

Detecting effects of upper basin riparian harvesting at downstream reaches using stream indicators

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Abstract

Stream evaluation field data from 44 basins in the Bowron River watershed were used in combination with results from GIS spatial analysis to investigate whether impacts from logging the riparian zone of upper-basin streams could be detected at downstream sites. The field data included responses to stream indicator questions taken from the BC Ministry of Forests and Range's Riparian Management Routine Effectiveness Evaluation (RMREE). The evaluation included questions associated with the following stream indicators: (1) channel bed condition, (2) channel bank condition, (3) in-stream large woody debris processes, (4) channel morphology, (5) aquatic connectivity, (6) fish cover, (7) moss, (8) fine sediment, and (9) aquatic invertebrates. This study examined the negative responses to these indicator questions in relation to the amount of upstream riparian harvesting that took place in each basin. Evaluated reaches that had been harvested to the stream bank were not significantly different from sample reaches with streamside buffers when both groups had harvested upstream riparian areas. Negative responses increased significantly at 30% upstream riparian harvest. Sites were grouped by this threshold (low/high) and compared to non-harvested sites to examine negative responses for each indicator. In discussing the results, we explore the potential role of recovery of harvested drainages, negative responses in the non-harvested group, elevation, soil erodibility, in-stream large woody debris processes, and aquatic invertebrate diversity (which may subsequently impact food and habitat supply for fish). The results support the best management practice of leaving a "no-harvest" riparian reserve on all small streams in order to mitigate downstream impacts.

KEYWORDS: *effectiveness evaluation, logging impacts, riparian harvesting, riparian reserves, stream indicators.*

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Introduction

Riparian areas are not only an important link between terrestrial and aquatic ecosystems, but they also mitigate land use impacts to streams. A mature, healthy, and intact riparian area protects stream structure and function from upland disturbance by slowing precipitation runoff (Welsch et al. 2000) filtering sediments (Hicks et al. 1991; Lee et al. 2000), and stabilizing stream banks (Beeson and Doyle 1995). It directly serves aquatic organisms by regulating bank microclimate and water temperature levels (Beschta et al. 1987) and providing food and habitat (Newbold et al. 1980; Gregory et al. 1991; Bilby and Bisson 1992; Bunnell et al. 2004; Olson et al. 2007). Harvesting riparian vegetation has been recognized as detrimental to stream function and aquatic health, prompting the proliferation of riparian zone management regulations in North America (Young 2000; Blinn and Kilgore 2001).

Restrictions on logging in the riparian zone are often set in place to maintain the integrity of stream function and to protect water quantity and quality. Legislation in British Columbia restricts streamside logging through Riparian Management Areas (RMAs), a provision first implemented with the *Forest Practices Act* in 1995. An RMA for fish-bearing streams more than 1.5 m wide consists of a reserve zone (where logging is restricted) and a management zone (which can be selectively logged to protect the reserve zone from the risk of windthrow). The size of these areas varies with stream size, but RMA width generally increases with increasing channel width. Currently, smaller fish-bearing streams (< 1.5 m in channel width) and non-fish-bearing streams do not require a reserve zone. In these cases, the management zone may be logged but it is expected that enough vegetation will be retained to protect the stream from channel bed or bank erosion, to reduce microclimate change, and to maintain and protect wildlife values (BC Ministry of Forests 1995). The interpretation of these objectives is subjective, leading to the application of a wide range of prescriptions, including the common approach of removing all commercial riparian vegetation. Many streams affected by this practice are found in the headwaters of drainage basins that may constitute 60–80% of the total stream length (Shreve 1969) and drain 70–80% of the total basin area (Sidle et al. 2000; Meyer and Wallace 2001).

Smaller fish-bearing streams and non-fish-bearing streams currently do not require a reserve zone. In these cases, the management zone may be logged but it is expected that enough vegetation will be retained to protect the stream from channel bed or bank erosion, to reduce microclimate change, and to maintain and protect wildlife values.

Riparian harvesting of headwater streams reduces the supply of large woody debris to the channel, which may affect other processes within the entire drainage. Normally, streams located in the upper portion of a basin supply water, nutrients, sediment, and organic matter to reaches at lower elevations (Chamberlin et al. 1991; Wipfli and Gregovich 2002; MacDonald and Coe 2007). Downstream habitat quality is partly determined by the delivery rate and timing of these transported materials (Wipfli et al. 2007). Headwater streams (first-order) of a watershed also serve as temporary storage sites for both fine particulate organic matter and sediment from the surrounding forest (Keller and Swanson 1979; Triska and Cromack 1980). Riparian harvesting of these headwater streams reduces the supply of large woody debris and diminishes sediment storage capacity, resulting in a decline in the biological processing of organic material and, consequently, the transfer of energy from terrestrial plants to aquatic organisms (Triska et al. 1982; Triska et al. 1984; Gregory et al. 1987). In addition, because their storage capacities are diminished, harvested first-order streams shunt sediment much more rapidly downstream to higher-order drainages in lower elevations, especially if channelization has occurred (Sedell and Beschta 1991; Hassan et al. 2005). When an increased influx of sediment is delivered to larger systems, species sensitive to turbidity and those that require clean gravel substrate for spawning are affected, ultimately leading to lower biological productivity and reduced species diversity (Platts and Megahan 1975; Berkman and Rabeni 1987).

Based on the above concepts, it is reasonable to assume that the effects of cumulative upstream

riparian disturbance can be detected by evaluating stream conditions lower in the watershed. This can be done through the application of a protocol containing appropriate indicators to a reach located downstream of harvest activity. The data used in this study are the results of the British Columbia Ministry of Forests and Range Riparian Management Routine Effectiveness Evaluations (RMREE) that were performed in the Bowron River watershed in 2007. The RMREE is a monitoring strategy that was developed to meet the sustainable management goals set forth in the *Forest and Range Practices Act (FRPA)*. The RMREE was created for non-specialist government staff to perform operational, routine-level assessments of stream and riparian functioning condition as they relate to fish habitat values. “Properly functioning condition,” first defined in the BC Forest Practices Code and since integrated into the current *FRPA*, is:

the ability of a stream, river, wetland, or lake and its riparian area to 1) withstand normal peak flood events without experiencing accelerated soil loss, channel movement or bank movement, 2) filter runoff, and 3) store and safely release water.

