University of Alberta

Forest Harvesting Effects on Benthic Macroinvertebrate Abundance in the Boreal Foothills of Alberta

by

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Abstract

Benthic macroinvertebrates (BMI) are well-known indicators of stream condition. Anthropogenic forest operations, specifically harvesting and the presence of stream crossings, contribute suspended sediments to streams and can be detrimental to BMI. This study examined the effects of these processes on BMI abundance in the boreal foothills of Alberta. Generally, BMI abundance did not differ between harvested and unharvested streams, and these streams were considered in good condition. Crossings negatively impacted the abundance of all taxa in the harvested watersheds. Stream gradients and riparian vegetation appeared to strongly mitigate the effects of both harvesting and crossings on BMI communities. However, Trichoptera were sensitive to both harvest and crossing effects, and may be valuable as initial indicators of stream integrity in the Grande Cache region. Similar indicators were not identified for the Drayton Valley or Edson regions, but might be observed with taxonomic resolutions greater than the Order level.

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Introduction: Stream Integrity & Benthic Macroinvertebrates

Streams in forested watersheds can be greatly impacted by both natural and anthropogenic forest processes, including fire and harvesting, respectively. Tree removal significant on a watershed scale, regardless of the operation involved, reduces the amount of evapotranspiration in the system (Campbell & Doeg, 1989). In addition, more precipitation reaches the forest floor where it infiltrates the soil or is transported as runoff (Ursic, 1991). After forest harvesting, stream crossings provide the greatest input of suspended sediment in harvested watersheds (Rothwell, 1983).

Suspended sediment can be detrimental to the health of aquatic invertebrates and fish. Excess sediment may limit gill function, reduce light available for primary production, overwhelm filtering abilities, and minimize the availability of attachment and habitat sites for invertebrates (Campbell & Doeg, 1989; Bottorff & Knight, 1996; Stone & Wallace, 1998). Streamside harvesting also reduces terrestrial carbon inputs, and changes flow and temperature regimes, which influences invertebrate feeding, metabolism and life cycles (Campbell & Doeg, 1989; Stone & Wallace, 1998).

Taxonomic richness and abundance of macroinvertebrates are highly effective measures of stream integrity, with poor conditions indicated by taxonomic losses and reduced abundances (Norris & Thomas, 1999; Death, 2003). Biotic indicators are used because they provide a direct measure of stream integrity, integrate the effects of past disturbance events, and are cumulative indicators of multiple disturbance episodes (Micacchiou, 2003). Benthic macroinvertebrates (BMI) are sensitive to a range of anthropogenic disturbances and exhibit species-specific responses to stress (Rosenberg & Resh, 1993; Fore et al., 1996; Morley & Karr, 2002). Composition of BMI communities fluctuates around a relatively stable state, but with strong interannual variation, some of which reflects predictable seasonality (Scarsbrook, 2002). Under normal levels of

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environmental variation, BMI adapt to local conditions and communities are sustained in a general seasonal pattern (Karr, 1999). These communities are resilient and can withstand large variations, providing they occur within the range of a system's evolutionary experience. Communities may be altered profoundly when a dramatic, often anthropogenic, change occurs (Norris & Thomas, 1999). Even the smallest disturbance beyond the range of experience can have serious consequences to richness and abundance (Karr, 1999).

No single BMI metric is sufficiently sensitive to universally indicate degradation and it is rare to causally link degradation to a single point-disturbance event (Karr, 1991). BMI communities are correlated with both local habitat conditions, including substrate size, and landscape scale conditions, including stream size and catchment area (Morley & Karr, 2002; Paavola et al., 2003; Herlihy et al., 2005). Community composition in undisturbed watersheds is reasonably predictable, with environmental conditions determining which species will persist as a consequence of the match between their traits and environmental conditions (Heino et al., 2003). Forest harvesting has been associated with decreased overall BMI richness and increased abundance of particular taxa in disturbed streams (Campbell & Doeg, 1989; Stone & Wallace, 1998). Biotic effects of harvest tend to indicate the final stage of environmental degradation (Norris & Thomas, 1999). When disturbance impacts have not yet reached their full potential, an undisturbed BMI community can exist in a disturbed stream. BMI communities in headwater streams are sensitive to forest succession following harvest, and clearly respond to changes and forest regrowth for decades (Stone & Wallace, 1998; Cole et al., 2003). To the extent that the responses are 'adaptive', bringing stream biota gradually back toward the regional norm, we might well choose to understand these changes as part of an overall healthy dynamic.

All communities exhibit some natural variation, and any two communities can be found to differ, provided they are examined in sufficient detail (Eberhardt & Thomas, 1991; Oksanen, 2001; Townsend et al., 2004). Results of

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bioassessments are the sum of many environmental factors, and not simply the factor of interest (Warren, 1986; Death, 2003). Thus, there is always a level of uncertainty related to the outcome. Streams are heterogeneous systems that change with some degree of predictability (Scarsbrook, 2002). Streams are said to have 'integrity' when they support communities that are balanced and well adapted to local conditions (Karr, 1999). Ecological processes in high integrity streams occur at rates and on scales that support this community (Dale & Beyeler, 2001). Lammert & Allan (1999) found that local habitat variables, including channel morphology and substrate, were better predictors of stream integrity than land use alone, and within-stream conditions can buffer a stream and its inhabitants against disturbances in the watershed.

The Reference Condition Approach

Traditional methods of biological monitoring faced issues of pseudoreplication, as upstream and downstream sites in the same stream were considered comparable and the additive effects of disturbance in a stream were unrecognized (Reynoldson et al., 1997). Reference Condition Approaches (RCA) address the problem of pseudoreplication in traditional methods, such that it is no longer necessary to compare upstream and downstream sites in the same stream. RCA, where potentially disturbed sites are compared against an undisturbed (reference) site, are useful in assessing stream integrity, although one must consider that the reference condition varies over time (Bailey et al., 1998; Scarsbrook, 2002). A stream is considered disturbed when there is a significant difference between measures of integrity for the disturbed and reference sites. When using the RCA it is crucial that the test and reference sites vary as little as possible, in terms of watershed size and topography, vegetation and stream characteristics. In the absence of an *undisturbed* reference site in a particular region, Ganason and Hughes (1998) recommended the use of the least disturbed site in the same region. Alternatively, a model reference site based on habitat requirements or

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historical conditions can be used, although extensive site and community information are required.

One of the primary advantages to using this method is that the difference between the test and reference sites reflects the magnitude of environmental degradation at the potentially disturbed site (Bailey et al., 1998). In addition, providing that reference sites are sampled in every year that assessments are being made, the RCA can be successfully used even when sampling and analytical methods have been modified over time.

Objectives

In this project I attempted to apply the above principles to understand the possible impacts of forest harvesting on low and moderate gradient watersheds of the western boreal forest of Alberta. Weyerhaeuser Canada Ltd. operates on three publicly owned land bases in Alberta, covering approximately 26,000 km^2 of the west-central and northwestern regions of the province. This study developed and tested a procedure for quantifying and monitoring stream health in these regions using a bioassay approach involving benthic macroinvertebrates (BMI). The identification of these impacts may help industry in this region to evaluate their management practices and, if necessary, improve them.

Specifically, the objectives of this study were:
- To quantify any effects of forest harvest on BMI communities and use this information to assess stream integrity in west-central Alberta,
- To quantify the effects of road stream crossings on BMI communities, and
- To review the Benthic Index of Biotic Integrity stream assessment method for its applicability in forested watersheds.

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References

Bailey R.C., Kennedy M.G., Dervish M.Z. & Taylor R.M. (1998) Biological Assessment of Freshwater Ecosystems Using a Reference Condition Approach: Comparing Predicted and Actual Benthic Invertebrate Communities in Yukon Streams. *Freshwater Biology,* 39, 765-774.

Bottorff R.L. & Knight A.W. (1996) The effects of clearcut logging on stream biology of the North Fork of Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA -1986 to 1994 Unpubl. Report prepared for the California Department of Forestry and Fire Protection, Contract No. 8CA3802, May 1996, Sacramento, CA. 177 p.

Campbell I.C. & Doeg T.J. (1989) Impact of timber harvesting and production on streams: A review. *Australian Journal of Marine and Freshwater Research,* 40, 519-539.

Cole M.B., Russell K.R. & Mabee T.J. (2003) Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon Coast Range Mountains. *Canadian Journal of Forest Research,* 33, 1433-1443.

Dale V.H. & Beyeler S.C. (2001) Challenges in the Development and Use of Ecological Indicators. *Ecological Indicators,* 1, 3-10.

Death R.G. (2003) Spatial patterns in lotic invertebrate community compositions: is substrate disturbance actually important? *Canadian Journal of Fisheries and Aquatic Science,* 60, 603-611.

Eberhardt L.L. & Thomas J.M. (1991) Designing environmental field studies. *Ecological Monographs,* 61, 53-73.

Fore L.S., Karr, J.R. & Wisseman, R.W. (1996) Assessing invertebrate responses to human activities: evaluating alternative approaches. *Journal of the North American Benthological Society,* 15, 212-231.

Ganason V. & Hughes R.M. (1998) Application of an Index of Biological Integrity (IBI) to Fish Assemblages of the Rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology,* 40, 367-383.

Heino J., Muotka T., Mykra H., Paavola R., Hamalainen H. & Koskenniemi E. (2003) Defining Macroinvertebrate Assemblage Types in Headwater Streams: Implications for Bioassessment and Conservation. *Ecological Applications,* 13, 842-852.

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Herlihy A.T., Gerth W.J., Li J. & Banks J.L. (2005) Macroinvertebrate community response to natural and forest harvest gradients in western Oregon headwater streams. *Freshwater Biology*, 50, 905-919.

Karr J.R. (1999) Defining and Measuring River Health. *Freshwater Biology*, 41, 221-234.

Lammert M. & Allan J.D. (1999) Assessing Biotic Integrity of Streams: Effects of Scale in Measuring the Influence of Land Use/Cover and Habitat Structure on Fish and Macroinvertebrates. *Environmental Management*, 23, 257-270.

Micacchiou M. (2003) Biological index uses, types, and development. Presented to the National Biological Assessment and Criteria Workshop, Coeur D'Alene, ID, U.S.A. www.epa.gov/waterscience/biocriteria/modules/index101-01-intro.pdf

Morley S.A. & Karr J.R. (2002) Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology*, 16, 1498-1509.

Norris R.H. & Thomas M.C. (1999) What is river health? *Freshwater Biology*, 41, 197-209.

Oksanen L. (2001) Logic of experiments in ecology: is pseudoreplication a pseudoissue? *Oikos*, 94, 27-38.

Paavola R., Muotka T., Virtanen R., Heino J. & Kreivi P. (2003) Are biological classifications of headwater streams concordant across multiple taxonomic groups? *Freshwater Biology*, 48, 1912-1923.

Rosenberg D.M. & Resh V.H. (1993) Introduction to freshwater biomonitoring and benthic macroinvertebrates. Chapman and Hall, New York, U.S.A.

Rothwell R.L. (1983) Erosion and sediment control at road-stream crossings. *The Forestry Chronicle*, 59, 61-66.

Reynoldson T.B., Norris R.H., Resh V.H., Day K.E. & Rosenberg D.M. (1997) The Reference Condition: A Comparison of Multimetric and Multivariate Approaches to Assess Water-Quality Impairment Using Benthic Macroinvertebrates. *Journal of the North American Benthological Society*, 16, 833-852.

Scarsbrook M.R. (2002) Persistence and stability of lotic invertebrate communities in New Zealand. *Freshwater Biology*, 47, 417-431.

Stone M.K. & Wallace J.B. (1998) Long-term recovery of a mountain stream from clear-cut logging: the effects of forest succession on benthic invertebrate community structure. *Freshwater Biology*, 39, 151-169.

Townsend C.R., Downes B.J., Peacock K. & Arbuckle C.J. (2004) Scale and the detection of land-use effects on morphology, vegetation and macroinvertebrate communities of grassland streams. *Freshwater Biology*, 49, 448-462.

Ursic S.J. (1991) Hydrologic effects of two methods of harvesting on mature southern pine. *Water Resources Bulletin*, 27, 303-315.

Warren W.G. (1986) On the presentation of statistical analysis: reason or ritual. *Canadian Journal of Forest Research*, 16, 1185-1191.

Effects of forest harvesting on benthic macroinvertebrate abundance in west-central Alberta, Canada

Introduction

Forest harvesting can simplify benthic communities in downstream watersheds (Benstead et al., 2003). Suspended sediment gathered in overland flow following forest harvesting is detrimental to the health of both aquatic invertebrates and fish (Bottorff & Knight, 1996; Kaller & Hartman, 2004). Excess sediment limits gill function, reduces light available for primary production, overwhelms filtering abilities, and minimizes availability of attachment and habitat sites for invertebrates.

Benthic macroinvertebrate (BMI) abundance is typically higher in harvested watershed streams than in similar unharvested streams (Newbold et al., 1980; Silsbee & Larson, 1983; Bottorff & Knight, 1996; Carlson et al., 1990; Dickes, 1999; Stone & Wallace, 1998; Kedzierski & Smock, 2001; Benstead et al., 2003; Benstead & Pringle, 2004). BMI communities in harvested areas become disturbance adapted and shift to dominance by small, filter-feeding species with high fecundity (Kedzierski & Smock, 2001). Conversely, several authors have found that BMI abundance decreased with increased disturbance intensity and area (Hartman & Scrivener, 1990; McCabe & Gotelli, 2000; Minshall et al., 2001). Other authors have found no difference in BMI abundance between harvested and unharvested sites (Carlson et al., 1990; Growns & Davis, 1991; Liljaniemi et al., 2002; Herlihy et al., 2005). Despite this wide range of findings, the majority of studies seem to support the hypothesis that forest harvesting increases BMI abundance. In general, increased levels of suspended sediment also reduces taxonomic richness, and eliminates or dramatically reduces populations of sensitive taxa, such as Ephemeroptera, Plecoptera and Trichoptera species, and these changes are associated with dramatic increases in burrowing taxa, dominated by dipterans (Bottorff & Knight, 1996).

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Biotic assessments in streams have been based on a range of biota, including fish, BMI and diatoms, and have focused primarily on abundance and diversity in large watersheds (> 100 km^2). When the condition of a large watershed is assessed, a 'dilution effect' may occur (Rothwell, 2003 personal communication); that is, the impact of the disturbance is less spatially concentrated and more difficult to detect. At larger scales, for example, Bunn and Davis (2000) found that community structure of BMI differed at sites, despite similar local site characteristics. By conducting similar studies in smaller watersheds (< 50 km^2) dilution effects are minimized and potentially more accurate impact assessments can be made.

Early stream monitoring programs in North America focused on measures of the physical habitat and water chemistry (Karr, 1981; Fore & Grafe, 2002). The examination and inclusion of biological measures in assessments over the past two decades has brought a wider perspective to stream monitoring and restoration programs. However, there have been few well-analysed and published studies of forest harvesting impacts on low and moderate gradient boreal streams. The objective of this study was to determine if there was a relationship between changes in BMI abundance in representative streams and forest harvesting in the boreal watersheds of west-central Alberta. The results are intended to support assessment of stream integrity in this region. Seasonal variation in BMI abundance was also examined and is reported here.

Methods

Six stream sites were monitored within small watersheds (< 50 km^2, headwater streams); three that were recently harvested and three that remained unharvested for the duration of the study. Harvested and unharvested watersheds were paired and located within each of three Forest Management Areas (FMA) allocated to

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Weyerhaeuser Canada by the Province of Alberta: Edson, Drayton Valley, and Grande Cache (Figure 1).
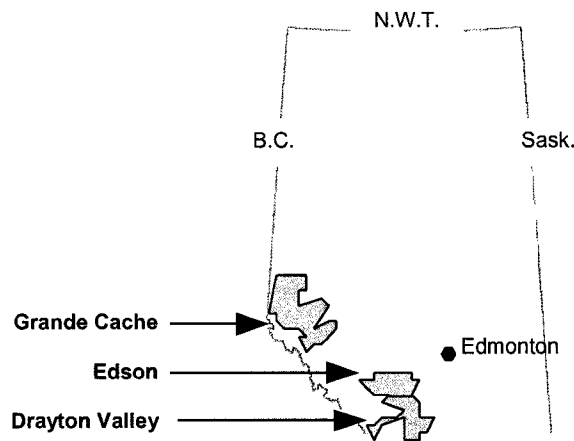
Figure 1. Forest management areas and study site locations at Grande Cache, Drayton Valley and Edson, Alberta.

Stream sites were selected with the aid of Digital Elevation Models (ArcView 3.2, ESRI Inc., Redlands, CA, U.S.A) to ensure similar basin topography and size. Harvested and unharvested watersheds were as evenly paired as possible, based on area, dominant vegetation type, streamflow and substrate (Table 1). Harvested streams were subject to the cumulative impacts of forest harvesting and activities related to oil and gas development. Harvest episodes at these sites occurred within the last one to five years, and covered between 10 and 20 percent of the watershed. Information detailing proximity to harvest was not available.

Site selection to meet the goals of this study was very limited, primarily by harvest schedules and road access, and stream pairs within FMAs inevitably differed to some extent. Based on data collected during 2004 from all three sites, I focused on the streams in the Grande Cache area in 2005. An additional harvested watershed, Kakwa Creek, was located and sampled in 2005 to determine if the generalized pattern of response to disturbance seen during 2004 applied to other sites within the Grande Prairie FMA.

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Table 1. Stream characteristics at representative stream sites in 2004. Location provided in units of latitude and longitude. All qualitative descriptions are relative and apply only to the six stream sites sampled.

