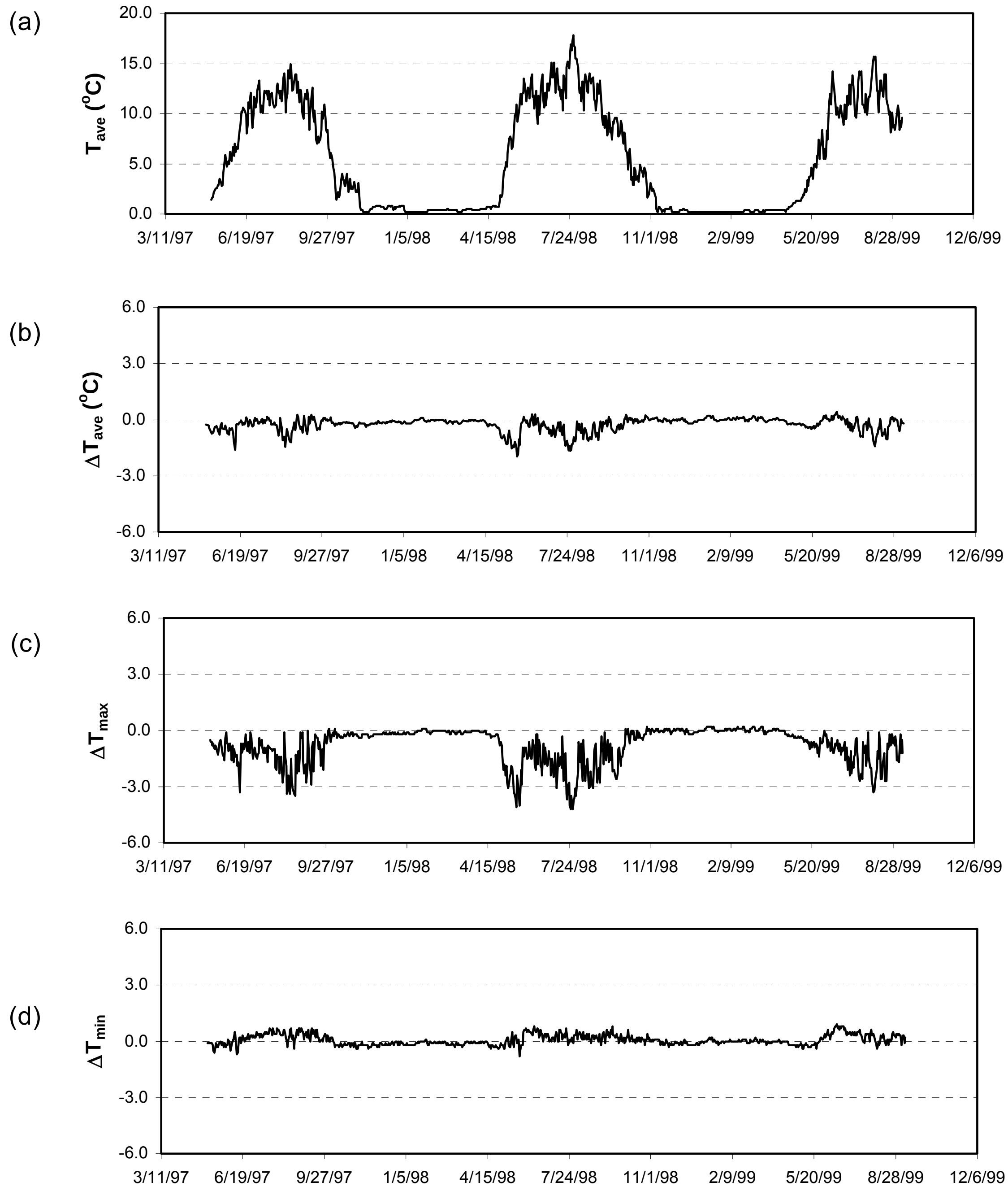

Fig 1. Stream temperature data for B3. (a) Mean daily stream temperature at lower edge of road crossing; (b) Difference in mean daily temperature between the forest and the road crossing; (c) Difference in maximum daily temperature between the forest and the road crossing; (d) Difference in minimum daily temperature between the forest and the road crossing. In plots (b), (c) and (d), a negative difference indicates downstream cooling.

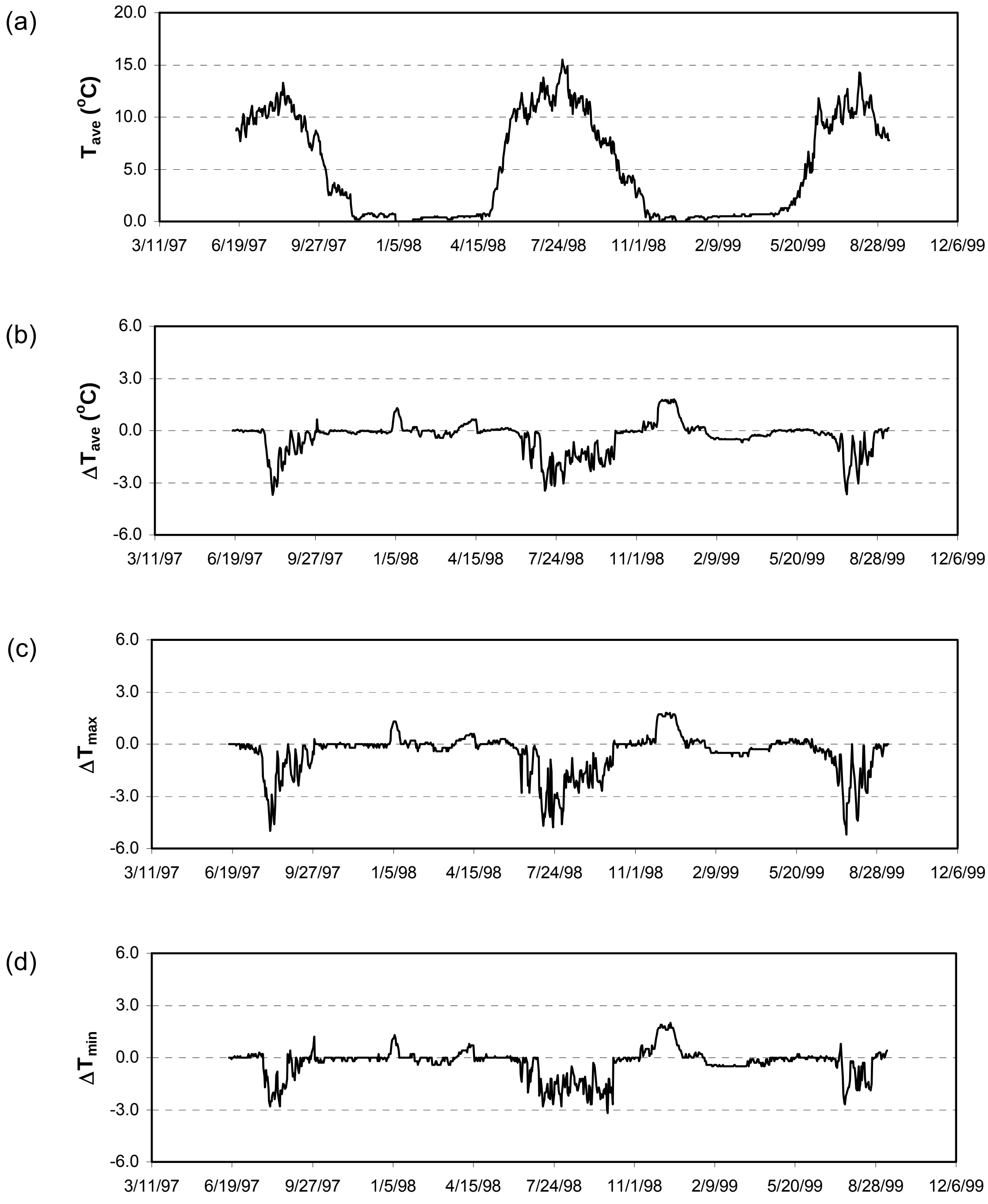

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Fig. 2. Stream temperature data for B5. See caption for Fig. 1 for detailed explanation.

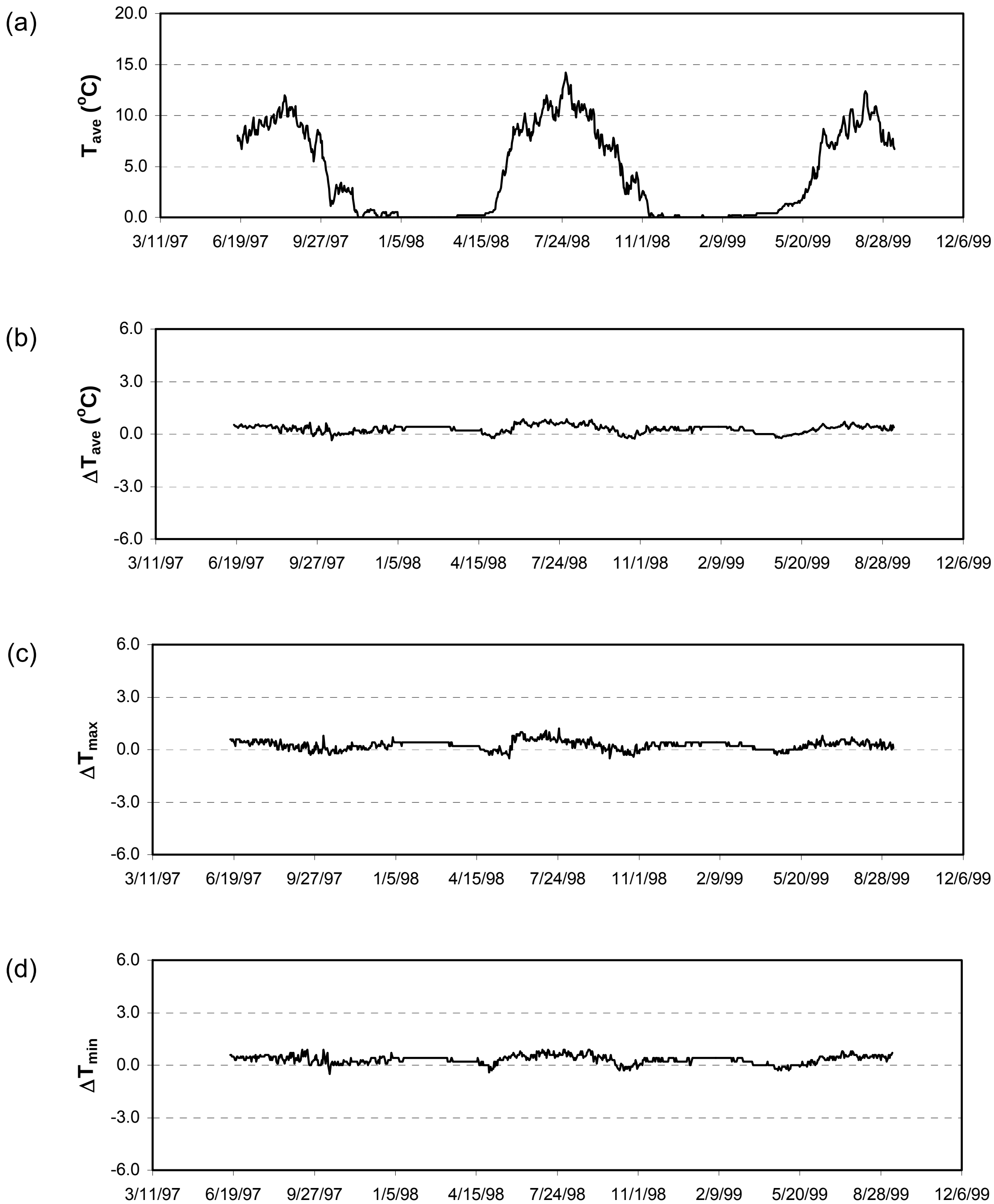

Fig. 3. Stream temperature data for B4. See caption for Fig. 1 for detailed explanation.

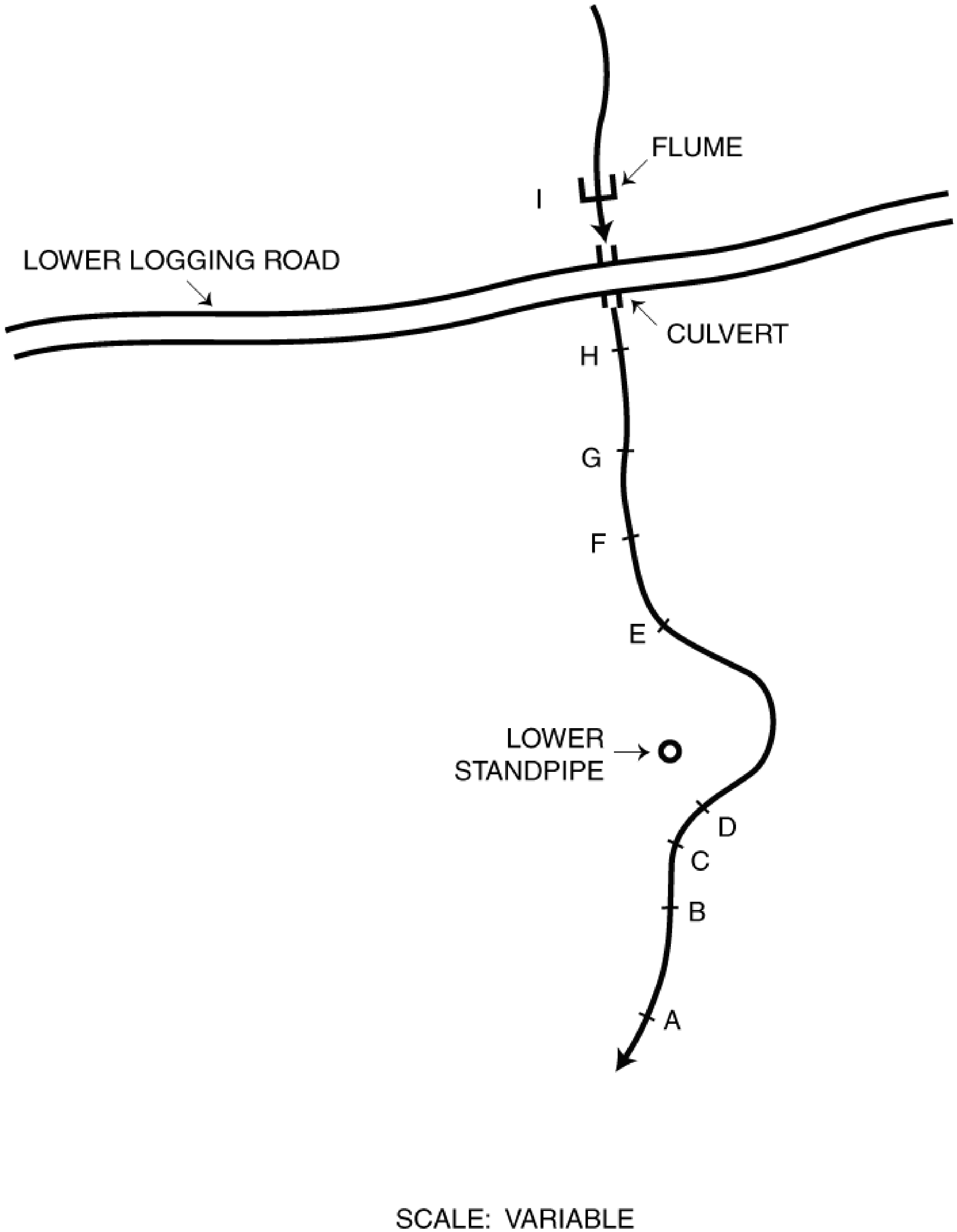

Fig. 4. Schematic diagram of B5, indicating locations where observations were made during pilot studies in summer 1999.

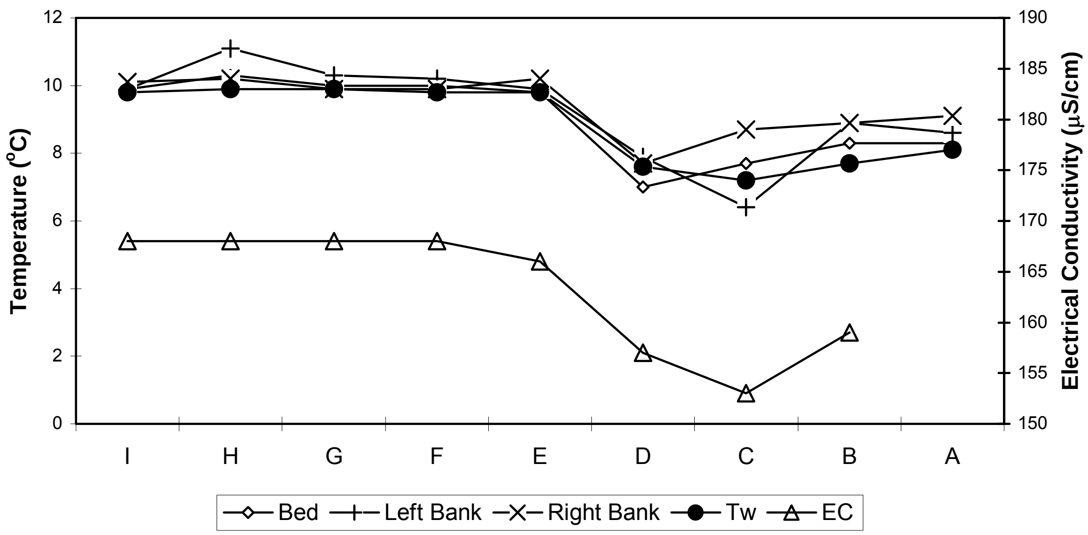

Fig. 5. Variations in stream temperature, subsurface temperatures in the stream bed and banks, and electrical conductivity along B5 on August 11, 1999.

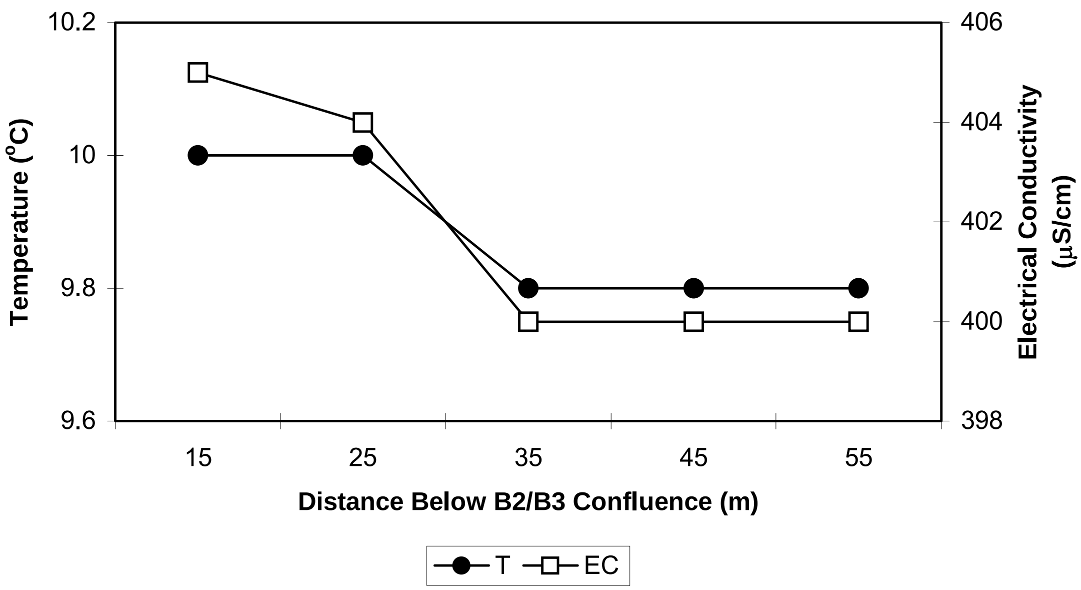

Fig. 6. Variations in stream temperature and electrical conductivity during steady-state salt injection in B3 on August 12, 1999. Distances below the B2/B3 confluence are approximate.

PAGE BREAK

Effects of Forest Harvesting On Stream Temperatures in the Central Interior of British Columbia: The Moderating Influence of Groundwater and Lakes

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INTRODUCTION

Of all the factors regulating biological processes in small streams, water temperature is one of the most important, and it is often a key consideration when planning timber harvesting activities around streams. As a result, the scientific literature is replete with studies that examine the impacts of forest harvesting practices on stream temperatures (thoroughly reviewed in Beschta et al. 1987 and Hicks et al. 1991).

Stream fish populations (and salmonid populations in particular) are often the primary resource fisheries managers wish to protect, and interactions between forestry practices and fish are numerous and potentially complex. For example, processes relevant to fish populations that are affected by water temperature include: stream primary and secondary productivity, dissolved oxygen content of the water, metabolic rates of instream fauna, egg development, growth and growth efficiency, activity levels, and fish behavioral responses (such as the timing of migration and spawning events; Beschta et al. 1987; Hicks et al. 1991). As a result, changes in stream temperatures (and associated light regimes) following streamside harvesting can have both beneficial and detrimental effects on fish. Beneficial effects can occur as a result of increased food supply (through increased sunlight and temperatures), leading to more efficient prey capture and increased growth (Gregory et al. 1987). However, those benefits may be offset if increased stream temperatures exceed the thermal tolerances of resident fish (Hicks et al. 1991).

Evidence from the literature is overwhelming in supporting the conclusion that streamside timber harvesting results in increased summer stream temperatures, whether the magnitude of the increase is large or small (see reviews in Beschta et al. 1987; Hicks et al. 1991; Anderson 1973; and Brown1985). However, most North American fish/forestry studies have been conducted in coastal regions, and this hinders the application of their results to other areas with different climatic regimes. In British Columbia (B.C.), fish/forestry studies have only recently been initiated in the central interior region of the province (Macdonald and Herunter 1998), which is characterized by a temperate climate. Furthermore, studies investigating the impacts of forestry activities on water temperatures tend to focus on headwater streams, with little attention being paid to streams headed by lakes. In this paper we hope to address two gaps in our current knowledge of timber harvesting impacts on stream temperatures: that of harvesting impacts in interior (temperate) regions of B.C., and of the potential influence of lakes in moderating those impacts. We describe an experiment comprising four small (first order), lake-headed streams that examines stream temperature responses to riparian harvesting. Our primary hypothesis is that the removal of >50% of the streamside vegetation, thereby increasing the amount of incident solar radiation reaching the stream surface, will result in increases in mean daily water temperatures during summer months. We follow up this hypothesis with an examination of some of the consequences of increased water temperatures to fish growth.

MATERIALS AND METHODS

Experimental Design

In order to examine stream temperature responses to clear-cut logging, we chose a paired stream case study approach using an “extensive before-after” design (Hall et al. 1978). This design requires pre- and post-harvesting monitoring of multiple streams (both controls and treatment), and has the advantage of allowing for true controls and replication of treatment effects. The main disadvantages lie in the inability to assess year-to-year variation and long term impacts (Hicks et al. 1991).

In August 1997, we began monitoring four small, lake-headed, unlogged, fish bearing streams within the Nation River watershed in central B.C. (Fig. 1). Two of the streams were logged by Canadian Forests Products Ltd. (CANFOR) in February 1998, and two streams served as controls and remained unlogged for the duration of the study (until September 1999). Because we were constrained by CANFOR’s operational timetable, we were only able to collect 6 months of pre-logging temperature data on the treatment streams.

Site Descriptions

All four streams were located in the Sub Boreal Spruce biogeoclimatic zone of B.C. (Farley 1979), and all streams were headed by small (<20 ha), relatively shallow lakes. Because we used a paired stream design, we attempted to match stream physical characteristics as closely as possible (Table 1). All streams were low gradient (<4%) with channel morphologies consisting mainly of pool-riffle sequences. The treatment streams varied in the linear length receiving streamside harvesting (118/48 had approximately 600m of stream length running through the cutblock while 118/16 had only 400m), and as a result the control streams were chosen to mimic their respective paired partners. Thus, PUR/9, with a study section length of 600m, was chosen as a control pair for 118/48 while HIP, with a length of 400m, was chosen as a control pair for 118/16.

The most common and most abundant fish species in all four streams was rainbow trout (*Oncorhynchus mykiss*), and this was the only fish species caught in HIP Creek. Small numbers of suckers (*Catostomus* spp) and lake chub (*Couesius plumbeus*) were captured in the two treatment streams (118/48 and 118/16), while suckers, lake chub, sculpins (*Cottus* spp), and burbot (*Lota lota*) were captured in control stream PUR 9 (Table 1).

Logging Treatments

Timber harvesting treatments consisted of clear-cut logging and included the removal of streamside timber within the riparian zone on both banks. The two treatment streams were originally classified as “S3” streams under the British Columbia Forest Practices Code (based on their bank-full widths and the presence of fish) and were thus originally allotted 20m buffer strips along each bank within which no harvesting was permitted. However, in order to assess the impacts of streamside harvesting on streams in the S3 category, harvesting was permitted within the riparian zones subject to the following conditions:

1) A 5 m machine exclusion zone was maintained around the streams.

