PHYSICAL AND ECOLOGICAL ORGANIZATION IN A LARGE, GRAVEL-BED RIVER AND RESPONSE TO DISTURBANCE

by

LAURA L. REMPEL

B.Sc., The University of Calgary, 1994
M.Sc., The University of British Columbia, 1997

A THESIS SUBMITTED IN PARTIAL FULFILMENT OF
THE REQUIREMENTS FOR THE DEGREE OF
DOCTOR OF PHILOSOPHY
in
THE FACULTY OF GRADUATE STUDIES
(Department of Geography)

We accept this thesis as conforming
to the required standard

Michael Church

Olav Slaymaker

THE UNIVERSITY OF BRITISH COLUMBIA

July 2004

Abstract

Along the 80-km gravel reach between Hope and Mission, Fraser River has a wandering morphology that includes secondary channels, gravel bars, and islands that together support a diverse and productive ecosystem. The wandering morphology is produced, in part, by annual sediment deposition within the reach. Sediment deposition, however, is perceived as a threat to flood security and gravel mining is proposed as a profitable solution to flood risk. This thesis presents a hierarchical habitat classification for the gravel reach, which provided a spatial framework to, first, examine habitat associations of benthic invertebrates and fish and, second, evaluate the physical and ecological responses to habitat disturbance by gravel mining.

At the highest level of the classification, the river is divided into 5 sub-reaches (10^4 m scale) that vary in morphological expression and sediment gradational tendency. The intermediate level specifies gravel bar units (10^3 m scale) each consisting of a riffle, gravel bar, and adjacent pool. Nested within gravel bars are physically distinct habitat units, which represent the finest level of the classification (10^1-10^2 m scale). Results demonstrated that the assemblages of invertebrates and fish associated with habitats are moderately distinct and differentiated along a hydraulic gradient corresponding to velocity. However, the congruence between habitat structure and the structure of aquatic communities was weakened by large spatial and temporal variability in the distribution and abundances of many species. This variability was predicted to afford the invertebrate and fish community resilience to physical disturbance.

The second component of this study examined the effect of disturbance by gravel mining from an exposed bar in 2000. Physical changes immediately after mining were substantial as the pre-existing cobble surface was replaced by loose gravel and sand. Two subsequent freshets transformed the site into a topographically complex area with similar substrate texture as before mining, although sediment replenishment to the site was negligible. Habitat availability at flows $<3000\ \mathrm{m^3\ s^{-1}}$ increased and habitat quality was comparable with reference sites. A third above-average freshet replenished 31% of the removal volume and restored average bar surface elevation to within 9 cm of the pre-scalped surface. Physical changes elicited a significant reduction in invertebrate density immediately after mining, however, the impact lasted less than one freshet cycle. No change in fish density as a result of mining was found, although statistical power to detect an impact was low and fish sampling was carried out at flows $<5700\ \mathrm{m^3\ s^{-1}}$. These results support the expectation that the invertebrate and fish community in the gravel reach has resilience to disturbance from a single gravel removal provided that site recovery by way of sediment transport and replenishment occurs.

Table of Contents

List of Tables

List of Figures

Acknowledgements

This project was made possible through the efforts and contributions of many individuals and organizations. First and foremost I would like to thank my supervisor, Dr. Michael Church, for giving me freedom and independence to carry out this research, and for providing stimulating discussion, critical reviews, and encouragement in the culminating stages. His support spanned many years, continents, and time zones, and this work would have been severely compromised without his geomorphological perspective and process-based approach to ecological research. I also extend grateful appreciation to members of my committee who provided very thoughtful and prompt reviews of thesis drafts and have been entirely supportive of this study: Dr. Michael Healey, Dr. John Richardson, Dr. Marvin Rosenau, and Dr. Olav Slaymaker. Dr. Richardson is further thanked for providing valuable statistical advice and Dr. Tom Johnston is acknowledged for his thoughtful review of chapters 7 and 8. This research also benefited from discussions with Dr. Stephen Rice (Loughborough University) and Mr. Chris Perrin (Limnotek Research & Development).

Financial support for this research was provided by the British Columbia Habitat Conservation Trust Fund, the National Science and Engineering Research Council (NSERC), and the Department of Fisheries and Oceans (DFO). Personal support was provided by NSERC and through UBC scholarships (Li Tze Fong Fellowship, University Graduate Fellowship).

Sincere appreciation is extended to the following people who helped me with field work and whose energy and enthusiasm, especially on the rainy days, were greatly appreciated: Dave Awram, Dave Campbell, Kathy Christie, Erica Ellis, Fly, Shawn Johnston, Alexis Heaton, Nick Manklow, Jacqui Morrisson, Dave Oldmeadow, Mike Papageorge, Teri Peterson, Karen Rempel, Hal Rempel, Steve Rice, Karyn Rilkoff, Catherine Shaw, Ian Skipper, Blair Tarling, Hamish Weatherly, Arelia Werner, Andre Zimmerman, and Zip. Fly is thanked for providing entertainment in the field and for assuring quality control during invertebrate and sediment sampling. Erica Ellis and Hamish Weatherly are especially thanked for their tireless contribution in the field and for assisting with countless duties throughout this project. Darren Ham is gratefully acknowledged for his technical assistance and GIS-wizardry.

Various agencies and institutions provided logistical support and equipment that were essential to this research. The former British Columbia Ministry of Fisheries (now Ministry of Water, Land and Air Protection, WLAP) provided the jet boat and free accommodation at the Fraser Valley Trout Hatchery in Abbotsford. Mr. Bob Land (WLAP) is acknowledged for providing logistical

advice and technical support in the field. Field sampling equipment was provided by the former BC Ministry of Environment, Lands and Parks (WLAP) and the Department of Geography, University of British Columbia. The DFO Laboratory at Cultus Lake continues to provide storage space for benthic invertebrate samples and fish stomach contents, and provided per-gratis laboratory equipment and space necessary for sample processing. All invertebrate and stomach samples were processed by Ms. Shirley Fuchs.

Thanks are extended to the staff in the Department of Geography (UBC). In particular, Vincent Kujala who went beyond the call of duty on many occasions with computing assistance, Sandy Lapsky who was tremendously helpful with administrative and financial duties, and the staff in the main office (Elaine, Jeanne, Junnie, and Mary) who always were helpful over the years.

Leaving the best for last, I want to sincerely thank special friends who have been a great source of support and provided much needed distractions at key times. These include Wendy & Bernie, Erica & Bob, Matthias & Monica, Matt, Pam & Clint, Kathleen, and Dave & Shannon. To my family, Scott & Mo, Christy, Telle, Marett & Graham, and most especially my parents, Karen & Hal-- thank you for your unstinting support and encouragement in all my endeavors. And to the love of my life, Hamish Weatherly, thank you for being there for me, in every capacity, always.

Dedication

This thesis is dedicated to Dr. John W. Adams and Dr. George C. Weatherly, who inspired myself and many others in their research and scholarship, but were unable to see this project come to fruition.

Chapter 1. Introduction

1.1 Introduction

In his keynote address to the International Large River Symposium, Hynes (1989) identified large rivers as the most understudied yet most threatened of all aquatic systems. He attributed the lack of study to a variety of factors, not the least of which are the logistical challenges and costs of conducting research in deep and fast-flowing water. The message delivered by Hynes (1989) evidently was heard because, since 1989, large rivers have been the focus of numerous studies to advance the understanding of their physical, chemical, and biological dynamics (http://isi10.isiknowledge.com/portal.cgi). Research efforts have been fuelled by an increasing awareness that management practices and land-use activities over the past century have placed the economic, social, and ecological values of large rivers at significant risk (Sparks 1995, Gore and Shields 1995).

The degradation of large rivers has occurred on a global scale, and impacts to rivers in developed regions of the northern hemisphere are particularly well documented (Dynesius and Nilsson 1994). Northern temperate rivers are the focus in this thesis, and common physical impacts include flow regulation and diversion, aggregate extraction, pollution, channelization for floodplain development, as well as basin-scale changes in land use. Such impacts may affect the flow regime or sediment load of a system, which in turn affects channel morphology and the physical character and overall availability and distribution of habitats (Stevens *et al.* 1995). Changes to habitat structure (usually significant simplification) and subsequent changes in the biological community may impact upon the health of the ecosystem and the economic and cultural values of a region (Bravard *et al.* 1986, Healey 1994, Beechie *et al.* 1994, Dynesius and Nilsson 1994).

For many large rivers, the period of impact extends beyond accurate records of historical channel morphology and biological productivity prior to impact (Hynes 1989); hence, present estimates as to the severity of river degradation are speculative at best (Ward and Stanford 1989). Few studies have been carried out on relatively pristine large rivers, despite the volume of recent research noted above, and an understanding of the physical dynamics and ecological function for such rivers is lacking. Meanwhile, public awareness of both real and perceived threats to river ecosystems is increasing and, in many regions, has led to demands for ecologically sustainable river management and the restoration of large rivers where habitat has been degraded (Dynesius and Nilsson 1994, Gregory *et al.* 1998, Ricciardi and Rasmussen 1999, Graf 2001, Abell 2002). Without an understanding of large river structure and function in the absence of human modifications, river restoration may proceed blindly attempting to achieve an uncertain target state and without re-establishing the physical and ecological processes necessary to sustain this state (Bayley 1991, Kondolf 1995, Kondolf 1998a, Trainor and Church 2003).

Large rivers that have so far escaped major modifications to their hydrological regime, and channel and floodplain, provide an exceptional opportunity for study. The urgency is more acute in populated areas of temperate biomes as compared to northern taiga and arctic regions, because a smaller proportion of large rivers remains relatively unmodified (Dynesius and Nilsson 1994). Information derived from such studies not only will improve the overall understanding of large rivers, but also can be used to direct best management practices that minimize impacts and protect habitat, and may be extrapolated to rivers of similar morphology to guide management decisions and restoration initiatives. Several recent studies have been conducted in medium-sized and relatively unmodified warm-water rivers of the southern United States (e.g., Benke 2001) and cold-water rivers of Europe (e.g., van der Nat *et al.* 2003) for this purpose; however, the habitat structure and ecology of large, cold-water rivers remain unstudied.

Fraser River, British Columbia, is one such large river where the mainstem channel has remained unregulated over its entire course and where channel confinement by dyking has been restricted mainly to the lower-most reach (Northcote and Larkin 1989). As is the case for many northern temperate rivers, physiographic confinement by mountains has limited extensive floodplain development in comparison to lowland, tropical rivers. However, the pronounced seasonal freshet and sediment supply have developed an expansive channel zone where Fraser River leaves the Coast Mountains downstream of the town of Hope. The reach of river between Hope and Mission, referred to as the *gravel reach*, includes secondary channels, backwaters, mid-channel bars, and islands that together support a diverse and productive ecosystem. The faunal diversity is testament to the high quality of physical habitat available in the gravel reach.

In addition to its exceptional ecological value, the Fraser Basin is home to 65% of BC residents, roughly 2.5 million people. At least 87% of these people are concentrated in the lower Fraser River downstream of Hope (The 2001 Census of Canada, www.bcstats.gov.bc.ca). Associated with the high density of people occupying the lower river basin are a variety of land use pressures that threaten the integrity and function of the riverine ecosystem. One such pressure stems from a concern that the natural processes of bank erosion and seasonal flooding pose an increasing and unacceptable risk to infrastructure and investment. Riprap has been placed along more than 63 km of bank line between Hope and Mission in an effort to control erosion (Church *et al.* 2001, Church and Ham 2004), and concerns of overbank flooding have led to dyke construction and the isolation of more than 100 km of side channels (Rosenau and Angelo 2000, Ellis *et al.* 2004). However, with continuing gravel deposition within the reach, there is concern that rising streambed and water levels will eventually compromise the security provided by the dykes. Gravel mining from within the main channel is now being investigated as an effective strategy to reduce flood risk and, in certain circumstances, counter erosion of the river banks; however, the ecological effects of this strategy remain uncertain.

This thesis presents results from a field-based study conducted in the gravel reach of Fraser River, British Columbia (**Figure 1-1**). Part One of the thesis characterizes the physical and ecological organization of the gravel reach in the framework of a hierarchical habitat classification, and evaluates the congruence between habitat structure and the distribution and abundances of invertebrates and fish. Part Two examines the short-term response of the physical habitat and local communities of invertebrates and fish to physical disturbance, with disturbance exemplified by gravel mining. The response to disturbance is assessed in the context of the strength of the association demonstrated between the physical habitat and local communities, which is predicted to influence the resilience of biological communities to disturbance and habitat change. This study advances the understanding of how physical habitat is organized in a large river, the degree to which this physical organization influences the structure of biological communities, and identifies factors influencing the sensitivity of large river ecosystems to disturbance. Results also contribute towards the understanding of a productive and diverse ecosystem that is under increasing threat from management impacts.

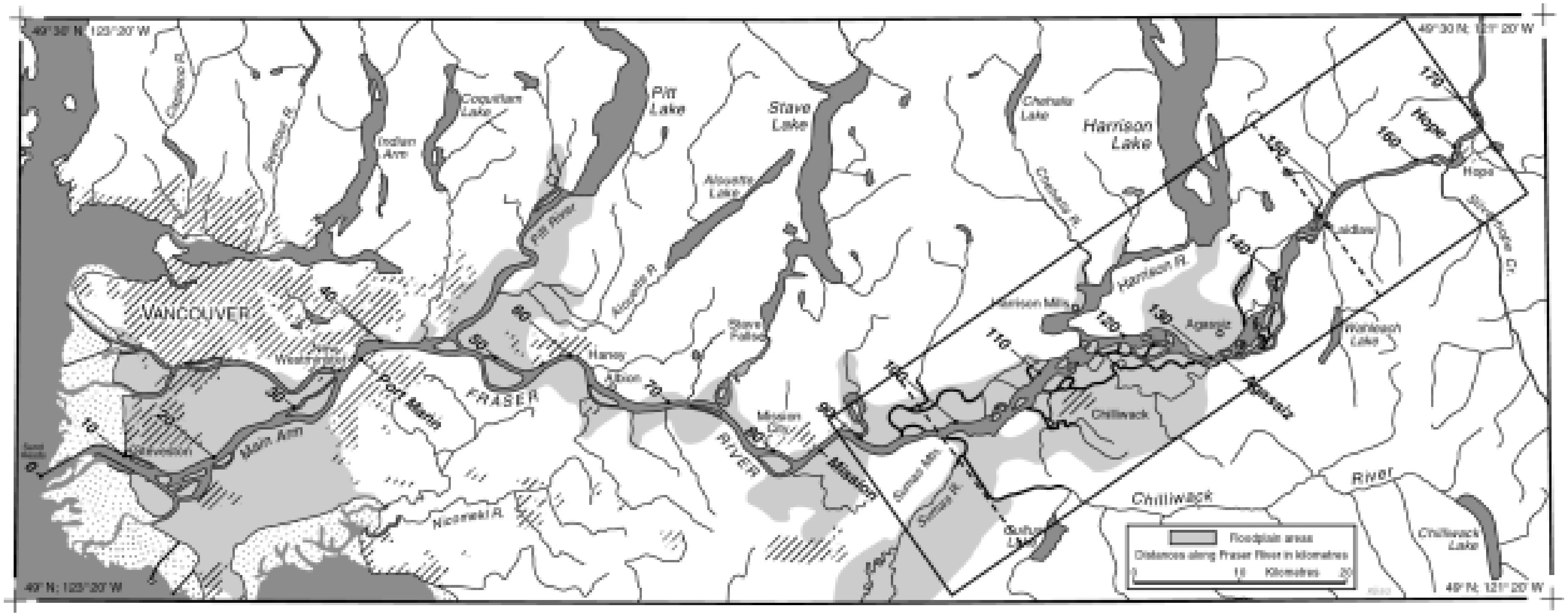


Figure 1-1. Location map of lower Fraser River downstream of Hope. The rectangle outlines the gravel reach and the dashed lines mark the portion of the gravel reach within which data were collected for this study.

1.2 Large Rivers

Large rivers can be defined on the basis of size-related characteristics such as channel width and depth, mean annual discharge, or drainage area. For example, Church (1992) described large rivers as those typically exceeding 20 to 30 m in channel width and with a bankfull discharge exceeding 20-50 $m^3 s^{-1}$. Dynesius and Nilsson (1994) defined a large river as one that has, anywhere in its catchment, a river channel section with a virgin mean annual discharge of at least 350 $m^3 s^{-1}$. No single definition appears superior, suggesting that the most appropriate definition may be determined by the hydrologic context in which it is applied. Stalnaker *et al*. (1989) suggested that large rivers also may be distinguished based on the relative importance of specialized or isolated habitat types associated with the river margin. More habitat zonation is predicted for large rivers, with “hydraulically efficient zones” in deep and fast-flowing habitats favouring few species, and “biologically rich zones” associated with edge habitats and islands favouring many others. For comparison, biologically rich zones are expected to overlap each other in small streams, with animals using a large proportion of the wetted channel area (Stalnaker *et al.* 1989).

Habitat zonation in large rivers occurs in multiple dimensions: longitudinally along the river, laterally across the channel, and vertically in deep water between the surface and river bottom (Kellerhals and Church 1989, Ward 1989). Vertical habitat zonation refers to the partitioning of habitat along a vertical velocity gradient and is a topic that has received little study, in part because relatively sophisticated sampling equipment is required (e.g., Gard and Ballard 2003). In the longitudinal dimension, channel gradient, physiographic confinement, and tributary inputs of water and sediment impose controls on channel morphology and floodplain development, which directly influence the attributes of local-scale habitat features (Davies *et al.* 2000).

In the lateral dimension, channel cross-sectional geometry influences velocity-depth distributions and the proportional area of shallow, near-shore habitat where biological productivity is typically highest (Kellerhals and Church 1989, Thorp 1992). The cross-sectional geometry of gravel-bed rivers is characterized by the thalweg with deepest water and the shallow channel margins surrounding bars and islands. Whereas high velocity in the thalweg maintains a relatively sterile substrate and is energetically costly for most fish, near-shore zones are often hydraulically favourable for fish and with increased light penetration to support a productive food base of primary (e.g., algae) and secondary (e.g., invertebrates) producers (Thorp 1992, Johnson and Jennings 1998). Also in the lateral dimension, floodplain habitats and secondary channels of multi-thread rivers (braided, wandering, and anastomosing) represent perhaps the most productive of large river habitats, in part because of the longer retention time of nutrients to facilitate primary production and the reduced energetic costs to animals (Smock *et al.* 1992, Ward *et al.* 1999). These habitats also are under greatest threat from flood management and land-use changes, and significant floodplain habitat has been lost from many northern temperate rivers (Bayley 1991, Beechie *et al.* 1994, Dynesius and Nilsson 1994).

1.2.1 Physical and Ecological Characteristics

The diverse and multi-dimensional array of habitat types collectively making up the active channel zone of large rivers poses a challenge for systematic study because channel size and inherent physical variability require significantly more sampling effort than is typical for small streams. As a result, the physical and ecological structure of large rivers in northern temperate regions have remained poorly described. In contrast, several tropical and warm-water rivers such as the Amazon and Mississippi have received considerable study, in part because they support economically valuable fisheries (Welcomme 1979, Fremling *et al.* 1989). The Large Rivers Working Group of the International Association of Geomorphologists, formed in 1997 as a forum for discussing the complexities of large rivers, also has focused primarily on warm-water rivers such as the Mekong and Yangtze (http://www.geomorph.org/wg/wglr.html).

Tropical rivers that have escaped major engineering alterations characteristically have a broad channel zone that includes a periodically inundated and expansive floodplain (Junk 1984). The Flood Pulse Concept (FPC) was developed as a conceptual model for the ecological functioning of these systems in recognition of the importance of long-lasting, seasonal floodplain inundation to support high levels of biological production and valuable fisheries (Junk *et al.* 1989). The flood pulse is credited with enabling the lateral exchange of nutrients between the river channel and floodplain, and supporting nutrient recycling within the floodplain. As a result, the majority of riverine biomass is derived from local floodplain production (Junk 1984).

Large rivers in northern temperate regions differ markedly from low-latitude systems. They typically have less extensive floodplain development due to physiographic confinement, although often still significant compared to smaller rivers, and the flood pulse is of shorter duration. Rivers draining mountainous and piedmont settings also have a steeper gradient and thus coarser bed material and reduced retention time for nutrients and organic matter. Seasonally elevated concentrations of suspended sediment may create turbid conditions that limit light penetration for photosynthesis while cold temperatures in winter limit autochthonous carbon production mainly to summer months. Bayley (1991) attributed the comparatively low productivity of northern temperate rivers to these factors and recognized the "flood pulse advantage" of tropical rivers that results from prolonged inundation of extensive floodplain areas.

The River Continuum Concept (RCC, Vannote *et al.* 1980) was the earliest conceptual model that made predictions for the trophic assemblage and ecosystem function of large, temperate rivers. Although overly simplistic in its portrayal of ecosystem structure (e.g., ignoring the floodplain influence), it was successful at generating testable hypotheses to stimulate large river research (Sedell *et al.* 1989, Walters *et al.* 2003). The Riverine Productivity Model (RPM, Thorp and Delong 1994) was proposed, based on criticisms of the RCC, as a conceptual model of carbon sources and ecosystem function for large rivers in temperate regions. The RPM was developed based on observations from the Ohio River and is intended to represent deep and laterally constricted rivers. It predicts that riparian inputs and local autochthonous production have been underestimated and the importance of the flood pulse overestimated as a carbon source by the RCC. Rather, the food web of northern temperate rivers is predicted to rely on carbon derived from local production and from the riparian zone because it is most easily assimilated and available for long periods. Neither the RCC nor RPM specifically addresses fish communities in large rivers.

No existing conceptual model appears appropriate for gravel-bed rivers such as the lower Fraser River that have an expansive channel zone subject to seasonal inundation. Gravel-bed rivers are common in mid- to high-latitude mountainous and piedmont regions, and the physical structure and ecological significance of the expansive channel zone remain uncertain (Welcomme 1995). Some of the uncertainty is attributed to the practical challenge of biological sampling in deep and fast-flowing water (Casselman *et al.* 1990). As well, uncertainty stems from an inadequate characterization of the fluvial processes and physical structure of gravel-bed rivers, which ultimately impose limits on the identity and diversity of species that inhabit the system, their spatial distributions, their life histories and seasonal phenologies, and the nature of species interactions (Power *et al.* 1988, Scarsbrook and Townsend 1993, Rice *et al.* 2001, Power 2001, Inoue and Nunokawa 2002, Parsons *et al.* 2003, Walters *et al.* 2003). Studies attempting to characterize the ecology of large, gravel-bed rivers, and large rivers in general, should be carried out in a spatial framework that recognizes the physical organization of the system and links fluvial processes to ecological attributes. Such studies stand to make the greatest contribution to the understanding of large rivers (Ward *et al.* 1999), and a geophysical process-based approach is increasingly favoured for habitat restoration and management (Kondolf 1995, Thomson *et al.* 2001). River classification has proven to be a useful tool to facilitate these goals.

1.2.2 Geomorphological River Classification

Geomorphological river classifications fulfill a large number of important functions: they consolidate and synthesize large amounts of biophysical information in a format that reduces variability and emphasizes patterns of significance; they identify key physical factors that limit or support particular species and populations; they provide a basis for estimating biotic responses to habitat availability; they provide an efficient means of estimating population characteristics at large spatial scales; they provide a framework on which to base experimental designs and stratify sampling effort; they ensure sampling consistency in long-term monitoring programs; they define a terminology that improves communication between scientists, managers and users; and they aid in the development of design criteria for river enhancement and restoration (Bisson *et al.* 1981, Rabeni and Jacobson 1993, Hawkins *et al.* 1993, Maddock 1999, Newson and Newson 2000, Thomson *et al.* 2001, Peterson and Rabeni 2001a). It is not surprising, therefore, that riverine habitat classifications are used extensively by researchers and fisheries managers (Bain *et al.* 1999).

The utility of river habitat classification stems from the expectation that physical habitat characteristics exert strong control on the species assemblage and distribution of organisms. This tenet is the basis of a habitat-centered view of ecology (Southwood 1977, Southwood 1988) that has gained wide acceptance by both researchers and managers of riverine ecosystems (Vannote *et al.* 1980, Scarsbrook and Townsend 1993, Newson and Newson 2000). An extension of this view is that a diverse and productive aquatic ecosystem relies upon the availability of high quality habitat (Maddock 1999). Furthermore, high quality habitat depends on intact regimes of water and sediment delivery to the channel, which produce the morphological complexity that gives rise to ecologically important habitats. Habitat classification provides a particularly useful framework for assessing habitat availability and quality, as well as identifying the geomorphological processes that produce and maintain productive habitats in river systems (Pitlick and Van Steeter 1998, Kemp *et al.* 1999, Thomson *et al.* 2001).

The overwhelming majority of habitat classification schemes have been developed for small streams. Meanwhile the need for large river research, and the utility of hierarchical habitat classification for providing a spatial framework within which to conduct research, is clear. A hierarchical classification applicable to large, gravel-bed rivers with a seasonally expansive channel zone would serve as a valuable tool in research and management. The exceptional fishery value of many coastal gravel-bed rivers further highlights the need for a classification that is based on geophysical habitat attributes and serves to elucidate how geomorphological processes at local and larger scales create and maintain ecologically important habitats (Davies *et al.* 2000, Thomson *et al.*

2001). The gravel reach of Fraser River, British Columbia, is an appropriate prototype from which to develop a hierarchical habitat classification with widespread applicability to gravel-bed rivers because the fluvial processes responsible for creating and maintaining habitat units are relatively intact.

1.3 Thesis Statement

I will present a three-level, hierarchical habitat classification developed for the gravel reach of Fraser River, but intended for general use in large, gravel-bed rivers. The classification has a geophysical basis and characterizes the physical organization of the gravel reach. Level 1 identifies sub-reaches of distinct channel morphology and gradient, and Level 2 corresponds with gravel bar units. Level 3 habitat types are nested within gravel bar units and represent a spatial scale expected to be ecologically relevant to resident and anadromous fish that use the gravel reach for rearing.

Within the framework of the hierarchical habitat classification, I will evaluate the expectation that physically distinct habitat types, which develop as a consequence of sediment transport and deposition, present fish with distinct functional opportunities for rearing, and that physical disturbance producing a change in habitat character will elicit a detectable ecological response. This expectation is embodied in three premises:

Premise 1: Morphological features defined by sedimentation patterns at the bar scale define habitat types at the scale of 10^1-10^2 m that are morphologically distinct, and occur ubiquitously along the gravel reach.

Fluvial processes of sediment deposition and erosion preserve themselves as signature features on gravel bar surfaces. These features are the building blocks of complex bar morphology and are expected to be associated with physically distinct habitat types. Evaluating this premise will entail examining the physical distinctiveness of habitat types and the spatial consistency of habitat characteristics along the gravel reach. That is, to what extent do bar-scale sedimentary features possess physically distinct attributes that qualify them as habitat units, and do reach-scale characteristics influence habitat attributes at the local-scale?

Premise 2: Habitat types created by within-bar sedimentation patterns specify a spatial scale that is ecologically relevant to invertebrates and fish, and the physical distinctiveness of habitat types lends each a predictable assemblage of invertebrate and fish species.

The physical characteristics that discriminate among habitat types are expected to influence the distribution of benthic invertebrate and fish species to the extent that species demonstrate specific habitat associations. Hence, evaluating this premise requires examination of the ecological significance of habitat types by determining the congruence of spatial relationships between habitat types and invertebrates and fish. That is, how well does ecosystem structure match the physical organization of the river and to what extent are invertebrate and fish assemblages within habitat types unique? Together, Premises 1 and 2 highlight the linkage amongst fluvial processes, their provision of physically distinct habitat types, and the assemblages of species that habitats support. Moreover, evaluating these premises is a necessary precursor to testing Premise 3, below.

Premise 3: The strength of the association between physical habitat structure and the distribution of invertebrates and fish determines the extent of ecological response to physical disturbance, exemplified by gravel mining.

Physical disturbance by gravel mining causes immediate changes to bar morphology that, in turn, may affect aquatic organisms because ecological impacts are transmitted mostly by way of physical alterations to river habitat. The ecological response to physical disturbance is predicted to be significant in a system where the species-habitat association is strong, compared to a system where species have widespread distributions and relatively indiscriminate habitat associations. Evaluating this premise includes, first, examining the physical changes caused by bar scalping and, second, determining whether or not these changes translate into a change in habitat use by local invertebrate and fish communities. That is, to what extent does bar scalping affect physical habitat structure and to what extent do the habitat-specific associations of invertebrates and fish influence the nature of ecological response?

The thesis focuses on alluvial habitats associated with the perimeter of gravel bars, which represent foci of habitat diversity and biological productivity in large rivers (Thorp 1992, Johnson and Jennings 1998). Fish sampling methods were chosen to be most effective in near-shore areas and, as such, were selectively biased towards juvenile fish. This bias was acceptable because juvenile fish typically have stronger habitat-specific associations compared to adults, and therefore are better indicators of the structural attributes and condition of river habitat (Schiemer *et al.* 1991). Moreover, the persistence of river populations is dependent on the survival of juvenile fish.

1.3.1 Outline

The thesis consists of nine chapters; Chapters 4 through 8 present core research results. These five chapters build on one another and are ordered as sequential contributions to the thesis. The format of core chapters was chosen to approximately match stand-alone documents, therefore consisting of comprehensive Introduction, Data Analysis, and Discussion sections. The overall sampling design, and field and laboratory methods are presented separately in Chapter 3 to avoid repetition among these chapters.

Chapter 2 describes the physical and ecological setting for this research. Emphasis is placed on characterizing the geomorphology and hydrology of the Fraser River basin generally, and the gravel reach specifically, in order to contextualize the hierarchical habitat classification, which is presented in Chapter 4.

Chapter 3 describes site selection, the sampling schedule for field data collection, and methodology for all field sampling and laboratory processing.

Chapter 4 presents a hierarchical habitat classification for the gravel reach of Fraser River and examines the physical distinctiveness of Level 3 habitat types that occur at a spatial scale expected to be most relevant to aquatic organisms, particularly juvenile fish. The spatial integrity of the habitat classification is evaluated by comparing the physical characteristics of habitat types among sub-reaches and channel types. The chapter concludes with a discussion of the geomorphological genesis of habitat types and the provision of small-scale habitat units as a function of larger-scale sedimentary features.

Chapters 5 and 6 assess the ecological relevance of Level 3 habitat types by characterizing the assemblages of benthic invertebrates (Chapter 5) and fish (Chapter 6) associated with habitat types, and evaluating the congruence between habitat structure and the distribution and abundances of organisms. Material pertaining to invertebrates and fish is divided into two chapters for convenience, but the chapters share a common set of objectives.

Chapters 7 and 8 examine the physical and ecological responses to gravel extraction by bar scalping. These responses are evaluated in the context of the physical and ecological organization of the gravel reach, as described in Chapters 4 through 6, and consider how the strength of the association affects the resilience of the ecosystem to physical disturbance.

Chapter 9 presents conclusions and recommendations for river management.

Chapter 2. Physical and Ecological Setting

2.1 Study Area

The present-day Fraser River drains approximately 233,000 km^2 of south-central British Columbia. A diversion in the upper Nechako basin effectively reduced the drainage area from 250,000 km^2 in 1952. Originating in the Rocky Mountain trench, the river flows northwest between the Columbia and Rocky Mountains before turning south. It flows south across the dry Interior Plateau and between the humid Coast Mountains and Cascade Mountains, before making an abrupt turn west at the town of Hope. From Hope, the river flows through the Fraser Valley and drains into the Strait of Georgia at Vancouver. Sandheads is the most downstream point of Fraser River and corresponds to river-km (rkm) 0. Herein, *lower Fraser River* refers to the segment extending 165 km upstream from Sandheads to Hope (**Figure 1-1**).

This research was conducted in the Fraser Valley downstream of Hope, between Laidlaw and Sumas Mountain (rkm 145 – 105). Within this reach of Fraser River, the riverbed consists predominantly of gravel-sized material and substantial volumes of gravel are deposited in the channel zone on an annual basis; hence, it is referred to as the *gravel reach* of Fraser River. The majority of data collection and field activities were conducted in the 25-km river segment between Chilliwack and Agassiz (**Figure 2-1**).

2.2 Hydrology

Over 300 major tributaries contribute to Fraser River flow, including drainage from large lakes such as Shuswap, Kamloops, Chilko, Quesnel, and Harrison. The mainstem channel is unregulated, although there are at least 802 licensed dams on tributaries within the basin (Fisheries and Oceans Canada 1985). The majority of dams are small and for irrigation purposes, particularly in the Interior Plateau. On a global scale, the annual mean flow of Fraser River is rated as “moderately affected” by flow regulation based on several criteria (Dynesius and Nilsson 1994): a dam regulates flow on a major tributary (Kenney dam on Nechako River); 26% of the mean annual discharge is contained in reservoirs (gross capacity); and one diversion (Nechako River to Kemano River) extracts flow from the basin. Along its 1350 km length, Fraser River spans 10 of the 14 provincial biogeoclimatic zones that differ with respect to physiography, geology, climate, and vegetation (Meidinger and Pojar 1991). Specifically, these zones, named for the dominant forest type, are coastal western hemlock, interior cedar-hemlock, interior Douglas-fir, Ponderosa pine, bunchgrass, montane spruce, Engelmann spruce-subalpine fir, mountain hemlock, sub-boreal spruce, and sub-boreal pine spruce.

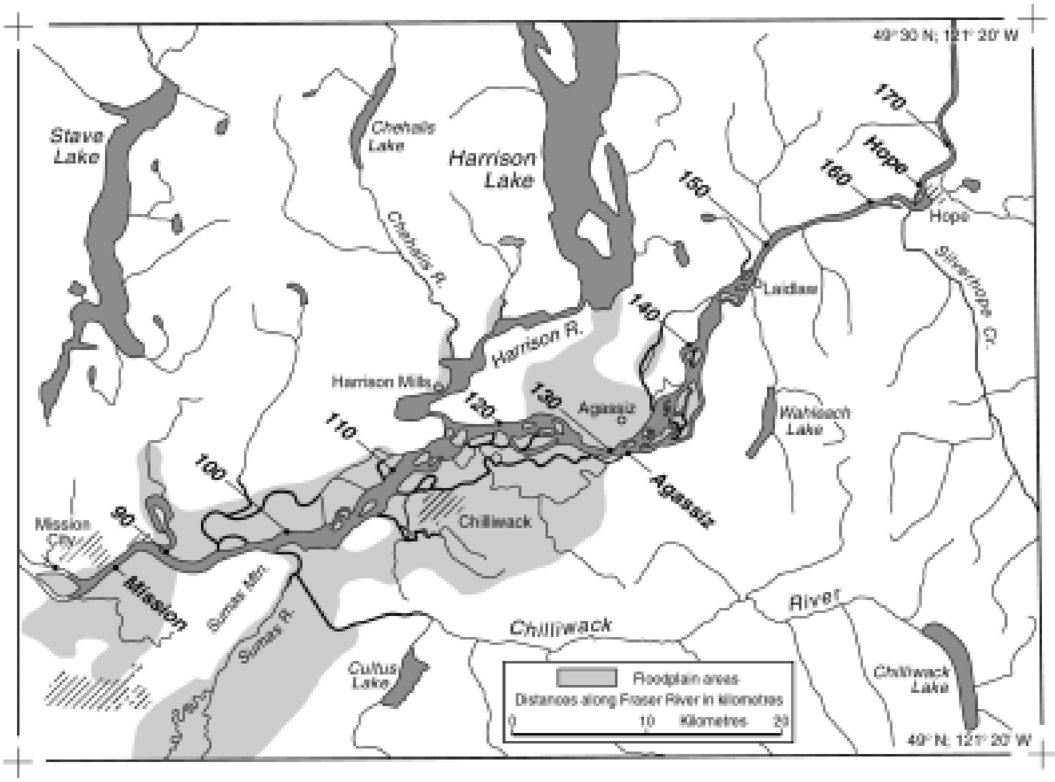


Figure 2-1. Location map of gravel reach of Fraser River, Hope to Mission (rkm 85 to 165). The majority of data collection occurred between Chilliwack and Agassiz, corresponding with rkm 110 to 135 (from McLean *et al*. 1999).

Annual precipitation varies greatly across the Fraser River basin, ranging from 280 to 500 mm yr^{-1} in the Interior Plateau (Ponderosa pine zone) and 1000 to 4400 mm yr^{-1} in the Coast Mountains and Fraser Valley (Meidinger and Pojar 1991). The climate of the lower Fraser River is dominated by maritime air masses and the passage of high and low pressure systems travelling generally from west to east. High-pressure systems are most common in summer months and can produce extended periods of dry weather. The frequency of low-pressure, cyclonic systems increases during autumn and winter, creating more unsettled weather and the period of highest precipitation.

Water Survey of Canada collects hydrologic data for Fraser River at Hope (Station ID 08MF005) and Mission (Station ID 08MH024), which approximately coincide with the upstream and downstream boundaries of the gravel reach. The Agassiz Bridge was a third measurement site within the gravel reach (Station ID 08MF035), but the gauge station was abandoned after 1986 due to cross-sectional changes. Discharge remains virtually constant between Hope and Agassiz due to the lack of tributary inputs. Between Agassiz and Mission, tributary inputs from Harrison and Sumas rivers increase drainage area by 4.6%, mean flow by about 18%, and mean flood flow by 10-15% (McLean *et al.* 1999). Unless otherwise stated, discharge values reported herein were measured at Hope, where the hydrometric station has been in operation since 1912 (Church and McLean 1994).

The annual hydrograph of Fraser River is unimodal, with low flow during winter and high flow in spring and early summer due to snowmelt. Freshet refers to the period of high discharge and suspended sediment transport, which generally occurs from May through July (**Figure 2-2**).

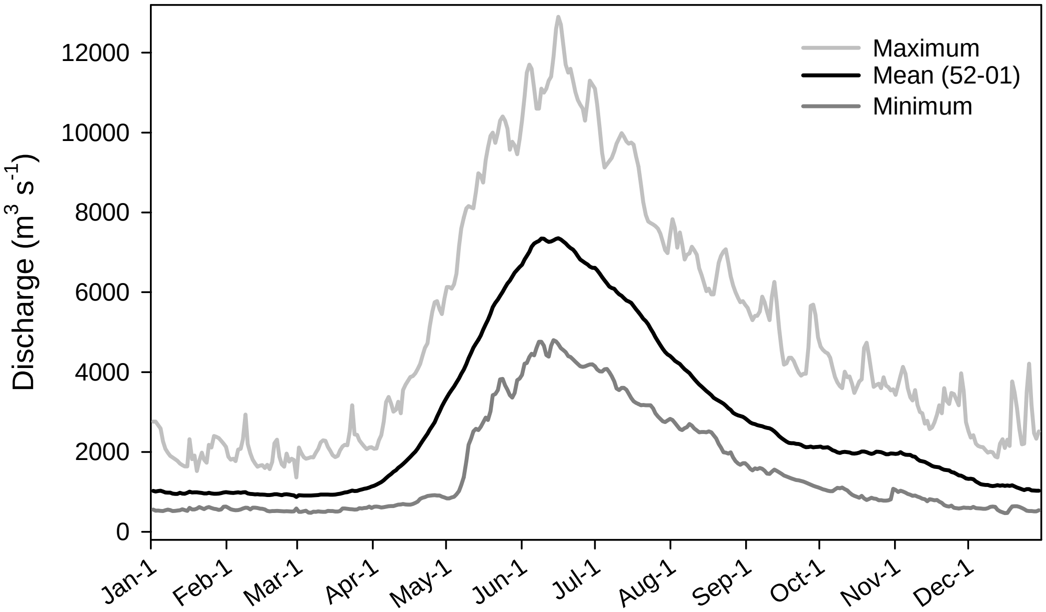


Figure 2-2. Average, minimum, and maximum daily discharge at Hope (1952-2001).

Water levels increase rapidly on the rising limb of the flood hydrograph in April, whereas the declining limb of the hydrograph is relatively prolonged and water levels recede slowly through August and September. Average peak flood discharge at Hope is 8766 $m^3 s^{-1}$ and the long-term mean annual flow is 3410 $m^3 s^{-1}$ (McLean *et al.* 1999). The largest flood on record reached 15,200 $m^3 s^{-1}$ at Hope in 1948. Although not officially measured, the even larger flood of 1894 has been estimated at 17,200 $m^3 s^{-1}$ at Mission. With winter low discharge being about 1000 $m^3 s^{-1}$ and flood discharge typically approaching 10,000 $m^3 s^{-1}$, the annual range of flows is approximately 10*x*.

2.3 Geomorphology of the Gravel Reach

2.3.1 Sediment Supply and Delivery

The Fraser River Basin was most recently glaciated in the Pleistocene Epoch and thick fills of glacial deposits were left in the main valley and its major tributaries. Upland areas of the basin are not major sediment sources; hence, Quaternary sediments from the last glacial period supply the main sediment load to the drainage network through erosion of river banks and terraces. The bulk of the source material is fine-grained and consists of glacial till, glacio-lacustrine silt, earth flow, and debris flow deposits (Church 1990). As a consequence, water turbidity due to elevated suspended sediment concentrations (sand, silt, clay) is high for much of the year. Larger sediments (gravel and cobble) make up a much smaller proportion of the total sediment load; only about 1% of the total sediment load at Agassiz is greater than 2 mm in size and transported as bed load (Church and McLean 1994).

Significant volumes of sediment are mobilized and transported from glacial deposits in the upper Fraser Basin due to the steep channel gradient (**Figure 2-3**). Fraser Canyon between Lytton and Yale is a major hydraulic pinch-point through which significant volumes of sediment are passed. Where the river emerges from Fraser Canyon near Hope, the channel gradient slackens and the river begins to adjust its form, carrying with it the substantial sediment load. Fine sediment remains in suspension, however, the rapidly declining gradient forces the deposition of cobble and gravel-sized sediment between Hope and Mission. Annual gravel deposition in the channel zone between Hope and Mission has produced, on a large scale, a confined alluvial fan. Mountainous terrain confines lateral growth of the fan, while dyking restricts sedimentation to the contemporary channel zone. Consequently, the alluvial fan is aggrading (building vertically) each year with sediment deposition. The transition from a gravel-bed to sand-bed channel is approximately at Mission, downstream of which the gradient is not sufficiently competent to transport coarse particles (>2 mm in size). The reach between Hope and Mission is referred to as the *gravel reach* because the material that forms the bed and lower banks, and thus defines the morphology of the channel, is predominantly gravel.

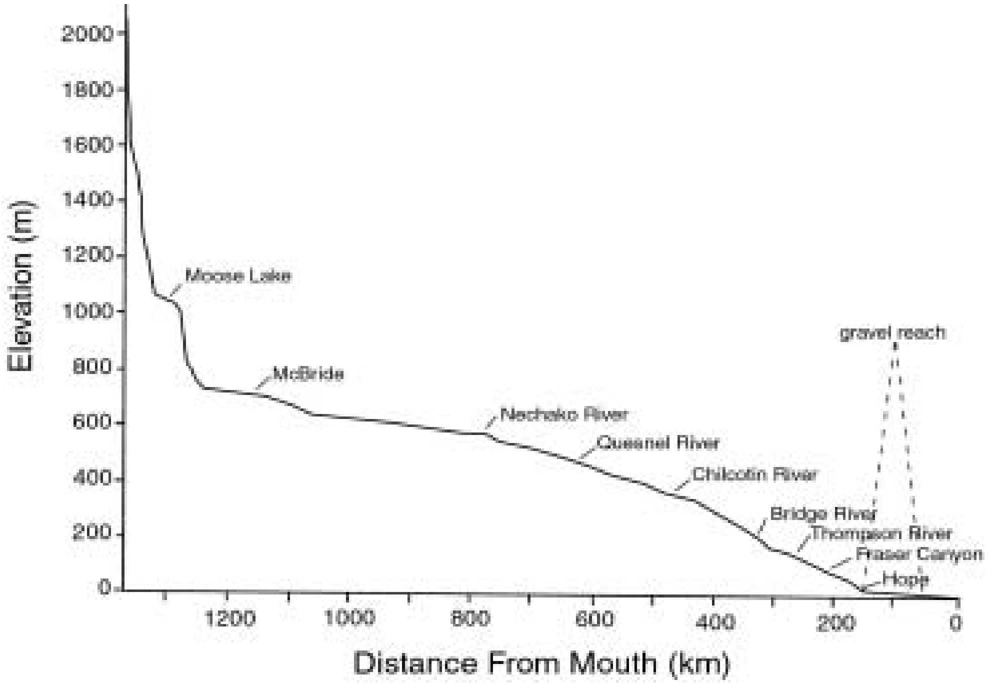


Figure 2-3. Channel gradient of Fraser River, headwater to estuary (from Northcote and Larkin 1989).

2.3.1.1 Suspended Load

The suspended sediment load constitutes material carried in suspension because water currents convey upward directed momentum that equals the submerged weight of the particle. In the gravel reach, the suspended sediment load is equated to *wash material* (McLean *et al.* 1999), which normally is transported directly through the reach but may represent a significant fraction of the upper banks, islands, and secondary channels where it is deposited during floods

The Sediment Survey Section of the Water Survey of Canada carried out a comprehensive sampling program over twenty years to measure suspended sediment transport in the gravel reach of Fraser River. Measurements were collected at Agassiz and Mission between 1966 and 1986, and at Hope between 1966 and 1979 by taking volumetric samples of the water column. Based on these measurements, annual total suspended sediment is practically identical at Hope, Agassiz, and Mission, averaging 17 x 10^6 t yr^{-1}. Suspended sediment concentration increases rapidly on the rising limb of the flood hydrograph through March, peaking by late April or early May, and declining at a moderate rate through the summer months (**Figure 2-4**). Peak suspended sediment concentration occurs, on average, 1 month prior to peak discharge. The largest fraction of the suspended load is transported by flows between 7500 and 8000 m^3 s^{-1}, which correspond approximately to a return frequency of 1.5 years (McLean *et al.* 1999). Higher magnitude floods transport a relatively smaller fraction of the long-term sediment load; this pattern is consistent with other large rivers (Wolman and Miller 1960).

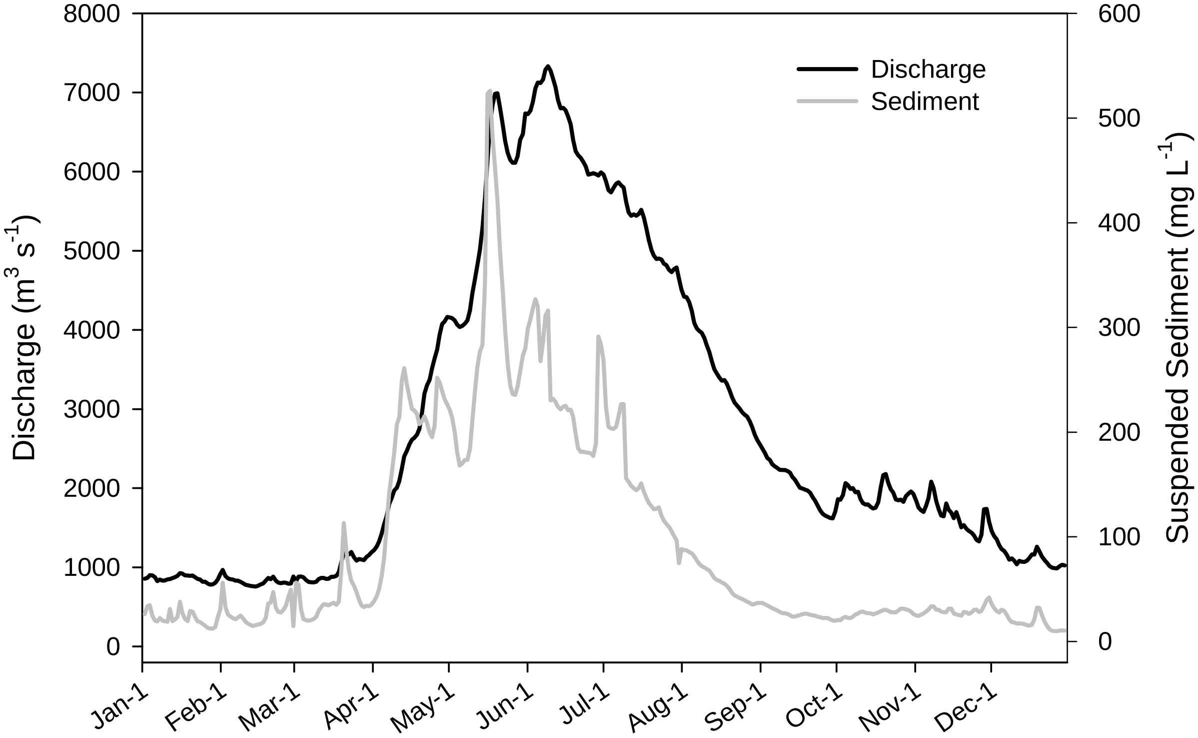


Figure 2-4. Annual hydrographs of discharge and suspended sediment, averaged over a 5-year period for which data were available (1993 to 1997).

Data presented by McLean *et al.* (1999) indicate that the suspended load at Agassiz during freshet typically consists of 35% sand, 50% silt, and 15% clay. Nearly half of the sand fraction is very fine sand (0.063-0.125 mm) and the coarsest particles in suspension rarely exceed 1.0 mm. Within each year, suspended sediment concentration follows a well-defined clockwise hysteresis characterized by an increasing load of suspended sand transported between January and May, and a decreasing load between June and December. This pattern, together with the low representation of sand in the channel bed, is indicative of supply limited conditions for fine sediment in the gravel reach with transport rates apparently governed by the supply of sediment rather than hydraulic conditions (McLean *et al.* 1999).

2.3.1.2 Bed Load

Bed load transport typically is measured by placing some form of sediment trap directly on the riverbed to collect particles too heavy to be suspended and therefore moving along the river bottom by rolling and sliding. Depending on the sampling device used, particles moving by saltation may or not be included in bed load measurements. Bed load transport was measured at the Agassiz Bridge between 1968 and 1976 using a basket sampler over a range of flows by Water Survey of Canada. On a given sampling date, multiple samples were collected from six or fewer vertical segments across the channel and results were then integrated to estimate total bed load transfer through the cross-section. The sampling program was summarized and evaluated for the precision of the measurements by McLean *et al*. (1999). Based on this analysis, annual bed load transport was estimated to be about 130,000 m^3 yr^{-1} ± 40% at Agassiz.

Estimating sediment transport through a channel cross-section (i.e., sediment transfer) by direct measurement has several inherent weaknesses: bed load movement is highly sporadic both in space and time near threshold transport conditions (Wilcock and McArdell 1993); bed load is believed to move in waves or clumps at all flows (Gomez *et al.* 1989); and collecting representative samples is both physically and logistically challenging. An enormous sampling effort is usually necessary in large channels for even moderately precise estimates of bed load transport by direct measurement. Moreover, the local transport rate estimated by direct measurement at a cross-section is most likely not representative of sediment transfer along an extended segment of natural channel. An additional criticism, relevant to this research, is that sediment transfer measurements provide no information on the spatial pattern of sediment recruitment and exchange (erosion and deposition) along the channel.

Because the morphology of a channel is a direct consequence of sediment transfer by the river, an alternative method of sediment transport estimation is to quantify morphological change from repeated cross-sectional or bathymetric channel surveys. This “sediment budget” approach was first proposed by Popov (1962) and Neill (1969), and has been used to estimate long-term bed material transport rates in the gravel reach of Fraser River (McLean and Church 1999, Church *et al.* 2001, Ham and Church 2003). Bathymetric surveys conducted in 1952 (Laidlaw to Mission), 1984 (Agassiz to Mission), and 1999 (Laidlaw to Mission) were used for the computations. The sediment budget approach is well suited to the gravel reach because several necessary conditions are met: gravel transport downstream of Mission is virtually zero; there are no intervening tributary sediment sources within the reach (Harrison Lake serves as a sediment trap for Harrison River); and repeated bathymetric surveys are available.

Whereas direct sediment transfer measurements isolate the sediment transported as bed load directly along the bed, sediment budget estimations include all material forming the channel bed and lower banks. In the context of sediment budget measurements, the term *bed material load* is therefore preferred over bed load, because it makes the connection between particle transport and the role served by the particles in morphological development and sediment storage within the channel. In the gravel reach of Fraser River, bed load and bed material load are practically interchangeable terms because the bed material consists mostly of particles >2 mm in diameter, which are always transported as bed load (McLean *et al.* 1999). The distinction, nevertheless, is worthwhile for this research because of the interest in linking sediment transport to channel morphology and fish habitat development. The proportion of wash material in the bed (mostly contained within interstitial spaces between particles) can be determined by sediment sampling, so that estimates of gravel transport (i.e., bed material load) can be derived from sediment budget computations.

The sediment budget approach is conceptually simple but computationally sophisticated, and influenced greatly by the availability and density of survey data along the channel. Interpolation of river topography is necessary where data are lacking (i.e., between cross-sections), which introduces bias and inaccuracies to the mathematical exercise (Ham and Church 2003). For this reason, sediment budget estimates are never final and subject to refinement and revision as additional information and improved computational techniques become available.

Based on the sediment budget approach, annual gravel transport averaged approximately 200,000 m^3 yr^{-1} ± 35% between 1952 and 1999 at Agassiz (Ham and Church 2003). Substantially larger volumes of sediment, on the order of 1 million m^3 yr^{-1}, are redistributed locally within the reach during annual freshet. Extending the calculations upstream to Laidlaw (**Figure 1-1**) yielded an estimate of 195,000 m^3 yr^{-1} ± 35% annual gravel transport since 1952, but calculations of flux between Agassiz and Laidlaw since 1952 indicate persistent sediment degradation in the upper part of this reach. This finding highlights the valuable spatial resolution gained by the sediment budget approach, which has obvious advantages for river management.

Estimated bed material deposition between 1952 and 1999 translates into an average aggradation rate of 2.3 mm yr^{-1} over the active channel zone between Agassiz and Sumas Mountain (**Figure 1-1**). Gravel deposition, however, is not evenly distributed but is highly variable in space and time, and zones of degradation occur despite the predominantly aggradational character of the reach (**Figure 2-5**). Zones of erosion, from which substantial quantities of sediment may be transferred from banks and islands into the channel, are associated with a downstream deposition zone (McLean and Church 1999). In Fraser River, erosion zones locally correspond with the locations of lateral channel shifting and bank instability that develop episodically along the channel and largely are responsible for the complex morphology of the gravel reach.