	Grande Cache		Drayton Valley		Edson	
Treatment	Unharvested	Harvested	Unharvested	Harvested	Unharvested	Harvested
Stream Name & Location	Sheep 54° 15' N 118° 51' W	Sheep 54° 18' N 118°42'W	Grey Owl 52° 35' N 115° 27' W	Rapid 52° 36' N 115°38'W	Swartz 53° 22' N 116° 27' W	Deer Hill 53° 24' N 116° 28' W
Riparian Vegetation	Mature mixed	Meadow	Mature coniferous	Mature coniferous	Mature deciduous	Meadow
Canopy Cover	30%	0%	40%	20%	20%	0%
Substrate	Cobble, gravel	Cobble, gravel	Cobble, gravel	Boulder, cobble	Gravel, sand	Cobble, gravel
Stream Width	6 m	5 m	8 m	5 m	9 m	3 m
Relative Stream gradient	Moderate	Moderate	Moderate	Moderate	Low	Low

Sampling took place during the same period (July through September 2004 and 2005) in both years to minimize effects of seasonal variation in invertebrate life cycles (Tikkanen et al., 1994). A Surber Sampler (12" diameter, 350 μm) was used to collect BMI at three locations within a 100 m reach at each harvested and unharvested stream site. Multiple locations were sampled at each stream site, as

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streams in these regions are naturally heterogeneous and single samples might not have accurately represented the natural variation in BMI community structure (Reece & Richardson, 2000; Scarsbrook, 2002). Sampling began at the downstream-most locations, to minimize substrate disturbance and reduce sampling errors. Each sample consisted of a 30-second active sampling session, wherein the substrate was disturbed to a depth of 10 cm. Prior to sampling, all large rocks and debris were hand-washed, stimulating the most strongly attached invertebrates to drift into the Surber sample.

A combination of pool and riffle habitats was sampled, as disturbance impacts organisms in these habitats in different ways (Kerans & Karr, 1994). Riffles and pools were selected to be representative of the natural variation in these habitats at each site. Riffles with boulders were avoided due to the well-known differences in community structure between the front and back regions of a boulder, a response to differences in particulate organic matter availability and turbulence (Bouckaert & Davis, 1998). Invertebrates were preserved in the field in 95 percent ethanol and taken to the laboratory, where they were transferred to 80 percent ethanol prior to processing. Sampling and sample processing methods were standardized, as described in Appendices I and II, to minimize the effects of human error and resultant variation on results (Clarke et al., 2002).

BMI were identified to the order level, the lowest taxonomic level possible within the constraints of this study, due to the presence of large numbers of early instar larvae that could not be identified further. Individuals were sorted into five categories: Ephemeroptera, Plecoptera, Trichoptera, Diptera, and "Other" (consisting primarily of non-arthropod groups, including Coleoptera, Collembola and Gastropoda). Following sorting, the total number of individuals in each sample in each order was counted and recorded.

Once Trichoptera were identified as strong indicators of stream integrity in the Grande Cache region, individuals of this order were further identified to the

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Family level, and the total number of individuals in each family was counted and recorded, in so far as possible. Identification of BMI to lower taxonomic levels (genus and species) does not seem to result in significantly different community response patterns than identification at the family or order levels (Bowman & Bailey, 1997). Family level identification is sufficient for samples containing large numbers of early instar larvae because further identification is difficult and does not contribute appropriately to increased accuracy (Reece et al., 2001).

Analysis

Although subsampling and compilation may be statistically acceptable, Kerans and Karr (1994) found that information regarding within-site variation was largely lost when these methods were used, providing an inaccurate representation of BMI community structure. Therefore, all individuals collected in this study were processed and included in analysis. All data were log transformed to normalize distributions.

ANOVAs were used to identify differences in BMI abundance between harvested and unharvested streams, and riffle and pool habitats (SPSS 14.0, SPSS Inc., Chicago, IL, U.S.A.). Repeated measures ANOVAs were used to identify seasonal effects on BMI abundance under the two harvest treatments (harvested and unharvested) (SPSS 14.0). Analysis focused more intensively on the Grande Cache sites, because they demonstrated stronger BMI patterns and responses to disturbance than the Drayton Valley and Edson sites. Indicator Species Analysis (ISA) was used to identify BMI groups that best reflected harvest and habitat effects at Grande Cache (PCOrd, MJM Software Design, Gleneden Beach, OR, U.S.A.), and Non-Metric Multi-Dimensional Scaling (NMDS) was used to ascertain which factors (habitat type or harvest treatment) determined taxon abundance in this region.

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Results

A total of 54,320 individuals were collected from the streams in all three FMAs: 10,432 in Edson; 22,461 in Drayton Valley; 9,437 in Grande Cache (2004); and 11,990 in Grande Cache (2005) (Table 2). The focus of this study shifted to streams and watersheds in the Grande Cache region in 2005 because they were most accessible for sampling and thus offered the most complete data sets.

Table 2. Total number of individuals of each taxa collected at each site.

	Edson 2004	Drayton Valley 2004	Grande Cache 2004	Grande Cache 2005
Ephemeroptera	3,354 (32.2%)	3,689 (16.4%)	3,615 (38.3%)	6,994 (58.3%)
Plecoptera	960 (9.2%)	9,196 (40.9%)	4,088 (43.3%)	2,103 (17.5%)
Trichoptera	791 (7.6%)	922 (4.1%)	134 (1.4%)	282 (2.4%)
Diptera	5,327 (51.1%)	8,654 (38.5%)	1,600 (17.0%)	2,611 (21.8%)
Total	10,432 (100%)	22,461 (100%)	9,437 (100%)	11,990 (100%)

Habitat effects on BMI abundance

Taxa exhibited different habitat associations in each of the three FMAs. Indicator Species Analysis showed that the Ephemeroptera, Plecoptera and Trichoptera were strong indicators of riffle habitats in Grande Cache in 2004 (Ephemeroptera:

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IV=84.7, p<0.001; Plecoptera: IV=86.0, p<0.001; Trichoptera: IV=76.9, p<0.05), although these findings were not supported by ANOVA results. Ephemeroptera were more abundant in riffles at Grande Cache, while Plecoptera and Trichoptera were significantly more abundant in riffles than pools at Edson (Table 3; Figure 2). Diptera abundance did not differ between riffles and pools at any of the sites, and no habitat associations were apparent. Based on these results, sampling in 2005 was restricted to riffle habitats, where BMI were generally most abundant.

Table 3. Differences in BMI abundance between riffles and pool habitats. Significant differences (*) were determined using two-factor ANOVA.

	Ephemeroptera	Plecoptera	Trichoptera	Diptera
Drayton Valley	$F_{1,8}$ = 0.28 p = 0.56	$F_{1,8}$ = 3.31 p = 0.11	$F_{1,8}$ = 0.27 p = 0.56	$F_{1,8}$ = 0.03 p = 0.87
Edson	$F_{1,8}$ = 1.93 p = 0.20	$F_{1,8}$ = 4.95 p = 0.06 *	$F_{1,8}$ = 2.14 p = 0.06 *	$F_{1,8}$ = 0.20 p = 0.60
Grande Cache	$F_{1,12}$ = 3.93 p = 0.07 *	$F_{1,12}$ = 0.95 p = 0.35	$F_{1,12}$ = 1.34 p = 0.41	$F_{1,12}$ = 0.03 p = 0.87
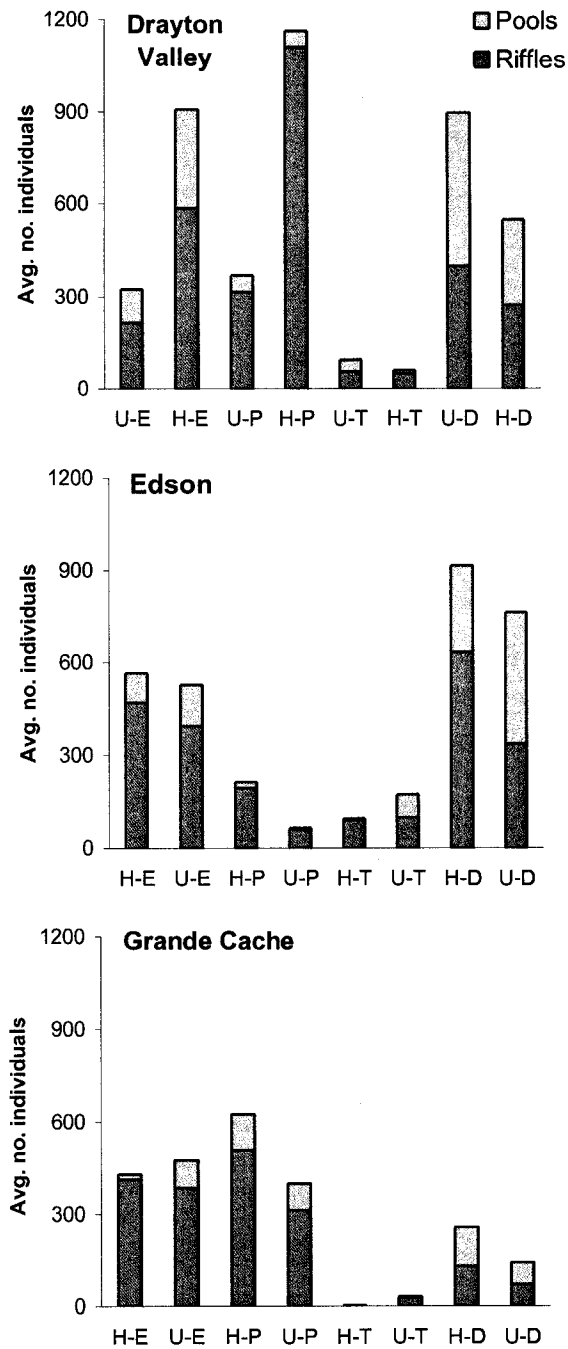
Figure 2. Average BMI abundance in unharvested and harvested riffles and pools at Drayton Valley, Edson and Grande Cache stream sites in 2004 (U: Unharvested, H: Harvested; E: Ephemeroptera, P: Plecoptera, T: Trichoptera, D: Diptera). Darker bars indicate data from riffles and lighter bars indicate data from pools. In general, EPT were more abundant in riffles than in pools, while there was no difference in Diptera abundance between the two habitats.

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In addition to the habitat associations demonstrated above, BMI differed significantly in abundance among the three FMAs (Ephemeroptera ($F_{2, 28} = 4.28$, $p < 0.05$), Plecoptera ($F_{2, 28} = 30.86$, $p < 0.0001$), Trichoptera ($F_{2, 28} = 19.45$, $p < 0.0001$) and Diptera ($F_{2, 28} = 18.56$, $p < 0.0001$)), indicating that watersheds in each FMA supported different communities. In general BMI were most abundant in watersheds in the Drayton Valley FMA and least abundant in the watersheds in the Grande Cache FMA. The relatively greater abundance of Plecoptera in Drayton Valley and lower abundance of Diptera in Grande Cache drove this pattern.

Overall abundance (sum of EPTD in riffles and pools) did not differ between harvested and unharvested BMI populations in 2004 at Grande Cache ($F_{1, 15} = 0.02$, $p = 0.89$), Drayton Valley ($F_{1, 46} = 2.01$, $p = 0.18$) or Edson ($F_{1, 38} = 1.04$, $p = 0.32$). Non-metric Multidimensional Scaling (NMDS) suggested that BMI abundance in these regions varied primarily by habitat type (Figure 3).

Taxa abundance did not differ between harvested and unharvested sites in any of the three FMAs in 2004 (Table 4; Figure 4). However, Indicator Species Analysis (ISA) determined that Trichoptera were strong indicators of unharvested riffles ($IV = 79.5$, $p < 0.01$) in Grande Cache. ISA also indicated that the Family Rhyacophilidae (Trichoptera) was a strong indicator of unharvested riffles ($IV = 88.9$, $p < 0.01$) and this family was the only trichopteran group to successfully distinguish between harvest treatments and habitat types ($F_{1, 32} = 5.09$, $p < 0.05$). Despite statistically non-significant results, strong patterns in EPT abundance were observed in all three FMAs. In Edson, for example, Trichoptera were more abundant at the unharvested sites and Plecoptera were more abundant at the harvested sites (Figure 4). In Drayton Valley, Ephemeroptera were more abundant in the unharvested site. In Grande Cache, Ephemeroptera were more

abundant at the harvested site, while Trichoptera were more abundant at the unharvested site.
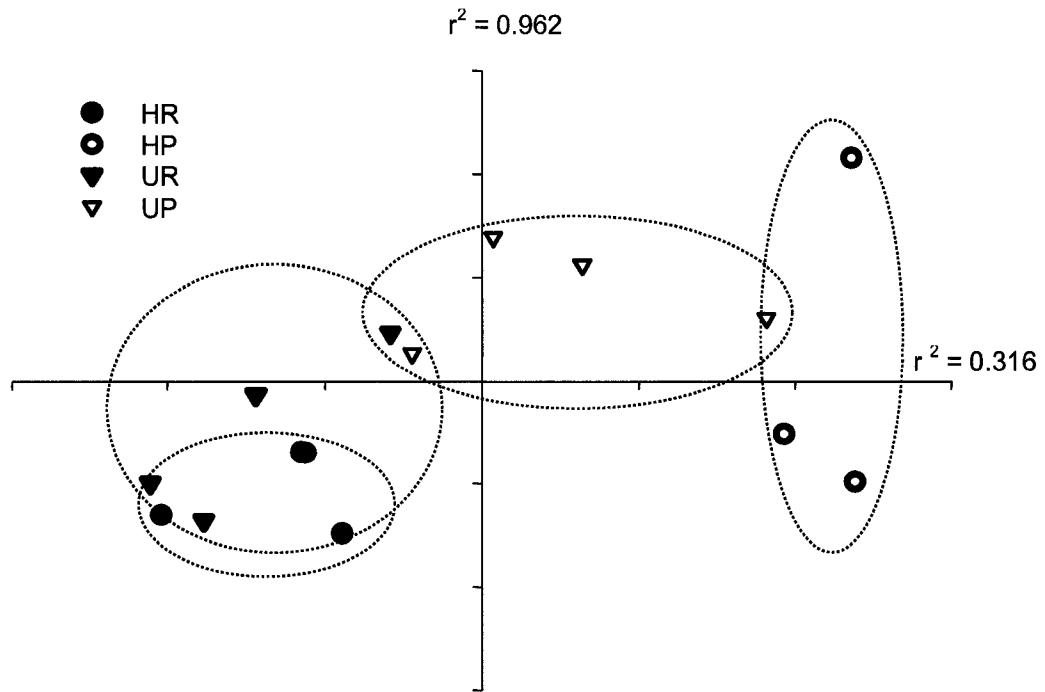
Figure 3. Non-metric multidimensional scaling ordination (using Bray-Curtis distance) indicated that BMI abundance varied primarily by habitat type (riffles (R) vs. pools (P)), rather than by harvest type (harvested (H) vs. unharvested (U)) in Grande Cache.

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Table 4. Differences in BMI abundance between unharvested (U) and harvested (H) sites calculated with two-factor ANOVA. * Indicates significant results, with direction of differences indicated in Figure 4.

	Ephemeroptera	Plecoptera	Trichoptera	Diptera
Drayton Valley (2004)	$F_{1,8} = 0.65$ $p = 0.44$	$F_{1,8} = 0.81$ $p = 0.39$	$F_{1,8} = 0.18$ $p = 0.68$	$F_{1,8} = 0.09$ $p = 0.77$
Edson (2004)	$F_{1,8} = 0.01$ $p = 0.92$	$F_{1,8} = 0.90$ $p = 0.37$	$F_{1,8} = 1.31$ $p = 0.29$	$F_{1,8} = 0.03$ $p = 0.87$
Grande Cache (2004)	$F_{1,12} = 0.28$ $p = 0.59$	$F_{1,12} = 0.17$ $p = 0.69$	$F_{1,12} = 1.95$ $p = 0.19$	$F_{1,12} = 0.09$ $p = 0.77$
Grande Cache (2005)	$F_{1,8} = 3.07$ $p = 0.12$	$F_{1,8} = 33.31$ $p < 0.001$ * U > H	$F_{1,8} = 15.35$ $p < 0.05$ * U > H	$F_{1,8} = 22.56$ $p < 0.001$ * U < H

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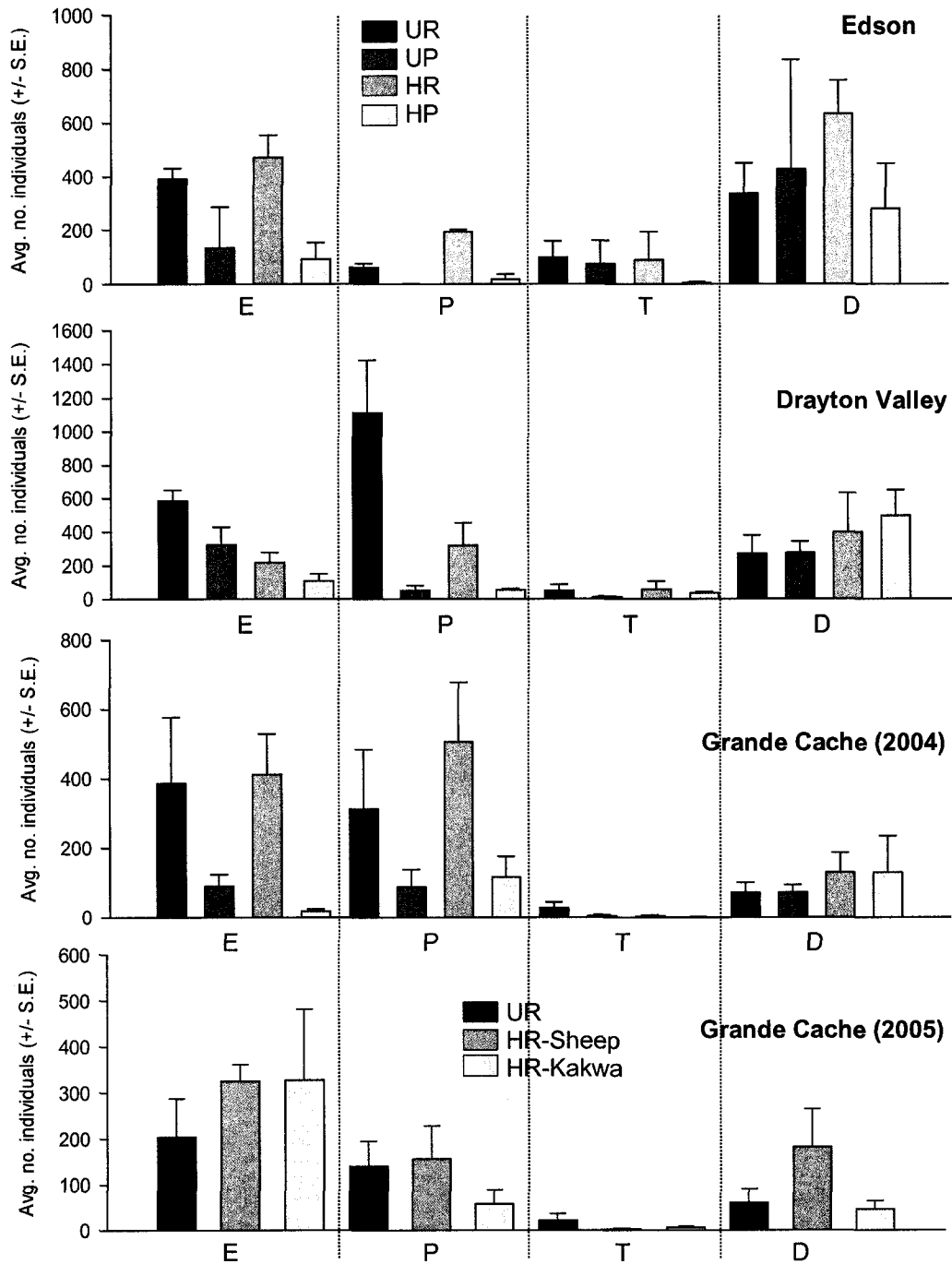
Figure 4. BMI abundance in unharvested and harvested streams (HR: Harvested Riffles, HP: Harvested Pools, UR: Unharvested Riffles, Unharvested Pools; E: Ephemeroptera, P: Plecoptera, T: Trichoptera and D: Diptera). Note that abundance for 3 FMAs presented on different scales.

