

**INTEGRITY OF RIPARIAN AND AQUATIC HABITAT IN TWO  
AGRICULTURALLY IMPACTED STREAM VALLEYS OF SOUTHERN  
BRITISH COLUMBIA**

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## **Abstract**

Habitat patterns in and along two midsize streams of the Thompson River drainage, British Columbia, were examined using multivariate - statistical (factor) analyses. Terrestrial data consisted of spot counts of floral-growth forms and substrata in the lower riparian zone and qualitative assessments of floral intactness in the upper riparian zone. Results of the analyses suggest that lower-riparian vegetation does not accurately reflect floral conditions in the upper riparian zone; thus, assessments of stream habitat integrity require examination of both zones. Aquatic data was collected on stream width, lateral microhabitats and hydraulic-mesohabitat characteristics along longitudinal and deforestation gradients. Expected habitat differences, such as increased pool:riffle ratio and habitat diversity, were not consistent along the longitudinal gradient, probably because sampling was limited to midsize streams and natural features confounded patterns. Similarly, expected habitat differences along the riparian-intactness gradient were inconsistent. The latter was probably due to unaddressed factors such as sedimentation. Further studies, including holistic physiochemical analyses are necessary to establish what riparian and aquatic habitat factors consistently change with deforestation in the southern interior of British Columbia.

## Résumé

Les configurations de l'habitat le long de deux cours d'eau de taille moyenne du bassin versant de la rivière Thompson (Colombie-Britannique) ont été examinées à l'aide d'analyses statistiques multivariées (factorielles). Les données terrestres consistaient en dénombrements ponctuels de formes de croissance florale et des sous-couches dans la zone riveraine inférieure et en évaluations quantitatives du caractère vierge de la flore dans la zone riveraine supérieure. Les résultats des analyses suggèrent que la végétation de la zone riveraine inférieure ne reflète pas avec exactitude l'état de la flore dans la zone riveraine supérieure; ainsi, les évaluations de l'intégrité de l'habitat des cours d'eau exigent l'examen des deux zones. On a recueilli des données sur la largeur des cours d'eau et sur les caractéristiques de leurs microhabitats latéraux et de leurs mésohabitats hydrauliques suivant des gradients longitudinaux et de déforestation. Les différences attendues au niveau de l'habitat, comme un accroissement du rapport bassins/seuils et de la diversité de l'habitat n'étaient pas uniformes suivant le gradient longitudinal, probablement parce que l'échantillonnage se limitait aux cours d'eau de taille moyenne et parce que les entités naturelles rendaient les configurations confuses. Semblablement, les différences attendues au niveau de l'habitat suivant le gradient de virginité de la zone riveraine étaient irrégulières. Ces dernières étaient probablement attribuables à des facteurs non abordés comme la sédimentation. D'autres études, notamment des analyses physico-chimiques holistiques, sont nécessaires afin d'établir quels facteurs de l'habitat riverain et aquatique changent de manière uniforme en fonction du déboisement dans l'intérieur méridional de la Colombie-Britannique.

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## 1.0 Introduction

Riparian (floodplain) vegetation is important for maintaining lotic and riparian ecosystems (habitat and biota) in the Pacific Northwest (Sedell and Swanson 1984; Salo and Cundy 1987; Raedeke 1988; Gresswell et al. 1989), including British Columbia (BC) watersheds (Morgan and Lashmar 1993). Fisheries biologists often assess riparian habitat because of the importance of trees and woody debris (snags) in providing shade, cover, habitat, food, and other benefits to salmonids (Theurer et al. 1982, 1985; Beschta and Platts 1986; Woessner and Potts 1989) and other lotic animals (Harmon et al. 1986; Salo and Cundy 1987; Platts et al. 1987; Bartholow 1989). Similarly, ornithologists often measure foliage-height (growth-form) and tree-species composition and snag density because of the importance of riparian vegetation and snags for supplying avian foods (Yeager 1955; Nudds 1977; McIntosh 1986; Verner et al. 1986) and structuring bird (DesGranges 1980; Stauffer and Best 1980; Rice et al. 1983, 1984; Diamond 1987) and other riparian-faunal assemblages (Harmon et al. 1986; Szaro and Rinne 1988).