These criteria are the basis for the attribute and indicator questions that structure the RMREE along with fish habitat and passage requirements (Tripp et al. 2007). The level of functioning condition of a reach is determined by the number of negative responses recorded for the set of indicator questions.

Nordin et al. (2009) found that many of the RMREE indicator responses from the 2007 sampling season in the Bowron River watershed were negative with respect to riparian vegetation, suggesting that this attribute had not fully recovered decades after harvesting. They concluded that this result was explained by harvest practices that had occurred prior to the Forest Practices Code, when many streams had been logged to the edge of the bank. Stream indicator responses from the same study were more difficult to interpret at the reach scale because of the lack of significant difference between control and treatment groups. Here, we consider these same results at a watershed scale to investigate whether the amount of upstream riparian harvesting has an effect on downstream indicator response. We examine responses to the nine stream indicator questions from the RMREE in relation to the amount of upstream riparian timber harvested in each of the study drainages to identify any relationships that may not have been evident at the reach scale.

Methods and materials

Site description and the RMREE

Stream evaluation results collected in 2007 from catchments in the Bowron River watershed were used in this study. The Bowron River watershed was extensively logged from the mid-1970s to 1987 in response to a spruce beetle outbreak, and many of the streams were logged to the stream bank, especially smaller tributaries. The 340 300-ha watershed is located in the Central Interior of British Columbia, about 50 km east of Prince George (Figure 1). Overall, the region has a cool and continental climate characterized by long, cold winters and moderately short, warm summers. The Sub-Boreal Spruce (SBS) biogeoclimatic zone is dominant in the watershed with Interior Cedar–Hemlock (ICH) at lower elevations and Engelmann Spruce–Subalpine Fir (ESSF) zones at higher elevations (BC Ministry of Forests 2007). Soils are generally composed of fine-textured surficial materials, including glacial-lacustrine and sandy glacial–fluvial deposits in the lower, middle, and to some extent, upper watershed. Elevation ranges between 600–2440 m, and the annual hydrograph is snowmelt-dominated. The watershed is primarily drained by the Bowron River, which runs north from Bowron Lakes Provincial Park to the Fraser River. The average annual peak flow, measured as a daily average, for the Bowron River (Bowron Box Canyon hydrometric station, 1977–2005) is 319 m³/s. The peak flow recorded in the spring prior to sampling was ranked fourth highest out of 30 since recording began (420 m³/s; Lynne Campo, Water Survey of Canada, pers. comm., September 2007). The Bowron River and its tributaries are important for spawning chinook (*Oncorhynchus tshawytscha*) and sockeye (*O. nerka*) salmon. Populations of Dolly Varden (*Salvelinus malma*), mountain whitefish (*Prosopium williamsoni*), rainbow trout (*O. mykiss*), burbot (*Lota lota*), and white sturgeon (*Acipenser transmontanus*) are also present (BC Ministry of Environment 2007).

Sample reaches, one from each of 44 sub-basins in the Bowron River watershed, were selected from a set of 70 for which data was gathered using RMREEs during the summer of 2007. Each of these 44 sub-basins eventually drains into the Bowron River, but was assumed to be independent based on the topography that dictates spatially distinct drainage patterns. The remaining 26 of the original 70 sampled reaches were located within these same catchments