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Several significant differences in abundance were observed at the Grande Cache sites between 2004 and 2005. Ephemeroptera were more abundant in the 2005 samples ($F_{1,28} = 10.19$, $p < 0.005$), while Plecoptera were more abundant in the 2004 samples ($F_{1,28} = 8.58$, $p < 0.01$). Harvest treatment influenced differences in Trichoptera and Diptera abundance between years, with Trichoptera more abundant in the unharvested sites in 2005 ($F_{1,28} = 35.53$, $p < 0.001$) and Diptera more abundant in the harvested sites in 2005 ($F_{1,28} = 8.30$, $p < 0.01$).

Harvest Effects on BMI abundance at Grande Cache (2005)

Based on the observed patterns in BMI abundance at Grande Cache in 2004 and access to an additional harvested site, I focused sampling exclusively on riffle habitats in the Grande Cache FMA in 2005. BMI abundance clearly differed between harvested and unharvested sites in this FMA in 2005 (Table 4; Figure 4).

However, there were significant differences in Plecoptera ($F_{1,16} = 19.34$, $p < 0.001$), Trichoptera ($F_{1,16} = 12.19$, $p < 0.01$) and Diptera ($F_{1,16} = 27.35$, $p < 0.0001$) abundance between the harvested Sheep and Kakwa Creek sites. In some instances, BMI abundance at the Kakwa Creek site more closely resembled abundances at the unharvested Sheep Creek site, weakening confidence in the general relationship between forest harvesting and BMI hypothesized based on the 2004 results.

Ephemeropteran abundance did not differ between the harvested sites ($F_{1,16} = 1.11$, $p = 0.31$), and Plecoptera were more abundant in unharvested sites, although their abundance was significantly greater at the Kakwa Creek harvested site ($F_{1,16} = 19.37$, $p < .001$) (Table 4; Figure 4). Trichoptera were significantly less abundant at the harvested sites ($F_{1,16} = 12.19$, $p < .005$), and Diptera were more abundant at the Sheep Creek harvested site ($F_{1,16} = 27.35$, $p = .001$).

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BMI abundance varied seasonally at Grande Cache. In 2004 the only taxa to exhibit significant differences in abundance from August to September were the Ephemeroptera at both the harvested and unharvested sites, and the Plecoptera at the unharvested site (Table 5; Figure 5); however, in 2005 the Ephemeroptera did not exhibit seasonal differences in abundance. Instead, during 2005, Plecoptera, Trichoptera and Diptera abundance changed with the sampling period at the harvested sites. Plecoptera were more abundant at later sampling dates (September), Trichoptera were more abundant at earlier sampling dates (August), and Diptera were significantly less abundant in mid-August than in early August or September. Dipteran abundance was significantly lower in late August at the unharvested site.

Table 5. Difference in BMI abundance between August and September sampling dates in Grande Cache, Alberta. Results calculated by two-factor ANOVA. * Indicates significant results, with direction of differences indicated in Figure 5.

	Ephemeroptera	Plecoptera	Trichoptera	Diptera
2004 Unharvested	$F_{1,4} = 56.98$ $p < 0.001$ *	$F_{1,4} = 6.48$ $p = 0.08$ *	$F_{1,4} = 0.77$ $p = 0.43$	$F_{1,4} = 0.15$ $p = 0.72$
2004 Harvested	$F_{1,4} = 20.47$ $p < 0.01$ *	$F_{1,4} = 1.55$ $p = 0.28$	$F_{1,4} = 3.23$ $p = 0.15$	$F_{1,4} = 0.59$ $p = 0.49$
2005 Unharvested	$F_{2,24} = 0.57$ $p = 0.57$	$F_{2,24} = 2.59$ $p = 0.10$	$F_{2,24} = 0.96$ $p = 0.40$	$F_{2,24} = 14.88$ $p < 0.005$
2005 Harvested	$F_{2,24} = 0.50$ $p = 0.61$	$F_{2,24} = 3.42$ $p < 0.05$ *	$F_{2,24} = 6.07$ $p < 0.01$ *	$F_{2,24} = 5.13$ $p < 0.01$ *
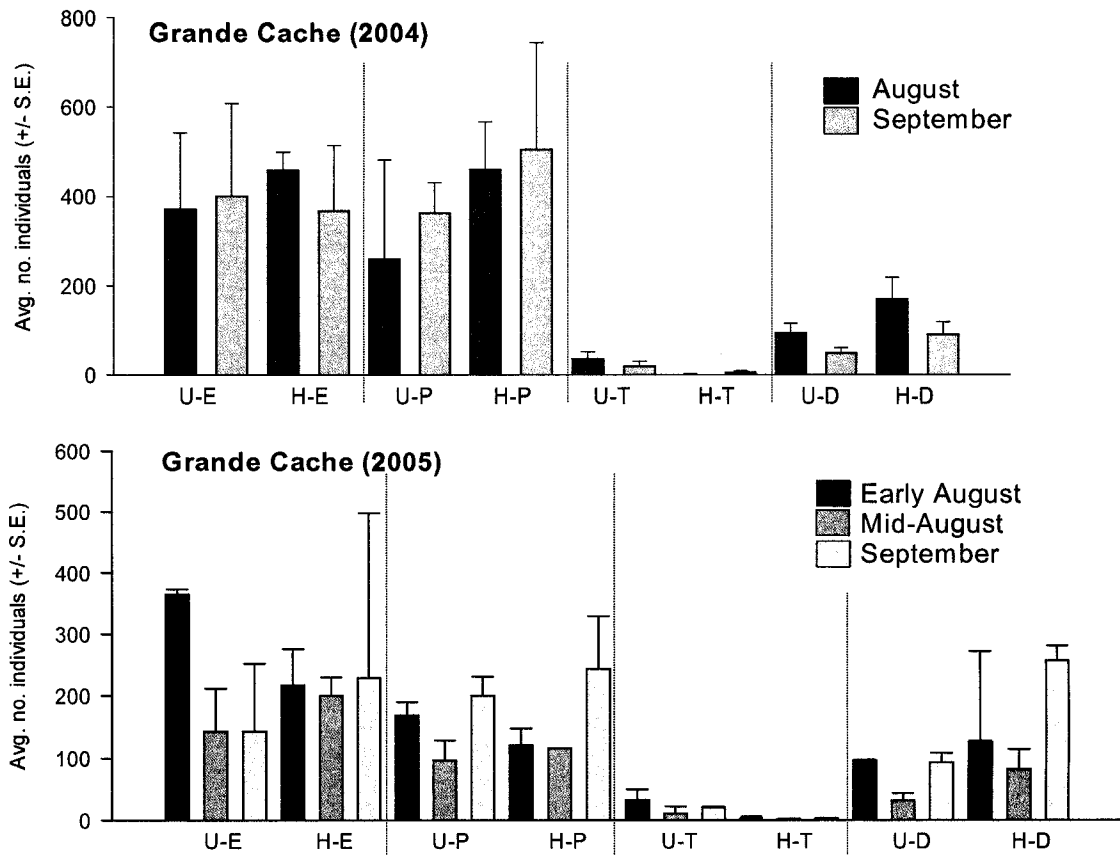
Figure 5. Seasonal effects of sampling on BMI abundance at the Sheep Creek tributaries of Grande Cache (U: Unharvested site, H: Harvested site; E: Ephemeroptera, P: Plecoptera, T: Trichoptera, D: Diptera). BMI abundance differed in both years, although seasonal abundance of each taxa differed significantly in one of the two years.

Discussion

Community structure in undisturbed streams is highly predictable, as local habitat conditions determine species presence and persistence (Heino et al., 2003). However, undisturbed streams are subject to high levels of natural variation and often have complex communities (Heino et al., 2003; Nijboer et al., 2005). BMI persistence in disturbed streams requires higher species tolerance and suitable adaptations to disturbed conditions. Because disturbance affects watersheds differently, due to differences in topography, geology, vegetation and flow, it can be difficult to predict which species will persist in these streams. BMI abundance

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has proven to be a reliable indicator of forestry-induced disturbance in many streams, while richness and diversity of BMI assemblages are better used as indicators of pollution (Liljaniemi et al., 2002).

Habitat associations observed for EPT

Statistical techniques are employed in biological data analysis to identify trends and patterns, and prevent users from drawing unsupported conclusions (Marshall & Elliot, 1997). However, significant differences can always be found between two groups, providing they are examined in sufficient detail (Eberhardt & Thomas, 1991; Oksanen, 2001; Townsend et al., 2004). Alternately, when the same groups are examined in less detail, these differences are not always apparent or statistically significant. Although these differences may become apparent with increased sample size or resolution, the lack of statistical significance should not imply that observed patterns are uninformative. The patterns may, in fact, be biologically significant and relevant.

In this study, a greater abundance of Ephemeroptera, Plecoptera and Trichoptera in riffle habitats was observed, despite the absence of statistically significant differences between the riffle and pool habitats. Differences in abundance between riffles and pools were likely driven by variations in oxygen and attachment site availability, streamflow, sediment deposition rates, or a variety of other taxon-specific habitat preferences (Plotnikoff & Ehinger, 1997). This pattern served as the basis for the decision to focus sampling in 2005 exclusively on the riffle habitats, which served to reduce data noise.

Diptera were not disproportionately collected in either riffles or pools. Aquatic representatives of this order are filter feeders and sediment burrowers, and thus tend to thrive in areas where fine organic matter and sediment accumulate (Clifford, 1991). Pools are generally the preferred habitat of this taxon, because they are dominated by fine-grained sediments, due to low streamflow and high

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deposition rates. Streamflow regimes may have been a major factor influencing abundance in these regions. Diptera were more generally abundant in the lower gradient streams of Edson and Drayton Valley, than in the higher gradient streams of Grande Cache. Streamflow in the higher gradient streams may have been sufficient to flush sediment from the stream fairly quickly, limiting sediment and organic matter deposition in riffles and pools, and thus suppressing populations of dipteran species. Any Diptera easily collected by my sampling technique in the high flow riffle habitats of Grande Cache would have thus been well adapted to local conditions, allowing for their persistence at levels similar to those in pools. Conversely, in the lower gradient streams of Edson and Drayton Valley, streamflow in the riffles may have been slow enough to allow for sediment build-up in riffles and to encourage dipteran populations similar to those observed in the pools. BMI adapt to local conditions and constant habitat conditions, whether in riffles or pools, promote persistence of particular BMI communities (Scarsbrook, 2002).

Harvest effects on BMI abundance

One basic assumption anchors this study. It was assumed that forest harvesting was of sufficient intensity at harvested watershed sites to have detectable influences on BMI communities in these regions. Biotic effects of harvest, such as changes in abundance or richness, tend to indicate the final stage of environmental degradation (Norris & Thomas 1999). Therefore, when the full potential of a disturbance event to impact stream biota has not been reached, it is possible for a healthy BMI community to exist in a disturbed watershed.

It was not surprising that BMI communities differed among the three FMAs because they are known to be correlated with both local habitat conditions, including substrate size, and landscape scale conditions, including stream size and catchment area (Morley & Karr, 2002; Paavola et al., 2003; Herlihy et al., 2005). Riparian vegetation and stream gradient also influence BMI abundance (Murphy

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& Giller, 2000; Kedzierski & Smock, 2001). Stream recovery post-harvest is slower in low gradient streams (Growns & Davis, 1991) and organisms in these streams are subject to lesser flow impacts (Holomuzki & Biggs, 1999). The Edson streams were relatively wide and shallow, and were located in low gradient, deciduous-dominated watersheds, and were thus dominated by slower flows and finer-grained substrates. Allochthonous inputs are higher in deciduous watersheds, promoting the abundance of taxa in the 'shredder' functional feeding group (Murphy & Giller, 2000). However, restriction of sampling to riffles and pools in Edson may have lead to the underestimation of BMI abundance in these streams, as BMI in low gradient streams tend to accumulate in high sediment areas, such as debris dams (Roeding & Smock, 1989; Jones & Smock, 1991).

Conversely, the Grande Cache streams were deeper and narrower, and were conifer-dominated, higher gradient streams with coarse substrates. Communities in high gradient streams are highly influenced by stream temperature, sediment input, increased primary production and decreased allochthonous input (Kedzierski & Smock, 2001). BMI in such watersheds must be well adapted to lower water temperatures and decreased sediment deposition rates. Allochthonous inputs in these fast-flowing streams are quickly flushed downstream, and BMI must match their feeding habits to take advantage of higher rates of primary production.

The Drayton Valley streams, also conifer-dominated, fell somewhere between the Grande Cache and Edson streams in terms of streamflow and substrate. Further statistical comparisons of the FMAs would have been relatively uninformative in the context of this study, due to the high level of variation between their watersheds in terms of stream gradient and vegetation cover.

Differences in community composition between harvested and unharvested streams can often be attributed to changes in the relative abundance of taxa, and not presence or absence (Death, 2003). In many cases, BMI abundance, most

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notably that of taxa in Edson and Drayton Valley, did not differ between harvested and unharvested streams. Ephemeroptera, Plecoptera and Trichoptera (EPT) are considered sensitive to disturbance and habitat degradation, and generally respond with decreased abundance (Morley & Karr, 2002; Benstead et al., 2003; Yuan & Norton, 2003; Quinn et al., 2004). In west-central Alberta, significant harvesting impacts on BMI communities were observed only in the Grande Cache 2005 samples. However, several patterns were observed. While the Ephemeroptera were less abundant and more sensitive in the harvested stream at Drayton Valley, they were more abundant and thus appeared to be more tolerant in the harvested site at Grande Cache. Plecoptera were also more abundant in the harvested stream at Edson. These patterns were not statistically significant but were dramatic enough that further sampling and taxonomic resolution may have made them so.

The lack of statistically significant differences in BMI abundance between harvested and unharvested sites might be attributed to the low level of taxonomic resolution used in this study. Because of the wide range of larval instars encountered in my samples, identification of all individuals to taxonomic levels beyond the order level was beyond the constraints of this project. However, further identification may have revealed differences in the taxonomic makeup of the Ephemeroptera communities. Species within the Ephemeroptera exhibit a wide range of tolerances to disturbance. In particular, the genus *Baetis* includes highly tolerant and resilient species that are often more abundant in harvested watersheds. Minshall et al. (2001) observed that chironomids (Diptera) and *Baetis sp.* comprised up to 60 percent of post-disturbance stream invertebrate assemblages, as opposed to accounting for only 18 percent of BMI in undisturbed streams. Baetids dominate in harvested streams because of their ability to take advantage of in situ production, rapid colonization rates and short generation times, which make them highly adaptable (Newbold et al., 1980; Benstead et al., 2003). Therefore, tolerant ephemeropteran species may have dominated harvested watersheds in Grande Cache, resulting in their increased abundance,

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while more sensitive species may have dominated at Drayton Valley, leading to their increased abundance in the unharvested stream. Similar explanations may apply to the higher abundance of the typically sensitive Plecoptera in the harvested stream at Edson.

The level of taxonomic resolution necessary to accurately infer environmental impact in freshwater streams is debatable (Reynoldson et al., 2001). Bowman and Bailey (1997) recommended sacrificing taxonomic resolution, rather than quantitative data, when trade-offs were necessary. For the purpose of stream monitoring, identification to the generic level does not appear to result in significantly different community response patterns than identification to family or order levels (Bailey et al., 2001). However, analysis using lower levels of taxonomic resolution reduces the amount of variability so that disturbance effects and their driving mechanisms may be more apparent. Identification to the order level (and to family for Trichoptera in Grande Cache) was necessary, due to the large number of unidentifiable early instars, whose removal would have dramatically reduced the number of individuals included in analysis and skewed observed response patterns (Reece et al., 2001).