2) Within a riparian management zone (extending 30m from each stream bank), mature commercial timber (>15cm DBH for lodgepole pine (*Pinus contorta*) and >20cm DBH for spruce (*Picea* species) and balsam fir (*Abies lasiocarpa*) was harvested, while non-commercial and deciduous timber was left standing where operationally feasible. Unlike the remainder of the cutblock, the riparian management zone was therefore not completely clear-cut, as only approximately 50% of the standing timber was removed.

3) Timber was felled and yarded away from the stream.

The above conditions are not artificial and represent the same level of restrictions that are applied to all streams where logging is permitted within the riparian zone (Anon 1995). Because of our desire to simulate an operational (versus purely scientific) treatment, we asked CANFOR to conduct logging to conform to their standard operating procedures to remove any potential scientific bias.

Temperature Monitoring

Automated temperature loggers (temperature range –5 ^{0}C to 35 ^{0}C; resolution 0.2 ^{0}C; Vemco Ltd) were installed at both the upstream (US) and downstream (DS) boundaries of the study sections to record hourly stream temperatures. For the treatment streams, the US and DS sites were at the uppermost and lowermost edges of the cutblock boundaries, respectively. As a result, our predictions regarding impacts of timber harvesting on stream temperatures apply only to the DS sites, as the US sites were unaffected by logging and served as additional controls to monitor year-to-year variation. Because the case study streams also formed part of a study examining the impacts of clear-cut logging on fish movement, the sites were visited on average every 2-3 days from May to November, and during each visit we ensured that the temperature loggers were fully submerged. At no time during this period were the loggers found out of water and exposed to air. During each visit a water temperature reading was also taken with a calibrated mercury thermometer to check the accuracy of the temperature loggers. The automated logger readings were within 0.1 to 0.3 ^{0}C of the thermometer readings.

Estimates of the amount of stream shading provided by the riparian vegetation (reported in terms of percent canopy cover) were obtained with a hand held densiometer following the methods outlined in Lemmon (1956). Five estimates were obtained along each study section and averaged. In order to determine the amount of shade that was lost following streamside harvesting, canopy cover estimates were measured at the same locations along the study sections during both pre- (1997) and post-logging (1998-1999) summers.

PresentatIon of Data

The hourly temperature data were averaged to produce daily mean, maximum and minimum values. The layout for the temperature graphs is identical for all four streams: in panel (a) the mean daily trends are plotted for the US and DS sites, as well as the difference between US and DS sites (a positive value indicating that the DS site is cooler than the US site). Panels (b) and (c) show the daily maximum, minimum, and temperature fluctuations (daily maximum-daily minimum) for the US and DS sites, respectively. The scales on all temperature graph axes are identical to allow for direct comparisons among similar graphs. For the period from August 01 to Sept. 30, we also calculated the average daily mean temperature and fluctuation for the US and DS sites, as well as the average net cooling (US-DS) for all four streams in each of the three sampling years (1997-1999; Table 2). The data were restricted to these 2 months because it is the only period for which we had relatively complete sets of both pre- and post-logging temperature records, and thus the only period for which comparisons across different years are valid.

RESULTS

Primary Hypothesis: Streamside logging will result in increases in mean daily temperatures at the DS sites of the treatment streams

In 1997, prior to logging, all four streams were relatively heavily shaded by streamside vegetation (canopy covers in excess of 60%; Fig.2), and all streams cooled as they traveled from the US to the DS boundaries (as shown by the positive difference between US and DS temperatures; Figs. 3a, 4a, 5a, and 6a). Although we were limited to 6 months of pre-harvesting data (of which four months were in the winter), examination of the daily temperature patterns did not reveal any substantial changes in the two treatment streams following streamside logging. Surprisingly, the two treatment streams continued to cool as they traveled through the harvested cutblocks (Figs. 3a and 4a), despite the increased incident solar radiation resulting from the removal of >50% of the streamside vegetation in 1998 (Fig. 2). Little interannual variation in cooling patterns was evident in the two control streams (Figs. 5a and 6a; Table 2), which suggests that climatic anomalies were not responsible for the trends seen in the treatment streams.

Although the two treatment streams continued to cool through the logged cutblocks, we were interested in determining if the *magnitude* of the cooling between US and DS boundaries changed following streamside harvesting. We can therefore modify our primary hypothesis in the following way:

An increase in incident solar radiation will warm the streams as they travel through the harvested cutblocks, but not enough for the temperatures at the DS site to exceed those at the US site. Consequently, the cooling trend will be less pronounced after logging.

An examination of Figures 3a and 4a does not reveal any obvious differences between the treatment streams before and after harvesting, as both cooled an average of 1-2 ^{0}C (with considerable variation ranging from 0-5 ^{0}C) during the summer months in each of the three years (Table 2). To further test for any changes in the magnitude of cooling between US and DS sites, we performed an analysis of covariance (ANCOVA) on the difference between US and DS temperatures for each treatment stream and its paired control for the pre- (1997) and post- (1998/1999) harvest years. This approach allowed us to look at cooling trends before and after logging while accounting for background interannual variation (by including data from the control streams). If the magnitude of the cooling between US and DS sites decreased following streamside harvesting, then we would expect the intercepts of the regression equations to be lower during post-harvest years (assuming that the slopes of the regressions were equal). Because we were primarily interested in impacts on summer temperatures, and because we were constrained by our relatively short pre-logging monitoring period, for these analyses we restricted our data to the period covering Aug. 01-Sept. 30. (Note that for treatment stream 118/48 and its control pair PUR 9, we had to omit the 1999 data from the ANCOVA analyses because no relation was found between the difference in US and DS temperatures during that year (ANOVA, $p>0.42$; Fig. 7b). We speculate that this was due to the construction of an extensive beaver dam at the outlet of the lake heading PUR 9 in October 1998, resulting in artificial flow regimes throughout 1999.)

For treatment stream 118/16 and its control pair (HIP), the slopes of the regressions were homogeneous across all 3 years (ANCOVA, $p>0.42$), but the intercepts for both post-logging years (1998 and 1999) were significantly *higher* than for the pre-harvesting year (1997); (ANCOVA, $p>0.001$; Fig. 7a). This indicates that not only did treatment stream 118/16 continue to cool following streamside harvesting, but that the cooling trend was more pronounced following logging. Because the intercepts for the 1998 and 1999 regressions were not significantly different from each other (ANCOVA; Bonferroni corrected $p>0.04$), we combined the data for these two years; (Fig. 7a). As an example to illustrate this point, when HIP (control) cooled by 0.4 ^{0}C as it went from the US to the DS boundary, stream 118/16 (treatment) cooled by approximately 1 ^{0}C in 1997, and by approximately 1.7 ^{0}C during 1998/1999 (Fig. 7a). The increased cooling following streamside logging is also illustrated by the data summary in Table 2.

A different pattern was seen for treatment stream 118/48 and its paired control (PUR 9). The results show that although the slopes were again homogeneous (ANCOVA, $p>0.31$), the intercept was higher for the pre-harvest period (1997) when compared to the post harvest period (1998; Fig. 7b). This indicates that treatment stream 118/48 cooled less as it traveled downstream following streamside logging. However, the difference in cooling between years was relatively slight, averaging only 0.2 ^{0}C (Table 2) which was within the resolution of the automated data loggers.

Additional comments regarding the temperature patterns

Several other points are worth mentioning regarding Figures 3, 4, 5, and 6.

(1) Mean daily temperature patterns at the US and DS sites are very similar to each other during the coldest months of the year in this temperate region (Oct. 01 to May 01), but de-couple during the summer months. The start and end of the de-coupling coincide roughly with the time of ice breakup (in the spring) and ice formation (in the fall) on the surface of the headwater lakes, and at a time when stream temperatures are near zero.

(2) Of particular interest is the degree to which the US site of stream 118/16 is naturally warm (Fig. 3b). Temperatures as high as 24 ^{0}C were recorded as early as May 24, 1998, and almost reached 30 ^{0}C in June of that year. The management implications relating to such naturally warm streams are discussed below.

Monitoring of additional streams: a comparison of headwater vs. lake-headed streams

In 1998 (the first year following logging), we began to suspect that the presence of headwater lakes was an important factor to consider when examining the effects of streamside harvesting on stream temperatures. To test this hypothesis, in the summer of 1999 we installed automated temperature loggers at the US and DS cutblock boundaries of two additional, previously logged streams within the same biogeoclimatic zone as our 4 case study streams (Fig. 1). These two streams were physically similar to our case study streams except that the cutblock sizes were an order of magnitude larger and consequently the length of stream we monitored was doubled (Table 1). One stream (727) was clear-cut in 1991 and was headed by a small lake, while the second stream (TCH CC) was a headwater stream (with no lake) that was clear-cut in 1996. Both streams were harvested prior to the introduction of the British Columbia Forest Practices Code and as a result no commercial timber was left standing along the riparian zones. Note that in these two cases we do not have pre-logging data and make no predictions about post-harvesting impacts (such as changes in the magnitude of cooling between US and DS sites). Instead, based on our observations from the two lake-headed treatment streams (118/16 and 118/48), we predict that water temperatures in 727 (lake-headed) will cool, while temperatures in TCH CC (headwater) will increase, as the streams travel through the cutblocks.

In the summer of 1999, mean daily temperatures in stream 727 ranged from 8-18 ^{0}C (Fig. 8a), similar to the other lake-headed streams examined in this study. As predicted, water temperatures cooled as the stream traveled through the cutblock, with the magnitude of the cooling ranging between 0.2-1.8 ^{0}C (mean 0.9 ^{0}C; Fig. 8a). In contrast, stream TCH CC warmed as it traveled from the US to the DS boundary, with warming ranging from 2-7 ^{0}C (mean 4.3 ^{0}C; Fig. 9a). This stream was also considerably cooler than the others examined in this study, with mean daily temperatures at the US (unlogged) boundary ranging from 4-9 ^{0}C (Fig. 9a).

DISCUSSION AND MANAGEMENT IMPLICATIONS

What factors contribute to the lack of warming in the treatment streams despite the removal of >50% of the streamside vegetation?

Using an energy balance model, Brown (1985) showed how increasing the direct solar heat load to a stream results in increased water temperatures, and literature reviews on impacts of clear-cut logging on stream temperatures overwhelmingly support this conclusion (Beschta et al. 1987; Hicks et al. 1991; Anderson 1973). However, we were unable to uncover a single published study that demonstrates continued (and in the case of stream 118/16, increased) cooling of water temperatures following streamside harvesting. Our canopy cover estimates (Fig. 2) leave no doubt that the amount of incident solar radiation increased following logging, yet the treatment streams continued to cool as they traveled through the cutblocks (Figs. 3a and 4a). In order to address these findings, we consider other physical features within the ecosystem that may have compensated for the increase in solar radiation, and discuss their relevance and applicability to our streams. We stress that what we propose below is speculative, as we have no extensive physical data other than the temperature records. However, we believe the following potential contributing factors to be plausible.

A) Groundwater inputs increased following logging:

Clear-cut logging has been shown to increase summer low flows in small streams by reducing groundwater uptake that occurs through evapotranspiration, resulting in more water reaching the stream and therefore augmenting flows (Rothacher 1971; Harr et al. 1979). We propose that this mechanism, whereby increased groundwater inputs compensate for the increase in solar radiation, is largely responsible for the continued cooling observed in our treatment streams following logging (note that Brown's (1985) energy balance model assumes no groundwater inputs to counteract increased solar radiation levels). In the case of stream 118/16, where cooling increased following logging, the cooling from added groundwater inputs may have exceeded solar radiation inputs. Furthermore, water temperature readings taken approximately every 25-30m along the study sections of our treatment streams in August 1999 showed a gradual decrease from the US to the DS sites, suggesting that groundwater inputs were diffuse and continuous along the entire study reaches.

B) Water volume/velocity were sufficient to compensate for the increased solar inputs:

The removal of >50% of the streamside vegetation may not have been sufficient to warm the streams if flows were high enough and travel times through the cutblocks short enough so that the duration the water was exposed to the increased solar radiation was minimized. For example, a stream with a discharge of 200 m^3/s and a travel time of 10 minutes through a cutblock will warm to a lesser degree than a stream with a discharge of 20 m^3/s and a travel time of 60 minutes through the same cutblock. This mechanism can be dismissed as contributing to the cooling seen in our harvested, lake-headed streams because of the low summer discharges observed in these streams

(<1 L/s in the case of stream 118/16 during August 1999; Table 1). Furthermore, given the summer discharges reported in Table 1, travel times through the cutblocks of approximately 30-70 minutes (estimated using average channel cross section water velocities), and Brown's (1985) energy balance model, we would have predicted unequivocally that all three harvested, lake-headed streams (118/16, 118/48 and 727) would be warmer at the DS sites relative to the US sites.

C) The remaining non-commercial and deciduous vegetation left standing in the riparian zone following harvesting continued to provide sufficient streamside shading:

Although logging within the riparian zone typically eliminates shading provided by large conifers, sufficient shade may still be provided by trees (small conifers and deciduous shrubs) left standing or regenerated along the stream banks. We do not believe this mechanism to contribute significantly to the cooling trends seen in our treatment streams, as only 20-30% canopy cover remained following logging (Fig. 2). However, it may partly explain the lack of warming seen in lake-headed stream 727, where deciduous shrubs and small trees continued to provide 46% canopy cover despite being clear-cut to both banks in 1991. In contrast, headwater stream TCH CC retained 0% canopy cover after being clear-cut and undergoing a broadcast burn in 1996, and was consequently completely exposed to incident solar radiation.

D) Stream aspect influenced stream cooling:

Although our densiometer estimates showed a reduction of >50% of the riparian canopy cover following logging (Fig. 2), stream aspect (as a measure of the sun's position relative to the stream) determines the actual amount of sunlight reaching the stream surface. It is therefore possible that a reduction in canopy cover may not translate into increased solar radiation reaching the stream itself. To test this hypothesis, in August 1999 we supplemented our densiometer data with estimates of a second measure of canopy cover, angular canopy density (ACD; Summers 1983). ACD takes into account the position of the sun relative to the stream and is therefore a more accurate measure of the amount of solar radiation reaching the water surface. In contrast to densiometer estimates, ACD estimates can vary with season and with time of day (because of changes in the sun's angle and position in the sky). Our ACD measurements were therefore recorded at the same locations as the densiometer readings, and were calculated using the position and angle of the sun at noon in the middle of August (the hottest part of the day during the hottest month).

The ACD canopy cover estimates for all six streams were as follows: 118/16 (19%); 118/48 (14%); HIP (52%); PUR 9 (67%); 727 (37%); TCH CC (0%). Canopy cover estimates obtained using the ACD method were therefore lower than those measured with a densiometer (except for TCH CC, which remained at 0% cover), and this suggests that stream aspect did not contribute significantly to the cooling trends observed in our harvested lake-headed streams.

E) The presence of headwater lakes mitigated stream warming:

In searching for factors that may have mitigated warming in streams 118/16, 118/48 and 727, the presence of headwater lakes was perhaps the biggest and most obvious difference between our study and those reported in the literature (which typically monitor headwater streams). Small, shallow lakes of the type encountered in our study are particularly susceptible to warming during summer months, as their large surface areas are directly exposed to solar radiation. Consequently, water leaving the lake and entering outlet streams can be considerably warmer when compared to headwater streams, even in forested systems. For example, during 1998 maximum water temperatures in excess of 25 ^{0}C were recorded in stream 118/16 at the US cutblock boundary (a site unaffected by logging) as early as June 01 (Fig. 3b). A similar, although not as extreme, trend was seen at the US site of streams 118/48 and PUR 9, where maximum daily temperatures exceeded 20 ^{0}C during the summer of 1998 (Figs. 4b and 6b). Even the coldest of the four case study streams, control stream HIP, reached daily maximum temperatures of between 15-20 ^{0}C at the US site in 1998 (Fig. 5b). In contrast, the US (forested) site of headwater stream TCH CC barely reached 10 ^{0}C in the summer of 1999, and only then for a short period of time (Fig. 9b). Nearby headwater streams have temperature regimes that are similar to the forested section of THC CC (Macdonald et al., this compendium), and we therefore conclude that lake-headed systems are naturally warmer than headwater systems in this geographical region.

If stream cooling is accomplished by groundwater inputs (which are typically colder than stream surface waters, especially in the case of lake-headed streams), then thermodynamic theory predicts that the *magnitude* of cooling will be greater for a stream that is initially warmer when compared to a colder stream (assuming all other factors such as the discharge and temperature of the groundwater, the discharge of the stream, and the composition of the streambed are held constant). This is shown graphically in Figure 10, where the relation between initial and final stream temperatures was calculated (assuming an input of 5 ^{0}C groundwater accounting for 20% of the total stream discharge) using a standard thermodynamic equation (Brown 1985). The actual numbers used in the equation are inconsequential: the figure is included to illustrate how the *difference* between initial and final temperature increases with increasing initial temperatures. The thermodynamic equation further predicts that: (a) increases in initial stream temperatures and in groundwater inputs, and (b) decreases in groundwater temperatures and in the discharge of stream surface waters, will all contribute to stream cooling (although interactions among these terms would increase the complexity among the relations). Summer conditions in streams 118/16 and 118/48 (low flows and initially warm temperatures promoted by the presence of head-water lakes) therefore favored substantial cooling by groundwater inputs, despite the increased exposure to solar radiation following logging.

Although we believe all of the above factors can potentially influence stream warming, we favor the combined mechanisms involving increased groundwater inputs (factor A) and the presence of headwater lakes (factor E) as being primarily responsible for the continued cooling trends seen in our treatment streams following streamside harvesting.

What are the fisheries management implications of warmer streams? Comparisons between our treatment streams and control stream HIP.

Although no temperature increases were observed in our two treatment streams (118/16 and 118/48) following logging, both streams were naturally warmer than control stream HIP. The temperature differences between these streams (averaging 2 ^{0}C; Table 2) also fall within the reported range of temperature increases resulting from streamside harvesting in temperate regions (Beschta et al. 1987). Therefore, the treatment streams can be used to investigate fish behaviour in systems that are considerably warmer than many unharvested streams in the central interior of B.C., including control stream HIP. (We exclude control stream PUR 9 from our comparisons because we did not collect extensive fish data from this stream). These three streams are ideal for this type of comparison because they are physically similar (Table 1) but differ in their thermal regimes.

Increased stream temperatures have generally been considered to be detrimental to fish development, physiology, behaviour, and activity (Beschta et al. 1987). However, potential impacts must be assessed in terms of the fish species that are likely to be affected, and this leads to some interesting management implications. For example, consider the life history strategy of sockeye salmon (*Oncorhynchus nerka*) within the Takla watershed in the central interior of B.C. In this region, sockeye salmon spawn in their natal streams in late summer, with fry emergence and emigration occurring the following spring. It has been speculated that increases in stream temperatures would lead to accelerated development to the fry stage and earlier, possibly untimely, emergence and emigration when environmental conditions are potentially unsuitable (Fig. 11a; Macdonald et al. 1998). However, the same forecast does not necessarily hold true for a species like rainbow trout, which typically spawns in the spring with fry emerging from stream gravels by mid to late summer. Increased stream temperatures could also lead to earlier fry emergence (Fig. 11a), but because rainbow trout are spring spawners, the advanced timing would allow fry additional time to grow and accumulate energy reserves before the onset of winter, thereby conferring a size advantage and increasing the likelihood of overwinter survival.

To test our hypothesis as it applies to rainbow trout, we compared the dates of earliest emergence for trout fry in our two treatment streams and in control stream HIP for 1999. We predict that the warmer temperatures in 118/16 and 118/48 would result in earlier fry emergence and consequently in larger fry by the end of the summer. The data show that not only did trout fry emerge approximately 2 weeks earlier in 118/16 (July 03) and 118/48 (July 05) than in HIP (July 19; Fig. 11b), but that by mid-September those fry were also significantly bigger and heavier than fry from HIP (ANOVA, $p<0.001$; Fig. 11c). While we cannot rule out other potential reasons for the increased growth observed by mid-September (for example, the treatment streams may also be naturally more productive than HIP), we propose that the naturally warmer waters provided a net benefit to rainbow trout fry in terms of accelerated emergence and growth.

Species-specific thermal tolerances and growth optima are also important fisheries management considerations when assessing logging impacts. Temperatures that exceed thermal preferences and optima are generally detrimental to fish, as they lead to increased metabolic requirements and to decreased food conversion efficiencies and dissolved oxygen levels in water (Spigarelli and Thommes 1979). Alternatively, water temperatures that are too cold can impede growth and development. In northern, temperate latitudes, small streams may be so well shaded and consequently so cold that fish growth is stunted, even for cold-water adapted species like trout. Because growth and food conversion efficiency typically increase with temperature up to a certain point, and then decrease as the optimum is surpassed (Hokanson et al. 1977), criteria must be developed to allow fisheries managers to assess under what conditions temperature increases following logging are likely to be beneficial or detrimental.

In laboratory studies, rainbow trout have been shown to not only have a wide range of thermal tolerances (0-29 ^{0}C; Lee and Rinne 1980; Charlon et al. 1970; Reiser and Bjornn 1979; Hokanson et al. 1977) and preferenda (11-22 ^{0}C; Spigarelli and Thommes 1979; Cherry et al. 1977; Javaid and Anderson 1967; Eaton et al. 1995), but that these temperatures were highly dependent on the initial acclimation temperature. However, Hokanson et al. (1977) demonstrated that the range of temperatures for optimum growth (i.e. the range within which growth is maximized) in rainbow trout was considerably narrower (15.5-17.3 ^{0}C in a thermally fluctuating environment). If we accept Hokanson et al.'s (1977) growth optima (even though these are also dependent on acclimation temperature and food availability), we can then compare the amount of time water temperatures in our streams fell within this range as a further test of the impacts of warmer water on fish populations.

Because the automated loggers took temperature readings every hour, we tallied the total number of times that stream temperatures were recorded within the optimum range and assumed that they remained within that range for the duration of the hour. Data were tallied for all 12 months of 1998 (the warmest of the three years) and for approximately 8 months (January to Sept. 7) for 1999 (by which time daily maxima were exceeding 15 ^{0}C only at the US site of stream 118/16). The results show that both treatment streams were within the optimum temperature range for trout growth on average two to five times longer than HIP across both years (Table 3). This suggests that in terms of temperature, the treatment streams were better suited for trout growth, despite the extremes in daily maxima recorded at the US site of stream 118/16. It is particularly striking that in 1999, stream temperatures at the DS site of HIP were within the optimum growth range for a total of only 34 hours.

If the treatment streams represent thermally superior environments for trout growth, we wold predict that fish in those streams would be larger across all age classes, not just within the fry class. Using enumeration data from streamwide electroshocking surveys conducted in August 1999, trout length-frequency distributions for our three streams support this hypothesis. The data show that despite considerable overlap in some age classes, the size range for each age class encompassed larger fish in the treatment streams than in HIP (Fig. 12). Furthermore, if we combine all age classes other than fry, fish in stream 118/16 were significantly larger than those in HIP (ANOVA $p<0.002$). Using the same combination of age classes, mean fish size in stream 118/48 was not significantly different from HIP (ANOVA, $p>0.15$), even though individual age classes were clearly larger (Fig. 12). This may have been due to heavy predation on older fish in stream 118/48 by mink in the month prior to our population census, reducing the overall number of larger fish and skewing the population size distribution toward smaller animals. It is therefore reasonable to suggest that the enhanced thermal environments in streams 118/16 and 118/48 resulted in larger rainbow trout. However, it is possible that intrinsic but undocumented differences amongst the streams, e.g., nutrient status, may also have played a role.

It can be argued that the higher mean daily temperatures recorded in stream 118/16 led to higher fish mortality, resulting in less competition for food resources and greater growth (Wedemeyer et al. 1984). This is unlikely to apply to our streams, as standardized density estimates from our 1999 census were higher in both treatment streams (5.2 fish/m^3 in stream 118/16 and 7.6 fish/m^3 in stream 118/48) than in HIP (4.4 fish/m^3).

Not only did stream temperatures in 118/16 fall within the thermal optima for a longer period than in HIP (Table 3), but they were also near (if not exceeding) the upper lethal limit for rainbow trout for longer periods as well. If we accept an upper lethal limit of 25.75 ^{0}C for this species (Jobling 1981 and references therein), then stream temperatures at the US site of 118/16 exceeded this limit for a total of 70 hours during the course of the summer in 1998 and for a total of 32 hours in 1999. No other site exceeded this limit during our three years of monitoring. However, fish in that section of the stream were relatively abundant and appeared healthy. The reasons for this may lie in a combination of behavioral thermoregulation and local adaptation. By actively seeking groundwater sources, fish are often able to survive short periods of time in water where temperatures are well in excess of their upper thermal limits (Beschta et al. 1987). The longest consecutive period in which lethal temperatures were reached in 118/16 was 3 hours (and then only on 2 separate occasions), and it is plausible to suggest that fish were able to survive these temperatures by seeking localized thermal refugia or by migrating to cooler areas. Fish may also be physiologically or genetically adapted to such extreme thermal environments (Matthews 1987). If streams are naturally warm and exhibit episodic bouts of extreme temperatures, then the ability to withstand these temperatures would be advantageous and indeed critical for survival. The potential role of local adaptation in allowing cold-water fish to tolerate high temperatures can be illustrated by considering the headwater lake for control stream HIP. This lake can best be described as a large pond, with an exposed (albeit small) surface area, shallow, unstratified waters, and a high potential for extreme summer temperatures (Tyermann 1999b). Given these conditions, we would have predicted that few trout would be able to survive in this lake, yet not only were rainbow trout the only species that were captured, but they were abundant, large (20-30 cm fork length) and apparently healthy (Tyermann 1999b) (comparisons to the other headwater lakes are not warranted because they stratify during summer months (Tyermann 1999a; Pillipow 1996; Hunter 1996) and thus offer thermal refuges to fish). The effect of increased temperatures on fish populations that are presumably not adapted to high temperatures (such as populations in cold headwater streams) is unknown, and should form part of future research.