River valleys often show greater diversity of floral growth forms and wildlife than other terrestrial habitats (Thomas et al. 1979a,b; Porter 1981). Streams in western North America show changing dominance from herbs to shrubs to deciduous trees in a lateral progression from stream edge to the upper-riparian zone, adjacent upland habitats being dominated by large conifers in mesic areas (Fonda 1974; Thomas et al. 1979b; McGarigal and McComb 1992; Kistritz and Porter 1993; Morgan and Lashmar 1993) and herb-shrub vegetation (e.g., bunchgrass and sagebrush) in drier, inland regions (Thomas et al. 1979a; Szaro and Rinne 1988; Hickey and Trask 1994). Deforestation from agricultural, logging, and other riparian activities often reduces growth-form diversity on the streambanks and floodplain by direct (cutting) and indirect means (loss of local seed sources) (Lynch et al. 1977; Kauffman and Krueger 1984; Swift 1984; Knopf et al. 1988). To my knowledge, however, statistical analyses have not been done to determine if all vegetation zones along the lateral gradient are similarly affected by human impacts.

Aquatic-habitat classification is commonly done by fisheries researchers in the northwestern U.S. (Helm 1985; Dolloff et al. 1993; Hawkins et al. 1993) and BC (de Leeuw 1982; DFO and MELP 1989) to assess habitat change along longitudinal (up- to downstream) and human-impact



gradients. Classification systems include stratification by lateral-macrohabitat (e.g., main-channel vs. edge habitats) and hydraulic-mesohabitat characteristics (e.g., pools vs. riffles) (Vadas 1992, 1994; Hawkins et al. 1993); these higher-level systems simplify and help standardize habitat assessment and may be more efficient than detailed microhabitat measurements (Karp and Matthews 1988; Vadas 1994; Anon. 1995c). In general, edge (backwater and side-channel) and slow-deep habitats (e.g., medium and deep pools) are larger (lower habitat density) and more abundant downstream (de Leeuw 1982; Hogan 1986; Baker et al. 1991; Myers and Swanson 1991), whereas fast-shallow habitats (e.g., fast riffles and waterfalls) are more prevalent in steeper-sloped, headwater streams (de Leeuw 1982; Grant et al. 1990; Kershner et al. 1992; Hubert and Kozel 1993). Deforestation and instream-habitat alteration from logging and agricultural activities in the western U.S. (Behnke 1977; Berkman and Rabeni 1987; Salo and Cundy 1987; Myers and Swanson 1991; Schroeder and Allen 1992) and British Columbia (Narver 1972; Hogan 1986; Roberts 1987; Tripp 1994) generally cause (1) increases in slow-riffle and run habitats, sedimentation, and erosion; (2) losses of edge and slow-deep habitats, instream-woody cover, and stream stability; (3) habitat homogenization (decreases in habitat-unit density and diversity); (4) divergence of pool:riffle [P:R] ratios away from unity; and (5) other physicochemical changes. In British Columbia (BC), deforestation usually causes channel widening (Hogan 1986; Roberts 1987), although not invariably (Narver 1972).

In the present paper, simple habitat-classification systems are developed to allow rapid assessment of lower-riparian vegetation and aquatic-habitat composition along two inland streams in BC. The study focuses on differences in riparian-floral growth forms and aquatic-habitat types in relation to deforestation on the floodplain, via multivariate-statistical (factor) analyses and various indices of habitat diversity and quality. The lower-riparian assessment was hypothesized to be a quick, efficient, surrogate method for measuring intactness of upper-riparian (floodplain) vegetation, although trees were expected to be less abundant in the lower-riparian zone. I hypothesized that cover diversity would be lower at deforested sites. Deforested sites were expected to have wider channels, lower diversity and density of aquatic habitats, and lower proportions of edge, deep, and fast habitats. I expected a higher diversity and lower density of aquatic habitats and increased abundance of deep habitats downstream, whereas fast habitats were hypothesized to be more abundant at sites of steeper channel gradient.

## 2.0 Materials and Methods

### 2.1 Study Watersheds

The two agriculturally impacted stream valleys studied were in the Thompson River (TR) drainage of the Fraser River basin, i.e., in the southern-interior (Kamloops) region of BC (Hume 1993), where soils and waters are calcareous (Clark and Bonham 1982). The streams included the mainstems of the Salmon River (near Salmon Arm) and Nicola River (near Logan Lake and Merritt), which flow into Shuswap Lake (South TR drainage) and mainstem TR, respectively. Whereas headwater vegetation is dominated by mesic conifers, valley vegetation in this region is 'dry forest': ponderosa pine (*Pinus ponderosa*) characterizes the uplands, cottonwoods (*Populus* spp.) and bunchgrass (erect-stemmed, perennial Poaceae) inhabit the floodplain zone, and sagebrush (*Artemisia* spp.) has invaded cattle-impacted riparian areas (Cowan and Guiguet 1965; van Ryswyk et al. 1992; Anon. 1993). Deforestation of riparian habitats, bank erosion, eutrophication, and lowered flows in both rivers are extensive due to agricultural and urban impacts (Wiens 1980; Woodward and Healey 1993; John and Geier 1994) and ice-flooding effects (McMullen 1985; Doyle 1988). Floodplain assessment and mitigation are being undertaken to protect fish and wildlife habitats in both the Salmon (Ross 1992; NIB 1993; Hamm 1995) and Nicola watersheds (McMullen 1983; Sahlstrom 1992; EC et al. 1993; Cantin 1995).