Overall, PTD abundance in the Grande Cache FMA in 2005 was affected by forest harvesting and related practices, and harvested watershed streams might thus be considered in relatively poor condition. Natural variation between sample sites in a stream can obscure differences in abundance between harvested and unharvested sites (Growns & Davis, 1991; Townsend et al., 2004), although the differences between the streams in this FMA were sufficient to override the effects of this variation. The addition of the second (Kakwa Creek) harvested stream site in the Grande Cache area was an attempt to limit the potential of falsely inferring a harvest effect, when in fact between-site differences were merely coincidental with natural patterns (Townsend et al., 2004). However, the BMI of the Kakwa Creek sites often more closely resembled those of the unharvested Sheep Creek tributary than they did the BMI community of the

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harvested Sheep Creek tributary. Most dramatically, dipteran abundance was greatest at the harvested Sheep Creek stream while their abundance did not differ between the unharvested Sheep Creek and harvested Kakwa Creek streams. Natural variability will thus be problematic when comparing streams that differ by more than the factor of interest, in this case, the presence or absence of harvesting (Death, 2003). Therefore, it is difficult to establish a strong cause-effect link between harvesting and observed changes in community structure. Reliable assessments of stream condition rely heavily on the strong pairing of reference and disturbed sites, particularly in highly variable watersheds. Unfortunately, due to the limitations imposed by site selection, proper pairing of reference and disturbed sites was not always possible and characterizes well the results at Grande Cache.

BMI abundance differed between years at Grande Cache. Comparisons of abundance between these two years could not reasonably be used to draw conclusions about harvest effects. Abundance data from each year were treated as independent observations of community composition, because I had insufficient information of temporal variability in this region to infer a link between the events of both years. Although abundance patterns in each year may be linked, the cause of the variation could not be identified and it was best to err on the side of caution. The Grande Cache region is topographically diverse and subject to both natural and anthropogenic disturbance events of differing magnitudes and frequencies. As a result, factors influencing BMI abundance likely differ between years and it is unlikely that abundance between two consecutive years will be similar. However, over a period of a decade or more BMI communities in undisturbed watersheds are generally stable (Scarsbrook, 2002).

Trichoptera as indicators of stream integrity

Ecological indicators are used as early warning signals of environmental change and are intended to reflect critical information about the biotic composition and

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function of the system in question (Dale & Beyeler, 2001). Ideal indicator taxa will respond predictably to stress, while exhibiting limited sensitivity to natural variation (Karr, 1991; Dale & Beyeler, 2001). While most indicator species are hardy and disturbance-adapted, the use of rare and sensitive taxa might be more ecologically relevant (Turak & Koop, 2003). Sensitive taxa can be indicative of smaller, less dramatic environmental changes and could provide sufficient time to initiate and implement appropriate actions before irreversible damage is done. Trichopterans were the most sensitive taxa in the Grande Cache streams and could be used as early indicators of change and measures of stream integrity in harvested watersheds in this FMA.

Members of the family Rhyacophilidae (Trichoptera) are typically found in fast-flowing, unpolluted streams, such as the unharvested stream site in Grande Cache (Elliot, 2005). Rhyacophilids were strong indicators of unharvested stream sites in Grande Cache, which suggested that these were sites of high integrity. However, it seems reasonable to use overall Trichoptera abundance as a measure of stream integrity in this region, as it requires less taxonomic expertise for sample processing and is equally as informative as rhyacophilid abundance.

No indicator taxon can be applied successfully across all regions and spatial scales (Dale & Beyeler, 2001). Streams are complex, heterogeneous systems, and dependence on a relatively small number of indicator taxa can lead to a misguided simplification of the system under observation (Stone & Wallace, 1998; Cao et al., 2003; Nijboer et al., 2005). However, harvesting can impact BMI communities for decades following initial harvest, and population recovery rates vary widely between species (Campbell & Doeg, 1989; Growns & Davis, 1991; Stone & Wallace, 1998; Cole et al., 2003; Herlihy et al., 2005). While the recovery of more tolerant taxa may be underway, sensitive taxa may still be in a post-disturbance suppression state. Therefore, the use of multiple taxa to assess stream integrity is not always informative, and may be misleading, particularly when that information is amassed in an index. The use of a single sensitive

attribute, such as trichopteran abundance, as an indicator of change can be useful for determining whether further, more intensive investigation is desirable. Results based on abundance of an indicator taxon should be used as a starting point from which to determine if further action (including detailed assessment) is necessary. This provides for effective monitoring that is less costly, both in time and resources, than measuring entire community structures.

Drivers of BMI abundance patterns

Establishing causal mechanisms to explain faunal dynamics in natural stream systems is difficult because streams vary by more than the disturbance factor (Death, 2003). Land use is not a generally strong predictor of stream integrity (Lammert & Allan, 1999). Rather, local habitat conditions, including stream morphology and substrate, are better predictors. In addition, riparian vegetation, watershed geology, topography, sediment levels and streamflow regimes can all have dramatic effects on streams and their communities (Carlson et al., 1990; Growns & Davis, 1991; Cole et al., 2003; Death, 2003). For example, streams of harvested watersheds typically have higher levels of suspended sediments and nutrients (Silsbee & Larson, 1983; Campbell & Doeg, 1989), to which EPT abundance is negatively correlated (Cole et al., 2003).

Several studies have suggested that the mechanism driving the shift in community structure with harvesting is a change in the availability of food resources, rather than abiotic processes, such as sedimentation and habitat degradation (Newbold et al., 1980; Fuchs et al., 2003; Benstead & Pringle, 2004). Harvested watershed streams shift from an allochthonous to autochthonous energy base (Campbell & Doeg, 1989; Stone & Wallace, 1998; Morley & Karr, 2002; Herlihy et al., 2005) and such shifts in the post-harvest food base can lead to a two- to six-fold increase in overall BMI abundance (Herlihy et al., 2005). However, not all taxa in Grande Cache, Drayton Valley and Edson reflected this pattern of increased abundance with harvesting and thus, other drivers were involved.

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In my study, as in many others, it appears that riparian vegetation, present along both harvested and unharvested streams, acted as a buffer sufficient to mitigate the negative impacts of forest harvesting, including changes in basal resource availability, on the abundance of some taxa (Newbold et al., 1980; Growns & Davis, 1991; Lammert & Allan, 1999; Quinn et al., 2004; Herlihy et al., 2005). Riparian vegetation limits sedimentation, supplies terrestrial carbon, maintains stream temperatures and regulates primary production (Newbold et al., 1980; Campbell & Doeg, 1989; Carlson et al., 1990; Hartman & Scrivener, 1990; Nerbonne & Voudracek, 2001; Benstead & Pringle, 2004). Differences in riparian vegetation type, mature forest at unharvested sites and low lying shrubs and grasses at harvested sites did not appear to affect BMI abundance (Nerbonne & Voudracek, 2001). However, several similar studies have observed that buffer width and stream integrity were uncorrelated for low levels of disturbance (Bunn & Davis, 2000; Morley & Karr, 2002) and that changes in BMI community structure could only be identified in situations of moderate-intensity harvesting (Kreutzweiser et al., 2005). It is possible, despite my assumption of the reverse, that harvesting in all three FMAs considered in my study was of insufficient intensity to negatively impact most taxa, with only the most sensitive taxa, such as the Trichoptera, being affected at most sites.

Behavioural adaptations allow various taxa to maintain their populations in disturbed watersheds. Many species that are abundant in disturbed watersheds are highly adaptable to environmental conditions and can respond to forest harvesting by altering their feeding habits (Liljaniemi et al., 2002). Harvesting can result in increased flow volumes and decreased lag times between peak flow events (Bottorff & Knight, 1996). Invertebrates exhibit a variety of adaptations to these events, including moving to bank-side refugia (ex. Borchardt, 1993) and entering dormant life stages (e.g., Lamouroux et al., 2004). BMI are well adapted to strong currents, although very few inhabit streams with extreme or unpredictable flow (Morley & Karr, 2002). Cobb et al. (1992) observed a 90 percent reduction

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in invertebrate density immediately following flood events. The number and magnitude of extreme flow events during the sampling period may have impacted BMI abundance in the sampled streams, independent of forest harvesting. In such cases, it must be assumed that the harvested and unharvested sites in each FMA were similarly affected by the extreme event and were appropriately paired for analyses.

Seasonal effects of sampling on BMI communities

Strong seasonal effects on BMI abundance can obscure the detection of disturbance effects (Reece et al., 2001). Karr (1999), for example, recommended that all samples be taken in a period of less than one month. However, in the case of a pseudoreplicated experiment, such as this one, sampling dates were interspersed such that some degree of recovery from the negative effects of sampling (e.g., substrate disturbance during sampling, removal of entire BMI community in the sampled area) could occur as the sample site returned to its pre-sampled status.

Wider ranges in water temperatures may have driven seasonal differences in BMI abundance at the harvested site in Grande Cache. Riparian vegetation insulates streams and maintains water temperatures within a range suitable to BMI persistence. Streams with little canopy cover, such as those at the harvested sites, experience a wider range of daily and seasonal temperatures (Campbell & Doeg, 1989; Benstead et al., 2003). Lower water temperatures can depress BMI populations and their food supplies, resulting in lower BMI abundances (Carlson et al., 1990). Thus, the lack of overhanging canopy at the harvested streams may have allowed for increased daytime and summer water temperatures, promoting increased BMI abundance in some of these streams.

If Trichoptera are to be used as indicator taxa for streams in the Grande Cache area, as previously suggested, it is crucial that differences in their abundance

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reflect changing patterns of land use and not seasonal differences. Trichopteran species exhibit a wide range of feeding habits, which are strongly influenced by the seasonally affected riparian vegetation. Therefore, it is recommended that all sampling of all taxa take place within no more than a one-month period. Ideally, single samples taken from a greater number of sites, along one or multiple streams, over a shorter period of time should be used for future assessments.

Summary

In conclusion, stream gradients and riparian vegetation appeared to strongly influence BMI abundance in the boreal streams of west-central Alberta. Natural variation among streams and watersheds in three different FMAs, including watershed topography and streamflow regimes, was a major factor influencing BMI abundance in this region and limited the generality of BMI indicators of harvesting. Ephemeroptera, Plecoptera and Trichoptera were much more commonly collected in riffle habitats than in pools, and future studies in this region could save time and resources by focusing on the riffle fauna exclusively. Because most taxa exhibited seasonal differences in abundance in disturbed watersheds BMI sampling intended to monitor stream condition should be conducted within a one-month period. Trichoptera were sensitive to harvest effects, and appear to be valuable as initial indicators of stream integrity in the Grand Cache region. Order level resolution was sufficient to identify disturbance effects in this taxon, although further resolution might further clarify these effects. Although Ephemeroptera and Plecoptera were abundant in the streams sampled their use as order-level indicators cannot be recommended by this work. Further study at more detailed taxonomic levels may reveal patterns that are useful for identifying indicators of disturbance.

References

Bailey R.C., Norris R.H. & Reynoldson T.B. (2001) Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. *Journal of the North American Benthological Society*, 20, 280-286.

Benstead J.P., Douglas M.M. & Pringle C.M. (2003) Relationships of stream invertebrate communities to deforestation in eastern Madagascar. *Ecological Applications*, 13, 1473-1490.

Benstead J.P. & Pringle C.M. (2004) Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. *Freshwater Biology*, 49, 490-501.

Borchardt D. (1993) Effects of flow and regufia on drift loss in benthic macroinvertebrates: implications for habitat restoration in lowland streams. *Freshwater Biology*, 29, 221-227.

Bottorff R.L. & Knight A.W. (1996) The effects of clearcut logging on stream biology of the North Fork of Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA -1986 to 1994 Unpubl. Report prepared for the California Department of Forestry and Fire Protection, Contract No. 8CA3802, May 1996, Sacramento, CA. 177 p.

Bouckaert F.W. & Davis A.J. (1998) Microflow regimes and the distribution of macroinvertebrates around stream boulders. *Freshwater Biology* 40, 77-86.

Bowman M.F. & Bailey R.C. (1997) Does taxonomic resolution affect the multivariate description of the structure of freshwater benthic macroinvertebrate communities? *Canadian Journal of Fisheries and Aquatic Science*, 54, 1802-1807.

Bunn S.E. & Davis P.M. (2000) Biological processes in running waters and their implications for the assessment of ecological integrity. *Hydrobiologia*, 422, 61-70.

Cao Y., Hawkins C.P. & Vinson M.R. (2003) Measuring and controlling data quality in biological assemblage surveys with special reference to stream benthic macroinvertebrates. *Freshwater Biology*, 48, 1898-1911.

Campbell I.C. & Doeg T.J. (1989) Impact of timber harvesting and production on streams: A review. *Australian Journal of Marine and Freshwater Research*, 40, 519-539.

Carlson J.Y., Andrus C.W. & Froelich H.A. (1990) Woody debris, channel features, and macroinvertebrates of streams with logged and undisturbed riparian

timber in northeastern Oregon, USA. *Canadian Journal of Fisheries and Aquatic Sciences,* 47, 1103-1111.

Clarke R.T., Furse M.T., Gunn R.J.M., Winder J.M. & Wright J.F. (2002) Sampling variation in macroinvertebrate data and implications for river quality indices. *Freshwater Biology,* 47, 1735-1751.

Clifford H.F. (1991) *Aquatic Invertebrates of Alberta.* The University of Alberta Press, Edmonton, AB, Canada.

Cobb D.G., Galloway T.D. & Flannagan J.F. (1992) Effects of discharge and substrate stability on density and species composition of stream insects. *Canadian Journal of Fisheries and Aquatic Science,* 49, 1788-1795.

Cole M.B., Russell K.R. & Mabee T.J. (2003) Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon Coast Range Mountains. *Canadian Journal of Forest Research,* 33, 1433-1443.

Dale V.H. & Beyeler S.C. (2001) Challenges in the development and use of ecological indicators. *Ecological Indicators,* 1, 3-10.

Death R.G. (2003) Spatial patterns in lotic invertebrate community compositions: is substrate disturbance actually important? *Canadian Journal of Fisheries and Aquatic Science,* 60, 603-611.

Dickes B. (1999) McDonald Creek benthic macroinvertebrate study: a cumulative impact assessment after timber harvesting in a small watershed. Washington State Department of Ecology Publication, 99-304.

Elliot J.M. (2005) Contrasting diel activity and feeding patterns of four instars of *Rhyacophila dorsalis* (Trichoptera). *Freshwater Biology,* 50, 1022-1033.

Fore L.S. & Grafe C. (2002) Biological conditions of large rivers in Idaho (U.S.A.). *Freshwater Biology,* 47, 2015-2037.

Fuchs S.A., Hinch S.G. & Mellina E. (2003) Effects of streamside logging on stream macroinvertebrate communities and habitat in the sub-boreal forests of British Columbia, Canada. *Canadian Journal of Forest Research,* 33, 1408-1415.

Growns I.O. & Davis J.A. (1991) Comparison of the macroinvertebrate communities in streams in logged and undisturbed catchments 8 years after harvesting. *Australian Journal of Marine and Freshwater Research,* 42, 689-706.

Hartman C.F. & Scrivener J.C. (1990) Impacts of Forestry Practices on a Coastal Stream Ecosystem, Carnation Creek, British Columbia. *Canadian Bulletin of Fisheries and Aquatic Sciences*, 223.

Heino J., Muotka T., Mykra H., Paavola R., Hamalainen H. & Koskenniemi E. (2003) Defining Macroinvertebrate Assemblage Types in Headwater Streams: Implications for Bioassessment and Conservation. *Ecological Applications,* 13, 842-852.

Herlihy A.T., Gerth W.J., Li J. & Banks J.L. (2005) Macroinvertebrate community response to natural and forest harvest gradients in western Oregon headwater streams. *Freshwater Biology*, 50, 905-919.

Holomuzki J.R & Biggs B.J.F. (1999) Distributional responses to flow disturbance by a stream-dwelling snail. *Oikos*, 87, 36-47.

Jones J.B.Jr. & Smock L.A. (1991) Transport and retention of particulate organic matter in two low gradient headwater streams. *Journal of the North American Benthological Society*, 10, 115-126.

Karr J.R. (1981) Assessment of Biotic Integrity Using Fish Communities. *Fisheries,* 6, 21-27.

Karr J.R. (1991) Biological integrity: A long-neglected aspect of water resource management. *Ecological Applications*, 1, 66-84.

Karr J.R. (1999) Defining and Measuring River Health. *Freshwater Biology,* 41, 221-234.

Kedzierski W.M. & Smock L.A. (2001) Effects of Logging on Macroinvertebrate Production in a Sand-Bottomed, Low-Gradient Stream. *Freshwater Biology,* 46, 821-833.

Kerans B.L. & Karr J.R. (1994) A Benthic Index of Biotic Integrity (B-IBI) for Rivers of the Tennessee Valley. *Ecological Applications,* 4, 768-785.

Kaller M.D. & Hartman K.J. (2004) Evidence of a threshold level of fine sediment accumulation for altering benthic macroinvertebrate communities. *Hydrobiologia*, 518, 95-104.

Kreutzweiser D.P., Capell S.S. & Good K.P. (2005) Macroinvertebrate community responses to selection logging in riparian and upland areas of headwater catchments in a northern hardwood forest. *Journal of the North American Benthological Society*, 24, 208-222.

Lammert M. & Allan J.D. (1999) Assessing Biotic Integrity of Streams: Effects of Scale in Measuring the Influence of Land Use/Cover and Habitat Structure on Fish and Macroinvertebrates. *Environmental Management,* 23, 257-270.