CONCLUDING REMARKS AND RECOMMENDATIONS

This study has shown conclusively that under certain conditions clear-cut logging within the riparian zones of small streams does not result in increased summer water temperatures, despite a significant reduction in canopy cover and low summer flows. It is important to note, however, that the analyses presented here are exploratory and may warrant subsequent revision. From a fisheries perspective, we have also tried to show how temperature impacts must be assessed in terms of the fish species that will likely be affected. Our evidence suggests that the naturally warmer waters of the treatment streams provided an overall net benefit for rainbow trout emergence and growth. It is therefore possible for streamside timber harvesting to be conducted in such a way as to accommodate the needs of both the forestry industry and fisheries managers.

It is also important to note the impacts and benefits we ascribe to streamside harvesting are short term: we have no long term data for our treatment streams and make no claims as to potential future consequences, such as loss of fish habitat resulting from long term reductions in large organic debris (LOD) recruitment. However, operational harvesting techniques and reforestation that follow best management practices are designed to minimize many of these long-term effects and protect against potential future habitat degradation. Furthermore, in the absence of additional data, we make no claims as to harvesting impacts on other biological components of stream ecosystems, such as primary and secondary productivity, or social interactions and competitive outcomes within fish populations. Nevertheless, clear-cut logging is generally associated with increases in stream primary and secondary productivity (Gregory et al. 1987). If this is coupled to an enhanced thermal environment then significant short term benefits could result for certain fish populations, as long as any temperature increases do not exceed the thermal tolerances of resident fish for prolonged periods of time.

We do not advocate a return to historical logging practices. However, in light of new evidence such as the results presented here, blanket restrictions on forestry activities around small streams (like those legislated by B.C.'s Forest Practices Code) may not be appropriate. It is likely that cold, well shaded, temperate streams would benefit most from streamside logging, especially if harvesting increased the amount of time water temperatures reached the optimum growth range for resident fish. For example, rainbow trout in control stream HIP would likely benefit if HIP was subjected to the same careful logging treatment that was applied to streams 118/16 and 118/48. The challenge facing forestry planners and fisheries managers alike is identifying those streams. Future research should focus on developing simple methods and guidelines to allow managers to accurately forecast impacts of clear-cut logging on stream temperatures and the consequences to resident fish populations. This is where multi-disciplinary collaborations (such as the Stuart-Takla Fish-Forestry Interaction Project; Macdonald and Herunter 1998) become particularly valuable and are to be encouraged.

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Table 1. Physical characteristics of the four study streamscomprising the “extensive before-after” experimental design and the two additional streams monitored during the summer of 1999. (T) beside the stream name indicates treatment, while (C) indicates control. Bankfull width, height, and stream and hillslope gradients represent averages of 5 measurements taken at equidistant points along the stream study sections. Hillslope gradients were measured on both banks and averaged. Discharge represents a single estimate taken in August, 1999 to provide an example of summer base flows (data are provided for both the upstream (US) and downstream (DS) sites). Lake characteristics are for the lakes at the head of each stream.

Stream	Cutblock Area (ha)	Stream length monitored (m)	Bankfull Width (m)	Bankfull Height (cm)	Aspect	Stream Gradient (%)	Hillslope Gradient (%)	Discharge (US/DS; L/sec)	Fish Species Present[1]	Lake Surface Area[2] (ha)	Lake Mean Depth[2] (Max Depth; m)
“Before-After” Design											
118/16 (T)	54	400	1.7	29.5	SE	1.8	19.3	0.8/1.0	RBT; SK	13.1	3.1 (8.7)
118/48 (T)	36	600	1.7	35.2	SW	1.9	23.5	4.9/4.3	RBT; SK	15.3	2.5 (5.5)
HIP (C)	n/a	400	1.0	26.7	N	3.1	34.2	5.4/6.5	RBT	4.7	1.5 (3.0)
PUR 9 (C)	n/a	600	2.4	36.3	SE	2.3	24.5	8.2/9.3	RBT; CHB; SK; BB; SC	18.1	1.7 (6.5)
Additional Streams											
727	207	1000	2.2	46	NE	2.5	25.8	11.3/13.2	RBT	11	Not available
TCH CC	507	1100	2.2	43	SE	7.5	6.8	12.7/17.0	RBT	No Lake	No Lake

[1]RBT=Rainbow trout (*Oncorhynchus mykiss*); SK=Sucker (*Catostomus* spp); CHB=Lake Chub (*Couesius plumbeus*); BB=Burbot (*Lota lota*); SC=Sculpin (*Cottus* spp).
[2]Data for lakes heading streams 118/16 and PUR 9 are from Pillipow (1996) and Hunter (1996), respectively. Data for lakes heading streams 118/48 and HIP are from Tyerman (1999a and 1999b).

Table 2. Summary of average mean daily temperatures and fluctuations (daily maximum-minimum) for the US and DS sites of the four case study streams during the period from Aug. 01 to Sept. 30, 1997-1999. A positive number under heading of “Net Cooling” indicates that the stream cooled as it traveled from the US to the DS sites. Streams 118/16 and 118/48 were treatment streams, and streams HIP and PUR 9 served as controls.

Strea	Yea	US (°C)	US (°C)	DS (°C)	DS (°C)	Net (°C)
118/16	1997	12.5	7.7	11.3	3.7	1.7
	1998	11.0	6.3	9.1	3.5	1.9
	1999	13.7	7.3	11.7	4.3	2.1
118/48	1997	10.6	2.2	9.2	1.6	1.5
	1998	9.7	4.3	8.4	3.3	1.3
	1999	14.4	3.4	12.3	3.6	2.1
HIP	1997	10.3	2.1	9.7	1.8	0.5
	1998	10.2	2.4	9.7	2.2	0.5
	1999	11.2	2.3	10.8	2.0	0.5
PUR 9	1997	13.1	2.1	12.0	2.2	1.0
	1998	12.0	2.3	10.8	2.5	1.2
	1999	13.6	1.3	12.3	2.0	1.3

Table 3. Number of hours that stream temperatures were recorded within the optimum range for rainbow trout growth (15.5-17.3 C; Hokanson et al. 1977) during 1998 and 1999. The mean time is the average of the US and DS times.

Stream	US Site (hrs)		DS Site (hrs)		Mean (hrs)	
	1998	1999	1998	1999	1998	1999
118/16	442	338	518	275	480	307
118/48	633	473	504	288	569	381
HIP	357	108	205	34	281	71

Fig. 1. Map showing the general location of the four case study streams monitored from 1997-1999 and the two additional streams monitored in the summer of 1999.

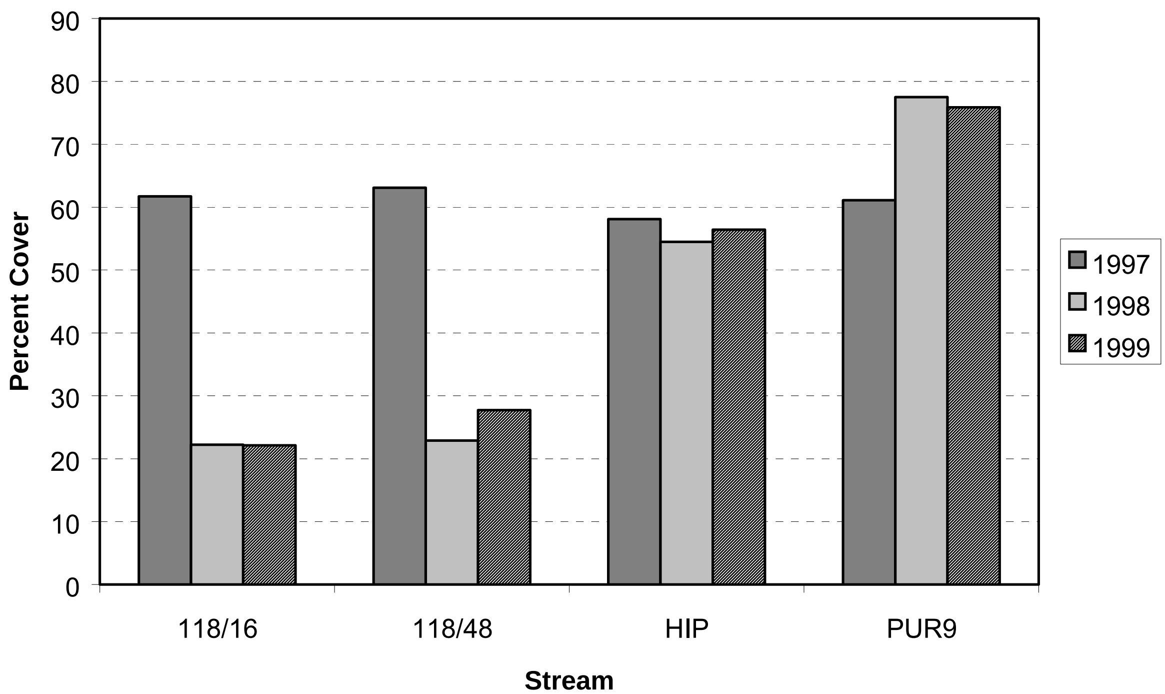


Fig. 2. Percent canopy cover for the two treatment (118/16 and 118/48) and two control (HIP and PUR 9) streams during pre (1997) and post (1998/1999) harvesting periods.

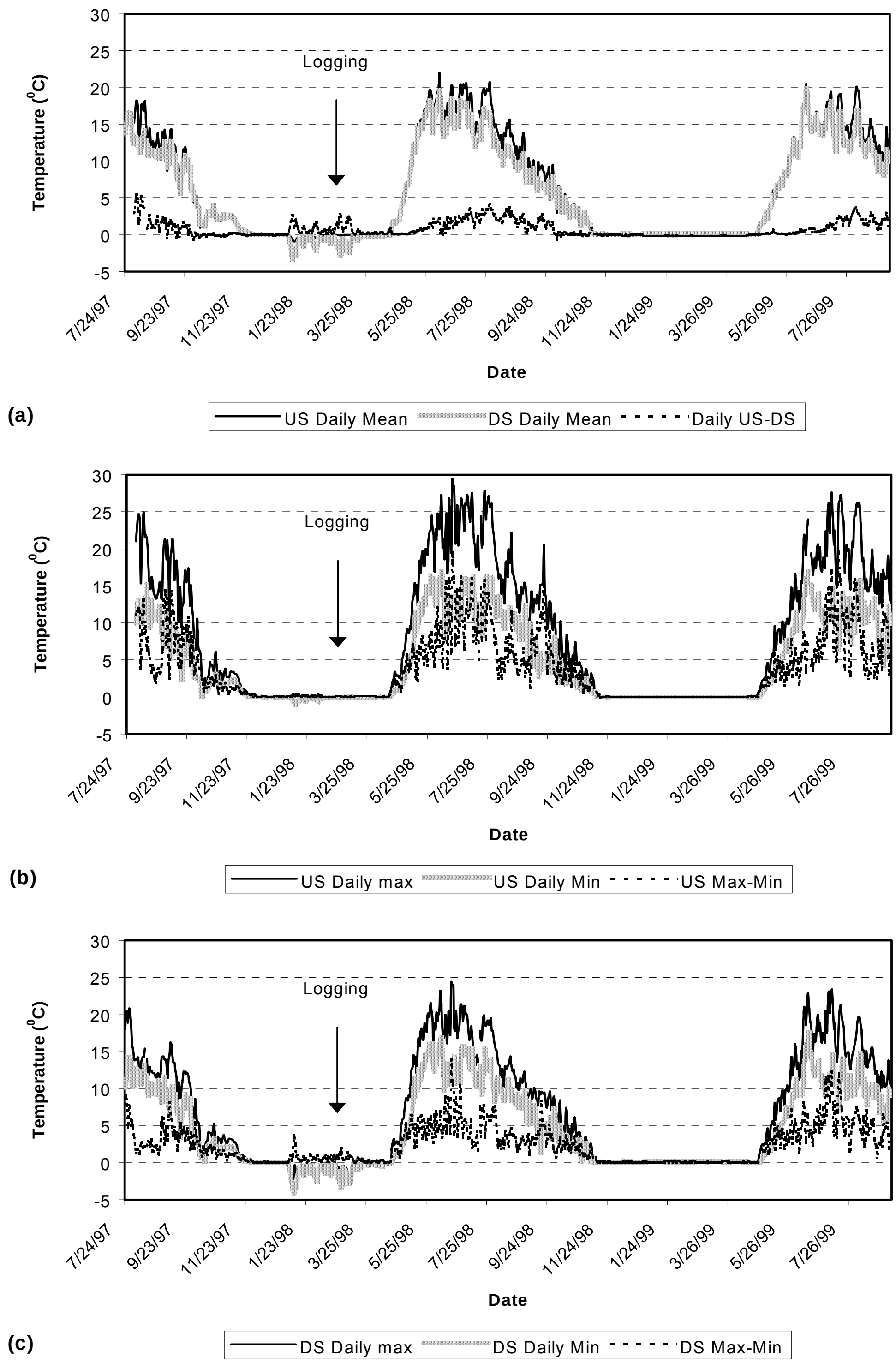
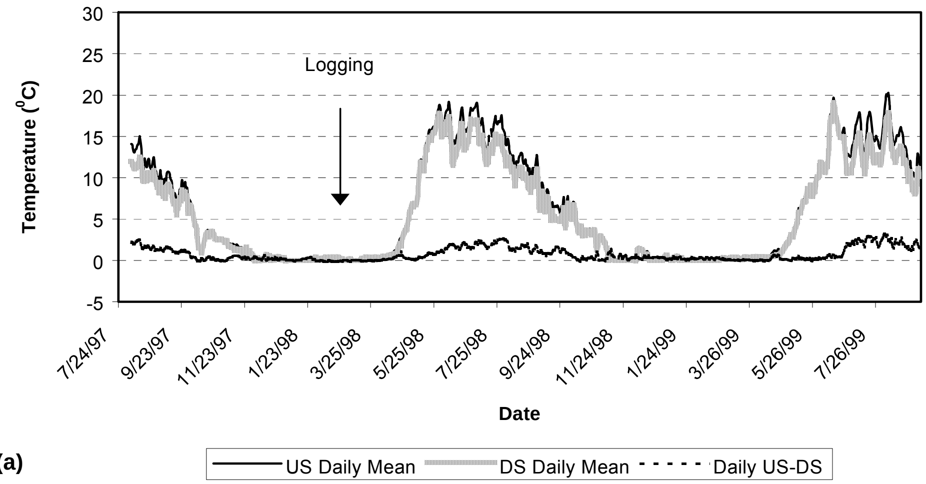
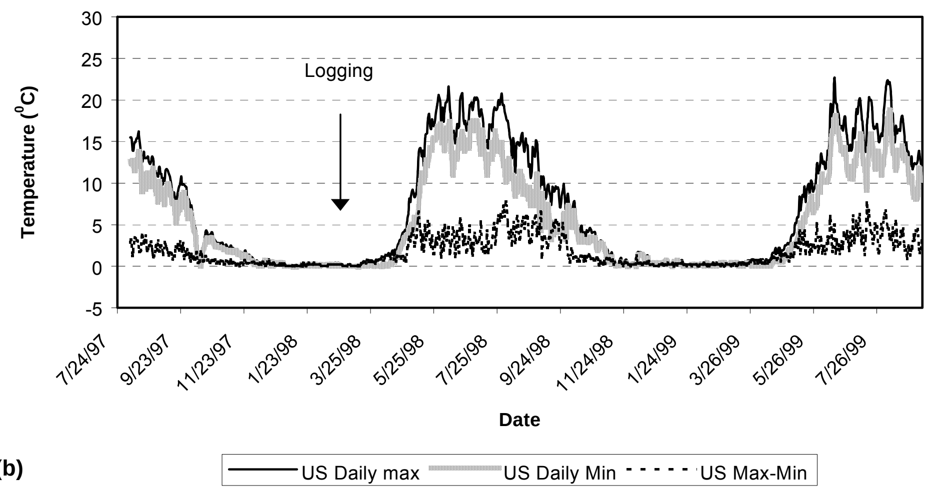


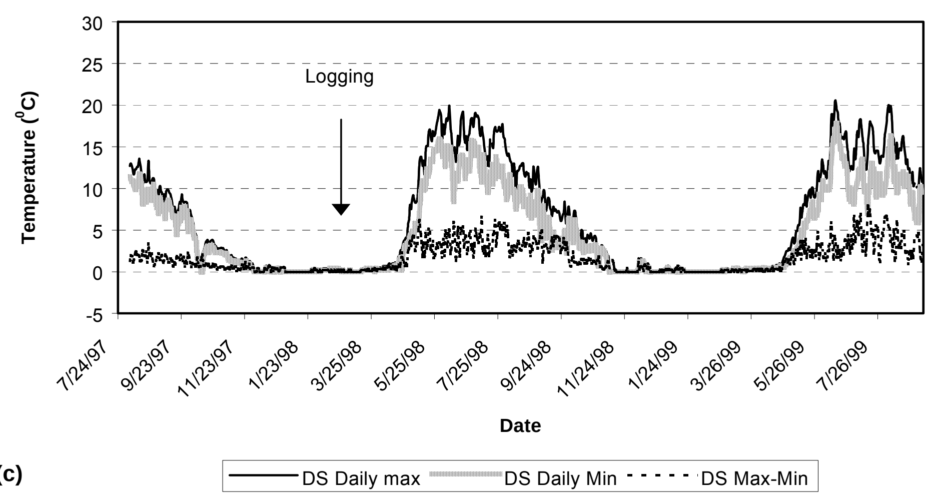
Fig. 3. Daily mean, maximum, and minimum temperatures recorded at the US and DS sites of treatment stream 118/16 during 1997-1999. The difference between US and DS sites, as well as the fluctuations (maximum-minimum) at each site, are also plotted. We suspect the negative values recordered at the DS site in the winter of 1997/98 are due to the automated logger becoming encased in ice.



(a)

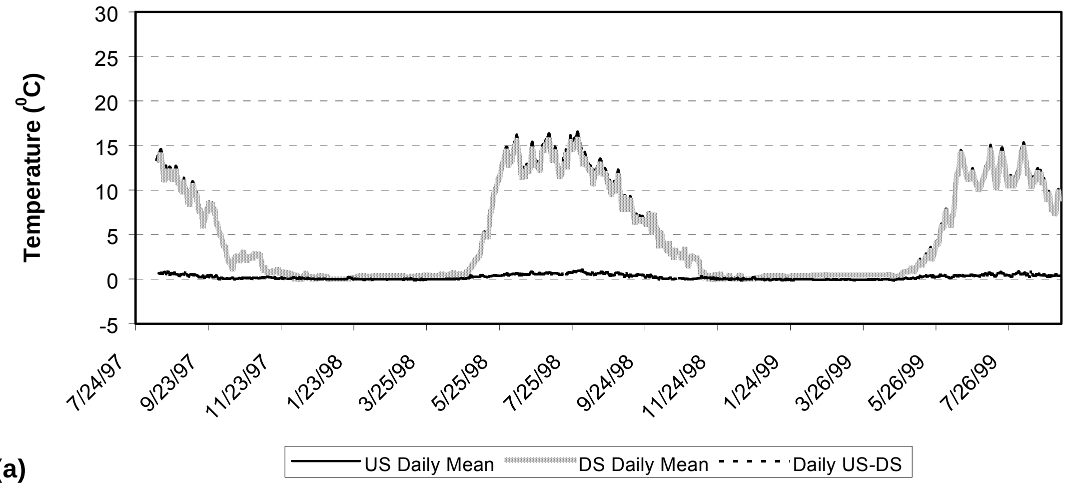


(b)

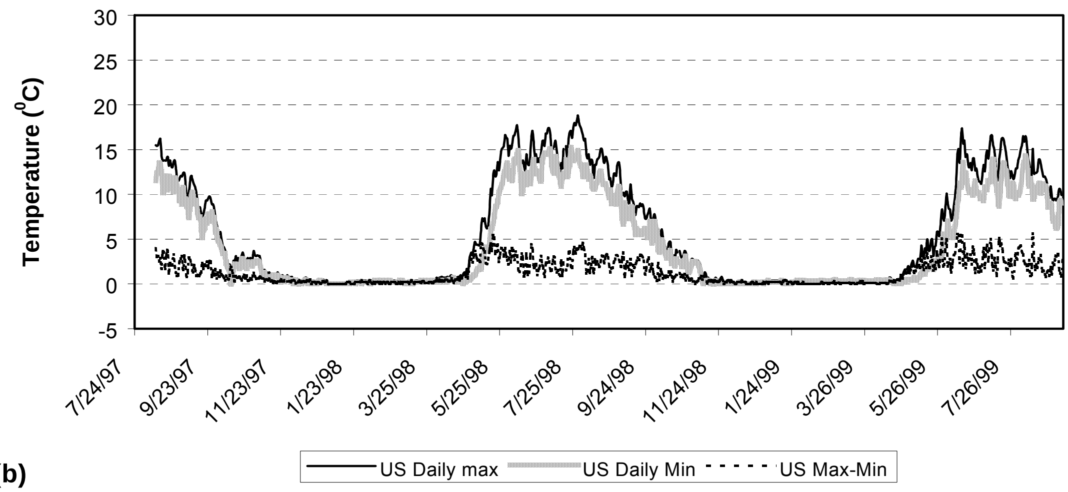


(c)

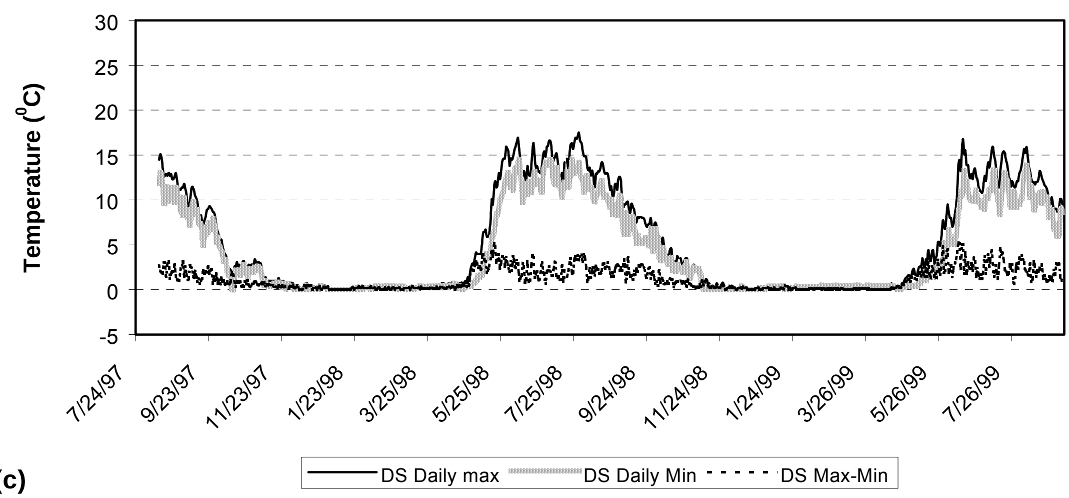
Fig. 4. Daily mean, maximum, and minimum temperatures recorded at the US and DS sites of treatment stream 118/48 during 1997-1999. The difference between US and DS sites, as well as the fluctuations (maximum-minimum) at each site, are also plotted.



(a)



(b)



(c)

Fig. 5. Daily mean, maximum, and minimum temperatures recorded at the US and DS sites of control stream HIP during 1997-1999. The difference between US and DS sites, as well as the fluctuations (maximum-minimum) at each site, are also plotted.