The study streams were moderate in elevation, size, and gradient, with meandering channels and some braiding (channel division). All study habitats were in the middle and lower river zones between 350 and 600 m in elevation (Figures 1 and 2). Based on analyses of topographical (1:50,000) maps showing intermittent and permanent streams, Salmon River (SR) reaches 1A and 1B were 4th order in size, whereas the downstream reaches (below Bolean Creek) were 5th order. Slopes of 4.0 m/km characterized both the middle and lower zones. On the Nicola River (NR), the upper site (#1) was 5th order and had a slope of 3.2 km/m, whereas the lower, canyon site (#2) was 6th order (below Spius Creek) and had a slope of 6.4 m/km (cf. Doyle 1988). Because the beds of both streams consisted especially of cobble, gravel, sand, and mud substrata

(R.L. Vadas, Jr., unpubl. data), SR and NR had characteristics of Fonda's (1974) 'floodway' and 'pastoral' stream types.

## **2.2 Riparian-Habitat Assessment**

Sampling was conducted during the fall of 1994 for SR (August 28 to September 12) and NR (September 24 to October 8). In both rivers, three reaches (i.e., A, B, and C) of differing upper-riparian intactness (see below) were studied at two sites: middle (#1) and lower (#2) mainstems (Figures 1 and 2). There were three vegetation types categorized in each river valley. 'Forested' (SR only) and 'semi-forested' reaches had intact deciduous-buffer strips on both vs. one side(s) of the river, respectively. 'Shrubby' reaches (NR only) had sagebrush-dominated buffer strips on both sides of the river, whereas 'grassy' reaches were dominated by low and/or high herbs. These vegetation categories are similar to those used by Batchelor et al. (1982) and Redpath (1990).

Floral-cover sampling in the lower-riparian zone was done to assess bank shade and stability (Moore and Archdekin 1980; Anon. 1995b,c; OES 1995; Johnston and Slaney 1996) in both rivers. Each reach was 300 to 400 m long, allowing at least three river bends (meanders) and several habitat types to be sampled (cf. de Leeuw 1982; Vadas 1991; Munro and Taccogna 1994). Spot counts of the dominant cover type on both shorelines were done every 25 paces, from the head marker downstream to the tail marker. The dominant cover type was the tallest floral-growth form and/or the most-abundant substratum type. The five categories in decreasing order of desirability (cf. Rounick and Winterbourn 1982; Platts et al. 1983; Myers and Swanson 1992) were 'tree', 'woody-shrub', 'tall-herb', 'bare-coarse', and 'bare-fine', the height criterion separating trees and shrubs being 10 m (Terrell et al. 1982; Steen and Roberts 1988; Kistriz and Porter 1993; Anon. 1995a). Woody plants were only counted if they overhung the stream, had trunks within 5 m of the bank, and/or had roots extending to the bank. The bare categories at most contained sparse, short (< 50 cm tall) vegetation. Bare-coarse spots, in increasing order of abundance, were large-woody debris, cobble-boulder riprap, and gravel-cobble bars. In contrast, fine-coarse spots were bars that contained some combination of clay, silt, sand, and/or gravel (sensu Platts et al. 1983).

### 2.3 Aquatic-Habitat Assessment

Assessment included measurements of wetted-stream width and habitat composition. Wetted widths were measured at 7 to 19 transects that were systematically placed to sample different aquatic-habitat types, more measurements being made when the channel was braided. Because of the limited number of lengthy stretches with homogeneous vegetation, some reaches within sites were separated by townships, confluences of major tributaries, and/or large distances ( $\geq 1$  km) (Figures 1 and 2); width comparisons across the deforestation gradient were thus best for reaches #1A and B in the middle SR, #2B and C in the lower SR, and #2A to C in the lower NR. Reach lengths were also measured so that mesohabitat density (the number of habitat units per stream-km) could be calculated.