Lamouroux N., Doledec S. & Gayraud S. (2004) Biological traits of stream macroinvertebrate communities: effects of microhabitat, reach, and basin filters. *Journal of the North American Benthological Society,* 23, 449-466.

Liljaniemi P., Vuori K., Ilyashuk B. & Luotonen H. (2002) Habitat characteristics and macroinvertebrate assemblages in boreal forest streams: relations to catchment silvicultural activities. *Hydrobiologia,* 474, 239-251.

McCabe D.J. & Gotelli N.J. (2000) Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. *Oecologia,* 124, 270-279.

Minshall G.W., Royer T.V. & Robinson C.T. (2001) Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1998 Yellowstone wildfires. *Canadian Journal of Fisheries and Aquatic Sciences,* 58, 1077-1088.

Morley S.A. & Karr J.R. (2002) Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology,* 16, 1498-1509.

Murphy J.F. & Giller P.S. (2000) Seasonal dynamics of macroinvertebrate assemblages in the benthos associated with detritus packs in two low-order streams with different riparian vegetation. *Freshwater Biology,* 43, 617-631.

Nerbonne B.A. & Voudracek B. (2001) Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, U.S.A. *Environmental Management,* 28, 87-99.

Newbold, J.D., Erman, D.C. & Roby K.B. (1980) Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and Aquatic Sciences,* 37, 1076-1085.

Nijboer R.C., Verdonshot F.M. & Van Der Werf D.C. (2005) The use of indicator taxa as representatives of communities in bioassessment. *Freshwater Biology,* 50, 1427-1440.

Norris R.H. & Thomas M.C. (1999) What is river health? *Freshwater Biology,* 41, 197-209.

Paavola R., Muotka T., Virtanen R., Heino J. & Kreivi P. (2003) Are biological classifications of headwater streams concordant across multiple taxonomic groups? *Freshwater Biology,* 48, 1912-1923.

Plotinkoff R.W. & Ehinger S.L. (1997) Using invertebrates to assess the quality of Washington streams and to describe biological expectations. Washington State Department of Ecology, no. 97-332.

Quinn J.M., Boothroyd I.K.G. & Smith B.J. (2004) Riparian buffers mitigate effects of pine plantation logging on New Zealand streams 2. Invertebrate communities. *Forest Ecology and Management*, 191, 129-146.

Reece P.F. & Richardson J.S. (2000) Benthic macroinvertebrate assemblages of coastal and continental streams and large rivers of southwestern British Columbia, Canada. *Hydrobiologia,* 439, 77-89.

Reece P.F., Reynoldson T.B., Richardson J.S. & Rosenberg D.M. (2001) Implications of seasonal variation for biomonitoring with predictive models for the Fraser River catchment, British Columbia. *Canadian Journal of Fisheries and Aquatic Science*, 58: 1411-1418.

Reynoldson T.B., Rosenberg D.M. & Resh V.H. (2001) Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. *Canadian Journal of Fisheries and Aquatic Science,* 58, 1395-1410.

Roeding C.E. & Smock L.A. (1989) Ecology of macroinvertebrate shredders in a low gradient sandy-bottomed stream. *Journal of the North American Benthological Society*, 8, 149-161.

Scarsbrook M.R. (2002) Persistence and stability of lotic invertebrate communities in New Zealand. *Freshwater Biology*, 47, 417-431.

Silsbee D.G. & Larson G.L. (1983) A comparison of streams in logged and unlogged areas of Great Smoky Mountains National Park. *Hydrobiologia*, 102, 99-111.

Stone M.K. & Wallace J.B. (1998) Long-term recovery of a mountain stream from clear-cut logging: the effects of forest succession on benthic invertebrate community structure. *Freshwater Biology*, 39, 151-169.

Tikkanen P., Laasonen P., Muotka T., Huhta A. & Kuusela K. (1994) Short-term recovery of benthos following disturbance from stream habitat rehabilitation. *Hydrobiologia* 273, 121-130.

Townsend C.R., Downes B.J., Peacock K. & Arbuckle C.J. (2004) Scale and the detection of land-use effects on morphology, vegetation and macroinvertebrate communities of grassland streams. *Freshwater Biology*, 49, 448-462.

Turak E. & Koop K. (2003) Use of rare macroinvertebrate taxa and multiple-year data to detect low-level impacts in rivers. *Aquatic Ecosystem Health & Management*, 6, 167-175.

Yuan L.L. & Norton S.B. (2003) Comparing responses of macroinvertebrate metrics to increasing stress. *Journal of the North American Benthological Society*, 22, 308-322.

Stream crossing effects on benthic macroinvertebrate abundance in Grande Cache, Alberta

Introduction

After sedimentation associated with forest harvesting itself, stream crossings provide the greatest input of suspended sediment in harvested watersheds (Rothwell, 1983; Campbell & Doeg, 1989; Carlson et al., 1990; Kaller & Hartman, 2004). Suspended sediment can be detrimental to aquatic invertebrates, fish and aquatic plants. Excess sediment may limit gill function, reduce light available for primary production, overwhelm filtering abilities, and reduce the availability of attachment and habitat sites for invertebrates (Bottorff & Knight, 1996). Headwater streams respond more quickly to such impacts than higher-order streams because of greater variation in sediment load and water volume (Minshall et al., 2001). As the effects of sedimentation on stream integrity have been recognized, stream crossings have been improved to minimize sedimentation rates. However, sedimentation remains a concern in both harvested and unharvested watersheds that are accessible by road.

Benthic macroinvertebrates (BMI) are strong and effective indicators of stream condition because they are key elements in aquatic food webs and exhibit species-specific responses to disturbance (Hilsenhoff, 1987; Fore et al., 1996; Dickes, 1999; Morley & Karr, 2002; Waters, 2000). Each group has a specific set of habitat preferences, with some more tolerant of environmental change than others. Previous evaluations of stream condition have focused on the effects of both anthropogenic (e.g., forest harvesting, urbanization) and natural (e.g., fire, drought) disturbances on BMI communities (Campbell & Doeg, 1989; Bottorff & Knight, 1996; Dickes, 1999; McCabe & Gotelli, 2000; Minshall et al., 2001; Morley & Karr, 2002).

The purpose of this study was to determine whether road-stream crossings affected benthic macroinvertebrate abundance in forested watershed streams in Grande Cache, Alberta. I predicted that the greatest abundance of all taxa (EPTD: Ephemeroptera, Plecoptera, Trichoptera and Diptera) would occur in samples taken above crossings in a paired study. Immediately below the crossing, a lower BMI abundance was expected due to sediment input from the crossing, with some degree of recovery at sample sites 50 metres downstream.

Methods

Two sites were monitored within small watershed basins (< 50 km^2, headwater streams): one that was recently harvested and one that remained unharvested for the duration of the study. Stream sites were located in Weyerhaeuser's Forest Management Area (FMA) in Grande Cache, Alberta and were tributaries of the Sheep Creek system.

Streams were selected with the aid of Digital Elevation Models (ArcView 3.2, ESRI Inc., Redlands, CA, U.S.A.) to ensure similar basin topography. The harvested and unharvested watersheds were as evenly paired as possible, based on riparian cover type, stream size and substrate. Site selection was very limited, primarily by harvest schedules and road access. Both watersheds drained conifer-dominated areas, with the riparian buffer in the harvested watershed composed of low-lying deciduous vegetation. These were high gradient streams dominated by coarse substrates, which are typical of high flow areas.

Samples were collected in August and September 2005. BMI were sampled using a Surber Sampler (350 μm) at three locations along each stream reach: 1. Above the crossing, 2. Immediately below the crossing, and 3. 50 m downstream from the crossing. Three representative riffles were sampled at each of these three locations, and both the harvested and unharvested streams were sampled on three

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separate sampling dates. Multiple riffles were sampled at each location, as single samples might not have accurately represented the natural variation in BMI community structure (Reece & Richardson, 2000; Scarsbrook, 2002). Riffles were selected to be representative of the natural variation (e.g., substrate size, algal cover) in these habitats at each site. Invertebrates were preserved in the field in 95 percent ethanol and taken to the laboratory, where they were transferred to 80 percent ethanol, for identification. BMI were identified to the order level, the lowest reasonable taxonomic level due to the presence of large numbers of early instar larvae that could not be identified further.

Individuals were sorted into five categories: Ephemeroptera, Plecoptera, Trichoptera, Diptera, and Other (consisting primarily of non-arthropod groups, including Coleoptera, Collembola and Gastropoda). Following sorting, the total number of individuals in each sample of each of the four major orders (EPTD) was counted and recorded.

Analysis

All data were included in analyses and distributions were normalized by log transformation. Single-factor ANOVAs (SPSS 14.0, SPSS Inc., Chicago, IL, U.S.A.) were used to identify differences in abundance between harvested and unharvested stream sites. Repeated measures ANOVAs (SPSS 14.0) were used to identify spatial differences in BMI abundance along the stream-crossing gradient. A critical alpha for significance of $p < 0.10$ was used.

Results

A total of 7,811 individuals from the orders Ephemeroptera (4,795; 61.4%), Plecoptera (1,793; 22.9%), Trichoptera (251; 3.2%) and Diptera (972; 12.4%) were counted and included in analyses.

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Although this study focused on the effects of stream crossings on BMI abundance, it was first important to consider whether BMI abundance was at all sensitive to the effects of forest harvesting in the Grande Cache region. As might be expected from Chapter 2, abundance of Trichoptera was greater in the unharvested watershed ($F_{1, 17} = 25.53$, $p < 0.001$), while abundance of Diptera was greater in the harvested watershed ($F_{1, 17} = 14.03$, $p < 0.005$) (Figure 1). Abundance of Ephemeroptera and Plecoptera in the samples did not vary significantly with forest harvesting ($F_{1, 17} = 1.05$, $p = 0.32$ and $F_{1, 17} = 0.37$, $p = 0.55$, respectively).
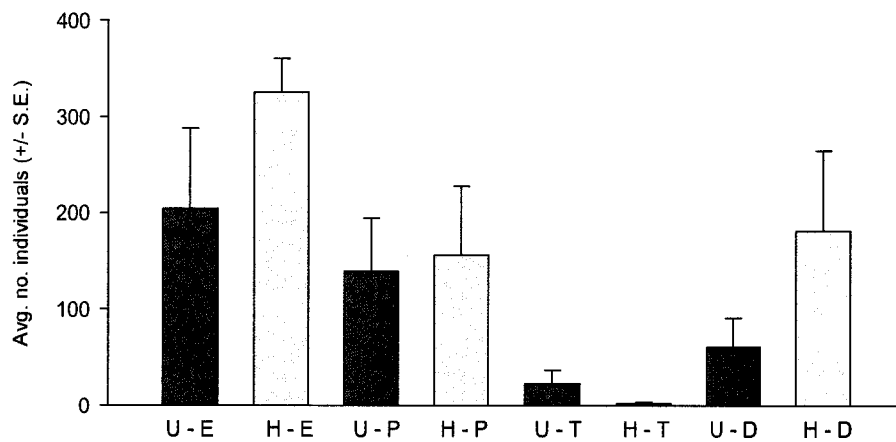
Figure 1. Harvest effects on taxa abundance (U: Unharvested, H: Harvested; E: Ephemeroptera, P: Plecoptera: T: Trichoptera, and D: Diptera) at Grande Cache, AB. Harvest activities significantly affected BMI abundance in this region.

Stream crossings did not affect all taxa equally. In the unharvested watershed, trichopteran and dipteran abundance was significantly lower immediately below crossings, with some degree of recovery evident 50 m downstream (Table 1; Figure 2). Dipteran abundance recovered to pre-crossing levels 50 m below crossings, but abundance of Trichoptera did not. However, neither trichopteran nor dipteran abundance was significantly affected by stream crossings in the harvested watershed. Ephemeropteran and plecopteran abundance was not

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significantly altered in relation to stream crossings at either the harvested or unharvested site. Nonetheless, variation in abundance of mayflies and stoneflies in the unharvested stream followed the significant patterns observed for abundance of caddis flies and aquatic Diptera at the same stream.

Table 1. Spatial effects of stream crossings on BMI abundance under two harvest treatments in Grande Cache, AB 2004. * Indicates significant results.

	Ephemeroptera	Plecoptera	Trichoptera	Diptera
Unharvested	$F_{2,26} = 0.54$ $p = 0.59$	$F_{2,26} = 0.83$ $p = 0.45$	$F_{2,26} = 2.97$ $p = 0.06$ *	$F_{2,26} = 5.58$ $p = 0.01$ *
Harvested	$F_{2,26} = 0.85$ $p = 0.44$	$F_{2,26} = 1.79$ $p = 0.19$	$F_{2,26} = 0.60$ $p = 0.56$	$F_{2,26} = 0.48$ $p = 0.62$

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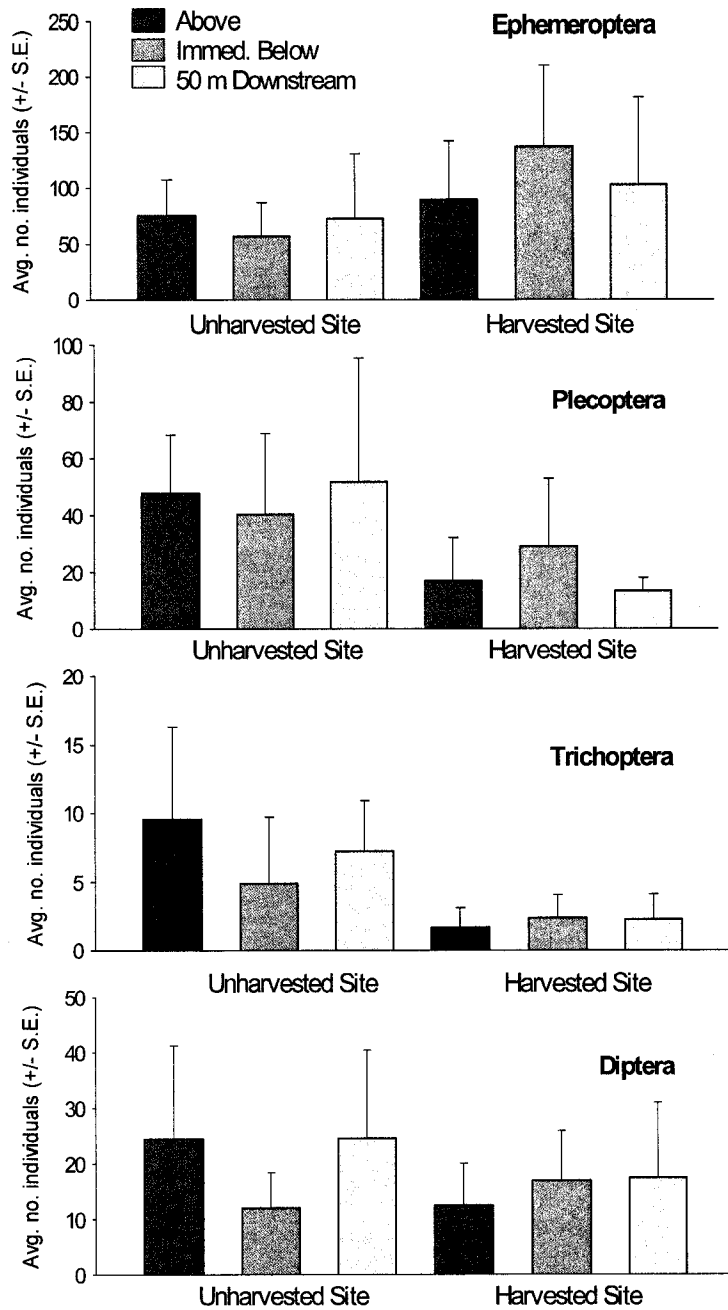
Figure 2. BMI abundance along a stream-crossing gradient, from above the crossing (control), immediately below the crossing and 50 metres downstream, in Grande Cache, AB. All taxa followed a pattern of reduced abundance immediately below the crossing, with some degree of recovery 50m downstream, at the unharvested site. Similar patterns were not observed at the harvested site.

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Discussion

The primary goal of this study was not to quantify differences between harvested and unharvested watershed stream communities. However, to realistically examine the effects of stream crossings on BMI abundance in these streams, the potential impact of watershed treatment was necessarily considered. Taxa did not respond equally to harvest, but as expected, trichopteran abundance decreased and abundance of aquatic dipterans increased in response to this disturbance. Higher sedimentation rates in harvested watersheds provide more habitat to sediment-inhabiting dipteran larvae (Bottorff & Knight 1996; Minshall et al., 2001). However, caddisflies were negatively affected by harvesting, perhaps because of their strong association with undisturbed riffle habitats (Chapter 2).

Ephemeroptera and Plecoptera are generally also considered sensitive to environmental change (Bottorff & Knight, 1996; Morley & Karr, 2002; Cole et al., 2003; Yuan & Norton 2003; Quinn et al., 2004), but such effects were not significant in data from the Grande Cache region. Newbold et al. (1980) found that although invertebrate diversity in disturbed streams was diminished by sedimentation, overall abundance did not change. In the present study, significant patterns may have been found at the genus or species levels. The order Ephemeroptera, for example, comprises many species with varying degrees of tolerance to environmental change. In particular, the genus *Baetis* includes many tolerant and resilient species that are often more abundant in harvested watersheds. Minshall et al. (2001) found that chironomids (Diptera) and *Baetis sp.* comprised up to 60% of post-disturbance BMI assemblages in Montana, as opposed to 18% in undisturbed streams. The present study, with identification to the order level, is insensitive to these sorts of patterns.

However, to meet the initial objectives of this study, it was deemed sufficient to show whether differences existed in the abundance of some BMI higher taxa between harvested and unharvested watershed streams at the order level.

Overall trichopteran and dipteran abundance were, in fact, sensitive to the effects of stream crossings in the unharvested watershed stream at Grande Cache. Ephemeroptera and Plecoptera exhibited the same pattern of reduced abundance immediately below the crossing with some recovery downstream, but the differences were not statistically significant.