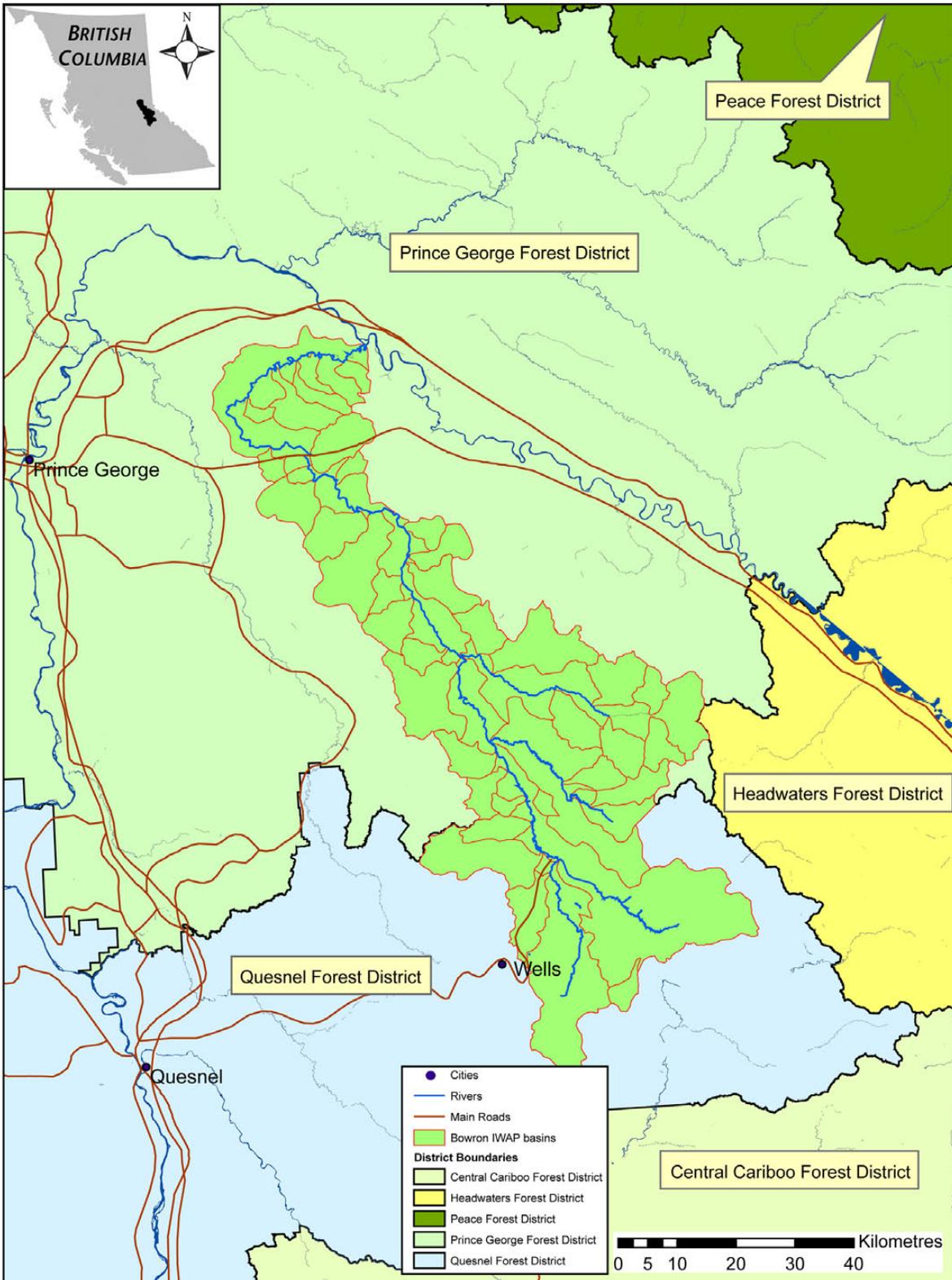


FIGURE 1. Location map of the Bowron river watershed (Source: Nordin et al. 2009; reproduced with the permission of Natural Resources Canada, Canadian Forest Service).

and thus were omitted from this study to reduce spatial autocorrelation in the data analyses. The original report on the functioning condition of these reaches had separated the data by their position (upper/lower) in each of the basins (Nordin et al. 2009), but because this study considers the effect of upstream harvest activity, only the results for the lower reaches were used.

The RMREE consists of a checklist with indicator questions, each of which addresses a component of a properly functioning stream and its adjacent riparian area. Stream and riparian indicators correspond to 15 main questions in the 2007 version of the protocol. Each indicator requires the consideration of one or more associated stream or riparian attributes that are either measured directly or are subjectively compared with an unmanaged reach in the same area. Quantitative measurements are compared to threshold values to help answer the main questions. The number of negative responses to the 15 indicator questions determines the functioning condition of the site. For more information on the procedure, see Tripp et al. (2007).

Out of the 15 main indicator questions in the 2007 version of the protocol, we examined the responses (yes/no) from nine stream indicator questions in relation to the amount of upstream riparian harvest in each drainage (see GIS methods for how this amount was calculated). The nine stream indicator questions and the attributes that are considered to answer the questions (Table 1) are described in detail and rationalized in Tripp et al. (2007). The threshold values that determine a positive or negative response for the attributes and the indicator questions vary with stream type (non-alluvial, riffle-pool, cascade-pool, step-pool) and can be found in the RMREE protocol (Tripp et al. 2007).

GIS

Watershed boundaries for each of the sub-basins ranging from 91 to 25 127 hectares in area were drawn in ArcMap using topography and water feature information (Figure 2). Sample reaches were located at the lowest point in each of the drainages. The amount of riparian harvesting that was done in the drainages was derived using a GIS spatial analysis model with provincial forest harvesting and stream data layers obtained from the BC Ministry of Forests and Range. Harvesting was coded as “riparian” if it occurred within 10 m of any stream bank. This distance was

selected based on the recommended riparian buffer for best management of small streams in the BC Ministry of Forests *Riparian Management Area Guidebook* (1995). The length of riparian harvesting in a drainage system was divided by total stream length to get percent riparian harvest for each sample drainage. The sample size included sites that had 0–94% of their total upstream drainage length harvested. An example of one of the drainages with notable riparian harvesting is in the Ketchum Creek area (Figure 3). Nine sample drainages were not harvested within their riparian areas. Out of these nine reaches, seven also had non-harvested upland areas and the remaining two had less than 5% of their uplands harvested. Percent upland harvesting was calculated using basin area and harvested polygons outside the riparian zone. Out of the 35 sites that were subject to upstream riparian harvesting, 24 had been logged within 10 m of the stream edge at the sample reach. Data from other possible covariates were retrieved using GIS queries for each site, including: number of upstream road crossings; elevation at the sample reach; total drainage basin area; and harvest date when the majority of the catchment area was logged.

Data Analyses

Stream indicator responses from RMREEs conducted in 2007 for the 44 sample reaches were used in regression analyses to identify relationships with the amount of upstream riparian harvesting in each drainage. Prior to this study, several of the evaluation answers were statistically adjusted to compensate for site-specific variability at the reach level, including stream gradient, channel width, coupling (the hillslope influence of material transfer to a stream), and channel soil erodibility. Specifically, the adjustment was done by applying a general linear model (Systat version 11, Richmond, CA) to the attribute measurements that were used to answer the indicator questions. The indicator responses were then re-evaluated based on the adjusted means of the attributes (see Nordin et al. 2009 for details). The removal of the effect of these localized covariates allows us to assume that interpretations from this analysis on watershed-scale characteristics will not be confounded by variability among reaches due to site-specific environmental heterogeneity and stream size. The adjustment was completed with only the indicators that were associated with quantitative and continuous attributes, so not every stream indicator was corrected in this manner.

TABLE 1. Stream indicator questions from the Riparian Management Routine Effectiveness Evaluation. Responses that were used in this study were given after consideration of the listed attributes. Thresholds for the attributes vary with stream type and can be found in Tripp et al. (2007).