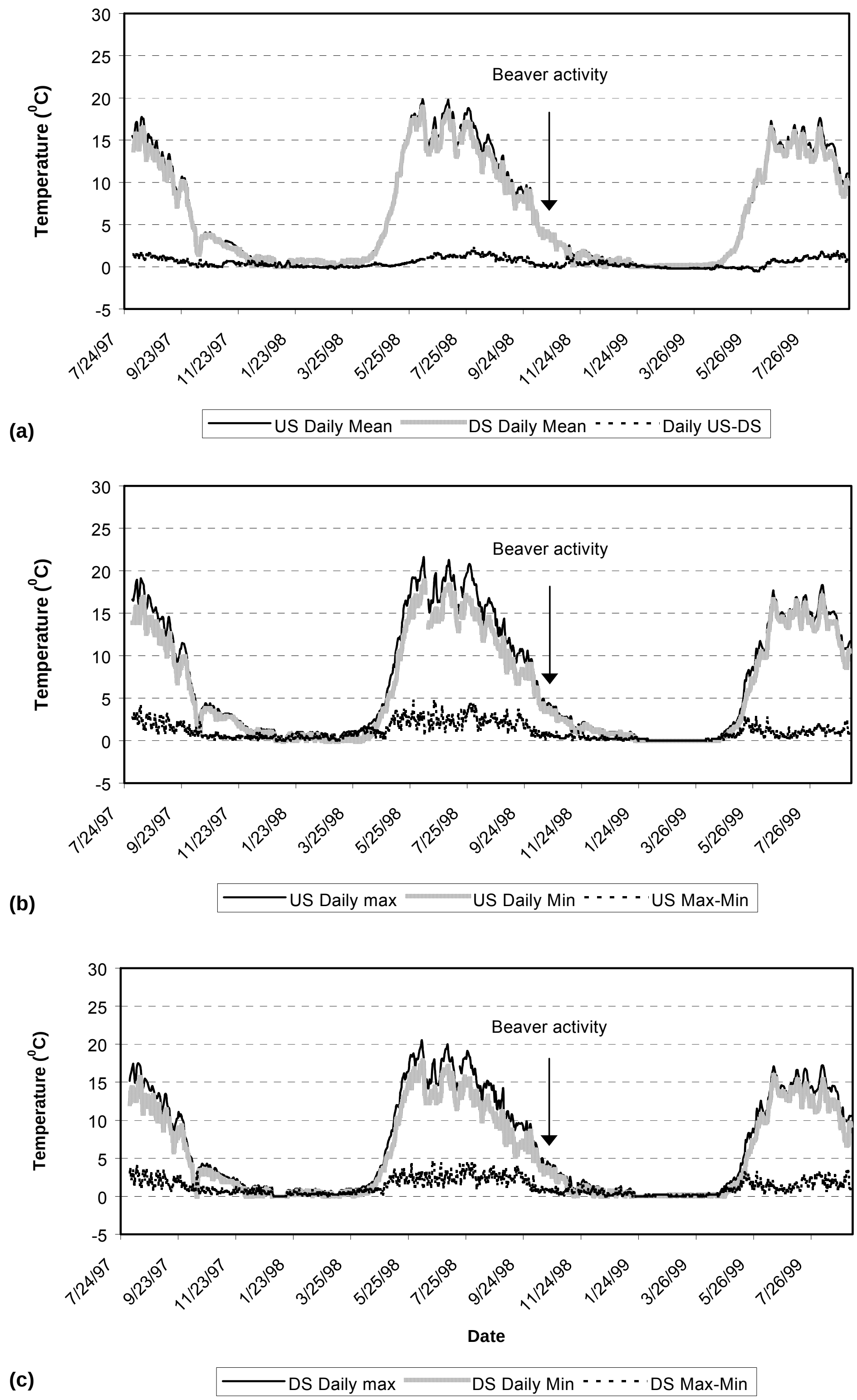
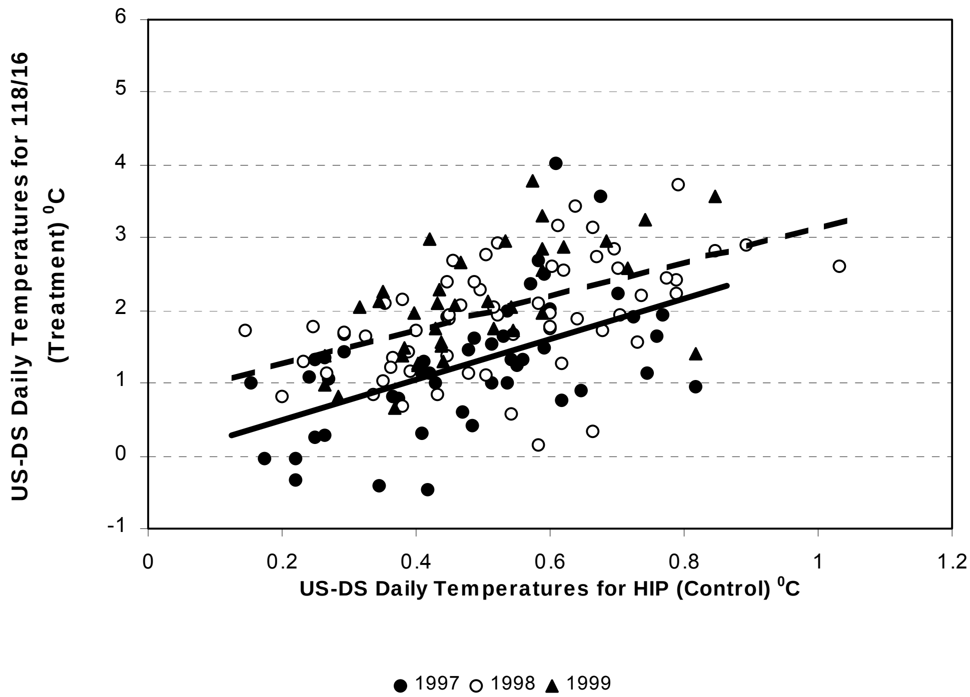


Fig. 6. Daily mean, maximum, and minimum temperatures recorded at the US and DS sites of control stream PUR 9 during 1997-1999. The difference between US and DS sites, as well as the fluctuations (maximum-minimum) at each site, are also plotted. The onset of beaver activities that resulted in artificial flows throughout 1999 is indicated.



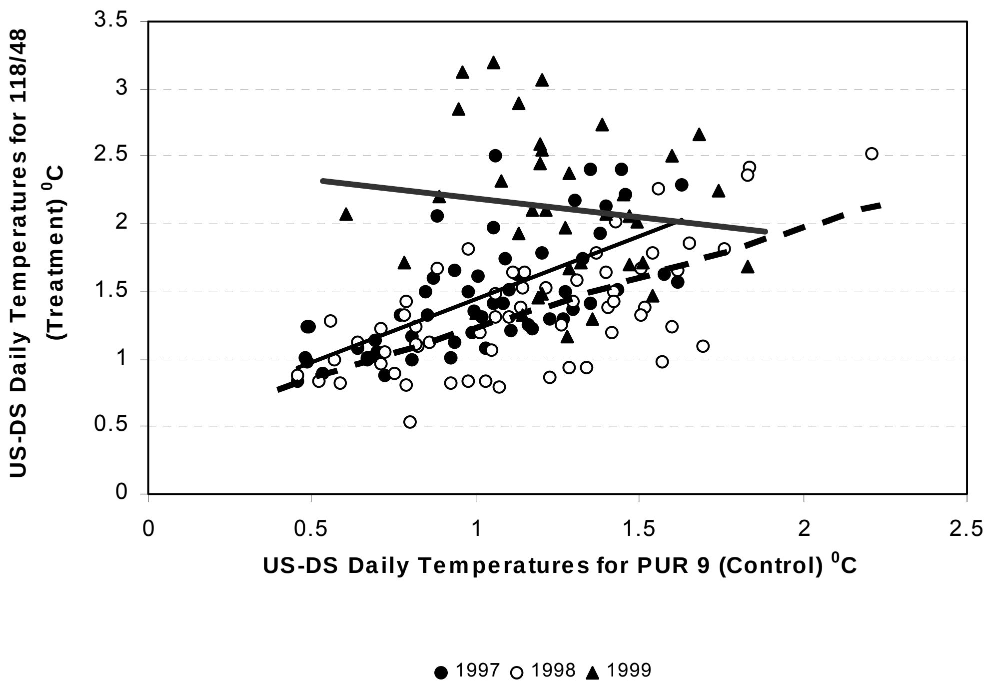
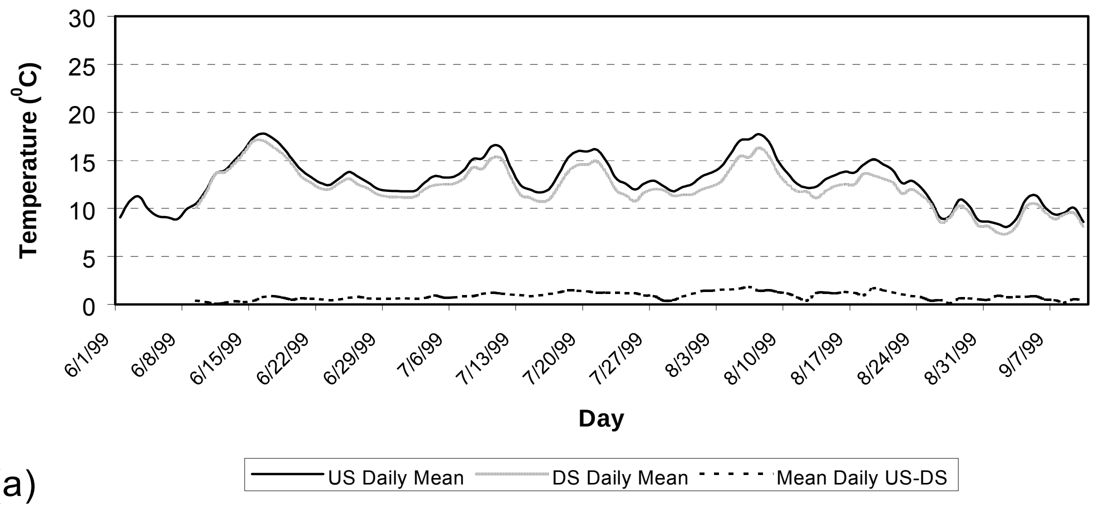
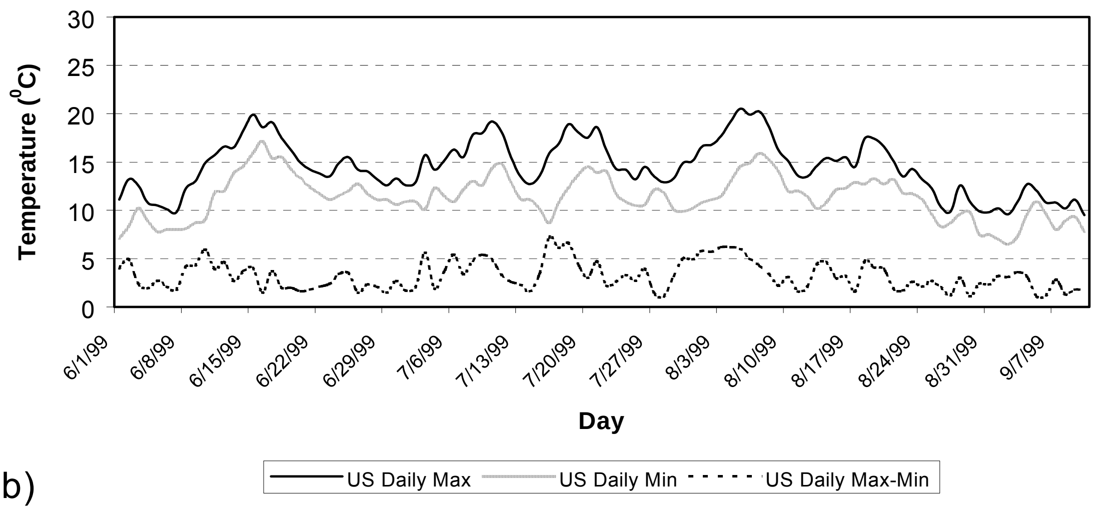


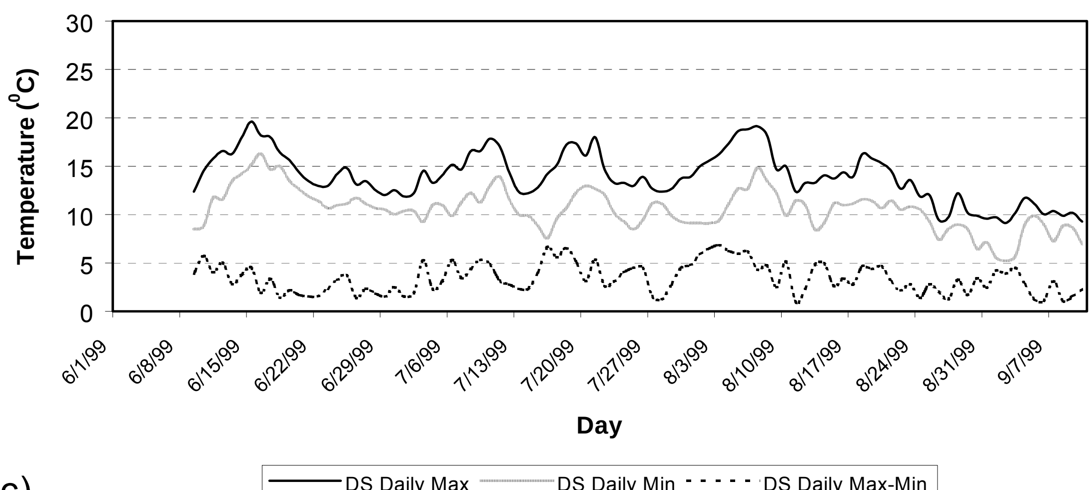
Fig.7. Relationship between the difference in US and DS temperatures for control streams and their paired treatment streams for the period Aug. 01 to Sept. 30 (see ANCOVA results in text). A) HIP and 118/16. Regression lines for 1998 and 1999 were coincident and are therfore represented by a single line. B) PUR 9 and 118/48. Due to beaver acitvity, the 1999 data were omitted from the ANCOVA (see text). Note the difference in ordinate and abscissa scales.



(a)



(b)



(c)

Fig. 8. Daily mean, maximum, and minimum temperatures at the US and DS sites for lake-headed stream 727. Monitoring was only conducted during the summer of 1999.

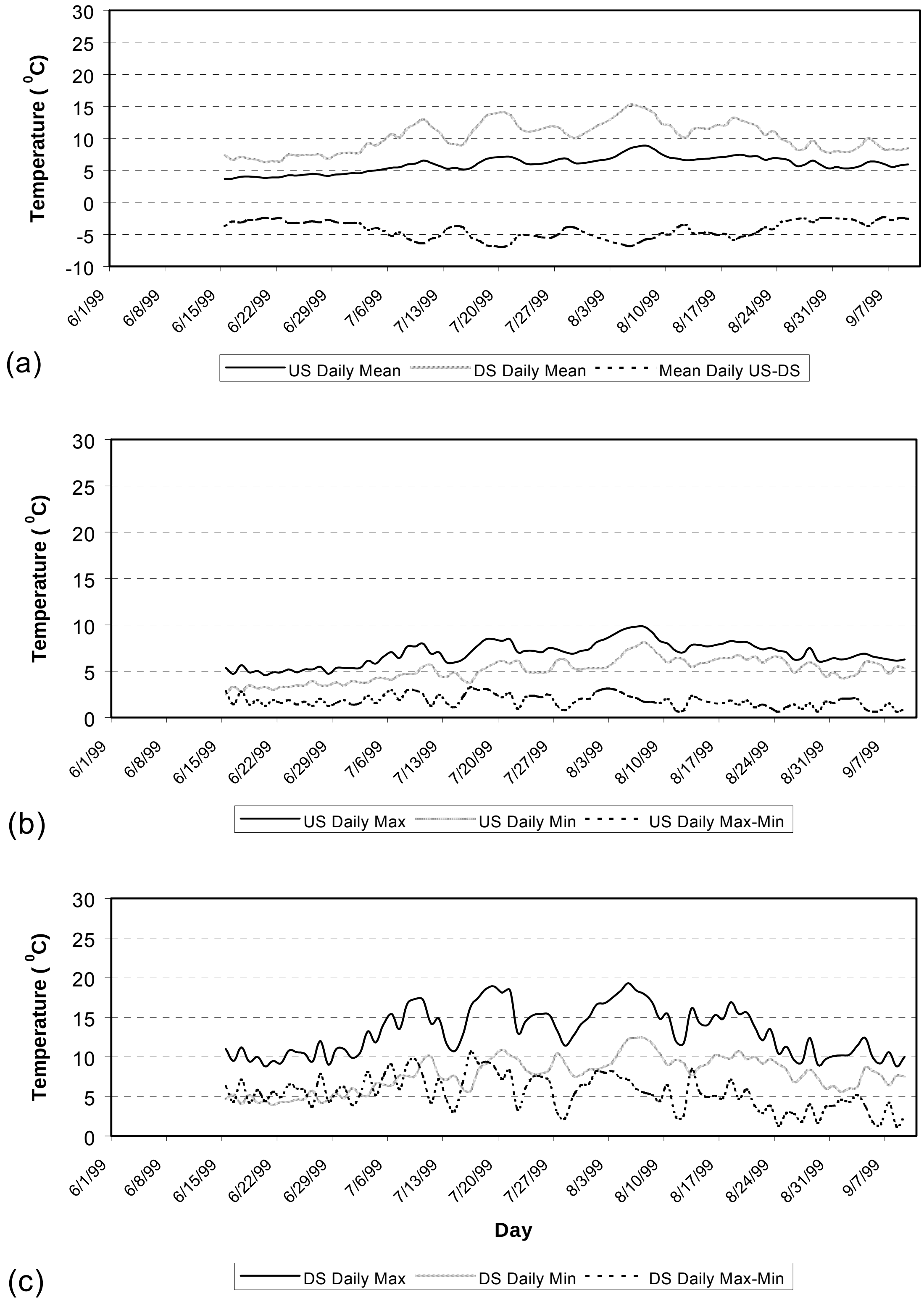


Fig. 9. Daily mean, maximum, and minimum temperatures at the US and DS sites for headwater stream TCH CC. Monitoring was only conducted during the summer of 1999. Note how in panel (a) the difference between US and DS sites is negative, indicating warming as the stream travels through the cutblock.

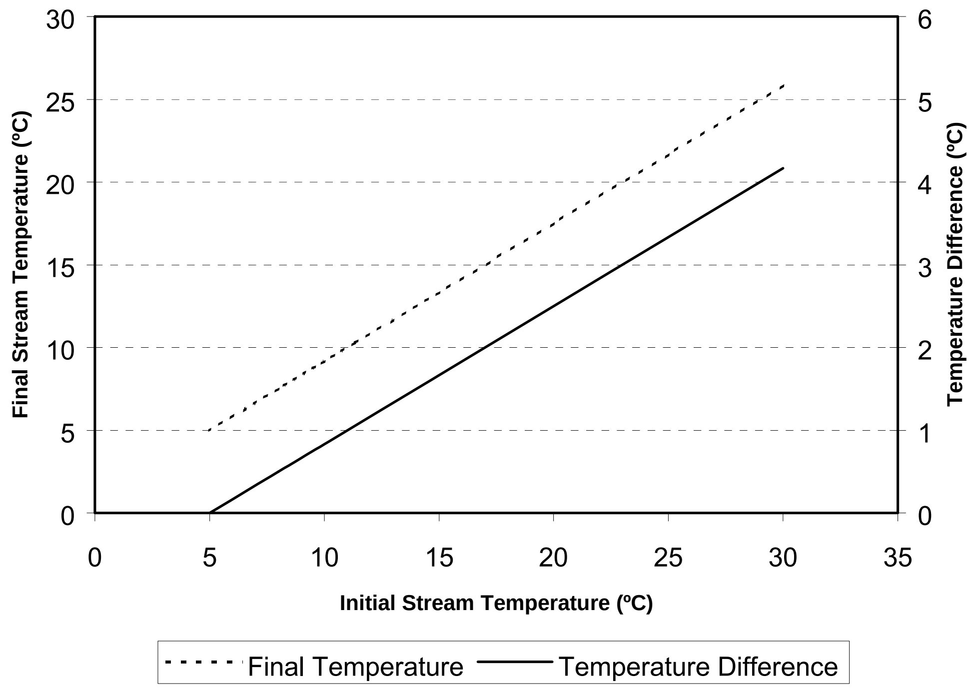


Fig. 10. Hypothetical stream temperature response to the addition of a constant volume of 5 degree Celsius groundwater as a function of the initial stream temperature. Note how the magnitude of the cooling increases as the initial stream temperature increases. A standard thermodynamic equation (equation 3.8 in Brown 1985) was used to generate the relationships (see text).

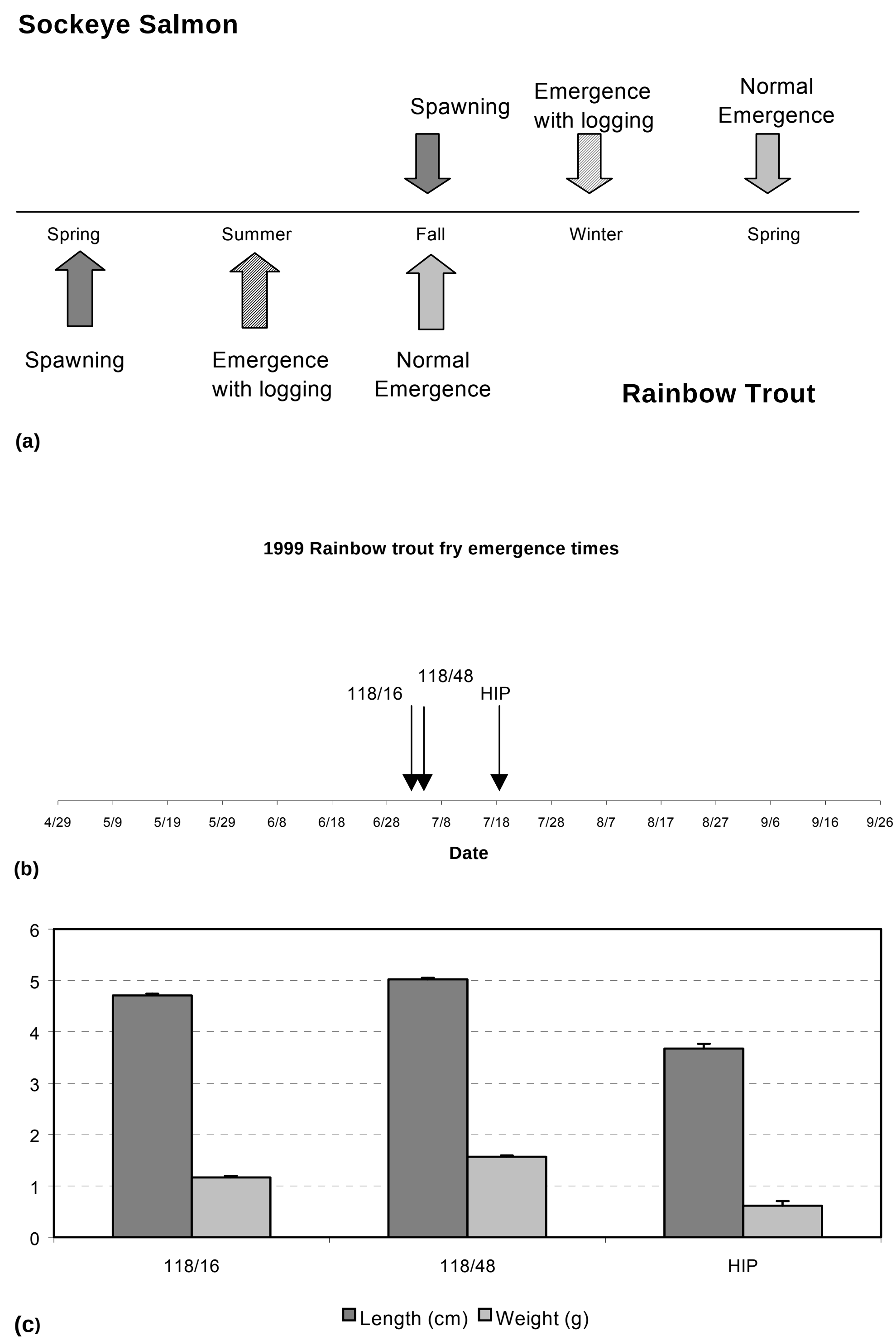


Fig. 11. Effect of increased stream temperatures on fry emergence. (A) Differences between sockeye salmon and rainbow trout. Logging is assumed to result in an increase in stream temperatures. (B) Observed emergence times of rainbow trout fry in the two treatment streams (118/16 and 118/48) and one control stream (HIP) during 1999. The time difference between the treatment and control streams is 2 weeks. (C) Observed lengths and weights of rainbow trout fry by mid-September 1999. Error bars represent standard errors of the mean. Sample sizes are as follows: 118/16 (252 fish); 118/48 (692 fish); HIP (95 fish).