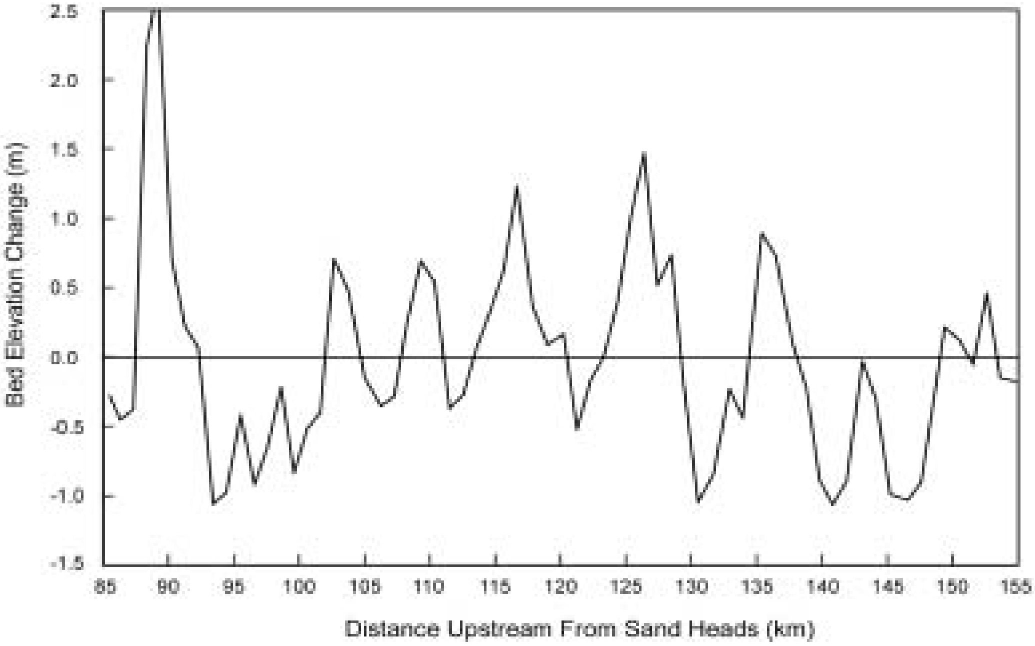


Figure 2-5. Distribution of sedimentation along the gravel reach based on average bed level change for 1-kilometre cells, 1952-1999 (from Church *et al*. 2001).

2.3.2 Sediment Transport and Channel Morphology

The morphology of the gravel reach of Fraser River is referred to as “wandering” (Neill 1973, Desloges and Church 1989), and is a direct consequence of sediment storage and processes of erosion and deposition in the reach (**Figure 2-6**). Wandering rivers are characterized by an irregularly sinuous channel, frequent and large wooded islands, and low-order braiding. An extensive network of perennial, seasonal, and abandoned channels within the floodplain is common, reflecting the lateral instability of the channel and often indicating locations where the main channel once flowed. A wandering morphology, with multiple channels separated by gravel bars and large islands, describes the gravel reach between Laidlaw and Sumas Mountain. Upstream of Laidlaw, the cobble-bed channel is single-thread and confined by Pleistocene terraces, landslide material, and bedrock. Downstream of Sumas Mountain, the mostly sand-bed channel is single-thread and confined by dyking.

Figure 2-6. Characteristic wandering river morphology of the gravel reach, Fraser River (at Agassiz). Flow is from right to left (photo taken March 27, 1999; discharge: 700 $m^3 s^{-1}$).

Wandering channels develop as a result of intermittent and modest volumes of bed material transport. They are common in piedmont valleys of mountainous landscapes such as the Canadian Cordillera, where the legacy of glaciation has left behind large volumes of sediment in river valleys (Church and Slaymaker 1989). Church *et al*. (2001) examined the interplay between discharge, channel gradient, and sediment load, and ascertained that the gravel reach of Fraser River maintains its wandering regime conditional on the intermittent supply and transport of bed material. A substantial increase in sediment load would shift the channel towards a braided morphology due to increased sediment storage within the channel zone (Desloges and Church 1989), whereas a decrease in sediment load or reduction in sediment availability would initiate bar erosion and lead to a single-thread and simplified meandering morphology. The latter condition is more likely to occur due to the increasing use of bank revetment (usually riprap) over the past century to resist bank erosion and protect land and infrastructure on the floodplain. Over 75 km of riverbank are protected between Hope and Mission, corresponding to 46% of the total bank length (Church *et al.* 2001). The effect of riprap and bank protection has been to decrease the active channel width by 22% since 1912, and thereby reduce sediment supply and exchange within the channel zone (Ham and Church 2002).

The past century encompasses a period of significant human modification to the lower Fraser River (Healey 1997). Riparian vegetation has been cleared from river banks and floodplain areas, reducing the quality of bank-line habitat and local recruitment of organic matter to the river. Moreover, large wood and snags were removed from the channel in the early century and, later, a debris trap was installed near Hope to intercept wood that posed a navigation hazard. The debris trap continues to operate during freshet, collecting approximately 65,000 m^3 yr^{-1} of wood debris (PGL Organix 1996) that would otherwise serve as an important morphological attribute, habitat element, and source of carbon to the gravel reach (Sedell and Froggatt 1984). Over 100 km of highly productive floodplain side channels, representing zones of nutrient exchange and rearing habitat for fish, have been isolated from the mainstem by dyking (Rosenau and Angelo 2000, Ellis *et al.* 2004). Major floodplain wetlands also were drained and isolated from the river for agricultural use in the early 20th century, most likely reducing overall productivity of the river. The comparable habitat in tropical rivers, várzea floodplain lakes, is largely responsible for the tremendous productivity of these systems (Junk 1984). Lastly, both flood management and the industrial need for aggregate has resulted in the extraction of at least 4.8 million m^3 of sediment from the channel zone of the gravel reach since 1964 (Weatherly and Church 1999), averaging 117,000 m^3 yr^{-1}. Evidence from other systems indicates that in-stream gravel mining can have detrimental effects on fish habitat and channel morphology (Kondolf 1994, Kondolf 1997, Brown *et al.* 1998), although the effects on Fraser River are uncertain. This topic will be examined in greater detail in Chapter 7.

Despite engineering actions over the past century, the gravel reach of Fraser River maintains a relatively natural wandering morphology and processes of sediment transfer are intact. Sedimentation features associated with processes of gravel bar development create habitat units of differing physical character and a range of channel sizes that together provide varied combinations of velocity, depth, and substrate that support a diverse assemblage of fish species. Side channels and off-channel bays provide rearing habitat for many species during freshet, and vegetated bank edges along the channels and islands provide riparian habitat where cover, terrestrial insects, nutrients and microhabitat features are available. The sequence of sediment erosion and deposition around bars also maintains fish habitat of high quality and supports benthic invertebrate production because the sediment is episodically reworked and cleaned. It is no coincidence that the gravel reach supports at least 28 species of fish and Fraser River is one of the great salmon producing rivers in the world (Northcote and Larkin 1989).

2.3.2.1 Gravel Bar Growth and Morphology

Along the gravel reach, prominent gravel bars are exposed at low water levels and are indicative of major zones of sediment deposition. Lateral bars attach themselves to the channel banks and islands, point bars develop at river bends, and mid-channel bars accumulate in areas of flow expansion (Kellerhals and Church 1989). Gravel bars reflect the fact that sediment aggradation within the reach is discontinuous and strongly localized. The transfer distance of bed material in any one flow event approximates the distance between bar deposits, which ranges between several hundred meters and 2 to 3 km (Church *et al.* 2001). The transport dynamics of bed load sediment contrast sharply with that of suspended sediment, which generally travels continuously over considerable distances.

Bed material, once entrained from an eroding bank or bar edge, moves by rolling, sliding, or saltation along the bed until insufficient hydraulic competence is encountered, usually at the next bar. Persistent deposition in zones of low competence leads to bar growth and, as a result, the channel cross-section area is reduced and the flow is forced to erode adjacent banks or bars to maintain conveyance (Knighton 1998). This pattern of sediment deposition and nearby sediment erosion repeats itself along the gravel reach to produce a repeating sequence of bar-pool units along the channel (**Figure 2-7**). Smaller particles are more easily entrained and will move greater distances than larger ones as a consequence of selective transport (Ashworth and Ferguson 1989), and the bed material may exhibit a downstream fining trend in grain size. This trend may be revealed both at the gravel bar scale (Smith 1974), with individual gravel bars consisting of coarser material at the bar head and finer sediment at the bar tail, and at the reach scale (Knighton 1998) corresponding to the decrease in channel gradient and flow competence.

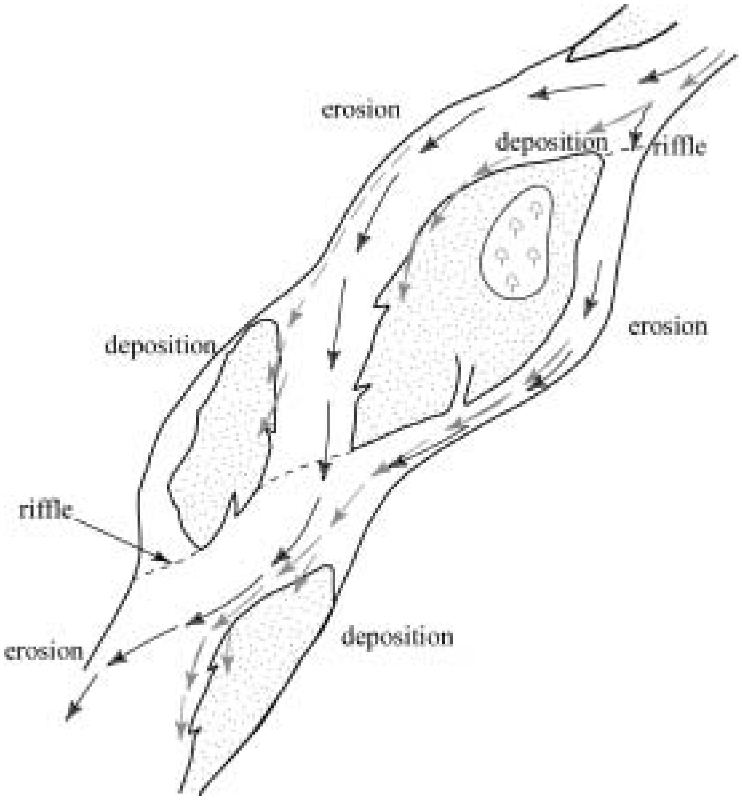


Figure 2-7. Schematic showing pattern of gravel movement and resulting pools and gravel bars. Black lines indicate flow of water and grey lines indicate pattern of gravel movement.

Sediment sampling in August 2000 confirmed this downstream trend at both the spatial scale of the gravel reach and individual bars (Church and Ham 2004). At the reach scale, average median grain size of surface materials increases from 22 mm at rkm 100, to 33 mm at rkm 125, and 59 mm at rkm 150. The median grain size of sub-surface material shows an identical trend: increasing from 9 mm at rkm 100, to 20 mm at rkm 125, and 26 mm at rkm 150. Substantial variability in sediment texture across bar surfaces distorts the fining trend at the bar scale. This variability is typical of the gravel reach, being a product of the antecedent conditions of bar growth, sediment transport, and channel instability that interact together to shape the present bar morphology. The fining trend is nonetheless discernible over most bar surfaces. For example, median surface grain size at Powerline Island, beneath the Agassiz Bridge, decreases from 45 mm at the bar head, to 38 mm at mid-bar, and 21 mm at the bar tail.

Three bed-surface types may be generalized for the gravel reach of Fraser River (Church *et al.* 2000). *Clean gravels* have little or no fine material present. *Sandy gravels* are those with the coarse gravel framework partially obscured by a thin, discontinuous veneer of sand and/or significant interstitial sand. *Blanket sands* are those where the gravel framework is buried beneath a sequence of sandy deposits. Clean and sandy gravels dominate the active portions of the gravel reach. Within annually inundated areas, blanket sands are limited to areas of low hydraulic stress, often in shallow secondary channels behind bars, at the downstream ends of bars, and overlying areas of high elevation where vegetation can establish.

Sub-surface sedimentary deposits within the channel zone are strongly bimodal, with the coarse fraction consisting of gravel-sized material (25-30 mm median diameter) and the fine fraction being medium sand (McLean *et al.* 1999). The sand fraction usually represents 15-30% of the bed material and occupies the interstitial spaces between stones. The size distribution of surface sediment is typically unimodal because the sand fraction is lacking. The gravel fraction of surface material is usually similar in texture to the sub-surface sediment (McLean *et al.* 1999).

Processes of sediment deposition and erosion preserve themselves as signature features on the surface of gravel bars. These features generally correspond with sedimentary units of relatively uniform grain texture, which are the building blocks of complex bar morphology. Systematic study of the topography and sedimentology of such features is a means by which processes of bar development and habitat creation may be elucidated. Closer examination of bars along the gravel reach of Fraser River reveals many such sedimentary features worthy of discussion.

Distinctive *gravel sheets*, one or many grains in thickness, are seen draping across bar surfaces and occasionally building a step-front of substantial height (**Figure 2-8**). The sheets generally have a steep leading edge, indicative of downstream advancement, and consist of a relatively uniform grain size that may be different from that of the host bar surface. The surface of gravel sheets typically resembles a high, advancing *gravel platform*, but in some cases may be dissected by secondary channels. Several sheets may be found on some bar surfaces, representing different transport events and sequentially layering themselves across the bar.

The deposition of gravel sheets also can produce *gravel lobes*, which deposit along the edges of bars and may extend some distance into the channel. These lobes are accretionary wings building both downstream and laterally into the channel, usually with a steep inner avalanche face and a moderately sloping outer edge that is worked by main channel flows. Successive deposition of relatively small lobes produces scalloped topography and nook-like features along the bar edge (**Figure 2-9**). Larger lobes may create quiet bays along their inner edge.

Figure 2-8. Examples of gravel sheets on Herrling Island (upper), Harrison Bar (middle), and Queens Bar (lower).

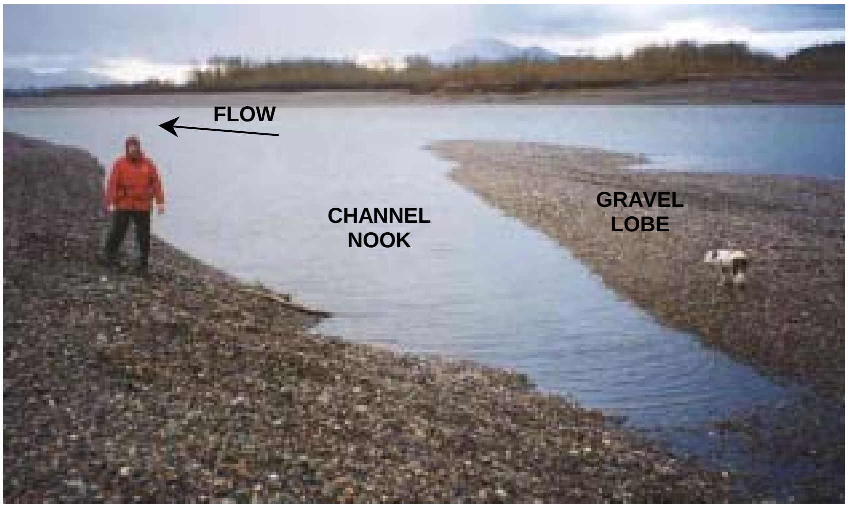


Figure 2-9. Accretionary gravel lobes creating bays (Calamity Bar, upper photograph) and channel nooks (Foster Bar, lower photograph).

On the receding limb of the flood hydrograph, flows washing across bar surfaces and gravel sheets may concentrate and dissect the bar front as small drainage channels. These channels tend to develop in clusters and produce characteristic *chute and lobe* couplets on the leading avalanche face (**Figure 2-10**). Couplets may vary in size and form, developing as relatively small features along riffle faces or as larger and highly textured avalanche faces elsewhere. The clean, well-sorted nature of these sedimentary features, completely void of wash material, indicates that a relatively rapid drop in water level is necessary for chute and lobe couplets to form.

Figure 2-10. Examples of chute and lobe couplets at Big Bar (upper photograph) and Webster Bar (lower photograph).

2.3.2.2 Island Development

Some gravel bars build to the point that they have slack water across the top, even in flood. This level usually corresponds to the level of the most probable annual flood with a 1.6-year return period (Desloges and Church 1989). The hydraulic conditions on these bar tops favour sand deposition and allow pioneering vegetation, mostly black cottonwood (*Populus balsamifera*) and willow (*Salix* sp.), to establish (Boniface 1985). Fine sediment becomes trapped more readily with the presence of vegetation and the bar surface accretes vertically to initiate island development. Islands are stable features relative to gravel bars. They may persist for centuries and support mature forests of cottonwood and spruce before eventually being attacked by the river again. The longevity of many islands, unlike gravel bars that may undergo minor changes in configuration on an annual basis, is an indicator of the slow pace of large-scale change along the river.

Concern over the systematic reduction in channel area due to bank hardening prompted a detailed historical analysis of channel islands in the gravel reach of Fraser River (Ham and Church 2002). Unlike many river banks, islands in the active channel are unprotected from erosion and are therefore vulnerable to long-term changes in flow regime or channel alignment, or to changes in sediment supply and transfer within the channel, all of which are possible responses to bank hardening. The study revealed a reduction in mature island area within the channel zone since 1912, due mostly to their isolation from the active channel by dyking, floodgate construction, and siltation. While many mature islands with well-developed forest cover were absorbed into the floodplain, the loss has been offset by a recent increase since the 1970s in young islands with early stage bush vegetation. As a result, total island area within the channel zone does not differ significantly between 1928 and 1999 (Ham and Church 2002).

Young islands typically are more fragmented and discontinuous than older island surfaces (**Figure 2-11**); hence, their functional role as a stabilizing element in the channel and source of wood and organic carbon to the ecosystem is reduced. Moreover, young islands lack the complex network of secondary channels typical of mature islands, which dissect their surface and provide refuge for fish during freshet. On mature islands, these channels are bounded by thick riparian vegetation that provides shade, cover, wood, and a source of drop-in terrestrial insects for aquatic organisms (**Figure 2-11**). The ecological significance of these channels has not been investigated thoroughly, however, they likely represent exceptionally productive habitats during freshet. Because young islands are lower in elevation (Ham and Church 2002), surfaces become submerged earlier than mature islands and their potential contribution as habitat of reduced hydraulic stress is likely reduced.

Large, mature islands are the most stable fluvially formed features in the gravel reach and appear to develop through one of several processes. Likely the most common process is sedimentation and the infilling of inter-island secondary channels, leading to the coalescence of several smaller islands over decades. Minto Island at the mouth of Harrison River appears to have developed in this manner over the past forty years (Church and Weatherly 1998). Alternatively, islands may form as a result of channel avulsion into the floodplain, which effectively increases the active channel area. Although perhaps occurring prior to settlement of the Fraser Valley, bank hardening over the past century now limits the possibility for bank avulsion along much of the reach.

Figure 2-11. Examples of a young island at Harrison Bar (upper photograph) and mature island at Herrling Island (lower photograph).

The progressive development of young islands since the 1970s appears to be related to a recent period of below-average flows, which coincides with a cyclic trend in flow regime over the last century. The flow record since 1912 reveals relatively abrupt regime shifts in 1925, 1948, and 1977, with the period between 1948 and 1977 corresponding to above-average flow and the periods before and since having below-average discharge (McLean and Church 1999). These shifts coincide with the Pacific Decadal Oscillation (PDO, Mantua *et al.* 1997), a stable climate signature of the North Pacific Ocean. Increased island growth since the 1970s appears to have been facilitated by an extended period of below-average flows that has allowed fine sediment deposition and vegetation establishment on low-elevation bar surfaces. If a regime of below-average flow persists, either as a result of the cyclic PDO or forecasted climate change (Morrison *et al.* 2002), island growth will continue. A negative implication, however, is that a low-flow regime is associated with reduced lateral instability, leading to increased island coalescence with the floodplain and eventually a simplified channel zone (Ham and Church 2002).

2.3.2.3 Secondary Channels

Discontinuous, low-order braiding is a characteristic of wandering channels and refers to the division of deepwater channels around bars and islands (Neill 1973, Desloges and Church 1989). The channel is not everywhere divided around bars (hence, discontinuous), and where it is, there are only two or a few deep channels (hence, low-order) that meander in an irregular fashion within the channel zone. A dominant thalweg remains discernible, regardless of the degree of braiding, effectively distinguishing the main channel from all secondary channels (**Figure 2-6**). By definition, the *main channel* conveys the highest discharge throughout the year and the surface sediment is subject to bed load transport during freshet. Bed material consists mostly of “clean gravels”, in which fine sediment has been winnowed away by selective transport. Secondary channels (*side* and *summer* channels) may be differentiated on the basis of flow activity and position relative to islands.

S*ide channels* convey substantial discharge (Ellis 2004) and minor bed load sediment during freshet but may have little or no flow during winter. Wetted areas at the upstream and downstream ends of the channels persist year-round. Their orientation is usually parallel to the main channel as they flow behind islands and major bar-island complexes. Side channels are relatively large and stable features. Bed material may contain a low to moderate amount of fine sediment (i.e., “sandy gravels”) at the upstream end and a moderate to high amount at the downstream end (i.e., “blanket sands”).

Smaller *summer channels* are seasonally inundated during freshet only and are often oriented diagonal to the main channel, dissecting islands and bar tops. Fine gravel may be transported during peak freshet flows, along with a substantial sand load. Characteristic of some summer channels is the intrusion of a gravel sheet (or wedge of superimposed sheets) into the upper end, which then stalls in the channel. The bed surface elevation of summer channels is high relative to the main channel and bed material contains a high proportion of fine sediment (i.e., “sandy gravels” and “blanket sands”). Summer channels may occur as a dense network of unstable channels dissecting unvegetated gravel bar deposits or as meandering single-thread channels dissecting vegetated island habitats; hence, the number of summer channels associated with any one bar-island complex is highly variable along the gravel reach.

2.4 Ecology of the Gravel Reach

2.4.1 Carbon - The Basis for the Ecosystem

The fundamental energy source for river ecosystems is carbon. Carbon is cycled through the food chain of an ecosystem, beginning with photosynthesis by autotrophs such as algae and plants to convert dissolved inorganic carbon (DIC) to its organic forms. The carbon stored in plant matter is metabolized by heterotrophs such as insects and some fish species through consumption, and organic carbon is also transferred between organisms when one eats another. Ultimately, fish production is dependent on a series of carbon transfers through the food chain of a river, beginning with the fixation of $C0_2$ by plants in photosynthesis. The carbon base of Fraser River is therefore critical to the productivity of the fishery resources of the gravel reach. However, only limited effort has been directed at documenting and monitoring the organic base of the ecosystem (Healey and Richardson 1996).

Carbon is supplied to the gravel reach from a variety of sources: autochthonous carbon (produced in situ) from local phytoplankton, algae and aquatic plants; allochthonous carbon (produced outside the river margins) from local, diffuse sources such as overland flow and seasonal inundation of riparian margins; and carbon delivered from upstream, or nearby tributaries and upland areas. The expectation for large, non-tropical rivers is that local carbon sources are insignificant in comparison with the carbon supply from upstream sources (Vannote *et al.* 1980). An intact riparian zone with floodplain areas and off-channel habitat can significantly improve retention time and carbon production in large rivers (Grubaugh and Anderson 1989, Brunet *et al.* 1994), thereby improving the rate by which organisms assimilate energy for growth. Quiet water habitats such as embayments and back channels also improve carbon production by supporting populations of phytoplankton and zooplankton in large rivers (Thorp *et al.* 1994).

Healey and Richardson (1996) determined that, on a seasonal basis, carbon from upstream sources is a significant contributor to the total carbon load (46-91%) of Lower Fraser River. During winter months in particular, organic carbon processing in the gravel reach exceeds inputs: total organic carbon declined by 10% between Hope and Mission and by 38% from Mission to the river mouth in March, based on data from 1973 and 1974 (Benedict *et al.* 1973, Hall *et al.* 1974). The reverse pattern occurs during April and May as the rising water level inundates islands, river margins, and riparian habitats, producing a positive carbon balance between Hope and Mission. Total organic carbon measured 4.3 mg L^{-1} at Hope and 8.4 mg L^{-1} at Mission, based on May samples in 1973 and

1974. Hence, local carbon inputs to the gravel reach from island and riparian habitats exceed processing during spring and surplus carbon supplies up to 50% of the carbon processing demand downstream of Mission. Carbon processing in the gravel reach is again higher during summer months than inputs when water level declines and juvenile fish density is highest.

Floodplain clearing for agriculture, channel confinement by dyking, and loss of floodplain habitat downstream of Mission likely have reduced total carbon production in the lower Fraser River by at least 2.5 times historic levels (Healey and Richardson 1996). Carbon processing and invertebrate production in seasonally flooded wetlands and back channels also were probably important contributors to the productivity of the river prior to dyking; however, many of these habitats have been lost over the past century (Healey 1997, Rosenau and Angelo 2000, Ellis *et al.* 2004). Historically, a more complex river channel and more abundant supply of wood along the channel margins would have resulted in greater trapping and processing of the organic carbon delivered to the main channel. Healey and Richardson (1996) speculated that such changes to the physical structure and productivity base of the lower Fraser River are at least partly responsible for significant declines in abundance of anadromous salmon species and white sturgeon over the past century.

2.4.2 Algae and Plants

Most planktonic algae (free-living) in Lower Fraser River are diatoms and cell densities are highest in spring (200-400 cells mL^{-1}) and lowest in summer and autumn (Northcote and Larkin 1989). Densities are consistently below those in Columbia River, where counts are approximately twice as high (Williams and Scott 1962). High turbidity combined with the absence of large mainstem lakes and impoundments on Fraser River likely contribute to relatively low phytoplankton density. However, secondary channels and quiet water habitats such as embayments most likely support significantly higher diatom concentrations than the main channel (Thorp *et al.* 1994).

Attached algae in various habitats of the Fraser River estuary have been studied in some detail (e.g., Pomeroy and Levings 1980). Periphytic and planktonic algae were sampled between 1972 and 1973 at sites in the gravel reach as well (Northcote *et al.* 1975). In general, high turbidity limits algal growth to rocks and logs in shallow water where light penetration is adequate. Macrophytes are assumed to have local importance in contributing to primary production in secondary channels and backwater habitats, but their overall contribution is likely low (Northcote and Larkin 1989). The contribution of nutrients from pink salmon carcasses and the resulting increase in productivity have not been investigated, however, filamentous algae was observed along main channel gravel bar margins in winter 2000 where pink salmon had spawned in the previous autumn. Periphyton chlorophyll-*a* in these areas averaged 25 ug cm^{-2} in March 2000, compared with 8 ug cm^{-2} in side channel areas where pink salmon had not spawned (L. Rempel, unpublished data). With the number of spawning adults averaging 5 million pink salmon in the gravel reach every two years (Fraser River Action Plan 1995), the nutrient input is possibly substantial albeit short-lived as discharge and turbidity increase with freshet.

2.4.3 Benthic Invertebrates

Benthic invertebrates reside in or near the substrate-water interface and represent a major food source for juvenile and adult fish in streams and lakes (Northcote *et al.* 1978b). The life cycle of most taxa in the gravel reach is univoltine (i.e., one year), and typically new larvae hatch in summer, mature through fall and winter months, and emerge in late winter or early spring (Hynes 1970, Reece and Richardson 2000). As a consequence, invertebrate density varies significantly by season in near-shore habitats of gravel bars: summer and fall densities average approximately 2000 animals m^{-2}, winter density typically exceeds 12,000 animals m^{-2}, and spring density between April and June averages less than 400 animals m^{-2} (L. Rempel, unpublished data).

Rempel *et al*. (1999) examined the invertebrate community associated with bars in the main channel of the gravel reach and demonstrated the importance of shallow sloping bar flanks as a flow refugium during seasonal flooding. Sampling took place between April and September 1995, during freshet, and sampling locations in each month were stratified by water depth. Whereas prior to flooding in April, total density was highest offshore at 1.5 m water depth, density was highest at shallow depths <0.5 m during freshet. A broad diversity of taxa with varying feeding behaviours and morphologies were concentrated along the margins of gravel bars during flooding, demonstrating the significant value of this habitat. The lateral distance over which these organisms migrated was between 30 and 100 m, corresponding to the horizontal distance that water levels shifted across the bar tops during flooding. Depth-averaged velocity remained less than 30 cm s^{-1} in shallow water throughout freshet while offshore velocity increased to over 160 cm s^{-1} during peak flooding. Only a select group of taxa specialized for habitats of high hydraulic stress were found offshore at peak flow.

At least 55 unique benthic taxa have been identified from main channel gravel bars in the gravel reach (Dymond 1998), however, taxonomic diversity is considered low relative to smaller tributary streams (Reece and Richardson 2000). Orthocladiinae, a midge sub-family of the order Diptera, largely predominates (usually >50%) the benthic invertebrate community (Reece and Richardson 2000, Rempel *et al.* 2000). Mayflies belonging to the families Baetidae, Ephemerellidae, and Heptageniidae are significantly less abundant, but also common. The majority of animals occupying gravel-cobble substrate along gravel bar flanks are collector-gatherers (usually >80%) that feed on particulate organic matter, whereas other feeding groups such as predators and grazers feeding on algae have low representation (Rempel *et al.* 2000).

The invertebrate community inhabiting low velocity, soft-bottomed habitats in Lower Fraser River was examined by Northcote *et al*. (1976). Data collection was carried out in 1973 and 1974 and sampling occurred in off-channel bays at two locations in the gravel reach, near the Harrison River mouth and immediately upstream of the Agassiz Bridge at Herrling Island. Sample collection was stratified by depth and substrate type (mud-sand and sand-gravel). Densities averaged 1400 animals m^{-2} in mud-sand substrates and both density and biomass increased with water depth (from <3 m to >6 m). In sand-gravel substrates, densities averaged approximately 600 animals m^{-2} and both density and biomass decreased with water depth.

A total of thirty-five taxa were found in soft-bottomed habitats of the gravel reach (Northcote *et al.* 1976), which is considerably less than the taxonomic richness in habitats of coarse sediment associated with gravel bars (Dymond 1998). Taxonomic resolution was lower in the study by Dymond (1998); hence the estimate of 55 taxa in coarse substrate habitats is conservative. Oligochaetes and two species of Diptera belonging to the sub-families Diamesinae and Chironominae were the most numerically abundant taxa collected in soft-bottomed habitats (Northcote *et al.* 1976). Lamprey ammocoetes were numerically uncommon but were the dominant contributor to total benthic biomass. Non-dipteran aquatic insects such as mayflies, stoneflies, and caddisflies made up a negligible proportion of the benthic community, both with respect to density and biomass.

The abundances and distributions of invertebrates, more so than fish, are strongly influenced by the physical habitat (Power *et al.* 1988, Scarsbrook and Townsend 1993). The relation is sufficiently strong that invertebrate distributions form the basis for habitat classifications in some river systems (Buffagni *et al.* 2000, Rabeni *et al.* 2002), and invertebrates are preferred for biomonitoring and environmental impact studies (Schindler 1987, Bailey *et al.* 1998). Within the gravel reach of Fraser River, major gradients of depth, velocity, and substrate create a complex variety of habitat features, occurring at multiple spatial scales, which are expected to correspond with the spatial distribution of invertebrate taxa. The basis for this expectation is the habitat templet concept (Southwood 1977, Southwood 1988), which postulates that the habitat is a templet (template) that imposes restrictions on the life history strategies and ecological traits of resident organisms. Numerous studies have presented evidence in support of the habitat templet concept, indicating that benthic community structure is primarily governed by the physical conditions of the habitat (Power *et al.* 1988, Scarsbrook and Townsend 1993, Rempel *et al.* 2000).

The habitat templet concept appears to hold at the spatial scale of major gravel bar-pool units in Fraser River: benthic invertebrates generally segregate themselves among bar habitats of gravel-cobble sediment and pool habitats of fine sand and mud. The association between invertebrate distributions and physical gradients of depth, velocity, and substrate previously was confirmed for the gravel reach at the smallest spatial scale of micro-habitat patches of coarse sediment (Rempel *et al.* 2000). A correspondence between the spatial organization of major habitat types around gravel bars (e.g., riffles, bar tails, bar heads) and the distribution of invertebrates remains unconfirmed and will be investigated in this thesis.

2.4.4 Fish

The fish species assemblage of the Fraser Basin is a legacy of the last glaciation, which reached its maximum in British Columbia 14.5 kya (thousands of years ago) and blanketed most of Canada and the northern United States with ice. All of British Columbia was glaciated except barren mountaintops, the northeast tip of Graham Island in the Queen Charlottes Archipelago, and the northwest corner of Vancouver Island (Tipper 1971, Warner *et al.* 1982). Fish species were confined to one or several areas of refuge on the fringes of the ice front: the Chehalis refuge; the Columbia Basin refuge; the Mississippi refuge; and the Yukon refuge. Deglaciation and ice-melt were underway by 16 kya and the sequence of deglacial events over the next 7,000 years produced enormous volumes of water, which gave way to temporary glacial lakes and spillways. Many drainage divides became inundated during this period and some watersheds were temporarily connected by large meltwater lakes, providing fish with the opportunity to disperse into newly deglaciated regions.

Recolonization of the Fraser Basin began from the Chehalis and Columbia refuges in the south, and later via the Mississippi refuge from the east. Some euryhaline fish gained access directly from the Pacific Ocean (e.g., landlocked smelts, *Spirinchus thaleichthys*, in Harrison and Pitt lakes appear to be glacio-marine relicts) and others during shifts in sea level. At least two species of fish from the Chehalis refuge followed lakes and meltwater channels northward up Puget Sound into the Fraser Lowlands. Glacial retreat from the Columbia River basin produced a complex series of lakes including Glacial Lakes Thompson and Shuswap in the southern interior, which drained south into the Columbia system and connected the Fraser Basin to the Columbia refuge. This route was the most direct and long lasting and, not surprisingly, the Columbia refuge became the primary source of postglacial immigrants to the Fraser River basin (McPhail and Carveth 1993). Although the Mississippi glacial refuge hosted significantly more species and was responsible for the recolonization of much of Canada, the migration route into the Fraser Basin was convoluted and few species succeeded (McPhail and Lindsey 1970). Dispersal northward into Canada was via Glacial Lake Agassiz, which then connected via a series of proglacial lakes and meltwater spillways into Glacial Lake Peace and, later via Clearwater River and Glacial Lake McConnell into the Peace River system (Rempel and Smith 1998). A waterway connection between Peace River and the Fraser Basin was via Glacial Lake Prince George, which developed several times as a result of ice blockage in the middle Fraser River (Tipper 1971).

Today, Fraser River supports 52 species of fish, of which 43 are native (McPhail and Carveth 1994). The majority are immigrants from the Columbia glacial refuge, but two species originated from the Mississippi refuge (white sucker, *Catostomus commersoni*, and brassy minnow, *Hybognathus hankinsoni*) and two species are from the Chehalis refuge (Salish sucker, *Catostomus* sp. and Nooksack dace, *Rhinichthys* sp.). At least 28 species representing 9 families are found in the gravel reach today (Chapter 3); this reach of Fraser River is more species-rich than all upstream reaches. Seven species are considered at risk (http://srmwww.gov.bc.ca/atrisk/index.html), including the red-listed and endangered white sturgeon (*Acipenser transmontanus*). Overall, however, a comparative study found the fish community of the lower Fraser River to be relatively unchanged from 1973 to 1994 with respect to species composition, density, and biomass (Richardson *et al.* 2000).

This diverse faunal assemblage is testament to the high quality of physical habitat in the gravel reach, which is a product of the geography and glacial history of the Fraser basin. The steep channel gradient upstream of Hope delivers fast, cold water and a supply of cobble and gravel sediment that becomes deposited and accumulates as gravel bars and islands between Hope and Mission. Where gravel accumulations build to become islands, the riparian perimeter represents a source of large woody debris and organic matter to the channel, as well as a source of terrestrial insects for the aquatic food web. Sand and silt are transported mostly in suspension through the reach and produce moderate turbidity that provides cover for aquatic organisms. Large lakes in major tributaries act as sediment traps, but also represent a critical habitat for anadromous fish species. The seasonal recruitment of gravel and cobble sediment from upstream and local redistribution within the gravel reach together produce topographically complex habitat features that support a range of fish species and life stages. Seasonal flooding also cleans and reworks the substrate on an annual basis, making conditions suitable for invertebrate production and spawning by fish.

The family Salmonidae dominates the faunal assemblage with 11 species found in the gravel reach. Ten of these 11 species are salmonines (sub-family Salmoninae). Only one other large river (Skeena River) equals this native salmonine diversity and Fraser River exceeds all rivers worldwide in terms of total stock abundances of salmonine fishes (Northcote and Larkin 1989). The five anadromous Pacific salmon species contribute the large majority to this total abundance. Of these five species, coho (*Oncorhynchus kisutch*) and sockeye salmon (*O. nerka*) are mainly migratory through the gravel reach and are captured relatively infrequently as juveniles during summer months (Brown *et al.* 1989). Spawning by sockeye occurs mostly in tributary streams in the upper basin, and in both upstream and downstream tributaries for coho.

Chinook salmon (*O. tshawytscha*) also spawn in the middle and upper Fraser Basin (except the notable Harrison River population, Healey 1998); however, substantial numbers rear in the gravel reach for up to one year before migrating to sea. In this study, chinook salmon were the second most common species collected after leopard dace (*Rhinichthys falcatus*). Based on DNA evidence, the majority (>70%) of these chinook originated from the Stuart and Nechako systems, as well as the North Thompson (<30%, **Appendix E**). Their emigration from natal spawning grounds and prolonged residency in the gravel reach suggest highly favourable conditions for rearing. Levings and Lauzier (1991) found that juvenile chinook in the gravel reach were larger than upriver fish in winter months and fed on a range of insect taxa including Diptera, Trichoptera, and Plecoptera. Juvenile chinook were mostly associated with steep, riprap banks in winter (Beniston *et al.* 1986) and shallow sloping bar flanks of the main channel during summer months (Zallen and Boyd 1986, Brown *et al.* 1989). Refuge habitats such as back channels that are protected from the main current appeared to be of importance during peak flow (Zallen and Boyd 1986).

Fraser River supports the largest population of pink salmon (*O. gorbuscha*) in the northeast Pacific basin south of Alaska (Fraser River Action Plan 1995), and approximately 85% of the population spawns in the mainstem gravel reach. Spawning takes place every two years in September and October on an odd year cycle, primarily on the flanks of gravel bars in the main channel. Recent winter sampling near the Agassiz Bridge found eggs in main channel habitats up to 1.8 m water depth (winter low flow) and at distances greater than 60 m offshore (Stables and Perrin 2002). Spawning substrate is typically gravel and cobble sediment with relatively little fine material (Heard 1991). Eggs incubate through the winter and emerge in early spring, migrating immediately to the Fraser estuary. Hence, no rearing takes place in the gravel reach.

In contrast, chum salmon spawn outside main channel habitats of the lower Fraser River, but fry (*O. keta*) spend a brief period rearing in the gravel reach in March and April as they migrate to the

Strait of Georgia. Stomach contents of chum fry captured in the gravel reach indicated feeding on aquatic insects (mostly Diptera and Ephemeroptera) and zooplankton (**Appendix G**). Fraser River is the largest producer of chum salmon in British Columbia, and the majority of fish spawn downstream of Hope in tributaries such as Harrison, Chilliwack, and Stave rivers. It is believed that most chum in the gravel reach spawn in side channels. Spawning takes place over gravel-cobble substrate where groundwater upwelling occurs (Salo 1998).

An anadromous, non-salmonid of special status in the gravel reach is eulachon (*Thaleichthys pacificus*). A member of the smelt family, eulachon spends most of its life in the ocean but spawns in coastal rivers. Eulachon are culturally significant to First Nations throughout coastal areas of British Columbia, and also are important as a food fish for white sturgeon that migrate downstream to intercept the spawning run each spring. The upstream limit of its spawning migration has remained speculative, however, recent acoustic and trawl net sampling found larvae in the gravel reach as far upstream as Herrling Island, above the Agassiz Bridge (Perrin *et al.* 2003b) and eulachon use deep water and gravelly sand substrate for spawning (Perrin and Rempel 2000a). Fraser River supports British Columbia's largest population of eulachon, but a dramatic population decline since 1994 led to the closure in 1998 of the Fraser River fishery (Fisheries and Oceans Canada 1999). Currently, the eulachon fishery is highly restricted based on stock assessment estimates and the species is blue-listed, in recognition of its imperiled status. Eulachon was not captured during this study.

A description of the biology and life history of each freshwater species in the gravel reach is not warranted here (refer to Scott and Crossman 1973), however, two freshwater species residing in the gravel reach deserve special mention. The first, white sturgeon (*Acipenser transmontanus*), is the largest freshwater fish in Canada and occurs throughout the mainstem Fraser River. The most abundant Fraser population resides in the gravel reach and is genetically distinct from upstream populations (R.L.&L. 2000, Smith 2002). Its residency in the gravel reach and its significant cultural and economic importance has been the primary force behind political and local support for habitat protection and ecologically sustainable management of the gravel reach. During this study, only two white sturgeon were collected by gill net and promptly released in a main channel bay off Queens Bar (286 mm and 340 mm fork length).

Adult white sturgeon can grow to over 635 kg and 6 m in length, and live more than 100 years. However, life history traits make white sturgeon highly vulnerable to population declines. Females reach sexual maturity at 25 to 35 years of age and the period between spawning episodes is up to 11 years. Major die-offs of large fish in 1993 and 1994, the majority being female, prompted a moratorium on the harvest of white sturgeon by commercial, sport and First Nations fishers and contributed to its red-listed status in British Columbia. Recent studies coordinated by the Fraser River Sturgeon Conservation Society and carried out in conjunction with a non-retention sport fishery and First Nations have led to a current population estimate of 47,000 sturgeon between Hope and Mission, and the population appears to be growing (Mr. T. Nelson; Executive Director, Fraser River Sturgeon Conservation Society, pers. comm.).

Until recently, all information on the habitat requirements of white sturgeon were based on observations from regulated rivers, primarily Columbia River (Parsley *et al.* 1993, Parsley and Beckman 1994), which suggested white sturgeon prefer high velocity habitats associated with dam tailraces for spawning. However, Perrin *et al*. (2003a) demonstrated the exclusive use of side channels by white sturgeon for spawning in the gravel reach.

A final freshwater species of special mention in the gravel reach is mountain sucker (*Catostomus platyrhynchus*), considered vulnerable and blue-listed in British Columbia. Until recently, only a few individuals had been confirmed in Lower Fraser River (McPhail and Carveth 1993). However, significant numbers of mountain sucker (2,637) were captured during this study. They are known to spawn in shallow, fast flowing riffles of mountain and piedmont streams between late spring and early summer (Wydoski and Wydoski 2002). Growth rates are very slow, as confirmed from individuals collected in the gravel reach in 2000 (Dr. J. D. McPhail, Zoology, The University of British Columbia, unpublished data). Mountain sucker feed almost exclusively on plant material, mostly algae, so the species is an important part of the food chain connecting primary producers to secondary consumers that feed on mountain sucker such as some trout species and white sturgeon (Scott and Crossman 1973).

Overall, the fish species diversity and productivity of the gravel reach are testament to the exceptional habitat it provides. Moreover, the fishery resource has been the cornerstone of aboriginal economies along Lower Fraser River for thousands of years. The wandering and laterally unstable channel pattern and dynamic fluxes of sediment and water are central to this productivity, and are responsible for providing passage for upstream anadromous stocks and spawning and rearing conditions for many species. These conditions include: fast, cool, soft water of moderate turbidity; a range of gravel sizes that are suitable for spawning and support a diverse benthic community; moderate gravel transport that cleans and reworks the substrate on an annual basis; predominantly seasonal flows that ensure substrate stability during egg incubation; a variety of main, side and back-channel morphologies of variable connectivity; an abundance of shoreline providing edge habitats and riparian energy inputs; and a range of depths and velocities at all stages.

Chapter 3. Field and Laboratory Methods

3.1 Site Selection and Sampling Units

Gravel bars were chosen as the spatial unit for study to determine habitat associations of fish and invertebrates and evaluate the ecological response to physical disturbance. Sixteen bars extending from rkm 105 to 145 (**Table 3-1**) were selected for repeated physical and biological sampling between 1999 and 2001. Gravel mining has occurred throughout the gravel reach for at least 50 years and several sites had a history of extraction, as detailed in **Table 3-1**. Including sites in this study with a history of gravel mining was virtually unavoidable because of the widespread extent of mining in the gravel reach. The majority of extractions occurred several years or more prior to the study and had been reworked by the river over multiple freshets. Overall, the sites were distributed evenly among morphologically distinct sub-reaches (described below) and together provided a good database to examine the range of habitats and patterns of habitat use by fish in the river.

Although gravel bars are easily identified as discrete morphological units and provide a spatial framework for sampling, their spatial scale is large relative to the scale of observation by fish. Recognizing this, a major goal of the study became the development of an ecologically relevant habitat classification applicable at the gravel bar scale. The classification also was intended to identify appropriate spatial units among which to stratify sampling effort for fish. Delineating habitat types was not straightforward because the types of aquatic habitats, as well as their extent and distribution, have been inadequately described for large gravel-bed rivers. Defining habitat types in the gravel reach of Fraser River was an iterative process, using a combination of both low flow air photograph interpretation and field surveys over a range of flows. The objective was to identify physically distinct habitat types occurring ubiquitously throughout the gravel reach that develop as a consequence of sediment transport and deposition, and that present fish with distinct functional opportunities for rearing.

The habitat classification was nested in a hierarchical framework first proposed by Church *et al.* (2000) that recognizes morphologically distinct sub-reaches within the gravel reach (Level 1), each consisting of repeating gravel bar-pool-riffle units (Level 2). Habitat units nested within gravel bar units therefore represent the finest level (Level 3) of the hierarchical habitat classification. A detailed description of Level 1 sub-reaches and Level 2 bar-pool-riffle units is provided in Chapter 4.

Table 3-1. Study sites (listed in upstream order), location, and gravel extraction history.

Gravel Bar Site	River km[+]	Sub-Reach[++]	Extraction History[o]
Webster	105	Chilliwack	76*k* in 1969, 48*k* in 1995
Wellington*	108	Chilliwack	No known extractions; Provincial Ecological Reserve #76 (est. 1977)
Queens*	110	Chilliwack	Originally thought to have no extraction history; however, scalping operation documented on 1949 air photograph
Calamity*	115	Chilliwack	No known extractions
Lower Minto* *(in-stream dredging and bar scalping)*	110	Chilliwack	Steelhead site (upstream): >830k (1964-98), averaging 23*k*/yr; removals likely began in 1950s Vosco site (downstream): 1,300k (1966-94), averaging 52*k*/yr 46*k* (1982-83)
Upper Minto*	119	Chilliwack	No known extractions
Harrison*	118	Rosedale	No extractions prior to this study; 70*k* (2000) for this study in February 2000
Foster*	120	Rosedale	76*k* (1971) for dyke upgrade; 230*k* (1974-78); 170*k* (1995); anecdotal evidence suggests mining since 1950s
Carey*	121	Rosedale	No known extractions
Hamilton*	126	Rosedale	4*k* (1978-80); >40*k* (1992-98) and 44*k* (since 2000) from side channel bar
Big	128	Rosedale	Uncertain history; scalping operation documented on 1974 air photograph
Powerline*	130	Cheam	>100*k* (1972) for dyke upgrade; >240*k* (1996-98) by Cheam Indian Band
Lower Herrling*	132	Cheam	34*k* (1978) by MOTH
Tranmer	137	Cheam	54*k* (1979) from main channel upper bar by MOTH; ~120*k* (1993-97) and 10*k* (2001) from side channel lower bar
Spring	140	Cheam	No known extractions
Peters	145	Cheam	No known extractions

*one of eleven sites originally selected to examine bar scalping impacts

[+] approximate river distance upstream from Sandheads (rkm 0)

[++] Five sub-reaches are designated in the gravel reach between Mission and Hope based on trends in channel gradient, riverbed sediment size, and sediment transport regime; three are examined in this study.

[o] *k* is thousand m^3 (Weatherly and Church 1999), private extraction unless indicated (MOTH: Ministry of Transportation and Highways).

The starting point for Level 3 habitat identification was field surveys and physical measurements over a range of flows, mostly on the declining limb of the hydrograph between July and September. Additional sources of information were recent air photographs (15BC99001/1-37 flown 27 March 1999 at a nominal scale of 1:40,000 and SRS 6164/1-92 flown 10 March 2000 at a nominal scale of 1:10,000) that were used to identify sedimentary features and morphologically homogeneous areas around bar perimeters. Observed spatial differences in the distribution of fish species served to validate the presence of habitat features and refine the boundaries of habitat types associated with them (methodology described in Section 3.3).

Together, physical characteristics and coarse spatial differences in the distribution of fish species formed the basis for *preliminary* field-based habitat type identification. The criteria for habitat identification was that replicate units occur *ubiquitously* along the gravel reach and at all stages. Otherwise, the number of habitat types identified would be impractically large because of the inherent physical complexity of the reach. Habitat units were classified as one of ten alluvial habitat types or three bank types on the basis of morphological, sedimentary, and hydraulic characteristics. The ten alluvial habitat types were: riffle, bar head, bar edge, bar tail, eddy pool, open nook, channel nook, bay, bar top, and vegetation. The three bank types were: cut bank, rock bank, and artificial bank. Sampling effort for fish and invertebrates was stratified among these habitat types, which were nested within the spatial unit of gravel bars. This sampling was used to test the hypothesis that the physical distinctiveness of habitat types lends each a predictable assemblage of invertebrate and fish species.

3.2 Sampling Schedule

Flow conditions varied over the three years (1999-2001) of data collection (**Figure 3-1**). Whereas low flow periods in late autumn and winter were similar with respect to discharge and duration, the freshet in each year differed. Ultimately, the freshets in each year provided a good representation of above-average, average, and below-average flood hydrographs. In 1999, peak discharge was substantially higher than average (11,000 $m^3 s^{-1}$ on June 22) and freshet was a prolonged event; water levels remained higher than average through September. Flooding followed an approximately unimodal pattern in 2000 with peak discharge (8000 $m^3 s^{-1}$ on July 6) approaching the mean annual flood discharge. The flood hydrograph was below average and bimodal in 2001, with discharge reaching 6940 $m^3 s^{-1}$ on June 6, subsequently dropping to a low of 4180 $m^3 s^{-1}$ on July 19, and peaking again on July 23 at 7210 $m^3 s^{-1}$. In each of the three years, maximum discharge occurred later than average; peak flow typically occurs by mid June (McLean *et al.* 1999).

Field data collection (habitat surveys and biological sampling) took place over 26 months between July 1999 and September 2001. The sampling schedule was divided into quarterly periods corresponding with season, which was based on the discharge hydrograph. Winter sampling corresponded to flows less than 1500 $m^3 s^{-1}$. Spring sampling was on the rising limb of the hydrograph between 1500 and 5000 $m^3 s^{-1}$. Flows exceeding 5000 $m^3 s^{-1}$ designated the summer sampling period and autumn sampling was associated with flows between 5000 and 1500 $m^3 s^{-1}$ on the declining limb of the hydrograph.

Physical and biological data were collected in all seasons, but seasonal sampling effort was unequal. Several factors influenced the timing and frequency of field sampling: discharge/weather conditions and the safety of personnel; the availability of necessary sampling equipment, including a motor boat; the life cycle of resident and anadromous fish species rearing in the gravel reach; and the life cycle and larval maturation period of benthic invertebrates.

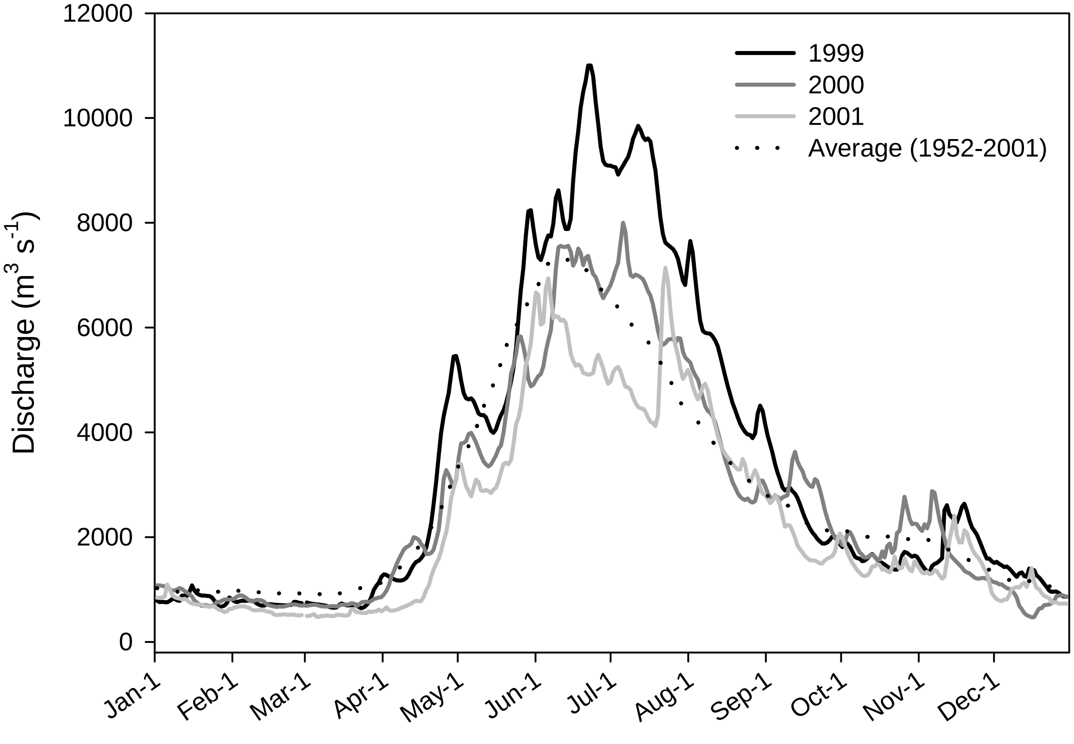


Figure 3-1. Discharge hydrograph for Fraser River in each year of data collection (1999-2001) and the 50-year average (1952-2001), measured at Hope.

Sampling for benthic invertebrates occurred at gravel bar sites throughout the gravel reach in September 1999 and 2001, and March 2000 and 2001 (**Table 3-2**). Invertebrate sampling was limited to three bars within the Rosedale sub-reach only (Harrison, Foster, Carey) for the remaining sampling episodes: April, May, August, September and November 2000, and January 2001. These sites were chosen as part of an experimental study, described in Chapter 7 and 8, to examine the physical and ecological response to disturbance by gravel mining at Harrison Bar. However, they served a dual purpose of providing information on the seasonal distribution and habitat use by invertebrate taxa during these months. In total, 10 "episodes" of invertebrate sampling took place between September 1999 and September 2001. Sampling infrequently over a large spatial scale and more frequently at a local scale allowed for comparisons of temporal and disturbance-related variance with spatial variance in invertebrate community composition within the reach.