Indicator question	Indicator question attributes
1. Is the channel bed undisturbed?	<ul style="list-style-type: none"> • mid-channel bars • sediment wedges • multiple channels • lateral bars
2. Are the channel banks intact?	<ul style="list-style-type: none"> • bank disturbance • deep-rooted vegetation • stable, undercut banks • recently upturned root wads
3. Are channel large woody debris (LWD) processes intact?	<ul style="list-style-type: none"> • logging-related LWD • accumulations of LWD which span the channel • LWD positioned parallel to the stream • apparent removal of LWD by equipment or weather events
4. Is the channel morphology intact?	<ul style="list-style-type: none"> • pools • deep pools (2× riffle depth) • sediment texture heterogeneity
5. Are all aspects of the aquatic habitat sufficiently connected to allow for normal, unimpeded movements of fish, organic debris, and sediments?	<ul style="list-style-type: none"> • post-harvest blockages anywhere in the reach • crossing structure-related accumulations • downcutting • channel diversion • dewatering
6. Does the stream support a good diversity of fish cover attributes?	<ul style="list-style-type: none"> • boulders • organic material • deep pools • aquatic vegetation • overhanging vegetation • undercut banks • a stable mineral substrate with void spaces
7. Does the amount of moss present on the substrate indicate a stable and productive system?	<ul style="list-style-type: none"> • presence and condition of moss
8. Has the introduction of fine inorganic sediments been minimized?	<ul style="list-style-type: none"> • fine sediment (< 4 mm in diameter) • substrate embeddedness • single large areas of particularly soft patches of sediment • sensitive invertebrates
9. Does the stream support a diversity of aquatic invertebrates?	<ul style="list-style-type: none"> • sensitive invertebrate species • insects • major invertebrate groups • total invertebrate species

The indicators that were not adjusted because they lacked appropriate data for a general linear model included aquatic connectivity, fine sediment, and in-stream large woody debris (LWD).

Several of the sites that had been harvested upstream had a riparian buffer at the sample reach. To test whether the indicator results were different between these sites and those without a riparian

buffer, we grouped all sites with upstream harvesting by presence or absence of a riparian buffer (10 m) at the sample reach. The negative indicator responses for the two groups of sites were tested for a significant difference using a Student's *t*-test. Once it was determined that the absence of a riparian buffer at the sample site did not influence negative indicator responses, a linear regression was done on the

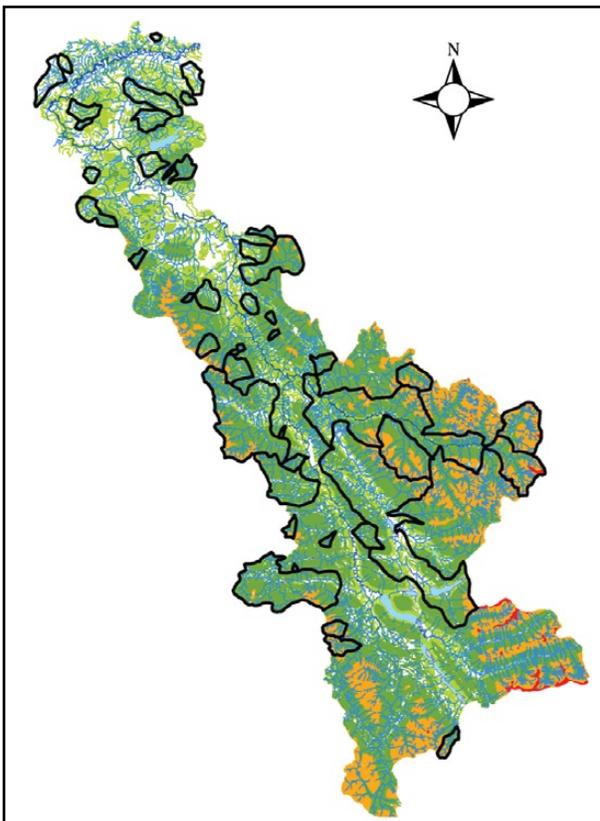


FIGURE 2. Location of sample drainages in the Bowron River watershed.

number of negative responses and percent upstream riparian harvesting using all of the data to explore any relation between headwater harvesting and downstream negative indicator response. A plot of the residuals against the estimate was checked to ensure homogeneity of variance.

Next, segmented regression analysis software was used (SegReg 2008) to identify any breakpoints in the data at which the number of negative responses per site increased with respect to upstream riparian harvest. Segmented regression applies linear regressions to a data set (x, y) that does not normally have a strong linear relation by introducing one or more breakpoints, whereupon separate linear regressions are performed for the linear segments. The breakpoints are located using a method to calculate confidence intervals, and the breakpoint contributing to the smallest interval (i.e., the ideal breakpoint) is selected using an appropriate function type for each unique data set. The selected function type is one that maximizes the coefficient of explanation and produces a breakpoint that passes

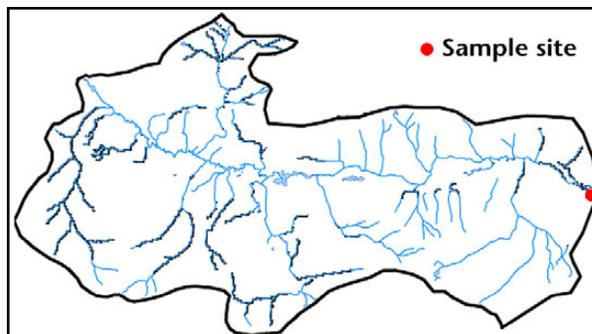


FIGURE 3. A sample catchment in the Ketchum creek drainage basin. Riparian harvesting is shown in black.

a test of significance based on an alpha value of 0.05 (Oosterbaan 2005). Significance is built into the results using this method and the outcome is graphed with confidence limits. This specific software has a wide application and has been used in other studies relating to crop production (Oosterbaan et al. 1990; Oosterbaan 1994), zooplankton size (Korosi et al. 2008), and fish distribution (Hilton et al. 2008). After a significant breakpoint was found, data were grouped using this figure as a threshold. Indicators were then examined separately among groups to identify negative responses within each indicator, and differences among the categories were tested using Chi-square tests of homogeneity (Sokal and Rohlf 1995).