Differences in the effects of crossings on BMI abundance between the harvested and unharvested watershed streams may have been more related to the type of riparian vegetation dominating each watershed than to the presence of a crossing or forest harvesting. Benstead and Pringle (2004) suggested that changes in BMI communities are driven by shifts in availability of basal food resources supplied by riparian vegetation, rather than due to abiotic disturbances, including sedimentation. Aquatic dipteran larvae, for example, rely heavily on terrestrial carbon inputs for filter feeding. At the harvested site a deciduous riparian buffer would have supplied higher quality food resources than did the coniferous vegetation at the unharvested site (Murphy & Giller, 2000). As a group, the Diptera may have been most sensitive to these differences since the EPT represent a wider range of functional feeding groups (FFG). However, shifts in patterns of abundance may have been observed in EPT at a lower level of taxonomic resolution, when individual genera or species could be characterized according to particular FFGs.

Although I tried to pair the harvested and unharvested watersheds, harvest schedules and limited access restricted site selection, such that the Sheep Creek tributaries were the only watersheds available in the FMA for this study. Unmeasured factors differing between these watersheds may have exerted more effects on community structure than did the anthropogenic stresses that I attempted to compare. Slight differences in stream gradient, sinuosity and riparian vegetation may have resulted in differences in flow regimes affecting the

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pattern and rate of sediment deposition and removal. If such factors do, in fact, explain the differences observed, they would have to have been consistent across all 3 replicate sites for each treatment in each stream.

The lack of a stream crossing effect on BMI abundance in the harvested watershed might be attributed to a behavioural response. Behavioural drift rates are higher in disturbed streams, as BMI actively seek more hospitable environments (Waters, 1965; Campbell & Doeg, 1989; Siler et al., 2001). It is possible that increased sedimentation from the stream crossing only lengthened drift distances and, as a result, BMI would have moved further downstream, to areas beyond the sample sites, in search of suitable habitat. As a result, there would be no expected difference in BMI abundance above and below the crossing and a stream crossing effect in this watershed would not be observed.

In general, abundances of Ephemeroptera, Plecoptera and Trichoptera (EPT) are reduced by fine sediments (Kaller & Hartman, 2004) but aquatic larvae of dipterans tend to be hardier and more tolerant, thriving in high sedimentation areas (Newbold et al., 1980). They have short generation times and rapid colonization rates, which enables them to deal with fluctuating environments and build up large populations opportunistically. However, Diptera in Grande Cache responded to crossings in the same way as other well-known sensitive taxa (EPT), with decreased abundance in more disturbed areas. Sediment generally accumulates in areas of slow flow, such as pools and slow moving riffles (Kaller & Hartman, 2004). Streamflow in this high gradient area may have been sufficient, such that these streams could not support large aquatic dipteran populations, as their food resources and habitat-base were too quickly washed downstream. As a result, the Diptera may have been narrowly adapted to habitat and resource conditions in this region, resulting in their high sensitivity to stream crossings.

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Trichoptera as indicators of environmental change

Trichoptera responded strongly to the effects of both stream crossings and forest harvesting, although responses to the two disturbance types did not interact statistically. Relative to other taxa, trichopteran abundance was consistently lower in both harvested and unharvested sites (3.2% of sampled BMI). Trichoptera appear to be the most sensitive group in the Sheep Creek system, and thus, they may be early indicators of change in harvested and crossed watersheds. In the interest of saving time and resources, order level identification of this group can be successfully used to identify potentially disturbed stream sites. Further work, however, could be quite useful in further understanding which particular species comprise the response and what ecological features are responsible for it.

Turak and Koop (2003) suggested that, while most indicator species are hardy and disturbance-adapted, the use of rare and sensitive taxa might be more ecologically relevant. Trichoptera have specific habitat and flow requirements, including clean, well-oxygenated water and can be considered environmentally sensitive taxa (Clifford, 1991). Sensitive taxa may be indicative of smaller, less dramatic environmental changes and could provide policy makers with sufficient time to initiate and implement change before irreversible damage is done.

Pseudoreplication and taxonomic resolution issues

Several recent studies have acknowledged the issue of pseudoreplication in their designs and have dealt with the issue by limiting the inference of their results to the region under investigation (Viroux, 1999; Kiffney et al., 2004; Whiles & Goldowitz, 2005). Similarly, it was not possible to achieve true replication in this study, due to limited site selection, and the results thus relate firmly only to the sites we sampled. However, these results may provide the basis for the development of further hypotheses (Hargrove & Pickering, 1992) and could be useful in subsequent meta-analyses. Oksanen (2001) suggests that replication is

most necessary only when the predicted response is very weak, in which case replication can be obtained by conducting similar experiments or by performing meta-analyses.

Effects of taxonomic resolution on the results of this study must also be addressed. There is considerable debate as to the level of taxonomic resolution necessary to reliably reveal environmental impact (Reynoldson et al., 2001). Arguments range from favouring a system of order-level resolution, for its savings in time and resources, to favouring a system of species-level resolution, which accounts for the species-specific evolution of variable tolerances to disturbance. In this study, samples included a wide range of larval instars, and identification of all individuals to lower taxonomic levels was beyond the constraint of available resources. Further taxonomic resolution may have revealed significant differences within the community at the genus or species level, which were not apparent with order-level resolution. Patterns observed in relation to forest harvest and stream crossings, particularly for the Ephemeroptera and Plecoptera, may have become statistically significant with greater taxonomic resolution or greater sampling effort (Warren, 1986). However, the patterns of significant response observed for the Diptera and Trichoptera may be useful in an applied management context as described above.

Summary

In conclusion, stream crossings affected BMI abundance only in the unharvested watershed. Riparian vegetation and streamflow regimes may be strong mitigating factors maintaining stream integrity against the effects of stream-crossings and forest harvesting in Grande Cache. However, the Trichoptera, were sensitive enough to respond to the earliest signals of environmental change. In the future, initial assessments of the anthropogenic effects on stream integrity in the Grande Cache region should focus their efforts on sampling Trichoptera abundance. Restoration programs and plans for the prevention of further degradation may be

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most effective with the use of sensitive taxa to identify potentially degraded streams in their earliest stages of decline, and subsequent action may prevent irreversible damage associated with harvest.

References

Benstead J.P. & Pringle C.M. (2004) Deforestation alters the resource base and biomass of endemic stream insects in eastern Madagascar. *Freshwater Biology*, 49, 490-501.

Bottorff R.L. & Knight A.W. (1996) The effects of clearcut logging on stream biology of the North Fork of Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA -1986 to 1994 Unpubl. Report prepared for the California Department of Forestry and Fire Protection, Contract No. 8CA3802, May 1996, Sacramento, CA. 177 p.

Campbell I.C. & Doeg T.J. (1989) Impact of timber harvesting and production on streams: A review. *Australian Journal of Marine and Freshwater Research, 40,* 519-539.

Carlson J.Y., Andrus C.W. & Froelich H.A. (1990) Woody debris, channel features, and macroinvertebrates of streams with logged and undisturbed riparian timber in northeastern Oregon, USA. *Canadian Journal of Fisheries and Aquatic Sciences, 47,* 1103-1111.

Clifford H.F. (1991) *Aquatic Invertebrates of Alberta.* The University of Alberta Press, Edmonton, AB, Canada.

Cole M.B., Russell K.R. & Mabee T.J. (2003) Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon Coast Range Mountains. *Canadian Journal of Forest Research*, 33, 1433-1443.

Dickes B. (1999) McDonald Creek benthic macroinvertebrate study: a cumulative impact assessment after timber harvesting in a small watershed. Washington State Department of Ecology Publication, no. 99-304.

Fore L.S., Karr, J.R. & Wisseman, R.W. (1996) Assessing invertebrate responses to human activities: evaluating alternative approaches. *Journal of the North American Benthological Society,* 15, 212-231.

Hargrove W.W. & Pickering J. (1992) Pseudoreplication: a *sine qua non* for regional ecology. *Landscape Ecology,* 6, 251-258.

Hilsenhoff W. (1987) An improved biotic index of organic stream pollution. *The Great Lakes Entomologist*, 20, 31-40.

Kaller M.D. & Hartman K.J. (2004) Evidence of a threshold level of fine sediment accumulation for altering benthic macroinvertebrate communities. *Hydrobiologia*, 518, 95-104.

Kiffney P.M., Volk C.J., Beechie T.J., Murray J.L., Pess G.R. & Edmonds R.L. (2004) A high-severity disturbance event alters community and ecosystem properties in West Twin Creek, Olympia National Park, Washington, U.S.A. *The American Midland Naturalist*, 152, 286-303.

McCabe D.J. & Gotelli N.J. (2000) Effects of disturbance frequency, intensity, and area on assemblages of stream macroinvertebrates. *Oecologia*, 124, 270-279.

Minshall G.W., Royer T.V. & Robinson C.T. (2001) Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1998 Yellowstone wildfires. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 1077-1088.

Morley S.A. & Karr J.R. (2002) Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology*, 16, 1498-1509.

Murphy J.F. & Giller P.S. (2000) Seasonal dynamics of macroinvertebrate assemblages in the benthos associated with detritus packs in two low-order streams with different riparian vegetation. *Freshwater Biology*, 43, 617-631.

Newbold, J.D., Erman, D.C. & Roby K.B. (1980) Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and Aquatic Sciences*, 37, 1076-1085.

Oksanen L. (2001) Logic of experiments in ecology: is pseudoreplication a pseudoissue? *Oikos*, 94, 27-38.

Quinn J.M., Boothroyd I.K.G. & Smith B.J. (2004) Riparian buffers mitigate effects of pine plantation logging on New Zealand streams 2. Invertebrate communities. *Forest Ecology and Management*, 191, 129-146.

Reece P.F. & Richardson J.S. (2000) Benthic macroinvertebrate assemblages of coastal and continental streams and large rivers of southwestern British Columbia, Canada. *Hydrobiologia*, 439, 77-89.

Reynoldson T.B., Rosenberg D.M. & Resh V.H. (2001) Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. *Canadian Journal of Fisheries and Aquatic Science*, 58, 1395-1410.

Rothwell R.L. (1983) Erosion and sediment control at road-stream crossings. *The Forestry Chronicle*, 59, 61-66.

Scarsbrook M.R. (2002) Persistence and stability of lotic invertebrate communities in New Zealand. *Freshwater Biology*, 47, 417-431.

Siler E.R., Wallace J.B. & Eggert S.L. (2001) Long-term effects of resources limitation on stream invertebrate drift. *Canadian Journal of Fisheries and Aquatic Science*, 58, 1624-1637.

Turak E. & Koop K. (2003) Use of rare macroinvertebrate taxa and multiple-year data to detect low-level impacts in rivers. *Aquatic Ecosystem Health & Management*, 6, 167-175.

Warren W.G. (1986) On the presentation of statistical analysis: reason or ritual. *Canadian Journal of Forest Research*, 16, 1185-1191.

Waters, T.F. (1965) Interpretation of invertebrate drift in streams. *Ecology*, 46, 327-334.

Waters T.F. (2000) *Wildstream: a natural history of the free flowing river*. Riparian Press, St Paul, Minnesota, USA.

Whiles M.R. & Goldowitz B.S. (2005) Macroinvertebrate communities in central platte river wetlands: patterns across a hydrologic gradient. *Wetlands*, 25, 462-472.

Yuan L.L. & Norton S.B. (2003) Comparing responses of macroinvertebrate metrics to increasing stress. *Journal of the North American Benthological Society*, 22, 308-322.

Use of the Benthic Index of Biotic Integrity for assessment of forest management practices

Biotic integrity has been defined as the ability of a system to support and sustain a balanced ecological community (Karr, 1981). A community is considered stable when changes in its structure are minimal and its persistence is high (Townsend et al., 1987). Stream invertebrates are well adapted to normal environmental variation, including extreme events, that is within their range of "evolutionary experience" (Karr, 1999). However, communities may not persist unchanged under conditions beyond their range of experience, most notably new episodes of anthropogenic disturbance. These disturbances change interactions between organisms, alter physical habitats and food bases, and modify hydrologic regimes. Benthic macroinvertebrate density is generally greater and diversity is lower in disturbed stream sites than in similar undisturbed sites (Carlson et al., 1990; Bottorff & Knight, 1996; Dickes, 1999; McCabe & Gotelli, 2000). Highly disturbed communities are dominated by disturbance-adapted species, consisting mostly of small, filter-feeding species with high rates of fecundity, such as chironomids and *Baetis spp.* (Kedzierski & Smock, 2001).

Assessments of stream integrity should be an important component of sustainable forestry. By quantifying the effects of anthropogenic disturbances, such as forest harvesting and industrial development, on streams we can address the effectiveness of current practices and policies for preserving the ecological function of watersheds. Historically the focus has been on point source detection, threshold contaminant levels and acute toxicity tests. However, biological monitoring is often preferable to chemical monitoring, as chemical pollution may be transient and unpredictable, and samples represent conditions at one point in time and space, rather than long-term cumulative effects (Guerold, 2000). Indices of stream condition have used a variety of organisms as indicators, including fish, benthic macroinvertebrates and diatoms. As biotic indicators selected taxa must

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be sensitive to disturbance and respond in a predictable manner, with minimally variable responses (Dale & Beyeler, 2001).

Several indices have been proposed for quantifying stream condition, including diversity, similarity, multivariate and multimetric indices. Recently a Benthic Index of Biological Integrity (B-IBI) was constructed to quantify stream condition in terms that are useful and easily understood by both the scientific and non-scientific community (Kerans & Karr, 1994; Morley & Karr, 2002). In contrast to other approaches, use of the B-IBI in stream impact assessments synthesizes impact of multiple disturbance events and also provides a clear quantitative measure of biological degradation.

The B-IBI (Kerans & Karr, 1994) is distinguished from historical methods of stream assessment by its ability to respond to cumulative impacts and the integration of human disturbance. It was developed based on Karr's (1981) Index of Biotic Integrity, which used fish communities to assess stream condition and detect degraded sites. Benthic macroinvertebrate communities provide an ideal basis for this model because they include taxonomically and ecologically well-known groups that exhibit species-specific responses to disturbance. They also respond rapidly to a broad range of environmental changes and disturbances, and are key elements in aquatic ecosystems (Kotze & Samways, 1999; Morley & Karr, 2002).

The B-IBI relies on comparison with a reference condition. A reference stream is one that has been least disturbed by human impacts in a particular region and is used as the basis for comparison against potentially disturbed sites based on a series of community descriptors (Bailey et al., 1998). Issues of pseudoreplication in biomonitoring, where upstream and downstream sites in the same stream were previously treated as comparable, are overcome by the incorporation of the reference site (Reynoldson et al., 1997).

Multimetric indices are designed to be sensitive to biological degradation and assess stream condition (Reynoldson et al., 1997; Karr, 1999; Morley & Karr, 2002). Under the B-IBI approach *"attribute"* refers to all possible measures of biotic integrity, whereas *"metric"* refers to those attributes that respond in a predictable and statistically significant manner to increasing levels of anthropogenic disturbance in a particular region. The use of multiple metrics increases the probability of accurately assessing stream condition (Fore et al., 1996; Ranasinghe et al., 2002). A multimetric index quantifies the magnitude of stream degradation and provides a single score, the B-IBI score, which integrates individual, population and community-level information (Kerans & Karr, 1994; Ganason & Hughes, 1998).

Metrics may be diversity or richness indices, or may reflect community function, including feeding groups and habits. Metrics are complementary and are intended to represent all possible facets of the community (Fore et al., 1996; Weigel et al., 2002). They can distinguish between reference and degraded sites, and correlate strongly with factors indicative of poor water and habitat quality (Kerans & Karr, 1994). Only those metrics that respond significantly to disturbance are included in the B-IBI and thus the number of metrics varies by region. Kerans and Karr's (1994) original model for the Tennessee Valley consisted of 13 metrics, while Morley and Karr's (2002) B-IBI for the Puget Sound Basin consisted of 10 metrics.

All metrics are assigned scores that reflect stream condition relative to the condition of the undisturbed reference stream, which is assigned values that are considered optimal. In most cases the optimal value is five (5), while metrics that differ most widely from the reference condition are assigned a value of one (1). Individual metric scores are then summed to obtain a final B-IBI score that is compared to the target reference score. Sites are ranked from very poor or

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disturbed (low score, same as number of metrics) to excellent or relatively undisturbed (maximum score, same as reference site score).

Objectives

The purpose of this review is not to discredit the B-IBI approach, but to recognize the sources of its limitations and explore its value, especially for forest management purposes. The B-IBI has proven to be a useful assessment tool in many circumstances but results are most useful and informative when interpreted within their limits. Having recognized the limits of the model we can determine where, when and how it can be most effectively applied.

Advantages of the B-IBI

Abundance and density measurements are not consistently reliable measures of water quality because they experience substantial natural variation (Karr, 1999). Since its development in 1994, the B-IBI has gained momentum and credibility, and has been adopted by several American state governments and the Environmental Protection Agency. Although initial setup and metric selection can be expensive and time consuming, once established a B-IBI can be used to make efficient assessments of stream condition in subsequent years.

The advantages of the B-IBI are numerous. It is a relatively fast and inexpensive assessment method. The model is powerful enough that, providing reference sites are sampled annually, the same B-IBI can be used even if sampling and analytical methods change. In terms of its overall goal of assessing the magnitude of stream degradation, it is a more reasonable to measure only those community attributes that respond significantly to disturbance than to measure all factors that potentially impact stream condition, and its results are cumulative. The B-IBI can distinguish between community responses to anthropogenic and natural

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disturbances, although anthropogenic disturbances may intensify the impacts of natural disturbance events (Fore et al., 1996; Ravera, 2001; Weigel et al., 2002). The magnitude of disturbance between reference and disturbed sites is comparable and is reflected by the B-IBI score (Kerans & Karr, 1994; Fore et al., 1996; Bailey et al., 1998). Larger differences between test and reference scores are suggestive of greater disturbance area, intensity or frequency. From a management perspective, results are easily communicated as a single score and can be interpreted and applied by individuals with limited biological experience or knowledge.