Table 3-2. Sampling seasons (S) designated according to the discharge hydrograph of Fraser River at Hope. Data collection periods for fish (F) are in italics and the number of sampling episodes for invertebrates (I) is indicated (1-2 days per episode).

Season	Discharge ($m^3 s^{-1}$)	1999	2000	2001
Winter	<1500	S: 1-JAN to 13-APR *F: -* *I: -*	S: 1-JAN to 6-APR *F: 7-Feb to 6-Apr (20)* *I: 1 episode*	S: 1-JAN to 21-APR *F: 27-Feb to 31-Mar (8)* *I: 2 episodes*
		S: 19-OCT to 31-DEC	S: 18-NOV to 31-DEC	S: 7-OCT to 31-DEC
Spring	1500 - 5000	S: 14-APR to 22-MAY *F: -* *I: -*	S: 7-APR to 21-MAY *F: 7-Apr to 9 May (13)* *I: 2 episodes*	S: 22-APR to 27-MAY *F: -* *I: -*
Summer	>5000	S: 23-MAY to 16-AUG *F: 21-Jul to 16-Aug (9)* *I: -*	S: 22-MAY to 5-AUG *F: 15-Jun to 3-Aug (9)* *I: -*	S: 28-MAY to 2-AUG *F: 26-Jun to 18-Jul (9)* *I: -*
Autumn	5000 - 1500	S: 17-AUG to 18-OCT *F: 19-Aug to 27-Sep (28)* *I: 1 episode*	S: 6-AUG to 17-NOV *F: 15-Aug to 15-Nov (28)* *I: 3 episodes*	S: 1-AUG to 6-OCT *F: 14-Aug to 24-Sep (20)* *I: 1 episode*

Number of fish sampling days in each season is indicated in parentheses.
'-' indicates no sampling.

The term “episode” is used herein to describe each round of invertebrate sampling, which was completed over one or two days and involved replicate sampling at all sites and in all available habitats. As shown in **Table 3-2**, multiple sampling episodes for invertebrates occasionally occurred within one season (e.g., autumn 2000), and each episode was considered to be a discrete sampling event. The time required for fish sampling was significantly greater than for invertebrates, and more than one day often was required for comprehensive sampling of all habitats even at a single site. Reach-wide sampling at all study sites required up to several weeks. Because fish sampling took place more or less continuously over extended periods, the temporal sampling unit for fish data collection was defined by season.

Fish sampling during the winter season took place in February, March, and early April 2000, as well as limited sampling in February and March 2001 (**Table 3-2**). Spring sampling on the rising limb of the hydrograph was in April and May 2000 only. Early summer sampling during peak freshet was limited due to dangerous boating conditions and the lack of exposed beach areas for fish sampling; hence, only nine days of summer sampling at >5000 $m^3 s^{-1}$ occurred in each year. Summer sampling was restricted to July and early August in 1999 and 2000, and June and July 2001. Autumn sampling took place in late August and September of 1999, 2000, and 2001. Overall, sampling effort for juvenile fish was concentrated between July and September of each year, when densities in the river were highest. Late autumn and winter fish sampling was less intensive because densities were lower and the combination of cold water temperatures and high water clarity made fish more difficult to catch.

3.3 Field Methods

3.3.1 Fish

A 17’ aluminum-welded boat with an outboard jet engine was used to travel on the river and access sites for sampling. Fish sampling was authorized under collection permits issued by the Department of Fisheries & Oceans Canada (DFO: 99.199; 00.55; 01.30), the former BC Ministry of Environment, Lands and Parks (now Ministry of Water, Land and Air Protection, MWLAP: FC99-57; FC00-20; FC01-73), and The University of British Columbia Committee on Animal Care (A99-0083, renewed over 3 years). All field sampling was carried out by the writer with the assistance of 1 or 2 technicians.

Fish were collected from habitat units using a variety of sampling gear: beach seine, gillnet, minnow trap, and electro-shocker. All methods were used extensively during the first summer and winter of sampling on a trial basis, and the beach seine was used almost exclusively in subsequent sampling seasons. The beach seine provided the most reliable and consistent catch data, and was the most versatile collection method for different habitat types. All data presented in Chapters 4 through 8 were obtained solely by beach seine sampling. A large beach seine (30 x 3.5 m, 9 mm mesh) was used infrequently and was deployed using a boat in deep and relatively quiet habitats. A smaller and highly versatile seine (12.5 x 2 m, 6 mm mesh) was deployed by wading from shore; its major limitation was that sampling extended to depths less than 1.2 m, the maximum depth one can safely wade in chest waders. Sampling was carried out within recognizable habitat units by dragging the net downstream along the shoreline over a distance of 10-50 m, depending on the size of the habitat unit. Fish became trapped in the net, which was then hauled on shore, and all fish were immediately transferred to holding buckets containing fresh river water for processing.

Although the beach seine was easily deployed in all habitat types and performed in a consistent and reliable manner, its capture efficiency likely varied depending on a variety of factors including habitat type, species of fish, fish size, time of year, and time of day. Each of these factors will bias fish sampling devices (Bayley and Dowling 1993), and some authors recommend attempting to quantify this bias and applying a correction factor to catch data (e.g., Parsley *et al.* 1989). However, Holland-Bartels and Dewey (1997) demonstrated that corrections to compensate for gear bias and environmental conditions are difficult and can be inaccurate because the error of the adjusted data remains high.

Sources of bias that contribute to an underestimate of fish density were from fish either *evading* the net (under the lead line or around the outside edge) or *escaping* through its mesh. To minimize fish evasion of the net, each seine was executed swiftly and only relatively short lengths of beach were sampled at a time. The catch data were discarded for any seine in which the net became snagged. Despite these efforts, it remains probable that bottom-dwelling fish such as longnose dace and sculpin species managed to evade the net in some instances, particularly over coarse substrate. Highly agile and fast-swimming fish such as some salmonid species may have evaded the net in some instances as well. The problem of fish escaping through the net pertains only to very small individuals (<20 mm) whose species identification would be difficult to determine, and to small individuals of longnose dace that are highly streamlined and could pass through the mesh.

The beach seine capture efficiency was not estimated, in part because the biases introduced by physical differences between habitat types and species-specific traits were variable and difficult to estimate, but also because turbidity during most months likely minimized sampling bias. Turbidity has been shown to decrease the reactive distance of fish (Sweka and Hartman 2001), and Gregory and Levings (1998) demonstrated that turbidity in Fraser River reduced the encounter rate between predacious adult fish and juvenile chinook salmon. Gregory and Levings (1998) also reviewed evidence that fish living in turbid water are active throughout the day and benefit from turbidity providing protective cover, which reduces the risk of occupying near-shore areas. Data presented in **Appendix B** supported these findings: summer fish density in daytime beach seines was similar or higher than at night whereas winter density averaged almost 5 times higher at night. Based on this collection of evidence, turbidity during most months of sampling (April through September) likely contributed to realistic estimates of fish density. Density was underestimated during daytime sampling in winter months because of the clear water (**Appendix B**). However, relative comparisons between sites and habitats during winter months should remain valid because density was underestimated by a common factor.

Sampling by beach seine depended on habitat types being present and accessible at each gravel bar site. Although all habitats had a likelihood of occurring at all sites and at all water levels, more often some habitats were absent at a site during any one sampling episode. Consequently, sampling effort varied among sites and dates, and effort could not be stratified equally among habitat types. Common habitat types (e.g., bar edge) were sampled with greater frequency than uncommon ones (e.g., eddy pool) but an attempt was made to sample fish from all available habitats during each sampling period. The unbalanced sampling effort among habitat types and dates was problematic for data analysis, as described in later chapters.

Once collected by each sampling method, all fish were identified to species (McPhail and Carveth 1994) and counted by the writer. Voucher specimens were routinely collected for species verification by Dr. G. R. Haas (BC Ministry of Water, Land and Air Protection). Species identification remained uncertain for very small fish (usually <20 mm) collected during summer months. A minimum of 15 fish representing non-anadromous species and all anadromous fish species in each haul were measured for fork length (mm) and, weather-permitting, weighed (g). Fish were returned to the river at the point of collection promptly after processing. Beach seine sampling sites are shown in **Figure 3-2**through **Figure 3-4**.

Of the 54,684 fish collected by various methods over three years of sampling, 26,771 (49%) were measured for fork length and 11,533 (21%) were weighed to the nearest 0.1 g. The 15 fish of each non-salmonid species were selected to approximately represent the size distribution of each species and the sub-sampled size distribution was then extrapolated to estimate biomass for fish not weighed. Fish weight is related to length and a log-log linear regression provides an accurate model from which to estimate weight (Murphy and Willis 1996). Refer to **Appendix A** for length-weight regression analysis for each fish species.

A substantial amount of effort was devoted to fish sampling between 1999 and 2001 in the gravel reach of Fraser River (**Table 3-3**). Beach seine sampling alone covered a channel area exceeding 340,000 m^2, from which more than 52,000 fish were identified and counted. Twenty-five species were identified over the three years of sampling (**Table 3-4**), including ten salmonid species, white sturgeon (red-listed in British Columbia) and 5 blue-listed species (bull trout, Dolly Varden, coastal cutthroat trout, brassy minnow, mountain sucker, http://srmwww.gov.bc.ca/atrisk/index.html). Juvenile chinook salmon were encountered more consistently in beach seine hauls than any other species, despite the fact that chinook only rear in the reach for up to one year before migrating to the Pacific Ocean. Three cyprinid species, leopard dace, longnose dace, and redside shiner, also were encountered in greater than 50% of beach seine hauls. Leopard dace, followed by chinook salmon, were the two most numerically abundant species in beach hauls.

Table 3-3. Fish sampling effort and catch summary, Fraser River gravel reach, 1999 to 2001.

Sampling Gear	# Events	Total Fish	Total Area (m^2)	Total Time (hr)
Electro-shocker	20	456	2491	-
Gillnet	61	719	-	351
Minnow Trap	534	1091	-	11,480
Beach Seine (all)	960	52,418	341,800	-
Beach Seine (12.5 x 2 m)	933	50,263	328,691	-
Beach Seine (30 x 3.5 m)	27	2,155	13,109	-
TOTAL	**1575**	**54,684**	**344,291**	**11,831**

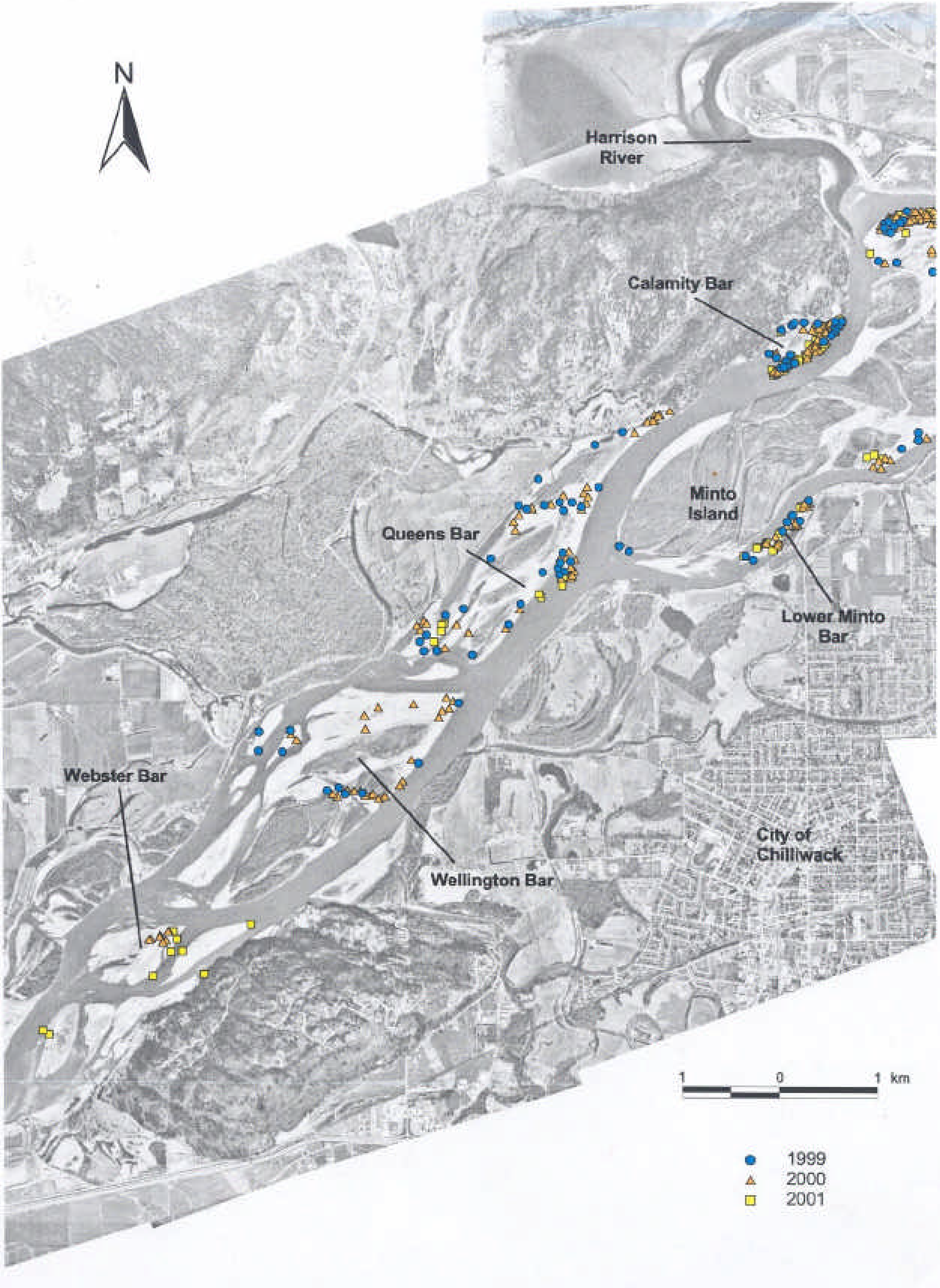


Figure 3-2. Beach seine sampling sites in the Chilliwack sub-reach (1999 - 2001).

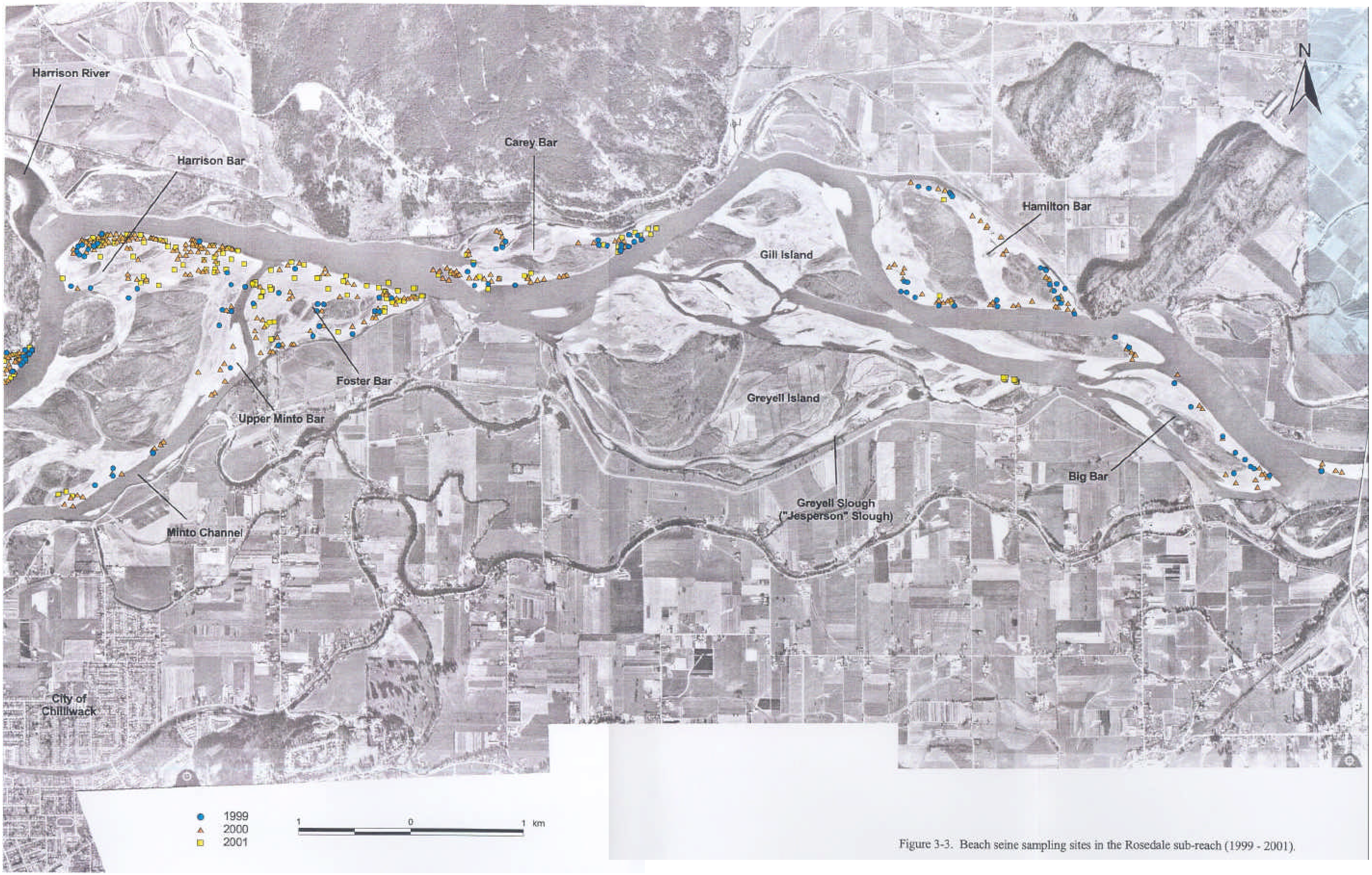


Figure 3-3. Beach seine sampling sites in the Rosedale sub-reach (1999 - 2001).

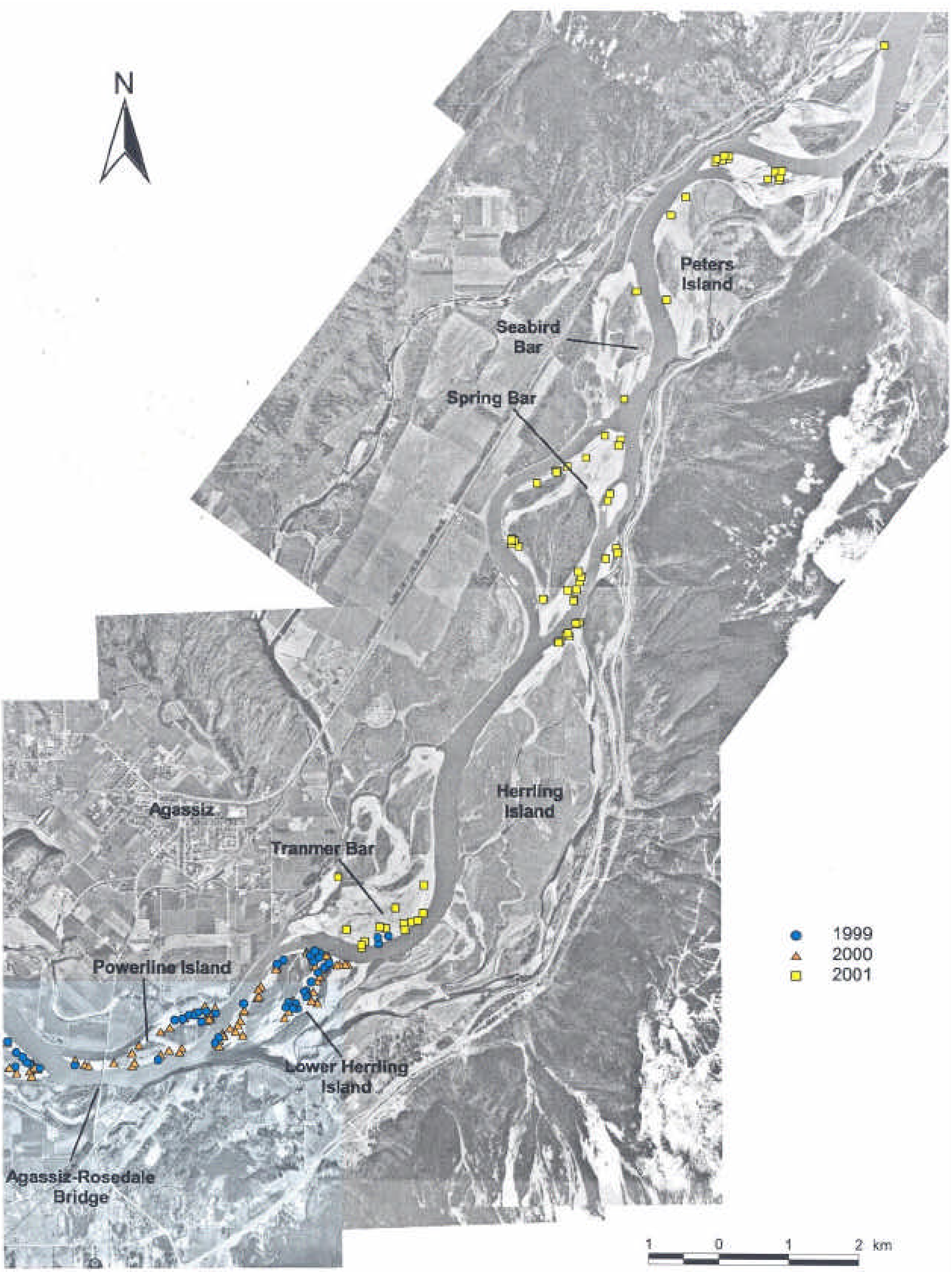


Figure 3-4. Beach seine sampling sites in the Cheam sub-reach (1999 - 2001).

Table 3-4. Fish species known to occupy the gravel reach of Fraser River and 3-letter identity codes assigned to those species captured in this study. The percent (%) of beach seines (n = 960) in which each species occurred is given.

Family	Species	Common name	Code	Presence%
Acipenseridae	*Acipenser transmontanus*	White sturgeon R	WST	0
Salmonidae	*Prosopium williamsoni*	Mountain whitefish	MWF	26.8
	Salvelinus confluentus	Bull trout B	BUL	0.1
	S. malma	Dolly Varden B	DOL	0.1
	Oncorhynchus clarki	Cutthroat trout B	CUT	3.6
	O. gairdneri	Rainbow trout	RBT	9.5
	O. gorbuscha	Pink salmon	PIN	5.4
	O. keta	Chum salmon	CHU	14.9
	O. kisutch	Coho salmon	COH	0.9
	O. nerka	Sockeye salmon	SOC	11.7
	O. tshawytscha	Chinook salmon	CHI	69.1
Cyprinidae	*Hybognathus hankinsoni*	Brassy minnowB	BRA	0.4
	Mylocheilus caurinus	Peamouth	PEA	37.1
	Ptychocheilus oregonensis	Northern pikeminnow	NPM	32.8
	Rhinichthys cataractae	Longnose dace	LND	49.5
	R. falcatus	Leopard dace	LED	65.6
	Richardsonius balteatus	Redside shiner	RSS	51.0
Catostomidae	*Catostomus macrocheilus*	Largescale sucker	LGS	32.5
	C. platyrhynchus	Mountain sucker B	MTS	33.8
	C. columbianus	Bridgelip sucker	BLS	0.1
Gasterosteidae	*Gasterosteus aculeatus*	Threespine stickleback	TSS	16.9
	G. aculeatus trachurus	Marine stickleback	MSB	10.6
Cottidae	*Cottus aleuticus*	Coastrange sculpin	CRS	2.1
	C. asper	Prickly sculpin	PRS	23.6
Petromyzontidae	*Lampetra ayresi*	River lamprey+	LAM	0.4
	L. richardsoni	Western brook lamprey+	LAM	-
	L. tridentata	Pacific lamprey+	LAM	-
Osmeridae	*Thaleichthys pacificus*	EulachonB *	nc	-

R: red-listed, B: blue-listed
+ presence documented in Northcote and Larkin (1989), specimens not identified to species
* presence of eulachon upstream of Agassiz is documented by Perrin *et al*. (2003b)
nc: not captured in this study

3.3.2 Benthic Invertebrates

Benthic invertebrates were collected near-shore using a Surber net (250-μm mesh, 0.09 m^2) by the writer and one field assistant. The substrate was disturbed vigorously by hand to a sub-surface depth of 5-10 cm and benthic material was washed into the net by the current of the river. A minimum of three replicate samples was collected at each location and on each date of sampling. Replicates were taken 5 to 10 m apart and sequentially in an upstream direction to ensure that each location was undisturbed prior to sampling. Samples were preserved in 4% formalin and later processed in the DFO Laboratory at Cultus Lake. Invertebrate sampling locations are shown in **Figure 3-5**.

The Surber net depends on the river current for effectiveness; hence, samples were collected only from habitat units with flowing water (i.e. riffle, bar head, bar edge, bar tail). Sites of representative velocity and substrate within habitat units were targeted for invertebrate sampling. In this way, variability due to environmental factors other than habitat type was minimized. Sampling effort was stratified by habitat type at a given site but varied among dates because the presence of habitat types varied as water levels fluctuated, particularly riffles. Bar edge habitat was the single habitat type that was sampled consistently in all ten sampling episodes, whereas sampling effort in riffle, bar head, and bar tail units was variable. Similar to fish, the unbalanced sampling effort among habitat types and sampling episodes was problematic for invertebrate data analysis, as described in Chapter 5.

Water depth and velocity at 6/10 depth from the surface were measured at the exact location of each Surber sample using a graduated wading rod and Marsh-McBirney velocity meter. The surface sediment was visually classified as the most dominant grain size class (sand, gravel, gravel/cobble, cobble). The degree of sediment embeddedness and any unique habitat features were noted.

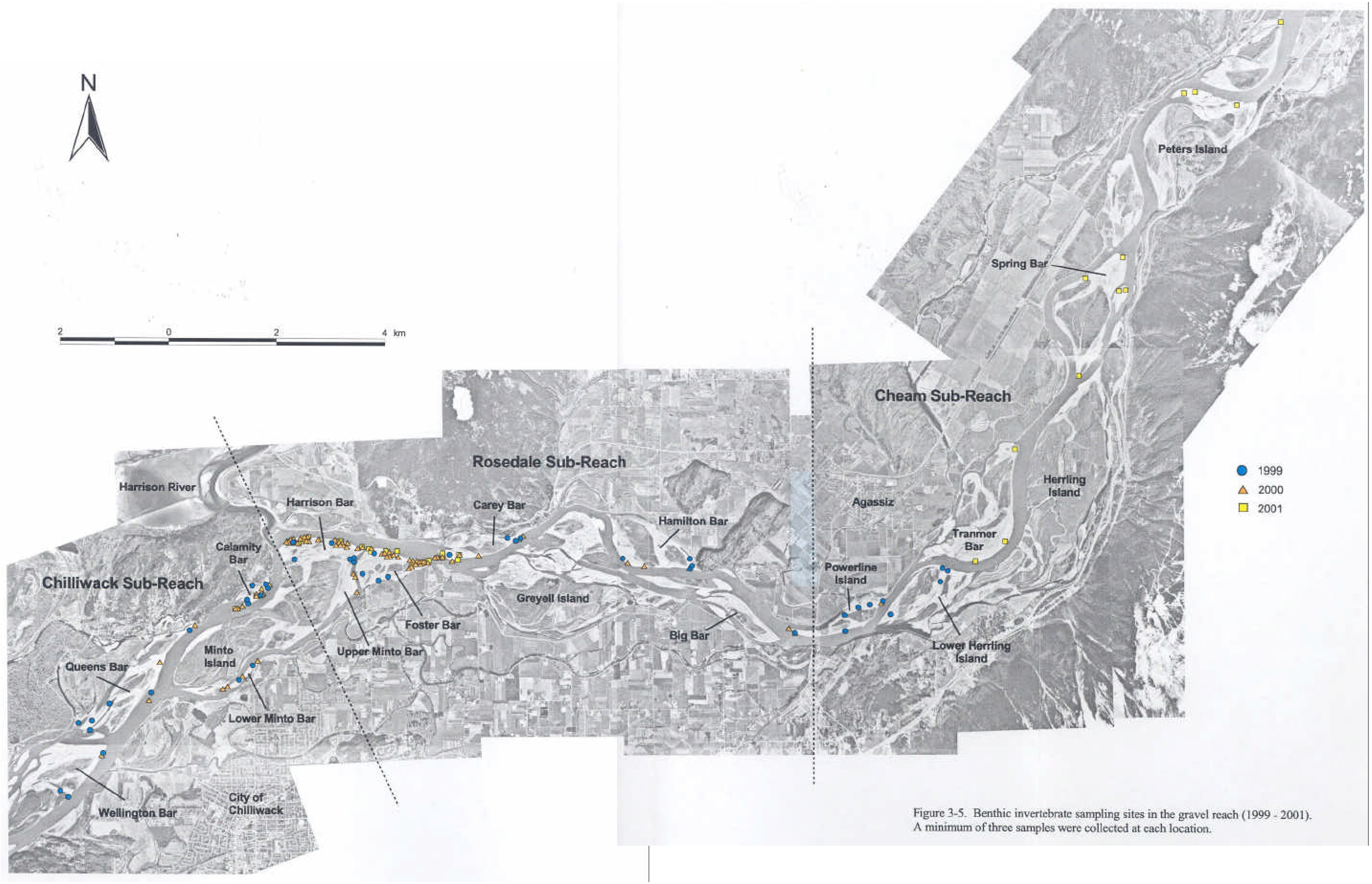


Figure 3-5. Benthic invertebrate sampling sites in the gravel reach (1999 - 2001). A minimum of three samples were collected at each location.

3.3.3 Habitat Surveys

Habitat surveys consisted of making observations and measurements of the physical characteristics within habitat units where fish sampling occurred. This methodology ensured that information on the distribution and life stage of species could be linked with habitat characteristics to elucidate species-specific habitat associations. Water depth and velocity at 6/10 depth from the surface were measured at nine locations within the sampled area using a graduated wading rod and Marsh-McBirney velocity meter. The 9 locations corresponded to the nearshore, midpoint, and most offshore extent of beach seine sampling along 3 transects positioned at the upstream, midpoint, and downstream boundaries of the sampled area. Mean velocity was the average of the nine measurements and maximum velocity was highest of the measurements. The surface sediment was visually classified for degree of embeddedness and percent representation by major grain size classes: sand (<2 mm), gravel (2 – 64 mm), cobble (64 – 128 mm), and large cobble (>128 mm). Embeddedness refers to the degree to which dominant gravel or cobble were embedded in the surrounding matrix material (Sylte and Fischenich 2002). The slope angle of the bar edge was estimated based on offshore width and maximum depth. Water temperature at the mid-point in the seine area was measured using a hand-held thermometer. A detailed sketch map was made of all sampling locations, depicting the local bar configuration, flow patterns, and nearby vegetation.

3.4 Laboratory Methods

3.4.1 Benthic Invertebrates

Preserved invertebrate samples were processed at the DFO Laboratory at Cultus Lake by Ms. Shirley Fuchs. Samples were first wet-sieved (250 µm mesh) and then invertebrates were picked using a dissecting microscope and preserved in 70% isopropanol. The entire contents of each sample were examined for animals; no sub-sampling occurred. Individuals were identified to the lowest possible taxonomic level principally according to Merritt and Cummins (1996). Additional references used were Wiggins (1996), Stewart and Stark (1988), Fitzpatrick Jr. (1983), Bland and Jacques (1978), and Borror and DeLong (1964). Mayflies, stoneflies, and caddisflies were identified to genus; dipterans to either family or subfamily (Chironomidae); beetles and true bugs to family; oligochaetes, leeches, crustaceans, and mites to order; and nematodes to phylum. Taxa were assigned the following functional feeding group (FFG) categories after Merritt and Cummins (1996): collector-gatherers (CG), scrapers (SC), collector-filterers (CF), predators (PR), shredders (SH), and parasites (PA). These categories are based on the general mechanism used by each taxon to feed. A list of taxonomic groups identified in this study and their feeding classification is provided in **Table 3-5**.

Table 3-5. Benthic invertebrate taxa collected in the gravel reach of Fraser River, 1999 to 2001.

ORDER	FAMILY	SUB-FAMILY	GENUS	FFG$^+$	Presence %
Ephemeroptera	Baetidae		*Baetis* sp.	CG/SC	61.0
	Heptageniidae		*Cinygmula* sp.	SC	43.1
			Epeorus sp.	SC	12.4
			Heptagenia sp.	CG/SC	24.5
			Rhithrogena sp.	CG/SC	49.7
	Ephemerellidae		*Drunella* sp.	SC	2.5
			Ephemerella sp.	CG	71.1
	Ameletidae		*Ameletus* sp.	SC	28.3
	Leptophlebiidae		*Paraleptophlebia* sp.	CG	0.4
Plecoptera	Capniidae		*Capnia* sp.	SH	35.3
			Utacapnia sp.	SH	0.4
	Chloroperlidae		*Plumiperla* sp.	PR	1.7
			Sweltsa sp.	PR	0.2
	Nemouridae		*Ostrocerca* sp.	SH	14.9
			Podmosta sp.	SH	0.4
			Zapada sp.	SH	1.0
	Perlidae		*Agnetina* sp.	PR	0.2
			Claassenia sp.	PR	0.8
			Hesperoperla sp.	PR	0.2
	Perlodidae		*Isogenoides* sp.	PR	5.8
			Isoperla sp.	PR	2.3
			Skwala sp.	PR	2.1
	Leuctridae		*Despaxia* sp.	SH	0.4
	Taeniopterygidae		*Taenionema* sp.	SH	31.3
Trichoptera	Brachycentridae		*Brachycentrus* sp.	CF	0.4
	Glossosomatidae		*Glossosoma* sp.	SC	1.2
	Hydroptilidae		*Hydroptila* sp.	SC	0.4
	Limnephilidae		*Onocosmoecus* sp.	SH	0.2
	Lepidostomatidae		*Lepidostoma* sp.	SH	7.2
	Hydropsychidae		*Hydropsyche* sp.	CF	38.4
	Polycentropodidae		*Polycentropus* sp.	PR	0.2

ORDER	FAMILY	SUB-FAMILY	GENUS	FFG[+]	Presence %
	Rhyacophilidae		*Rhyacophila* sp.	PR	1.7
Diptera	Athericidae		*Atherix* sp.	PR	0.2
	Blephariceridae		*Bibiocephala* sp.	SC	0.2
	Ceratopogonidae	*s.f.* Ceratopogoninae.		PR	19.2
	Chironomidae	*s.f.* Orthocladiinae		CG	97.5
		s.f. Tanypodinae		PR	41.7
		s.f. Chironominae		CF	51.1
		s.f. Diamesinae		CG	3.1
	Empididae		*Chelifera* sp.	PR	1.2
			Hemerodromia sp.	PR	35.1
	Simuliidae		*Simulium* sp.	CF	4.5
	Tipulidae		*Antocha* sp.	CG	0.4
			Dicranota sp.	PR	9.5
			Erioptera sp.	CG	0.8
			Hesperconopa sp.	CG	0.4
			Limnophilia sp.	PR	0.4
			Ormosia sp.	CG	0.8
Coleoptera	Dytiscidae		*Brachyvatus* sp.	PR	0.4
	Elmidae		*Heterolimnius* sp.	SC	0.2
	Gyrinidae			PR	0.2
	Hydrochidae		*Hydrochus* sp.	SH	0.6
	Hydrophilidae			PR	0.8
	Corixidae		*Corisella* sp.	PR	0.2
Homoptera			*Tricorixa* sp.	PR	0.8
Lepidoptera				SH	0.2
Nematoda*				PA	47.8
Oligochaeta*	Naididae			SC	66.2
	Tubificidae			SC	30.3
Acarina[+]				PA	27.2

*Presence expressed as percentage of samples in which taxon was present (n = 485)

[+] Functional Feeding Group classification (Merritt and Cummins 1996): CG, collector-gatherer; SC, scraper; SH, shredder; PR, predator; CF, collector-filterer; PA, parasite.

3.5 Data Analysis

Data collection throughout this study was stratified among several tiers of spatial and temporal organization: habitat units, gravel bar units, channel types, sub-reaches, and seasons (spatial units described in detail in Chapter 4). To avoid confounding variance attributed to these factors, a core dataset was designated and included only those observations collected in *autumn (discharge: 1500-5000* m^3 s^{-1}*) from main channel habitat units within the Rosedale sub-reach*. The Rosedale sub-reach, situated between the Harrison River confluence and the Agassiz-Rosedale Bridge, includes the following gravel bar sites: Harrison, Foster, Carey, Hamilton, and Big. The Rosedale sub-reach was designated as the core reach because it best reflects the characteristic morphology of wandering, gravel-bed rivers and has been under the greatest pressure for gravel removal in recent years to mitigate a perceived flood risk. It was the core dataset on which the majority of analyses are based.

Descriptions of data analyses and statistical methods are provided in subsequent chapters where appropriate. Unless otherwise stated, Statistica v.6 (StatSoft, http://www.statsoft.com) was used for all statistical analyses and a significance level of $\alpha = 0.05$ was applied.

Chapter 4. Hierarchical Habitat Classification

4.1 Introduction

In wandering gravel-bed rivers, processes of sediment transport and deposition are agents of frequent morphological change that maintain a high degree of physical complexity in the channel. Complexity occurs over a range of spatial scales and is expressed in the network of channels of varying size that flow around and dissect topographically variable gravel bars and islands. Collectively, these features present organisms with an array of functional opportunities that influence profoundly the structure of biological communities (Hawkins 1981, Poff and Allan 1995, Rempel *et al.* 2000). Variations in water depth, velocity, and substrate texture produce distinctive physical units within a channel, and certain life stages of species and assemblages occupy or avoid particular units depending on their immediate habitat requirements (Inoue and Nunokawa 2002). Natural processes of erosion and deposition produce short-term (i.e., inter-annual) changes in the availability and distribution of individual habitat units, as do seasonal fluctuations in water level. However, in the absence of climatic change or human intervention, water and sediment loads should be consistent over the long-term and the overall variety and frequency of habitat units are likely to remain constant.

A tool that seeks to characterize the complexity and attributes of habitat units in river channels is habitat classification. Classification systems are effective because they simplify a seemingly complex and variable landscape by defining a framework of spatial organization along the channel. For many river classifications, this framework is derived from a geomorphological context and identifies discrete "habitat types" that develop as a result of governing fluvial processes. Habitat types occur ubiquitously along the channel and are characterized by a unique combination of depth, velocity, and substrate conditions. A channel reach consists of many *habitat units*, each classified exclusively as a particular *habitat type*. Habitat structure, as defined by a classification, is linked to ecosystem function by the assumption that the physical structure and organization of stream habitat, combined with the pool of species available for colonization, largely determine the structure and function of stream communities (Frissell *et al.* 1986). Considerable evidence supports this habitat-centered view for stream ecosystems (Southwood 1977, Aadland 1993, Peterson and Rabeni 2001a), to the extent that habitat classification is used to fulfill a large number of functions in ecological research and management (Bisson *et al.* 1981, Hankin and Reeves 1988, Rabeni and Jacobson 1993, Hawkins *et al.* 1993, Maddock 1999, Newson and Newson 2000, Thomson *et al.* 2001, Peterson and Rabeni 2001a).

The physical character and distribution of units within a stream are controlled in part by local features such as riparian vegetation, as well as basin-scale features such as discharge, sediment supply, and process domains (Montgomery 1999). As a result, habitat classifications often are organized in a hierarchy of spatial scales, which provides the basis for discerning local and basin-scale influences on ecosystem structure (Peterson and Rabeni 2001a). Hierarchical systems may be constructed for large physiographic regions by first classifying drainage basins of differing lithology, geomorphology, and land-use history, and progressively identifying smaller-scale classification levels nested within the drainage basin scale. Other hierarchical systems are developed for single watersheds. Regardless of the spatial scope, hierarchical classifications encourage the user to see beyond local site conditions to understand the influence of catchment attributes and large-scale processes on local habitat structure (Kondolf 1995).

Despite the utility of river habitat classification, at least two criticisms have been raised. First, not all classification schemes acknowledge that habitat structure within a channel develops as a consequence of fluvial processes and the interaction with underlying physiography and climate (Poole *et al.* 1997). Ignoring this link promotes a naïve approach to river management because making available a diverse range of habitat types without maintenance of geomorphic processes will not sustain a productive ecosystem for the long-term (Kondolf 1995, Power *et al.* 1996). Classification-based monitoring programs that focus strictly on habitat structure may establish target states for the habitat-unit composition of streams (e.g., threshold pool frequencies, pool-to-riffle ratios), rather than strive to maintain natural fluvial processes such as regimes of water, sediment, and wood delivery to the channel (Poole *et al.* 1997, Jungwirth *et al.* 2002). For these reasons, a process-based approach to habitat classification is preferred (Whiting and Bradley 1993, Harvey *et al.* 1993, Kondolf 1995, Poole *et al.* 1997, Pitlick and Van Steeter 1998).

A second criticism of habitat classification is the possibly simplistic underlying assumption that river habitat structure is organized as discrete physical units. Wright and Li (2002) addressed this issue and presented a conceptual framework for evaluating community composition along an environmental gradient using multivariate techniques. The conceptual framework applies equally well to physical habitat structure, the focus of this chapter. Their choice of multivariate analysis, principally ordination, was to accommodate the numerous variables of interest and the inherent variation within environmental data. They defined three models for habitat structure, the first being the “ideal continuum” where all habitat variables are strongly correlated because they consistently and gradually change along some environmental gradient (i.e., longitudinal gradient). An ordination of sample units would yield only one significant dimension that explains 100% of the variation in the data because only one gradient exists (**Figure 4-1a**).

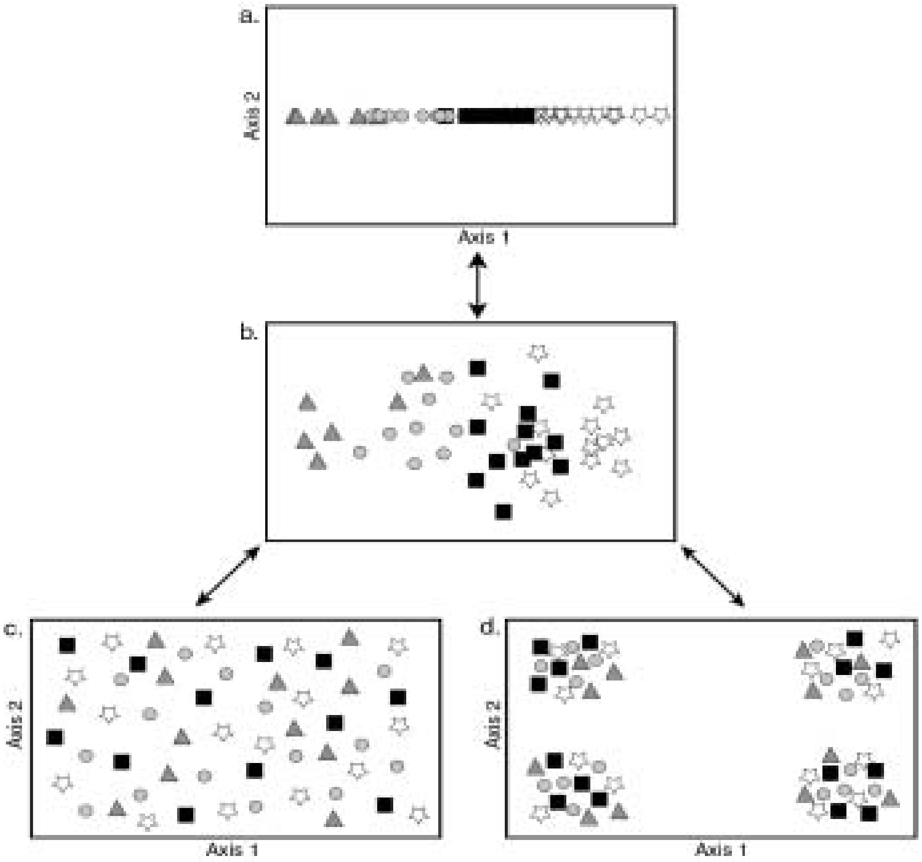


Figure 4-1. Conceptual framework of river habitat structure based on ordination patterns: a) an ideal continuum; c) random; d) patchy. The expected ordination of actual data (b) contains attributes of each conceptual model. Individual symbols represent sites and each shape indicates a state along a longitudinal gradient, e.g., stream order or elevation (from Wright and Li 2002).

The "patchy model" (**Figure 4-1d**) represents conditions where no variables change consistently along the longitudinal gradient. However, discontinuous or smaller-scale gradients exist and there are strong correlations between certain variables, causing habitat types in ordination space to group together in clusters. The sites within a cluster are more similar to each other than to sites in any other cluster. The "random model" (**Figure 4-1c**) implies no strong correlations between or among the variables and no variables being related consistently to the longitudinal gradient. The three models represent useful *a priori* null hypotheses for examining habitat structure although, realistically, the underlying structure of river habitat is likely a combination of these models because both stochastic factors, and large- and small-scale gradients are likely to exist. An ordination of actual habitat data is expected to include attributes of each of the models and allow examination of the degree to which habitat structure differs from the models (**Figure 4-1b**).

Wright and Li (2002) applied this conceptual framework to several Oregon streams between 2 m and 11 m in wetted channel width and representing a gradient of stream order (1st- through 5th-order). Habitat types for streams of this size have received considerable research attention and are well defined (Bisson *et al.* 1981, Frissell *et al.* 1986, Kondolf 1995). In contrast, the types of aquatic habitats in large, gravel-bed rivers and their extent and distribution have been inadequately described. The species assemblages of fish and invertebrates large river habitats support have remained equally uncertain. Classifications for small streams may not apply directly to large rivers because of the significant difference in channel size, influence of riparian vegetation, and major differences in physical conditions and processes spanning the channel.

4.1.1 Objectives

This chapter presents a hierarchical habitat classification for a large, gravel-bed river that is derived from a geomorphological understanding of the physical processes involved in habitat development. The classification consists of three levels (described in detail in Section 4.2): mnorphologically based shore-zone habitat units (Level 3) spatially nested within gravel bar units and associated channel types (Level 2), which are nested within sub-reaches (Level 1) of the gravel reach of Fraser River. The characterization of Level 3 habitat units is a major focus of this research and is presented in a spatially hierarchical context because of the expectation that large-scale morphological attributes and environmental gradients influence local habitat structure (Montgomery 1999).

The objectives of this chapter are, first, to present the hierarchical habitat classification and, second, to characterize the habitat structure of the gravel reach at the scale of Level 3 habitat units (10^1-10^2 m), which are intended to be ecologically relevant to juvenile fish. Three hypotheses were evaluated to meet these objectives: 1) habitat types are physically distinct with respect to velocity, depth, and substrate attributes, and the physical relation among habitat types corresponds to a combination of patchy and gradient-based habitat structure; 2) the characteristics of habitat types are consistent over the spatial scales of Level 1 sub-reaches and Level 2 channel types; and 3) morphological features associated with the fluvial processes of sediment deposition and erosion create these consistently identifiable habitat types. A reduced “hydraulic model” for Level 3 classification is also evaluated for comparison with the morphological model, because of the widely recognized influence of velocity on the distribution of aquatic organisms (e.g., Beecher *et al.* 1995)

Hypotheses were evaluated following several steps. First, the physical distinctiveness of habitat types was determined; this assumption is fundamental to the utility of a habitat classification but is rarely assessed. Modifications to the definitions and groupings of habitat types were then made to maximize the physical distinctiveness between habitats. Second, habitat types were compared among Level 1 sub-reaches to determine the spatial variability in habitat characteristics within the gravel reach and relate potential differences among sub-reaches to local and large-scale gradients. Third, habitat types were compared among Level 2 channel types (main and secondary) in a similar manner. Lastly, the fluvial process-based origin of Level 3 habitat types was examined.

The classification was specifically developed for the gravel reach of Fraser River, British Columbia; however, it is intended to serve as a general model for large gravel-bed rivers in northern temperate regions. Such rivers are common in mountainous and piedmont settings and, as with large rivers in general (Hynes 1989), their ecology has not been studied extensively. For Fraser River, there was a clear need for a habitat classification system for use in habitat inventory and management because pressures on the gravel reach for flood and erosion control have increased significantly in recent years.

4.2 Hierarchical Habitat Classification

The hierarchical habitat classification consists of three levels of increasing detail based on the morphological and sedimentary attributes of the gravel reach of Fraser River. The two coarsest levels of the classification, sub-reaches and gravel bar units, were introduced previously (Church *et al.* 2000) and descriptions of each have been expanded upon herein. The development of Level 3 habitat units has been an iterative process and represents a major contribution of this thesis. A schematic representing the original hierarchical classification structure, as it was formulated at the beginning of the study, is presented in **Figure 4-2**.

At the highest level, the river is divided into five sub-reaches that differ with respect to channel gradient, sinuosity, and sediment load. As a consequence, the morphology and gradational tendency of each sub-reach differs. Sub-reach breaks are imposed by channel constrictions, river valley geometry, and tributary inputs. Within each sub-reach, gravel material accumulates where the hydraulic conditions are favourable. These conditions repeat themselves along the river to form gravel bars, which are ubiquitous features of all gravel-bed channels.

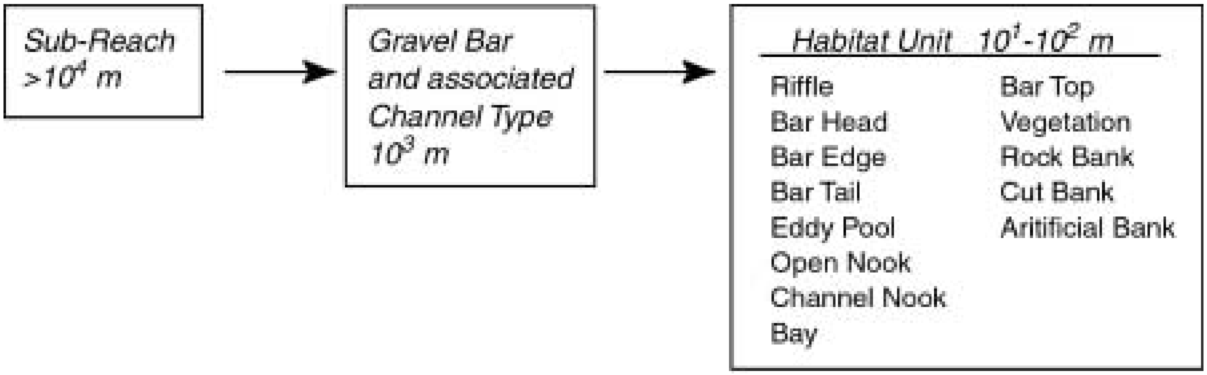


Figure 4-2. The preliminary hierarchical habitat classification structure for the gravel reach of Fraser River and approximate spatial scale of each level.

Gravel bars are nested within sub-reaches and represent the intermediate level of physical organization within the gravel reach. Gravel bars often are vegetated over areas of highest elevation, reflecting their longevity and the role of island development in promoting bar stability along the river. Associated with gravel bars are habitat units, which represent the finest level of the classification. Habitat units are discrete units of relatively homogeneous morphological and physical character that occur around the perimeter of gravel bars and along river banks. Habitat units are classified according to one of thirteen habitat types and membership is determined on the basis of morphological identity and physical character.

4.2.1 Level One – Sub-Reaches

Trends in channel gradient (**Figure 4-3**), riverbed sediment size, and sediment transport regime have been collated to discriminate five sub-reaches of distinct morphological character within the gravel reach of Fraser River (**Table 4-1**, **Figure 4-4**). Sedimentation patterns and the gradational tendency of each sub-reach contribute to a distinctive channel form and determine the physical complexity of each sub-reach. These differences in channel form are expected to lend Level 2 gravel bar units within each sub-reach a particular morphology as well as produce a unique distribution of Level 3 habitat units (though not individually unique habitat types). Together, these factors probably influence the fish assemblages dominantly found in each sub-reach. Sub-reach characteristics will remain unchanged for many decades as a consequence of the large volume of stored sediment that would have to be moved to produce a morphological change; hence, sub-reaches are suitable for strategic management planning within the gravel reach.

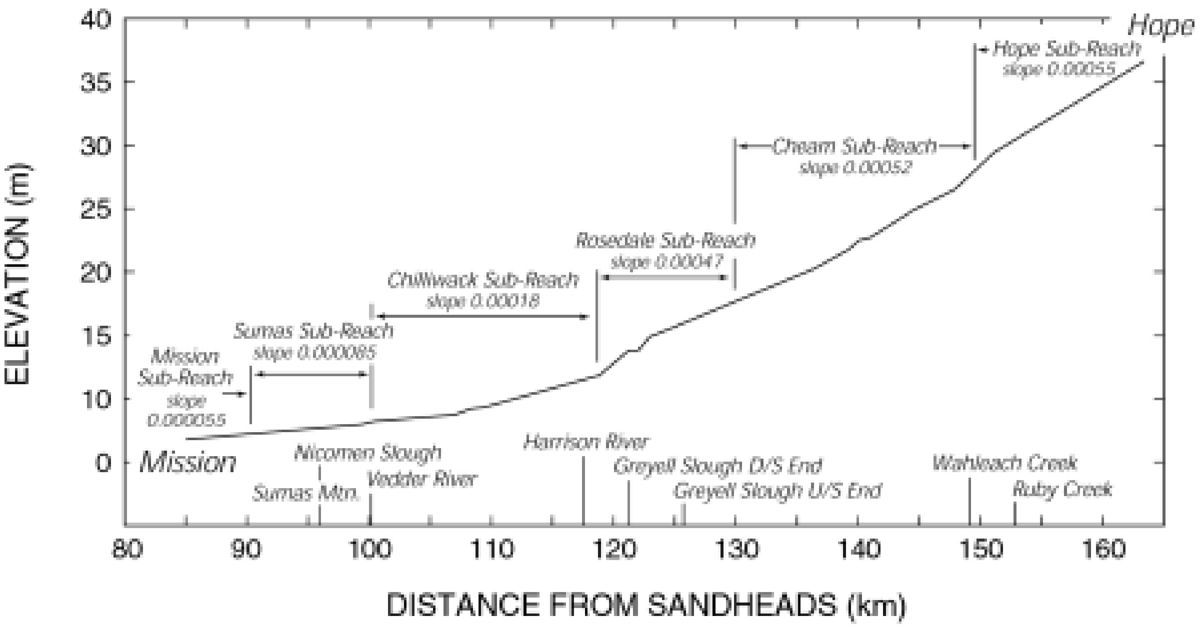


Figure 4-3. Water surface profile of the gravel reach (1972), Mission to Hope, Fraser River (McLean *et al.* 1999). The boundaries of sub-reaches are indicated.