To assess the influence of other watershed factors on indicator response, multiple logistic regressions were performed on each indicator that showed no significant difference among harvest groups (channel bed disturbance, fine sediment, moss, fish cover, fish connectivity, morphology, and bank disturbance). This was done to look for relationships that may not be directly due to upstream riparian harvesting and to explain any negative indicator responses in non-harvested drainages. The watershed factors that were included as independent variables were: drainage area, elevation (at lowest part of survey), stream density, number of road crossings, percent upland harvesting, and major soil type for each basin (Table 2). The year when most of the harvesting occurred within each catchment was also included because, in some cases, logging of specific drainages occurred years earlier than others, which allowed for potential recovery. Reaches that had been marked as “N/A” for an indicator (as in the case of channel morphology for non-alluvial streams) were omitted from analyses for that indicator. Channel width was included with the independent factors in the logistic

TABLE 2. Negative responses were used from the listed indicator questions in logistic regressions with the specified watershed variables to test for influences in negative indicator responses that could not be linked to riparian vegetation removal.

Watershed variables	Indicator question number
Percent upland harvest	1,2,4,5,6,7,8
Number of road crossings	1,2,4,5,6,7,8
Main soil type	1,2,4,5,6,7,8
Stream density	1,2,4,5,6,7,8
Harvest year	1,2,4,5,6,7,8
Elevation (m)	1,2,4,5,6,7,8
Channel width	5
Soil erodibility at site	8

regressions for aquatic connectivity because it was not adjusted for in the previous linear corrections to the data. Similarly, soil erodibility at the channel reach was included for the fine sediment regressions as this data was also not appropriate for the corrections previous to this analysis. With the threshold for a negative response for the aquatic connectivity indicator question set at just one blockage and in consideration of recent flooding (2007), the number of blockages at each site with a negative response was analyzed further using ANOVA (with one outlier identified and removed).

Results

Indicator response

Many reaches where the evaluations took place had been logged to the stream bank, which could make the evaluation of effects related to upstream riparian harvesting difficult to determine. To assess whether the removal of the riparian timber at the sample site influenced our results, we compared the number of negative responses from sites representing complete riparian harvest at the sample reach with those from sites that had a riparian buffer of at least 10 m in the harvested group. Although the mean number of “no” answers was lower for buffered sites (mean 2.8, $n = 11$) than non-buffered sites (mean 3.4, $n = 24$), this difference was not significant (Student’s t -test $p = 0.19$). We then assumed that there was no confounding effect on the data from riparian harvesting at the sample reach and all data were grouped together for subsequent analyses.

A positive relationship was found between the number of negative answers per site and the percent of

upstream harvesting that was done when tested with a linear regression ($p = 0.049$). The breakpoint analysis found a significant threshold in the data using a best-fit function of two horizontal segments at different levels. This means that the regression coefficient values to the left and right of the breakpoint were determined to be insignificant, while the average Y values (negative responses) on each side of the breakpoint differed significantly with 95% certainty. These results suggest a riparian harvesting threshold of 30% for increasing stream indicator failures (Figure 4). Further results are based on the grouping of sites by this threshold value.

The average number of negative responses for the nine stream indicators was 2.4, 2.7, and 3.6 for the non-harvested, 1–30%, and greater than 30% riparian harvested groups respectively. Hereafter, the 1–30% group will be known as “low” and the greater than 30% group will be known as “high” with respect to riparian harvesting.

Although the overall number of negative answers in response to the stream indicator questions fell into a predictable pattern with respect to riparian harvesting, the distribution of negative responses (plotted as a percent per harvest group) among the specific stream indicators was varied (Figure 5). The non-harvested group did not have any negative responses for channel bed disturbance, LWD processes, or aquatic invertebrate diversity, but did have unexpected negative responses to the remaining indicator questions.

Only two indicators showed a significant difference in response among the sites grouped by upstream riparian harvest. First, there were higher negative responses for the aquatic invertebrate indicator in the high harvested group compared to the other two groups ($\chi^2 > 5.99$, $df = 2$, $p = 0.034$). Second, the LWD indicator had higher failures for both harvested groups compared to the non-harvested group ($\chi^2 > 5.99$, $df = 2$, $p < 0.001$). Most of the LWD failures were attributed to the abundance of logging-related debris observed in the channel and identified as such by their mechanically cut ends. The non-harvested, low, and high groups were not as strongly divided for each of the other seven indicators and, with the exception of the channel bed disturbance indicator, the non-harvested group displayed negative responses.

Influences from watershed characteristics

No significant prediction relationships were found between in-stream fine sediment and the watershed characteristics (basin soil type, stream density, harvest date, elevation, road crossings, and percent upland

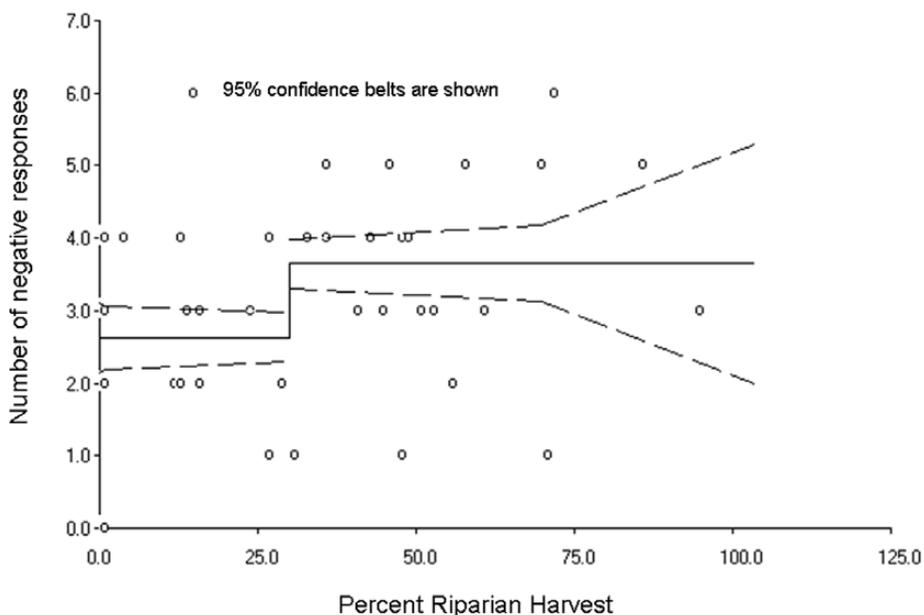


FIGURE 4. Segmented regression analysis showing a breakpoint in the number of negative indicator responses at 30% upstream riparian harvest. *Note: Graph does not delineate multiple points with identical values.