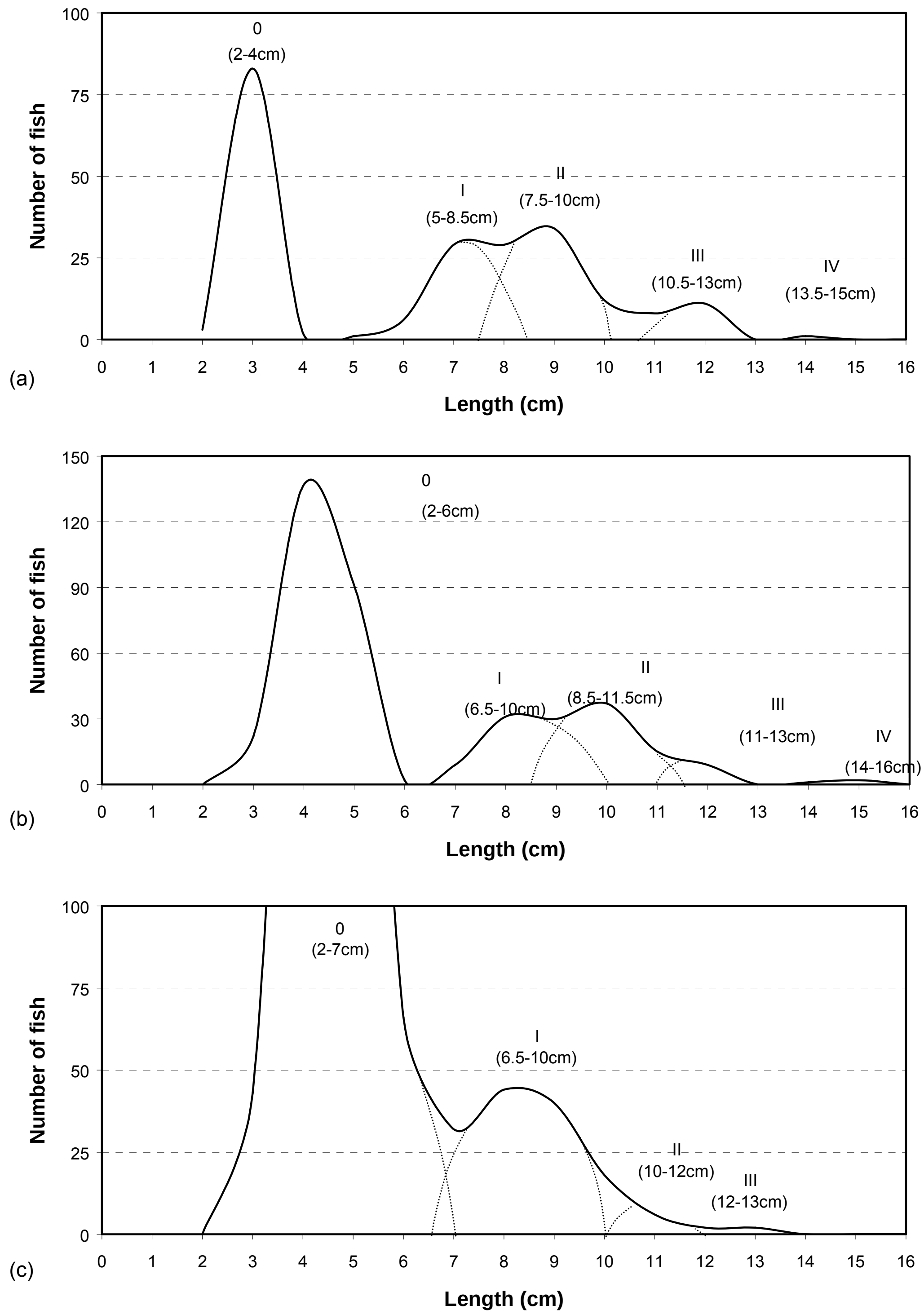


Fig. 12. Length-frequency distributions for rainbow trout in our case study streams. (A) Control stream HIP. (B) Treatment stream 118/16. (C) Treatment stream 118/48. The ordinate scale in panel C has been truncated so that the distribution of fry (age 0) does not obscure the distributions of older fish. The distribution of fry peaks at 299 fish, corresponding to a size of 5 cm. Roman numerals correspond to our estimated age classes. The dotted lines represent our estimates of the size class boundaries (given in parentheses), and these were corroborated by age analyses conducted using otoliths.

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Influence of Logging Road Right-Of-Way Size on Small Stream Water Temperature and Sediment Infiltration in the Interior of B.C.

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INTRODUCTION

Logging roads and their associated stream crossings have long been identified as sources of impacts to streams and fish habitat (Burns 1972; Beschta 1978; Everest et al. 1987; Harper and Quigley 2000). Historically sediment generation has been a prime concern related to road use and maintenance (Reid and Dunne 1984; Nistor and Rood 1999), road type (Luce and Black 1999), and road density (Cederholm et al. 1981). Stream temperature fluxes associated with clearcuts are well documented and have been attributed primarily to riparian canopy removal (Brown and Krygier 1971; Beschta et al. 1987; McGurk 1989). However, little work has focussed specifically on the thermal impacts of roads and their associated structures (e.g. ditches and crossings) on streams. Neither stream sediments nor temperature impacts have been correlated with road right-of-way width.

Road right-of-ways are continuous linear areas cleared for road construction. Depending on the geography of an area (e.g. soil type and slope) they are generally several times wider than the road itself. Factors such as visibility (for safety), snow removal, and slash disposal are also taken into account when determining road right-of-way widths in the interior of B.C. (Anonymous 1995). The riparian area of a stream encompassed by a road right-of-way is generally severely disturbed by both bank alteration (e.g. culverts and bridge abutments) and vegetation removal, providing opportunity for both sediment intrusion and thermal impact to occur. Although not well documented, road right-of-way width may be an important factor regulating sediment generation and thermal impact at small streams crossings.

As a component of the on-going Stuart-Takla Fish-Forestry Interaction Project (STFFIP) we had an opportunity to investigate the effect of road right-of-way width on sediment generation and thermal impact at four culvert stream crossings. Surface water temperature, interstitial sediment deposition, and groundwater characteristics (level, chemistry, and temperature) were monitored on either side of each stream crossing. Surface water temperature and interstitial sediment deposition results are presented here.

MATERIALS AND METHODS

Study Area

The Baptiste study watershed is part of the STFFIP, full descriptions of the project are provided in Macdonald (1994), and Macdonald and Herunter (1998). Four stream crossings on three creeks (B3, B4 and B5, Fig. 1) were investigated. The streams are small groundwater fed head waters which are hydrologically dominated by a spring snowmelt event followed by a stable base flow with occasional peaks in the late summer due to storm events. Average wetted widths are about 1 m with peak flows ranging from 0.03-0.05 m^3/s and base flows of about 0.01 m^3/s. Complete physical details of the creeks including discharge and suspended sediment loadings are presented in Beaudry (2001). Collet and Ryder (1997) describe the surficial geology in the study area primarily as basal till.

Rainbow trout spawn and rear 100 m downstream of the study sites. Spawning occurs in early June and fry are present well into September. The streams are classified under the *Forest Practices Code of B.C. Act* as S6 (non-fish bearing; < 3 m width) above the road and are S4 (fish bearing; < 1.5 m width) below the road.

Experimental Design

The construction of a new 8-m wide logging road in the winter of 1996 allowed us to examine the influence of road right-of-way width on stream temperature and sediment deposition. This road, along with an existing “old road” incorporated into the study, had a low gradient and was constructed using cut and fill methods. Both roads were used extensively during harvesting between the winter of 1996 and the fall of 1997. During construction 50 m, 30 m, and 20 m road right-of-ways were left on B3, B5, and B4, respectively. A 50 m right-of-way exists on B3 as a result of the old road (Fig. 1, Table 1).

Table 1. Right-of-way width at the study stream crossings. Stream temperature and sediment monitoring stations were established on either side of each crossing.

Stream	Road Age	Right-of-way width (m)
B3	Old	50
B3	New	50
B5	New	30
B4	New	20

Road right-of-way width for harvesting operations in the area is typically 30 m which allows room for burying of slash and opens the road prism for drying (personal communication Greg Pearson, Canfor, Fort St. James). Depending on slope and soil texture this can range up to 150 m (Anonymous 1995).

During snowmelt in the spring of 1997 stream temperature and sediment monitoring stations were established on either side of each crossing, so that the upstream stations were out of right-of-way influences and downstream stations captured all of the right-of-way influences (e.g. road run-off from ditches). The B3 old road station was comprised of the new road upstream station (B3AR) and a previously established station at the bottom of a cut block (B3Lo). All of the crossings were built using corrugated metal pipe (CMP) culverts approximately 16 m in length and 1 m in diameter. The B3 and B5 watersheds both had 60 ha cut blocks harvested upstream of the study sites at the time the road was built. The B4 watershed remained as a non-harvested control.

Sediment Traps

Three samplers were placed in the thalweg at each station in riffle/glide habitat. Grab samples of road bed adjacent to two of the stream crossings (B3 and B5) were taken for size class analysis.

Samplers consisted of a 2 litre (135-mm diameter, 150-mm high) tub filled completely with clean sieved spawning size gravel (12-40 mm) (Larkin and Slaney 1996). It became apparent during the first field season that the sample volume of this trap sampler was insufficient so a larger sampler was developed combining the original design with that of Fletcher (1995). A 2 litre tub was used with an internal Plexiglas frame (110-mm high) which supported a 10-mm mesh. A 40 mm layer of clean spawning size gravel was placed on top of the mesh, level with the rim of the tub. This resulted in a sampler that had a “filter” of gravel on top and a compartment below, in which depositional fines were collected. This version of sampler was installed at all of the sampling stations in May 1998 and had a much larger sample capacity.

Both trap types used were designed to sample fine depositional sediments that infiltrate spawning gravel. Results are not comparable to sediment budgets produced from suspended load or total bedload samples. The 40 mm layer of spawning gravel and the 10 mm mesh on top of the sample chamber restrict entry of large size fractions into the sampler. Interstitial deposition may not be proportional to total deposition as the pores within the “filter” can become bridged and sediment can then overlay the sampler without being collected below the gravel “filter”. Actual fine sediment infiltration is likely underestimated because sediment that moves laterally through the streambed is not sampled.

All sediment traps were buried flush with the stream substrate. Samplers were recovered and reset generally in May or June, July, August, and September of 1997, 1998, and 1999. Upon recovery any scour or overlying sediment was noted and removed and discarded to locate and withdraw the samplers. Samples from each station and sampling date were pooled, weighed and dry sieved to 9.5, 5.6, 2.36, 1.18, 0.3, 0.075, and 0.0 (mm) size fractions at the DFO Cultus Lake Laboratory. Weighing and size fraction analysis was performed on approximately 45% of the samples from a station to provide an indication of within station variability. When three samples from a station were processed individually, standard deviations and coefficient of variation (CV) were calculated and applied to the annual data set for that particular station.

All samples were truncated at 9.5 mm for analysis as the sampling chamber screen only allowed particles less than 10 mm to enter the sampler. Total annual deposition rate was calculated by summing the sample weights from each year and standardizing them to a 12 month deployment period. Geometric mean diameter (Dg) was calculated for each sample set as:

$$Dg = d_1^{w1} \bullet d_2^{w2} \bullet \underline{\quad\quad} \bullet d_n^{wn},$$

where d = the mean diameter between two adjacent sieve sizes and w = the proportion of the layer retained by the smaller sieve. Average annual Dg was calculated by taking the average of each sample set for a given year. Approximately 10% of the samples were sieved three times to calculate laboratory sieving error. The Dg standard deviations ranged from 2 to 7% of sample average, so sieving error was considered negligible.

Temperature

Temperature data were recorded using Vemco minilogger dataloggers (precision +/- 0.2^{o}C) at each monitoring station. A datalogger was located on the streambed in the thalweg of the stream in riffle/glide habitat. Data downloads were performed up to 4 times per year. Accuracy of the loggers was checked on site with a calibrated mercury thermometer during each download. The loggers record an instantaneous reading on an hourly basis. Creek temperature data for the stream crossing study began in May 1997, after the road was established and has been recorded continuously since then, the most recent download occurring in September 1999.

Temperature records of less than zero occurred at some stations during the winter as the sensors were exposed to ice or air. These temperatures were assumed to be zero. Stream temperatures above and below each crossing were compared (ΔT^{o} = downstream temperature – upstream temperature) to detect cooling ($-\Delta T^{o}$) or warming ($+\Delta T^{o}$) trends. Cumulative temperature differences were calculated by adding daily average ΔT^{o} for spring/summer (May 1 to September 30) and fall/winter periods (October 1 to April 31).

Summer rainfall measurements were recorded at the Middle River DFO camp meteorological station (approximately 50 Km from the study site). Continuous air temperature was attained by combining DFO camp meteorological station data with data from an adjacent monitoring site. Specific equipment and methods are described in Andersen (1997).

RESULTS

Sediment traps

Samplers were occasionally buried by bedload following freshet events, particularly in the spring, but samples were recovered in most cases. During the May 1998 freshet, the entire set of station B3 Above-Road samplers was lost. The 1997 analyses do not include over winter sampling (September 1996 to April 1997) because the samplers were set in May 1997. While no major slope failures occurred in any of the watersheds upstream of our sampler locations, some upstream sediment sources were identified during the study. Bank destabilization occurred in 1997 as a result of riparian windthrow in the B3 cutblock and as a result of a skidder crossing culvert removal in the B5 cutblock. No vegetation re-growth occurred in any of the riparian areas within the right-of-ways, resulting in no temporal increase in bank stabilization or stream cover.

Twelve month interstitial sediment deposition rates (ending in September each year) in the samplers ranged from about 1000g/year to 4000 g/year (Fig. 2). When sample variability is taken into account there is no indication of a difference between upstream and downstream stations.

Average geometric mean diameter deposition within the samplers was about 1 mm (Fig. 3). Again there was little difference between upstream downstream stations, with the exception of B4 which had consistently coarser material deposited in the downstream samplers. Road bed samples had Dg's of 1.17 (B5) and 1.24 (B3) mm.

Temperature

Annual average daily creek temperature below the stream crossings ranged from 0.0 ^oC to 13.0 ^oC (B3 old road), 14.3 ^oC (B3 new road), 13.2 ^oC (B5), and 12.7 ^oC (B4) (Fig. 4). The non-harvested watershed and smallest right-of-way, B4, tended to be slightly cooler than the other creeks.

During the spring and summer (May-September), stream temperatures increased as they passed through the road right-of-ways, but showed no change or declined slightly in the winter months (Fig. 5). Wider right-of-ways have a greater influence on stream temperature. Increases of 1.0 ^oC (maximum 2.2 ^oC) were not uncommon with the 50 m widths. Smaller increases of 0.5 and 0.2 ^oC (maximums of 1.5 and 0.4 ^oC) were detected at the 30 and 20 m right-of-ways, respectively. Temperature differences recorded at the narrowest right-of-way were within the cumulative precision of the dataloggers. Ditch flow generally only occurred during the freshet/snowmelt event (April) when the creeks were less than 5 ^oC. Thermal impact from ditch flow was considered negligible as the ΔT^o for this period was small.

During the summer, much larger cumulative ΔT^o occurred at the wider right-of-ways (Fig. 6). The 50 m right-of-ways attained cumulative ΔT's ranging inter-annually from 70–140, the 30 m ranged from 17–40, and the 20 m ranged from 0-17. The greatest temperature accumulations occurred during the summer of 1998 although it was neither the warmest nor the wettest of the three summers (Table 2).

Table 2. Cumulative rainfall and air temperature from the DFO Middle River Camp meteorological station for the period June 1 to September 13 each year.

Year	**1997**	**1998**	**1999**
Cumulative rainfall (mm)	179.4	151.8	143.2
Cumulative daily average summer air temperature (^oC)	1160.4	1043.8	1011.8
Maximum daily average summer air temperature attained (^oC)	22.6	17.8	15.4

DISCUSSION

Sediment Traps

Both the interstitial deposition and size class analysis results from the sediment traps were unexpected. We anticipated a downstream increase in interstitial fine sediment deposition and a decrease in geometric mean diameter of the deposited sediments due to fine sediments being liberated from the road and the right-of-ways. Our findings contradict those of Spillios and Rothwell (1998) and Clarke et al. (1998) who detected an increase in fine sediment concentration downstream of stream crossings using freeze-core techniques and Wesche samplers, respectively. The Baptiste road bed samples consisted of a fairly fine distribution (Dg of 1.17 and 1.24 mm) so the right-of-ways had the potential to affect the Dg of the sediment deposited within the downstream samplers. However, the particle distributions of the road samples are consistent with that of till in the area (Collet and Ryder 1997) and its cohesive properties may restrict its erosion potential and therefore entrainment to our study streams. Bilby (1985), using freeze core techniques, was unable to detect a difference in streambed percent fines downstream of a stream crossing, however he did measure increased suspended load. Likewise, in our study, high suspended loads may have occurred as a result of the stream crossings and their right-of-ways but those particles may not have been deposited in our samplers.

Temperature

The effect of right-of-way size on stream temperature heating is more pronounced than temperature increases measured through cutblocks (Table 3). The 50 m right-of-ways had average summer heating rates exceeding those reported in interior BC cutblocks and maximum values far exceeding those found in California. Macdonald et al. (1998) and Moore et al. (2001) have suggested that groundwater intrusion and hyporheic exchange mitigate stream heating effects. Groundwater flow and exchange at the stream crossings may have been severely altered by the road bed, and CMP culverts preclude any exchange. Limited groundwater intrusion and exchange may be the reason why we observed such large temperature increases over a relatively short distance.

Table 3. Stream temperature increases through cut blocks compared to increases found in this study at stream crossings with various right-of-way widths. All values are standardized to temperature increase per 100m; averages are taken for the summer period (May 1 to September 30). The Baptiste B5 value is relative to a control stream.

<table>
<tr><td colspan="4">Cutblocks</td><td colspan="3">Stream Crossings</td></tr>
<tr><td>Source and Location</td><td>McGurk (1989) California</td><td>Macdonald (1998) interior B.C.</td><td>Macdonad (2001) Baptiste B5</td><td>50 m right-of-way</td><td>30 m right-of-way</td><td>20 m right-of-way</td></tr>
<tr><td rowspan="2">^{o} C rise/ 100m</td><td rowspan="2">1.5</td><td rowspan="2">0.6</td><td rowspan="2">0.4</td><td>Max: 4.4</td><td>5.0</td><td>2.0</td></tr>
<tr><td>Avg: 1.3</td><td>0.6</td><td>0.2</td></tr>
</table>

The stream "heating" experienced at the B3 and B5 crossings is in addition to elevated temperatures attained as they both flow through cut blocks prior to flowing through the crossings (see Macdonald et al. 2001). The B3 new road crossing exhibited temperature increases after the stream flowed through both the cut block and the old road stream crossing. This suggests that the heating effect is cumulative and that several stream crossings could markedly elevate stream temperatures.

There was little ditch flow at any time of year other than during the snowmelt event thus changes in temperature are attributed specifically to the stream crossings. As all of the culverts were the same size, right-of-way width was the determining factor in the amount of heating which took place. Riparian vegetation re-growth within the right-of-ways was negligible, even 2.5 years after the initial disturbance, leaving the streams exposed to solar radiation. The onset of positive ΔT's corresponds to increases in summer air temperature indicating that the mechanism of heating was largely direct solar exposure.

The timing of temperature increases corresponds to the presence of rainbow trout spawning immediately downstream of the study site. Anthropogenically induced thermal fluctuations in streams can have implications for early life stages of salmonids (Macdonald et al. 1998) and the production of invertebrates they consume (Noel et al. 1986). Although daily average temperatures were well within the tolerance limits of rainbow trout, the large changes in cumulative ΔT could have significant effects on their incubation and growth. The effect may be positive or negative depending on biological circumstances within a geographical area. In this case, the effect may be reduced as there is a suggestion of spatial thermal recovery downstream of the study site (see Moore et al. this Compendium) but it may be more pronounced in areas lacking thermal recovery.

Management Implications

The effect of road right-of-way width on stream temperature on these particular systems is clear; the larger the right-of-way the greater the thermal impact across it.

This effect appears to be additive, so several crossings on a single stream could lead to a large increase in downstream temperatures.

Impacts from stream crossings are thought to occur primarily during crossing installation and removal; guidelines and best management practices mitigate these acute impacts by requiring crossing installation and removal during fisheries sensitive time windows (e.g. at times when fish or their eggs are not spawning, incubating, migrating or rearing in a particular area). However results from this study indicate chronic thermal effects do occur. Decreasing the right-of-way width at stream crossings to a minimum (while incorporating necessary safety factors) would be beneficial from a temperature regulation perspective. This would also, in turn, protect more riparian area leading to an overall reduction in stream bank disturbance and a decrease in sediment intrusion opportunity.

ACKNOWLEDGEMENTS

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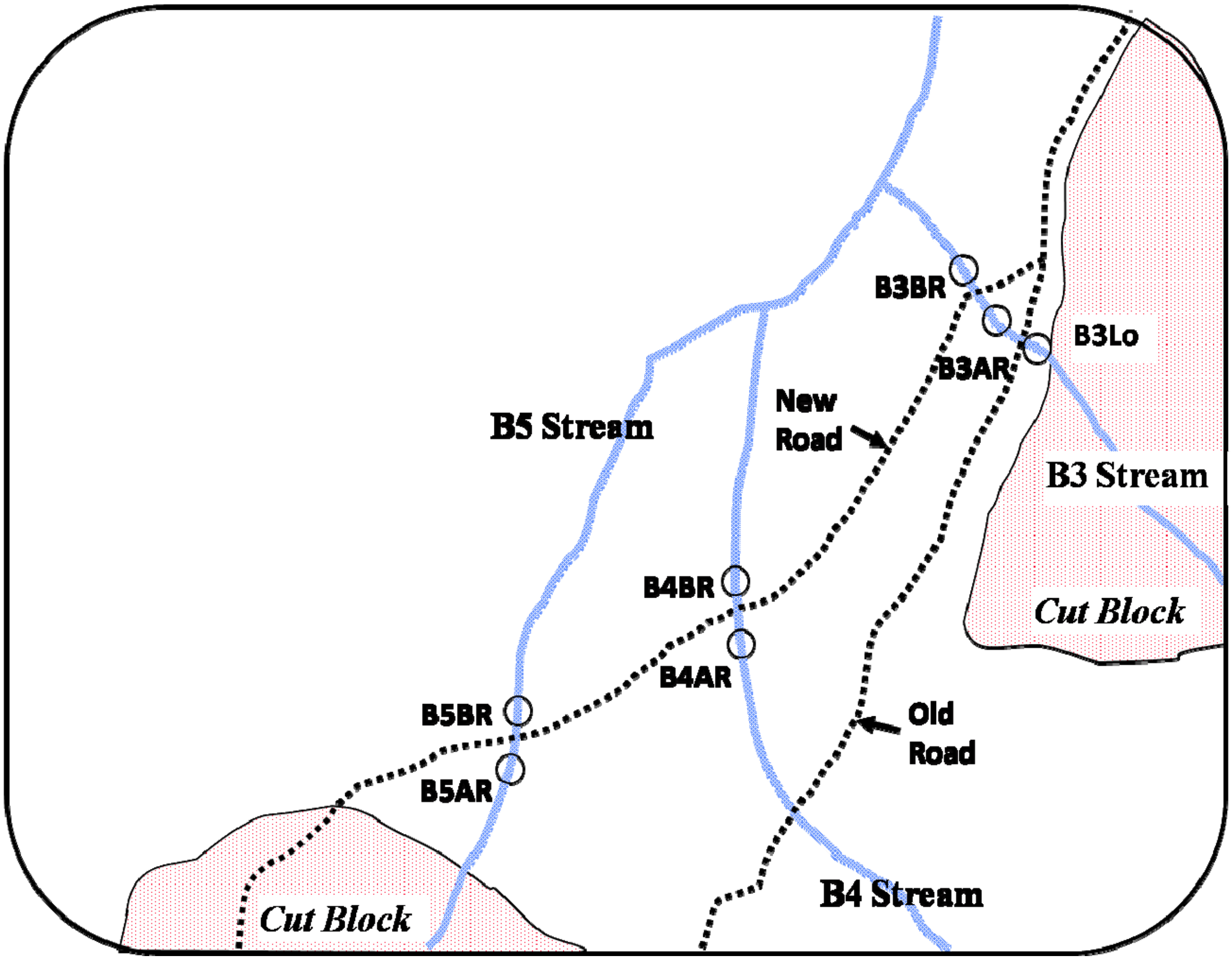

Fig. 1. Road Right-of-way width and stream crossing study station locations. Stations on creeks B3 and B5 are located downstream of cutblocks, creek B4 is a nonharvested control. AR=Above Road, BR= Below Road, and Lo= below cutblock. Right-of-way widths were 50 m, 20 m, and 30 m on streams B3, B4, and B5, respectively.