Hope is the most upstream sub-reach and extends from the town of Hope to Wahleach Creek. River valley walls and debris from ancient landslides largely confine the sub-reach. As a consequence, the river flows in a nearly straight, single-thread channel that is essentially stable. Much of the gravel load delivered from Fraser Canyon and Coquihalla River is transported through this sub-reach and only one substantial sedimentary accumulation occurs at the town of Hope. Here, the river negotiates a 130^o bend that contributes significant flow resistance and causes slackening of the current. The locally complex morphology around the bend represents the most diverse habitat in this sub-reach. Downstream, there is a sequence of alternating lateral bars with a spacing of approximately 6.3w_s (surface width of the channel), which is the classical riffle-pool spacing for large gravel-bed rivers (Leopold *et al.* 1964).

Table 4-1. Level One sub-reaches of the hierarchical habitat classification for Fraser River gravel reach.

Sub-Reach Name	Downstream Boundary	River km Limits	Mean Gradient	Mean Grain Size (mm)	Discharge at MAF ($m^3 s^{-1}$)*	Mean Gravel Transport ($m^3 a^{-1}$)+	Gradation Tendency	Major Morphological Features
Hope	Wahleach Creek	149-165	0.00055		8766	215 000	stable	single-thread cobble and gravel channel with stable lateral bars
Cheam[**]	Agassiz Bridge	130-149	0.00052	50	8766	215 000	mildly degrading	large, mature islands with surrounding bars; single dominant channel and major secondary channels
Rosedale[**]	Harrison River	118-130	0.00047	40	8766	95 000	strongly aggrading	multi-thread gravel channel with large island-bar complexes; laterally unstable
Chilliwack[**]	Vedder River	100-118	0.00018	26	9790	15 000	mildly aggrading	multi-thread gravel channel with diagonally extending bars and subordinate islands
Sumas	Matsqui Bend	89-100	0.000085	16 - sand	9790	0	degrading	single-thread, gravel-sand transition; submerged bars

* Based on gauges at Hope (first three reaches) and Mission (last two reaches). MAF = mean annual flood.

\+ Transport is averaged for the period 1952-1999, estimated at the downstream end of the reach (Church *et al.* 2001).

** Sub-reaches within which data collection occurred.

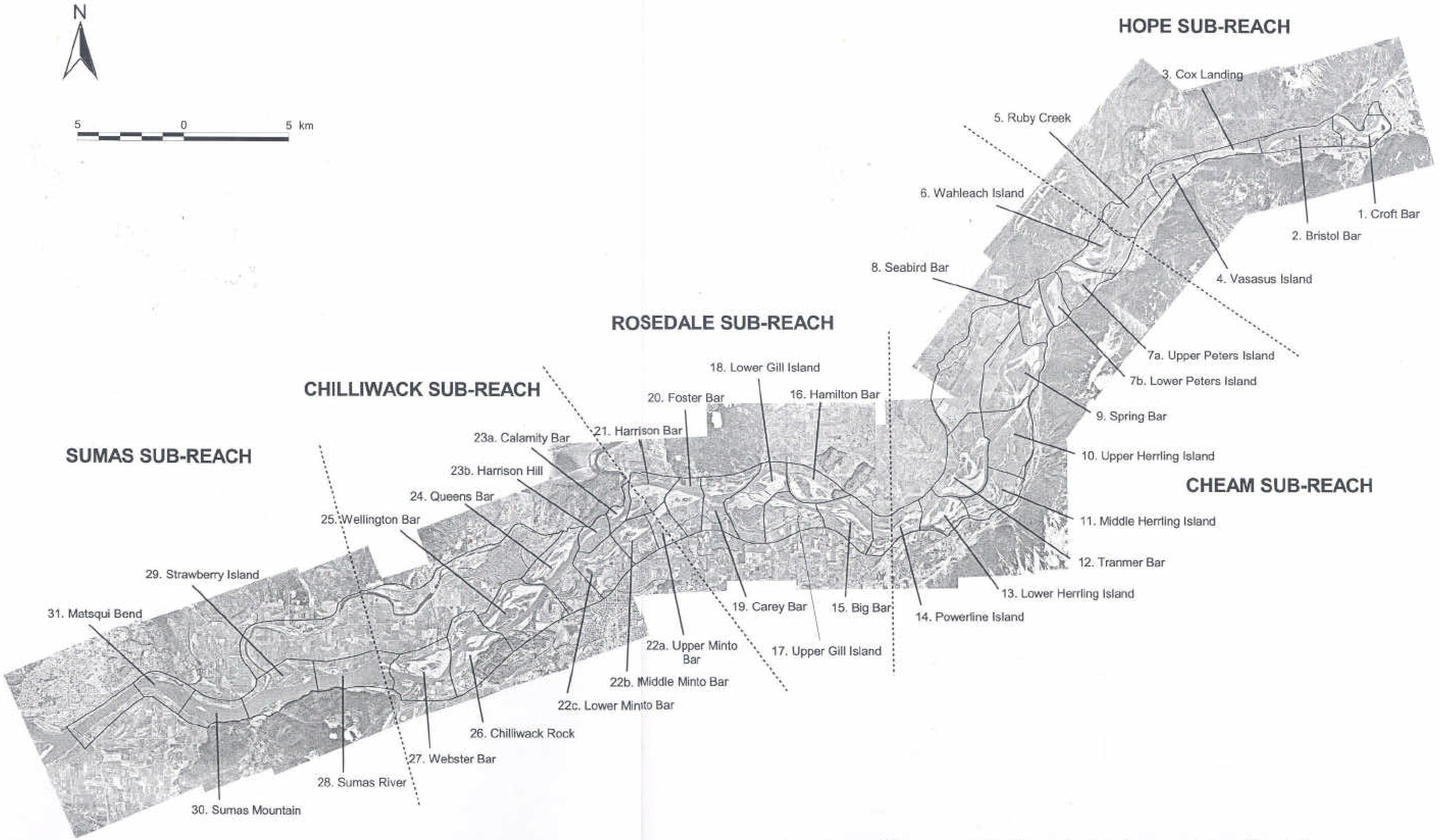


Figure 4-4. Fraser River gravel reach classified according to 5 sub-reaches and 31 gravel bar units that correspond to Level 1 and Level 2 of the hierarchical habitat classification, respectively.

The river ceases to be confined in the *Cheam* sub-reach, which extends downstream from Wahleach Creek to the Agassiz Bridge. However, gradient remains steep and bed material in transport generally moves through the sub-reach. The channel maintains a semi-regular sinuosity as it flows around a series of mature islands, on top of and behind which there is a modest length of perennial and seasonal secondary channels. These channels likely constitute important rearing and refuge areas for fish during spring and summer months (Zallen and Boyd 1986), although some have been isolated or are regulated at the entrance (e.g., Peters Island, Maria Slough). Overall, this sub-reach is mildly degrading along its length, with localized aggradation on lower Tranmer Bar and lower Herrling Island, in part because of the backwater effect from severe natural channel constriction at the Agassiz Bridge.

The *Rosedale* sub-reach extends from the channel constriction at the Agassiz Bridge to the mouth of Harrison River. This sub-reach has been a zone of substantial aggradation over the past 50 years. Major deposition is presently concentrated immediately upstream of the Harrison River confluence where the main channel is forced to turn sharply against Calamity Rock. Flow resistance due to the bend, coupled with the backwater effect created by Harrison River flow, is causing significant sediment deposition upstream. Bar and island complexes with secondary channels occur throughout the sub-reach and a substantial number of sloughs intersect the south bank floodplain, although many have been isolated or are regulated by head gates at the main dyke (Ellis *et al.* 2004). Overall, the sub-reach presents a diverse range of habitat opportunities that change and are renewed frequently.

Channel gradient is substantially reduced in the *Chilliwack* sub-reach, which extends downstream to the confluence of Vedder River. Gravel bars built around island cores dominate the sub-reach; however, islands are smaller, more fragmented, and relatively less common than upstream. The exception is Minto Island, which is bounded by Minto Channel along its south bank. Minto Island developed about 35 years ago by re-alignment of the main channel past Harrison River mouth and the coalescence of several smaller islands (Church and Weatherly 1998). Aggradation in this sub-reach is mild and has been partially offset by major gravel removals from Minto Channel (Weatherly and Church 1999). The sediment is finer in size than in upstream sub-reaches and the bar gravels are relatively less stable. There are major secondary channels on the back sides of bars and many sloughs intersect the floodplain, although some are isolated from the river at their upstream end by channel works (e.g., Shefford Slough).

The most downstream sub-reach, *Sumas*, extends from the Sumas River confluence to Matsqui Bend and channel sediment is transitional from gravel to sand. Bars are submerged at most flows, producing a single-thread channel mostly confined within dykes or by Sumas Mountain. The sub-reach is much less varied than those upstream, with little morphological complexity except around Strawberry Island and the entrances to Nicomen and Hatzic sloughs.

4.2.2 Level Two – Gravel Bar Units

Within each sub-reach, the river is organized into a sequence of gravel bar units (**Table 4-1**). Such units are characteristic of the natural organization of all gravel-transporting channels (Knighton 1998), and correspond with the characteristic step-length for gravel transport in Fraser River. The foundation of a gravel bar unit is a cross-over riffle, with a superimposed gravel bar (possibly vegetated), and adjacent downstream pool (**Figure 4-5**). However, some units are more complex. More than one gravel bar unit may occur on a single, extended riffle and sometimes a riffle is entirely coincident with the bar and is not separately identifiable. Multi-unit complexes with a long history of development are divided for practical purposes (e.g., Herrling Island), although their division is somewhat arbitrary. In addition, the deep-water boundaries between adjacent units are arbitrarily drawn. Exact boundaries, which would be defined by the thalweg, shift from year to year, making them somewhat variable in any case.

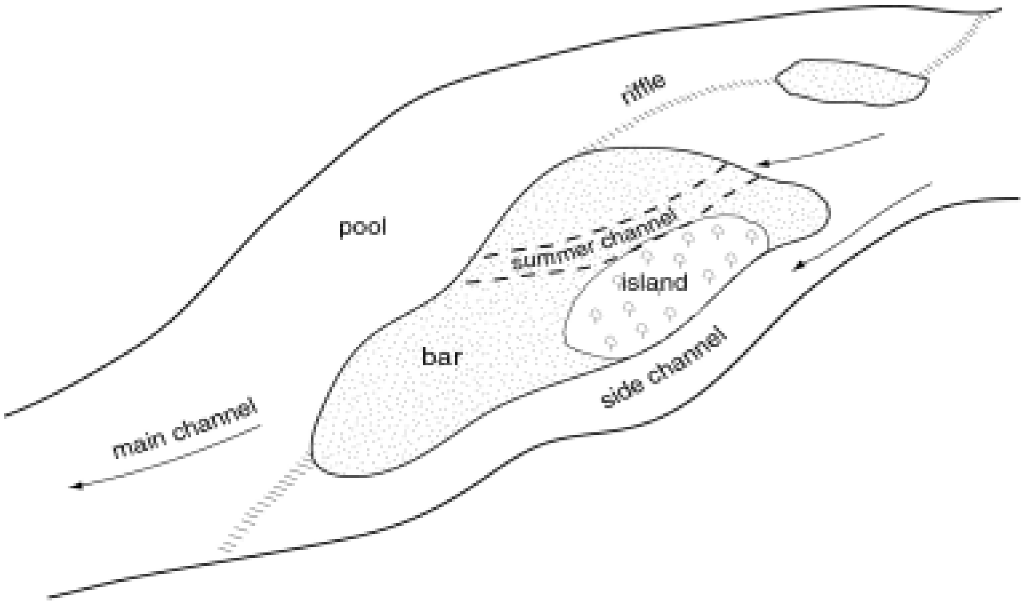


Figure 4-5. Schematic of a basic gravel bar unit and associated channel types in the gravel reach of Fraser River.

Many sedimentary features are associated with gravel bars, including gravel lobes, gravel sheets, and chute-lobe couplets (described in Chapter 2). These features are replicated from bar unit to unit, only changing their relative frequency and physical character among sub-reaches. Fluvial processes modify gravel bars in most years and, occasionally, an avulsion within the channel zone causes a substantial reorganization of bars and secondary channels locally. Hence, gravel bar units are somewhat changeable but major bars remain identifiable for many decades. The size of gravel bar units, and the fact that they are the largest units within which the full range of local habitats may be found, makes them suitable for operational management along the river and appropriate as planning units for scientific studies of river sedimentation and ecology.

Bar units average 700 metres in width (**Table 4-2**), where bar width is the area of unvegetated channel bed that is seasonally exposed at low flow in winter and constitutes intermediate and shallow water habitat during higher flows. This width is about 1.4 times the apparent equilibrium width of the principal channel (Church *et al.* 2000). The number of bar units classified within the gravel reach was 31, for an average bar unit length of 2.6 km (**Table 4-2**).

Associated with gravel bar units are three channel types (**Table 4-3**), which scale in typical length to the bar units. Generalizations can be made for each channel type about the size of bed sediment, the frequency of sediment transport, and flow conveyance. The *main* channel includes the thalweg and is the path along which the majority of flow is conveyed. Secondary channels include side and summer channels, which differ mostly according to size and the annual period of flow conveyance. *Side* channels typically are perennial, but the annual duration of flow conveyance depends on the degree of bar attachment to the main bank. *Summer* channels dissect the tops of gravel bars, thus being influenced by the sedimentation history of the bar top, and convey water at high flow (**Figure 4-6**). The amount of main and side channel area associated with each gravel bar unit is generally similar, however, the number of summer channels tends to vary between bars.

Bar units are numbered in downstream order; units 1 through 5 occur in the Hope sub-reach. *Croft Bar* (Unit 1) includes substantial bar areas around Hope Bend and represents the most complex morphology and most varied habitat opportunities of all units in this sub-reach. *Bristol Bar* (2) and *Cox Landing* (3) are each simple units with right diagonal riffles and limited sediment accumulation in the form of lateral bars. *Vasasus Island* (4) includes a right-bank island and small back channel immediately upstream from Ruby Creek confluence, from which a submerged diagonal riffle is anchored and extends through Unit 5, *Ruby Creek*. Within the Hope sub-reach, 64% of the channel area is permanently wetted at low flow.

Table 4-2. Attributes of Level Two gravel bar units based on low flow, 700 $m^3 s^{-1}$.

Bar Unit No.	**Bar Unit Name**	**Water Surface (m^2)***	**Bar Surface (m^2)$^{+}$**	**Island Surface (m^2)^{o}**	**Unit Length (m)**	**Unit Width (m)**
1	Croft Bar	698301	1216698	196548	2731	701
2	Bristol Bar	1054449	314670	-	3680	372
3	Cox Landing	688058	304380	-	3176	312
4	Vasasus Island	590340	305572	61122	2413	371
5	Ruby Creek	1093709	217354	19682	2988	439
6	Wahleach Island	761153	1176447	529708	2091	927
7a	Upper Peters Island	433614	1110075	663860	2136	723
7b	Lower Peters Island	667700	851366	470134	2120	717
8	Seabird Bar	414418	1117190	359624	1835	835
9	Spring Bar	1354470	1579372	1514575	3990	735
10	Upper Herrling Island	696436	697629	2130024	1819	766
11	Middle Herrling Island	886998	1231391	3746925	4352	487
12	Tranmer Bar	971651	2226377	878789	4571	700
13	Lower Herrling Island	631003	1626305	521103	2561	881
14	Powerline Island	668422	770153	206388	2354	611
15	Big Bar	683517	1430895	334585	2614	809
16	Hamilton Bar	405184	1073095	567353	2774	533
17	Upper Gill Island	89170	1338319	1521870	2513	568
18	Lower Gill Island	544912	2311756	1934397	2212	1292
19	Carey Bar	468651	790289	276078	2082	605
20	Foster Bar	528684	804594	130408	1566	851
21	Harrison Bar	508472	1204031	970942	2144	799
22a	Upper Minto Bar	146620	304781	182866	1173	385
22b	Middle Minto Bar	283746	452814	916162	1843	400
22c	Lower Minto Bar	411264	395031	725566	2435	331
23a	Calamity Bar	571485	271487	184240	2157	391
23b	Harrison Hill	553518	223374	885175	1765	440
24	Queens Bar	1213943	1461902	542619	2652	1009
25	Wellington Bar	1822592	2213577	1012307	3273	1233
26	Chilliwack Rock	1558072	1981732	668485	2718	1303
27	Webster Bar	1378493	2414917	226975	3110	1220
28	Sumas River	2692033	473280	-	3504	903
29	Strawberry Island	2358156	492996	1489512	3677	775
30	Sumas Mountain	2052029	84635	-	3115	686
31	Matsqui Bend	1213820	597304	-	2662	680

*Water surface represents the main channel that is permanently wetted.

$^{+}$ Bar surface is the area of unvegetated, seasonally exposed channel bed.

^{o} Island surface is the vegetated area within the channel zone that remains emergent except in high freshets.

Table 4-3. Channel types associated with Level Two bar units in the gravel reach of Fraser River.

Channel Type	Description
Main	Channel conveys flow year-round and includes the thalweg. Bed material consists mostly of clean gravels with a low proportion of fine sediment. The surface material is subject to bed load transport during freshet.
Side	Channel conveys flow during freshet but may have little or no flow during winter. Wetted habitats at the lower end of the channel persist year-round. Orientation is usually parallel to the main channel. Bed material contains a low to moderate amount of fine sediment (i.e., sandy gravels) at the upstream end and a moderate to high amount at the downstream end (i.e., blanket sands). Minor bed load transport during freshet.
Summer	Channel is seasonally inundated during freshet only and is often oriented diagonal to the main channel and intersecting the bar top. Bed surface elevation is high relative to the main channel and the bed material contains a high proportion of fine sediment (i.e., sandy gravels and blanket sands). Fine gravel may be transported; heavy sand load.

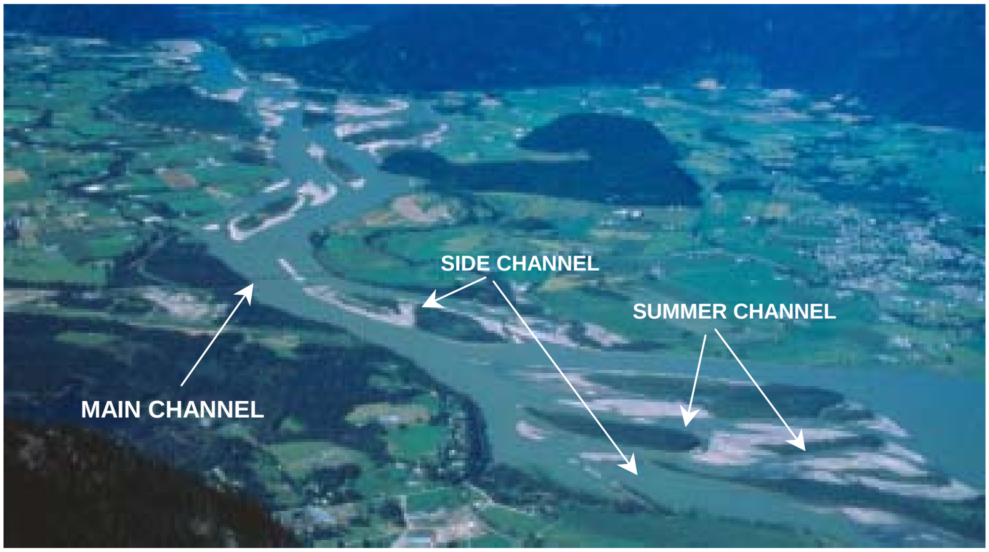


Figure 4-6. Channel types in the gravel reach of Fraser River (photo courtesy of A. Zimmerman). Flow direction is from bottom-right to top-left of the photo.

Bar units 6 through 14 occur within the Cheam sub-reach, which is characterized by large, mature islands and a dominant main channel. *Wahleach Island* (Unit 6), *Seabird Bar* (8), and *Tranmer Bar* (12) are each bar-island complexes situated on the right bank that offer extensive habitat opportunities at higher flow in spring and summer. *Peters Island* (7) is divided into upper and lower sub-units because of the overall size and complexity of the unit; however, it presents the same basic morphological characteristics as other units in the sub-reach. *Spring Bar* (9) is a medial island-bar complex with deep water on all sides on an extended riffle that connects Seabird Bar with Herrling Island. Herrling Island is sub-divided into *Upper* (10), *Middle* (11) and *Lower Herrling Island* (13) based on differing morphology and stages of development. Cheam View has a lower density of back channels than Middle Herrling Island, which presents a stable complex of mature islands and perennial back channels and offers high quality habitat opportunities. Lower Herrling Island is a large complex of bars and young islands that have undergone substantial recent aggradation. *Powerline Island* (14) straddles the boundary between Cheam and Rosedale sub-reaches and the secondary channel along its right bank has deposited substantial sediment in past years. Overall, Cheam sub-reach has a high incidence of side and back channels, and 38% of the channel area is permanently wetted.

The Rosedale sub-reach includes bar units 15 through 21, and is similar in morphology to the Cheam sub-reach except for a greater degree of lateral instability. *Big Bar* (Unit 15) lies parallel to a long, right-diagonal riffle that extends downstream to Hamilton Bar. The main channel flows to the north of Big Bar, but a perennial side channel follows the left bank. *Hamilton Bar* (16) and *Upper Gill Island* (17) are situated opposite one another and the opposing position of each unit emphasizes the complexity of this area and the wide variety of habitat opportunities available. The island extending off Greyell Slough (locally referred to as Jesperson side channel) is sub-divided between *Upper* (17) and *Lower Gill Island* (18) because of the large and highly complex area of bars and secondary channels. Within these units, significant lateral instability and changes in channel alignment have occurred in recent years. *Carey Bar* (19) is a lateral bar on the north bank that consists of young islands and open bar with good seasonal habitat opportunities. *Foster* (20) and *Harrison* (21) bars are relatively simple lateral bars with a mixture of both young and mature island areas; both have been sites of major aggradation in recent decades. Only 27% of the channel area is wetted at low flow within the Rosedale sub-reach, reflecting the large sediment accumulations in the Harrison-Foster and Gill Island units.

The mildly aggrading Chilliwack sub-reach consists of bar units 22 through 27. *Unit 22* is divided into three sub-units corresponding to *Upper* (22a), *Middle* (22b) and *Lower Minto Bar* (22c), and determined by major riffle positions along Minto Channel. Minto Channel is the most significant secondary channel in the gravel reach, draining a major portion of summer flow but conveying only minor flow in winter because of sedimentation at the channel entrance. The main channel is severely constricted where it flows around Harrison Knob and *Calamity Bar* (Unit 23), and two minor riffles associated with left and right bank lateral bars have developed. *Queens Bar* (24) and *Wellington Bar* (25) together represent a long diagonal riffle extending from Harrison Hill, but a reverse-oriented riffle has more recently developed from the tail of Minto Island across lower Queens Bar and upper Wellington Bar. Each of these bars consists of mature island cores intersected by secondary channels with rich and diverse habitat opportunities. Wellington Bar is provincially designated an Ecological Reserve (ER#76). *Chilliwack Rock* (26), opposite Yaalstrick Island, is a zone of active sedimentation on an extended left-diagonal riffle over which the river divides into several shallow channels. The channels are notably unstable and prone to sedimentation. *Webster Bar* (27) represents the most distal gravel accumulation that remains exposed at low flow. Excluding Minto Channel, 45% of channel area in the Chilliwack sub-reach is wetted at low flow, indicating a large area of shallow-water habitat is available on a seasonal basis during freshet.

The most downstream sub-reach, Sumas, includes bar units 28 through 31, and channel morphology is consistently simple. The last notable gravel bar area is opposite the *Sumas River* confluence (Unit 28), and some complexity occurs at the mouth of Nicomen Slough and the channels around *Strawberry Island* (29). Strawberry Island is the only significant island in this sub-reach and originated as a bar that coalesced with the floodplain due to siltation of the back channel. The gravel-sand transition occurs in the vicinity of *Sumas Mountain* (30) and *Matsqui Bend* (31). The most notable channel features in these units are a large bar composed mainly of sand on the right bank upstream of Matsqui Bend, and a medial bar with a gravel veneer on a long riffle. Approximately 84% of channel area in the Sumas sub-reach is wetted at low flow, the highest of all sub-reaches.

4.2.3 Level Three – Habitat Units

Associated with the perimeter of gravel bars and river banks are habitat units, which represent the finest level of the hierarchical classification. *Habitat units* are discrete areas of relatively homogeneous physical character that were classified in the field as one of ten alluvial *habitat types* or three bank types primarily on the basis of morphological identity (**Table 4-4**). Habitat types are expected to have physically distinct sedimentary and hydraulic characteristics, and the spatial scale of individual units is intended to be most ecologically relevant to aquatic organisms, particularly juvenile fish.

Habitat units occur at all stages of the hydrograph throughout the gravel reach and, unlike sub-reaches and gravel bars, *habitat units shift laterally with changing stage*. Hence, the positions of units are not fixed but rather shift in concert with changing water levels. All habitats are likely to occur at each gravel bar unit and within each of the three channel types (main, side, summer). A schematic of alluvial habitats, thereby associated with the perimeter of gravel bars and developing through sediment transport processes, is presented in **Figure 4-7**. Photographic examples of each are shown in **Figures 4-8** through **4-13**. The remaining three habitats were bank types (**Figure 4-14**); sampling challenges limited the extent of physical and ecological characterization of these bank types. This research was restricted in focus to 7 of the 10 alluvial habitat types because sampling methods for juvenile fish and aquatic insects could be standardized and were highly effective in these habitats throughout the gravel reach. The three excluded alluvial habitat types (riffle, bar top, vegetation) could not be sampled effectively and consistently using a beach seine.

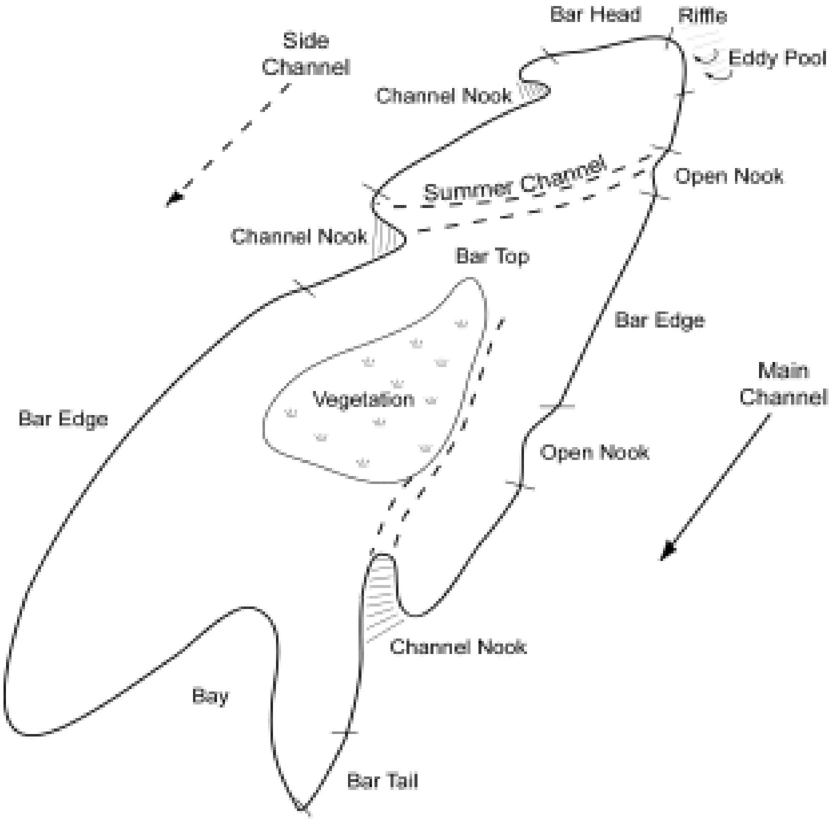


Figure 4-7. Illustration of channel types and Level 3 alluvial habitat types in the gravel reach of Fraser River.

Table 4-4. Preliminary habitat types corresponding to Level Three of the habitat classification for the gravel reach of Fraser River (two-letter abbreviations are given in parentheses).

Habitat Type	Definition
Riffle (RI)	High-gradient area of shallow, fast water flowing over well-sorted substrate that often has granular structures and is stable. The flow is rough. Common at bar heads.
Bar Head (BH)*	Upstream end of a gravel bar. Surface substrate is characteristically coarse and flow velocity is usually high.
Bar Edge (BE)*	Any length of bar edge not occurring at the head or tail of a bar that is oriented parallel to the flow and subject to constant and consistent flow forces. Bank slope is variable and a range of velocities and substrate types is possible. Riparian influence is variable.
Bar Tail (BT)*	Downstream end of a gravel bar, usually with moderate flow velocity. The habitat is often depositional and surface substrate consists of smaller cobbles and gravels.
Eddy Pool (EP)*	Area bounded by fast, rough water that creates a back eddy in the lee of the flow. Common on the inside edge of riffles and at the upstream end of some bar head habitats. Bank slope is invariably steep and the substrate is usually embedded cobble.
Open Nook (ON)*	Shallow indentation along a bar edge of reduced velocity and variable substrate that is openly connected to the channel with no sedimentary barrier (unlike channel nook). An ephemeral habitat that often disappears with a relatively small change in water level.
Channel Nook (CN)*	Dead-end channel or narrow embayment of standing water and concave geometry. Substrate material usually consists of sand/silt and embedded gravel.
Bay (BA)*	Semi-enclosed area with no flow velocity and fine bed material (sand/silt). Occurs on the lee side of large sediment accretions that are deposited in the shape of a crescent-dune.
Bar Top (BP)	Bar top surface inundated only during high flow with reduced velocity and shallow water depth relative to open water and the thalweg. Substrate is variable.
Vegetation (VG)	Area of flooded island or bank vegetation where velocity is reduced and substrate is relatively fine. Submerged only at very high flow.
Cut Bank (CB)	Eroding bank of fine sediment that is steeply sloped or vertical. Dense riparian vegetation is often present. Large woody debris is common and flow is variable.
Rock Bank (RB)	Natural rock bank, possibly with openings and cracks, that is invariably steep. The water is deep immediately offshore and currents are either fast or form a back eddy.
Artificial Bank (AB)	Bank is invariably steep and consists of riprap or rubble rock that may have significant openings within its structure. The water is usually deep and fast immediately offshore, particularly at high flow.

* alluvial habitat types effectively sampled in this study.

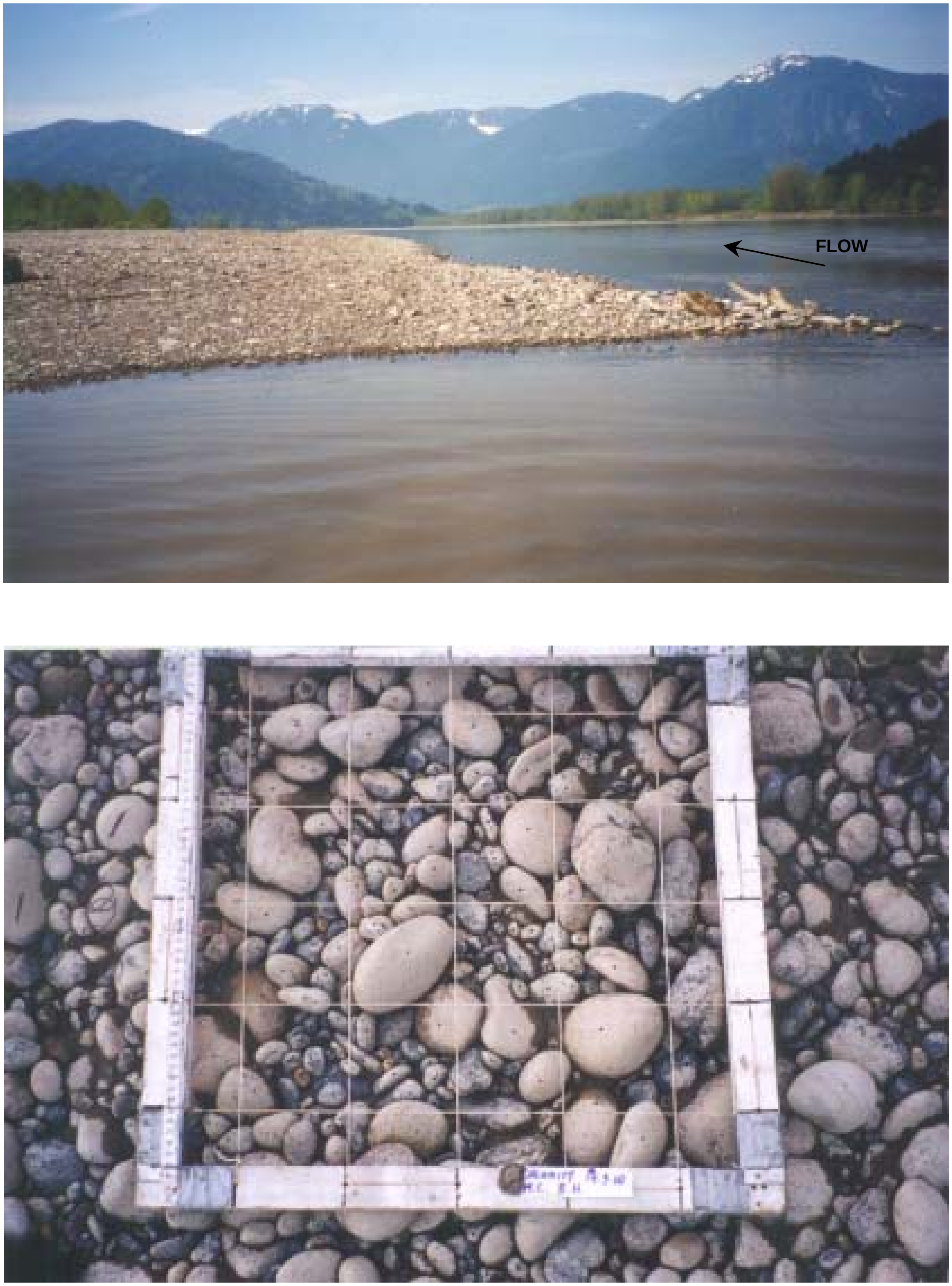


Figure 4-8. Example of a bar head habitat unit (Foster Bar) and characteristic surface sediment (1/2-m quadrat for scale).

Figure 4-9. Example of a bar tail habitat unit (Calamity Bar) and characteristic surface sediment (1/2-m quadrat for scale).

Figure 4-10. Examples of bar edge habitat units of varying bank steepness (flat angle, Carey Bar, upper panel; steep angle, Queens Bar, lower panel). Surface sediment characteristics are variable, consisting of cobble, gravel, and sand.

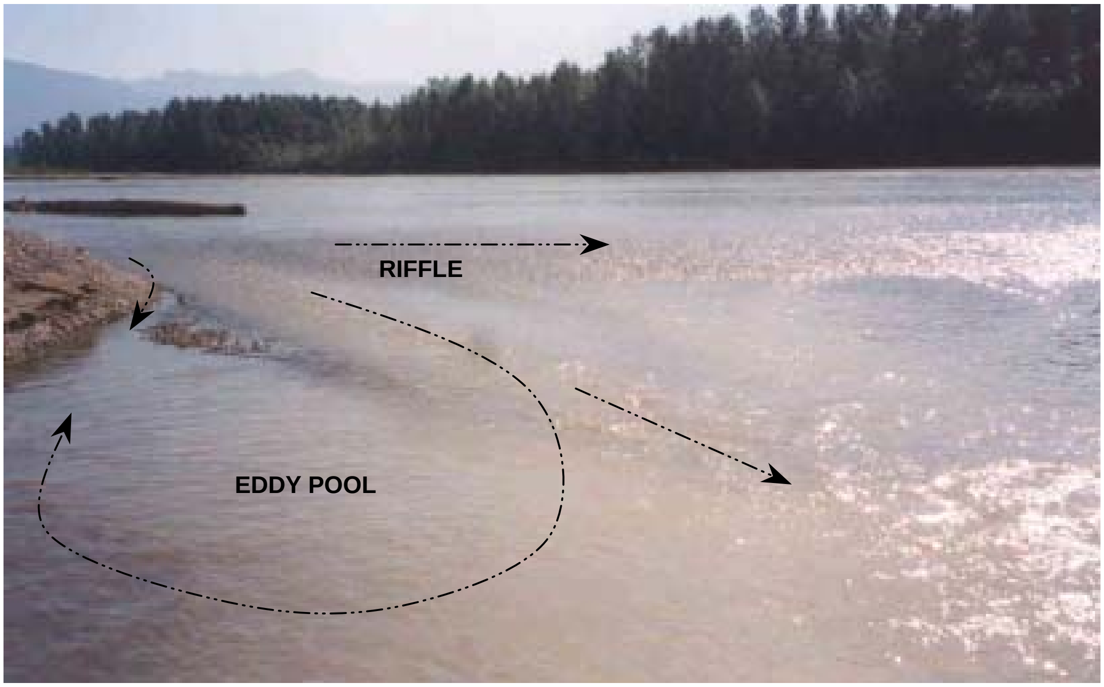


Figure 4-11. Example of an eddy pool habitat unit created in the lee of a riffle (Hamilton Bar) and characteristic eddy pool surface sediment (1/2-m quadrat for scale).

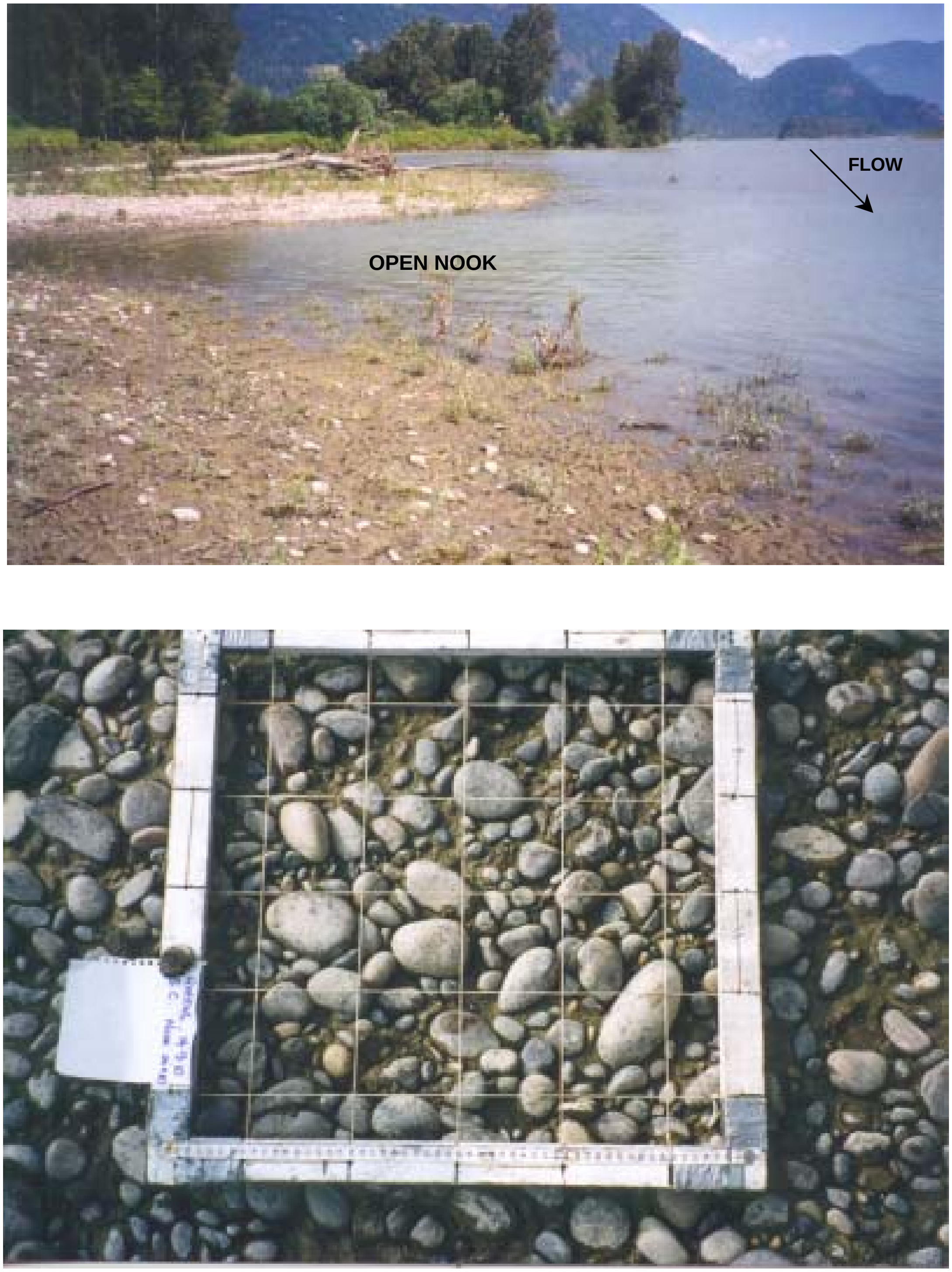


Figure 4-12. Example of an open nook habitat unit (Queens Bar) and characteristic surface sediment (1/2-m quadrat for scale).

Figure 4-13. Examples of a channel nook (Foster Bar, upper panel) and bay (Lower Herrling Bar, lower panel). Surface sediment typically consists of sand and silt or gravel and cobble sediment heavily embedded with sand and silt.

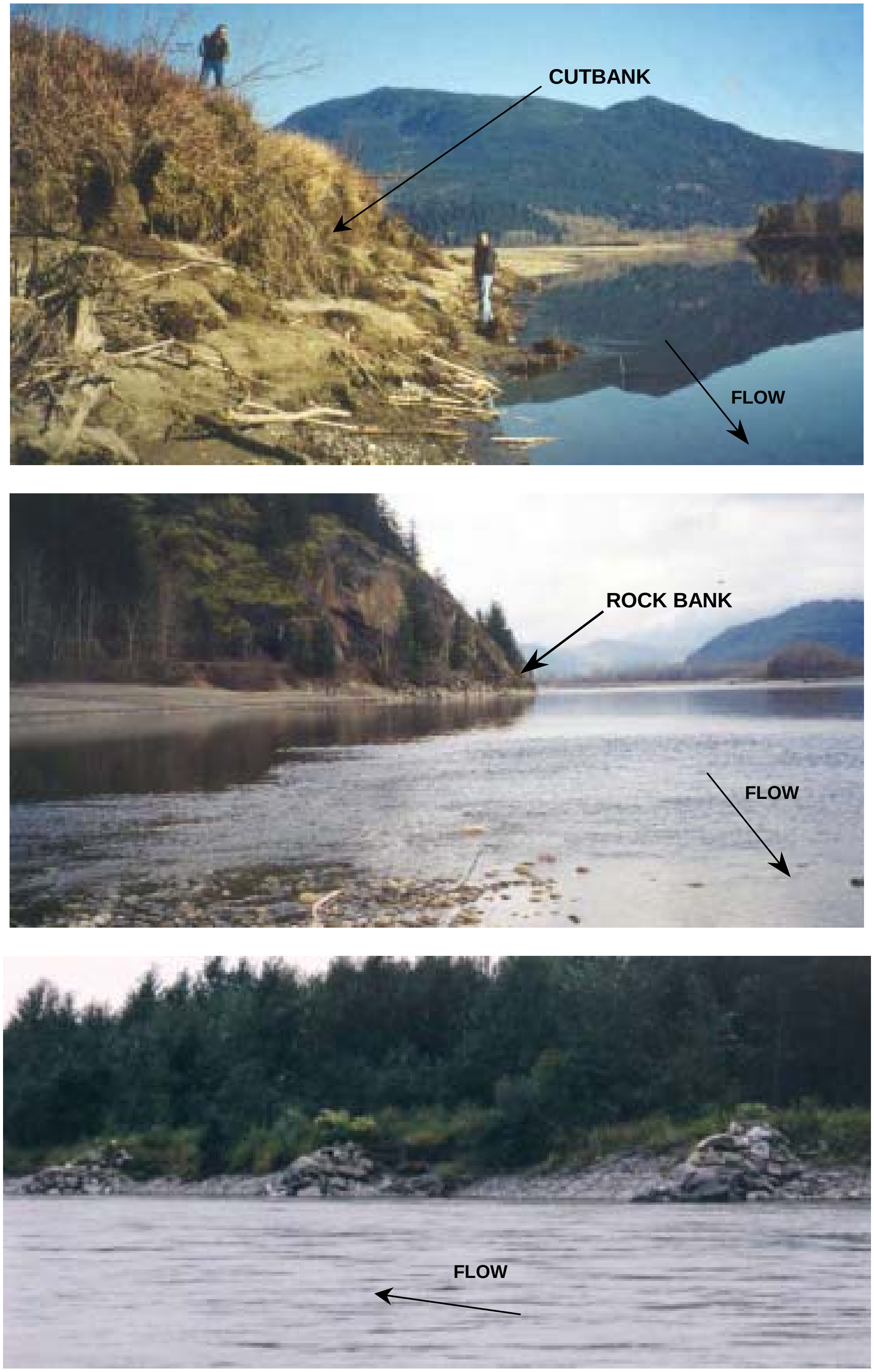


Figure 4-14. Examples of a cutbank at low flow (Lower Minto Bar, upper panel), rock bank (Hamilton Bar, middle panel) and artificial bank with groynes (Spring Bar, lower panel). Surface sediment associated with bank types is variable.

4.3 Data Analysis

Data collection throughout this study was stratified among several tiers of spatial and temporal organization: habitat units, gravel bar units, channel types, sub-reaches, and seasons (defined by river discharge). A core dataset was designated to avoid confounding variance attributed to multiple levels of these factors, and included observations collected in *autumn from main channel habitat units within the Rosedale sub-reach*. Autumn sampling corresponded to flows of 5000 to 1500 $m^3 s^{-1}$ on the declining limb of the hydrograph, typically from early August to late September in each year. It is not believed that habitat characteristics varied significantly over this range of flows compared to differences at high flow in summer and low flow in winter. It was the core dataset on which the majority of analyses were based. The dataset consists of 122 sampling observations made between 1999 and 2001 within habitat units where physical habitat measurements and fish sampling by beach seine occurred.

Discriminant analysis (DA) was used to address research hypotheses. Other analytical methods were considered, including multinomial logistic regression by maximum likelihood estimation and classification and regression trees (CART). Discriminant analysis was chosen because of its parsimonious capacity to identify variables that discriminate among groups (i.e., habitat types) and because of its convenient application as a predictive tool. As well, discriminant analysis has been applied successfully in previous habitat studies. Jowett (1993) classified pool, run, and riffle habitats by DA for a gravel-bed stream in New Zealand with “good” classification success between 50% and 85%. Peterson and Rabeni (2001a) had “fairly high” prediction of similar habitats in a Missouri stream with classification rates between 39% and 100% and averaging 75% to 80%.

Briefly, discriminant analysis seeks to classify data into mutually exclusive groups (i.e., habitat types) on the basis of a set of variables. DA is alternatively referred to as canonical discriminant analysis or canonical variates analysis (CVA) when used to discriminate between more than two groups; however, the term discriminant analysis is most commonly used (Dillon and Goldstein 1984). The key assumption is that all cases can be assigned to only one group in advance, through some means external to the data analyzed (Dillon and Goldstein 1984). In this way, it differs from principal components analysis (PCA), which assumes that any sub-structuring in the data is unknown prior to analysis. Whereas PCA maximizes the total variation explained by each principal component, DA maximizes the among-group variance explained by each canonical variate. As such, it focuses not on the overall variation in the data, but on the extent to which that variation is partitioned among groups to maximize group separation.

The separation of two groups is accomplished by finding a linear combination of the original variables that maximizes the ratio of between-group to within-group variability within a multi-dimensional space defined by the sample attributes (Dillon and Goldstein 1984). In the case of more than two groups (i.e., habitat types), multiple discriminant functions (canonical variates or roots) are produced that are orthogonal to one another and whose contributions to the discrimination between habitat types do not overlap. A discriminant function then becomes a set of coefficients for computing a discriminant score for each case (i.e., sample). The maximum number of functions will equal the number of habitat types minus one, or the number of variables in the analysis, whichever is smaller.

Only those discriminant functions found to be statistically significant by Chi-Square analysis and contributing to the discrimination between habitat types should be interpreted. Those variables with the largest standardized discriminant coefficients for each significant function contribute most to the discrimination between habitat types. Substantive "meaning" for each significant function is derived from the factor structure matrix of correlations between the variables in the model and the discriminant functions. Plotting the case scores for the two most significant functions (referred to as a canonical plot) is a means of assessing how well the functions discriminated between habitats and how dissimilar habitats are from one another. Centroids mark the average score for each habitat type in multivariate space and the distance between group centroids is indicative of habitat type distinctiveness.

A common application of discriminant analysis is the predictive classification of cases, which is accomplished either *post-hoc* with the same data used to derive the discriminant functions (also known as resubstitution), or *a priori* by predicting new cases that were not used in the original analysis (also known as cross-validation). *Post-hoc* classification accuracy always will be higher than *a priori* classification because of the bias associated with classifying cases using a model that has been optimized to deal with the unique characteristics and "noise" in the dataset. *A priori* classification is more rigorous and is carried out in one of two ways, depending on sample size. *Data-splitting* requires a sufficiently large sample population that is split into a "learning sample", used to derive the classification functions, and a "test sample", used to test the discriminant model. The j*ackknife* method is less limited by sample size and systematically excludes one observation at a time, constructing a model with the remaining *n-1* observations, and then predicting the response of each excluded observation using the model. The procedure is repeated *n* times so that each observation, in turn, is excluded in model construction in order to predict its response. Olden *et al.* (2002) recommended the jackknife approach because it produces relatively unbiased estimates of model performance (i.e., classification success of habitat types) and has the advantage of not requiring a large sample size.

The DA model for Fraser River habitat types was constructed as a standard, non-stepwise analysis, due to criticisms that stepwise methods introduce bias to significance tests because, given enough variables, some combination of them will usually produce significant discriminant functions by chance alone (Manly 1986). The following physical variables were included: proportions of gravel and sand/silt, bank angle, mean velocity, and mean depth. The proportion of cobble was excluded from analyses because it is a linear function of the other sediment variables, and therefore violates the assumption of non-multicolinearity. Maximum velocity and depth were excluded as well, because they were highly correlated with mean velocity and depth, respectively. DA assumes that the distribution of each variable is normal, which was tested by plotting the data as histograms and applying the Kolmogorov-Smirnov test for normality. DA also assumes homogeneity of variances, which was evaluated by plotting the within-groups variances against the within-groups means and applying Levene's test. Three variables required transformation based on these analyses: bank angle (logarithm) and the proportions of gravel and sand (arc-sine square root).

The first hypothesis to test was that Level 3 habitat types are physically distinct. This question was addressed using the core dataset (Rosedale sub-reach, main channel habitat units during autumn season) by graphical examination and by discriminant analysis to (1) quantify the degree of physical distinctiveness among habitats, (2) identify the physical factors most responsible for discriminating between habitat types, and (3) characterize the underlying physical gradients that structure fish habitat in the gravel reach. Classification success of the discriminant model was evaluated by the *jackknife* method and calculated as the *weighted* average for units of all habitat types to ensure that classification accuracy was adjudicated based mainly on dominant habitat types. Expecting that longitudinal gradients in substrate size and channel slope may influence the local environmental character of habitat types and therefore confound habitat discrimination, variables were first standardized by transforming case scores within sub-reaches into *z*-scores (mean = 0, standard deviation = 1). In this way, units with a similar morphological expression were compared while controlling for sub-reach variability.

Principal components analysis (PCA) was used as a complementary analysis to DA to determine the primary physical gradient structuring bar edge and bar tail habitat units, which had low classification success. All physical variables (normal-transformed) were included in the analysis and factor loadings on principal component axes were determined by correlation analysis.

The second hypothesis was that habitat types are similar among the three sub-reaches (Cheam, Rosedale, Chilliwack) and three channel types (main, side, summer). This question was addressed by applying classification functions derived from the Rosedale sub-reach to observations from the Cheam and Chilliwack sub-reaches and evaluating the predictive accuracy of habitat classifications (*data splitting* cross-validation). Variables were standardized into *z*-scores within each sub-reach to eliminate possible downstream trends in the local environmental characteristics. Classification functions derived from main channel habitats in the Rosedale sub-reach (autumn only) then were applied to side and summer channel observations (Rosedale sub-reach, autumn only) to determine how accurately the functions characterized habitat types (*data splitting* cross-validation). These analyses were supported by graphical comparisons among groups to explore the spatial integrity of habitat type characteristics. The final hypothesis, that morphological features associated with the fluvial processes of sediment deposition and erosion create consistently identifiable habitat types, was evaluated based on visual observations and photographic interpretation.

4.4 Results

4.4.1 Are Level Three Habitat Types Unique?

A graphical comparison of main channel habitat units in the Rosedale sub-reach (autumn season only) revealed physical differences among most habitat types. Bar head and bar edge units consisted of a higher than average proportion of large cobble and cobble-sized sediment, and had the lowest proportions of sand/silt of all habitat types (**Figure 4-15**). Water velocity in bar head units was highest of all sampled alluvial habitats (**Figure 4-16**). Bar edge units were not readily differentiated from bar head units based on hydraulic or sedimentary characteristics and, consistent with field observations, these habitat types were relatively similar. Bar tails consisted of well-sorted gravel sediment and a low proportion of cobble, and on this basis were differentiated from bar head and bar edge units. Open nook substrate was similar to that of bar tails, but bank angle was lower, as was water depth and velocity. Eddy pools, which morphologically are associated with bar head units, were differentiated based on a steep bank angle and cobble-sized channel substrate embedded with sand/silt. The physical characteristics of channel nooks and bays were highly similar, but distinguishable from all other habitat types based on negligible water velocity, deep water, and predominantly sand/silt substrate.

Also of note from these plots are correlations among several physical variables (**Table 4-5**). Inverse correlations between the proportion of sand/silt and the proportions of both gravel and cobble represented a sedimentary gradient. This gradient also represented a hydraulic gradient in that velocity was positively correlated with gravel and negatively correlated with the proportion of sand/silt, and water depth was positively correlated with sand/silt. Only main channel data from the Rosedale sub-reach were included in this analysis, but results are representative for the entire gravel reach. As a result of significant correlations among variables, only the proportions of gravel and sand/silt, bank angle, mean water velocity, and mean water depth were included in discriminant analyses.

Table 4-5. Correlation matrix of all physical variables measured to characterize main channel habitat units in the Rosedale sub-reach (autumn only).