From a design perspective, the B-IBI is statistically sound, stable, sensitive and robust (Alden et al., 2002). Sampling is replicated, providing statistical power and validity to the results. Results are quantitative and comparable, and are specific to the habitat and region under investigation. Alden et al. (2002) found that single metrics classified relative site quality as effectively as a multimetric index. However, the redundancy of using multiple metrics increases confidence in the results (Ranasinghe et al., 2002). In addition, large changes to a single metric's value did not alter overall B-IBI scores, supporting the method's robustness.

Limitations of the B-IBI

Statistical techniques are used to identify patterns in biological data and reduce the possibility of drawing unsupported conclusions (Marshall & Elliot, 1997). The sole purpose of the B-IBI is to assess the degree of stream degradation. However, users must be aware of the limitations to the application and interpretation of its results.

Several studies have examined the limitations of the B-IBI. Suter (1993), for example, offered an extensive list of criticisms for the index, including the

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ambiguity of the B-IBI score and post hoc justification of results. Other, more specific, problems have also been identified. Errors in construction and calculation of a B-IBI are compounded and can severely misrepresent stream condition (Reynoldson et al., 1997). Sampling effort affects B-IBI measures (Cao et al., 2003) and the rare taxa that are often most sensitive to disturbance and representative of pristine conditions can be ignored by the B-IBI, which requires a minimum sample size of only 100 individuals (Weigel et al., 2002). Realistically, a minimum of 2000 individuals should be sampled to account for rare taxa (Li et al., 2001).

Despite the list of advantages mentioned above, there are several broad sources of additional concern that may limit the applicability and interpretation of B-IBI results:

Metric selection and weighting do not account for natural variation

Metric selection provides the foundation of the B-IBI. All metrics are not created equally. Although metrics respond predictably to increasing levels of human disturbance, individual metrics may have different sensitivities to different disturbance types or events. Latitudinal filters, such as air and water temperature, strongly influence taxa distribution and community structure (Heino et al., 2003). Metric response may be spatially variable at a scale smaller than is being sampled, such that what is useful for one site will not necessarily be useful for another. For example, changes in physical stream characteristics, including morphological changes and increases in stream size, can lead to unrealistically high B-IBI scores because metrics respond very differently to these changes (Ganason & Hughes, 1998).

In addition, metric response may be temporally variable, due to external environmental factors not directly related to a disturbance event. It is typically assumed that environmental factors affect stream sites equally in a given region.

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However, this also assumes that reference and degraded sites have been perfectly paired. Realistically, there are likely to be differences between sites that may affect metric responses at a site, including, but not limited to, differences in elevation, stream gradient or flow regimes, and surrounding riparian vegetation. For example, differences in stream gradient can lead to differences in substrate types and sedimentation rates, which results in natural differences in BMI community structure unrelated to anthropogenic disturbance. Therefore, the magnitude of difference between B-IBI scores for reference and degraded sites can be exacerbated by naturally occurring differences and poor sample design. This could be problematic, when in fact no anthropogenic effect exists, or the particular effects of a human-induced disturbance event are not as severe as indicated by the model. Through the use of consistent sampling methods and timing, and the use of multiple sample replicates, this variability can be minimized. In most cases, it must be assumed that any factors that impact one site will similarly impact all sites, including the reference site, because it is logistically impossible to identify and quantify all differences between a reference and degraded stream.

All metrics are given equal weighting in the final B-IBI score and thus have an equal influence on the final outcome. The use of multiple metrics increases the model's statistical robustness (Alden et al., 2002; Ranasinghe et al., 2002), despite the fact that some metrics are potentially more influential than others. Influential, inclusive metrics vary by habitat and region. By the model's design, undisturbed sites are assigned a full set of optimal metric scores. However, sites evaluated as degraded by the B-IBI exhibit lower than optimal scores in only one or two of the inclusive metrics, making them the most influential metrics in the set (Weigel et al., 2002). For example, proportion EPT (Ephemeroptera, Plecoptera and Trichoptera), Shannon diversity and percent dominance can discriminate between ecoregions and are highly influential metrics (Li et al. 2001). Thus, when highly influential metrics are combined with less influential, although still significant metrics, anthropogenic effects are minimized.

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Single metrics cannot adequately represent overall community structure (Cao et al., 2003). Ranasinghe et al. (2002) found that large changes to an individual metric's score would not affect the final B-IBI score. However, this likely depends on the number of metrics in the model. A model with fewer metrics is more sensitive to changes in single metric values than a model with more metrics.

We must also be able to discern whether selected metrics provide an informative and useful signal, or respond simply because the metric is too vaguely defined. The most obvious way to overcome this problem is with increased taxonomic resolution. In their assessment of river integrity, Bady et al. (2005) required that all individuals be identified to the lowest common base due to potentially influential differences in taxonomic resolution. Different behaviours, including habits and manifestations of functional feeding groups, are observed at different taxonomic scales. Broad generalizations can be applied to lower resolution groups (ex. order-level), whereas taxon-specific details can be applied when higher resolution is used (ex. genus- or species-level). However, the added time, expertise and expense for this increased resolution must be considered. Potentially significant and influential metrics could be ignored when low resolution is used, which will mislead B-IBI results.

Unidirectional, linear metric scoring

A gradient of disturbance, represented by multiple stream sites with differing exposures to anthropogenic stress, is necessary for the B-IBI to accurately quantify the impact of disturbance on a stream (Lammert & Allan, 1999). When metrics are assigned scores relative to a reference stream, it is assumed that they respond linearly to disturbance.

The assumption that all useful metrics respond linearly to anthropogenic disturbance may be presumptuous. Metrics will only rarely respond independently in a dynamic system, such as a stream. Their responses are not

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driven solely by external environmental factors, including stream morphology and climate, but also by the responses of community members and related metrics. The density of one population may increase or decrease in response to changes in the density of a competitive, predatory or immigrant population. Recovery of an entire community to its pre-disturbance state reflects the sum of responses of individual populations and their related metrics. Therefore, the entire recovered community may exhibit linear responses while individual metrics may not.

In addition, the use of metrics that only respond linearly limits the number of potentially effective metrics, and other useful and informative metrics might be ignored. The use of non-linear metrics may be useful, providing they respond predictably to disturbance and similarly to those at the reference site. The problem with the inclusion of non-linear metrics arises in the assignment of metric scores. In this case, there is potential for the use of a presence and absence scheme, where presence (of degradation) indicates some deviation from the reference condition and absence (of degradation) indicates no deviation from the reference condition. This method may not be as detailed or accurate as the original model, but will still meet the primary goal of a B-IBI assessment, which is to assess whether a site is degraded. Even if the magnitude of degradation cannot be assessed, non-linear or poorly defined metrics could provide a preliminary assessment to determine whether further investigation is warranted.

Inability to Identify Causal Mechanisms

Biological assessments are excellent tools for quantifying the effects of a disturbance, but are not designed to identify the root causes of these effects. The identification of causal mechanisms is beyond the scope of B-IBI methods, which are intended only as an assessment tool. Ultimately, however, an understanding of the causes of the observed patterns may be more important for forest managers than the basic knowledge that a stream is degraded. Bunn and Davis (2000) suggest that we cannot deem our biomonitoring efforts successful unless those

processes that influence and define stream structure and dynamics are identified and understood. Deviations from the reference stream's condition can be misinterpreted as changes in the stream's integrity, when in fact they are due to natural stochastic processes. Again, it is important to note that for the B-IBI to be accurately interpreted and applied, it must be assumed that reference and degraded sites have been effectively paired. Streams are heterogeneous systems, which fluctuate around a relatively stable state. Natural stochastic processes and differences between reference and degraded sites (gradient, flow regimes, elevation, riparian vegetation, etc.) can have substantial and significant effects on BMI communities without necessarily affecting stream condition.

Karr (1999) suggests that the role of multimetric indices, such as the B-IBI, is to tease out the impacts of anthropogenic disturbance and not to explain variation, including natural variation, within that system. Identification of causal mechanisms would require measurements of chemical and physical water parameters, and watershed- and riparian-scale habitat assessments to understand specifically how and why the system responded to a particular disturbance in a particular manner. However, the B-IBI measures the cumulative impacts of disturbance and some causes of the observed degradation are not effectively identified by B-IBI methods.

When identifying causal mechanisms it is crucial to consider all potential causes, and eliminate bias toward singular explanations that have precedent or seem apparent. In the headwater tributaries of southeastern Michigan, Lammert and Allan (1999) found that habitat, immediate land use and dominant substrate were all strong predictors of macroinvertebrate community condition and biotic integrity. However, stream invertebrates, and other members of the aquatic food chain, do not respond only directly to stresses in the watershed (Novotny et al., 2005). The impact of each stressor is filtered, changed in combination with others from its original form into a more complex overall effect on the biota. Thus, it is not reasonable to correlate B-IBI scores directly with single or multiple

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disturbances in a watershed because they may not represent a direct cause-effect relationship. The use of the B-IBI for the sole purpose of identifying stream degradation is recommended. The identification of causal mechanisms requires strict adherence to sample designs that are difficult to achieve in natural settings. Locating test and reference sites that vary only by the degree of disturbance is near impossible, due to natural environmental variation.

Loss of potentially relevant ecological information

In the communication and interpretation of a B-IBI score all potentially important and relevant ecological information is lost. Without an understanding of how a score was derived and its ecological implications, decision makers and non-scientists may misinterpret and misuse the results. Although the overall B-IBI is useful to identify when a problem exists, consideration of the individual metric score components is required to reveal where and potentially why the problems exist. Biological information becomes relevant when the B-IBI has identified a degraded system and forest management practices and policies come into question as contributing to causation. Although we 'blow the whistle' when the overall community has responded to disturbance in a number of interacting ways, we must understand how each component group responds to present and test hypotheses of causation as is required to correct the situation.

Early use of benthic macroinvertebrates in bioassessments focused on indicator species. Such assessments assumed that any response and variation in that species would correlate with the response and variation in the entire community and reflect disturbance impacts (Carignan & Villard, 2002). Individual metrics can be assigned to indicator species, with the knowledge of how each metric responds to disturbance and this would help to make sense of B-IBI results. The advantage to using indicator species metrics is the extensive knowledge base available, detailing species biology and providing a strong basis for predicting its anticipated responses to disturbance. The disadvantage to using indicator species

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is that these are often hardier, disturbance-adapted groups, while the rare and sensitive taxa may provide more ecologically important information (Turak & Koop, 2003). The use of a keystone species, rather than an indicator species, may be more useful for developing a B-IBI, as changes in keystone species could be used to predict future changes to the BMI community and stream health. For example, salmon (Willson & Halupka, 1995; Cederholm et al., 1999), *Daphnia spp.* (Steiner, 2002) and crayfish (Creed, 1994; Parkyn et al., 1997) have been identified as aquatic keystone species, and their presence in the system maintains the populations of the entire aquatic community.

Behavioural components may also be lost through use of a single score. Natural and disturbance-induced variation in biological processes, including competition and predation, can significantly influence community structure and are often ignored in favour of the more obvious community patterns and biophysical processes when attempting to identify patterns of stream response to disturbance (Bunn & Davis, 2000). These changes may, in fact, be early warning signals of disturbance and would be overlooked by a standard B-IBI. Inter- and intra-group interactions and dynamics can be indicative of population and community response. Bunn and Davis (2000) emphasized the need to select metrics based on their strong functional importance. Incorporation of behavioural metrics is possible, although it would require a mathematical basis (e.g., ratio of active to passive drift) and a strong understanding of the behavioural complexities and interactions in a particular region.

Regional Specificity

B-IBI metrics vary regionally because benthic communities evolve under different sets of limits and opportunities (Karr, 1999). Metrics that are statistically significant in one region may be insignificant in another, based on extensive regional patterns, including topography, vegetation cover, and weather

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patterns. Application of the B-IBI in managed forests is confounded when managed areas are larger than those where one set of metrics can be reasonably applied, and can make for an inefficient use of resources, including time and funding. Thus, it is important to select appropriate reference sites and acknowledge sources of natural variation between test and reference sites that might explain any observed patterns of degradation. Forest management areas are not necessarily designated based on regional characteristics or patterns. FMAs that cross transition zones, such as the foothills or aspen parkland, may be subject to higher variability than other, less heterogeneous zones, such as a plantation, where variability across larger physical areas is relatively lower.

Regionally based B-IBIs have been developed for Chesapeake Bay and the Pacific Northwest (Kleindl, 1995; Ranasinghe et al., 2002; Mebane et al., 2003). A universal set of metrics applicable to all regions and yielding an accurate B-IBI score seems to be a goal worthy of pursuit, although none currently exists. A universal model could, it seems, be constructed by broadening the scope of inclusive metrics. By shifting metric focus from community structure to metrics that reflect community functions, behaviours and interactions it may be possible to derive universally applicable metrics, as all biological systems involve predators and prey, competition, births and deaths, immigrations and emigrations.

Lack of Natural Reference Streams

One of the more practical limitations of the B-IBI is the lack of natural reference streams in most regions of North America and Europe, particularly in those under strong anthropogenic pressure where there are the most serious needs for application of B-IBI methods. The B-IBI model requires that an undisturbed watershed be well paired with the stream under investigation, ideally differing only by the factor of interest, in this case anthropogenic disturbance. Landscape disturbances brought about by forest management generally manifest over large spatial scales and can complicate selection of appropriate reference sites. Perhaps

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forest policy ought to set aside reference watersheds for primary and secondary streams on some sort of landscape basis until disturbed watersheds have recovered.

When a pristine reference site cannot be located, the least disturbed site in the region can be used as a reference site (Ganason & Hughes, 1998). Alternatively and perhaps more ideally, a model reference site based on historical conditions and the habitat requirements of the community could be constructed. However, construction of this model reference site is likely to be expensive and would require extensive biological and regional expertise and historical data. Geological records might also be used to assess the pre-disturbance state of a stream or watershed and construct a reference model. This requires either strong documentation or extensive field data and analysis. Once the initial state of the reference stream has been recognized (i.e. how far does it deviate from pristine conditions?) it can be used as a restoration goal.

Due to the severity and frequency of human disturbance in a watershed, streams are often irreversibly damaged and restoration to the original (reference) state is impossible (Novotny et al., 2005). There is virtually no hope for these techniques with respect to large river systems on the planet because those in inhabited regions have endured decades, if not centuries, of significant disturbance. However, the approach is viable for streams in most regions. Locating natural reference streams requires a strong knowledge of the natural state of a given area. Streams and their communities are dynamic systems, and the natural state is likely to be variable around some optimum, based on additional factors, including climate and seasonal patterns. This makes separating naturally occurring variation from that caused by anthropogenic disturbances difficult. In severely degraded, high use regions, issues of permanent degradation, where sites are beyond recovery to their natural states, arise and the construction of a B-IBI requires the use of the best possible site as the new "normal." Therein lie problems of unidirectional metric scoring, where metric scores cannot be

improved beyond the reference score. The use of a degraded stream as a reference site will reduce the magnitude of difference between more severely degraded sites and reference sites in the final score. Resultant management goals and approaches may not be ambitious enough to maintain site condition, such that a stable stream community persists.

B-IBI scores can be misleading because they rely so heavily on initial conditions (Fore et al. 1996). If the reference stream is even slightly degraded, B-IBI scores are skewed such that the score reflects a less degraded stream than was actually present. Therefore, the initial condition of the reference site, which might be difficult to assess, should be reported and speculation about its deviation from the pristine state should be included.

Summary

Streams are complex systems of great functional importance to ecosystems and human welfare. Therefore a workable concept of stream integrity must therefore be respected as a complex concept. This review of the B-IBI and its proper use has yielded several recommendations. Although one of the advantages of the index is its communication as a single score, it would be more informative and useful to report raw index values, the magnitude of degradation (e.g., as a percentage), the initial reference site status and a list of all possible disturbances working at the local and watershed scales. Consistency is crucial for sampling and identification, and multiple samples must be taken to reduce the effects of natural variability on the model. There is also immense untapped potential for the use of indicator or keystone species and the incorporation of behavioural or functional metrics.

The Benthic Index of Biotic Integrity can be an effective and useful method for assessing stream condition in forested areas. However, in order for this method to

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be most usefully applied, it is necessary to recognize its limitations and limited scope. Some of these limitations include: metric selection and weighting that do not account for natural variability, unidirectional and linear metric scoring, an inability to identify causal mechanisms, the loss of potentially relevant ecological information, regional specificity and the lack of natural reference streams. There are risks associated with the over-interpretation of B-IBI scores. Forest managers may find that its results are best used as a starting point from which to determine whether further, more detailed investigation is warranted. Provided that it is used as an initial assessment tool, the advantages of the model outweigh its limitations for use in forested watersheds.

References

Alden R.W. III, Dauer D.M., Ranasinghe J.A., Scott L.C. & Llanso R.J. (2002) Statistical Verification of the Chesapeake Bay Benthic Index of Biotic Integrity. *Environmentrics,* 13, 473-498.

Bady P., Doledec S., Fesl C., Gayraud S., Bacchi S. & Scholl F. (2005) Use of Invertebrate Traits for the Biomonitoring of European Large Rivers: The Effects of Sampling Effort on Genus Richness and Functional Diversity. *Freshwater Biology,* 50, 159-173.

Bailey R.C., Kennedy M.G., Dervish M.Z. & Taylor R.M. (1998) Biological Assessment of Freshwater Ecosystems Using a Reference Condition Approach: Comparing Predicted and Actual Benthic Invertebrate Communities in Yukon Streams. *Freshwater Biology,* 39, 765-774.