Habitat classification was similar to that of Vadas (1992, 1994), being based on lateral (macrohabitat) and depth and turbulence (mesohabitat) characteristics. There were five lateral-macrohabitat and seven mesohabitat types, which were visually assessed during stream walks and measured with a tape measure. The macrohabitat categories included three major types (main-channel, side-channel, and backwater) and two transitional ones (backwater/side-channel and main/side-channel). 'Side channels' were under 50% of the total stream width (and flow), and were located on the back sides of well-vegetated (with riparian trees and/or shrubs) islands. 'Backwaters' were partially segregated from the 'main channel' by (1) small, unstable (sand-gravel) isles that lacked perennial vegetation (cf. Chamberlin 1980), (2) instream cover (organic debris), and/or (3) rapid changes in current velocity. In contrast to side channels, backwaters were stagnant, contained high amounts of algae, and were only connected to the main channel by a single chute. Stagnant, algal-covered habitats with connections at the up- and downstream ends were assigned 'backwater/side-channel' status. In cases where shallow, rocky shoals upstream of an island caused separation of flow between the left and right halves of the stream, the area upstream of the side channel was assigned 'main/side-channel' status. Isolated backwater ponds (*sensu* Vadas 1992) were not assessed.

The mesohabitat types included medium and shallow pools, medium and shallow runs, slow and fast riffles, and medium torrents (cf. Vadas 1992, 1994). Pools were smooth at the surface and nonturbulent for at least 75% of their areas (cf. Pearlstone 1976; Ward and Slaney 1979).

Whereas 'medium pools' were above waist-level in maximum depth, 'shallow pools' were usually at or below knee-level ( $\leq 55$  cm). 'Shallow' and 'medium runs' were differentiated based on the same depth criteria; they were turbulent in up to half of their surface areas and generally undular (wavy) at the water surface (cf. Pearlstone 1976; Ward and Slaney 1979). Riffles were shallow, with a majority of their surface broken by turbulence (cf. Ward and Slaney 1979; Courtney et al. 1997). In contrast to 'slow riffles', 'fast riffles' (rapids) contained whitewater turbulence, were generally steeper in slope, and were often formed by channel constrictions (cf. Pearlstone 1976; Johnson 1985; Courtney et al. 1997). 'Medium torrents' were similar to fast riffles but deeper ( $> 55$  cm). Because the streams were moderately narrow, particularly SR (see below), mesohabitat units were usually assigned for the full main-channel width, although non-pool habitat types (e.g., slow vs. fast riffles) were sometimes distinguished laterally.

Given that pool:riffle (P:R) ratios are often calculated to assess habitat integrity for fish and wildlife (Platts et al. 1983; Schroeder and Allen 1992; Anon. 1995c), I divided the areal abundance of the two pool habitats by that for the three riffle/torrent habitats to obtain P:R ratios; runs were ignored for this index.

## **2.4 Statistical Analysis**

Univariate comparisons of three aquatic-habitat variables were made for reaches within sites. Average wetted width and coefficient of variation of wetted width ([standard deviation / average width] X 100) were calculated for each stream reach. Density of aquatic-mesohabitat units within each reach was calculated as the total number of individual units divided by reach (thalweg) length.

Multivariate (varimax-factor) analysis and the Simpson-Levins diversity index (cf. Vadas 1991, 1992, 1994) were used to examine habitat patterns among sample reaches, multivariate analyses being used to efficiently and objectively cluster sites together with similar habitat compositions (cf. Fredette et al. 1990). Two sets of analyses were done on percent-abundance data, including overall and river analyses, to examine riparian-, macro-, and mesohabitat patterns; this yielded six total analyses. Overall analyses were done across the 12 study reaches (6 each for SR and NR), there being 12 variables (reaches) and 5 to 7 observations (habitat types). Separate factor

analyses were done on the two BC rivers for comparison's sake, given that the latter analysis showed less disparity (and statistical distortion) between the number of observations (5 to 7) and variables (6).

Because of the complexity of results for the overall factor analyses (FA), multivariate-similarity tables were formulated (cf. Vadas 1991, 1992, 1994). Variables showing highest loadings on the same factor axis were considered completely similar, and were stacked vertically in the diagram. A variable showing only a moderately high loading (within 0.10 Pearson units of the variable's highest loading) on an axis where other variables loaded highly was considered moderately similar to the latter variables; the former variable was stacked horizontally and connected by a similarity bar to the other variables. Other variables, which loaded on different axes because they were independent of each other, were not connected by similarity bars. 'Important' factor axes were those with high or moderately high loadings for at least one variable. The simple presentation of results provided by these tables facilitated assessment of habitat trends along the longitudinal and riparian-floral gradients.