Variable	**LC**	**CO**	***GR***	***SS***	***BkAng***	***V-Mean***	**V-Max**	***D-Mean***
CO	**0.25**	-	-	-	-	-	-	-
GR	***-0.32***	***-0.46***	-	-	-	-	-	-
SS	*-0.18*	***-0.42***	***-0.54***	-	-	-	-	-
BkAng	*0.17*	*0.23*	***-0.36***	*0.13*	-	-	-	-
V-Mean	*0.07*	*0.14*	***0.33***	***-0.50***	*0.18*	-	-	-
V-Max	0.15	0.16	**0.26**	**-0.45**	**0.24**	**0.93**	-	-
D-Mean	*0.16*	*0.20*	*-0.37*	*0.18*	***0.93***	*0.10*	*0.10*	-
D-Max	0.23	**0.26**	**-0.40**	0.13	**0.91**	0.13	0.19	**0.91**

LC: large cobble; CO: cobble; GR: gravel; SS: sand/silt; BkAng: bank angle; V-Mean: mean velocity; V-Max: maximum velocity; D-Mean: mean depth; D-Max: maximum depth.
Bold-font numbers are significant at the 99% confidence level.
Italicized text indicates the variables included in discriminant analyses.

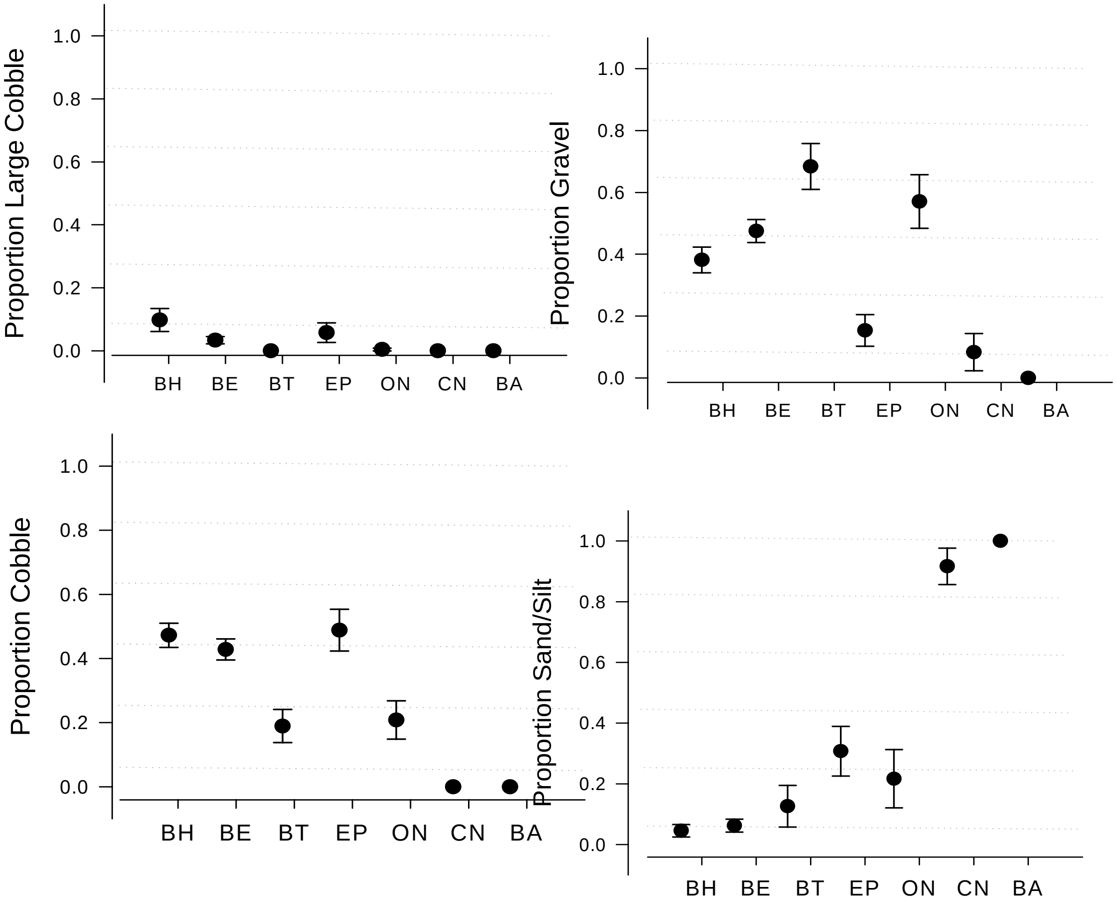


Figure 4-15. Substrate characteristics (mean ± SE) of main channel habitat types in the Rosedale sub-reach of Fraser River. Data were collected during the autumn season (1999-2001). Two-letter abbreviations of habitat types are in **Table 4-4**.

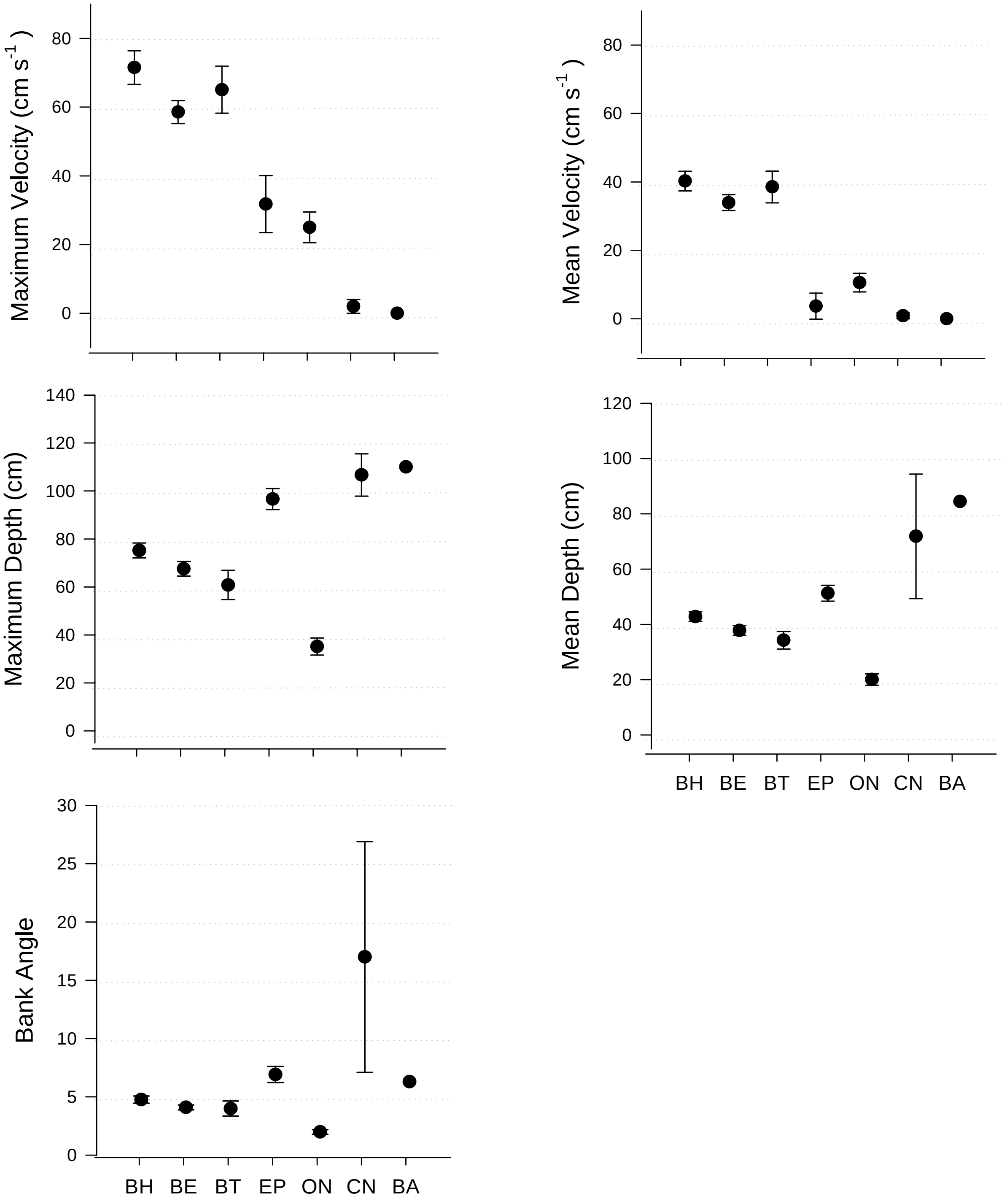


Figure 4-16. Water velocity, depth, and bank slope characteristics (mean ± SE) of main channel habitat types in the Rosedale sub-reach of Fraser River. Data were collected during the autumn season (1999-2001). Two-letter abbreviations of habitat types are in **Table 4-4**.

Discriminant analysis of main channel habitat units in the Rosedale sub-reach discriminated among habitat types based mostly on a hydraulic gradient. Two discriminant functions (canonical roots) were significant by Chi-Square analysis. Root 1 explained 67% of the variance in the data and was correlated with velocity (negative) and bank angle (positive, **Table 4-6**). The proportion of sand/silt was positively correlated with Root 1 as well. Units exposed to the flow (bar head, bar edge, bar tail, open nook) were differentiated along Root 1 from those with minimal flow, deep water, and steep bank angle (eddy pool, channel nook, **Figure 4-17**). However, there was relatively poor discrimination among those habitats exposed to the flow. Root 2 accounted for 20% of the variance and was negatively correlated with gravel and sand/silt, and positively correlated with bank angle and velocity (**Table 4-6**). Bar head, bar edge, and bar tail units were distributed along Root 2 in an order of decreasing velocity and bank angle, but the habitats overlapped considerably in physical character. Open nooks were differentiated along Root 2 from other habitat types exposed to flow based on a high proportion of gravel-sized sediment. Only some bar edge and bar tail units, which characteristically have a high proportion of gravel, overlapped with open nooks in canonical space. Bar edge units showed the highest degree of variability in physical characteristics and overlapped substantially with bar head and bar tail units. Note that ellipses in **Figure 4-17**, and subsequent canonical plots, are not statistically derived but intended to assist in visual interpretation by encompassing the majority of cases belonging to a particular group.

Table 4-6. Factor loadings on each significant canonical root based on correlation analysis for main channel habitat units of the Rosedale sub-reach.

Variable	**Root 1**	**Root 2**
Gravel	-0.40	-0.33
Sand/Silt	0.46	-0.36
Bank Angle	0.50	0.75
Mean Velocity	-0.51	0.68
Mean Depth	0.45	0.65
Variance Explained (%)	67	20

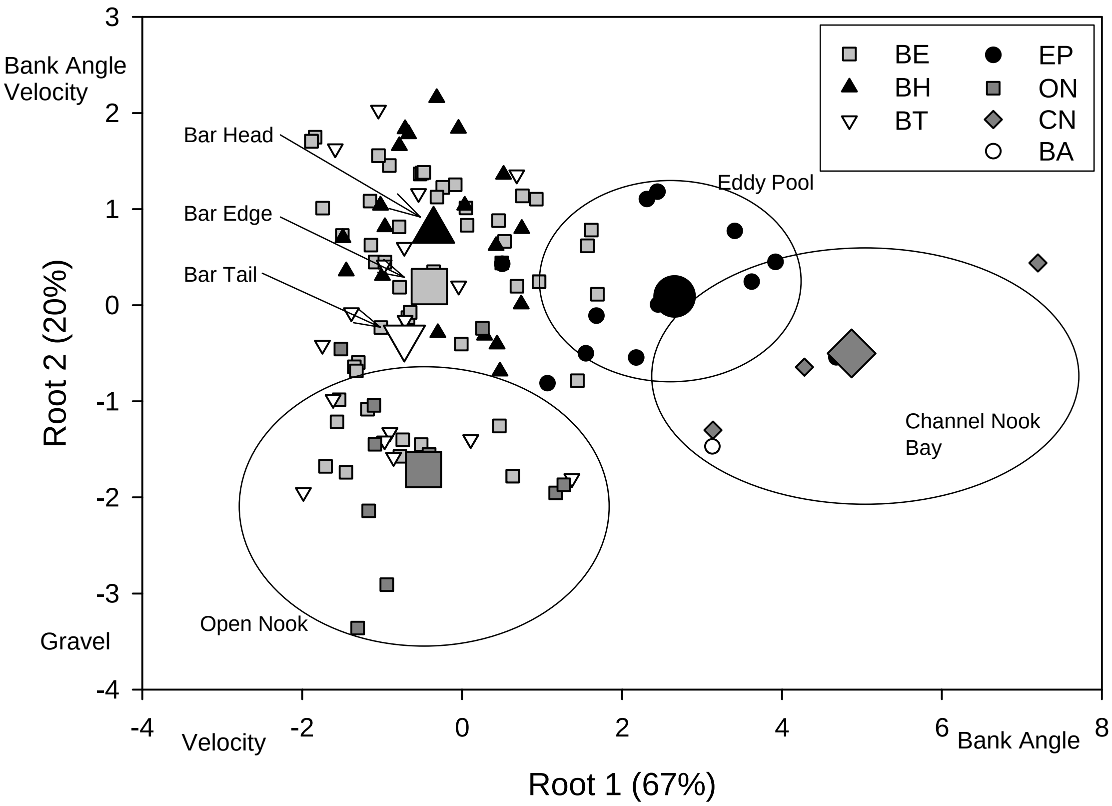


Figure 4-17. Canonical plot of main channel habitat units in the Rosedale sub-reach (autumn only). No ellipse was drawn around bar head, bar edge, and bar tail units for clarity. Enlarged symbols are the mean coordinates for habitat types on each canonical root. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

The resulting classification functions from DA correctly classified 42% of habitat units in the Rosedale sub-reach by the jackknife method. Bar edge units had a particularly low classification rate of 19% (**Table 4-7**). Excluding bar edge units, the average classification rate was 59%. The classification rate of open nooks was highest (83%), despite the apparent overlap in the canonical plot with bar edge and bar tail units. Two of 10 open nooks were misclassified as bar edge units. Nine of 13 eddy pool units (69%) were classified correctly, and misclassified units were mistaken to be bar edges and eddy pools. Bar head and bar edge units were mistaken for each other in a large number of cases, and bar edges were misclassified frequently as bar tails and open nooks as well. Bar tail units had a classification rate of 37% and the majority of misclassified units were classified as open nooks. Bar head and bar tail units were mistaken for one another in relatively few cases, indicating good discrimination between these habitats. The sample size of channel nooks was low and 2 of 3 were classified correctly by the jackknife method. The single observation of bay habitat was classified as a channel nook based on the classification functions.

Table 4-7. Classification matrix for main channel habitat units in the Rosedale sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat	% Correct	BH	BE	BT	EP	ON	CN	Total
BH	59	13	6	3	0	0	0	22
BE	19	20	10	9	4	10	0	53
BT	37	3	2	7	0	7	0	19
EP	69	0	2	0	9	0	2	13
ON	83	0	2	0	0	10	0	12
CN	67	0	0	0	1	0	2	3
Average/Total	42	36	22	19	14	27	4	122

Two-letter abbreviations of habitat types are in **Table 4-4**.

The problematic misclassification of bar edge units prompted an analysis of the physical gradients underlying bar edge variance. Principal components analysis (PCA) was applied to main channel bar edge units in the Rosedale sub-reach for this purpose, and all normal-transformed physical variables were included. The first three principal components (PC) collectively explained 86% of the total variance in the data. Bank angle and water depth were correlated significantly with the first PC (eigenvalue = 3.8), and the proportion of gravel was correlated with PC 2 (eigenvalue = 2.1). No variables were correlated with PC 3. A plot of principal component scores grouped samples in order of increasing bank angle along PC 1, with a clear break in unit groupings corresponding to 4^o (**Figure 4-18**). This separation of units according to bank angle suggested that the bar edge class may consist of two distinct sub-classes: steep bar edge ($\geq 4^o$) and flat bar edge ($< 4^o$). Bank angle was favoured as the distinguishing variable over water depth because depth varies with distance offshore and is determined by the width of the beach seine during sampling.

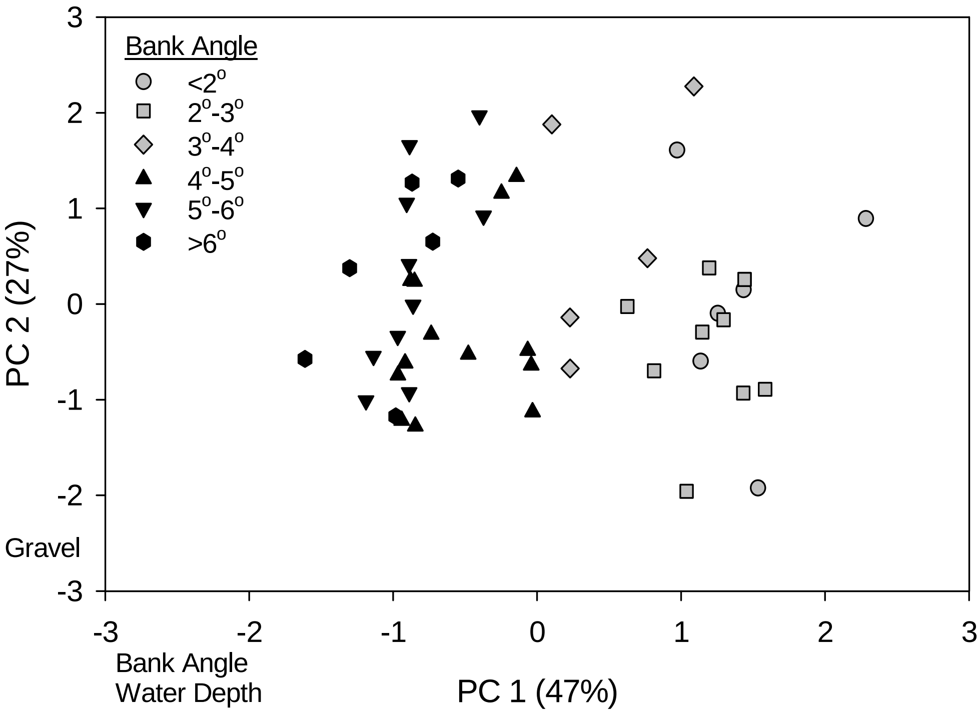


Figure 4-18. Principal component scores for main channel bar edge units in the Rosedale sub-reach, autumn sampling only. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses.

Based on PCA results, DA was applied with bar edge units separately classified as steep and flat units, and including all other original habitat types. Discriminant analysis revealed two significant canonical roots that collectively explained 89% of the total variance in the data. Root 1 was correlated positively with bank angle and water depth, and negatively correlated with the proportion of gravel (**Table 4-8**). Root 2 represented a hydraulic gradient of increasing velocity (positive correlation) and sand/silt (negative correlation). The canonical plot revealed almost complete overlap among bar heads and steep bar edges, which had strong positive loadings along Root 2 associated with velocity (**Figure 4-19**). Eddy pools and channel nooks each were associated in canonical space with high bank angle and sand/silt. There was overlap between open nook and flat bar edge units, which were associated with a high proportion of gravel-sized sediment. Bar tail units overlapped considerably with several habitat types, namely open nook, flat bar edge, steep bar edge, and bar head, thereby demonstrating high variability in physical character. Collectively, these five habitats were related along a hydraulic gradient corresponding to increasing velocity and bank angle.

Table 4-8. Factor loadings on each significant canonical root based on correlation analysis for main channel habitat units of the Rosedale sub-reach (bar edge units split based on bank angle).

Variable	Root 1	Root 2
Gravel	-0.41	0.07
Sand/Silt	0.25	-0.51
Bank Angle	0.86	0.34
Mean Velocity	-0.17	0.83
Mean Depth	0.83	0.36
Variance Explained (%)	54	35

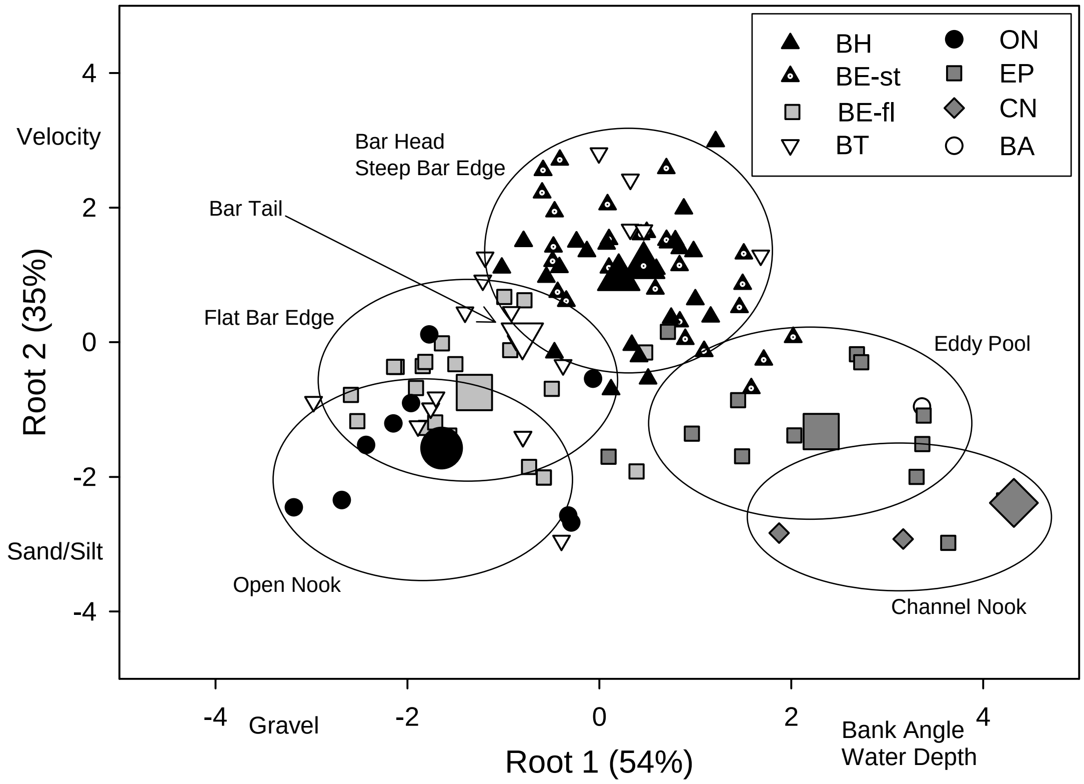


Figure 4-19. Canonical plot of main channel habitat units in the Rosedale sub-reach. No ellipse was drawn around bar tail units for clarity. Enlarged symbols are the mean coordinates for habitat types on each canonical root. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

The weighted average classification accuracy was 44% and identification of each bar edge sub-class improved to 41% (steep) and 52% (flat), respectively (**Table 4-9**). An equal proportion of steep bar edge units was classified correctly as was misclassified as bar heads. The classification accuracy of bar head units was reduced to 41%, and 8 of 13 misidentified units were classified as steep bar edge. Hence, steep bar edge and bar head units had similar physical character. Only 3 of 19 bar tails were classified correctly; misidentified units were mistaken as several other habitat types. Open nooks were classified correctly in 67% of cases and remaining cases all were mistaken to be flat bar edge units. The classification of eddy pool units was unchanged from the original analysis and the correct classification of relatively uncommon channel nooks was reduced to 33%.

Table 4-9. Classification matrix for main channel habitat units in the Rosedale sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat	**% Correct**	**BH**	**BE-s**	**BE-f**	**BT**	**EP**	**ON**	**CN**	**Total**
BH	41	9	8	0	5	0	0	0	22
BE-steep*	41	13	13	0	3	3	0	0	32
BE-flat*	52	3	0	11	1	0	6	0	21
BT	16	2	4	5	3	0	5	0	19
EP	69	0	1	1	0	9	0	2	13
ON	67	0	0	4	0	0	8	0	12
CN	33	0	0	0	0	2	0	1	3
Average/Total	44	27	26	21	12	14	19	3	122

* steep: $\geq 4^{o}$; flat: $< 4^{o}$.

Two-letter abbreviations of habitat types are in **Table 4-4**.

The physical similarity of bar head and steep bar edge units prompted a final DA of habitat types, with bar head and steep bar edge units merged as one group. Two canonical roots were significant and collectively explained 90% of the total variance in the data. Root 1 (54% variance explained) was negatively correlated with bank angle and water depth (**Table 4-10**), and positively correlated with gravel. Root 2 explained 36% of the variance in the data and represented a hydraulic gradient of velocity (positive) and sand/silt (negative).

Table 4-10. Factor loadings on each significant canonical root based on correlation analysis (bar head and steep bar edge units grouped for analysis).

Variable	Root 1	Root 2
Gravel	0.41	0.05
Sand/Silt	-0.27	-0.50
Bank Angle	-0.85	0.36
Mean Velocity	0.18	0.83
Mean Depth	-0.82	0.37
Variance Explained (%)	54	36

Habitats with deep water and a steep bank angle (eddy pool, channel nook, bay) were differentiated along Root 1 from shallower flat bar edges, open nooks, and bar tails, each of which also has a relatively high proportion of gravel substrate (**Figure 4-20**). High-velocity habitats (bar head and steep bar edge) were differentiated from slow-velocity habitats (open nook, eddy pool, channel nook, bay) along Root 2. Eddy pools were not discriminated from channel nooks. Root 2 achieved modest discrimination among habitats with a shallow bank angle (bar tail, flat bar edge, open nook) based on a decreasing hydraulic gradient.

The weighted-average jackknife classification accuracy was 59%, substantially higher than previous analyses, and merged bar head/steep bar edge units had a relatively high classification accuracy of 74% (**Table 4-11**). Flat bar edge units were identified correctly in 52% of cases, with units most commonly misclassified as open nooks (6/21 cases). Bar tail units had the lowest classification rate (16%), and units were mistaken as bar head/steep bar edges, flat bar edges, and open nooks in almost equal proportions. The classification success of remaining habitats (eddy pool, open nook, channel nook) was unchanged from the previous analysis. The single bay observation was most similar in physical character to channel nook units. The grouping of bar head and steep bar edge units appears to be appropriate based on a high degree of physical similarity and improved classification accuracy.

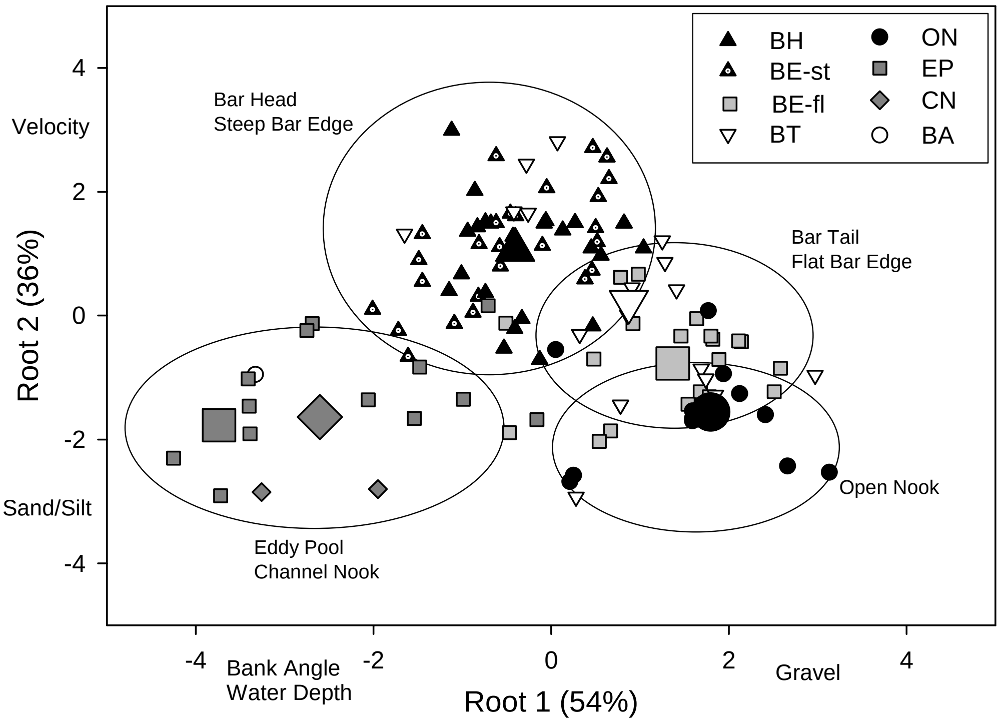


Figure 4-20. Canonical plot of main channel habitat units in the Rosedale sub-reach. Bar head and steep bar edge units are distinguished in the plot but were pooled for analysis. Enlarged symbols are the mean coordinates for habitat types on each canonical root. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

Table 4-11. Classification matrix for main channel habitat units in the Rosedale sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat	% Correct	BH+BE-s	BE-f	BT	EP	ON	CN	Total
BH+BE-steep*	74	40	0	9	5	0	0	54
BE-flat*	52	2	11	2	0	6	0	21
BT	16	5	6	3	0	5	0	19
EP	69	1	1	0	9	0	2	13
ON	67	0	4	0	0	8	0	12
CN	33	0	0	0	2	0	1	3
Average/Total	59	48	22	14	16	19	3	122

* steep: $\geq 4^{o}$; flat: $< 4^{o}$.

Two-letter abbreviations of habitat types are in **Table 4-4**.

The problematic misclassification of bar tail units prompted an analysis of the physical gradients underlying bar tail variance. Principal components analysis (PCA) was applied to main channel bar edge units in the Rosedale sub-reach for this purpose, and all normal-transformed physical variables were included. The first three principal components (PC) explained 95% of the total variance in the data. Bank angle, water depth, and velocity each were correlated significantly with PC 1 (eigenvalue = 4.7), and the proportion of gravel was correlated with PC 2 (eigenvalue = 1.6). No variable was correlated with PC 3. Unlike bar edge habitat that was clearly subdivided based on bank angle, a plot of principal component scores showed no meaningful grouping of samples along PC 1 or PC 2. Rather, bar tail units plotted haphazardly along PC 1. Results from DA indicated that bar tail units overlapped in physical characteristics with three habitat types, namely bar head/steep bar edge, flat bar edge, and open nook (**Table 4-11**). However, open nooks and flat bar edges were not misclassified as bar tails. Because of the variable misclassification of bar tails, they were retained as a separate habitat class for subsequent analyses.

4.4.2 Are Habitat Types Variable Among Sub-Reaches?

The similarity of Level 3 habitat units among Level 1 sub-reaches was examined based on the autumn sampling period, first, by graphical comparison and, second, by classifying habitat units in the Cheam and Chilliwack sub-reaches based on the discriminant model derived for habitat types in the Rosedale sub-reach. A downstream gradient among sub-reaches was evident in the proportions of large cobble and sand/silt, where the upstream Cheam sub-reach had the highest proportion of large cobble and lowest proportion of sand/silt for all but one habitat type (**Figure 4-21**). The proportion of cobble was highest in the Cheam sub-reach for hydraulically sheltered habitat types (eddy pool, open nook, channel nook) as well, but was lowest of all sub-reaches for all hydraulically exposed habitats except flat bar edge. Bar head and steep bar edge units were comparable in sedimentary character, which was similar among sub-reaches. Flat bar edge units showed variable proportions of cobble and gravel among sub-reaches. A longitudinal gradient in sediment size was evident in open nooks from the Cheam to the Chilliwack sub-reach corresponding to a decreasing proportion of cobble and increasing proportion of sand/silt in the downstream direction. Channel nooks and bays consisted almost entirely of sand/silt in the Rosedale sub-reach but were more variable in the Cheam and Chilliwack sub-reaches.

Water velocity represented a physical gradient among habitat types, with hydraulically exposed habitats (bar head, bar edge, bar tail) having higher velocity than sheltered habitats (eddy pool, open nook, channel nook) in all sub-reaches (**Figure 4-22**). The hydraulic gradient was most evident in mean velocity, with maximum velocity being variable between sub-reaches and habitat types. Whereas mean water depth was similar among sub-reaches for most habitat types, maximum water depth was lower in the Rosedale sub-reach for all habitats except channel nooks and bays. As expected, steep and flat bar edge units differed with respect to water depth; flat bar edge units were most similar to open nooks. Bank angle consistently was steepest in the Cheam sub-reach for all habitat types. However, a downstream gradient among sub-reaches was not evident because the most downstream Chilliwack sub-reach had a consistently higher bank slope than the Rosedale sub-reach.

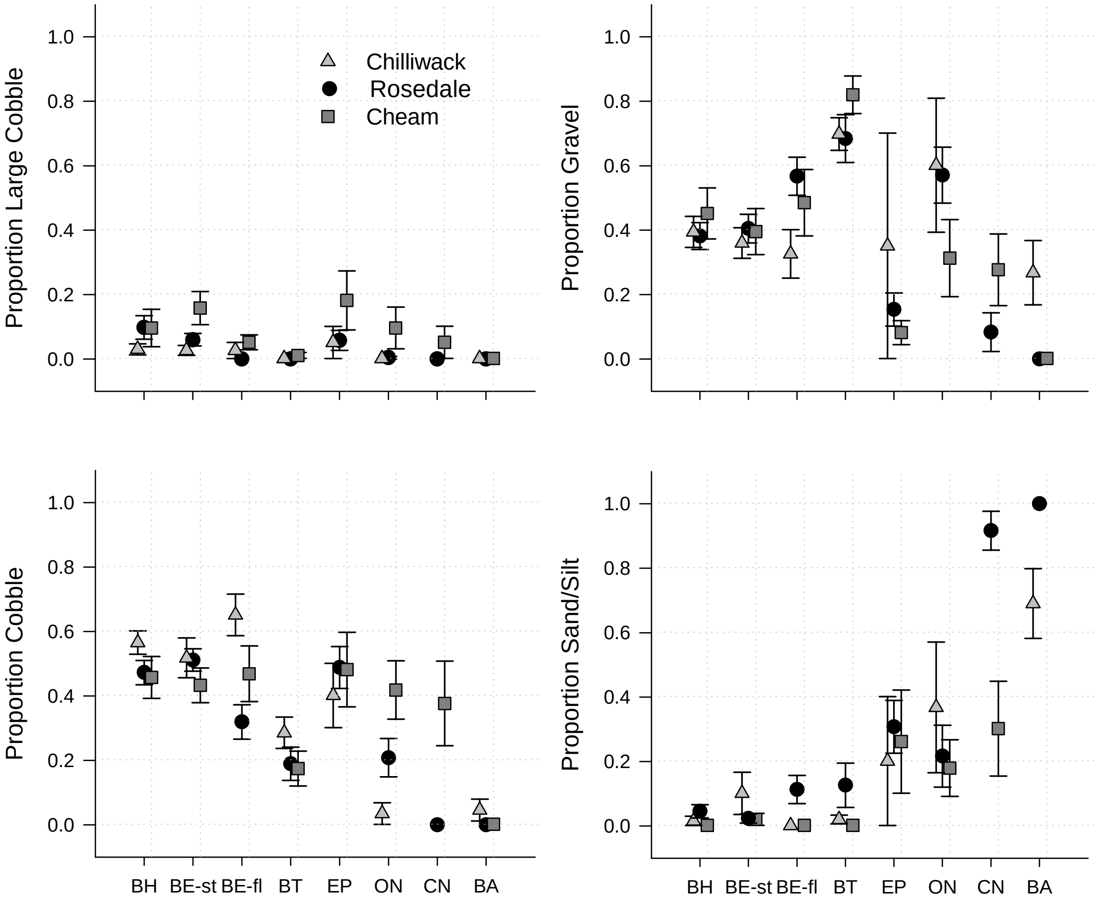


Figure 4-21. Substrate characteristics (mean ± SE) of main channel habitat types in the Chilliwack, Rosedale, and Cheam sub-reaches of Fraser River. Data were collected during the autumn season (1999-2001). Two-letter abbreviations of habitat types are in **Table 4-4**. BE-st: steep bar edge; Be-fl: flat bar edge.

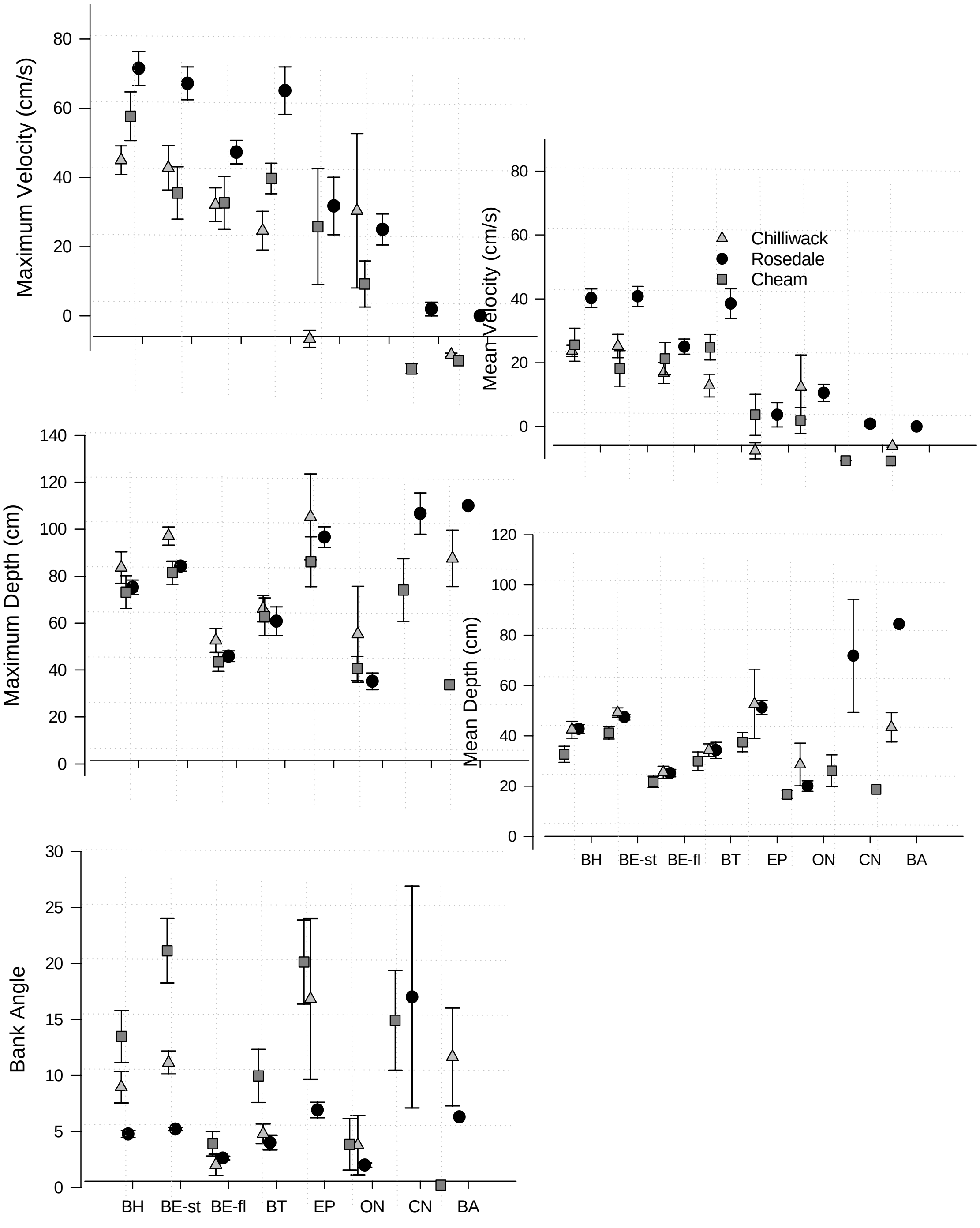


Figure 4-22. Water velocity, depth, and bank slope characteristics (mean ± SE) of main channel habitat types in the Chilliwack, Rosedale, and Cheam sub-reaches of Fraser River. Data were collected during the autumn season (1999-2001). Two-letter abbreviations of habitat types are in **Table 4-4**. BE-st: steep bar edge; Be-fl: flat bar edge.

The weighted average classification success of habitat units in the Chilliwack sub-reach based on classification functions derived from the Rosedale sub-reach was 63% (**Table 4-12**). Flat bar edge units had a perfect classification accuracy and the merged group of bar head and steep bar edge units was classified correctly in 72% of cases; most misidentified cases were classified as flat bar edges. The two eddy pool units in the Chilliwack sub-reach had a perfect classification rate (100%). In contrast, bar tail and open nook units were classified correctly in less than 40% of cases; the majority of misidentified bar tail units were mistaken to be open nooks. Of the three open nooks sampled, one was classified correctly and mistaken units were classified as bar tails. Nine bay habitats were sampled in the Chilliwack sub-reach and the classification functions of the Rosedale sub-reach were applied to determine the habitat that they most likely resembled. Six bays in the Chilliwack sub-reach (67%) were classified as channel nooks, 2 as open nooks, and 1 was classified as an eddy pool.

Table 4-12. Classification matrix for main channel habitat units in the Chilliwack sub-reach based on classification functions from the Rosedale sub-reach. Row values represent the field-based classification and column values represent the predicted classification.

Habitat	% Correct	BH+BE-s	BE-f	BT	EP	ON	CN	Total
BH+BE-steep*	72	23	6	1	1	0	1	32
BE-flat*	100	0	4	0	0	0	0	4
BT	38	1	3	6	1	5	0	16
EP	100	0	0	0	2	0	0	2
ON	33	0	0	2	0	1	0	3
CN	-	-	-	-	-	-	-	0
BA	-	0	0	0	1	2	6	9
Average/Total	63	24	13	9	5	8	7	66

* steep: $\geq 4^{o}$; flat: $< 4^{o}$.

Two-letter abbreviations of habitat types are in **Table 4-4**.

Classification accuracy averaged 53% for Cheam habitat units based on classification functions derived from the Rosedale sub-reach. Bar head/steep bar edge units had a classification accuracy of 60%, with cases being misclassified as flat bar edge, bar tail, and eddy pool units in equal proportions (**Table 4-13**). Bar tails and eddy pools each had high classification success (73% and 80%, respectively). Open nooks and flat bar edge units had relatively low classification accuracy;

most open nooks were mistaken to be flat bar edges. The majority of misidentified flat bar edge units were classified as bar head/steep bar edges. Three of 4 channel nooks were mistaken to be eddy pool units. The one bay habitat unit sampled in the Cheam sub-reach was classified as an open nook based on the classification functions derived for the Rosedale sub-reach.

Table 4-13. Classification matrix for main channel habitat units in the Cheam sub-reach based on discriminant functions from the Rosedale sub-reach. Row values represent the actual classification and column values represent the predicted classification.

Habitat	% Correct	BH+BE-s	BE-f	BT	EP	ON	CN	Total
BH+BE-steep*	60	15	3	3	3	1	0	25
BE-flat*	42	4	5	1	1	1	0	12
BT	73	1	1	8	0	1	0	11
EP	80	0	1	0	4	0	0	5
ON	33	0	5	0	1	3	0	9
CN	0	0	0	0	3	1	0	4
BA	-	0	0	0	0	1	0	1
Average/Total	53	20	15	12	12	8	0	67

* steep: $\geq 4^{o}$; flat: $< 4^{o}$.

Two-letter abbreviations of habitat types are in **Table 4-4**.

DA was applied separately to each of the Chilliwack and Cheam sub-reaches to determine the physical factors that discriminated most among habitat types and the degree to which habitat types were differentiated in comparison with the Rosedale sub-reach. For the Chilliwack sub-reach, two canonical roots were significant ($p = 0.009$). Root 1 explained 66% of the variance in the data and represented a hydraulic gradient of mean velocity (positive) and sand/silt (negative). Root 2 (23% variance explained) was correlated with the proportion of gravel (negative) and bank angle (positive). Habitat types were differentiated from one another reasonably well along the hydraulic gradient of Root 1 (**Figure 4-23**), suggesting habitats were relatively distinct in hydraulic character. Steep bar edge/bar heads, bays, and eddy pools with characteristically deep water were differentiated along Root 2 from flat bar edge, bar tail, and open nook habitats, which were associated with high loadings of gravel substrate and shallow water depth along Root 2.

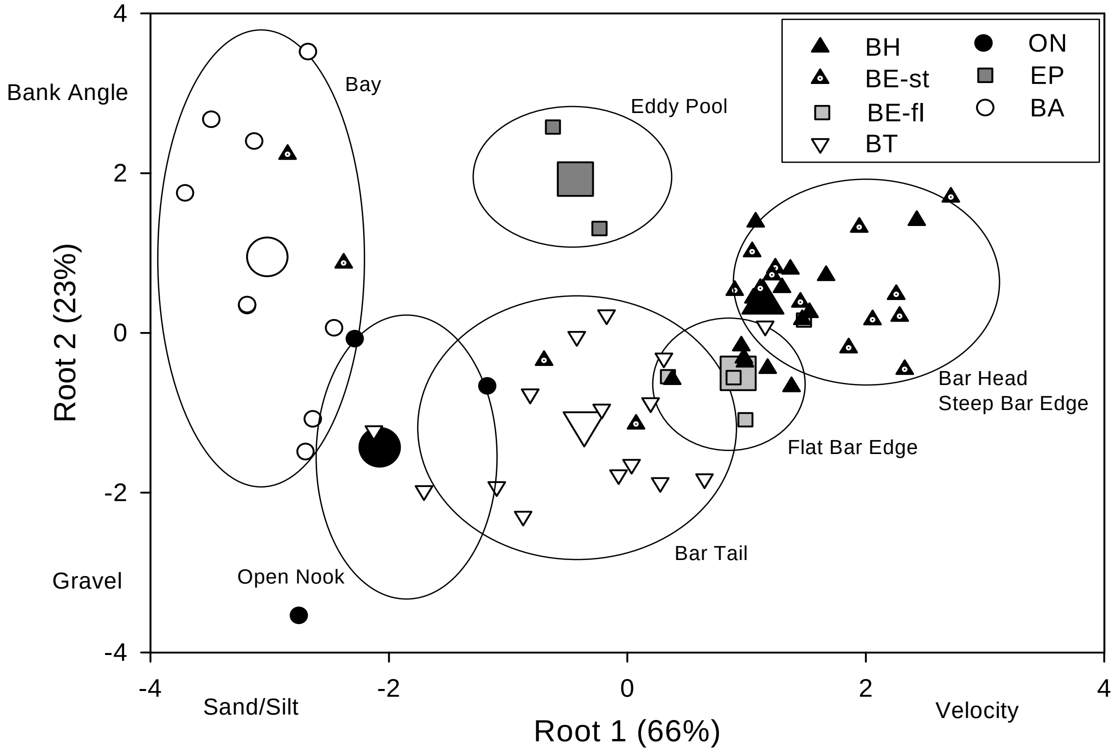


Figure 4-23. Canonical plot of main channel habitat units in the Chilliwack sub-reach. Bar head and steep bar edge units are distinguished in the plot but were pooled for analysis. Enlarged symbols are the mean coordinates for habitat types on each canonical root. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

The average jackknife classification success was 61% for habitat units in the Chilliwack sub-reach. This success was comparable to *a priori* classification of 63% based on Rosedale classification functions, and to the jackknife success for Rosedale units of 59%. Bar heads and flat bar edges had the highest classification accuracy (72% and 75%, respectively). Bar tails were classified correctly in 50% of cases, and misclassified cases were mistaken to be all other habitat types except bays (**Table 4-14**). The small sample size of eddy pools (n = 2) may have contributed to a jackknife classification of 0%, because *post-hoc* classification accuracy by resubstitution was 100%. Open nooks had relatively low classification accuracy (33%), but sample size also was low. Bays were correctly identified in 5 of 9 cases (56%), and misclassified units were mistaken to be open nooks, eddy pools, and bar tails. Incidentally, the classification accuracy of Rosedale and Cheam sub-reach units based on classification functions from the Chilliwack sub-reach averaged 58% and 41%, respectively.

Table 4-14. Classification matrix for main channel habitat units in the Chilliwack sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat	% Correct	BH+BE-s	BE-f	BT	EP	ON	BA	Total
BH+BE-steep*	72	23	4	2	0	1	2	32
BE-flat*	75	0	3	1	0	0	0	4
BT	50	1	2	8	2	3	0	16
EP	0	0	1	1	0	0	0	2
ON	33	0	0	1	0	1	1	3
BA	56	0	0	1	1	2	5	9
Average/Total	61	24	10	14	3	7	8	66

* steep: ≥ 4°; flat: < 4°.
Two-letter abbreviations of habitat types are in **Table 4-4**.

Only one canonical root was significant in the discriminant analysis of main channel habitat units in the Cheam sub-reach ($p = 0.0005$), which explained 57% of the variance in the data. The canonical root represented a hydraulic gradient with significant positive loadings of velocity and a negative loading of sand/silt. Habitats were distinguished along the hydraulic gradient of Root 1 with only moderate success, whereas Root 2 differentiated habitats with a steep bank angle (channel nook, eddy pool, bar head/steep bar edge) from those with a shallow bank angle and high proportion of gravel (open nook, flat bar edge, bar tail, **Figure 4-24**). Root 2 was not significant by Chi-Square analysis ($p = 0.18$), but accounted for 33% of the variance and was correlated with bank angle (positive) and sand/silt (negative).

Average classification accuracy of units in the Cheam sub-reach was 50% (**Table 4-15**), compared with 61% for habitat units in the Chilliwack sub-reach and 59% in the Rosedale sub-reach. The *a priori* classification success of Cheam units based on Rosedale classification functions averaged 53%. Bar head/steep bar edge units had a classification rate of 56%, being mistaken for flat bar edge, bar tail, and eddy pool units in almost equal proportions. Flat bar edge units were identified correctly in 50% of cases and were misclassified as bar heads, bar tails, and open nooks. Similarly, misclassified bar tail units were mistaken as flat bar edge, bar head, and open nook units. Bar tail units had the highest classification success in the Cheam sub-reach (73%). Incidentally, the classification accuracy of Chilliwack and Rosedale sub-reach units based on classification functions from the Cheam sub-reach averaged 60% and 61%, respectively.

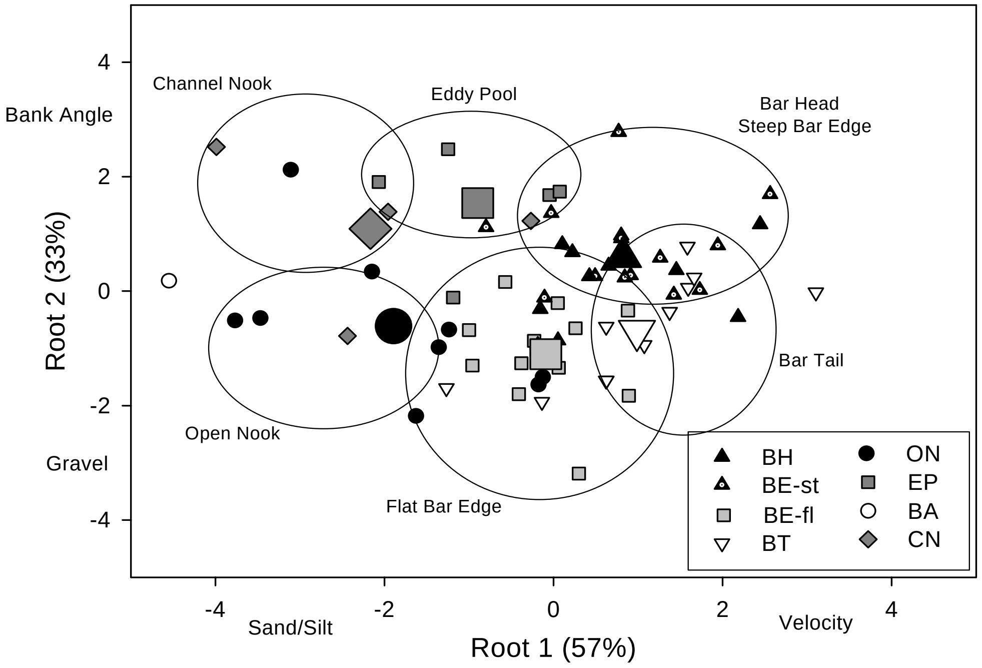


Figure 4-24. Canonical plot of main channel habitat units in the Cheam sub-reach. Bar head and steep bar edge units are distinguished in the plot but were pooled for analysis. Enlarged symbols are the mean coordinates for habitat types on each canonical root. Physical variables correlated significantly with each axis are indicated and the percentage of variance explained is given in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

Table 4-15. Classification matrix for main channel habitat units in the Cheam sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat	% Correct	BH+BE-s	BE-fl	BT	EP	ON	CN	Total
BH+BE-steep*	56	14	4	3	4	0	0	25
BE-flat*	50	2	6	3	0	1	0	12
BT	73	1	1	8	0	1	0	11
EP	0	2	0	0	0	1	2	5
ON	33	0	3	0	0	3	3	9
CN	50	1	0	0	0	1	2	4
Average/Total	50	20	14	14	4	7	7	66

* steep: ≥ 4^o; flat: < 4^o.

Two-letter abbreviations of habitat types are in **Table 4-4**.

4.4.3 Are Habitat Types Variable Among Channel Types?

Habitat characteristics of main channel units were compared with side and summer channel units based on observations from the Rosedale sub-reach (autumn sampling only). Large cobble was rare in side and summer channels and only observed in bar head units (**Figure 4-25**). Both main channel steep bar edges and eddy pools had a high proportion of cobble substrate whereas units in side and summer channels consisted predominantly of gravel. The main channel had a lower or equal proportion of sand/silt than side and summer channels in all habitat types except channel nooks and bays. This pattern was particularly notable for open nooks, which consisted predominantly of sand/silt in side and summer channels as compared to gravel in the main channel. For bar head, bar edge, and bar tail habitats, there was a general gradient corresponding to channel size of decreasing cobble sediment and increasing gravel and sand/silt.

Main channel bar head and bar edge units had higher water velocity than side and summer channels (**Figure 4-26**). Bar tail units were similar within main and side channels, but had notably higher velocity than summer channels. Main channel open nooks had higher water velocity compared with side and summer channels. The maximum depth of habitat units was relatively variable among channel types. The average depth of bar heads, channel nooks, and bays was notably higher in the main channel whereas average depth was relatively similar among channels for other habitat types. Side and summer channels had a consistently steeper bank angle for all habitat types except channel nook. Overall, bank angle was highly variable for hydraulically sheltered habitats (eddy pool, open nook, channel nook, bay) within side and summer channels.

The classification functions from discriminant analysis of main channel habitat units were applied to side and summer channel observations to evaluate the similarity of physical characteristics among channel types. Only data collected from channels of the Rosedale sub-reach were included. The *a priori* classification accuracy for side channel habitat units averaged 41%. Fifteen bar head/steep bar edges and 8 bar tails were classified correctly in only 13% and 25% of cases, respectively. In contrast, the classification accuracy of flat bar edges was 77%. Eddy pools were classified correctly in 50% of cases and misclassified units were classified as bar tail, flat bar edge, and channel nook. Six of 9 open nooks were correctly classified, and 2 of the 3 misclassified units were mistaken to be flat bar edges. All channel nook units were misidentified, the majority being misclassified as open nooks (3 of 5 cases).