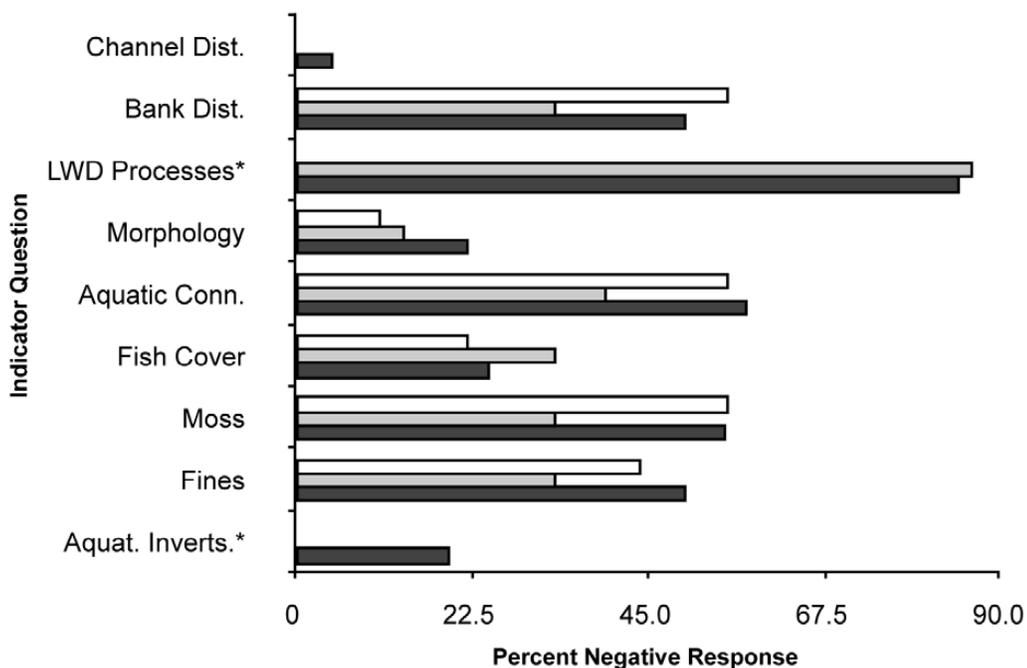


FIGURE 5. Percent of negative responses for the stream indicators by riparian harvest groups. Asterisk denotes significant difference among groups.

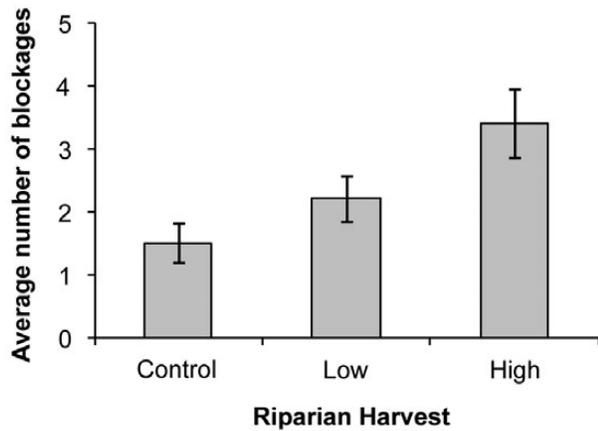


FIGURE 6. Mean number of blockages at sites with negative blockage indicator response grouped by riparian harvest. Error bars represent standard error.

harvest) using multiple logistic regression. However, the fine sediment indicator had not been adjusted for site-specific factors prior to this analysis because of the wide range of conditions under which it was found. It was included here and was found to be a significant predictor variable at the reach scale ($p = 0.049$). Sites that were found to have bed and bank material that were highly erodible at the sample reach displayed significantly higher failure rates for the fine sediment indicator.

Elevation was the only watershed variable that the logistic regression identified as a significant predictor for the moss indicator ($p = 0.049$). The presence of moss was more prominent at lower elevations, with the average elevation for a passing indicator score at 884 m compared to the average of 962 m for a failing score. When average elevation of each sample group was compared, the non-harvested reaches were found to be at higher elevations (median 1091 m) than the streams representing riparian harvesting (median 914 m).

None of the watershed-scale variables were significant predictors for the aquatic connectivity indicator. However, channel width proved to be a significant predictor variable for aquatic connectivity at the reach scale ($p < 0.01$) with more failures in smaller channels. Seventy-eight percent of the non-harvested channels were less than 5 m wide, and 33% were less than 2 m wide. The non-harvested group had the lowest average number of aquatic blockages and the “high” harvested group had the highest (Figure 6), but the difference among groups was not statistically significant (ANOVA, $p = 0.38$).

The channel bed, fish cover, stream morphology, and intact bank indicators were not significantly associated with any of the watershed-scale variables.

Discussion

Indicator response

This study found that harvesting 30% of the upstream riparian zone in a drainage basin can influence stream processes enough to be detected at a point downstream using a comprehensive set of indicators. Although it is recognized that headwaters contribute to downstream aquatic ecological processes (Wipfli and Gregovich 2002; Compton et al. 2003) and that harvesting can affect the distribution of materials downstream (Wipfli et al. 2007), there are no current suggestions as to the quantity of headwater riparian vegetation necessary to maintain downstream structure and ecological function. This is likely because supporting studies measuring change in downstream habitat in relation to upstream riparian harvest activity are scarce. However, our breakpoint of 30% is comparable to harvesting thresholds as they relate to detectable changes in annual water yield. Bosch and Hewlett (1982) found a 10% change in cover caused approximately a 40-mm change in annual water yield in coniferous forests, but logging effects could not be identified below a threshold of 20%. A compilation of catchment studies in Canada also suggests a 20% minimum of catchment area harvest for a measurable annual water yield increase and that each 10% increase in area harvested may increase water yield by approximately 15 mm, but their results were variable (Hetherington 1987). Stednick (1996) found that annual water yield change was detected at 15–45% of basin harvesting, depending on the region. Their Rocky Mountain/Inland Intermountain region data suggest that a 15% harvest area results in a measurable annual water yield increase, but that the rate of increase is variable, especially when the harvest area exceeds 30%.