A moderate-loading criterion of 0.20 Pearson units (instead of 0.10) was used for analyses to examine lateral-macrohabitat data. These factor analyses lacked loadings  $\geq 0.8$  because all variables showed substantial loadings on the first 2 to 3 axes. The latter pattern reflected statistical distortion that occurs in varimax-FA (in contrast to quartimax-FA) when all variables are similar (pers. obs.).

## **3.0 Results**

### **3.1 Riparian-Habitat Patterns**

Based on the data of Table 1, the overall FA yielded four clusters (important factor axes) of study reaches based on their floral compositions (Table 2). Reaches that loaded highest on the first two factor axes were generally less diverse in streamside cover because of dominance by fewer cover types. The first factor axis consisted of most NR reaches, regardless of floral composition in the upper-riparian zone. These reaches had extensive depositional bars of coarse substrata and relatively little shoreline vegetation. Factor #2 was for SR reaches lacking full forests, where tall

herbs dominated the banks. Factor #3 consisted of woody reaches on both rivers, where bare depositional bars and bank vegetation were common. Factor #4 was for more-forested reaches of SR, where bank vegetation was extensive. Separate factor analyses on the two rivers yielded similar results for reaches within rivers, except that treed (FO-SF) reaches in the middle SR loaded independently. Although treed reaches showed higher floral diversity than unforested (SH-GR) reaches in SR, the opposite trend was apparent in NR.

The analysis also shows that NR reaches usually clustered together, regardless of their upper-riparian intactness, because rocky depositional bars (and thus bare-coarse substrata) were dominant rather than vegetation as in SR. In contrast, treed reaches generally showed a greater abundance of shrubs and trees than grassy sites, where tall herbs were dominant. Upstream sites on the two rivers generally showed higher growth-form diversity (3.1 to 3.5) than did lower-river sites (1.7-2.5), as summarized by median-diversity values (Table 1). Despite the lesser floral correspondence between the upper- and lower-riparian zones for NR than SR, growth-form diversity was generally higher at more-forested sites for the two rivers: FO (3.1-3.9) > SF (2.0-3.9) > SH (1.6-2.7) > GR (1.3-1.7).

### **3.2 Wetted-Width Patterns**

Trends for wetted widths were more apparent along the longitudinal than deforestation gradient (Table 3). Downstream sites were generally wider, the 15-m increase apparent between the upper two NR reaches reflecting the entry of the Coldwater River just downstream of Merritt township (Figures 1 and 2). Treed reaches were not consistently wider or more diverse in width than nearby unforested reaches (i.e., comparisons of reaches #1A vs. B in the middle SR, #2B vs. C in the lower SR, and #2A vs. B-C in the lower NR).

### **3.3 Aquatic-Macrohabitat Patterns**

The overall FA yielded three important factor axes, but most sample reaches were completely similar (based on the 0.20 loading criterion) because main-channel habitat was predominant and habitat diversity was low (median = 1.1) (Table 4). The only divergent reaches were two partially wooded samples (SL-SF and NL-SH), which were partially similar to the other sites but not to

each other. These two reaches showed higher abundance of edge habitats and greater macrohabitat diversity (1.6-1.7). Separate factor analyses on the two rivers yielded similar results for reaches within rivers, except that the above two divergent reaches were independent of the others (i.e., they loaded alone on factor #2, whereas the others loaded highly on factor #1).

Downstream reaches in SR, but not NR, showed higher macrohabitat diversity than did middle-river sites. Although grassy reaches were less diverse (1.0) than treed habitats in SR (1.1-1.7), semi-forested reaches in NR (1.0-1.1) actually showed lower macrohabitat diversity than did unforested reaches (1.0-1.6). Hence, edge habitats were somewhat more common downstream and at treed sites only in SR.

### **3.4 Aquatic-Mesohabitat Patterns**

Longitudinal and deforestation trends in mesohabitat density and P:R ratio were ambiguous (Table 5). Although mesohabitat densities were higher for treed (32-89 per km) than grassy reaches in SR (17-19 per km), partially wooded sites in NR (21-31 per km) showed lower densities than for grassy reaches (38-53 per km). P:R ratios unexpectedly decreased downstream in both rivers, being especially low in the lower NR, reflecting the high abundance of slow and fast riffles in this naturally steeper zone. Given these results and the fact that P:R ratios were closest to unity for semi-forested reaches in SR and unforested habitats in NR, the P:R ratio was of limited use for assessing habitat damage by humans.