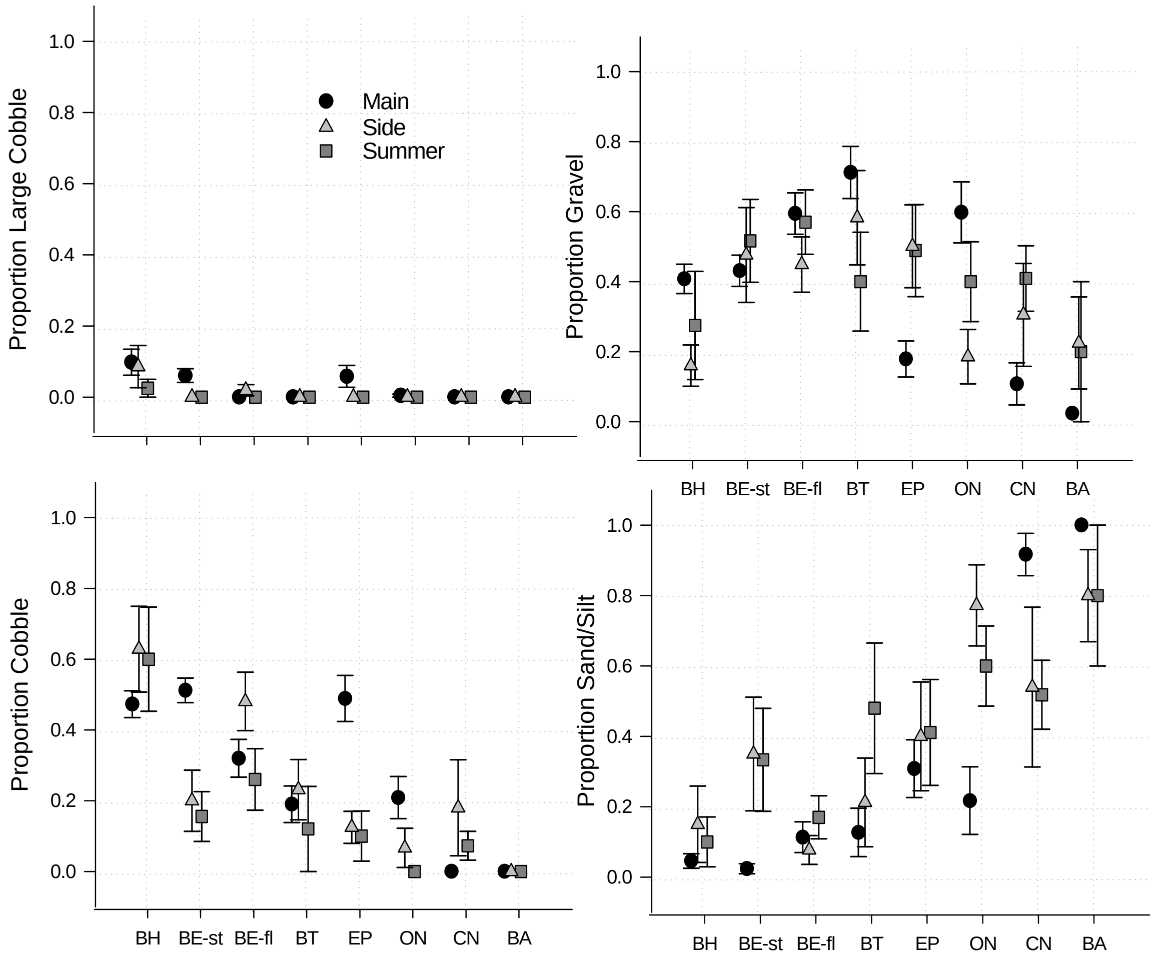


Figure 4-25. Substrate characteristics (mean ± SE) of habitat types in main, side, and summer channels of the Rosedale sub-reach of Fraser River. Data were collected during the autumn season (1999-2001). BE-st: steep bar edge; Be-fl: flat bar edge. Two-letter abbreviations of habitat types are in **Table 4-4**.

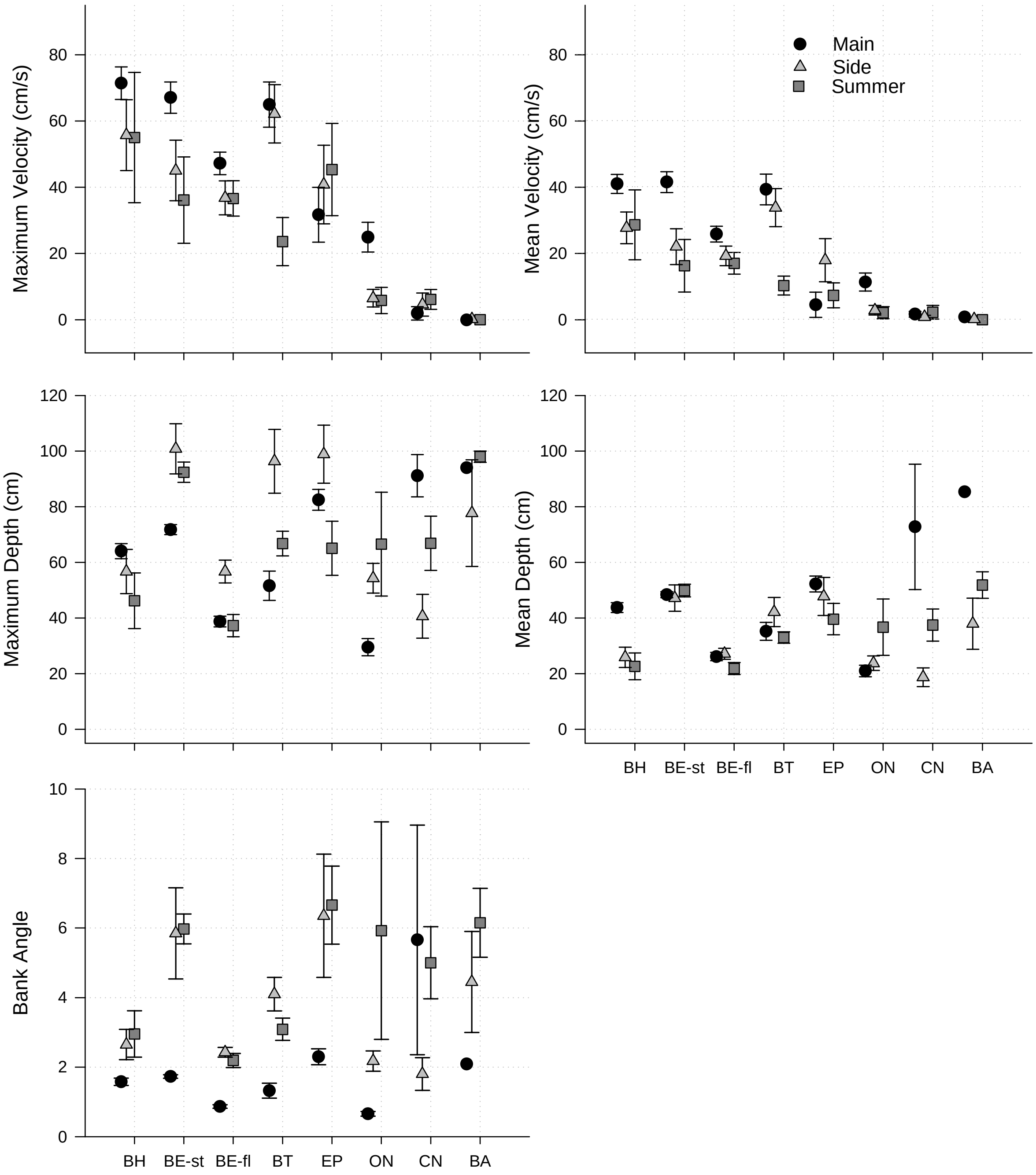


Figure 4-26. Water velocity, depth, and bank slope characteristics (mean ± SE) of habitat types in main, side, and summer channels within the Rosedale sub-reach of Fraser River. Data were collected during the autumn season (1999-2001). BE-st: steep bar edge; Be-fl: flat bar edge. Two-letter abbreviations of habitat types are in **Table 4-4**.

The *a priori* classification success of summer channel habitat units based on main channel classification functions was low and averaged 30%. The classification accuracy of bar head/steep bar edge was 23% and misidentified units were classified as all other habitat types in almost equal proportions. No bar tail units were classified correctly and 80% were misidentified as open nook units. The majority of misclassified flat bar edge units were mistaken to be open nooks as well. Eddy pools and open nooks each were most commonly mistaken to be channel nooks. Only 1 of 11 channel nooks was classified correctly, with most misidentified units being mistaken for higher velocity habitats such as eddy pools and open nooks.

DA was applied separately to side and summer channel habitat data (Rosedale sub-reach, autumn season only) to evaluate the distinctiveness of habitat types in secondary channels. The analysis of side channel data identified one significant canonical root ($p = 0.05$), which was correlated with velocity (negative) and sand/silt (positive). Habitat types showed modest clustering along Root 1 in the canonical plot (not shown). Classification success by the jackknife method averaged 43% and ranged between 0% and 63%. Flat bar edge and bar tail units each had >50% classification accuracy and misclassifications were most commonly the other habitat type. Bar head/steep bar edge units were misclassified in 10 of 15 cases (33% accuracy), and most units were mistaken to be eddy pools. Eddy pools were misclassified in 5 of 8 cases as bays, bar tails, and flat bar edges. Open nooks and channel nooks and bays were each mistaken as the others in several cases. All bays ($n = 3$) were misclassified as a channel nook, eddy pool, and open nook unit.

The analysis of summer channel data identified no significant canonical roots and the canonical plot (not shown) approximately matched the random model of Wright and Li (2002). Average classification accuracy was 24% by the jackknife method (0% - 50% range). Classification accuracy did not appear related to sample size because 11 channel nooks were classified incorrectly (0%). Flat bar edge units and bays had highest accuracy (50%); all other groups were misclassified frequently as many different habitat types. Flat bar edges, bar tails, and open nooks were each most commonly mistaken for each other, and eddy pools, bays, and channel nooks were mistaken for each other most frequently as well.

4.4.4 Morphological Habitat Classification Summary

A summary of results from discriminant analyses and the *jackknife* classification of habitat units is presented in **Table 4-16**. The revised classification, which differentiated bar edge units based on bank angle, showed a substantial improvement in classification success over the original model (59% versus 42% for the Rosedale sub-reach). Two canonical roots were significant in most analyses and together explained, on average, more than 85% of the variance in the data. For all analyses, Root 1 and Root 2 were correlated with water velocity and/or bank angle, indicating that a hydraulic gradient primarily discriminated among habitat types. A significant loading of velocity most often corresponded with an inverse loading of sand/silt, which suggests that habitat types also were discriminated by a secondary gradient of sediment size. Grain size is naturally correlated with hydraulics by the flow competence to move material. Comparing sub-reaches, factor loadings on the canonical roots were highly similar for all sub-reaches.

Table 4-16. Summary of significant canonical roots (% variance explained and significantly correlated variables) and jackknife classification accuracy of habitat units as determined by discriminant analysis. The grouping of bar head (BH) and bar edge (BE) units for each data set is indicated.

Data Set	**Root 1**			**Root 2**			**% Classification Accuracy**
Sub-Reach, Channel	**% Var.**	**+ve**	**-ve**	**% Var.**	**+ve**	**-ve**	
Rosedale Main *BH, BE*	67	B. Angle	Velocity	20	B. Angle	Sand/Silt	42
Rosedale Main *BH, BE-st, BE-fl*[+]	54	B. Angle	Gravel	35	Velocity	Sand/Silt	44
Rosedale Main *BH+BE-st, BE-fl*	54	B. Angle	Gravel	36	Velocity	Sand/Silt	59
Chilliwack Main *BH+BE-st, BE-fl*	66	Velocity	Sand/Silt	23	B. Angle	Gravel	61
Cheam Main *BH+BE-st, BE-fl*	57	Velocity	Sand/Silt	33*	B. Angle	Sand/Silt	50
Rosedale Side *BH+BE-st, BE-fl*	78	Sand/Silt	Velocity	13*	Sand/Silt	-	43
Rosedale Summer *BH+BE-st, BE-fl*	52*	B. Angle	-	31*	Velocity	Sand/Silt	24

Grey-shaded row represents the revised and final grouping of morphological habitat types.
*not significant by Chi-Square test ($\alpha = 0.1$)
+ Steep bar edge (BE-st): ≥ 4°; Flat bar edge (BE-fl): < 4°.
All data were collected from main channel habitat units of the Rosedale sub-reach in autumn unless otherwise indicated.

Classification accuracy of habitat units averaged 54% by the jackknife method across all sub-reaches and channel types. Overall, jackknife classification success was only slightly higher than classification accuracy by *a priori data splitting* (54% versus 47%), where classification functions derived from Rosedale main channel units were used to predict habitat membership for other sub-reaches and channel types (**Table 4-17**). For main channel units in the Chilliwack and Cheam sub-reaches, *a priori* classification success was actually higher based on Rosedale classification functions than by the jackknife method (58% versus 56% weighted average). Classification accuracy for Cheam habitat units was lower than in downstream sub-reaches. Secondary channels had the lowest classification accuracy and, for summer channel units, *a priori* classification accuracy was higher than by jackknife classification (30% versus 24%).

Table 4-17. Summary of habitat classification accuracy (%) based on discriminant analysis for 3 sub-reaches and 3 channel types. Data were collected from main channel units of the Rosedale sub-reach during autumn, unless otherwise specified.

Group	**BH+BE-s**	**BE-f**	**BT**	**EP**	**ON**	**CN**	**BA**	**Mean***
Rosedale S-R	74	52	16	69	67	33	-	59
Chilliwack S-R	72	75	50	0	33	-	56	61
R→Chilliwack	*72*	*100*	*38*	*100*	*33*	-	-	*63*
Cheam S-R	56	50	73	0	33	50	-	50
R→Cheam	*60*	*42*	*73*	*80*	*33*	*0*	-	*53*
Side Channel	33	62	63	38	44	20	0	43
Main→Side	*13*	*77*	*25*	*50*	*67*	*0*	-	*41*
Summer Ch.	31	50	0	33	0	0	50	24
Main→Summer	*23*	*50*	*0*	*44*	*60*	*9*	-	*30*
Mean % (jackknife)*	64	57	41	48	50	21	55	54
Total # Cases	134	58	59	31	32	19	11	344
Mean % (a priori)*	*51*	*62*	*40*	*58*	*50*	*5*	-	*47*
Total # Cases	*85*	*39*	*40*	*24*	*26*	*20*	-	*234*

Two-letter abbreviations of habitat types are in **Table 4-4**.
*Means are the weighted average based on the number of cases representing each habitat type.
Non-italicized entries are made by jackknife classification from functions of the specified sub-reach or channel type.
Italicized entries were made *a priori* based on classification functions from main channel units in the Rosedale sub-reach during autumn sampling. E.g., *R→Chilliwack: Rosedale functions applied to Chilliwack cases; Main→Side: Main channel functions applied to side channel cases.*

4.4.5 Hydraulic Habitat Classification

Modifications to the original habitat classification by dividing bar edge units based on bank angle, and amalgamating of steep bar edge and bar head units, have produced a six-member habitat classification. The habitat types are morphologically distinct and visually discriminated from one another; hence, this classification represents a "morphological model" for physical organization of the gravel reach at the habitat scale.

Parsimony is an important trait of habitat classifications, thus further simplification to the habitat classification was evaluated. Habitats were grouped according to hydraulic character because a hydraulic gradient consistently captured the greatest variance in the data for all discriminant analyses. Furthermore, flow velocity is a widely recognized factor influencing the distribution of aquatic organisms (Beecher *et al.* 1995). This "hydraulic model" for habitat structure in the gravel reach consists of four groups (**Figure 4-27**) and represents an alternative hypothesis to evaluate the habitat associations of invertebrate taxa (Chapter 5) and fish species (Chapter 6).

Figure 4-27. Alternative models of habitat structure in the gravel reach of Fraser River. Dashed lines indicate the expected relations for those habitat types with an insufficient sample size for statistical analysis.

Riffles were not consistently sampled and were retained as a distinct hydraulic group. Bar head and steep bar edge units represent the "exposed" habitat group. Bar tails, flat bar edges, and open nooks were combined to represent the "normal" habitat group, being characterized by moderate flow velocity aligned parallel to the banks and with a high proportion of gravel-sized sediment. Moreover, misidentified units of bar tail, flat bar edge, and open nook were most often mistaken to be one of the other two habitat types in this group in previous analyses. The term "normal" was chosen because moderate velocity and parallel flow alignment evoke a “normal” condition. Channel nooks and bays are morphologically different and easily distinguished by visual assessment based on size, but are highly similar in hydraulic character and provide sheltered habitat from the main flow. Eddy pools, although morphologically distinct, were misclassified as channel nooks in several analyses and the back eddy flow character provides hydraulic shelter from the main flow; hence, these units were grouped together to represent the “sheltered” habitat group.

DA was applied to the simplified hydraulic habitat classification (only 2 roots are possible with 3 classes). Along Root 1, bank angle had the highest negative loading and the proportion of gravel had the highest positive loading (**Table 4-18**). Root 2 represented a hydraulic gradient of velocity (negative correlation) and sand/silt (positive correlation), and discriminated exposed habitats from other hydraulic groups based on high velocity.

Table 4-18. Factor loadings on each significant canonical root based on correlation analysis for main channel habitat units of the Rosedale sub-reach (hydraulic model).

Variable	Root 1	Root 2
Gravel	0.47	0.04
Sand/Silt	-0.28	0.51
Bank Angle	-0.72	-0.38
Mean Velocity	0.28	-0.75
Mean Depth	-0.71	-0.39
Variance Explained (%)	**66**	**34**

The canonical plot revealed less overlap among hydraulic habitat groups compared with all previous analyses (**Figure 4-28**). Along Root 1, hydraulic habitat groups were clustered in order of increasing bank angle and water depth. Normal habitats had the shallowest bank angle and also were correlated with gravel along this axis. Exposed habitats had a higher bank angle than normal habitats, which were intermediate in bank angle between exposed and sheltered habitats. Sheltered and normal habitats were discriminated based on bank angle along Root 1, but were not discriminated by the velocity gradient represented by Root 2. Jackknife classification accuracy of hydraulic habitat groups was high, averaging 84% (**Table 4-19**). Exposed and normal units each were misclassified as the other in several cases whereas sheltered units had high classification success (88%). Of the 17 sheltered units, only 1 unit was misclassified as each of the exposed and normal groups.

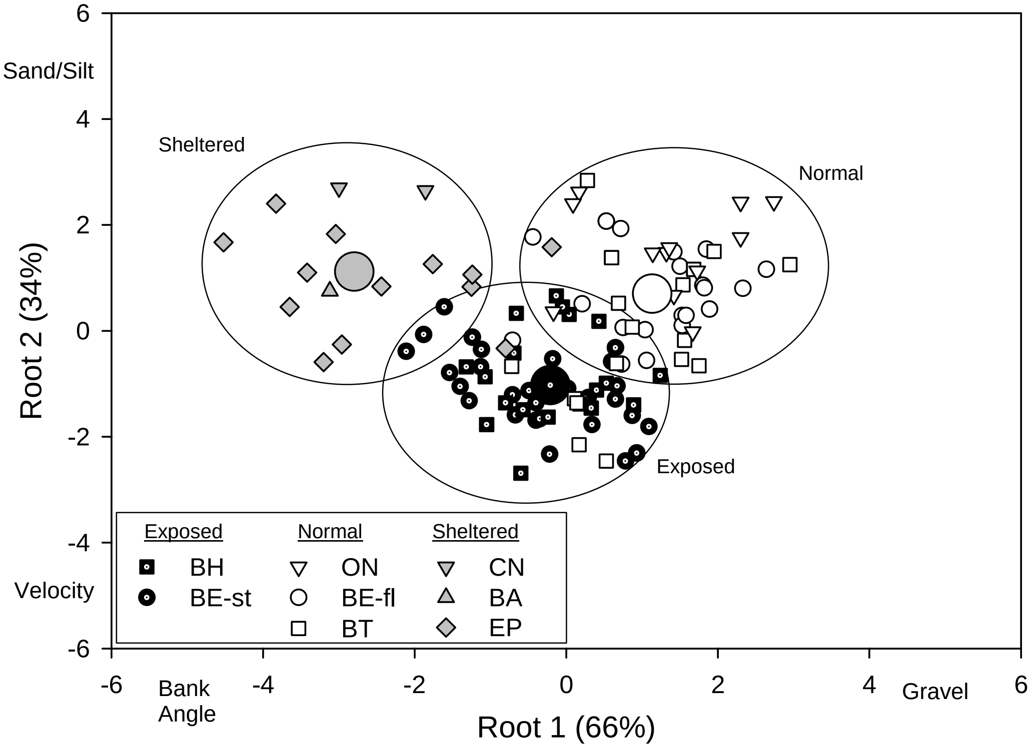


Figure 4-28. Canonical plot of main channel habitat units in the Rosedale sub-reach during autumn sampling. Habitat types were grouped by hydraulic character (indicated by symbol shade) for the analysis. Enlarged symbols are the mean coordinates for hydraulic habitat groups on each canonical root. The percentage of variance explained by each root and correlated variables are indicated in parentheses. Two-letter abbreviations of habitat types are in **Table 4-4**.

Table 4-19. Classification matrix for main channel habitat units in the Rosedale sub-reach based on discriminant analysis. Row values represent the actual classification and column values represent the predicted classification.

Habitat Group	% Correct	Exposed	Normal	Sheltered	Total
Exposed **(BH, BE-st$^+$)**	83	45	6	3	54
Normal **(BE-fl$^+$, ON, BT)**	83	9	43	0	52
Sheltered **(EP, CN, BA)**	88	1	1	15	17
Average/Total	84	55	50	18	123*

* One bay included in sheltered group that was excluded from previous analyses.
+ Steep bar edge (BE-st): ≥ 4°; Flat bar edge (BE-fl): < 4°.
Two-letter abbreviations of other habitat types are in **Table 4-4**.

The identical sequence of analyses presented above for the morphological habitat classification was applied to the various sub-reaches for hydraulic habitat classification. Results from these analyses are summarized in **Table 4-20**. Average habitat classification success by the jackknife method increased from 54% to 69%, and average *a priori* classification success increased even more substantially from 47% to 67%. The exposed habitat group had highest jackknife classification accuracy, whereas the normal hydraulic group had highest *a priori* classification success. Sheltered habitat units in the main channel of the Cheam sub-reach had highest variability in physical character and lowest classification success of all main channel units in the gravel reach. Classification success was similar by *a priori* and jackknife methods for the normal habitat group in Chilliwack and Cheam sub-reaches, and average classification accuracy by each method was similar as well. The classification of main channel units, overall, was good by both classification methods, ranging between 50% and 91%.

Similar to the morphological habitat classification, secondary channels had lower classification accuracy than main channel units, and *a priori* classification based on main channel characteristics had higher success than the jackknife method. Exposed hydraulic habitats in secondary channels had a particularly low classification accuracy compared with main channel units. The weighted average classification accuracy of main channel exposed units was 81% compared to 75% when secondary channels were included. Within side channels, normal habitats overlapped both exposed and sheltered habitats substantially (based on jackknife classification). In contrast, exposed habitats showed the greatest variability in physical character within summer channels.

Table 4-20. Summary of habitat classification accuracy (%) based on discriminant analysis for 3 sub-reaches and 3 channel types. Data were collected from main channel units of the Rosedale sub-reach during autumn, unless otherwise specified.

Group	**Exposed** **(BH, BE-st)**	**Normal** **(BE-fl, ON, BT)**	**Sheltered** **(EP, CN, BA)**	**Mean***
Rosedale S-R	83	83	88	**84**
Chilliwack S-R	81	87	73	**82**
R→Chilliwack	*78*	*91*	*64*	***80***
Cheam S-R	76	66	50	**67**
R→Cheam	*60*	*66*	*60*	***63***
Side Channel	53	23	50	**38**
Main→Side	*27*	*77*	*50*	***57***
Summer Ch.	46	65	50	**55**
Main→Summer	*38*	*85*	*73*	***69***
Mean % (jackknife)*	75	66	62	**69**
Total # Cases	139	157	76	**372**
Mean % (a priori)*	*58*	*78*	*63*	***67***
Total # Cases	*85*	*105*	*59*	***249***+

*Means are the weighted average based on the number of cases representing each habitat type.

+Discrepancies with Table 17 are because bay habitat units were included in analyses.

Non-italicized entries are made by jackknife classification from functions of the specified sub-reach or channel type.

Italicized entries were made *a priori* based on classification functions from main channel units in the Rosedale sub-reach during autumn sampling. E.g., *R→Chilliwack: Rosedale functions applied to Chilliwack cases; Main→Side: Main channel functions applied to side channel cases.*

4.5 Discussion

4.5.1 Are Level Three Habitat Types Unique?

Morphologically distinct habitat types in the gravel reach of Fraser River showed modest predictability based on physical characteristics, particularly in the main channel where classification success ranged between 33% and 100%. The greatest discrimination between habitats was achieved along a hydraulic gradient of increasing velocity. Habitat types at the extreme ends of the hydraulic gradient (i.e., channel nooks/bays and bar heads/steep bar edges) were non-overlapping in physical character, thereby corresponding to the “patchy” model for habitat structure described by Wright and Li (2002). However, those habitats occupying intermediate positions along the hydraulic gradient shared some physical attributes. A particularly high degree of overlap occurred within groups of hydraulically “exposed” (bar head, steep bar edge), hydraulically “normal” (flat bar edge, bar tail, open nook), and hydraulically “sheltered” habitats (channel nook, bay, eddy pool), which reinforced the significance of a hydraulic gradient as the primary agent of physical organization in the gravel reach. Water depth and bank angle were associated with this hydraulic gradient and contributed to the discrimination of steep (e.g., eddy pool, channel nook, bay) and flat (e.g., open nook, bar tail) angled habitats. A sedimentary gradient, correlated with velocity and bank angle, also contributed to the discrimination of habitat types, but sedimentary attributes were secondary in importance to hydraulic attributes for habitat discrimination.

Whereas some habitat types were defined based on morphological identity, bar head, bar edge, and bar tail units were defined based on geographic position with respect to the gravel bar unit. Bar head and bar edge units showed consistent overlap in physical character, which upon further analysis by ordination was related to bank angle. The division of flat bar edge units ($<4^o$ bank angle) with low to moderate flow velocity and a mixed gravel-cobble substrate from steep bar edge units ($>4^o$ bank angle) with high flow velocity and predominantly cobble substrate greatly improved overall classification accuracy. Steep bar edge units shared sufficient physical similarity with bar heads to warrant grouping as a single habitat, and because field observations provided no basis to expect that the assemblage of fish species occupying the habitat differed, this grouping was justified.

The poor physical discrimination of bar tails was not surprising because, at flows exceeding 1500 $m^3 s^{-1}$, the “morphological” bar tail is submerged and the exposed bar tail in fact corresponds in position with mid-bar habitat. This association highlights the stage dependency of the habitat classification; units defined as bar tail at moderate discharge may have different physical characteristics than true bar tail units at low flow. The same stage-dependence problem applies to bar head units, for which geographic position did not yield distinct physical identity. In this way, the boundaries for bar head and bar tail units are moving targets along a continuum of hydraulic and sedimentary gradients that define habitat types in the gravel reach.

Sedimentary characteristics, in particular, will vary both laterally across a bar surface and longitudinally from bar head to tail. Non-systematic variability in sediment texture across bar surfaces is typically high as well (Wolcott and Church 1991). The pooling of sampling effort by season (i.e., autumn: 1500 to 5000 $m^3 s^{-1}$) may have confounded stage-related physical variability within some habitat types as units shifted laterally across the bar with changing water levels. This temporal influence on sedimentary characteristics of habitat types also may have contributed to their low classification accuracy. Over the autumn range in flows, Discriminant analysis was applied to all seasons combined, but yielded even lower classification accuracy, suggesting that characteristics of habitat types vary with stage as water levels shift laterally across the bar surface. This stage-dependence problem has not been identified previously in river habitat classification.

The degree to which a given habitat type is discriminated by the hydraulic gradient likely affects the stage-dependency of its classification accuracy. For instance, channel nooks and bays are readily discriminated from all other habitat types based mostly on negligible flow velocity, a condition that is unaffected by stage. In the same way, riffles and eddy pools are differentiated from other habitat types based on velocity and flow state (slow, back eddy flow and fast, turbulent/rough flow, respectively). The classification of these habitat types should be least sensitive to differences in stage. In contrast, habitats discriminated from one another mostly based on geographic position and sedimentary characteristics (bar heads, bar edges, bar tails) may demonstrate high stage-sensitivity in classification accuracy because of the tremendous variability in sediment texture across gravel bars. Any general longitudinal trend of decreasing substrate size from the head to the tail of bars is punctuated by locally varying sediment texture and sedimentation history, which are heterogeneous within the scale of habitat units.

4.5.2 Are Habitat Types Variable Among Sub-Reaches?

Morphological habitat units in the Chilliwack sub-reach were predicted with reasonable classification accuracy based on the physical characteristics of Rosedale habitat types, suggesting that variability between sub-reaches did not exceed within-reach variability. Both sub-reaches share similar morphology, consisting of a multi-thread channel with large island-bar complexes and persistent sediment aggradation. Classification success of Chilliwack units was even higher based on the hydraulic model for habitat classification, and the high similarity with Rosedale habitat attributes was consistent between the hydraulic and morphological models.

It is uncertain why Cheam sub-reach habitat units were less predictable than downstream sub-reaches; however, the morphology of the Cheam sub-reach is notably different. The sub-reach is more laterally confined and consists of a single, dominant channel with large mature islands. Moreover, the Cheam sub-reach is mildly degrading, in contrast to downstream sub-reaches, and has a steeper channel gradient. A steeper channel gradient influences sediment texture, the gradational tendency of the reach, hydraulics, and channel geometry, including bank angle, which was correlated significantly with several canonical roots in discriminant analysis. Bank angle was significantly higher for several habitat types in the Cheam sub-reach compared with other sub-reaches, including bar head, steep bar edge, flat bar edge, and bar tail units. Following this, the most striking difference in classification accuracy for the Cheam sub-reach was in the classification accuracy of bar head units (56%) compared with downstream sub-reaches. Open nooks and bays had low classification rates as well, and units were mistaken to be higher velocity habitats. From field observations, the presence of nook-like features was noted along the flanks of Cheam sub-reach gravel bars, which upon closer examination did not meet the functional definition of open nooks because flow velocity was relatively high.

The large-scale downstream physical gradients of decreasing channel slope and lateral confinement, encompassed in Level 1 sub-reaches, appeared to influence the character of small-scale Level 3 habitat units but only to a modest extent between the degrading Cheam sub-reach and aggrading Rosedale and Chilliwack sub-reaches. Hence, habitat characteristics for the gravel reach of Fraser River show reasonable spatial integrity in the longitudinal dimension.

4.5.3 Are Habitat Types Variable Among Channel Types?

Morphological habitat units in secondary channels had poor discrimination both by jackknife and *a priori* classification methods. Habitat predictability for side channels units was slightly better than for summer channels, suggesting that the discrimination between habitat types may be related to the stability and maturity of secondary channels, as well as their sedimentation history. Summer channels, in particular, have lower flow conveyance, inundation frequency, and duration. They are worked less frequently by flow, which means that less sediment sorting and redistribution can take place during any given flow event. Moreover, the range of sediment sizes conveyed in side and summer channels is small (Ellis 2004) compared with the main channel, so the contribution made by a sedimentary gradient to the discrimination of habitat types is reduced. The hydraulic gradient, which primarily differentiated habitat types in the main channel, may also be reduced in secondary channels as differences in hydraulic characteristics among units become small at high flow.

These results imply that neither the patchy nor gradient-based models for habitat structure presented by Wright and Li (2002) adequately characterize habitat structure in secondary channels and that the spatial integrity of the morphological habitat classification extends laterally only to a limited degree from the main channel to secondary channels.

4.5.4 Bar Morphology and Processes of Habitat Development

Habitat types in the gravel reach of Fraser River have been identified at a spatial scale intended to be ecologically relevant to juvenile fish, on the order of 10^1-10^2 m in length. The physical processes responsible for habitat creation, however, occur over larger spatial scales extending from the gravel bar to the entire river basin. Elucidating these processes and characterizing the geomorphologic genesis of habitat structure in the gravel reach is the ultimate goal for understanding habitat development. The starting point towards this end is to determine whether or not fluvial processes and associated sedimentary features give rise to consistently identifiable morphologies at the scale of 10^1-10^2 m that represent distinct habitat units for fish. This objective represents the final contribution of this chapter.

Gravel bar units represent the most useful scale for examination because large-scale fluvial processes of sediment deposition and erosion preserve themselves as signature features on bar surfaces. These features generally correspond with sedimentary units of relatively uniform grain texture, and collectively represent the building blocks of complex bar morphology and fish habitat. Several fluvial sedimentary features have been described previously (Chapter 2), and are broadly categorized as gravel sheets, gravel lobes, and chute-lobe couplets. Variable expressions of these features occur along the gravel reach that, together, produce topographically complex bar surfaces and appear to offer high habitat diversity over a range of water levels.

Sedimentary features are best examined on exposed bar surfaces at low flow; however, they develop and are functionally important as fish habitat at higher stages. Gravel lobes deposited as accretionary wings typically have a shallow sloping outer edge of well-sorted gravel and a steeply sloped inside edge, often resembling an avalanche face (see inset, **Figure 4-29**). Over a range of water levels, this feature is a nucleus for habitat diversity: quiet bay habitat is available in the lee of the gravel lobe along the inside edge, flat bar edge or bar tail habitat develops along its outer edge, a riffle extends downstream, and a large circulating eddy pool may develop off the bar tail. Back eddy flow extending off the bar tail may scour a deep hole on the inside edge of the riffle and cause fine sediment deposition in the bay, producing a heavily embedded substrate. Many examples of this habitat-complex exist in the gravel reach, perhaps the most notable being Queens Bar (**Figure 4-29**), but also including Calamity Bar (**Figure 4-30a**), Gill Island, Spring Bar, and Peters Island (**Figure 4-31a**).

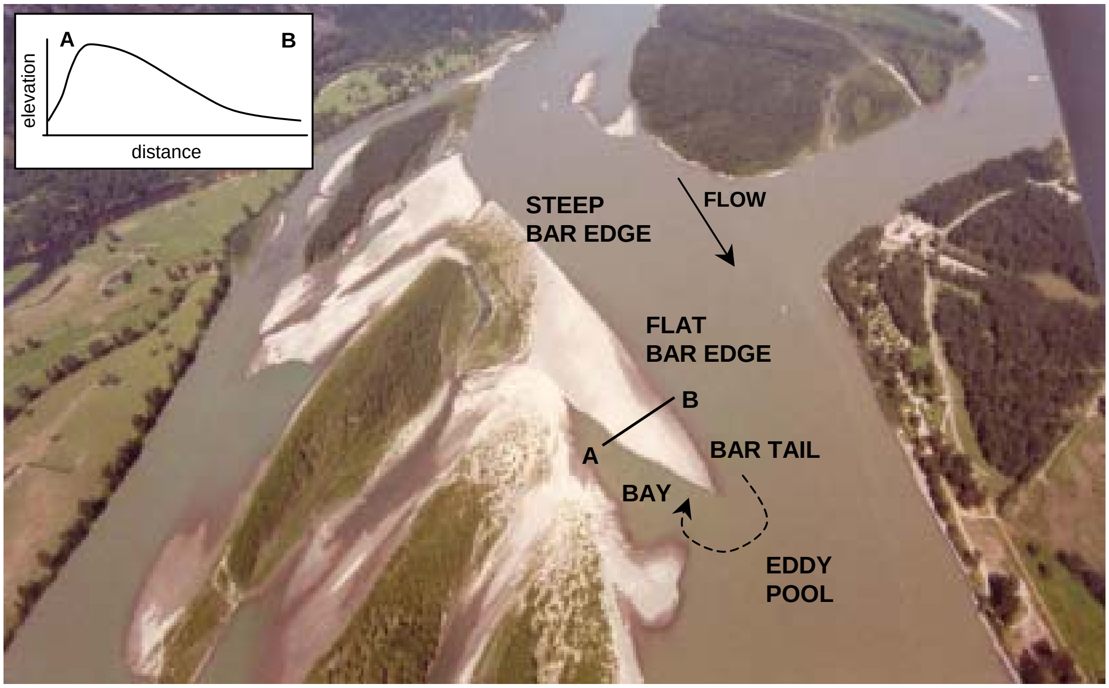


Figure 4-29. Habitats associated with an accretionary sediment lobe at Queens Bar: bar edge, bar tail, eddy pool, and bay habitat units (photo courtesy of V. Galay, August 12, 2002; 2971 $m^3 s^{-1}$). Inset represents the cross-sectional shape of the sediment lobe.

Channel nooks may develop in the same way as bays, with a small gravel lobe extending off a bar flank and creating sheltered habitat along its inside edge (**Figure 4-30a**). In such cases, channel nooks are morphologically similar to bays, differing mostly in scale. The complex of associated habitats (e.g., riffle, eddy pool) is sometimes, but not always present, in part because of the smaller size of the sedimentary feature. Alternatively, channel nooks may form at the inlet or outlet of a secondary channel, being flow-through during freshet and providing sheltered habitat at lower water levels (**Figure 4-30b**). This variation in genesis among channel nooks may have contributed to its relatively low jackknife classification rate; in some cases channel nooks may be physically similar to bays but smaller in size, whereas other channel nooks may be more similar to hydraulically exposed habitats with respect to bank angle, depth, and substrate composition, only with negligible velocity.

Bar surfaces typically consist of many overlapping gravel sheets of varying age and spatial extent. Some sheets become stalled on the elevated, central rib of a bar with the leading edges of the sheet being stretched downstream to create a dune-shaped deposit. The leading edge of high, advancing platforms typically is steep and, depending on river stage, quiet bay habitat may develop in the lee of the dune-shaped feature (**Figure 4-30a**). A sequence of overlapping sheets creates stair-step topography that, when inundated, produces a scalloped water line. This represents a third process by which channel nooks may form; where a recently deposited sheet has substantially greater relief than the underlying surface and produces a deeply scalloped and sheltered unit (**Figure 4-30c**). Open nooks may develop by this process as well but occupying more subtle irregularities along a bar edge where multiple sheets coalesce (**Figure 4-31**). In this case, the sheets have low differential relief and the associated nooks are relatively transient on the rising or falling limb of the hydrograph. Even a slight change in stage may create or eliminate open nook units along a scalloped bar flank and thereby prompt reclassification of the unit. Not surprising, open nooks were most commonly misclassified as flat bar edge units. Whereas most habitat types are associated with major features, open nooks are relatively transient, occupying topographical irregularities at the interface of adjacent sediment sheets or exploiting undulating topography of a single sheet (**Figure 4-30a, 30c**).

The leading edge of gravel sheets may produce riffle fronts extending into the channel. Eddy pool habitat may develop in association with the riffle, depending on flow conditions. Such features usually develop at the head of gravel bars with the gravel sheet consisting of relatively coarse substrate (**Figure 4-30b**). Selective sorting by hydraulic forces contributes to the coarsening of bar head environments with finer sediment being transported downstream. Differences in sediment texture between a bar head and bar tail depend on a pronounced hydraulic gradient and selective sediment transport processes. If this gradient is not strong, or selective transport processes are punctuated by local discontinuities in sediment texture associated with coalescing gravel sheets, then the distinction of bar head and bar tail habitats from bar edge will be weak.

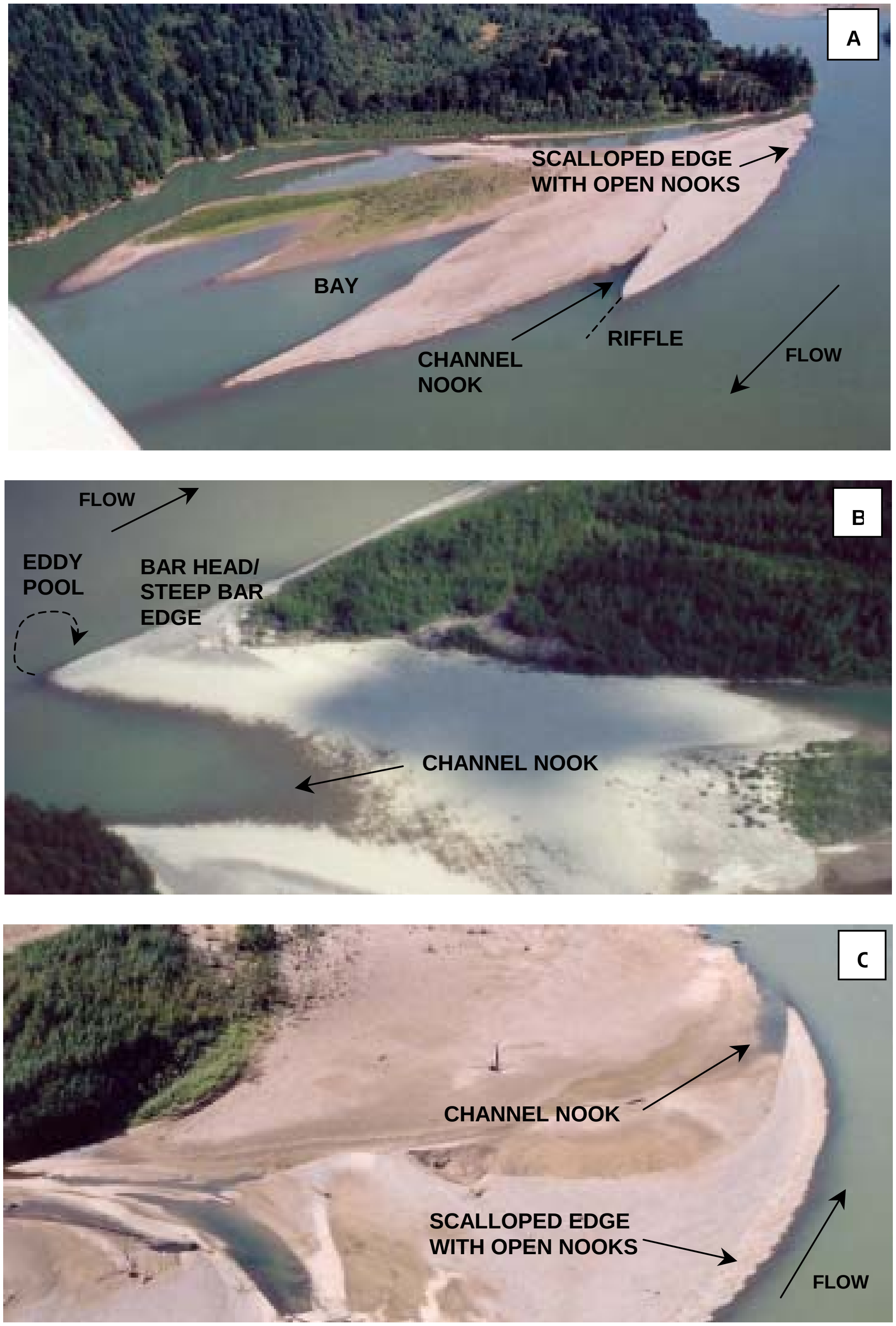


Figure 4-30. Morphological genesis of channel nooks: A) In the lee of a small gravel lobe (upper photo, Calamity Bar, September 20, 2002; 1880 $m^3 s^{-1}$); B) At the inlet to a dry secondary channel (middle photo, Seabird Island, August 20, 2001; 3340 $m^3 s^{-1}$); C) Along the contact between an over-riding gravel sheet and a lower elevation surface (lower photo, Harrison Bar, September 20, 2002; 1880 $m^3 s^{-1}$).

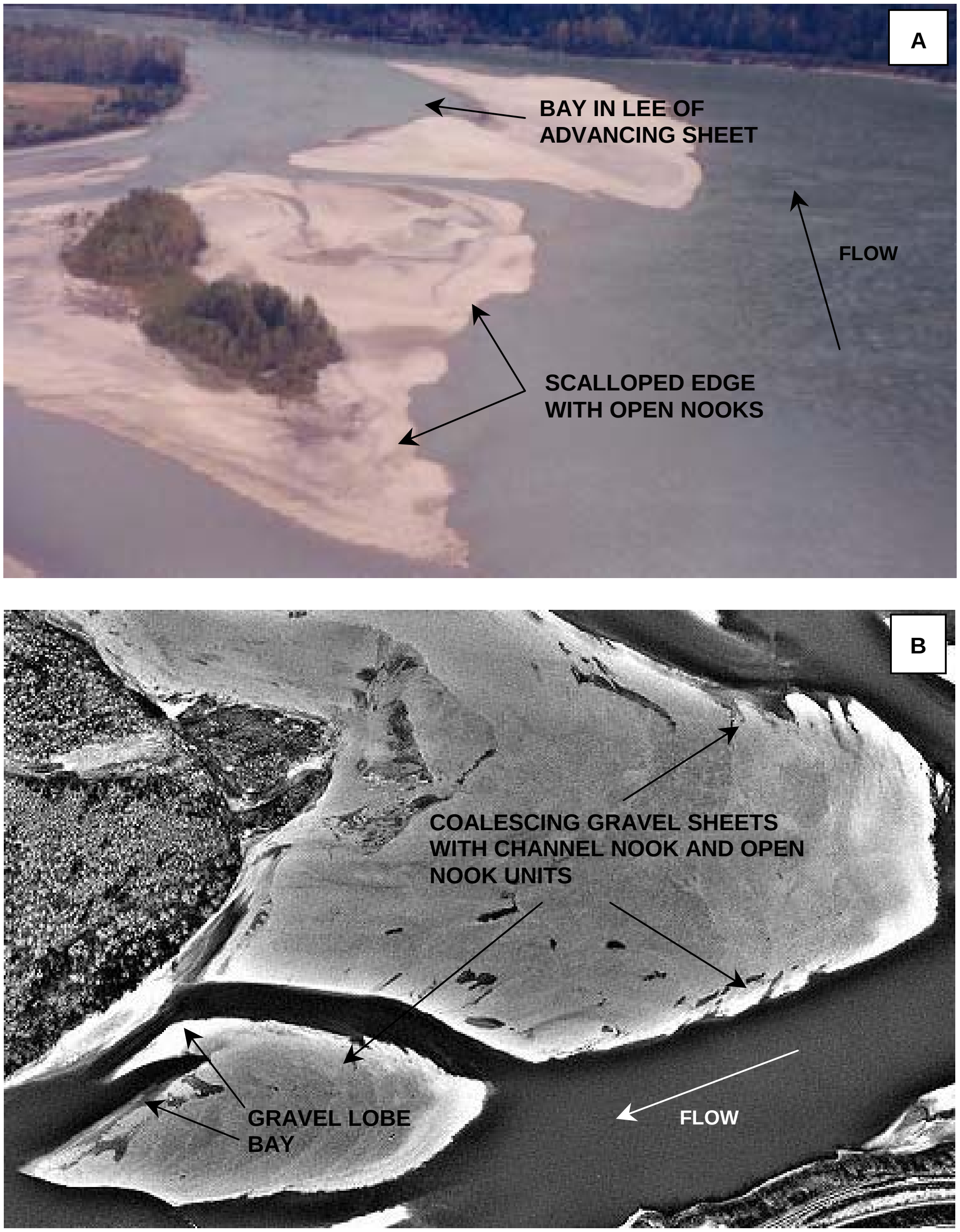


Figure 4-31. Morphological genesis of hydraulically sheltered units: A) low relief gravel sheets coalescing to produce a scalloped water edge with hydraulically sheltered open nook and channel nook units (upper photo, Peters Island, September 20, 2001; 1560 $m^3 s^{-1}$; lower photo, Spring Bar, March 10, 2000; 678 $m^3 s^{-1}$). Bay habitat develops in the lee of high, advancing gravel sheets.

4.5.5 Summary

Alluvial habitat types in the gravel reach of Fraser River possess a distinct combination of physical attributes known to influence the distribution and abundance of fish and other aquatic organisms. Although some overlap in physical character exists among habitat types, morphological units were predicted with reasonable accuracy (averaging 58% in the main channel). Previous studies classifying pool, run, and riffle habitats in small streams considered classification accuracy between 39% and 100% acceptable (Jowett 1993, Peterson and Rabeni 2001a). Habitat structure in each of these systems likely was much simpler than the physical complexity presented in the gravel reach. It is assumed that Fraser River habitat types were identified correctly in the field because the writer both developed the classification scheme and carried out all habitat typing over the course of the study. Hence, misclassifications by the jackknife method probably were due to variations in the physical characteristics of the habitats themselves. The overlap among several habitat types revealed by graphical analysis confirms this variability.

The simplified hydraulic habitat classification, whereby morphological habitat types were grouped according to hydraulic character, has practical advantages for field application. Previous investigations have indicated that field personnel often misclassify habitat units, resulting in biased, unrepeatable, and/or imprecise estimates of habitat availability (Roper and Scarnecchia 1995). The most simplified classification possible is recommended by Roper and Scarnecchia (1995) to reduce observer bias and error likelihood when applying a classification scheme as a tool in habitat assessment. The results of this study suggest that grouping habitat types according to three classes of hydraulic habitat improved classification accuracy substantially, particularly in the main channel. However, the “multivariate” morphological model for habitat classification integrates information on the fluvial processes involved in habitat development, which is otherwise lost with application of the hydraulic model. Instead, the simplified hydraulic classification incorporates considerable physical heterogeneity into estimates of habitat availability with a major loss in information about the morphological complexity of the reach. This has disadvantages from the perspective of habitat assessment and is problematic if fish species exhibit a preference for one or a limited number of morphological habitat types. This topic will be evaluated in Chapter 6.

Chapter 5. Habitat Associations of Benthic Invertebrates

5.1 Introduction

The preceding chapter examined the physical distinctiveness of seven morphological habitat types that are ubiquitous in the gravel reach of Fraser River and occur at a spatial scale intended to be relevant to aquatic organisms, particularly juvenile fish. Confirming the physical distinctiveness of habitat types is fundamental to the utility of any habitat classification and is a prerequisite to the expectation that habitat types have ecological importance for aquatic organisms. Of the original 7 morphological habitat types considered, the definitions of 5 remained unchanged based on discriminant analyses (bar tail, open nook, eddy pool, channel nook, bay), and the definitions of 2 habitat types were revised to reflect patterns of physical distinctiveness: bar edge units were split according to bank angle (*flat* units $< 4^{o}$ and *steep* units $\geq 4^{o}$ bank angle), and bar head units were merged with steep bar edge units. For simplicity, the term bar head is used from this point forward, in reference to the combined group of bar head and steep bar edge units.

The revised morphological habitats were discriminated with reasonable predictive accuracy based on hydraulic and sedimentary attributes in three sub-reaches of Fraser River. Accordingly, the habitat types may have ecological importance to aquatic organisms because hydraulic and sedimentary factors are known to influence the distribution and abundances of benthic invertebrates (Quinn and Hickey 1994, Rempel *et al.* 2000, Rice and Greenwood 2001, Rabeni *et al.* 2002) and fish (Rosenfeld *et al.* 2000, Peterson and Rabeni 2001b). Alternatively, a more simplified classification that differentiates three habitat types on the basis of hydraulic character (exposed, normal, sheltered) may more clearly discriminate the distribution of aquatic organisms. The following two chapters evaluate the ecological merit of these alternative models for habitat classification by examining the association of habitat units with macroinvertebrate (Chapter 5) and fish (Chapter 6) assemblages in Fraser River. The material is divided into two chapters for convenience, but the chapters share a common set of objectives and analytical methods.

5.1.1 Background

Studies relating habitat attributes to the distribution and abundances of benthic invertebrates have been carried out at many spatial scales. Invertebrate distributions are more convenient for study than those of fish because invertebrates are considerably less mobile and, consequently, it is reasonable to suppose that the habitat in which they are collected represents favourable conditions for residency. The focus of many studies has been to examine the relative importance of different physical factors in structuring the benthic community. Hydraulic and substrate conditions have been identified consistently as dominant factors that affect community composition and the abundances and distributions of the constituent populations (Wetmore *et al.* 1990, Cobb *et al.* 1992, Growns and Davis 1994, Quinn and Hickey 1994, Rempel *et al.* 2000). Other factors also have been shown to play important roles, including resource availability (Culp *et al.* 1983, Richardson 1991), water chemistry (Leland and Ford 1998), temperature (Hawkins *et al.* 1997), as well as the overall degree of environmental heterogeneity (Scarsbrook and Townsend 1993, Lancaster and Belyea 1997, Palmer *et al.* 1997) and hydraulic variability (Poff and Ward 1989, Blinn *et al.* 1995) that characterize the system.

The influence of physical habitat characteristics on benthic invertebrates is believed to be sufficiently strong that invertebrate distributions have formed the basis for habitat classifications in some river systems (Buffagni *et al.* 2000, Rabeni *et al.* 2002, Heino *et al.* 2003). Such a relation has clear advantages for promoting habitat classification as a useful tool in habitat inventory and bioassessment. In the study by Buffagni *et al*. (2000), “potential habitats” in a relatively unmodified Italian stream first were identified as visually distinct habitat types, and “functional habitats” then were distinguished as those habitats supporting different assemblages of invertebrates. Although the definition is based on structural attributes of the invertebrate community, the term “functional habitats” is intended to recognize the collective contribution and importance of habitat units to supporting ecosystem processes, productivity, and biotic diversity in streams (Kemp *et al.* 1999). The “functional habitats” approach also helps to address a major criticism of habitat classification: that it is overly simplistic to expect physical attributes and the taxonomic assemblage of river systems to be organized as predictable and distinct units.

The question of whether biotic communities exist as discrete units corresponding to habitat features, or as a continuum of gradually changing taxonomic composition has been a topic of debate in community ecology for several decades (McIntosh 1995). Shipley and Keddy (1987), and more recently Wright and Li (2002), demonstrated that the two alternatives are not necessarily mutually exclusive and, in fact, some combination of the two models is most realistic for stream communities. This is, in part, because communities are made up of many species, each with particular habitat preferences that may be highly specialized or relatively indiscriminate, and some of whose habitat preferences change with age. This also is because both discrete and continuously varied physical processes, operating at multiple spatial and temporal scales, influence habitat structure in streams, which in turn influence the structure of aquatic communities (Pringle *et al.* 1988, Wiley *et al.* 1997). An extension of the debate has arisen from authors arguing that it is not as much the structure of the habitat, but the degree of spatial heterogeneity and temporal variability characterizing the system, which mostly influence community structure (Death and Winterbourn 1995, Palmer *et al.* 1997).