This study found that harvesting 30% of the upstream riparian zone in a drainage basin can influence stream processes enough to be detected at a point downstream using a comprehensive set of indicators.

LWD and aquatic invertebrates

LWD processes and aquatic invertebrate diversity were the indicators that showed significant differences in responses among sites grouped by degree of riparian harvest, with zero negative responses in the non-harvested category. Field observations determined that the LWD failures were mainly due to logging-related debris that remained in the stream channel as identified by their mechanically cut ends. The presence of this 20–30-year-old woody debris not only contributes to large quantities of blockages, but also is no longer useful as a source of nutrients, which are mainly found in the needles and twigs of a tree (Hyvönen et al. 2000).

Streams might even be nutrient-poor because of logging, which could explain the invertebrate indicator responses in “high” harvested catchments. While nutrients such as nitrates may increase initially after harvest (Likens et al. 1970; Brown et al. 1973; Swank 1988; Martin et al. 2000), levels may eventually drop to below pre-harvest conditions (Johnson and Swank 1973; Vitousek and Reiners 1975; Vitousek 1977) as concentrations begin to accumulate in recovering plant biomass, reducing levels of nutrients in soil solution and stream water (Likens et al. 1970; Brown et al. 1973). Thus, even when sunlight to the stream increases due to a harvested riparian area, primary production may be limited. A post-harvest study on small streams in the same watershed has shown that levels of nitrogen and phosphorus are low and may be limiting (E. MacIsaac, Department of Fisheries and Oceans, pers. comm., September 2007). This would potentially restrict the autochthonous food supply for invertebrates.

Logging the riparian area also reduces input of fine and coarse organic particulate matter normally supplied to streams as litterfall from streamside and overhanging vegetation. Riparian vegetation is important to streams and the biota they support because their allochthonous inputs contribute to the energy supply of the aquatic food web (Bilby and Bisson 1992; Naiman et al.

2005). Piccolo and Wipfli (2002) found that in stands 35–40 years post-harvest, abundance and biomass of macroinvertebrates were lowest in streams buffered by young conifers, and highest in those with alder-dominated riparian zones. However, this correlation is not apparent for all woody shrubs and deciduous trees typical of forest regeneration. Irons et al. (1988) determined that alder was preferred by a shredder invertebrate species over poplar, birch, and willow, and that consumption was negatively associated with tannin content. They concluded that it is the species and nutrient composition of riparian vegetation which influences detrital food webs in streams. It is possible that due to extensive harvesting, the treatment streams in the Bowron River watershed are now low in the required nutrients for algal growth and/or do not have the riparian regrowth and related food supply necessary to support a level of invertebrate diversity comparable to non-harvested reaches.

Natural variability and potential recovery

There are two possible reasons that the harvested groups did not have significantly higher negative responses than the non-harvested group for the channel bed, fine sediment, moss, fish cover, aquatic connectivity, channel morphology, and intact banks indicators. One explanation is that there were negative responses in the non-harvested group for 6 out of 7 of these indicators, possibly due to natural variation in watershed parameters. For example, several of the non-harvested reaches had a negative response for the fine sediment indicator, and we found a significant statistical relationship between fine sediment and soil erodibility at the sample site. This suggests that pre-existing local soil conditions dictated the negative response at the non-harvested reaches and likely influenced the responses at several of the harvested reaches, confounding the results. Negative responses in the non-harvested category for the moss and aquatic connectivity indicators may be due to elevation and narrow channel widths, respectively. High water levels the previous spring may have contributed to some of the aquatic connectivity blockages seen in smaller channels for both the harvested and the non-harvested channels. Field observations confirm that many of these blockages were recent log jams and associated sediment wedges.

The negative responses for channel bed, intact banks, stream morphology, and fish cover indicators were more difficult to explain for the non-harvested group. A number of attributes are measured and compared

LWD processes and aquatic invertebrate diversity were the indicators that showed significant differences in responses among sites grouped by degree of riparian harvest, with zero negative responses in the non-harvested category.

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to threshold values in consideration of an indicator response, and it is possible that some of these are sufficiently correlated with natural variability to produce an occasional negative response. For example, the indicator question regarding fish cover considers seven different types of attributes (boulders, organic material, deep pools, aquatic vegetation, overhanging vegetation, undercut banks, and a stable mineral substrate with void spaces) that could vary independently with environmental factors. According to the RMREE, if three of the seven types of fish cover were absent, the indicator was to be given a negative response (Tripp et al. 2007). Field observations confirmed that boulders and a stable mineral substrate with void spaces were usually absent at sites in areas of glacial lacustrine soils where the channel bed and bank material were composed mainly of fine sediments. Consequently, this may weight these sites unfavourably when considering the fish cover indicator question. Similarly, moss was rarely seen at these same sites, thereby influencing a negative response. Recent flooding likely contributed to many of the negative responses for channel bed and bank condition in all groups, potentially masking any previous disturbance discrepancy attributable to logging.

The second explanation for non-significant differences in indicator results among groups is the long period of recovery since harvest, which allowed for the restoration of many hydrologic processes (e.g., streamflow magnitude), thereby diminishing impacts from logging. Increases in streamflow following harvesting can directly affect the physical, chemical, and biological processes of a stream (Naiman et al. 2001), and many of these impacts would be detected through the indicators in this study. However, changes in streamflow in response to forest harvesting generally do not persist over time due to vegetative regrowth (Harr 1976; Austin 1999; Jones 2000) and recovery

time may vary in relation to regeneration stand height, canopy density, elevation, species, precipitation regimes, site index, and relative maturity (Winkler et al. 2005; Strimbu et al. 2006), making any predictions complex. Others have found that the combination of forest harvest and roads can increase, decrease, or have no significant effects on peak flows (Harr 1976; Harr and McCorison 1979; Austin 1999; Moore and Wondzell 2005). These discrepancies may depend on the size of the basin as well as the structure and morphology of the drainage network and valley bottoms (MacDonald and Coe 2007). Direct comparisons of any affected indicators can be difficult due to different patterns of flow-related disturbance specific to each basin. Further study is required to determine the complex interactions of weather events, watershed characteristics, streamflow response, and associated indicators.