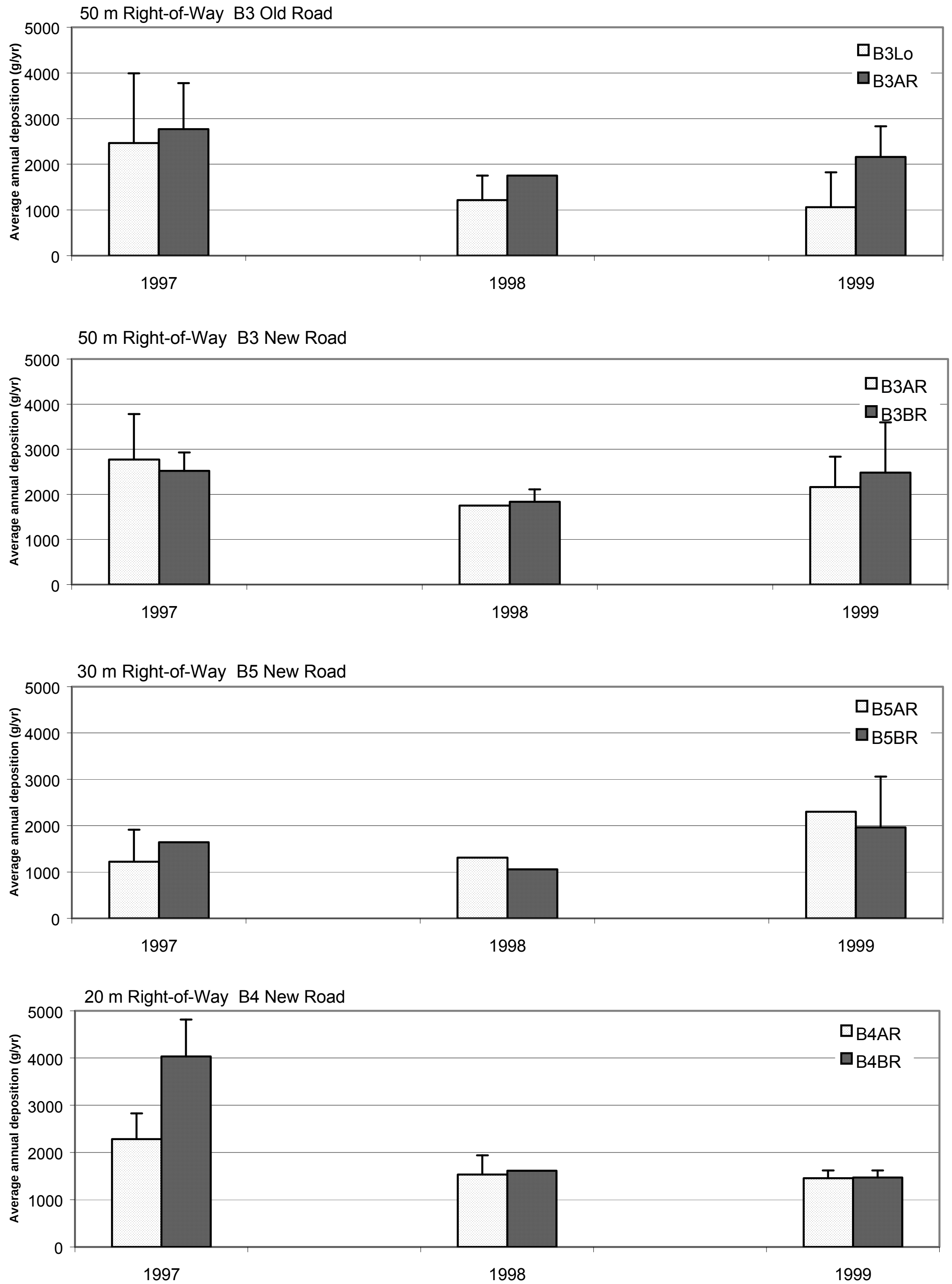

Fig. 2. Average annual fine interstitial sediment deposition in small streams above and below four stream crossings with three different road right-of-way sizes. Values were obtained by taking the average of three samplers at each station, summing the four sampler deployment periods, and standardizing deployment time to one year. Error bars are Coefficient of Variation (CV) relative to total deposition and were developed when three samples from a station were processed individually (~45% of the samples).

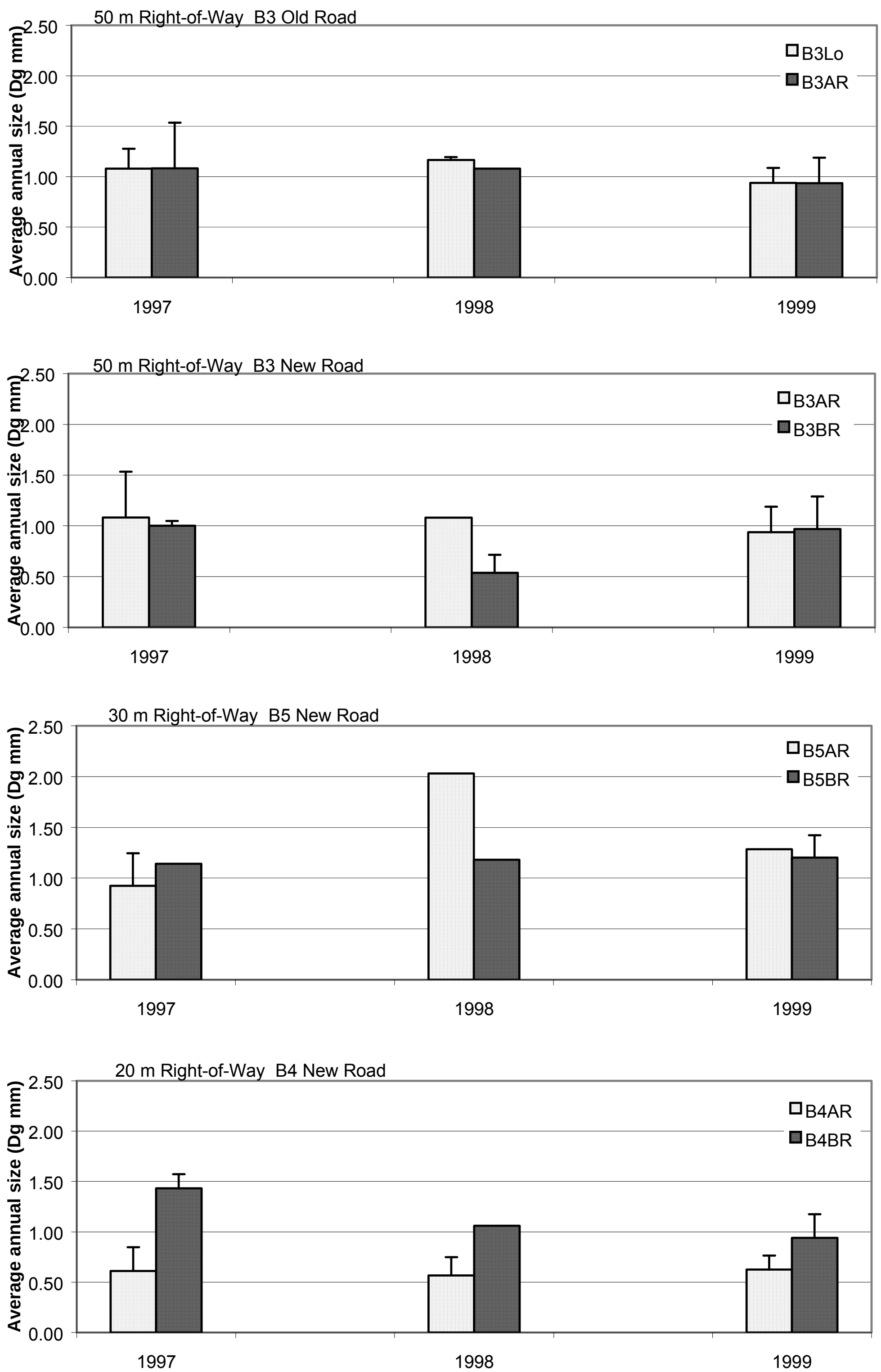

Fig. 3. Average annual sediment trap sample geometric mean diameter (Dg) of sediment deposited in small streams above and below four stream crossings with three different road right-of-way sizes. Values were obtained by taking the average of three samplers at each station, and averaging the four sampler deployment periods within one year. Error bars are one standard deviation relative to average deposition and were developed when three samples from a station were processed individually (~45% of the samples).

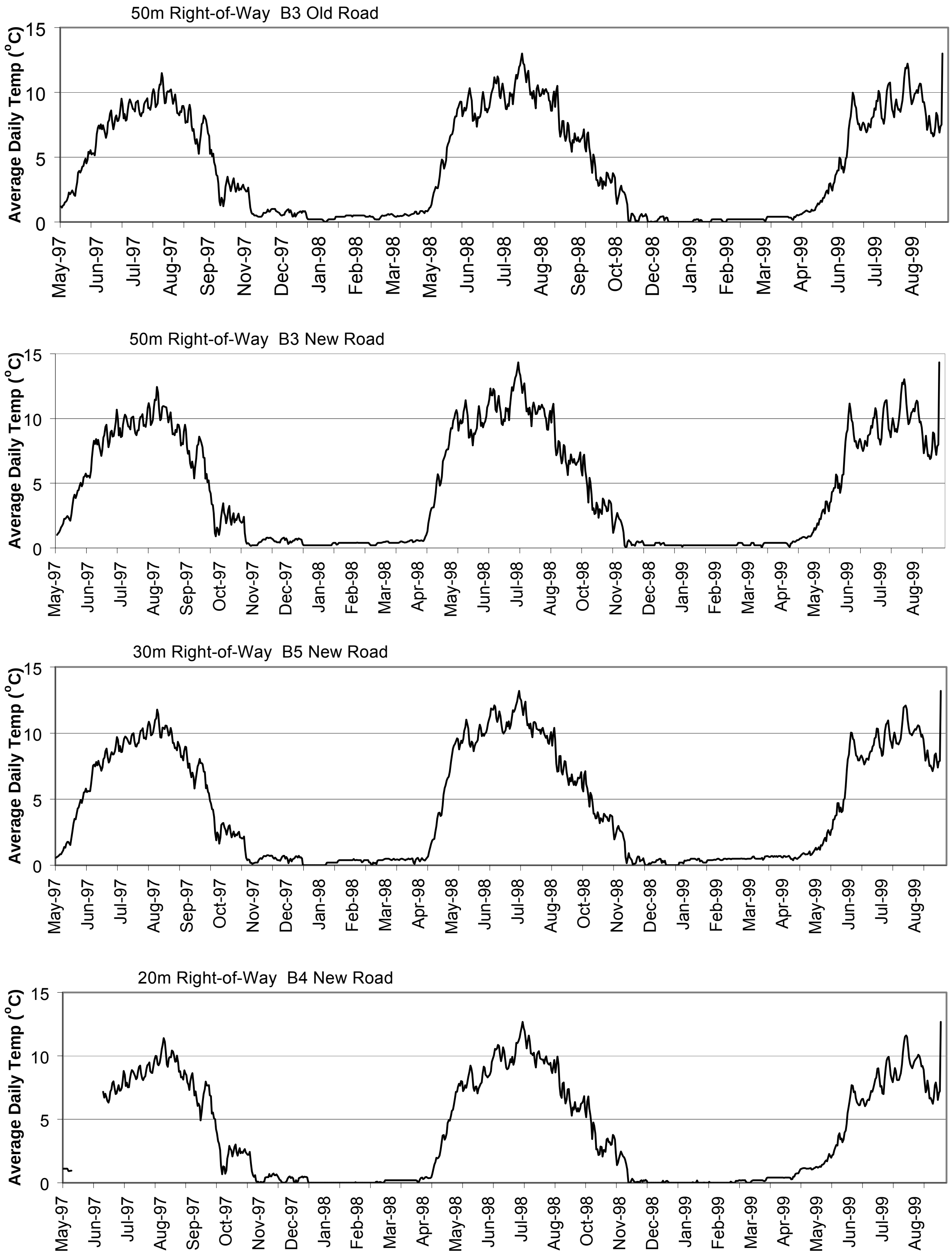

Fig. 4. Average daily creek temperature downstream of four road crossings with three different right-of-way widths.

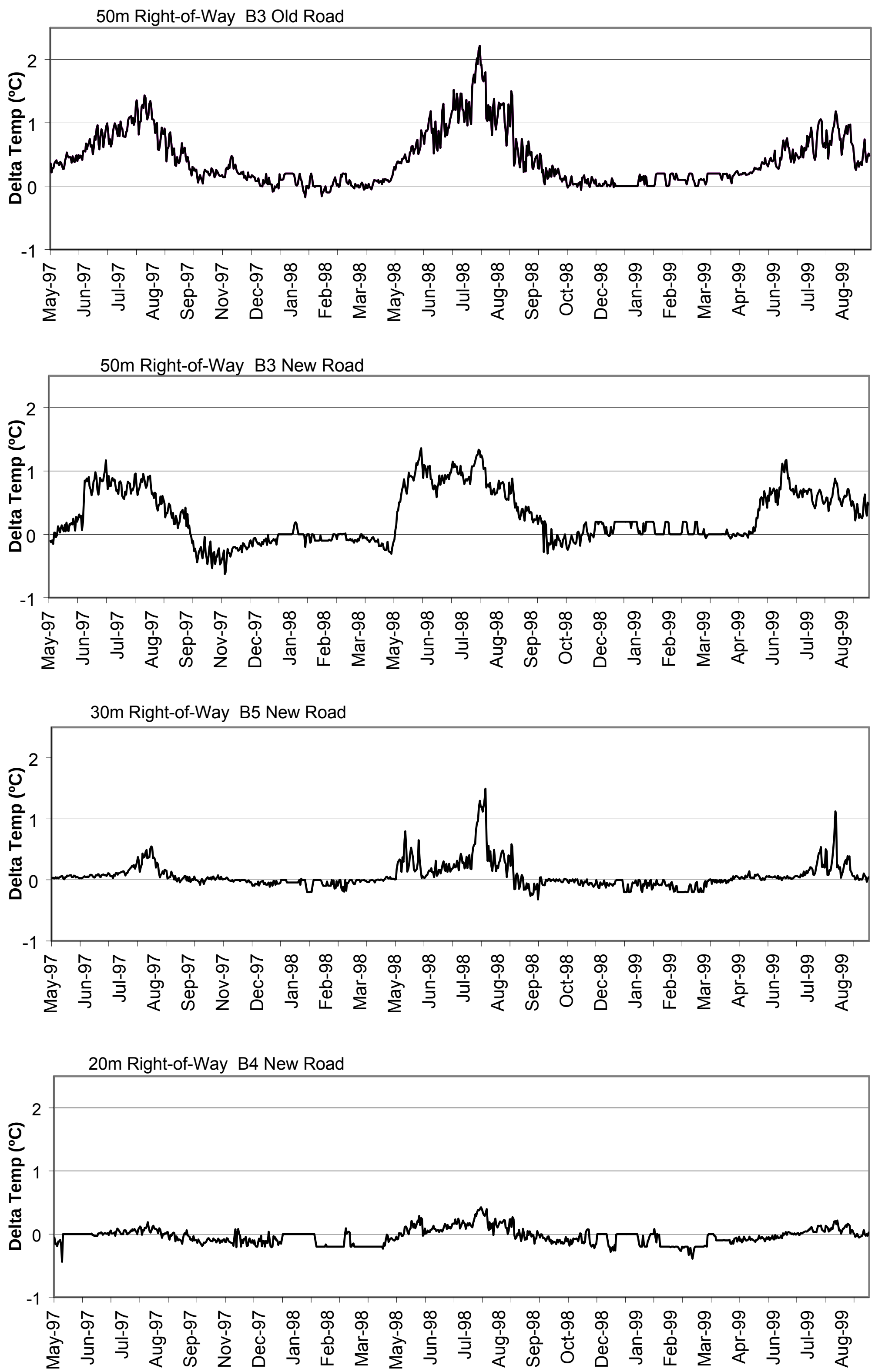

Fig. 5. Daily average temperature difference (delta T) between downstream stations and upstream stations at four stream crossings with three different right-of-way widths.

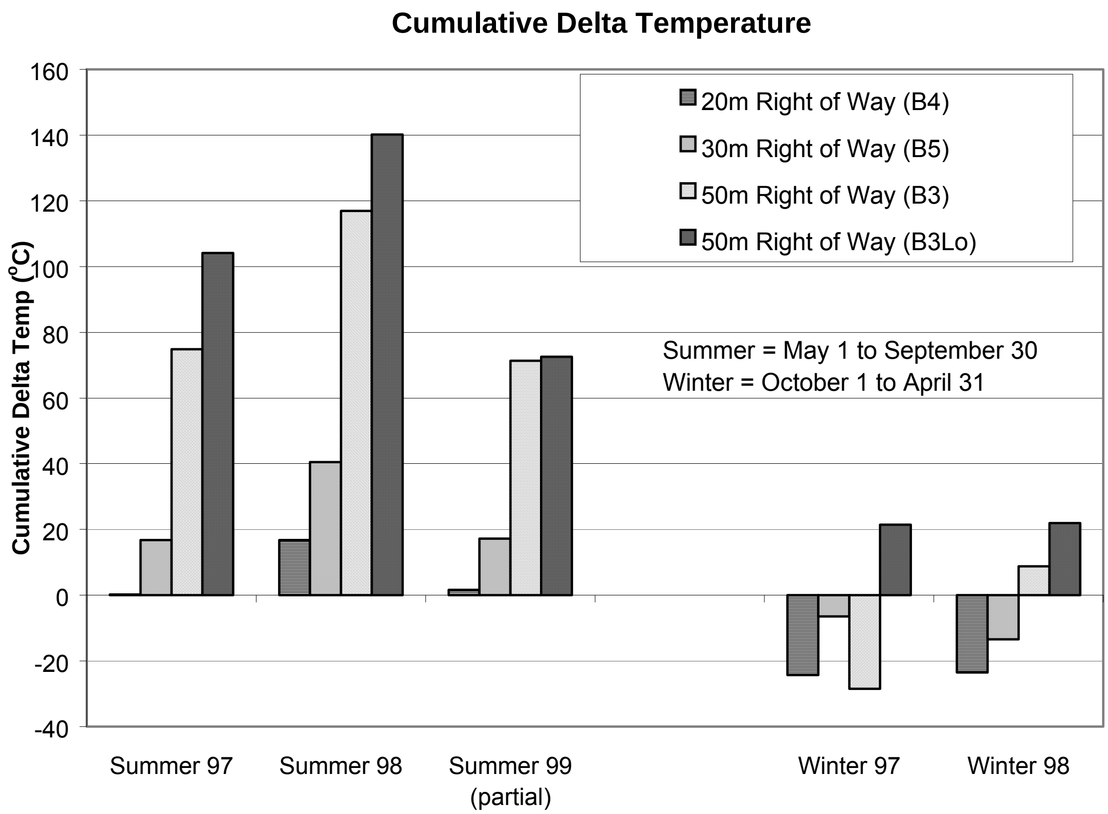

Fig. 6. Cumulative delta creek temperature of four stream crossings and three right-of-way widths. Values were calculated by summing the difference of average daily temperature between downstream and upstream stations on either side of the right-of-ways. Summer 1999 values end September 9.

The Effects of Logging and Solar Ultraviolet Radiation on Benthic Invertebrates in Baptiste (B5) Creek

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INTRODUCTION

The ecological health and stability of streams and rivers are intimately linked to the surrounding terrestrial ecosystem by riparian zones. Destruction or alteration of riparian zones through forestry activity can affect many physical, chemical, and biological properties of the stream. Clear-cut logging may change solar exposure (Brosofske et al. 1997; Hetrick et al. 1998a; Chen et al. 1999), thermal regime (Brown and Krygier 1970; Lynch et al. 1984; Holtby 1988; Garman and Moring 1991; Hetrick et al. 1998a), flow levels (Lisle 1982), channel width and depth (Heede 1991; Ryan and Grant 1991), sediment transport (Lisle 1982; Heede 1991) and concentrations of suspended and dissolved matter in the stream (Hobbie and Likens 1973; Borman et al. 1974; Meyer and Tate 1983; Garman and Moring 1991). These changes in physical and chemical character may in turn lead to large changes in benthic algal (Stockner and Shortreed 1976; Lowe et al. 1986; Robinson and Rushforth 1987; Hetrick et al. 1998a) and invertebrate (Newbold et al. 1980; Murphy and Hall 1981; Carlson et al. 1990; Brown et al. 1997; Hetrick et al*.* 1998b) productivity and assemblage composition.

Several studies have examined the effect of riparian logging and forest canopy removal on increasing the exposure of streams to solar radiation (Robinson and Rushforth 1987; Hetrick et al. 1998a). However, none of these studies have specifically examined the effect of increased ultraviolet (UV) radiation. Current levels of UV may have significant consequences for aquatic ecosystems, either through direct sensitivity of organisms to exposure (e.g. Bothwell et al*.* 1993,1994; Karentz et al. 1994; Häder et al. 1995, 1998) or indirectly through altered trophic (Bothwell et al. 1994; Hessen et al. 1997; Williamson et al. 1999) or chemical (Scully et al. 1996; Häder et al 1998; Zepp et al. 1998) interactions. In freshwater systems the biological effects of UV can be substantially altered by the presence of dissolved organic carbon (DOC) in the water. This is because DOC is the major constituent in water which absorbs UV radiation (Kirk 1994; Scully and Lean 1994; Williamson et al*.* 1996; Morris et al*.* 1995). DOC concentrations of only a few milligrams per litre can dramatically attenuate UV intensity, even in shallow water (Scully and Lean 1994). Increased concentrations of DOC of this magnitude in woodland streams following logging operations have been documented (Meyer and Tate 1983) and could potentially protect aquatic organisms from the harmful effects of elevated UV radiation reaching streams following canopy removal.

The objective of this study was to examine the effects clear-cut logging on invertebrate communities in a headwater boreal stream. Specifically, we examined the role of increased solar ultraviolet radiation in driving the response of invertebrate communities to logging. We tested the following four hypotheses: (1) that clear-cut logging and removal of the forest canopy would increase stream exposure to UV radiation, (2) that logging would result in significant changes in the density and composition of invertebrate communities, (3) that exposure to UV would significantly affect invertebrates in logged reaches of the stream, and (4) that DOC increases following logging would help ameliorate the biological impacts of UV exposure.

MATERIALS AND METHODS

Experimental Design

During the winter of 1996/1997, stream B5 of the Baptiste Watershed (hereafter referred to as Baptiste Creek) was commercially logged as part of the Stuart-Takla Fish-Forestry Interaction Project. Sixty hectares of the 150-hectare watershed were clear-cut, resulting in over half of the total channel length running through the cut-block. A Riparian Management Zone (RMZ) was left along side the stream channel as prescribed by the British Columbia Forest Practices Code (B.C. Ministry of Forests 1995). The 20m wide RMZ left along each side of the stream was comprised of non-merchantable timber (diameter at breast height <15 cm for *P. contorta* and <20cm for *P. glauca* and *A. lasiocarpa*. Heavy machinery was permitted to work within 5 m of the stream channel to remove merchantable timber, resulting in significant alteration of understory vegetation. Despite the application of these management practices, the retention of only non-merchantable trees within the RMZ resulted in significant disturbance along the entire stream channel

The effect of the winter harvest on Baptiste Creek (stream B5) was examined during July and August of 1997. A 3x2 factorial experiment considered three logging treatments (unlogged reference, top of clear-cut, and bottom of clear-cut) and two light treatments (photosynthetically active radiation (PAR) +UV or PAR –UV). Paired light treatments were replicated four times in the unlogged reference (500-250 m upstream of the forest/clear-cut edge), and twice at each of the clear-cut sites (top of the clear-cut = 150-200 m downstream of the forest edge; bottom of the clear-cut = 500-600 m downstream of the forest edge) for a total of 16 replicates and 6 treatments.

The two light treatments were established by placing large UV filters across the entire stream channel, 10 to 20 cm above the water surface. Acrylic sheets that either transmitted full spectrum solar radiation (Acrylite OP4; CRYO Industries) or excluded solar UV (Plexiglas UF4; Rohm and Hass) were fastened to 1.22 x 1.22 m wooden frames which were suspended with rebar from the stream bank. Acrylite OP4 transmitted 70 to 90% of total solar radiation below 400 nm (Bothwell et al. 1994). Plexiglas UF4 blocked out both UVA and UVB radiation (50% cutoff = 398 nm) (Bothwell et al. 1994).

Under each screen 6 plastic trays (25 x 15.5 x 6 cm) filled with uncolonized, clean gravel substrate were placed to lie flush with the streambed. Benthic communities were permitted to naturally colonize the trays undisturbed until sampling.

Sampling Protocol and Analysis

Photosynthetically active radiation (PAR), UVA, and UVB were continuously recorded in the unlogged reference and bottom clear-cut sites. Stream temperature was also continuously recorded in the unlogged reference, top clear-cut and bottom clear-cut sites. Total dissolved phosphorus, total dissolved nitrogen, nitrate, DOC, alkalinity, and stream pH were all measured upstream of the clear-cut and within the clear-cut at 2 week intervals. Measurements of DOC concentration were used to estimate the attenuation of UVA and UVB radiation within the water column (Scully and Lean 1994).

Benthic invertebrates were sampled every two weeks by removing a substrate-filled tray from under each UV shield. All invertebrates collected on a 1-mm sieve were identified and enumerated by light microscopy. Insects were identified to the lowest taxonomic level (Resh and Jackson 1993) (usually family or genus level), except dipterans, which were only identified to family. Identifications were done using keys by Clifford (1991), and Merritt and Cummins (1996).