River systems such as Fraser are inherently variable, both in space and time, which raises an important issue for habitat classification and its application in biological inventory and assessment. The utility of habitat classification stems from an expectation that species assemblages occur as predictable, discrete entities associated with identifiable habitat types. For this expectation to hold, animals should have relatively specialized habitat requirements that restrict their distribution to a single, or limited number of habitats. Such a community, however, is apt not to occur in spatially or temporally heterogeneous environments (Townsend 1989). Moreover, such a community is predicted to have low resilience to physical disturbance resulting in habitat change or loss (Southwood 1988, Poff and Ward 1990), because animals will not have the means to adapt to the new system configuration, and the low spatial heterogeneity will make no provision for spatial refugia to lessen the impacts of disturbance (Scarsbrook and Townsend 1993). In this context, the strength of the association between habitat structure and species' distributions may represent an inverse measure of resilience to disturbance. Environments subject to disturbance, or characterized by high spatial heterogeneity and poor habitat zonation, are predicted to favour animals with indiscriminate habitat associations and generally widespread distributions (Scarsbrook and Townsend 1993).

Within the gravel reach of Fraser River, the integrated influence of velocity, depth, and sediment texture, encompassed in the distinctiveness of morphological habitat types, is hypothesized to govern the spatial distribution of invertebrate taxa and species of fish. Alternatively, the dominant hydraulic gradient that defines hydraulic habitat classes may more clearly discriminate the distribution of organisms. These hypotheses are exclusive, but not exclusive of the possibility that other biological (e.g., species-specific interactions) or environmental factors operating at a different spatial scale (e.g., smaller patch-scale) may more strongly govern animal distributions. For instance, invertebrates may key on habitat attributes scaled more appropriately with body size, and which are expressed heterogeneously at the scale of habitat units. These hypotheses also are not exclusive to the possibility that the spatial heterogeneity and pronounced hydrologic cycle of Fraser River favour a community with weak habitat associations and widespread distributions. Whether or not the spatial organization of habitat units around gravel bars corresponds with the distribution of invertebrates will be evaluated in this chapter.

5.1.2 Objectives

The chapter has three objectives. The first and most basic objective is to describe the faunal composition of invertebrate assemblages occupying physically defined habitat units in the gravel reach of Fraser River. The second objective is to determine whether or not the invertebrate assemblages associated with habitat types are consistent and if the spatial distribution of animals corresponds either with the morphological or hydraulic habitat classification. The morphological and hydraulic models were compared based on which model maximizes within-habitat taxonomic similarity and between-habitat dissimilarity. The final objective is to compare the habitat-specific species assemblage among channel types and sub-reaches to determine if patterns are consistent in the lateral and longitudinal dimensions.

5.2 Data Analysis

The Surber net used for invertebrate sampling is ineffective in habitats of negligible velocity (i.e., all "sheltered" habitats: eddy pool, channel nook, bay). As a result, only four habitat types were sampled between September 1999 and September 2001: riffle, bar head, bar edge, and bar tail. (All sampled bar edge units had a flat bank angle $< 4^{o}$.) Open nooks were sampled in September 1999 but the low velocity, typical of open nooks, was found to compromise the effectiveness of the net. Using a single sampling method in a limited number of habitat types was preferred over multiple methods to sample in all habitat types because sampling techniques vary in efficiency and performance; data standardization would be required in order to carry out comparative analyses. Within the four sampled habitat types, sampling effort was not stratified equally because differences in water level between sampling episodes affected habitat availability, particularly riffles, which were sampled in only 3 of 10 episodes.

A total of 356 samples, collected from the main ($n = 300$) and side ($n = 56$) channels in the Cheam, Rosedale, and Chilliwack sub-reaches, were included in analyses. An additional 129 samples were excluded because they were collected from sites where gravel mining had taken place in the recent past. The majority of analyzed samples were collected from *main channel habitat units in the Rosedale sub-reach* ($n = 176$), and it is with this core dataset that the first two chapter objectives were evaluated. Because sampling effort was considerably greater in the main channel compared to side channels, emphasis was placed on contrasts in the invertebrate assemblage among habitats and sub-reaches rather than among channel types.

The univoltine life cycle of many invertebrate taxa in the gravel reach, which hatch as aquatic larvae in early autumn and emerge as terrestrial adults in spring (Dymond 1998), was a significant determinant of abundances over time. Hence, all analyses included time as an independent factor and proceeded systematically to examine habitat-specific patterns along a scale of increasing temporal resolution: 1) all sampling episodes pooled, 2) all episodes within a single season, and 3) a single episode. A combination of univariate and multivariate approaches was used, each having particular merits that collectively provided a comprehensive evaluation of chapter objectives. Graphical comparisons and univariate statistical contrasts were used to evaluate changes in community-based summary metrics (e.g., total density) over time and between habitats. Multivariate methods were used to characterize temporal changes in invertebrate community structure and to evaluate the similarity/dissimilarity in community patterns among habitat types, as well as to identify factors influencing invertebrate assemblages.

5.2.1 Univariate Analysis of Community Metrics

Seven community metrics were calculated based on invertebrate samples. Several different richness and diversity indices were examined because each differs in the weight placed on common and rare taxa. Formulae are from Krebs (1998).

1. **Total Density (N)**: total number of benthic invertebrates in a Surber sample divided by sampling area. Sampling area was 0.09 m^2 for all invertebrate samples.
2. **Taxon Richness (*S*)**: total number of unique taxonomic groups in a sample.
3. **EPT Richness (*S'*)**: total number of unique taxonomic groups belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera (mayflies, stoneflies, caddisflies).
4. **Margalef's Taxon Richness (*d*)**: $d = (S-1) / \mathrm{Log}(A)$ where S is the number of unique taxa in the sample and A is the number of individuals. The metric is similar to taxon richness but weighted by the number of animals collected in the sample.
5. **Simpson's Diversity (*D'*)**: $D' = 1 - D = 1 - \sum_{i=1}^{S} P_i^2$ where P is the proportion of individuals of the i^{th} species. D refers to Simpson's Index. The index places more weight on common species and reflects the probability of picking 2 invertebrates at random that are different taxa. Values range from 0 to 1.

6. **Shannon-Wiener Diversity (*H'*)**: $H' = -\sum_{i=1}^{S} P_i \times Log_2 P_i$ where P is the proportion of individuals of the i^{th} species. The index is more sensitive to rare species in the sample compared to Simpson's diversity.
7. **Pielou's Evenness (*J'*):** $J' = H' / Log\ (S)$, derived from the Shannon-Wiener index to express observed diversity as a proportion of the maximum possible diversity. *J'* approaches 1 as the proportions of all taxa in the sample approach equality.

Community metrics are presented graphically to examine temporal changes between sampling episodes. Single episodes then were chosen to examine spatial differences between habitat types and sub-reaches. Two-factor analysis of variance (ANOVA) was applied to five community metrics based on these data to examine habitat and sub-reach effects. The two excluded metrics, taxon richness and EPT richness, had distributions that could not be normalized by transformation. Invertebrate density was log ($X + 0.5$) transformed, and other variables did not require transformation to meet assumptions of normality and homogeneity of variances. A significance value of $\alpha = 0.01$ was applied, after Bonferroni's correction for multiple contrasts (0.05 / 5 metrics). Pairwise contrasts were made by Tukey's Test for unequal sample sizes.

5.2.2 Multivariate Analysis of Community Structure

Non-metric Multi-Dimensional Scaling (MDS) was chosen for multivariate analyses based on several attributes. MDS is a complex numerical algorithm, but is conceptually simple and makes a relatively transparent link between the original data and output, which is derived from ranked similarities of the data. The objective of the technique is to achieve a low-dimensional ordination map of the samples such that the ecological distances among samples (i.e., similarity) are preserved as rank distances in ordination space. MDS has greater flexibility in the definition of dissimilarity compared to principal components analysis (PCA), the latter using Euclidean distance which is not suited to abundance data in which zeros are common (Clarke 1993). Moreover, it is superior at preserving ranked dissimilarities as distances between samples in a 2-dimensional ordination plot. The non-parametric approach eliminates statistical assumptions about the distribution of the data, such as normality, and relaxes requirements for a balanced sampling design when comparing between groups. A limitation of MDS is that it has no capacity to classify new samples in a predictive mode; all samples are ordinated simultaneously and the MDS solution is unique to the samples included.

MDS was introduced by Shepard (1962) and Kruskal (1964) for application in psychology, and was shown by Minchin (1987) and Clarke (1993) to be a robust technique for analyzing ecological data. MDS is now commonly used to examine patterns in stream invertebrate (Hawkins *et al.* 1997, Wright and Li 2002, Heino *et al.* 2003) and fish (McCormick *et al.* 2000, Walters *et al.* 2003) communities. The lag time between its development and popular use is attributed to the high computational demand of the algorithm (Clarke 1993). The analytical procedure is described in moderate detail below, because of its relatively recent popularity.

Analyses were carried out on invertebrate groups identified to the lowest taxonomic level (usually genus, refer to Chapter 3), with the expectation that higher taxonomic resolution may improve discrimination between habitat types (Waite *et al.* 2004). Including a large number of taxa does not reduce the discriminatory power of MDS, but may improve the resolution to detect differences among habitats. All taxa occurring in greater than 2 samples were included. Exploratory analyses were carried out on taxa groups classified to family, as well as using a more stringent cut-off for genus-based taxon representation (e.g., occurring in >5 samples or representing >1% of total abundance), and results were similar. Overall, 58 unique taxa were identified and analyses were based on 36 groups representing several orders and classes: Ephemeroptera (8 groups), Plecoptera (9), Trichoptera (4), Homoptera (1), Diptera (9), Oligochaeta (2), and Acarina, Bivalvia, and Nematoda.

All multivariate analyses were carried out using PRIMER-e software (version 5.2 Clarke and Gorley 2001). **Figure 5-1** is a schematic diagram of the analytical framework followed. The starting point was calculation of a similarity matrix of samples using the Bray-Curtis coefficient, which measures the similarity in abundance between samples for each species, averaged over all species (Clarke and Warwick 1994). Taxon abundances first were averaged across replicate samples (usually n = 3) from each sampling location, and then square-root transformed to down-weight the influence of highly abundant taxa (Clarke and Gorley 2001). Hierarchical group average cluster analysis was carried out on the rank similarities to provide a visual display of the similarity among samples in the form of a dendrogram. The clusters can be useful as a complement to MDS ordination for highlighting structure and irregularities in community patterns (Clarke and Warwick 1994).

MDS ordination plots illustrate the relation in community structure among samples by arranging the samples in multi-dimensional space such that the rank order of the similarities among samples is maintained; the axes of the plots therefore have no units but the scaling distance is proportional to the ecological distance (i.e., similarity) between samples. MDS plots are interpreted by comparing the relative spacing between samples. Those samples positioned closely, or grouped together in an isolated cluster, are most similar in community structure. Samples spaced far apart in the MDS plot have high dissimilarity.

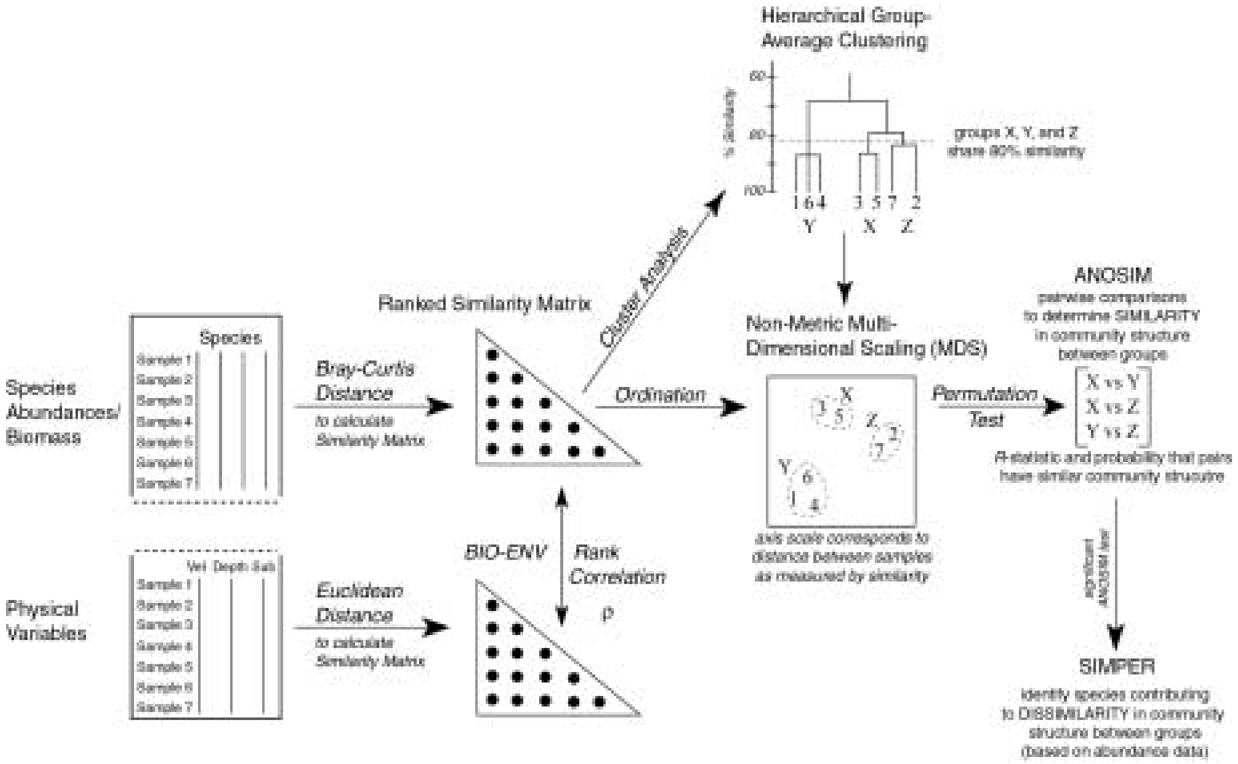


Figure 5-1. Schematic diagram of the procedure followed for multivariate analysis of invertebrate and juvenile fish data (modified from Clarke and Warwick 1994). Refer to text for details.

The MDS algorithm works on an iterative cycle by repeatedly arranging the samples into positions so that distances between samples match as closely as possible the rank order of the original similarity matrix. The extent to which the algorithm succeeds in this process is measured by the stress coefficient, calculated using Kruskal's stress formula, whereby stress tends towards zero as the rank orders reach perfect agreement. The stress value therefore measures the distortion between the similarity rankings and the corresponding distance rankings in the ordination plot. Stress values less than 0.1 correspond with an excellent representation in 2-dimensional space and values less than 0.2 provide a good representation, particularly for values approaching 0 (Clarke 1993). If the stress level of the 2-D plot exceeds 0.2, it should be interpreted in combination with the 3-D plot. All MDS analyses were run at least 30 times to ensure that a global minimum of the stress function had been reached. In some cases, clusters from group-average agglomerative cluster analysis were superimposed on the MDS plots to highlight structure in plots and quantify the degree of similarity between groups.

Statistical differences in community structure between groups (e.g., sampling episodes, habitats, sub-reaches) were evaluated using the ANOSIM ('Analysis of Similarities') permutation test, which is applied to the rank similarity matrix (Clarke 1993). ANOSIM compares the average rank of similarities among samples *between groups* (i.e., habitat type) with similarities from samples *within a group*. The null hypothesis is that there are no differences in community composition between groups. The test statistic, R, is the difference between the average ranks of similarities between and within groups. R is calculated from corresponding rank similarities between samples in the underlying similarity matrix whereby

$$R = (\acute{r}_B - \acute{r}_W) / (M / 2)$$

and $\acute{r}_B$ is the average of rank similarities from all samples *between* different groups, $\acute{r}_W$ is the average of all rank similarities among samples *within* groups, and

$$M = n(n-1) / 2$$

where n is the total number of samples. R approaches zero if the null hypothesis is true, indicating that similarities between groups are the same, on average. It is important to note that R may be significantly different from zero yet very small; it is therefore more useful to evaluate the statistical significance of R, rather than its actual value.

The statistical significance of R is determined by a continuous resubstitution procedure in which the group identity of each sample is randomly switched. For example, a bar tail sample is reclassified randomly as riffle, and a bar head sample is reclassified randomly as flat bar edge, and so on. The premise of the permutation test is that, following the null hypothesis of no difference between groups, there will be negligible effect on the value of R if the labels identifying which replicates belong to which groups are arbitrarily rearranged. In general there are $(kn)! / [(n!)^k k!]$ distinct ways of permuting labels for n replicates in each of k groups. The full set of permutations (typically very large) is randomly sampled up to a maximum of 999 times (with replacement) to give the null distribution of R. The significance level is determined by referring the observed R-value to its permutation distribution. If the observed R appears unlikely to have come from this distribution, then there is evidence to reject the null hypothesis. The procedure was run first as a "global" test to determine if there are group differences in the dataset worthy of further examination. Pairwise comparisons between groups (i.e., habitats) then were made by extraction and re-ranking of the similarities from habitats of interest, and repeating the permutation test. Due to the exploratory nature of the analysis, an adjusted significance value of $\alpha = 0.1$ was applied.

In cases where the analysis included two independent factors, such as habitat types sampled in multiple episodes, two-way ANOSIM was applied. This is an extension of one-way ANOSIM, whereby the randomization procedure was constrained to control for possible dissimilarity between groups of one factor, while evaluating the dissimilarity between groups of the second factor. For example, the dissimilarity between habitat types sampled in multiple episodes was evaluated by constraining the analysis to compare habitats within each sampling episode separately, and then averaging results across episodes to calculate the test statistic, R. The continuous resubstitution procedure to derive R is constrained, such that labels are randomly switched across habitat types within each episode, but labels are not interchanged between episodes.

Significant dissimilarities between groups, as revealed by ANOSIM, prompted analysis by SIMPER (‘Similarity in Percentages’) to determine which taxa contributed most to the overall dissimilarity between groups. SIMPER is an exploratory procedure based on the original abundance data, in contrast to ANOSIM that is based on the Bray-Curtis similarity matrix. SIMPER compares the percentage composition each species made to the average *similarity within groups*, and the percentage contribution each species made to the average *dissimilarity between two groups*. The goal is to identify which taxa contributed most to the dissimilarity between groups, as originally portrayed in the MDS plot and subsequently confirmed by ANOSIM. Abundance data first were square-root transformed to remove excessive bias of the most abundant taxa (Clarke and Warwick 1994). In one analysis (all episodes pooled), the fourth-root transformation was used because of large differences in abundance between sampling episodes.

Finally, the BIO-ENV procedure of PRIMER-e was used to evaluate the influence of physical factors (velocity, water depth, and substrate size class) on community patterns. A similarity matrix was produced from normal-transformed environmental variables using the normalized Euclidean Distance. The BIO-ENV procedure calculates the correlation between the environmental and the benthic invertebrate similarity matrices following methods outlined in Clarke and Ainsworth (1993). This approach was chosen because it first analyzes the biological data by MDS and then evaluates how well the structure of environmental variables matches the revealed community structure. The non-parametric procedure determines which combination of environmental variables has a multivariate pattern that best fits the multivariate pattern of the biological data. The fit is measured by the ρ statistic, a rank correlation of the two underlying similarity matrices, and a value approaching 1 indicates that the biological and the environmental matrices have an exact fit to each other (Dr. K. R. Clarke, Primer-E Ltd., pers. comm.). The procedure seeks to identify the subset of environmental variables that maximizes the fit, and the value of ρ degrades with the inclusion of irrelevant variables. The approach differs from other methods such as canonical correlation (e.g., Rempel *et al.* 2000) and canonical correspondence (Ter Braak 1986), which embed the environmental data within the biological analysis to identify specific gradients defining the species-environment relation. Also, the approach is more appropriate because of the limited number of measured physical variables.

5.3 Results

5.3.1 Univariate Analysis of Community Metrics

A total of 130,733 invertebrates were collected in 300 samples between September 1999 and September 2001 from main channel habitats. Fourteen invertebrate taxa were relatively common, each representing at least 2% of invertebrates collected during any one sampling episode and, together, representing between 96% and almost 100% of all invertebrates (**Table 5-1**). Orthocladiinae, a sub-family of the family Chironomidae, was most abundant in all months except April 2000 when the mayfly *Ameletus* sp. was proportionately more common. Another sub-family of Chironomidae, Chironominae, also was abundant in virtually all months of sampling. There was high variability in the proportional representation of some taxa between consecutively sampled months, irrespective of season.

Total invertebrate density fluctuated substantially between sampling episodes (**Figure 5-2**) and was highest in March of each year. September density was variable in all three years of sampling and river discharge in September of each year differed as well. Measures of diversity showed similar temporal variability between sampling episodes (**Figure 5-2**) and fluctuations were correlated negatively with density. Simpson's diversity, which places more weight on common taxa, was correlated more strongly with density (r = -0.3) than Shannon-Wiener diversity (r = -0.17), which gives more weight to rare taxa. Taxon richness and EPT richness were highly correlated with density (r = 0.68 and 0.67, respectively) as well as with each other (r = 0.94), indicating that the collective sum of mayflies, stoneflies, and caddisflies in samples was changing over time in a similar manner to the entire community. Margalef's richness, which weights the number of taxa in samples by the total abundance, was positively correlated with density (r = 0.37), but to a lesser extent than its non-weighted counterparts (**Figure 5-2**). November 2000 was the single month in which all measures of richness and diversity were high. In other months, high values of richness corresponded to low values of diversity, and vice versa. Pielou's evenness was negatively correlated with density (r = -0.55) and values were generally low in winter and high in summer and autumn (**Figure 5-2**). The one exception was August 2000, when evenness was relatively low.

Two-factor ANOVA applied to September 1999 data and examining habitat and sub-reach effects on community metrics indicated a significant difference among sub-reaches in diversity, evenness, and richness, but not in total density (**Table 5-2**). The Rosedale sub-reach had significantly higher diversity than either the Chilliwack or Cheam sub-reaches (**Figure 5-3**), and significantly higher richness and evenness than the Cheam sub-reach. No detectable difference among habitat types was found for any parameter.

March 2000 was a second month in which sufficient data were collected for graphical comparisons between habitats and sub-reaches (ANOVA was not applied due to unequal sampling effort between groups). Density, averaged over sub-reaches, was similar among the four habitat types (**Figure 5-4**). The Rosedale sub-reach had higher density in riffle and bar tail habitats, and average density in bar heads of the Chilliwack sub-reach was higher than upstream sub-reaches. Simpson's diversity was substantially higher in all sampled habitats of the Cheam sub-reach compared to downstream reaches, unlike in September 1999.

Table 5-1. Average percent (%) representation of the fourteen most common invertebrate taxa in samples collected from the main channel of the gravel reach in ten sampling episodes.

Taxon	**Sep-99**	**Mar-00**	**Apr-00**	**May-00**	**Aug-00**	**Sep-00**	**Nov-00**	**Jan-01**	**Mar-01**	**Sep-01**	**Mean**
Sampling Episode	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**-**
***Baetis* sp.**	**2.2**	1.8	0.6	1.3	1.8	0	**3.3**	1.6	**6.7**	**5.8**	**2.7**
***Cinygmula* sp.**	0	1.2	0.3	0.9	0.03	0	1.5	1.0	1.7	**5.9**	1.4
***Heptagenia* sp.**	0	0.2	0	0	0.4	0.8	0.4	0.07	0.1	**2.9**	0.5
***Rhithrogena* sp.**	**7.9**	1.6	0	0.8	0.3	0.7	0.6	0.2	**4.6**	**5.5**	**3.4**
***Ephemerella* sp.**	**8.3**	**4.0**	0.3	**11.2**	1.1	1.9	3.1	1.6	1.5	**6.9**	**4.8**
***Ameletus* sp.**	0.2	1.8	**37.3**	**5.2**	0.2	0	0.2	0.5	0.05	0.02	**3.4**
***Capnia* sp.**	0	1.4	0.7	0.4	0.06	0	**2.8**	1.8	1.4	0.3	0.8
***Taenionema* sp.**	0.1	0.7	0	0	0	0	**2.0**	1.1	1.1	0.2	0.5
***Hydropsyche* sp.**	**2.0**	0.6	0	0.5	0.2	0.5	0.5	0.1	0.2	1.7	0.9
***s.f.* Orthocladiinae**	**71.4**	**78.4**	**31.0**	**40.1**	**74.7**	**45.2**	**57.9**	**75.5**	**73.6**	**59.4**	**65.2**
***s.f.* Chironominae**	0.06	**2.6**	**8.8**	**5.5**	**19.4**	**8.0**	**19.2**	**5.9**	**4.0**	**3.0**	**5.1**
Nematoda	0.4	0.7	**5.0**	0.2	0.1	**4.2**	1.1	**3.8**	1.3	0.3	1.3
Naididae	**2.6**	2.0	1.1	**7.6**	0.3	**11.3**	**5.2**	**4.6**	**2.1**	**5.0**	**3.5**
Tubificidae	0.4	0.8	**11.0**	**25.2**	1.0	**23.4**	0.4	0	0	0.06	**3.7**
Total	**95.6**	**97.7**	**95.9**	**98.9**	**99.6**	**95.9**	**98.1**	**97.7**	**98.4**	**96.7**	**97.0**

Bold type indicates a sampling episode in which the taxon represented >2%, on average, of all invertebrates collected.

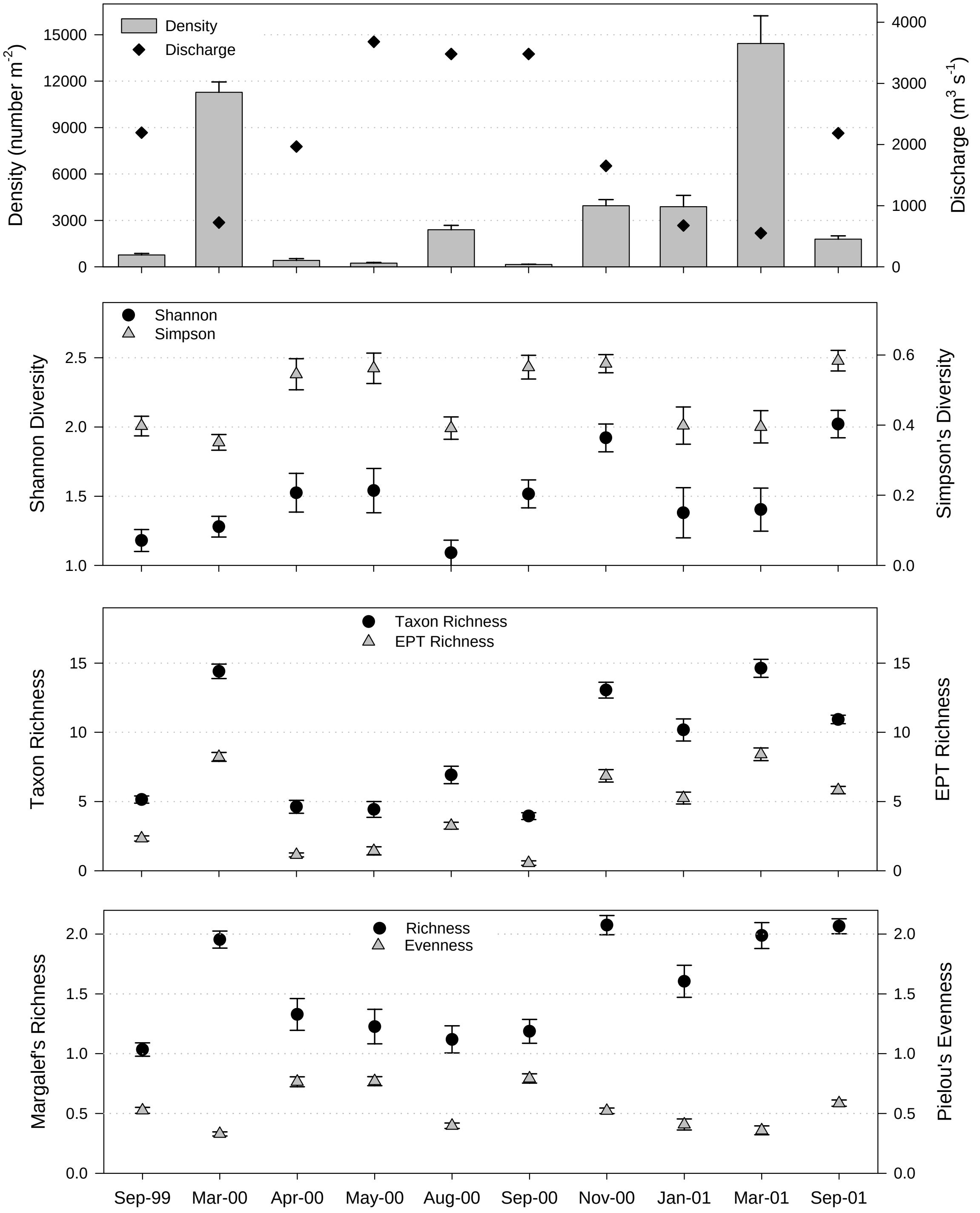


Figure 5-2. Community metrics (mean ± SE) characterizing invertebrate samples collected in the main channel between September 1999 and September 2001. Habitat and sub-reach data are pooled on each date. Discharge was measured at Hope.

Table 5-2. Two-factor ANOVA results examining effects of habitat type and sub-reach on community metrics of invertebrate data collected in September 1999 from the main channel of Fraser River.

Variable	**SS**	***df***	**MS**	**F**	***p***
Density					
Sub-reach	0.56	2	0.28	0.43	0.65
Habitat	1.93	3	0.64	0.99	0.41
Sub-reach*Habitat	4.76	6	0.79	1.22	0.31
Error	35.73	55	0.65		
Shannon-Wiener Diversity					
Sub-reach	5.10	2	2.55	10.94	0.0001*
Habitat	2.36	3	0.79	3.38	0.03
Sub-reach*Habitat	3.42	6	0.57	2.44	0.04
Error	12.82	55	0.23		
Simpson's Diversity					
Sub-reach	0.62	2	0.31	11.20	0.00008*
Habitat	0.29	3	0.10	3.52	0.02
Sub-reach*Habitat	0.40	6	0.07	2.40	0.04
Error	1.53	55	0.03		
Pielou's Evennness					
Sub-reach	0.26	2	0.13	4.86	0.01*
Habitat	0.11	3	0.04	1.33	0.28
Sub-reach*Habitat	0.25	6	0.04	1.58	0.17
Error	1.32	50	0.03		
Margalef's Richness					
Sub-reach	2.40	2	1.20	7.40	0.001*
Habitat	0.46	3	0.15	0.95	0.42
Sub-reach*Habitat	1.90	6	0.32	1.95	0.09
Error	8.91	55	0.16		

* significant at the $\alpha = 0.01$ level (adjusted by Bonferroni's correction for multiple contrasts).

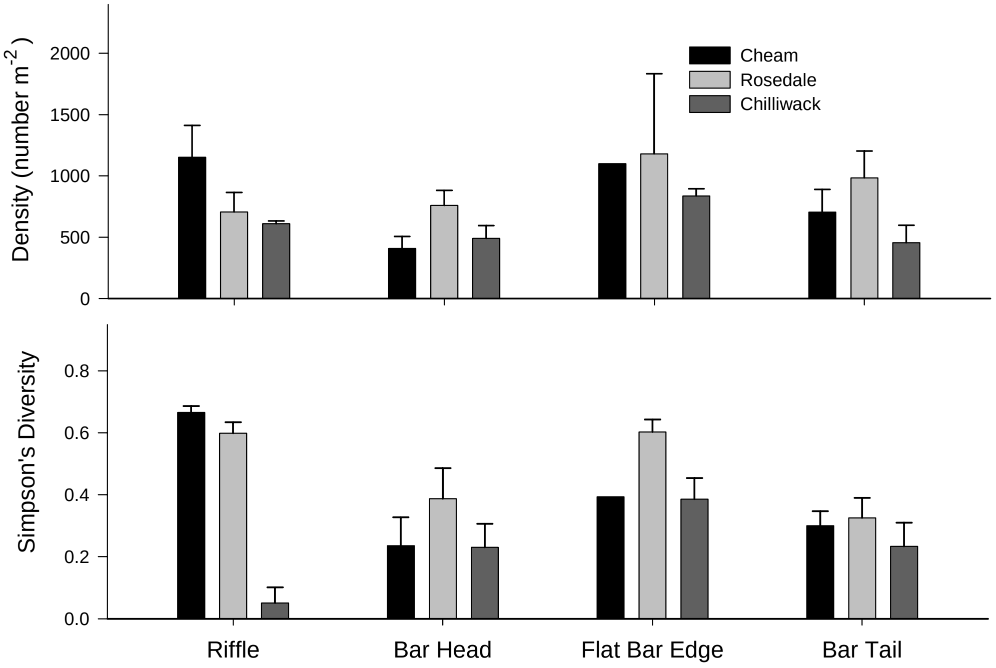


Figure 5-3. Community metrics (mean + SE) of density and Simpson's diversity for main channel invertebrate samples collected from four habitat types in sub-reaches of Fraser River in September 1999.

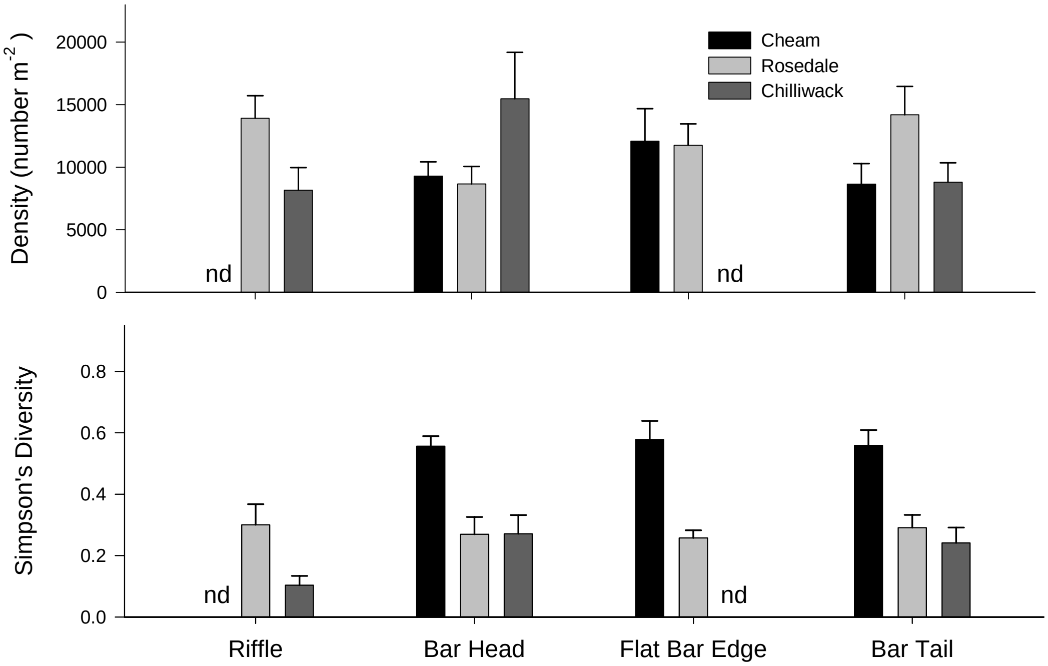


Figure 5-4. Community metrics (mean + SE) of density and Simpson's diversity for main channel invertebrate samples collected from four habitat types in sub-reaches of Fraser River in March 2000. "nd": no data collected.

5.3.2 Multivariate Analysis of Community Structure

Multivariate analyses examined differences in invertebrate community structure among habitat types at multiple temporal scales to determine at what scale, if any, habitat differences were detected. Analyses revealed a high degree of similarity among habitat types, prompting additional confirmatory analyses that evaluated various options for data transformation. The options explored were three levels of data transformation (i.e., raw, square-root, fourth-root), combined with the inclusion/exclusion of the most abundant taxon, Orthocladiinae. The outcome of each analysis was evaluated based on which produced an MDS ordination with lowest stress (and therefore most reliable ANOSIM), and which produced the most informative SIMPER results.

Some general comments regarding confirmatory analyses can be made. First, results were highly similar for all options of data transformation. The square-root transformation was chosen as the standard for MDS analyses because it achieved the lowest stress value in all cases. This is consistent with recommendations of Clarke and Gorley (2001) that the fourth-root transformation is too widely applied in ecological studies where the severity of the transformation is not required. Second, excluding Orthocladiinae from analyses had only minor effects on MDS ordinations and ANOSIM results. The *percent contribution* made by less abundant taxa to the overall dissimilarity between groups was greater, but the *relative contribution* of less abundant taxa remained virtually unchanged and the magnitude of dissimilarity between groups was unchanged. This pattern was observed at all temporal scales of examination except the winter season (presented below). These results, and the fact that Orthocladiinae contribute significantly to the diet of many resident fish species and juvenile chinook salmon (**Appendix G**), justified including Orthocladiinae in analyses.

The first analysis, examining all episodes combined, is presented comprehensively to familiarize the reader with the analysis. Only pertinent results are presented for subsequent analyses; a summary of all results is provided in Section 5.3.3, **Table 5-10**, and additional results are presented in **Appendix C**.

5.3.2.1 Habitat Associations – All Episodes Combined

Ordination of invertebrate abundances by MDS achieved a very low stress level (0.09), indicating that the distance between samples in the 2-D plot accurately portrayed the degree of taxonomic similarity between them. Samples classified as either morphological or hydraulic habitat types showed poor grouping in the ordination plot (**Figure 5-5a, b**), compared with distinct groupings of samples classified by episode and season (**Figure 5-5c, d**). Overlaid on the ordination in **Figure 5-5** are three clusters, identified by group-average cluster analysis, that correspond to a dissimilarity measure of 55% (i.e., clusters shared 45% similarity in community structure). Winter samples collected between November and March were most similar in community structure, as indicated by the tight clustering of samples in ordination space. Samples from September 2000 were more similar to spring samples collected in April and May 2000, than to those collected in September 1999 or 2001.

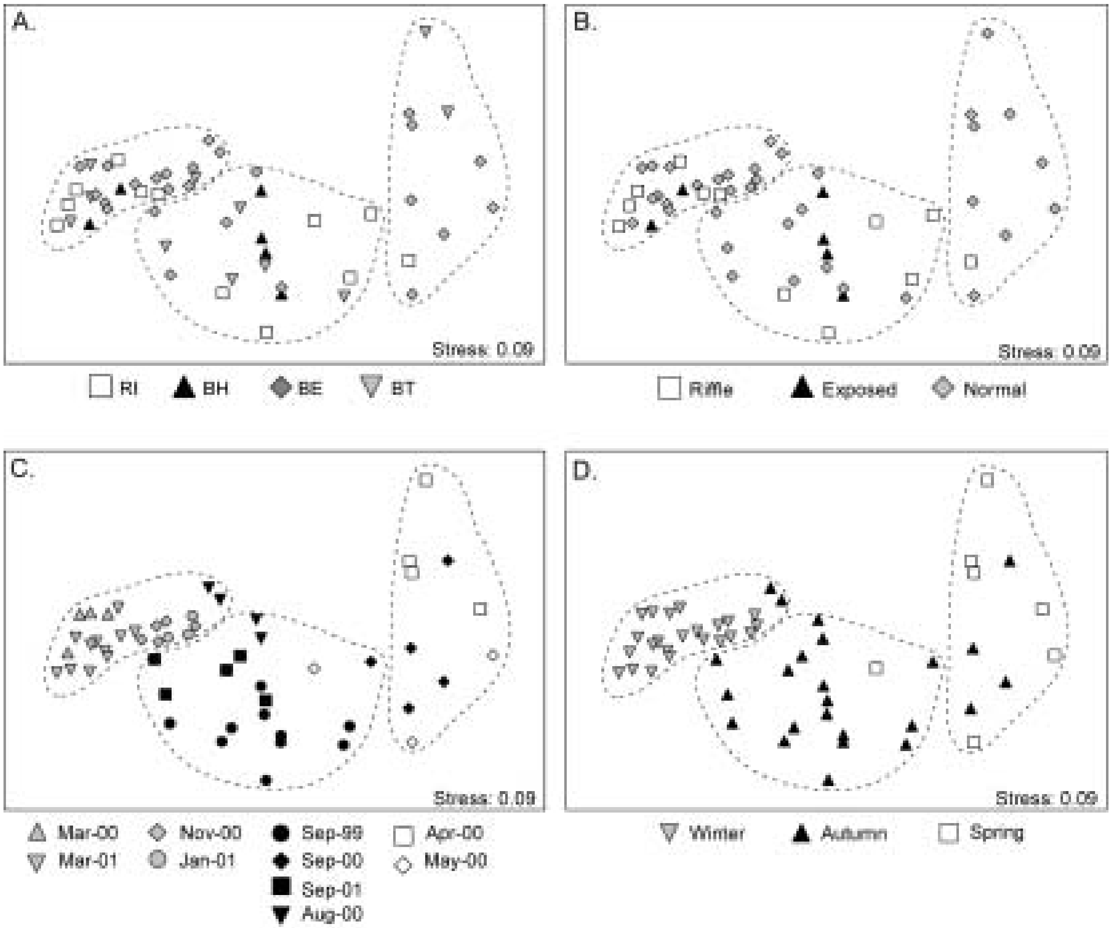


Figure 5-5. MDS ordination of square-root transformed invertebrate abundances in main channel samples collected in the Rosedale sub-reach between September 1999 and September 2001. Each plot shows the identical ordination with samples classified according to A) morphological habitats, B) hydraulic habitats, C) sampling episodes, and D) seasons. Dotted outlines indicate sample groupings clustered at 45% similarity.

Two-way ANOSIM was applied to the Bray-Curtis similarity matrix based on square-root transformed data to determine the degree of similarity in community structure between seasons and habitats. All seasons were highly dissimilar from each another ($p < 0.001$, **Table 5-3**), but morphologic and hydraulic habitat types averaged across seasons were similar ($p = 0.5$ and $p = 0.7$, respectively). These results support the interpretation of the MDS ordination, and preclude an analysis of community related differences between habitat types by SIMPER.

Table 5-3. Results of 2-way ANOSIM tests comparing community patterns between seasons and habitat types, based on main channel data collected in the Rosedale sub-reach between September 1999 and 2001.

Comparison	**Possible Permutations**	**# Cases ≥ Observed$^+$**	***R*-statistic**	***p*-value**
Differences between seasons averaged across habitat types				
Global			0.67	<0.001*
Spring, Autumn	>999	4	0.46	<0.001*
Spring, Winter	>999	0	1.00	<0.005*
Autumn, Winter	>999	0	0.55	<0.001*
Differences between morphologic habitat types averaged across seasons				
Global			0.006	0.50
Differences between hydraulic habitat types averaged across seasons				
Global			-0.041	0.69

* significant at the adjusted value of $\alpha = 0.1$.
\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

Abundance data were fourth-root transformed for comparisons by SIMPER of community structure among seasons because of the substantially greater abundances of virtually all taxa in winter. Prior analysis of square-root transformed data found that Orthocladiinae alone contributed up to 76% of the dissimilarity between seasons, thereby overwhelming contributions made by most other taxa. SIMPER analysis revealed high similarity among winter samples collected in November through March, compared to autumn and spring, and all taxa except the mayfly *Ameletus* sp. were most abundant in winter (**Table 5-4**). Besides seasonal dissimilarities contributed by chironomid taxa, winter samples were differentiated from autumn samples by a greater proportion of the stonefly *Capnia* sp. and the mayflies *Cinygmula* sp. and *Baetis* sp. Seasonal differences in the abundance of *Capnia* sp. also contributed to the dissimilarity between winter and spring samples. Spring and autumn samples, all of which had substantially lower abundances than winter samples, were differentiated by differences in abundance of the mayflies *Ephemerella* sp. and *Rhithrogena* sp.

Bubble plots showing the abundances of individual taxa overlaid on the MDS ordination assisted in assessing seasonal dynamics in relation to the overall community pattern. The plots indicate that most taxa including Orthocladiinae were most abundant in winter and least common in spring (**Figure 5-6**). Many taxa such as the mayflies *Rhithrogena* sp. and *Ephemerella* sp. increased in abundance systematically from autumn to winter, and most were virtually absent from spring samples on the rising limb of the hydrograph. The exceptions were *Ameletus* sp. and Tubificidae, which were relatively abundant in spring.

Table 5-4. Results of SIMPER analysis based on fourth-root transformed data indicating the average abundance (untransformed) of taxa in each season that contributed most to the *dissimilarity* in community structure between seasons.

Taxon	**Average Abundance**			**% Contributed to Dissimilarity**		
	Spring	**Autumn**	**Winter**	**S/A**	**S/W**	**A/W**
***s.f.* Orthocladiinae**	5.1	79.3	793.7	**10.6**	**13.8**	**10.6**
***s.f.* Chironominae**	1.2	8.2	42.9	**7.9**	**6.6**	**7.3**
***Baetis* sp.**	0.2	2.5	26.1	6.4	**6.5**	5.4
***Ephemerella* sp.**	1.0	9.3	24.3	**9.7**	6.1	4.4
***Rhithrogena* sp.**	0.1	5.4	18.6	**8.6**	5.3	4.3
***Capnia* sp.**	0.1	2.2	17.5	2.3	**6.4**	**7.7**
***Cinygmula* sp.**	.1	1.0	14.0	3.4	5.7	**6.6**
Taenionema sp.	0	0.1	10.0	-	5.9	6.3
Hemerodromia sp.	0.1	0.4	3.8	-	4.2	4.6
Ameletus sp.	1.6	0	0.7	7.3	2.5	-
Naididae	0.7	2.4	10.0	5.9	4.3	3.3
Tubificidae	2.3	0.8	4.1	6.7	2.9	3.0
Nematoda	0.3	0.2	8.1	4.0	4.6	5.6
Mean Similarity	45.7	53.3	73.2	-	-	-
Mean Dissimilarity	-	-	-	60.5	69.2	54.5

Data are from the main channel of the Rosedale sub-reach collected between September 1999 and 2001. Bolded text highlights the four taxa contributing most to the dissimilarity *between pairs*.

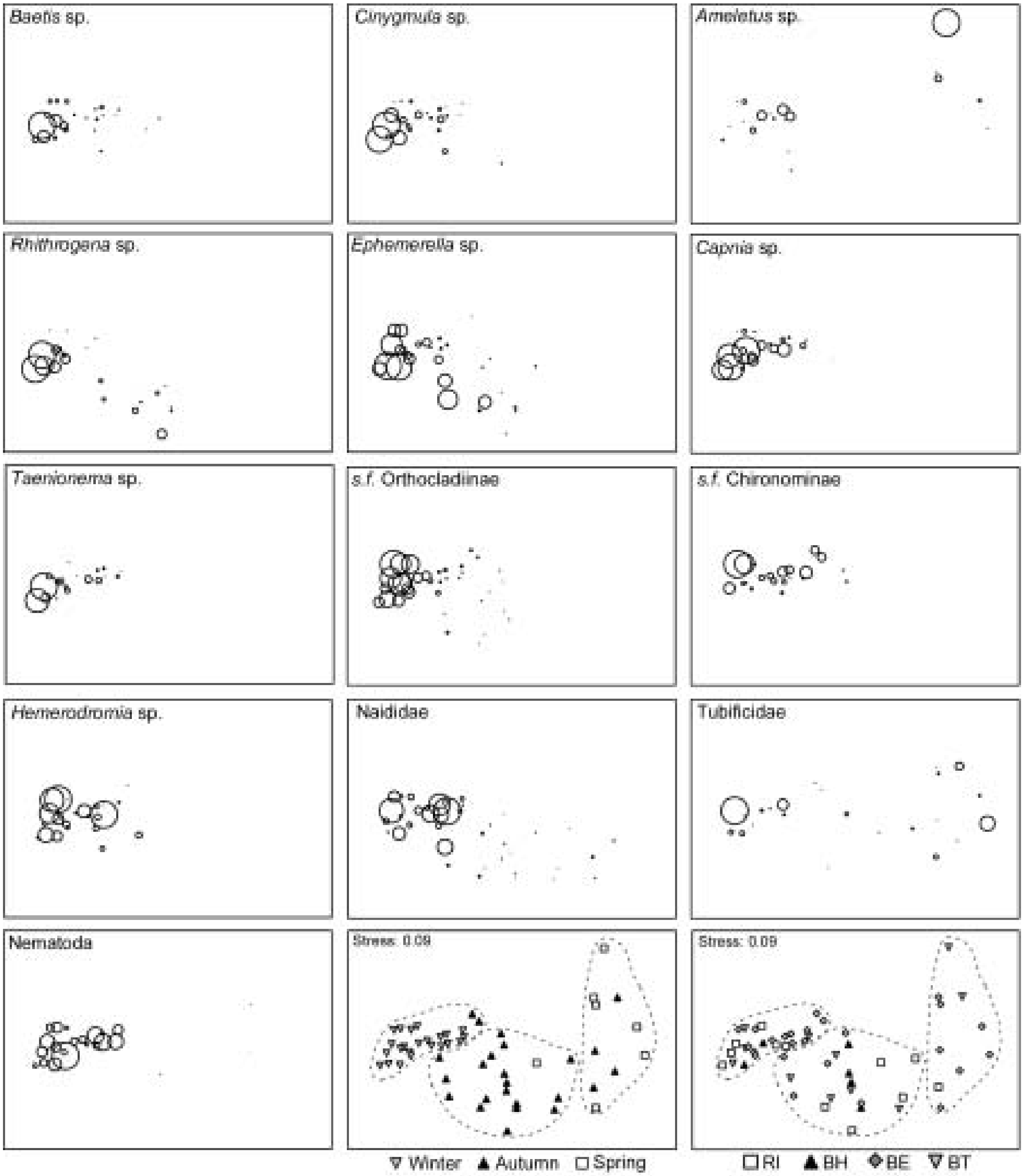


Figure 5-6. Bubble plots corresponding to the untransformed abundances of select invertebrate taxa that contributed to the dissimilarity in community structure between seasons. Plots are overlaid on the MDS ordination of main channel samples collected in the Rosedale sub-reach between September 1999 and September 2001 (**Figure 5-5**). Bubble symbols are sized proportionately to individual taxon abundance.

5.3.2.2 Habitat Associations – Winter Season

Invertebrate samples collected in winter (discharge <1500 $m^3 s^{-1}$) had the highest taxonomic similarity of all seasons and were chosen for more detailed examination of temporal and habitat-specific differences in community structure. Sampling took place four times in winter at low discharge, and MDS ordination grouped all episodes separately (**Figure 5-7a**). The low stress level (0.1) indicates that the 2-D plot accurately represents the taxonomic similarity among samples. Combined, all winter months shared approximately 60% similarity, as revealed from group-average cluster analysis, and monthly clusters were separated at 70% similarity (i.e., 30% dissimilarity between months, **Figure 5-7b**). Monthly clusters did not correspond with morphological habitat classes (**Figure 5-7c**), but showed some organization by hydraulic habitat class within November and March 2000 clusters (**Figure 5-7d**). In these clusters, "normal" habitats are separated from riffles in ordination space.

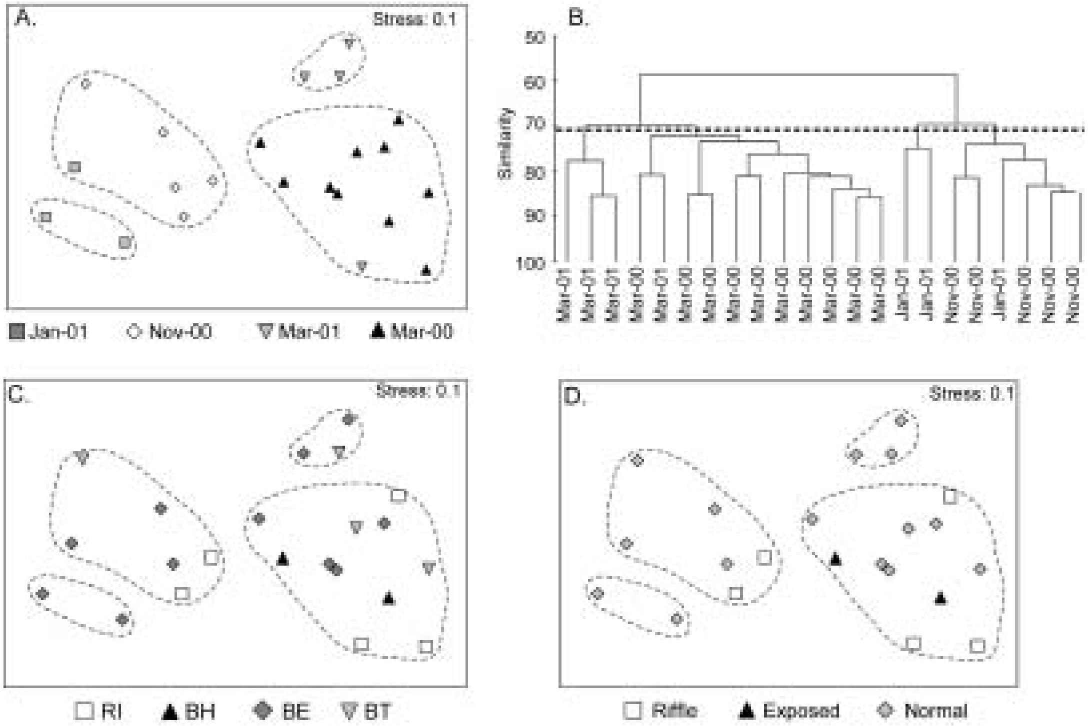


Figure 5-7. MDS ordination and cluster analysis of main channel invertebrate samples collected in the Rosedale sub-reach during winter months. Each plot shows the identical ordination with samples classified according to A) months, C) morphological habitats, and D) hydraulic habitats. Dotted outlines indicate sample groupings clustered at 70% similarity.

Two-way ANOSIM indicated significant dissimilarity between episodes ($p < 0.001$) and hydraulic habitat types ($p < 0.03$), but dissimilarity among morphologic habitats was marginally insignificant ($p = 0.16$). Samples collected in March 2000 were dissimilar from all winter months in 2001 (**Table 5-5**), and samples collected repeatedly over one winter between November 2000 and March 2001, and generally representing the same cohort, were similar in community structure (**Table 5-5**). Pairwise comparisons among hydraulic habitats showed that "normal" habitats (flat bar edge and bar tail) were highly dissimilar from riffles ($p = 0.04$), whereas contrasts between morphological habitats indicated that only riffles and flat bar edges were dissimilar ($p = 0.02$).

Table 5-5. Results of 2-way ANOSIM tests comparing episodes, and hydraulic and morphological habitat types, based on main channel data collected from the Rosedale sub-reach in winter months.

Comparison	**Possible Permutations**	**# Cases ≥ Observed**[+]	***R*-Statistic**	***p*-value**
Differences between episodes averaged across hydraulic habitat types				
Global			0.86	0.001*
March 2000, November 2000	252	1	0.95	0.004*
March 2000, January 2001	84	1	0.99	0.01*
March 2000, March 2001	252	5	0.63	0.02*
November 2000, January 2001	10	1	0.74	0.10
November 2000, March 2001	30	4	0.99	0.13
January 2001, March 2001	10	2	0.99	0.20
Differences between hydraulic habitat types averaged across episodes				
Global			0.37	0.03*
Riffle, Exposed	3	3	-0.50	1.00
Riffle, Normal	>999	36	0.47	0.04*
Exposed, Normal	28	5	0.30	0.18
*Differences between morphologic habitat types averaged across episodes***				
Global			0.18	0.16
Riffle, Bar Edge	135	3	0.43	0.02

* significant at the adjusted value of $\alpha = 0.1$.