Based on the data of Table 5, the overall FA yielded four clusters (important factor axes) of reaches based on their mesohabitat compositions (Table 6). Reaches that loaded highest on the first two factor axes were of various riparian intactness; they were dominated by shallow-pool and/or slow-riffle habitats and often showed subdominance by medium pools. Factor #3 consisted of treed sites, which were dominated by shallow runs. Factor #4 consisted of a shrubby site, which was dominated by medium pools. Separate factor analyses on the two rivers yielded similar results for reaches within rivers, except that the two semi-forested reaches in SR loaded highly on the same axis. Only NR showed downstream increases in mesohabitat diversity. Hence, deep habitats (medium pools) were more common upstream, fast habitats (fast riffles) were more



common at the steepest site (lower NR), and upstream sites were less diverse because only pool habitats were abundant.

The FA results and other indices do not provide strong support for consistent differences in mesohabitat composition among riparian-floral types; treed sites did not have consistently higher mesohabitat diversity and abundance of deep or fast habitats than did unforested sites. Indeed, mesohabitat diversity was higher for semi-forested (3.4-4.0) than forested (1.4-2.7) and grassy reaches (1.7-2.3) in SR, whereas grassy habitats showed higher values (3.1-3.8) than partially wooded reaches (1.6-3.4) in NR.

## **4.0 Discussion**

### **4.1 Habitat Patterns**

The results suggest that assessment of lower-riparian vegetation does not accurately reflect floral conditions in the upper-riparian zone. Whereas more-forested reaches on SR did show taller bank vegetation (especially tall herbs and woody shrubs), most NR reaches were bare along shore. Flood and ice damage, common in the NR watershed (McMullen 1985; Doyle 1988; Doyle et al. c.1993), may have limited floral colonization of the lower-riparian zone (Yanosky 1982; McBride and Strahan 1984). Alternatively, given that bank erosion can reduce tree distance from the water (Kilpatrick and Barnes 1964), more erosion may have occurred along SR, although these trees should reduce present erosion potential. Given that substratum sizes of depositional bars and adjacent riffles tend to be similar in stable streams (Kappesser 1993), a third possibility is that the relatively coarse channel substrata of the five downstream Nicola reaches (R.L. Vadas, Jr., unpubl. data) have promoted development of coarse depositional bars that limit vegetation development. Clearly, sampling of both the upper- and lower-riparian zones is needed to adequately assess riparian intactness; Moore and Archdekin (1980) and OES (1995) recommended that floral transects extend 35 to 40 m from water's edge for BC streams.

The aquatic results show that hypothesized trends for width, habitat composition, diversity, and mesohabitat density were often not realized. Macrohabitat diversity in SR and NR was not consistently higher downstream and in more-forested habitats. Literature data support the

ambiguous (partially unexpected) width and lateral-macrohabitat trends found in SR and NR. Although riparian grazing by livestock can cause streams to become wider and shallower, forested streams do not necessarily have lower width:depth ratios (Gresswell et al. 1989). Channel widening can occur in wooded (Trotter 1990) and deforested reaches (Lyons and Beschta 1983) because of the influx of woody debris (Trotter 1990) vs. fine sediments (Schumm and Khan 1972; Osterkamp 1978; Lyons and Beschta 1983; Jackson and Beschta 1984), respectively. Such organic and inorganic inputs both cause increases in channel complexity (braiding) and thus percent abundance of side-channel and other edge habitats, such that cover and sediment variables (R.L. Vadas, Jr., unpubl. data) require examination to better understand the lack of trend.

Hypothesized mesohabitat patterns (see the introduction) were often not observed. As expected, fast riffles were more common in the steepest stream reaches (lower NR). Contrary to expectations, pool habitat was more common upstream, habitat diversity was not always more common downstream, and treed reaches did not show higher habitat diversity and abundance of deep and fast habitats. Other researchers in the Pacific Northwest have also obtained unexpected results. Platts (1974, 1979) found that pools were less abundant at wider or steeper sites in Idaho, such that longitudinal changes in pool:riffle (P:R) ratios were ambiguous. Hogan (1986) and Myers and Swanson (1991) only found decreases in habitat quality (e.g., P:R ratio) for western streams subjected to severe riparian-habitat impacts. Admittedly, a better test of longitudinal hypotheses would require sampling longer stretches of river, and not just mid-sized reaches as I have done. The SR-NR results suggest that differences in aquatic biota along the deforestation gradient (Vadas 1997a,b) are more likely to result from factors not addressed here, e.g., sedimentation, eutrophication, heating, and changes in food and cover availability (R.L. Vadas, Jr., unpubl. data). For example, study reaches in the lower SR and middle NR showed higher abundance of fine substrata (sand and/or mud) than the other two sites (Gregory 1989; R.L. Vadas, Jr., unpubl. data), likely reflecting the proximity of the lower SR to Shuswap Lake and middle NR to Nicola Lake (Figures 1 and 2). Holistic physicochemical analyses are needed to establish what factors consistently change with deforestation in the southern interior of BC.