In addition to buffering storm runoff, vegetation regrowth can also reduce the input of sediment to a stream. MacDonald et al. (2003) found that it took only 2 or 3 years after harvesting for sediment levels to decrease to pre-harvest conditions in streams located in the province's Central Interior, depending on the amount of riparian retention. Field observations confirm the abundance of young deciduous trees species and woody shrubs such as trembling aspen (*Populus tremuloides*), red-osier dogwood (*Cornus stolonifera*), alder (*Alnus* spp.), and willow (*Salix* spp.) now established along many of the stream banks and providing an effective filter for sediments and increasing bank stability (Hicks et al. 1991). It was also noted that at most catchments, the only roads currently still in use were at the sample reaches located at the lowest part of the drainage, and evaluations were conducted upstream from the road. Most of the secondary and tertiary logging roads farther upstream had been deactivated for well over a decade and were partially if not fully revegetated, with bare sections armoured with coarse gravels. The elimination of these sources of fine sediment would potentially improve the fine sediment indicator response as well as indicators affected by fine sediment, such as invertebrate diversity.

Conclusions

Studies have shown headwater streams to be directly linked to downstream ecosystem processes (Wipfli and Gregovich 2002; MacDonald and Coe 2007) but measuring the effect of harvesting in these areas can be difficult, especially after a period of recovery has taken place. When all RMREE indicators were grouped

The objective of many carefully researched protocols is to provide an assessment based on a set of applicable indicators rather than numerous precise measurements for any one indicator.

together, the negative responses increased significantly at 30% riparian harvesting (meaning that riparian vegetation within 10 m of the stream bank was harvested along 30% of the total upstream drainage length), which is comparable to the results of studies investigating changes in annual water yield following logging (Bosch and Hewlett 1982; Hetherington 1987; Stednick et al. 1996). When indicators were examined separately however, effects from harvesting were difficult to confirm, suggesting that the whole (set of protocol indicators) is more significant than the component parts (individual indicators) in an ecological context. A comprehensive evaluation is designed to capture a range of possible negative impacts and any one watershed may respond to disturbance differently, depending on its characteristics. By basing the evaluation on a set of applicable indicators, we are able to effectively capture the range of variation and also identify any possible cumulative effects that may be more apparent when examined together than when they are considered separately. The objective of many carefully researched protocols is to provide an assessment based on a set of applicable indicators rather than numerous precise measurements for any one indicator.

Indicators that showed significant relationships with only watershed characteristics may have started to recover from the impacts of riparian timber removal. Also, the attribute responses by which the indicators are scored could be confounded by variability of landscape characteristics and hydrologic processes among watersheds. MacDonald and Coe (2007) found that the high spatial and temporal variability in the delivery of materials from the hillslopes into headwater streams and from headwaters to downstream reaches made predictions complex.

We saw significantly higher negative indicator responses for the LWD and aquatic invertebrate diversity indicators in reaches that drained streams with “high” harvested riparian areas. The LWD negative responses were attributed to 20–30-year-old residual

logging debris in the stream channel, which could also affect fish passage by contributing to blockages. Logging riparian vegetation completely will result in inadequate protection and diminished supply of LWD to headwater streams for decades, which can affect the sustainability and integrity of downstream habitats (Wipfli et al. 2007). The lower diversity of aquatic invertebrates is likely a related effect that could be caused by a reduced supply of autochthonous and/or allochthonous food sources. If macroinvertebrate biomass decreased concurrently with diversity, food supply for fish would be compromised. It is possible that macroinvertebrates in the harvested basins have been affected by other logging-related factors such as large temperature fluctuations or fine sediment fluxes that occur commonly when riparian vegetation is removed—these factors could also be detrimental to fish as both groups are associated with similar suites of environmental variables (Kilgour and Barton 1999). This investigation emphasizes the importance of riparian retention in smaller headwater streams, and also the need for further studies on fish populations in the Bowron River watershed.

Although harvesting in this study took place prior to the Forest Practices Code, the results are applicable to today’s practices which do not currently include a mandatory buffer for smaller streams. The BC Ministry of Forests and Range (1995) best management practices recommend retaining all trees within 10 m of small streams to provide protection for fish, wildlife, and water quality. Rex (2009) found that there was a significant improvement in the RMREE indicators in streams near Vanderhoof that had 5–10 m of riparian vegetation compared to streams with a 0–5-m buffer at the sample reach. This study further strengthens the recommendation of increased retention, not only to improve stream processes at the reach scale, but also to mitigate impacts further downstream.

The BC Ministry of Forests and Range (1995) best management practices recommend retaining all trees within 10 m of small streams to provide protection for fish, wildlife, and water quality.

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Test Your Knowledge . . .

Detecting effects of upper basin riparian harvesting at downstream reaches using stream indicators

How well can you recall some of the main messages in the preceding Research Report?

Test your knowledge by answering the following questions. Answers are at the bottom of the page.

1. Current legislation in British Columbia restricts riparian logging by requiring a reserve zone along:
 - A) Streams of all sizes
 - B) Streams where cattle grazing is abundant only
 - C) All fish-bearing streams
 - D) Fish-bearing streams with channel width of 1.5 m or more

2. Headwater streams supply lower reaches with:
 - A) Nutrients
 - B) Sediment
 - C) Organic matter
 - D) All of the above

3. A significant increase in negative responses to downstream indicator questions was seen in the Bowron River watershed when the amount of upstream riparian harvesting exceeded:
 - A) 10%
 - B) 20%
 - C) 30%
 - D) 40%

ANSWERS

1. D 2. D 3. C