RESULTS

Physical and Chemical Data

Logging significantly increased solar radiation reaching the stream surface. Removal of the forest canopy resulted in an average 8.3-fold increase in mean daily PAR in the clear-cut over the unlogged reference (Fig. 1a). Similarly, UVA and UVB fluxes were approximately 5.4 and 2.8 times greater respectively at the bottom of the clear-cut (Fig.1b,c).

The concentration of DOC at each site influenced the proportion of incident solar radiation reaching the stream bottom. DOC averaged 6.2 mg C l^{-1} in the unlogged reference and 10.5 mg C l^{-1} at the bottom of the clear-cut (Table 1). In the unlogged reference, this resulted in an average 21% reduction in UVA and 44% reduction in UVB penetrating to 5 cm, and an average 39% and 75% reduction in UVA and UVB in the clear-cut. While higher concentrations of DOC in the clear-cut greatly diminished the effect of logging in increasing biologically active UV radiation, average UVA and UVB remained higher in the clear-cut than in the unlogged reference.

Incident solar radiation was assumed to be nearly equal between the bottom clear-cut and top clear-cut sites. The sites were separated by 300m, had the same orientation and bank height, and the riparian canopy was not significantly different. Canopy closure measured with a spherical densiometer (Lemmon 1957) averaged 73.3% in the unlogged reference, 1.5% at the top of the clear-cut and 2.2% at the bottom of the clear-cut. DOC was not measured at the top of the clear-cut.

Stream temperature was significantly higher in the logged reach of Baptiste Creek. During the summer prior to logging, mean summer stream temperature was 6.3ºC in both the unlogged reference and bottom clear-cut sites (Steve Macdonald, Department of Fisheries and Oceans, unpublished data). During our experiment in the summer following logging, mean temperature was 8.1ºC in the unlogged reference, 9.6ºC at the top of the clear-cut, and 10.6ºC at the bottom of the clear-cut (Fig. 2a) This resulted in total accumulated degree days being 395 in the unlogged reference, 489 at the top of the clear-cut, and 565 at the bottom of the clear-cut (Table 1). Throughout the day, stream temperature fluctuated an average 1.5ºC in the unlogged reference, 3.8ºC at the top of the clear-cut, and 4.1ºC at the bottom of the clear-cut (Fig. 2b).

Concentrations of inorganic nutrients were higher in the logged reach of Baptiste Creek than in the upstream reference. Mean total dissolved phosphorus was 7.4 μg l^{-1} in the unlogged reference and 16.7 μg l^{-1} in the bottom clear-cut (Table 1). Mean total dissolved nitrogen was 224 μg l^{-1} in the unlogged reference and 318 μg l^{-1} at the bottom of the clear-cut. The increase in TDN resulted from an increase in nitrogen species other than nitrate, which remained relatively constant at 20 μg l^{-1} in the reference and 21 μg l^{-1} in the clear-cut. pH was similarly neutral between the two sites, however, alkalinity dropped from 88.2 μeq l^{-1} in the unlogged reach to 74.0 μeq l^{-1} in the bottom of the clear-cut.

Invertebrate Community

Logging decreased invertebrate diversity (Repeated Measures Analysis of Variance $P<0.001$; Fig. 3). Diversity in the bottom of the clear-cut was significantly lower than either the unlogged reference or the top of the clear-cut (post-hoc Student-Newman-Keuls $P<0.05$). However, there was no significant difference in invertebrate diversity between the unlogged reference and the top of the clear-cut. Ultraviolet radiation did not significantly affect invertebrate diversity in any of the logging treatments.

Total invertebrate density at the bottom of the clear-cut was significantly higher than both the unlogged reference and top clear-cut sites (RM-ANOVA $P=0.0045$; SNK $P<0.05$; Fig. 4a). The increase was driven by an explosion of Diptera, primarily chironomids, in the bottom of the clear-cut (RM-MANOVA $P=0.0036$; Table 2; Fig. 4b). During the first 42 days of the experiment, chironomid density averaged approximately 350% higher in the bottom of the clear-cut than either the unlogged reference or top of the clear-cut. Throughout the entire experiment, total invertebrate and chironomid densities at the top of the clear-cut were not significantly different than the unlogged reference.

While chironomids increased at the bottom of the clear-cut, Plecoptera decreased (RM-MANOVA $P<0.001$; Table 2; Fig. 4c). Total stonefly density averaged 68% lower in the bottom of the clear-cut than both the unlogged reference and top clear-cut. Stonefly response to logging was driven by 2 taxa that declined at the bottom of the clear-cut (RM-ANOVA Leuctridae $P<0.001$; Chloroperlidae $P=0.0014$) and one species that increased at the bottom of the clear-cut (*Zapada spp.* $P<0.001$). Again, there was no significant difference in Plecoptera density between the top of the clear-cut and the unlogged reference. Finally, total Ephemeroptera density was not significantly affected by any logging treatment (RM-MANOVA $P=0.23$; Fig. 4).

Despite no significant UV x Site interaction in any of the invertebrate groups (Table 2), there were different effects of UV exposure in the different sections of Baptiste Creek. While the effect of logging was strongest at the bottom of the clear-cut, invertebrates there did not respond to UV exposure (Fig. 4). Similarly, there was no clear difference in invertebrate density between UV exposed and UV shielded treatments in the unlogged reference. However, at the top of the clear-cut, mayfly density averaged 50% greater in UV protected environments, a difference that increased steadily over the course of the experiment (Fig. 4d). Similarly, during the first

42 days of the experiment, chironomid and total stonefly densities at the top of the clear-cut were consistently higher when shielded from UV (Fig. 4b,c).

DISCUSSION

Clear-cut logging in Baptiste Creek led to significant increases in light exposure, stream temperature, and concentrations of dissolved nutrients. In addition to increasing PAR, the results of our experiment clearly demonstrate that removal of the forest canopy also increases the exposure of the benthic community to UV radiation.

Our experiment demonstrates that the effect of logging was not consistent throughout the entire cut-block. Logging resulted in an instantaneous increase in some variables, such as light exposure, while effects on other variables, such as stream temperature, increased with distance downstream of the forest/clear-cut edge. This result suggests that the size of the cut-block, and the length of channel harvested, might be extremely important in determining the overall effect of clear-cut logging on streams.

In addition to the size of the cut-block, other characteristics of the logging operation will influence what impact clear-cut logging may have on streams. The size and character of the buffer strip left alongside the stream channel may significantly mitigate the impact of logging (Castelle et al. 1994; Chen et al. 1999). For example, Newbold et al. (1980) recommended that complete buffer strips of 30 m would prevent significant effects on a wide range of abiotic and biotic factors. Our results suggest that a 20 m buffer strip comprised only of non-merchantable timber is insufficient to protect small headwater streams from the effects of clear-cut logging. In addition, the number of stream crossings within the cut-block, the method of logging employed (machine or hand felled), and whether or not trees are felled and yarded across the stream channel may also influence the final effect of logging on streams.

The characteristics of the stream and watershed are likely just as important in determining the effects of logging, as the nature of the logging operation itself. Channel characteristics which may be important include width, depth, and volume. For example, riparian vegetation has less influence on wider streams and rivers (Naiman et al. 1992), and benthic communities in deeper streams enjoy greater attenuation of UV by the water column. In addition, larger volume streams will have increasingly greater thermal inertia, and will require greater inputs of energy per degree increase in temperature. Characteristics of the watershed may also be important. The aspect of the watershed, particularly in higher latitude areas, will affect the number of hours a channel is exposed to direct sunlight each day. Similarly, the slope of the catchment, and particularly the slope of the hillside immediately adjacent to the stream channel, will affect the exposure of the channel to solar radiation, runoff from the catchment, and the severity of soil erosion and sediment inputs. For example, streams at the bottom of steep gullies may not experience increases in solar exposure with riparian logging, although they may be more susceptible to high erosion and increased suspended sediments. Finally, the chemical properties of the water may be important in determining the effect of logging. For example, streams already nutrient rich will likely not respond to an increase in nitrogen or phosphorus with logging. Also, the natural concentration of DOC may affect the proportion of incident UV radiation that reaches to ecologically significant depths.

Because the effects of logging on the stream environment differed within the clear-cut, so too did its effects on the biological community. In our experiment, large changes in invertebrate communities did not occur until lower down in the clear-cut. Obviously, streams represent a continuum, and disturbances in the watershed may not be observed immediately. Our results suggest that the biological effects of logging within a cut-block accumulate with distance downstream of the forest/clear-cut edge. This observation may have significant implications for other studies of the effects of logging on streams, as results will depend on where within the cut-block studies are conducted.

At the top of the clear-cut, exposure to UV may have had an effect on invertebrates, while at the bottom of the clear-cut, UV had no observable impact. The 4 mg/L increase in DOC between the top and bottom of the clear-cut may have sufficiently shielded invertebrates from potentially deleterious effects of UV. Consequently, the removal of the forest canopy along streams, and the resulting increase in solar radiation, may not automatically result in a response of the biotic community to UV radiation. Organisms are continually responding to an infinite number of niche dimensions, and the relative importance of UV radiation will therefore depend on the amount of UV individuals are exposed to, plus the relative severity of other environmental factors.

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Table 1. Select limnological variables for Baptiste Creek. Water samples were collected bi-weekly during the experimental period.

Variable	Unlogged Reference Mean	Unlogged Reference Standard Deviation	Top of Clear-cut Mean	Top of Clear-cut Standard Deviation	Bottom of Clear-cut Mean	Bottom of Clear-cut Standard Deviation
DOC (mg C l^{-1})	6.2	1.3	–	–	10.5	2.5
TDP (µg l^{-1})	7.4	1.2	–	–	16.7	5.1
Nitrate (µg l^{-1})	20.1	14.7	–	–	20.6	28.3
TDN (µg l^{-1})	223.6	39.1	–	–	317.7	125.6
Temperature (°C)	8.1	0.8	9.6	1.2	10.6	2.1
Degree Days	395	–	489	–	565	–
pH	7.7	0.2	–	–	7.6	0.2
Alkalinity (µeq l^{-1})	88.2	20.7	–	–	74.0	8.0
Depth (cm)	8.1	2.8	5.8	2.2	5.8	2.5

Table 2. Results of RM-ANOVA of the effects of ultraviolet radiation (UV) and logging treatment (Site) on density of Diptera, Plecoptera, and Ephemeroptera.

Source	df	**Diptera** F	**Diptera** P	**Plecoptera** F	**Plecoptera** P	**Ephemeroptera** F	**Ephemeroptera** P
UV	1	0.26	0.78	1.12	0.42	0.65	0.60
Site	2	5.17	0.0036	10.76	<0.001	1.53	0.23
UV x Site	2	0.43	0.78	0.88	0.56	1.20	0.35
Time	3	2.37	0.041	6.25	<0.001	6.12	<0.001
Time x UV	3	0.94	0.47	1.07	0.40	1.00	0.45
Time x Site	6	1.58	0.12	2.11	0.0058	1.11	0.36
Time x UV x Site	6	0.87	0.58	1.95	0.012	0.59	0.90

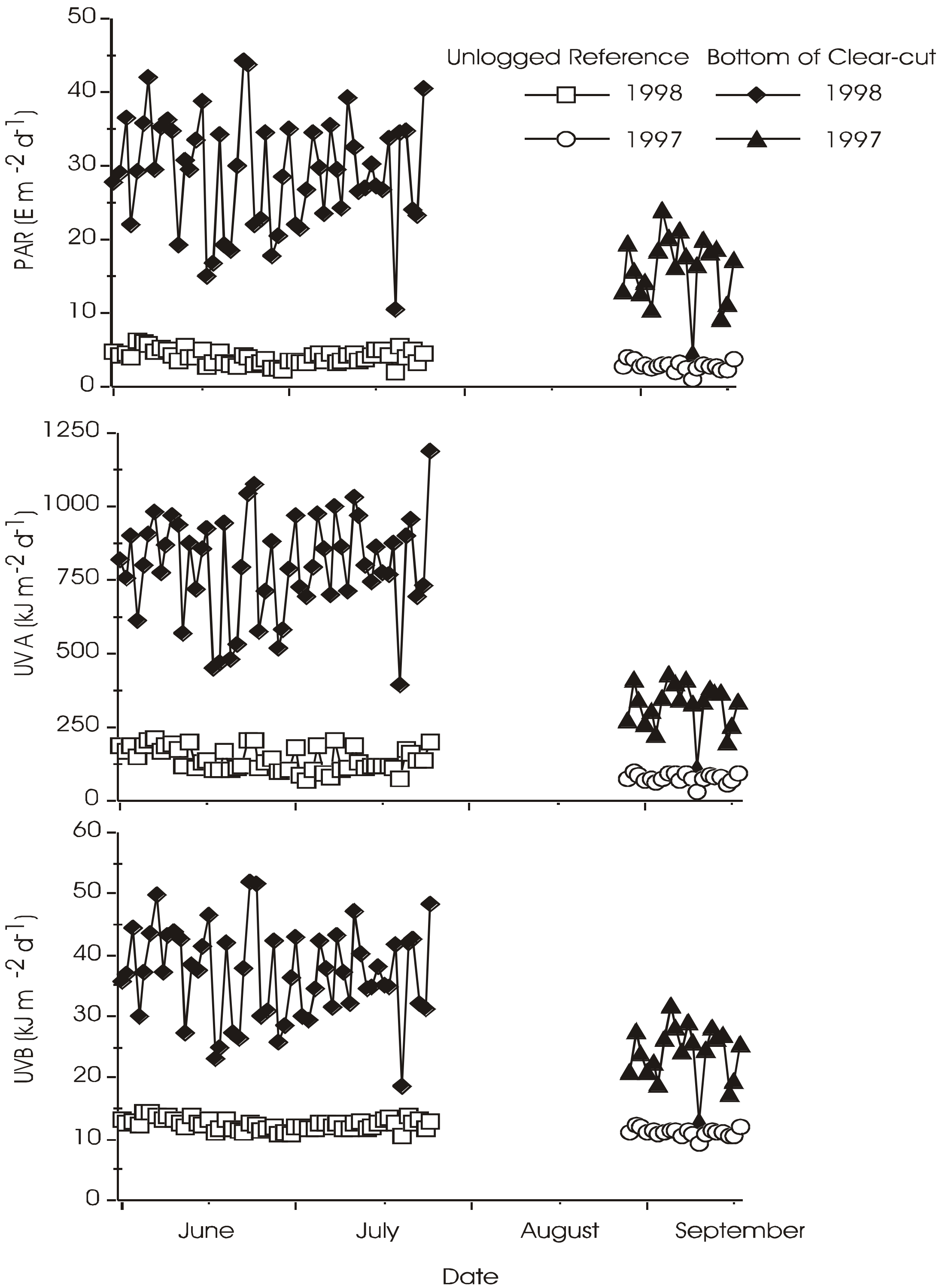

Fig. 1. Solar radiation (PAR, UVA and UVB) in the unlogged reference and bottom clear-cut sites during the recorded periods of 1997 (mid August to mid September) and 1998 (June to mid July).

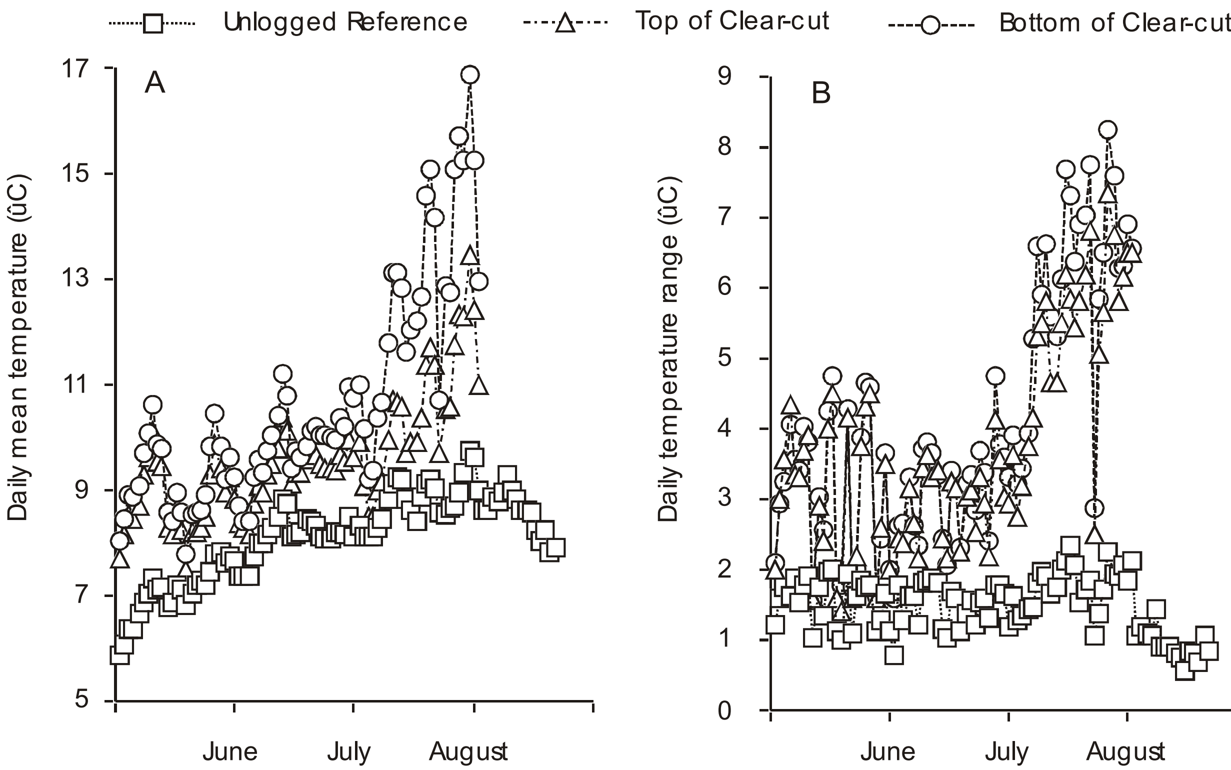

Fig. 2. Daily mean temperature (A) and range in temperature (B) in Baptiste Creek at the unlogged reference, top clear-cut and bottom clear-cut sites.

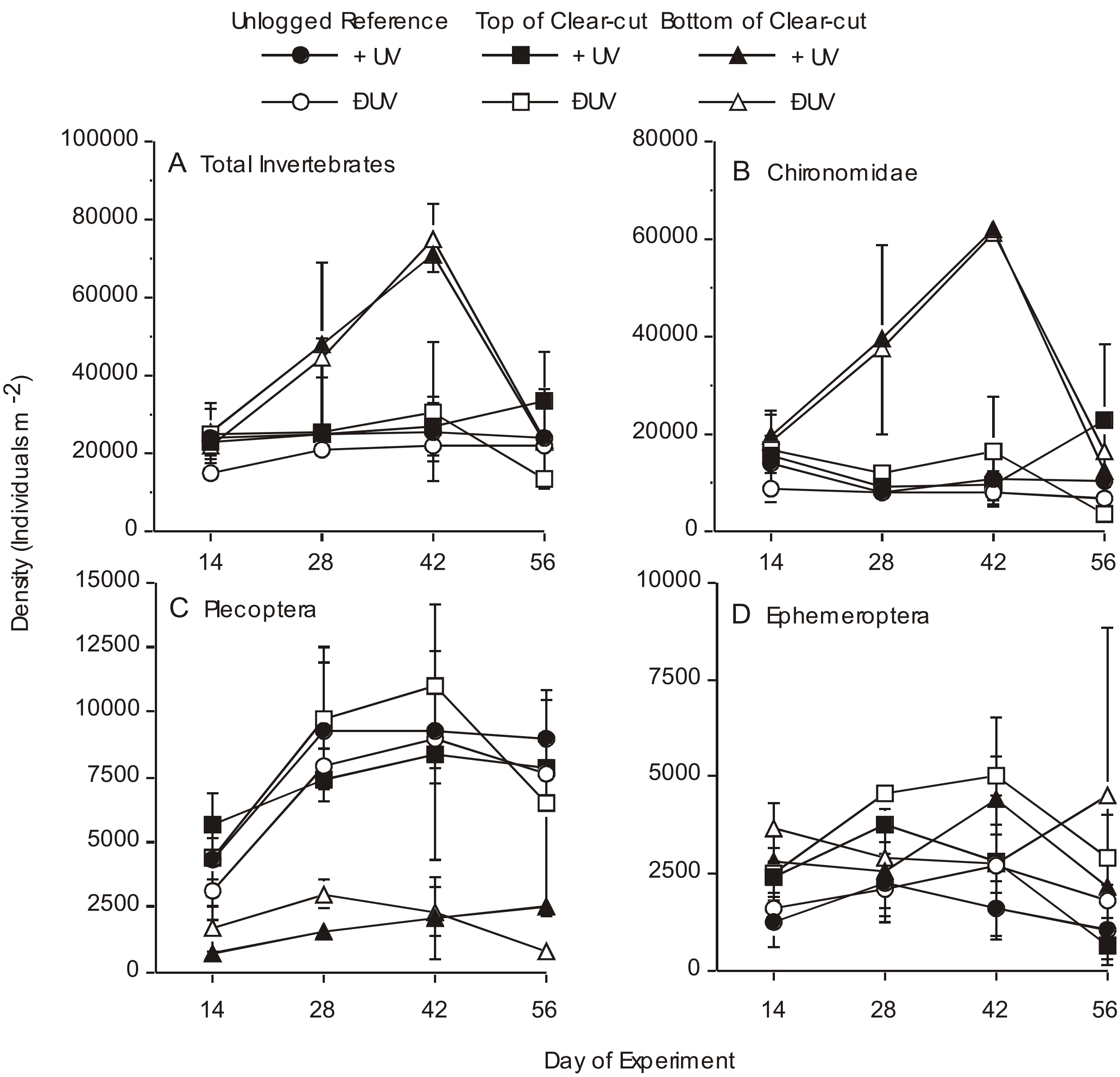

Fig. 3. Total invertebrate (A), Chironomid (B), Plecoptera (C), and Ephemeroptera (D) densities between UV exposed (black) and UV shielded (white) treatments in the unlogged reference (circle), top clear-cut (square), and bottom clear-cut (triangle) sites. Error bars are ± standard error.

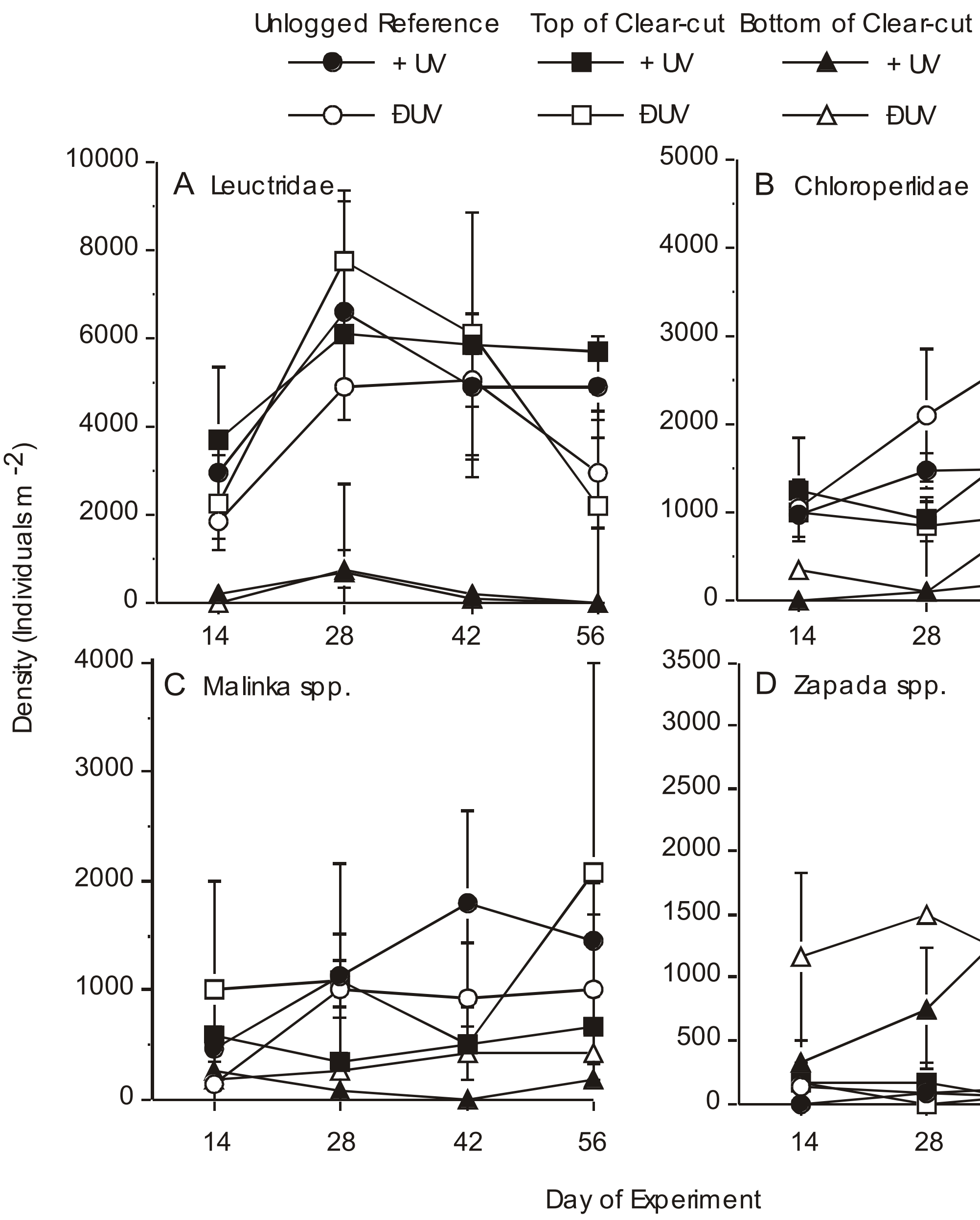

Fig. 4. Density of stonefly (Plecoptera) taxa in both the UV exposed (black) and UV shielded (white) treatments in the unlogged reference (circle), top clear-cut (square) and bottom clear-cut (triangle) sites. Error bars are ± standard error.