## 4.2 Management Implications

Riparian habitats and biota in North America are more heavily impacted by logging, agriculture, urbanization, and other human activities than are upland-forest ecosystems (Lynch et al. 1977; Kauffman and Krueger 1984; Swift 1984; Knopf et al. 1988). Such environmental effects have stimulated interdisciplinary-research efforts to assess deforestation impacts in the northwestern U.S. (Salo and Cundy 1987; Janik and Williams 1993) and BC (Narver and Chamberlin 1976; Poulin 1984; Poulin and Morris 1987), including assessment of habitat and biota in the upper- and lower-riparian and aquatic zones. Indeed, riparian zones with wider bands of tall grasses or woody vegetation provide better habitat for riparian vertebrates and water quality for lotic animals than do logged streamside zones (Vadas 1997a,b; Vadas and Newman 1997). Therefore, planting of native, riparian plants of various growth forms (herbs, shrubs, and deciduous and coniferous trees) along impacted streams should improve habitat for fish and wildlife in BC (Carr 1985; McLennan 1993; Hickey and Trask 1994; Anon. 1995f), the northwestern U.S. (Gresswell et al. 1989; Berg 1995), and elsewhere in the temperate zone (Lewis and Williams 1984; Dunn 1995). Indeed, preservation and restoration of riparian vegetation is a better management technique than merely rip-rapping banks and/or using instream structures (Platts and Rinne 1985; Donat 1995), such that riparian management can speed up healing processes in impacted streams when used in combination with instream-habitat improvements (USFS 1985; Elmore and Beschta 1987; Ferguson 1991). As noted by various BC researchers, provincial watershed management will improve with standardization of riparian and aquatic habitat-classification systems and training programs in habitat restoration for biologists and loggers (Mather et al. 1985; Anon. 1995d,e; OES 1995; Ward and Plackett 1995).

Although habitat restoration and formation of interdisciplinary partnerships are important goals in the Fraser River basin of BC (FRAP 1995), there has been inadequate focus on riparian research and management (Morgan and Lashmar 1993; Pearce 1993; Rautio and Bunnell 1994). Because riparian zones in the Pacific Northwest are heavily used by loggers, farmers, and recreationists, multiple-use planning will be needed to protect riparian and stream ecosystems (Thomas et al. 1979a,b). Such management will require the input of economists as well as biologists and managers; socioeconomic assessments (Brinson et al. 1981; Theurer et al. 1985; Salo and Cundy

1987; Braden et al. 1989), and economic incentives to developers (Sharpe 1975; Golde 1986; Allen 1993; Rolfe 1993) are both needed for successful watershed management. My research on the SR and NR watersheds has indeed been ecosystem-oriented (Michel 1997; Vadas 1997a) and will hopefully provide local citizens with the habitat and biological tools that they need to effectively manage and restore these stream systems.

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## **TABLES**

**Table 1. Percent abundance of cover types in the lower-riparian zone for reaches of two BC study streams.** Habitat types were forested (FO), semi-forested (SF), shrubby (SH), or grassy (GR). TOTAL # = total number of data points and DI = Simpson-Levins diversity index.

Cover Type	Salmon River						Nicola River					
	Middle			Lower			Middle			Lower		
	GR	FO	SF	GR	FO	SF	GR	SH	SF	GR	SH	SF
Tree	3	27.5	20	9	6	11	0	2.5	15	4	2	10
Woody-shrub	20	30	32	2	32	8	8	50	13	0	5	10
Tall-herb	73	40	26	77	32	68	8	27.5	15	6	16	17
Bare-coarse	0	0	22	0	18	13	77	0	52	88	77	62
Bare-fine	3	2.5	0	11	12	0	6	20	4	2	0	2
TOTAL #	30	40	50	44	50	38	48	40	46	52	44	52
Site DI	1.7	3.1	3.9	1.6	3.9	2.0	1.6	2.7	3.0	1.3	1.6	2.3
Median DI	3.1			2.0			2.7			1.6		



**Table 2. Multivariate-similarity table to examine riparian-cover differences among the 12 study reaches, namely the middle and lower Salmon and Nicola rivers (SM, SL, NM, and NL, respectively). Dominant (DM) and subdominant (S) floral-substratum types and median species diversity are shown for each factor axis. See Table 1 for data and other abbreviations.**

Cover Type	NM-SF NM-GR NL-SF NL-GR NL-SH	SM-GR SL-SF SL-GR	SL-FO NM-SH	SM-FO SM-SF
Tree	-	-	-	DM
Woody-shrub	-	-	DM	DM
Tall-herb	-	DM	S	DM
Bare-coarse	DM	-	S	S
Bare-fine	-	-	S	-
Median diversity	1.6	1.7	3.3	3.5

**Table 3. Wetted-width statistics for the 12 study reaches.** CV = coefficient of variation. See Table 1 for abbreviations.