Bottorff R.L. & Knight A.W. (1996) The effects of clearcut logging on stream biology of the North Fork of Caspar Creek, Jackson Demonstration State Forest, Fort Bragg, CA--1986 to 1994 Unpubl. Final Report prepared for the California Department of Forestry and Fire Protection, Contract No. 8CA3802, May 1996, Sacramento, CA. 177 p.

Bunn S.E. & Davis P.M. (2000) Biological Processes in Running Waters and Their Implications for the Assessment of Ecological Integrity. *Hydrobiologia,* 422, 61-70.

Cao Y., Hawkins C.P. and Vinson M.R. (2003) Measuring and Controlling Data Quality in Biological Assemblage Surveys with Special Reference to Stream Benthic Macroinverterbates. *Freshwater Biology,* 48, 1898-1911.

Carignan V. & Villard M.A. (2002) Selecting Indicator Species to Monitor Ecological Integrity: A Review. *Environmental Monitoring and Assessment,* 78, 45-61.

Carlson J.Y., Andrus C.W. & Froelich H.A. (1990) Woody Debris, Channel Features, and Macroinvertebrates of Streams with Logged and Undisturbed Riparian Timber in Northeastern Oregon, USA. *Canadian Journal of Fisheries and Aquatic Sciences,* 47, 1103-1111.

Cederholm C.J., Kunze M.D., Murota T. & Sibatani A. (1999) Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. *Fisheries,* 24, 6-15.

Creed R.P.Jr. (1994) Direct and indirect effects of crayfish grazing in a stream community. *Ecology,* 75, 2091-2103.

Dale V.H. & Beyeler S.C. (2001) Challenges in the Development and Use of Ecological Indicators. *Ecological Indicators,* 1, 3-10.

Dickes B. (1999) McDonald Creek Benthic Macroinvertebrate Study: A Cumulative Impact Assessment After Timber Harvesting in a Small Watershed. *Washington State Department of Ecology Publication,* no.99-304.

Fore L.S., Karr, J.R. & Wisseman, R.W. (1996) Assessing Invertebrate Responses to Human Activities: Evaluating Alternative Approaches. *Journal of the North American Benthological Society,* 15, 212-231.

Ganason V. & Hughes R.M. (1998) Application of an Index of Biological Integrity (IBI) to Fish Assemblages of the Rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology,* 40, 367-383.

Guerold F. (2000) Influence of Taxonomic Determination Level of Several Community Indices. *Water Research,* 34, 487-492.

Heino J., Muotka T., Mykra H., Paavola R., Hamalainen H. & Koskenniemi E. (2003) Defining Macroinvertebrate Assemblage Types in Headwater Streams: Implications for Bioassessment and Conservation. *Ecological Applications,* 13, 842-852.

Karr J.R. (1981) Assessment of Biotic Integrity Using Fish Communities. *Fisheries,* 6, 21-27.

Karr J.R. (1999) Defining and Measuring River Health. *Freshwater Biology,* 41, 221-234.

Kedzierski W.M. & Smock L.A. (2001) Effects of Logging on Macroinvertebrate Production in a Sand-Bottomed, Low-Gradient Stream. *Freshwater Biology*, 46, 821-833.

Kerans B.L. & Karr J.R. (1994) A Benthic Index of Biotic Integrity (B-IBI) for Rivers of the Tennessee Valley. *Ecological Applications*, 4, 768-785.

Kleindl W.J. (1995) A Benthic Index of Biotic Integrity for Puget Sound Lowland Streams, Washington, USA. Unpublished M.Sc thesis, College of Forest Resources, University of Washington, Seattle USA.

Kotze D.J. & Samways M.J. (1999) Support for the Multi-Taxa Approach in Biodiversity Assessment as Shown By Epigaeic Invertebrates in an Afromontane Forest Archipelago. *Journal of Insect Conservation*, 7, 125-143.

Lammert M. & Allan J.D. (1999) Assessing Biotic Integrity of Streams: Effects of Scale in Measuring the Influence of Land Use/Cover and Habitat Structure on Fish and Macroinvertebrates. *Environmental Management*, 23, 257-270.

Li J., Herlihy A., Gerth W., Kaufmann P., Gregory S., Urquart S. & Larsen D.P. (2001) Variability in Stream Macroinvertebrates at Multiple Spatial Scales. *Freshwater Biology*, 46, 87-97.

Marshall S. & Elliot M. (1997) A Comparison of Univariate and Multivariate Numerical and Graphical Techniques for Determining Inter- and Intraspecific Feeding Relationships in Estuarine Fish. *Journal of Fish Biology*, 51, 526-545.

Mebane C.A., Maret T.R. & Hughes R.M. (2003) An Index of Biotic Integrity (IBI) for Pacific Northwest Rivers. *Transactions of the American Fisheries Association*, 132, 239-261.

McCabe D.L. & Gotelli N.J. (2000) Effects of Disturbance Frequency, Intensity, and Area on Assemblages of Stream Macroinvertebrates. *Oecologia*, 124, 270-279.

Morley S.A. & Karr J.R. (2002) Assessing and Restoring the Health of Urban Streams in the Puget Sound Basin. *Conservation Biology*, 16, 1498-1509.

Novotny V., Bartosova A., O'Reilly N. & Ehlinger T. (2005) Unlocking the Relationship of Biotic Integrity of Impaired Waters to Anthropogenic Stresses. *Water Research*, 39, 184-198.

Parkyn S.M., Rabeni C.G. & Collier K.J. (1997) Effects of crayfish (*Paranephrops planifrons*) on in-stream processes and benthic faunas: a density manipulation experiment. *New Zealand Journal of Marine and Freshwater Research*, 31, 685-692.

Ranasinghe J.A., Frithsen J.B., Kutz F.W., Paul J.F., Russell D.E., Batiuk R.A., Hyland J.L., Scott J. & Dauer D.M. (2002) Application of Two Indices of Benthic Community Condition in Chesapeake Bay. *Environmetrics,* 13, 499-511.

Ravera O. (2001) A Comparison Between Diversity, Similarity and Biotic Indices Applied to the Macroinvertebrate Community of a Small Stream: The Ravella River (Como Province, Northern Italy). *Aquatic Ecology,* 35, 97-107.

Reynoldson T.B., Norris R.H., Resh V.H., Day K.E. & Rosenberg D.M. (1997) The Reference Condition: A Comparison of Multimetric and Multivariate Approaches to Assess Water-Quality Impairment Using Benthic Macroinvertebrates. *Journal of the North American Benthological Society,* 16, 833-852.

Steiner C.F. (2002) Context dependent effects of *Daphnia pulex* on pond ecosystem function: observational and experimental evidence. *Oecologia,* 131, 549-558.

Suter G.W. (1993) A Critique of Ecosystem Heath Concepts and Indexes. *Environmental Toxicology and Chemistry,* 12, 1533 – 1539.

Townsend C.R., Hildrew A.G. & Schofield K. (1987) Persistence of Stream Invertebrate Communities in Relation to Environmental Variability. *Journal of Animal Ecology,* 56, 597-613.

Turak E. & Koop K. (2003) Use of Rare Macroinvertebrate Taxa and Multiple-Year Data to Detect Low-Level Impacts in Rivers. *Aquatic Ecosystem Health and Management,* 6, 167-175.

Weigel B.M., Henne L.J. & Martinez-Rivera L.M. (2002) Macroinvertebrate-Based Index of Biotic Integrity for Protection of Streams in West-Central Mexico. *Journal of the North American Benthological Society,* 21, 686-700.

Willson M.F. & Halupka K.C. (1995) Anadromous fish as keystone species in vertebrate communities. *Conservation Biology,* 9, 489-497.

Synthesis: Stream Integrity & Macroinvertebrates

Streams in forested watersheds subject to harvesting may be considered at risk for reductions in ecological integrity. In this study, I set out to determine whether anthropogenic forest processes, including harvesting and the presence of stream crossings, affected benthic macroinvertebrate (BMI) abundance in the western boreal forests of Alberta. For this purpose I studied BMI in three pairs of watersheds. One watershed in each pair had experienced no harvesting, while in the other harvesting of approximately 10 to 20 percent of the watershed had been conducted in the past five years. The objective of the study was to assess if in-basin harvesting and the presence of crossings altered BMI abundance and to identify potential indicators of stream integrity in this region.

In west-central Alberta most BMI taxa did not respond to ambient levels of harvesting with reduced abundance, suggesting that stream communities were unaffected by the harvesting activity experienced to date (Chapter 2). However, Trichoptera appeared to be very sensitive to harvesting in this region and abundance of larval caddisflies was reduced, and were thus strong indicators of unharvested (high integrity) streams. Abundance of sensitive taxa, such as this, in stream health assessments may be a robust indicator of early environmental change and serve usefully in watershed monitoring, management and conservation. While more tolerant species may have begun their post-harvest recovery or yet be unaffected by forest harvesting, sensitive taxa may be in a state of post-disturbance suppression. The identification of such change at its earliest detectable stages may provide watershed managers with sufficient time to initiate and implement the changes necessary to prevent further, irreversible damage to streams at risk.

Trichoptera and Diptera abundance responded negatively to stream crossings (Chapter 3) and forest harvesting (Chapter 2). Dominance of early instars in all samples made identification further than the order level difficult and beyond the

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constraints of this study. Nonetheless, order-level resolution was adequate to identify harvest and stream-crossing effects in the Trichoptera. Increased levels of taxonomic resolution may have revealed differences in Ephemeroptera and Plecoptera species compositions that could be correlated with stream crossings and forest harvest, as both groups exhibit a wider range of tolerances to disturbance at the genus and species levels (Newbold et al., 1980; Minshall et al., 2001; Benstead et al., 2003).

Riparian vegetation appeared to strongly mitigate the effects of both harvest and sedimentation from stream-crossings, or at least delay the effects of disturbance on the BMI community. It is possible that harvesting in all FMAs was of insufficient intensity to negatively impact all taxa, with only the most sensitive taxa such as the Trichoptera being affected at most sites.

All efforts were made to pair similar harvested and unharvested watersheds for study. However, harvest schedules and limited access restricted site selection, such that the Sheep and Kakwa Creek tributaries were the only useful and practical watersheds available in the Grande Cache forest management area. Unstudied factors may have existed within these watersheds that influenced community structure more strongly than anthropogenic stresses. Slight differences in stream gradient, sinuosity and riparian vegetation, for example, might have resulted in differences in flow regimes affecting the pattern and rate of sediment deposition and removal. Natural riparian buffers were present along all streams, although the composition of the buffers differed. While the unharvested stream was bordered mainly by mature coniferous forest, the harvested stream ran mainly through grasses and low-lying deciduous vegetation. Therefore, streamflow regimes and deciduous riparian vegetation may have been more influential than the effects of stream crossings or harvests on BMI abundance in the harvested watershed streams.

Biotic indices can be useful tools for assessing stream integrity in forested watersheds (Kerans & Karr, 1994; Fore et al., 1996; Danilov & Ekelund, 2001). Although abundance data and indicator taxa can be used to initially assess stream integrity, further, more detailed assessments of stream integrity might be developed using the Benthic Index of Biotic Integrity (Chapter 4), which boasts multiple advantages. However, the B-IBI should be employed with a strong understanding of its construction and its limitations. There are several broad sources of concern that may limit the applicability and interpretation of B-IBI results, including: the selection and weighting of metrics, metric scoring methods, an inability to identify causal mechanisms, the loss of ecological information in final scores, regional specificity and the lack of natural reference streams. Results obtained from this index would best be used as starting points from which to determine whether and where further, more intensive investigation is merited.

Streams are complex, naturally variable systems. Therefore, care and consideration should be given to ensure that sampling is both consistent and accurate, in order to reduce noise. However, all practical applications will face a trade-off between affordable sampling effort and ability to account for this variability. Stream assessments in west-central Alberta should focus sampling efforts on riffle habitats, whose BMI were more abundant than those in pools, whose BMI did not provide new or informative results (Chapter 2). The use of a single habitat for such analyses may also reduce the noise to signal ratio reflected by the natural variability in BMI abundance. BMI abundance patterns in stream assessments should reflect land use patterns and not seasonal differences. Therefore, it is recommended that sampling in this region be completed within a one-month period, as strong seasonal differences in abundance can occur.

Norris & Thomas (1999) suggest that the most useful goal of stream management is to aim for the best possible condition, under reasonable land and water use practices. The use of minimally disturbed sites as reference sites might also be valuable in the rehabilitation of the most disturbed streams when a return to

natural conditions is considered unrealistic (Ganason & Hughes, 1998). In the end, it is far easier to protect streams of high ecological integrity than to restore degraded streams (Morley & Karr, 2002). Careful studies of benthic macroinvertebrates may provide efficient tools for correcting or limiting watershed disturbances before any serious functional ecological damage is done. A reasonable goal is to keep the fauna of watersheds in a condition that permits their recovery to pre-disturbance condition as the main physical-chemical impacts of disturbance decrease with time.

References

Benstead J.P., Douglas M.M. & Pringle C.M. (2003) Relationships of stream invertebrate communities to deforestation in eastern Madagascar. *Ecological Applications*, 13, 1473-1490.

Danilov, R.A. & Ekelund, N.G. (2001) Comparative Studies in the Usefulness of Seven Ecological Indices for the Marine Coastal Monitoring Close to the Shore on the Swedish East Coast. *Environmental Monitoring and Assessment*, 66, 265-279.

Fore, L.S., Karr, J.R. & Wisseman, R.W. (1996) Assessing Invertebrate Responses to Human Activities: Evaluating Alternative Approaches. *Journal of the North American Benthological Society*, 15, 212-231.

Ganason V. & Hughes R.M. (1998) Application of an Index of Biological Integrity (IBI) to Fish Assemblages of the Rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology*, 40, 367-383.

Kerans, B.L. & Karr, J.R. (1994) A Benthic Index of Biotic Integrity (B-IBI) for Rivers of the Tennessee Valley. *Ecological Applications*, 4, 768-785.

Minshall G.W., Royer T.V. & Robinson C.T. (2001) Response of the Cache Creek macroinvertebrates during the first 10 years following disturbance by the 1998 Yellowstone wildfires. *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 1077-1088.

Morley S.A. & Karr J.R. (2002) Assessing and Restoring the Health of Urban Streams in the Puget Sound Basin. *Conservation Biology*, 16, 1498-1509.

Newbold, J.D., Erman, D.C. & Roby K.B. (1980) Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and Aquatic Sciences*, 37, 1076-1085.

Norris R.H. & Thomas M.C. (1999) What is river health? *Freshwater Biology*, 41, 197-209.

Appendix I:
BMI Sampling Procedures

- Begin sampling at the most downstream point and work upstream, so as not to disturb substrate and samples.
- Always stand downstream from Surber when sampling so as not to disturb or alter streamflow.

Riffles

- Place Surber in part of riffle that is visually most representative of the substrate and flow. Select an area of relatively straight flow to minimize losses and maximize catch
- Wash all rocks by hand to remove any attached BMI and remove rock from sampling quadrat (place removed rocks downstream so as not to disrupt or alter flow regime).
- Stir sediment in quadrat with long-handled stick to depth of ~10 cm for 30 seconds.
- Allow water to clear and remove Surber.

Pools

- More difficult to sample due to lack of flow
- Place Surber in part of pool that is visually most representative of the substrate. Select an area of very low flow (test this by placing leaf or stick in middle of pool and observing its movement)
- Wash all rocks and debris by hand to remove any attached inverts and from quadrat (place them away from the pool so as not to kick up any sediment)
- Stir sediment in quadrat with long handled stick
  - Stir for 5 seconds
  - Use hands to wash disturbed sediment into the net
  - Repeat 6 times
- Allow sediment to settle and water to clear, remove Surber

Field Preservation Method

- Take full sediment sample, so as not to lose the smaller inverts
  - Also take any small sticks, algae, etc. and rinse smaller rocks in ethanol
- Preserve in 95% ethanol until sample processing
  - Alcohol that is too concentrated will dehydrate/burn the inverts

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- o BUT there should be enough water in the sample (sediment or organisms) to drop the concentration adequately until you change the alcohol in the lab

Lab Preservation Method

- Strain liquid portion of sample through cheesecloth
- Examine cheesecloth under dissecting scope and replace all BMI into sample
- Add 500 ml of 95% ethanol and 100 ml distilled water to sample, which reduces alcohol concentration to ~80%

Appendix II:
Sorting Procedures

Sample Coding:

U: Unharvested   **DV**: Drayton Valley
H: Harvested   **EDS**: Edson
R: Riffle   **GC**: Grande Cache
P: Pool

- Samples coded as: *Location – Treatment – Habitat*
  - Ex. *DV – U – R*

- Sample label must match vial label. Vials should be half filled with 80 percent ethanol for preservation.

- Small amounts of sediment (i.e. a subsample) are removed from the sample and placed in a dish for sorting under a dissecting microscope. Large pieces of debris (leaves, twigs, etc.) and rocks are washed lightly with 80% EtOH and placed in a 1L beaker. Any matter inspected and removed from the sample should be placed in this beaker – no material is to be discarded.

- Invertebrates are removed individually from the sediment and placed into one of the following six labeled vials:
  - Ephemeroptera (mayflies)
  - Trichoptera (caddisflies)
  - Plecoptera (stoneflies)
  - Odonata (dragon- and damselflies)
  - Diptera
  - Other

- Sorting of each subsample is complete once the sorter has not found an individual for at least five minutes. Once complete, all remaining sediment or matter should be placed in the 1L beaker.

- Once all subsamples (i.e. the entire sample) have been sorted, the remaining ethanol must be inspected. When completed this ethanol should also be placed in the 1L beaker. All sediment and ethanol in the 1L beaker are to be replaced in the original sample container.