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Macroinvertebrate Sampling Design and Preliminary Results from Small Streams in the Baptiste Watershed Study

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INTRODUCTION

Current logging practices in British Columbia provide no mandatory riparian buffers on small streams (< 1.5 m bankfull width) regardless of the presence of fish. These streams can be headwater drainages or small streams tributary to larger streams and lakes and they are ubiquitous in BC watersheds. Small streams can be spawning and rearing habitats for fish and they export cool water, nutrients, sediment and organic matter to downstream aquatic habitats (Vannote et al. 1980). Logging is known to affect many of the processes of riparian–stream ecosystems (Salo and Cundy 1987, Brewin and Monita 1998, Naiman et al. 2000) but compared to larger stream systems, relatively little is known about how logging affects small streams and how these effects translate into impacts on downstream aquatic habitats.

The production of invertebrate forage and drift for rearing salmonids may be one of the most important functions of small streams in interior BC watersheds. Logging riparian vegetation alters both the aerial input of terrestrial insects and the instream production of stream invertebrates (Hetrick et al. 1998b). Reduced organic matter inputs from allochthonous leaf litter sources affects invertebrate abundance and production and the structure of stream foodwebs (Gregory et al. 1987, Webster et al. 1990, Wallace et al. 1991, Bilby and Bisson 1992, Hetrick et al. 1998a). For example, in larger streams, increases in light and summer stream temperatures after logging have increased stream algae production and the abundance of invertebrates that feed on periphyton (Hawkins et al. 1982; Hetrick et al. 1998a; 1998b).

Logging also affects many other attributes of small streams such as solar radiation exposure, thermal regimes, streambed morphology, hydrology, organic matter retention and nutrient pathways that can interact and influence invertebrate communities. Reduced inputs of large woody debris physically alter sediment and organic matter storage and the stability and composition of the stream bed (Murphy et al. 1986 and Chamberlain et al. 1991) and these substrate composition changes can directly influence the abundance of certain macroinvertebrate taxa (Hawkins et al. 1982; Everest et al. 1987). Logging can also affect stream hydrology by increasing peak flows, which in turn cause benthos disturbance and influence macroinvertebrate drift (Anderson 1992). Thus, invertebrate communities are sensitive to many disturbances to stream and riparian conditions and for this reason they have become a useful indicator of stream health and degradation (Rosenberg and Resh 1993).

As a component of a multidisciplinary small stream study in the Baptiste watershed of the Stuart-Takla Fish-Forestry Interaction Project (STFFIP), invertebrate abundance and downstream export were examined in three small streams before and after selective riparian forest harvesting took place. There are two main objectives of the macroinvertebrate study in Baptiste. First, is to examine the impact of forest harvesting on the benthic macroinvertebrate community for 3 years post harvest. Second, is to quantify and compare the export of invertebrate drift from small streams with different levels of riparian vegetation retention.

We choose two invertebrate sampling techniques to measure both instream community structure and downstream export. Colonization baskets were used to track changes in community structure both upstream and downstream of forestry activity and drift samplers were used to estimate export of drifting macroinvertebrates. Terrestrial invertebrates entrained in the water column and float ing on the neuston layer are included in our drift samples (Wipfli 1997). Drift samples were collected in fishless areas avoiding predation as a confounding factor.

This report provides a description of our sampling techniques, experimental design, and provides preliminary analysis of some of the samples processed thus far. Although all samples have been collected, only the colonization basket samples have been analyzed to date (enumeration only). We provide a preliminary analysis of the abundance and distribution of benthic invertebrates two years post harvest and describe site specific differences in abundance and diversity of major insect taxa related to the different riparian treatment prescriptions.

MATERIAL AND METHODS

Study Area and Sampling Stations

Three small streams located in the Baptiste watershed received 3 different riparian treatments: B4 spatial ‘control’, B5 ‘aggressive’ treatment (no merchantable timber retained), and B3 ‘conservative’ treatment (merchantable timber < 20cm d.b.h. retained) (Fig. 1). Forest harvesting occurred in the winter of 1996/97 within the B3 and B5 watersheds.

Benthic macroinvertebrates were sampled using colonization baskets during the summers of 1996, 1997, and 1998 in the upper Baptiste watershed and in 1999 both colonization baskets and invertebrate drift samples were collected. Samples were taken from three creeks, B3, B4, and B5 at seven station locations (B3Hi, B3Lo, B4, B5Hi, B5Mid, B5Lo, B5AR) during that period (Table 1). B4, B3Hi, and B5Hi are reference or control sites not directly affected by harvesting. B5Mid is in the middle of a cutblock while B3Lo and B5Lo are at the bottom of the cutblocks. B5AR is 150m downstream of cutblock edge in a forested area and is considered a recovery zone station (Table 1).

Colonization Basket

Benthic invertebrates were collected using colonization baskets, a type of artificial substrate sampler. Each sampler consisted of a cylindrical plastic colander (11 cm high X 16 cm diameter; 200 cm^2 surface area; 2200 cm^3 volume; solid bottom; 1 cm vertical slots). The sampler was filled with clean gravel obtained from a recently excavated gravel pit that was sieved to between 1-5 cm diameter (Koning et al. 1995). Three baskets were installed flush with the streambed at each station. Stream depth and velocities in some cases were recorded. The baskets were installed in riffle/glide reaches, although reach features sometimes changed during the sampling period depending on flow.

Colonizing benthic invertebrates were sampled after 4 to 6 weeks by pulling the basket from the substrate and placing it in a clean 4 L bucket. Each sample was then placed in a 15 L bucket with 1 L of filtered (183 μm) water. The water and invertebrates were decanted off and rinsed through a 183 μm screen. This process was repeated three times for each sample and the remaining coarse material was scanned for residual invertebrates. If remnants were found an additional rinse was completed to remove the remaining fauna. All samples were immediately preserved in 10% formalin. The sample invertebrates > 250 μm were then enumerated and the insects identified to family level and functional group at the Fisheries and Oceans Canada Cultus Lake Laboratory using Merritt and Cummins (1996) as an identification reference.

The two major concerns with colonization baskets are species selectively and the time to full colonization (Clements 1991). We attempted to standardize the length of colonization period, but there were variances in this time. However, colonization basket work in the Baptiste streams using sequential sampling every week found that peak abundance and diversity occured within 4 weeks after which there was little variance in total numbers and diversity within a site (John Clare pers. comm.). All of the sample sites remained wetted during the sampling periods with the exception of B3Hi and we excluded B3Hi in comparisons from this analysis.

Invertebrate Drift

In 1999, invertebrate drift samples were collected from B4, B3Lo, B5Lo, and B5Hi stations. Drift samples were taken at B3Lo and B5Lo to estimate the export of invertebrate drift from a cutblock to fish bearing waters downstream. B4 and B5Hi stations were spatial and upstream references respectively.

Paired Mason drift samplers (200 μm mesh) were set monthly from June to September for two consecutive 24-hour periods, as described by Choromanski et al. (1996). Eight samplers were used to collect simultaneous samples from the 4 locations. Water velocities at the mouth of each sampler and cross sectional wetted area were recorded to calculate the total volume of water sampled. The depth of the samplers was greater than stream depth allowing the sampling of the stream surface layer as well as the main water column. The samples were preserved in 10% formalin.

Data Analysis

Invertebrate samples from July 1998 are presented because they are the most complete data set available at this time. Presence and abundance of each taxon per sample were calculated for each site and summarized based on diversity and abundance of the three inclusive categories, invertebrates, insects, and EPT (Ephemeroptera, Plecoptera, and Trichopteran). EPT index is used as an indicator of stream ecosystem health because these particular insects are considered sensitive to environmental change (Lenat 1988; Rosenberg and Resh 1993). Between creek differences were assessed by comparing the abundance of the major taxa (dipteran, ephemeroptera, plecoptera, and trichopteran) at the four different sites. Within creek differences were examined by graphically comparing the abundance of different plecopteran families divided into their two dominant functional feeding groups (shredder/detritovore and predator) (Bottorff and Knight 1988; Merritt and Cummins 1996).

RESULTS

Colonization Baskets

Among Creek Differences

The July 1998 colonization basket results for B4 and B5Hi (control stations), B3Lo (conservative treatment) and B5Mid (aggressive treatment) are presented in Fig. 2. There were no large differences in the total invertebrate abundance among the stations but there was a clear impact of the aggressive riparian treatment on the benthic invertebrate community structure. Distributions of the major taxa, ephemeroptera, plecoptera, trichoptera, and diptera, were similar for the control sites (B4 and B5Hi) and the conservative site (B3Lo) but ephemeroptera and plecoptera abundances were markedly lower in the aggressive treatment site (B5Mid). Higher numbers of diptera compensated in part for the much lower numbers of plecoptera. The community composition recovered to a certain degree after the stream flowed back into a forested area (B5AR results not shown).

Within Creek Differences

The previous comparison among creek stations found a major decline in the abundance of plecopterans at the aggressive treatment station (B5Mid). A more detailed analysis of the major families of plecopterans shows there are differences in numeric abundance between the different stations within creek B5 (Fig. 1) related to functional feeding groups (Fig. 3). The forested upstream B5Hi control site had higher numbers of shredders and detritivores (Capniidae, Leuctridae and Nemouridae) than the more open canopy B5Mid site. There was no obvious difference in two other families of predatory plecopterans, Perlodidae and Choroperlidae, between the two sites. The abundance of shredders and detritovores in the downstream forested B5AR recovery site was intermediate between B5Hi and B5Mid. Overall, there was a marked decrease in the abundance of shredders and detritivores in the clearcut area of B5 with some evidence of recovery of plecoptera numbers within 150 m of the streams re-entry into the forest.

DISCUSSION

There is a paucity of information relating forest harvesting activities to impacts on aquatic invertebrate communities in small streams in the BC central interior. Our preliminary results indicate that changes in invertebrate community structure occurred in the aggressively logged stream on both the functional feeding group and taxonomic order levels. Hetrick *et al.* (1998a & 1998b) provide some of the most recent information on canopy cover removal and changes in invertebrate community in coastal Alaskan streams. They found that ephemeropterans were more prevalent in closed canopy stream reaches while dipterans dominated in recently opened canopy areas. Our results showed a similar trend in ephemeropteran and dipteran abundance comparing B5Mid with B4 (Fig. 2). However, we also found a marked decline in the abundance of plecopterans with respect to canopy cover removal, something not found by Hetrick et al. (1998b).

Contrary to most other studies from larger streams, we did not find higher numbers of invertebrates after logging (Hawkins et al. 1982; Noel et al. 1986; Carlson et al. 1990; Hetrick et al. 1998b). In aquatic systems where light limits primary production of attached algae (periphyton), the removal of canopy cover increased periphyton growth and the abundance of invertebrate scrapers that feed on them (Noel et al. 1986; Gregory et al. 1987). However, to date we have not found a marked increase in the benthic invertebrate scrapers that are known to feed on periphyton in the logged sites (B5Mid and B3Lo).

The composition of lotic macroinvertebrate communities is highly dependent on food quality and availability; the relative contributions of allochthonous versus autochthonous organic matter influence the relative abundance of different macroinvertebrate feeding functional groups within a stream (Merritt and Cummins 1996). Riparian vegetation plays an important role in controlling the balance between these two processes. The amount and type of riparian vegetation determines the availability of leaf litter and other detrital organic matter for shredders and detritivores (Cummins et al. 1989) and determines the light availability for primary production of periphyton for macroinvertebrate periphyton scrapers. It is possible that periphyton primary production in the Baptiste streams is not limited by light but by nutrients such as phosphorus and nitrogen, which could account for the lack of an increase in invertebrate scraper abundances. Substrate stability may also be limiting periphyton abundance because the streambeds of the Baptiste streams are comprised of relatively fine materials subject to frequent bedload movement during storm events which could limit the availability of stable periphyton substrates for accrual.

Invertebrate drift is an important prey resource for stream salmonids and invertebrates from small fishless streams become available to downstream foraging salmonids through drift and entrainment of terrestrial insects (Rader 1997; Wipfli 1997) and quantification of invertebrate biomass from drift samples can be useful in evaluating food availability for fish. Analysis of the inverebrate drift samples from the Baptiste streams has not been completed yet but it is known that forest harvesting has the potential to affect many stream conditions and processes that influence invertebrate drift. Changes in the allochthonous and autochthonous organic matter inputs to streams will affect invertebrate abundances and the amount of drift (Keith et al. 1998). Invertebrate drift can also be accidental or intentional (Rader 1997). Accidental displacement of invertebrates from the stream bottom is often associated with increases in stream velocities or stream substrate disturbances. Macroinvertebrates will intentional drift in response to extreme physical conditions, like low oxygen, increased solar radiation, or reduced flow. A patchy distribution of food resources can also initiate an intential drift response. Future analysis of the Baptiste stream drift data will address both the amount of invertebrate production exported from these small streams and determine whether any change in drifting invertebrate composition has occurred with respect to different riparian treatments.

Management Implications

Preliminary analysis from this study shows a distinct shift in community structure associated with the aggressive riparian treatments but it isn't known how these disturbances to the invertebrate community will affect downstream food availability for fish or what level of downstream recovery is possible. Therefore, the most prudent advice for managing for overall ecosystem health, and not just invertebrate production for fish, would be to favour a more conservative riparian treatment prescription (Newbold et al. 1980). These results are preliminary and only provide a small glimpse of the entire data set, July 1998 only. Further analysis is ongoing and caution should be used in the application of these preliminary results.

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Table 1: A description for each sample location and the number of Colonization Basket and Invertebrate Drift Samples collected at each time period for the three small Baptiste streams (B3, B4, B5). "x" denotes samples have been analyzed.

Sample Site		**B3Hi**	**B3Lo**	**B4**	**B5Hi**	**B5MID**	**B5Lo**	**B5AR**
Site Description								
Location in Cut Block		Above	Bottom	N/A	Above	Middle	Bottom	150m Below
Riparian Treatment		None	High retention	None	None	Low retention	Low retention	None
Category		Ctrl	Conserv	Ctrl	Ctrl	Aggress	Aggress	Recover
Year	**Month**							
Colonization Baskets								
1996	Sept				3 x			3 x
1997	Aug	3 x	3 x	3 x	3			3
	Sept	3 x	3 x	3 x	3			3
1998	July	*3 x*	*3 x*	*3 x*	*3 x*	*3 x*		*3* x
	Sept	3 x	3 x	3 x	3 x	3 x		3 x
1999	July	3	3	3	3	3	3	3
	August	3	3	3	3	3	3	3
	Sept	3	3	3	3	3	3	3
Drift Samples								
1999	June		4	4	4		4	
	July		4	4	4		4	
	Aug		4	4	4		4	
	Sept		4	4	4		4	

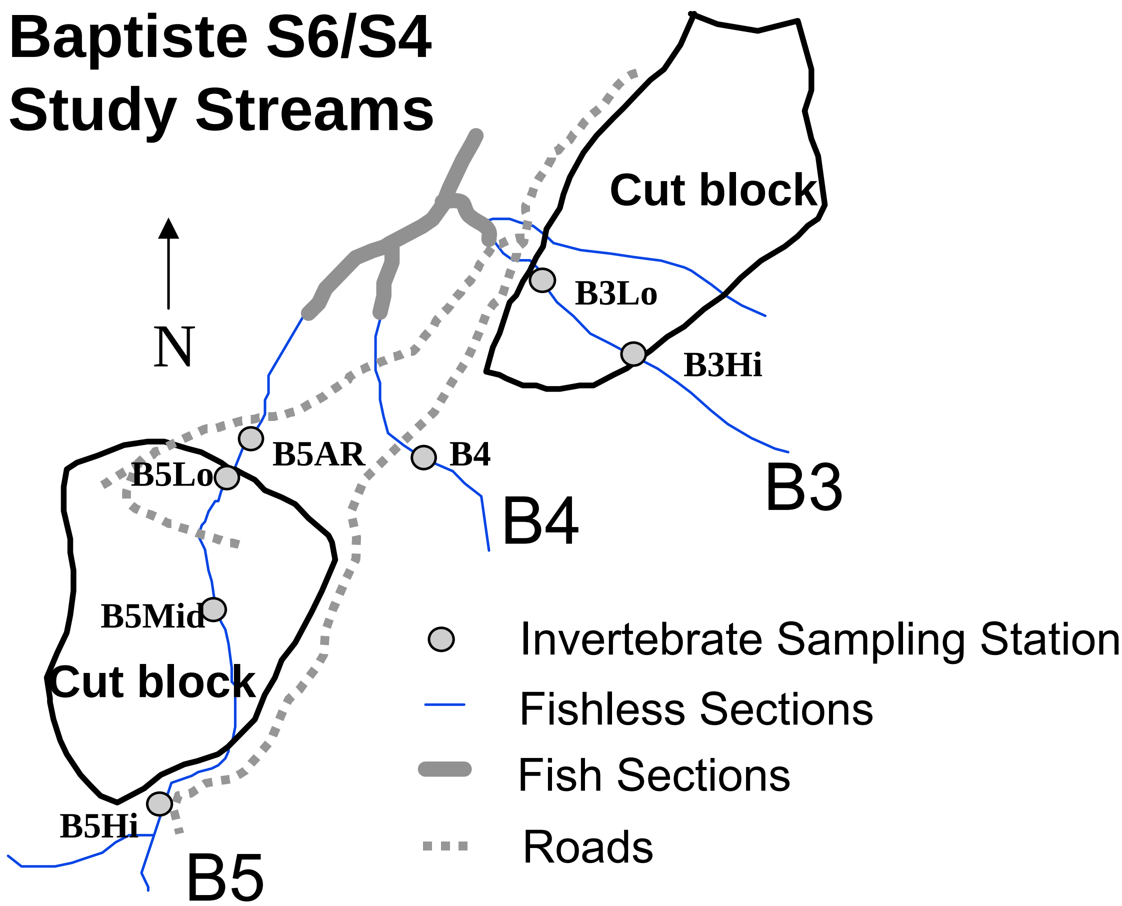

Fig. 1. Map of the Baptiste streams, the invertebrate sampling stations, the cutblock boundaries and the logging roads.

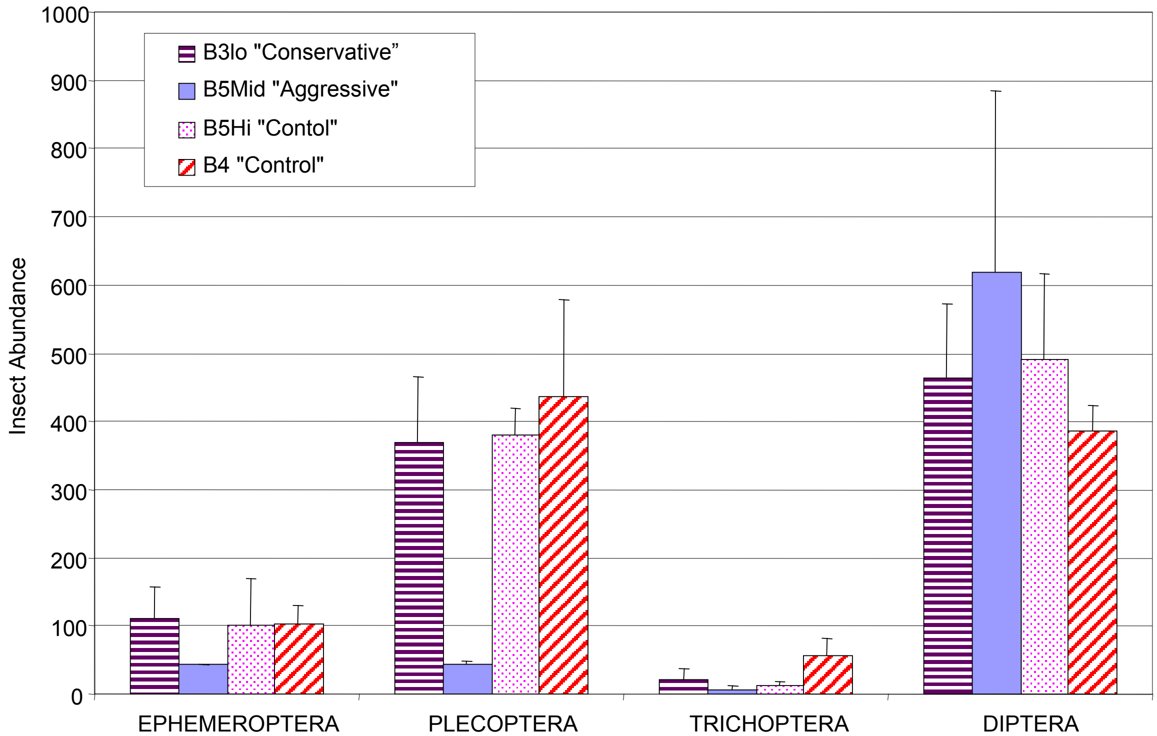

Fig. 2. An abundance comparison of major insect orders (larval or nymph life stages) at two control stations and two different riparian treatment sites (among three creeks). The values given are the mean of 3 samples per station taken in July 1998 and bars represent the upper 95% confidence limits.

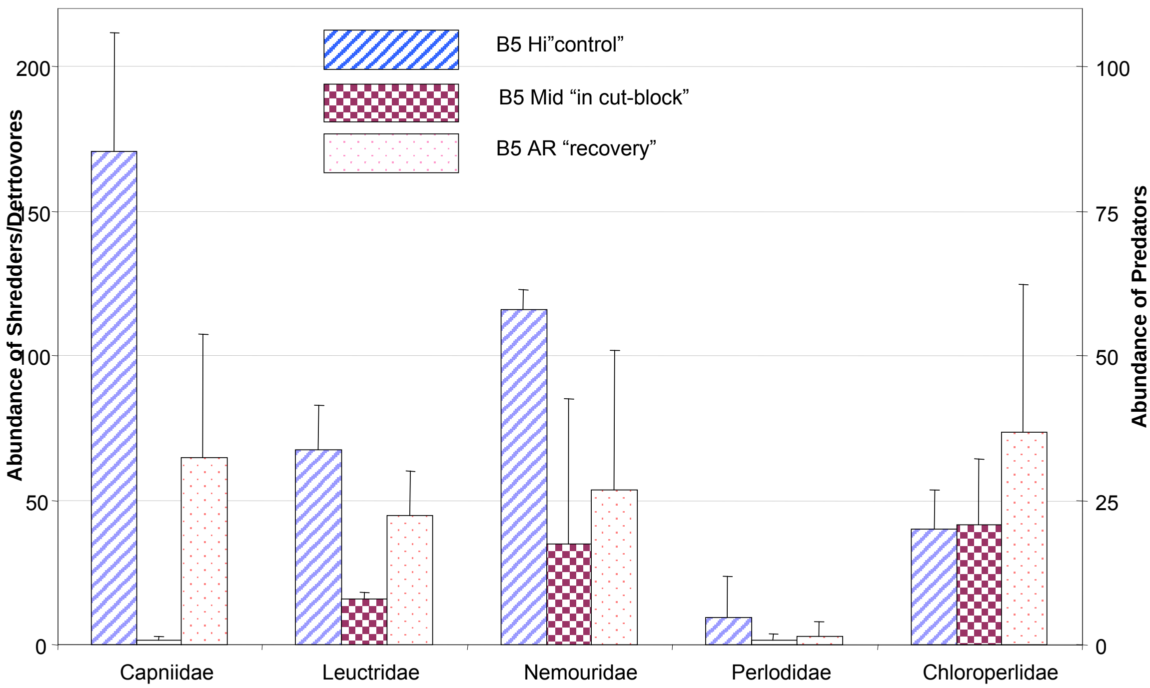

Fig. 3. An abundance comparison of different plecopteran families between three sites on creek B5 ranging from the upstream control station through the cutblock to the downstream recovery station. The families are grouped based on their repective functional feeding groups. Perlodidae and Chloroperlidae are predators and abundance is on the right axis. The others are shredders/detritivores. The values given are the means of 3 samples taken in July 1998 and are based on larval or nymph life stages. Bars represent the upper 95% confidence limits.