Parameter	Salmon River						Nicola River					
	Middle			Lower			Middle			Lower		
	GR	FO	SF	GR	FO	SF	GR	SH	SF	GR	SH	SF
Average width (m)	9	10	10	14	12	12	20	12	27	25	36.5	27
CV for width (%)	6	17	28	20	16	38	28	12	19	28	34	22
# of transects (N)	9	11	11	7	11	19	9	8	8	8	8	7

**Table 4. Percent abundance of aquatic-macrohabitat types for the 12 study reaches.**  
 MC/SC and BW/SC were transitional habitats. See Table 1 for format.

Habitat Type	Salmon River						Nicola River					
	Middle			Lower			Middle			Lower		
	GR	FO	SF	GR	FO	SF	GR	SH	SF	GR	SH	SF
Main-channel (MC)	100	95	94	100	84	73	86	100	94	97	75	100
MC/SC	0	0	0	0	0	0	0	0	0	0	22	0
Backwater (BW)	0	0	6	0	8	2	2	0	6	3	0	0
BW/SC	0	0	0	0	3	0	0	0	0	0	0	0
Side channel (SC)	0	5	0	0	5	25	12	0	0	0	3	0
Site DI	1.0	1.1	1.1	1.0	1.4	1.7	1.3	1.0	1.1	1.1	1.6	1.0
Median DI	1.1			1.4			1.1			1.1		

**Table 5. Percent abundance of aquatic-mesohabitat types for the 12 study reaches. TOTAL # = total number of habitat units sampled, DENSITY = number of habitat units per stream-km, P:R ratio is the ratio of pool vs. riffle/torrent abundance, and \* = infinity (undefined ratio). See Table 1 for format.**

Habitat Type	Salmon River						Nicola River					
	Middle			Lower			Middle			Lower		
	GR	FO	SF	GR	FO	SF	GR	SH	SF	GR	SH	SF
Medium pool	30	18	20	0	2	13	10	68	2	7	0	0
Shallow pool	70	56	31	59	12	38	41	27	76	16	25	12
Medium run	0	0	0	0	0	1	0	0	0	0	0	0
Shallow run	0	15	9	24	85	11	24	0	2	8	14	40
Slow riffle	0	9	31	17	1	34	13	4	18	50	43	23
Medium torrent	0	0	0	0	0	0	1	0	+	0	0	0
Fast riffle	0	3	9	0	0	2	11	0	2	20	19	25
TOTAL #	6	20	19	6	12	32	20	9	12	16	8	7
DENSITY	19	57	58	17	32	89	53	25	31	38	28	21
Site P:R ratio	*	6.2	1.3	3.5	14	1.4	2.0	24	3.8	0.33	0.40	0.25
Median P:R ratio		6.2			3.5			3.8			0.33	
Site DI	1.7	2.7	4.0	2.3	1.4	3.4	3.8	1.8	1.6	3.1	3.4	3.4

**Table 6. Multivariate-similarity table to examine aquatic-mesohabitat differences among the 12 study reaches. See Table 2 for format.**

Habitat	SM-GR SM-FO SL-GR NM-SF NM-GR	SL-SF	SM-SF NL-GR NL-SH	SL-FO NL-SF	NM-SH
Medium pool	S	S	-	-	DM
Shallow pool	DM	DM	S	S	S
Shallow run	S	S	-	DM	-
Slow riffle	S	DM	DM	S	-
Fast riffle	-	-	-	S	-
Median DI	2.3	3.4	3.4	2.4	1.8

## **FIGURES**

**Figure 1. Map of the Salmon River watershed.** The study sites (#1 and 2) on the river include three reaches (A, B, and C) of differing riparian intactness (as indicated). The Salmon watershed is northeast of the Nicola watershed (figure 2), originating near 50<sup>0</sup>10' north and 119<sup>0</sup>45' west coordinates. Asterisks (\*) indicate reaches with obvious cattle damage.

**Figure 2. Map of the Nicola River watershed.** The study sites (#1 and 2) on the river include three reaches (A, B, and C) of differing riparian intactness (as indicated). The Nicola River originates near 50°10' north and 119°45' west coordinates. Asterisks (\*) indicate reaches with obvious cattle damage.