\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

** only contrasts with $p < 0.20$ are reported.

SIMPER analysis (**Appendix C**) showed that several common taxa decreased in abundance from November 2000 to January 2001, before increasing between January and March 2001. Total density, however, remained constant over this period (**Figure 5-2**). Between November 2000 and March 2001, all common taxa increased in abundance except Naididae, which contributed 6.5% to the total dissimilarity between November and January samples. Significant differences between March samples in 2000 and 2001 were attributed to differences in the abundance of chironomids, as well as the mayflies *Baetis* sp. and *Rhithrogena* sp. Bubble plots of taxon abundances overlaid on the ordination support these patterns (**Appendix C**).

The SIMPER procedure was used to identify which taxa contributed to the dissimilarity between hydraulic habitats. The mayflies *Baetis* sp. and *Rhithrogena* sp., and the stonefly *Taenionema* sp. were substantially more abundant in riffle habitats and significant contributors to the dissimilarity of riffle units from exposed and normal habitats (**Table 5-6**). Orthocladiinae and Chironominae also contributed substantially to the dissimilarity of riffles, but based on a lower abundance in riffles than in exposed and normal habitats. All common mayfly taxa were most abundant in riffles except *Ephemerella* sp., which was more abundant in exposed bar head units and contributed substantially to the dissimilarity between exposed and normal habitats.

Analysis by BIO-ENV showed a correlation between velocity and the community structure of invertebrates ($\rho = 0.13$), but no correlation with water depth or substrate size. A velocity gradient, nested within the temporal gradient of sampling episodes, was detected particularly in March 2000 and November 2000 clusters (**Figure 5-8**).

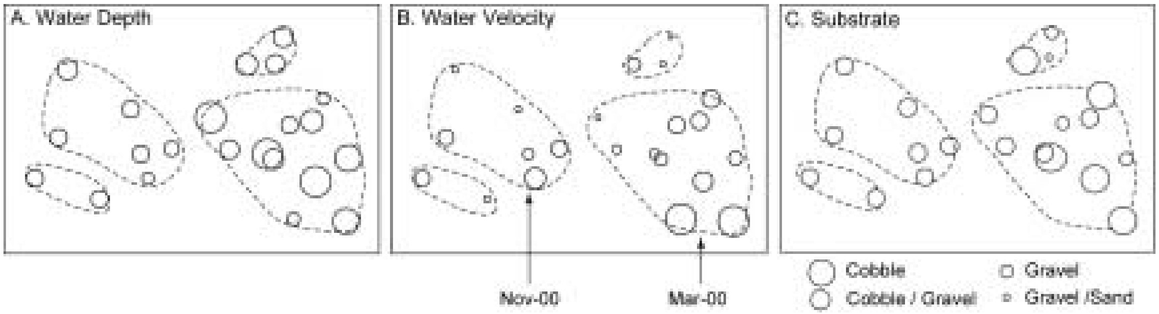


Figure 5-8. Bubble plots representing values of water depth, velocity, and substrate size that corresponded to invertebrate samples collected in winter from the main channel of the Rosedale sub-reach. Depth and velocity symbols are scaled proportionately to a continuous scale of measurement. Substrate symbols are scaled according to size categories. Dotted outlines indicate sampling episodes clustered at 70% similarity.

Table 5-6. Results of SIMPER analysis based on square-root transformed data indicating the average abundance (untransformed) of taxa in each habitat that contributed most to the *dissimilarity* in community structure between habitats.

Taxon	**Average Abundance**			**% Contributed to Dissimilarity**		
	Riffle	**Exposed**	**Normal**	**R/E**	**R/N**	**E/N**
***s.f.* Orthocladiinae**	720.3	765.7	821.9	**17.3**	**23.4**	**22.4**
***s.f.* Chironominae**	32.9	22.2	49.0	**7.0**	**7.7**	**8.0**
***Baetis* sp.**	44.9	15.2	21.3	5.0	**6.0**	3.7
***Ephemerella* sp.**	25.8	47.2	20.7	7.0	5.7	**8.0**
***Rhithrogena* sp.**	38.4	19.8	11.8	**8.5**	**7.4**	**6.1**
***Taenionema* sp.**	26.2	2.5	5.6	**7.1**	5.4	<3
Cinygmula sp.	34.3	24.2	5.9	4.4	6.3	5.5
Capnia sp.	28.3	13.0	14.4	4.1	5.3	3.6
Hydropsyche sp.	8.9	13.2	1.1	4.9	<3	5.1
Naididae	14.9	12.7	8.0	<3	4.3	3.2
Mean Similarity	66.8	74.9	65.7	-	-	-
Mean Dissimilarity	-	-	-	27.5	34.8	32.1

Data are from the main channel of the Rosedale sub-reach collected in winter months.
Bolded text highlights the four taxa contributing most to dissimilarity *between pairs*.

5.3.2.3 Habitat Associations – September 1999 and March 2000

Rosedale sub-reach samples collected in September 1999 and March 2000 were chosen for separate analyses at the finest scale of temporal resolution, thereby eliminating time-course changes that may confound the influence of habitat attributes. Sampling effort was highest in these months and, together, they provided a contrast between autumn and winter seasons. The episodes also were suited to contrasts between sub-reaches and channel types because main channel sampling took place in all sub-reaches in each month, and samples in September 1999 were collected both from main and side channels.

MDS ordination of main channel samples from September 1999 revealed no apparent groupings by morphological or hydraulic habitat types (**Appendix C**). The very low stress level (0.06) indicates that the ordination accurately represents the taxonomic similarity between samples. The similarity in community structure among habitat types was confirmed by 1-way ANOSIM (morphological habitats $p = 0.95$; hydraulic habitats $p = 0.80$), and precluded SIMPER analysis.

Samples also were analysed among sites to determine if site-specific differences contributed to sample groupings. However, no consistent patterns were noted and 1-way ANOSIM found no site differences ($p = 0.49$). This result supports the use of entire gravel bars as replicate sampling units for invertebrates to investigate habitat differences within sub-reaches.

MDS ordination of main channel samples collected in March 2000 identified distinct clusters corresponding to hydraulic habitat types (**Figure 5-9b)**, with all but one "normal" unit separated from exposed and riffle units at a similarity level of 80% (**Figure 5-9d**). The single "exposed" unit grouped with a "normal" unit had lower velocity than is typical of bar heads. The clusters of hydraulic habitat types corresponded to differences in water velocity (**Appendix C**) and BIO-ENV showed a correlation between velocity and community structure ($\rho = 0.16$). One-way ANOSIM found no global dissimilarity among morphologic habitat types ($p = 0.25$), but revealed global dissimilarity among hydraulic habitat types ($p = 0.07$, **Table 5-7**). Pairwise comparisons showed a difference between riffle and normal habitats, though marginally insignificant ($p = 0.11$). These results are considered in the context of the overall large degree of similarity shared among *all* March 2000 samples (almost 75% similarity, **Figure 5-9d**); hence, dissimilarity among groups was relatively low. Excluding Orthocladiinae and applying different data transformations produced identical results. Similar to the analysis of September 1999 samples, no systematic grouping by sites was revealed in the ordination or by ANOSIM ($p = 0.4$, **Figure 5-9c**).

Table 5-7. Results of 1-way ANOSIM comparing morphological and hydraulic habitat types based on main channel samples collected in the Rosedale sub-reach in March 2000.

Comparison	**Possible Permutations**	**# Cases ≥ Observed**$^+$	***R*-Statistic**	***p*-value**
Differences between morphologic habitat types				
Global			0.13	0.25
Differences between hydraulic habitat types				
Global			0.35	0.07*
Riffle, Exposed	3	3	-0.5	1.0
Riffle, Normal	28	3	0.47	0.11
Exposed, Normal	28	5	0.30	0.18

* $\alpha = 0.1$, adjusted for the exploratory nature of the analysis.
\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

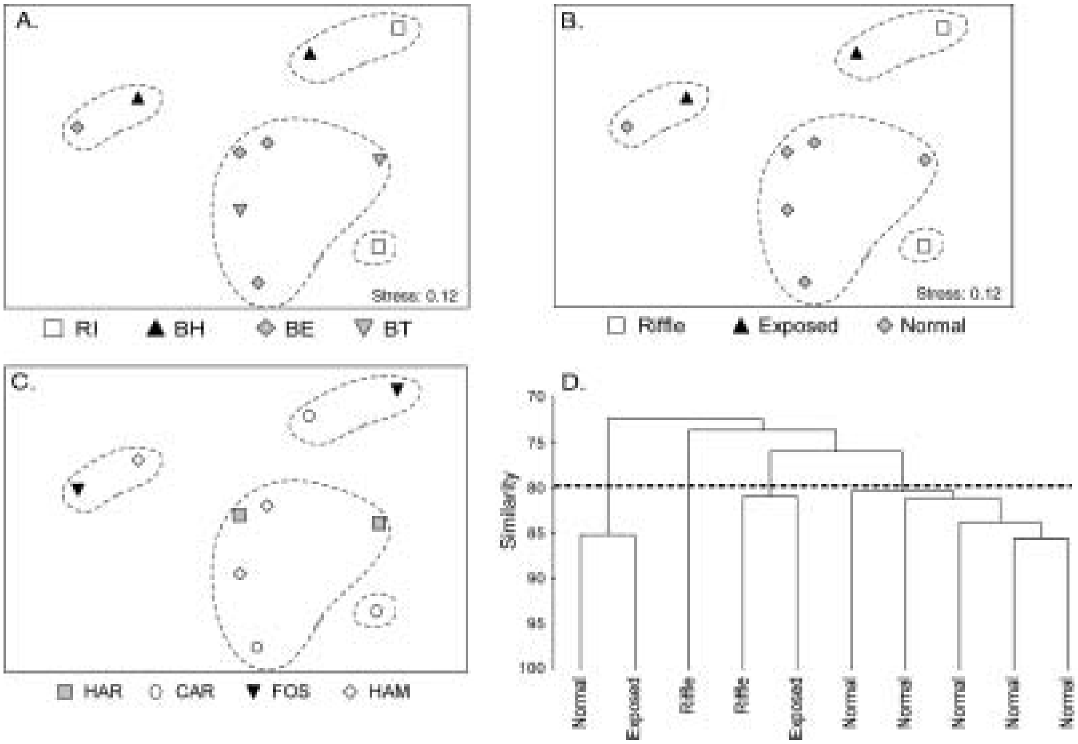


Figure 5-9. MDS ordination and cluster analysis of main channel invertebrate samples collected in March 2000 from the Rosedale sub-reach. Each plot shows the identical ordination with samples classified by A) morphological habitats, B) hydraulic habitats, and C) sites. Legend lists sites in order from downstream (left) to upstream (right). Dotted outlines indicate sample groupings corresponding to 80% similarity.

SIMPER analysis indicated that high densities of *Rhithrogena* sp. and *Taenionema* sp. in riffles contributed substantially to the dissimilarity with other habitats (**Table 5-8**), similar to results of all winter months combined. In contrast to the latter analysis, however, *Baetis* sp. was most abundant in "normal" habitats (mostly bar tail units) and contributed substantially to the dissimilarity of normal habitats from exposed and riffle units. *Hydropsyche* sp., though relatively low in overall abundance, was most abundant in high velocity habitats (riffle and exposed) and contributed to the dissimilarity with "normal" habitat types.

Table 5-8. Results of SIMPER analysis (square-root transformed data) indicating average abundance (untransformed) of taxa contributing most to the *dissimilarity* in community composition between hydraulic habitats. Data are from the main channel of the Rosedale sub-reach collected in March 2000 (square-root transformation applied to abundance data to derive dissimilarity contributions).

Taxon	**Average Abundance**			**% Contributed to Dissimilarity**		
	Riffle	**Exposed**	**Normal**	**R/E**	**R/N**	**E/N**
***s.f.* Orthocladiinae**	1149.3	765.7	1076.0	**14.0**	**13.6**	**13.9**
***s.f.* Chironominae**	4.0	22.2	19.9	**7.3**	**6.8**	**7.1**
***Baetis* sp.**	29.7	15.2	33.3	3.6	5.1	**7.0**
***Rhithrogena* sp.**	47.5	19.8	26.9	**9.6**	**8.9**	**7.6**
***Cinygmula* sp.**	44.7	24.2	8.9	4.2	**8.0**	4.9
Ephemerella sp.	52.7	47.2	35.2	5.3	4.5	6.1
***Taenionema* sp.**	27.5	2.5	9.7	**7.6**	4.8	3.9
Capnia sp.	23.8	13.0	27.8	4.2	4.9	4.9
Ostrocerca sp.	14.0	2.5	6.1	4.6	3.2	2.2
Hydropsyche sp.	20.3	13.2	0.9	5.8	6.3	5.9
Naididae	14.2	12.7	3.1	5.9	5.7	4.8
Mean Similarity	67.3	74.9	78.7	-	-	-
Mean Dissimilarity	-	-	-	25.1	26.3	23.9

Bolded text highlights the four taxa contributing most to dissimilarity *between pairs*.

5.3.2.4 Sub-Reach Differences – September 1999

Ordination by MDS to evaluate dissimilarity in community patterns among sub-reaches based on September 1999 data showed relatively poor clustering among samples (**Appendix C**). The stress value (0.14) indicates good representation of the similarity between samples in 2-D ordination space. One-way ANOSIM revealed significant dissimilarity between sub-reaches ($p = 0.04$) and pairwise comparisons indicated that community structure differed between the Rosedale and Cheam sub-reaches (p = 0.009). Based on SIMPER analysis (**Appendix C**), the Rosedale sub-reach had higher abundances of Orthocladiinae and the mayflies *Ephemerella* sp. and *Rhithrogena* sp., compared to other sub-reaches. *Hydropsyche* sp. also contributed substantially to the dissimilarity of the Rosedale sub-reach, being relatively more abundant than in the Chilliwack and Cheam sub-reaches. The Chilliwack sub-reach was dissimilar based on the presence of the filter-feeding midge *Simulium* sp., which was absent from samples in upstream reaches in September 1999.

5.3.2.5 Sub-Reach Differences – March 2000

MDS ordination of main channel samples collected in March 2000 identified sub-reach clusters that were differentiated at a similarity level of 70% (**Figure 5-10**). One sample from the Cheam sub-reach at Powerline Island was grouped with Rosedale samples, and two samples from Carey Bar were grouped with Chilliwack samples. Other than these cases, all samples were grouped appropriately by sub-reach. Sub-reach clusters were organized in an upstream order along the horizontal axis of the ordination plot, indicating that Chilliwack and Cheam samples were most dissimilar in community structure. One-way ANOSIM revealed significant dissimilarity between Chilliwack and upstream sub-reaches (**Table 5-9**). The dissimilarity between Rosedale and Cheam sub-reaches was marginally insignificant ($p = 0.11$).

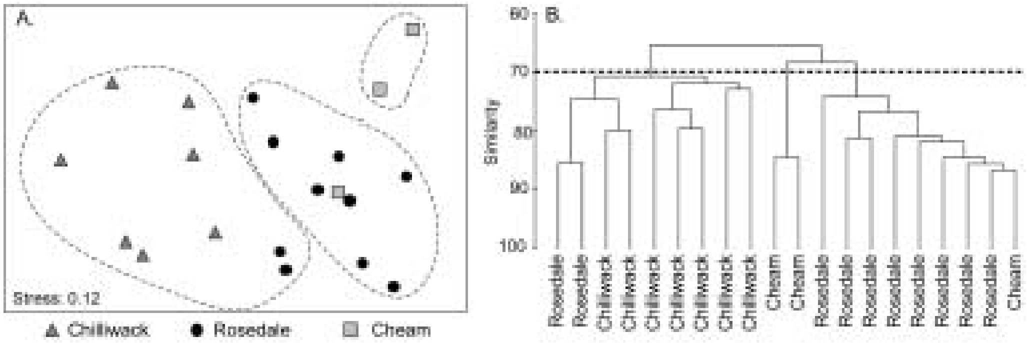


Figure 5-10. MDS ordination and cluster analysis of main channel invertebrate samples collected in March 2000 from all sub-reaches. Dotted outlines indicate sample groupings corresponding to 70% similarity.

Table 5-9. Results of pairwise tests from 1-way ANOSIM comparing sub-reaches based on main channel samples collected in March 2000.

Comparison	**Possible Permutations**	**# Cases ≥ Observed$^+$**	***R*-Statistic**	***p*-value**
Global			0.59	<0.001*
Chilliwack, Rosedale	>999	0	0.67	<0.001*
Chilliwack, Cheam	120	1	0.87	0.008*
Rosedale, Cheam	286	31	0.27	0.11

* $\alpha = 0.1$, adjusted for the exploratory nature of the analysis.
\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

No significant dissimilarity between morphologic or hydraulic habitat types was detected among sub-reaches by ANOSIM. However, a hydraulic gradient of velocity nested within sub-reach clusters was revealed (**Figure 5-11**); samples from high velocity sites within the Chilliwack and Rosedale clusters were grouped separately from low velocity sites. Analysis by BIO-ENV confirmed the correlation between velocity and the community structure of invertebrates ($\rho = 0.33$), nested within the spatial scale of sub-reaches.

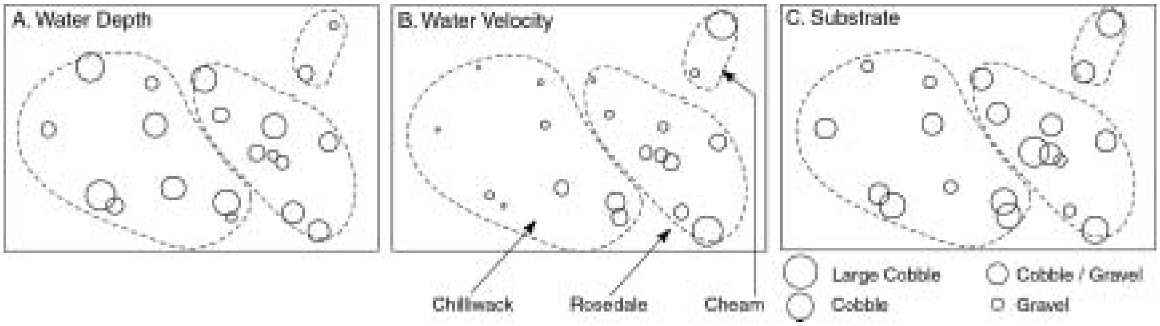


Figure 5-11. Bubble plots representing values of water depth, velocity, and substrate size that correspond to invertebrate samples plotted in ordination space (**Figure 5-10**). Substrate symbols are scaled according to size categories. Depth and velocity symbols are scaled proportionately to a continuous scale of measurement. Dotted outlines indicate sampling episodes clustered at 70% similarity.

SIMPER analysis (**Appendix C**) revealed taxonomic dissimilarities between sub-reaches that were substantiated by bubble plots of taxon abundances overlaid in ordination space (**Figure 5-12**). The bubble plots assisted in the interpretation of community patterns and highlight the greater dissimilarity of the Chilliwack sub-reach from both upstream reaches. Several taxa including Orthocladiinae and the stoneflies *Taenionema* sp. and *Capnia* sp. had similar abundances in the Rosedale and Cheam sub-reaches. The latter two taxa were relatively rare in the Chilliwack sub-reach. Chilliwack samples also were dissimilar from upstream reaches based on high abundances of the mayfly *Ameletus* sp. and Naididae. A substantially lower abundance of *Ephemerella* sp. in Cheam samples compared to downstream sub-reaches contributed to the dissimilarity of the Cheam sub-reach. Several taxa were more abundant in the Rosedale sub-reach, but only Orthocladiinae contributed substantially to the sub-reach dissimilarity.

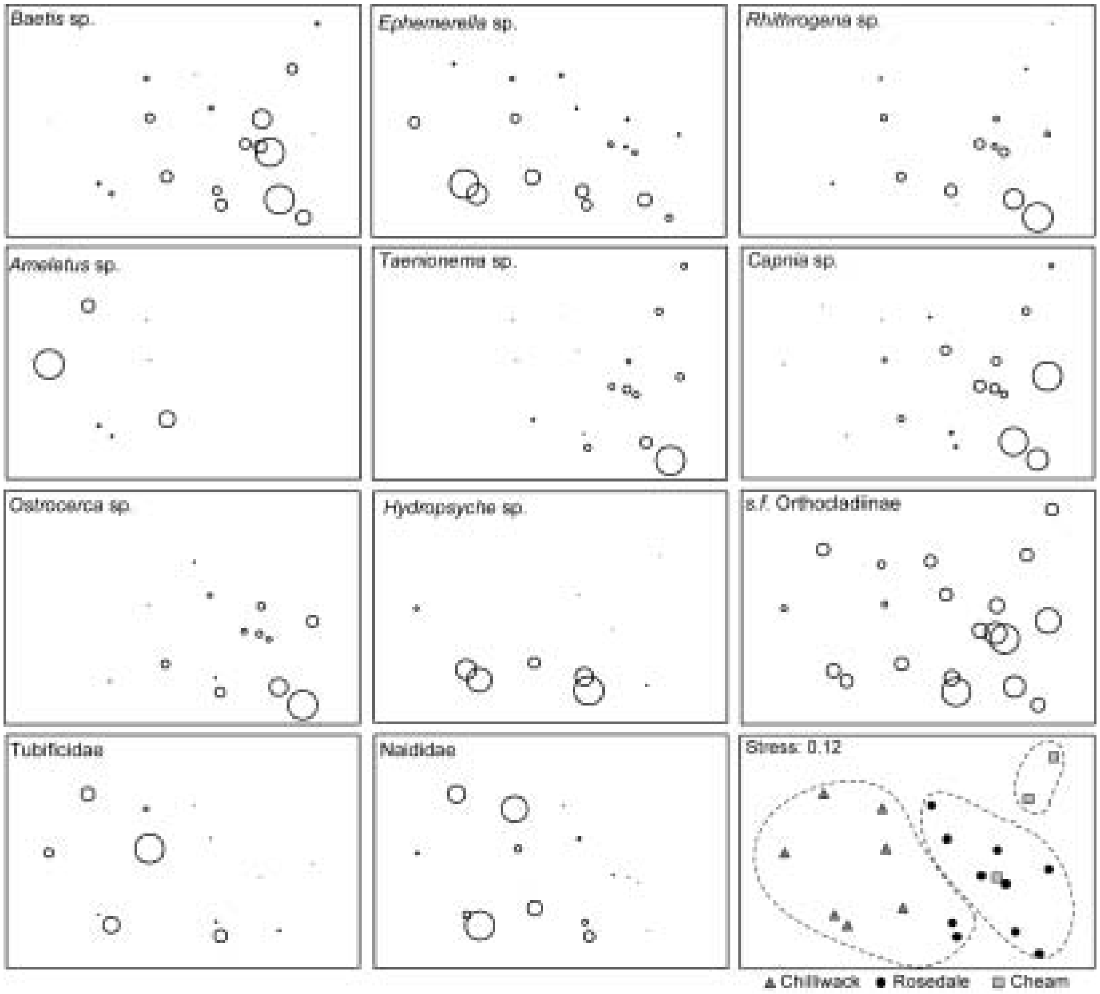


Figure 5-12. Bubble plots corresponding to the untransformed abundance of selected invertebrate taxa that contributed to the dissimilarity in community structure between sub-reaches. Plots are overlaid on the MDS ordination of main channel samples collected in March 2000 (**Figure 5-10**). The horizontal axis corresponds with a longitudinal gradient from downstream (left) to upstream (right). Bubble symbols are sized proportionately to individual taxon abundances.

5.3.2.6 Channel Type Differences – September 1999

MDS ordination showed no distinct clustering of main and side channels, both at the finer spatial scale of the Rosedale sub-reach and at the reach-scale with all sub-reaches combined (**Appendix C**). One-way ANOSIM found that samples from each channel type in the Rosedale sub-reach were similar (*p* = 0.14). Two-way ANOSIM examining channel type dissimilarity averaged across sub-reaches showed comparable results (*p* = 0.17).

5.3.3 Summary of Results

Table 5-10 provides a summary of results from multivariate analyses of invertebrate data. Overall, temporal changes in invertebrate community composition dominated results. Isolating samples collected in winter months (discharge <1500 $m^3 s^{-1}$) revealed significant dissimilarity in community structure between riffle and "normal" habitats ($p = 0.04$), the latter consisting of flat bar edge and bar tail units. Increasing temporal resolution and examining only samples collected in March 2000 found similar results, although marginally insignificant: riffles were dissimilar in community structure from "normal" habitats ($p = 0.11$). The mayflies *Rhithrogena* sp. and *Cinygmula* sp., and the stonefly *Taenionema* sp. were relatively abundant in riffles and consistently contributed to riffle dissimilarity from other habitat types. A relatively high abundance of *Ephemerella* sp. in "exposed" units compared to other habitats contributed substantially to dissimilarity. No dissimilarity between habitats was revealed in an analysis of autumn samples collected in September 1999.

Single-month analyses also revealed dissimilarity in community structure between sub-reaches. Samples collected from the Chilliwack sub-reach in March 2000 were dissimilar from both upstream sub-reaches ($p < 0.01$). A low proportion of *Ephemerella* sp. in Cheam samples, and a relatively high abundance of *Ameletus* sp. and Naididae in Rosedale samples, contributed substantially to sub-reach dissimilarity. Samples collected from the Rosedale sub-reach in September 1999 were dissimilar in community structure from Cheam samples ($p = 0.009$). This dissimilarity was associated with a relatively high abundance of *Ephemerella* sp., *Rhithrogena* sp., and *Hydropsyche* sp. in Rosedale samples. Chilliwack samples were differentiated from upstream sub-reaches by the presence of *Simulium* sp.

Table 5-10. Summary of results from multivariate analysis of the benthic invertebrate assemblage associated with sampling episodes and habitat types in the gravel reach of Fraser River. Analyses are based on square-root transformed data unless otherwise indicated.

Data Source	MDS stress	ANOSIM	SIMPER	BIO-ENV
ALL EPISODES (10) Sep-99 to Sep-01 Rosedale sub-reach Main channel	*2-D*: 0.09 *3-D*: 0.06	<u>*Two-way tests: Season & Habitat*</u> *Seasons*: Global-R, $p < 0.001$ • autumn, winter: $p < 0.001$* • autumn, spring: $p < 0.005$* • winter, spring: $p < 0.001$* *Morphologic*: Global-R, $p = 0.50$ *Hydraulic*: Global-R, $p = 0.70$	Almost all taxa more abundant in winter; *Capnia* sp., *Cinygmula* sp., and *Rhithrogena* sp. contributed to dissimilarity between winter and autumn seasons; *Capnia* sp. contributed to difference between winter and spring seasons; *Ephemerella* sp. and *Rhithrogena* sp. contributed to dissimilarity between spring and autumn seasons. *analysis based on fourth-root transformation	
WINTER EPISODES (4) – Mar-00, Nov-00, Jan-01, Mar-01 Rosedale sub-reach Main channel	*2-D*: 0.10 *3-D*: 0.06	<u>*Two-way tests: Episode & Habitat*</u> *Episodes*: Global-R, $p < 0.001$ • Mar-00, Nov-00: $p = 0.007$* • Mar-00, Jan-01: $p = 0.03$* • Mar-00, Mar-01: $p = 0.04$* • all other pairwise comparisons NS	Abundances of most taxa declined between Nov-00 and Jan-01, and increased to Mar-01; Orthocladiinae contributed >30% to the dissimilarity between most months; *Baetis* sp., *Ephemerella* sp., and *Rhithrogena* sp. contributed substantially to dissimilarity between four pairwise contrasts; Naididae contributed to dissimilarity between Nov-00 and Jan-01.	
		Morphologic: Global-R, $p = 0.16$ *Hydraulic*: Global-R, $p = 0.03$* • riffle, exposed: $p = 1.0$ • riffle, normal: $p = 0.04$* • exposed, normal: $p = 0.18$	*Taeniomena* sp. and *Rhithrogena* sp. contribute to dissimilarity of riffles from other habitats; *Ephemerella* sp. contribute to dissimilarity between exposed and normal habitats; Orthocladiinae and Chironominae relatively uncommon in riffles.	Velocity correlated with taxa similarity ρ=0.13
SEPTEMBER 1999 Rosedale sub-reach Main channel	*2-D*: 0.06 *3-D*: 0.02	<u>*One-way test: Habitat*</u> *Morphologic*: Global-R, $p = 0.95$ *Hydraulic*: Global-R, $p = 0.80$		

Data Source	MDS stress	ANOSIM	SIMPER	BIO-ENV
MARCH 2000 Rosedale sub-reach Main channel	*2-D*: 0.12 *3-D*: 0.01	<u>*One-way test: Habitat*</u> *Morphologic*: Global-*R*, $p = 0.25$ *Hydraulic*: Global-*R*, $p = 0.07$* • riffle, exposed: $p = 1.0$ • riffle, normal: $p = 0.11$* → • exposed, normal: $p = 0.18$	Orthocladiinae contributed substantially to dissimilarity between all habitats; *Rhithrogena* sp., *Taenionema* sp., and *Cinygmula* sp. contributed to dissimilarity of riffles from other habitats; *Hydropsyche* sp. and *Baetis* sp. differentiated riffles and exposed habitats from normal units; *Ostrocerca* sp. differed between exposed and riffle habitats.	Velocity correlated with taxa similarity ρ=0.16
SEPTEMBER 1999 All sub-reaches Main channel	*2-D*: 0.14 *3-D*: 0.07	<u>*One-way test: Sub-reach*</u> *Sub-reach*: Global-*R*, $p = 0.05$* • Chilliwack, Rosedale: $p = 0.27$ • Chilliwack, Cheam: $p = 0.33$ → • Rosedale, Cheam: $p = 0.009$*	*Ephemerella* sp., *Rhithrogena* sp., *Hydropsyche* sp., and Orthocladiinae were more abundant in Rosedale and contributed to its dissimilarity with Chilliwack and Cheam; The presence of *Simulium* sp. differentiated the Chilliwack sub-reach.	
MARCH 2000 All sub-reaches Main channel	*2-D*: 0.12 *3-D*: 0.07	<u>*One-way test: Sub-reach*</u> *Sub-reach*: Global-*R*, $p = 0.001$* • Chilliwack, Rosedale: $p < 0.001$* • Chilliwack, Cheam: $p = 0.008$* → • Rosedale, Cheam: $p = 0.11$	*Ameletus* sp., Orthocladiinae, Naididae, and Tubifidae were more abundant in Rosedale and contributed to its dissimilarity with Chilliwack and Cheam; *Ephemerella* sp. and Chironominae had low abundance in Cheam and contributed to its dissimilarity with Chilliwack and Rosedale.	Velocity correlated with taxa similarity ρ=0.33
SEPTEMBER 1999 Rosedale sub-reach Main, side channel	*2-D*: 0.09 *3-D*: 0.05	<u>*One-way test: Channel Type*</u> *Channel*: Global-*R*, $p = 0.14$		

5.4 Discussion

5.4.1 Local-Scale Habitat Associations

The invertebrate assemblages associated with four physically distinct habitat units (riffle, bar head, flat bar edge, bar tail) were highly similar in community structure. This is in contrast to the significant amount of dissimilarity revealed among 10 sampling episodes spanning a 2-year period from 1999 to 2001. Community structure was highly variable during spring and autumn seasons, both between months in a single year and in the same month between years. In contrast, winter samples collected at low flow were similar in community structure and also contained the highest number of unique taxa and highest invertebrate densities. These patterns of richness and density are typical of the invertebrate community in Fraser River (Dymond 1998). Isolating winter samples for analysis and thereby eliminating seasonal variability revealed that, even within the winter season, community structure differed significantly between months. Only at the finest temporal scale of examination, a single month, was dissimilarity in community structure between habitat types detectable and significant. However, this result applies to winter months only; no association between the invertebrate community and habitat types was found in autumn months.

Significant temporal changes in community structure are a consequence of the life cycle of invertebrate populations, being one-year in duration for most taxa in the gravel reach and referred to as “univoltine”. Over the short life cycle, significant morphological, physiological, and behavioural changes take place as larvae mature in preparation for emergence, and the timing and nature of these changes varies significantly among taxa. Habitat requirements are likely to change over this period as well, in response to morphological and behavioural changes, which may explain in part why community structure showed no consistent relation with habitat types when examined at a coarse temporal scale. For example, the stonefly *Capnia* sp. contributed significantly to the dissimilarity between seasons and occupies primarily hyporheic habitat below the substrate surface for most of its life cycle until shortly prior to metamorphose and emergence in late winter (Stanford and Ward 1988).

The most significant life cycle event for aquatic insects is emergence, which coincides with the onset of freshet for most taxa in the gravel reach (Reece and Richardson 2000). Spring emergence explains the dramatic reduction in total density recorded between sampling in March and April 2000, and the virtual absence of many taxa from spring samples. Invertebrates also respond to comparatively smaller changes in discharge (Poff and Ward 1989, Boulton *et al.* 1992, Palmer *et al.* 1992), which may explain the synchronous reduction in density of virtually all taxa between August and September 2000, and subsequent increase between September and November 2000. Samples collected in September 2000 consisted of relatively few taxa and community structure was more similar to samples collected in spring than samples collected in August 2000, one month prior. Over the ten-day period prior to sampling on September 12th, river discharge was less than 3000 $m^3 s^{-1}$ and declining, but flow increased to 3500 $m^3 s^{-1}$ over 48 hours prior to sample collection. This increase in flow was rapid and occurred on the declining limb of the hydrograph. Shallow-water sampling locations were displaced laterally with the rise in water level, and invertebrates may not have had sufficient time to colonize the newly inundated bar surface. Invertebrates respond to the rising water levels in spring by migrating laterally across the shore-zone of gravel bars (Rempel *et al.* 1999), but the response time for lateral migration is uncertain, and whether or not a brief and unseasonable spike in discharge would elicit a similar response is unknown. Clearly, any response due to flooding in September 2000 was expressed as a temporary redistribution of organisms locally, rather than a widespread loss, because densities in November were higher than in August 2000.

Only by examining a single sampling episode during a period of highly stable flow and when invertebrate density was highest, March 2000, were differences in community structure among habitats detected. In the month prior to sampling in March 2000, discharge fluctuated less than 200 $m^3 s^{-1}$ and averaged 780 $m^3 s^{-1}$. With stable water levels, invertebrates had the opportunity to seek out favourable habitat with low risk of becoming stranded and minimal energy expenditure to maintain a stable depth stratum. This is in contrast to the four-week period prior to sampling in September 1999 during which time discharge declined over 2000 $m^3 s^{-1}$. Temporally variable water levels through most of the year poses a persistent risk to invertebrates of becoming stranded, and this risk may have been sufficiently great to restrict spatial partitioning among habitats, contributing to widespread taxon distributions (Death 1995, Blinn *et al.* 1995).

Despite detecting a significant difference in community structure among habitat types in March 2000, all samples shared a high level of similarity, almost 75%. Dissimilarity among habitat types was based on differences in the *relative abundances* of taxa occurring in all habitats, as opposed to particular taxa being distributed exclusively within one or a limited number of habitats. Hence, most taxa have widespread distributions and weak habitat specialization. Two factors may have contributed to these results. First, invertebrates may key on habitat characteristics scaled more appropriately with body size, and which are expressed heterogeneously at the scale of habitat units. Support for this hypothesis is that invertebrate community structure was correlated with velocity at several temporal scales of examination, which was found previously to represent the primary physical gradient along which the distribution of invertebrates is organized in Fraser River (Rempel *et al.* 2000).

The second factor likely contributing to the weak association between habitat structure and invertebrate assemblages is the fact that the four sampled habitat types are relatively similar in physical character. The difference in morphological and hydraulic attributes between riffles, bar heads, flat bar edges, and bar tails, while detectable, is less than between riffles and, for example, bays. Invertebrate assemblages in Fraser River are predicted to differ more significantly between habitat types exhibiting a greater difference in hydraulic and sedimentary character. The basis for this prediction is a study by Northcote *et al*. (1976) that reported different taxonomic composition in samples collected from sand-mud substrate compared to sand-gravel substrate at Lower Herrling Bar (see Chapter 2), as well as studies in small streams that have found differences in community structure between riffles and pools (Scullion *et al.* 1982). In the study by Northcote *et al*. (1976), average density was more than double in sand-mud substrate compared with sand-gravel substrate (1400 animals m^{-2} versus 600 animals m^{-2}).

High similarity in community structure among the four sampled habitat types, with most taxa having widespread distributions and weak habitat specialization, fits with predictions for environments characterized by large spatial heterogeneity and temporal variability (Poff and Ward 1990). Moreover, these ecological traits are expected to lend aquatic communities resilience to habitat variability and physical disturbance causing habitat change (Palmer and Poff 1997). It is predicted that the more highly specialized a population is to specific habitats, the less resilient it may be to physical disturbance. Hence, in a moderately and repeatedly disturbed environment like Fraser River, habitat specialization may be a detrimental trait. A mechanistic test of this prediction, with physical disturbance exemplified by gravel mining, was carried out in this study and is reported in Chapters 7 and 8.

5.4.2 Reach-Scale Differences In Community Structure

Although community structure within locally occurring habitat types was generally overlapping, congruence between the geomorphological organization of the gravel reach and invertebrate assemblages was revealed at a larger spatial scale corresponding to sub-reaches. Differences in community structure between sub-reaches were detected at the temporal scale of single sampling episodes both in winter and autumn months, although the taxa contributing most to reach-scale dissimilarity differed in each season. A longitudinal gradient corresponding to invertebrate community structure has been reported in several stream studies (Rice *et al.* 2001, Wright and Li 2002, Parsons *et al.* 2003, Heino *et al.* 2003), as well as a regional study by Northcote *et al*. (1976) extending from estuarine conditions at Sandheads to Hope, in Lower Fraser River. The strength of the association between invertebrate assemblages and sub-reaches was greater, compared to the association with habitat types within a sub-reach, because sub-reach differences were consistently identified in autumn and winter. Even more importantly, sub-reach dissimilarity was based, in part, on the restricted spatial distributions of particular taxa between sub-reaches. This trend was most prominent in September 1999, when *Simulium* sp. was collected only in the Chilliwack sub-reach, *Hydropsyche* sp. were absent from the Cheam sub-reach, and the mayflies *Rhithrogena* sp. and *Ephemerella* sp. were found in substantially higher proportions within the Rosedale sub-reach.

Dissimilarity in community structure between sub-reaches, but similarity between gravel bar sites within a sub-reach, suggest that sub-reaches represent a scale within which invertebrates, sampled at the habitat scale, appear to be homogeneously distributed. Furthermore, the moderate congruence with invertebrate assemblages qualifies sub-reaches as units of ecological significance in the hierarchical habitat classification. Sub-reaches are geomorphologically derived units based on differences in sediment transport regime and channel gradient, which differ in their expressions of sinuosity, lateral channel confinement, and sediment texture. These large-scale physical attributes may act collectively as a "filter", selecting a subset of the regionally available taxa that possess traits favourable to the prevailing habitat conditions (Poff 1997).

5.4.3 Summary

Invertebrate community structure showed modest dissimilarity among alluvial habitat types, which were shown in Chapter 4 to possess distinct morphological and hydraulic attributes. Community related differences among habitats were revealed only at a relatively fine temporal scale, during the winter season, when discharge was relatively stable. The same temporal scale of examination, but during autumn when discharge was more variable and invertebrate density was lower, revealed no association between the invertebrate assemblage and habitat structure. The limited range of habitat conditions over which sampling took place is believed to have contributed to the modest habitat association overall, but it remains uncertain whether or not the spatial scale of habitat units is, in fact, ecologically significant to invertebrates. It also is possible that increased sampling effort, particularly in spring and autumn when invertebrate densities are low, would yield greater congruence between habitat structure and invertebrate assemblages because the number of samples required for a specified precision is inversely related to invertebrate density (Downing 1979, Resh

1979). Regardless, the widespread distribution of taxa in the gravel reach is predicted to afford the invertebrate community some resilience to habitat variability and physical disturbance resulting in habitat change.

The fact that only a sub-set of habitat types was sampled, among which invertebrate taxa were widely distributed, precludes a definitive choice being made between the morphological and hydraulic classifications. Although significant dissimilarity was detected more consistently among hydraulic habitats, differences among morphological habitats were only marginally insignificant in some cases. Moreover, contrasts between riffles and flat bar edges showed greater dissimilarity than between riffles and bar tails. Given that the level of sampling effort may have inadequately described the spatial pattern of variance, marginally insignificant results may correspond with ecologically significant patterns (constituting a Type II error). Additional sampling effort is required to resolve this issue.

Congruence between the geomorphological organization of the gravel reach and invertebrate assemblages was revealed at a larger spatial scale corresponding to sub-reaches. Factors contributing to this large-scale pattern remain uncertain, but likely involve a combination of environmental gradients such as sediment texture and channel gradient, as well as biological factors such as species-specific dispersal ability, that together influence community structure at a regional scale. The congruence between sub-reaches and invertebrate assemblages qualifies sub-reaches as an ecologically important level in the hierarchical habitat classification.

Chapter 6. Habitat Associations of Fish

6.1 Introduction

The preceding chapter evaluated the premise that the assemblage of macroinvertebrates associated with physically distinct habitat types is unique and predictable. Results indicate that invertebrate distributions are correlated with velocity and that the species assemblage differs among habitat units defined by morphological or hydraulic attributes, but only at a fine temporal scale of examination. Chapter 5 also demonstrated that a larger-scale longitudinal gradient, which divides sub-reaches and influences the physical character of habitat types at a local scale, affects the distribution and abundances of invertebrates. Chapter 6 represents a continuation on this theme, and focuses on habitat associations of resident and anadromous fish species that occupy the gravel reach for rearing.

6.1.1 Background

Several studies have related the distribution of fish to habitat types, with earlier studies examining patterns of habitat use by individual species (e.g., Bisson *et al.* 1981) or species guilds (e.g., Lobb and Orth 1991, Caron and Talbot 1993) in small streams. More recent studies have shifted from a species-centric focus to examining the predictability of habitat-specific densities, biomass, and diversities both for species of interest and the entire species assemblage (Peterson and Rabeni 2001b, Walters *et al.* 2003). Such community-based examinations of habitat use have greater utility for extrapolating to site- or reach-specific estimates of productive capacity (Hankin and Reeves 1988, Randall and Minns 2000) and for estimating the impacts of habitat restoration or disturbance on the entire fish community (Reeves *et al.* 1995, Maddock 1999).

One recent study examined the predictability of the relation between habitat units and the fish assemblage in a small, warm-water stream in Missouri (Peterson and Rabeni 2001b). Although habitat types were found to be physically distinct along a longitudinal gradient from a 3rd- to 5th-order reach of the Missouri stream (Peterson and Rabeni 2001a), the associated fish assemblages varied significantly between reaches and seasons. Only at a local scale within reaches was the association between habitats and fish assemblages predictable. Consequently, Peterson and Rabeni (2001b) proposed a hierarchical model to improve the utility of the habitat classification, with reach-level constraints corresponding to longitudinal position.

A second study examined associations between fish species and spatially nested habitat features around vegetated islands in a regulated reach of the Upper Mississippi River (Johnson and Jennings 1998). The study demonstrated that local-scale characteristics associated with islands, particularly the aerial coverage and biomass of shoreline vegetation, were correlated significantly with fish density. However, fish density was not associated with larger-scale attributes such as island shape, shoreline development, and proximity to the main channel. In each of these studies, a hierarchical framework was advantageous for interpreting the association between habitat types and the distributions and abundances of fish. Moreover, fish distributions were more closely associated with local-scale habitat attributes than with reach-scale factors.

Results from several additional studies conducted at the landscape scale within a hierarchical framework support the expectation that local-scale factors explain the greatest amount of variation in fish community structure (McCormick *et al.* 2000, Oswood *et al.* 2000). Hawkins *et al*. (2000) summarized several studies, all of which were carried out in small streams, and concluded that although landscape-level classification accounted for more biotic variation than is expected by chance alone, the amount of variation related to landscape features was insignificant compared with the influence of local habitat factors, zoogeographic patterns, and land-use history. Hawkins *et al*. (2000) went on to predict that the composition of freshwater fish and invertebrate assemblages is most accurately predicted at the reach- or local-scale for assessment and biomonitoring purposes.

Both reach- and local-scale factors have been incorporated into the proposed hierarchical framework for habitat classification in the gravel reach of Fraser River, because of the expectation that large-scale morphological attributes and environmental gradients influence local habitat structure. The habitat classification therefore provides a comprehensive model for examining fish community structure and identifying physical factors affecting the distribution and abundances of species. The key element in the classification is the definition of physically distinct habitat types, which are identified at a spatial scale intended to be most ecologically relevant to fish. Results from Chapter 4 demonstrate that the physical character of locally occurring habitat units is influenced by a reach-scale longitudinal gradient corresponding to sub-reaches, as well as a lateral gradient associated with channel types. These physical gradients influence the physical character of habitat units, which in turn may affect the associated assemblage of fish species. Whether or not the distribution and abundances of fish species correspond with habitat units, and furthermore respond to differences in habitat character in the lateral and longitudinal dimensions, is the focus of this chapter.

6.1.2 Objectives

Chapter objectives are identical in character to those previously stated for Chapter 5. The first objective is to present habitat associations of common fish species occupying the gravel reach of Fraser River. The second objective is to determine whether or not the fish assemblages associated with habitat units are consistently identified and if the morphologic or hydraulic habitat classification more accurately characterizes the spatial distribution of species. The final objective is to compare the habitat-specific species assemblage among channel types and sub-reaches to determine if patterns are consistent in the lateral and longitudinal dimensions. Based on results from Chapter 4, which showed a greater physical difference in habitat characteristics between channel types compared with sub-reaches, community structure is expected to be most dissimilar in the lateral dimension.

6.2 Data Analysis

All fish data presented in this chapter were obtained by beach seine sampling between July 1999 and September 2001. Data were excluded that had been collected at sites recently disturbed by gravel mining, or that were collected specifically for paired day-night contrasts (**Appendix B**). The majority of sampling took place between July and September in each year, which overlapped with the summer and autumn seasons. Modest sampling occurred during winter and spring in 2000, as well as in spring 2001. Catch data were grouped according to “season”, defined by discharge (refer to Chapter 3), because fish sampling was time intensive and several weeks were required to systematically sample habitats at all study sites. This was in contrast to invertebrate sampling, where each sampling “episode” was completed in one or two days, with replicate samples collected at all sites and in all habitats. Water level fluctuations between seasons affected habitat availability, and some habitats, such as eddy pools, were relatively uncommon whereas others such as flat bar edge were common at all flows, and therefore sampled more frequently.

Sampling at sites in the Rosedale sub-reach took place throughout the study, whereas most sites in the Chilliwack and Cheam sub-reaches were added to the sampling program in the second year of data collection. Compared with sampling for invertebrates, fish sampling effort was adequately distributed among channel types to allow contrasts in fish community structure among habitats, channel types, and sub-reaches.

Fish data were analyzed following the same methods used for invertebrate data analysis. First, graphical and univariate statistical contrasts evaluated differences in community metrics between habitats and channel types. Second, habitat associations of individual species were examined, both including all size classes and also separating various size classes for each species, to evaluate the degree of habitat specialization and if habitat associations changed with age. Third, multivariate techniques were used to evaluate the similarity/dissimilarity in community patterns between and within habitats, channel types, and sub-reaches. Analyses were based primarily on the core dataset consisting of beach seines collected in the *autumn season from the main channel of the Rosedale sub-reach* (n = 122). Recall that the physical habitat data associated with these 122 beach seines were the principal focus of analyses in Chapter 4, which evaluated the physical distinctiveness of morphological and hydraulic habitat types.

6.2.1 Univariate Analysis of Community Metrics

Six community metrics were calculated based on fish samples collected by beach seine. Formulae are from Krebs (1998).

1. **Total Density (N)**: total number of fish collected in a beach seine haul divided by sampling area.
2. **Salmonid Index (*%Sal*)**: the percentage of fish in a beach seine haul belonging to the family Salmonidae (i.e., including salmon, trout, char, and whitefish).
3. **Margalef's Species Richness (*d*)**: as defined in Chapter 5.
4. **Simpson's Diversity (*D'*)**: as defined in Chapter 5.
5. **Shannon-Wiener Diversity (*H'*)**: as defined in Chapter 5.
6. **Pielou's Evenness (*J'*):** as defined in Chapter 5.

Two-factor analysis of variance (ANOVA) was applied to the six community metrics based on the core dataset to examine differences among habitats and channel types. Fish density was log ($X + 0.5$) transformed, the salmonid index was arcsine-square root transformed, and other variables did not require transformation to meet assumptions of normality and homogeneity of variances. A significance value of $\alpha = 0.008$ was applied, after Bonferroni's correction for multiple contrasts. Pairwise contrasts were made by Tukey's Test for unequal sample sizes.

6.2.2 Habitat Associations Of Common Fish Species

Species-specific fish density was compared among habitat types by graphical presentation and single-factor ANOVA to evaluate patterns of habitat use and determine if differences in density among habitats were significant. Comparisons were made based on the core dataset of main channel habitats in the Rosedale sub-reach. The densities of several species could not be normalized by transformation and, although ANOVA is relatively robust to violations of normality, the Kruskal-Wallis (K-W) non-parametric test was applied as well. In only 4 of 22 tests, results differed between the parametric and non-parametric tests (2 tests were significant by ANOVA but not by K-W, and 2 were significant by K-W but not ANOVA). In these cases, the most conservative test result is reported. A significance value of $\alpha = 0.002$ was applied to all tests, after Bonferroni's correction for multiple contrasts. Only significant results are reported in the text.

Graphical examination of species-habitat associations served a second purpose of evaluating whether associations changed with size (age), because a relatively broad size range of some species was collected. A size-related shift in habitat preference, referred to as an ontogenetic habitat shift, would introduce significant variability to an analysis of habitat associations. Size class divisions originally were intended to correspond with age classes in order to be "biologically meaningful". Size-frequency plots were constructed for each species (**Appendix D**) to identify age classes, in consultation with previous studies in the gravel reach (Northcote *et al.* 1978a), and biosystematic references (Carlander 1969, Scott and Crossman 1973). However, this procedure was deemed ineffective given that the goal was to identify size divisions only for those species demonstrating an ontogenetic shift. Instead, an iterative approach was adopted that involved constructing a sequence of plots for each species comparing fish density among habitat types, with the plots corresponding to size classes set at regularly spaced intervals. Bivariate plots were constructed for each species as well, relating fish size classes to water velocity, which was shown in Chapter 4 to represent the primary physical gradient that discriminated among habitat types. The plots were used to identify which species showed a size-related change in habitat association, and the approximate size at which the change occurred. For those species showing a change, the distribution was split into appropriate size classes that were incorporated in multivariate analyses, described below.

6.2.3 Reach-Scale Distribution of Chinook Salmon

The distribution of juvenile chinook salmon was of particular interest because no spawning takes place within the gravel reach, however, relatively high densities were encountered throughout the study. In addition to the examination of local-scale habitat use described above, chinook salmon density also was compared among channel types and sub-reaches to determine the reach-scale distribution of fish. Comparisons were made by graphical examination and two-factor ANOVA based on autumn densities (1999-2001). This comparison was supplemented by DNA analysis of chinook salmon collected in August 2000 from the main channel of each sub-reach. The analysis was carried out by Dr. J. Irvine (Department of Fisheries and Oceans, Pacific Biological Station) and detailed results are provided in **Appendix E**.

6.2.4 Multivariate Analysis of Community Structure

Of the 25 fish species identified in beach seine hauls, 16 were included in multivariate analyses of community structure. Seven excluded species were considered "rare" because less than 10 fish were captured throughout the study: brassy minnow, bridgelip sucker, bull trout, coho salmon, Dolly Varden, lamprey species, and white sturgeon. (Bull trout was confirmed by Dr. G. R. Haas (BC Ministry of Water, Land and Air Protection) and Dr. J. D. McPhail (Zoology, The University of British Columbia), who participated in sampling on a day the species was captured.) An additional two species, marine stickleback and pink salmon, were excluded because neither rears in the gravel reach for extended periods. Although a large number of anadromous marine stickleback was identified (n = 3909), almost all were captured in dense schools during a brief period in spring 2000, and are believed to use the gravel reach primarily as a migratory route to access smaller streams for spawning (Dr. J. D. McPhail, pers. comm.). Five of the 16 species were divided into two size classes for analysis: largescale sucker less than and greater than 75 mm, mountain sucker and northern pikeminnow less than and greater than 100 mm, and leopard dace and prickly sculpin less than and greater than 50 mm. Hence, analyses were based on 21 "species" groups.

Analyses were carried out using PRIMER-e software (version 5.2 Clarke and Gorley 2001), following the analytical framework described in Chapter 5, **Section 5.2.2**. The framework included MDS ordination based on the Bray-Curtis similarity matrix to characterize the similarity in community structure among habitat units, followed by the ANOSIM permutation test to compare community structure between groups, and then the SIMPER procedure to identify species contributing most to between-group dissimilarity. Lastly, the BIO-ENV procedure was applied to the normalized Euclidean matrix of environmental variables to determine the combination of physical variables that maximized the correlation with the Bray-Curtis similarity matrix. The following variables were included: mean velocity, mean water depth, log-bank angle, and the arcsine-square root transformed proportions of large cobble, cobble, gravel and sand/silt.

MDS ordination requires that no sample has a zero-catch, which occurred in 10 of 122 beach seine hauls. To account for the zero result, a 'dummy' species was included at a density of 0.0005 m^{-2} across all samples, which has a negligible effect on the overall analysis (Dr. K. R. Clarke, Primer-E Ltd., pers. comm.). The Bray-Curtis similarity matrix was based on square-root transformed densities of each species in beach seine hauls. The choice of data transformation was made following extensive exploratory analyses to evaluate which produced the lowest stress value in MDS ordination. The critical value for ANOSIM results was $\alpha = 0.1$, adjusted for the exploratory nature of the analysis.

Each beach seine haul was treated as an independent sample because a single haul often encompassed the majority or entire area of a habitat unit. This is in contrast to invertebrate sampling where each Surber sample covered a relatively small area (0.09 m^2) and replicate samples were necessary to adequately characterize each habitat unit. Analyses first examined fish community structure in main channel habitat units during autumn months, pooling observations from Rosedale sites across years (1999-2001). Community structure then was analyzed with increased temporal resolution by isolating an individual year. The year 2000 was chosen because sampling effort was the highest of all years and the flood hydrograph typified mean annual conditions. Exploratory analyses isolating 1999 and 2001 data also were carried out, and results are briefly described. Third, comparisons in community structure based on 2000 data were made among channel types within a sub-reach, and finally among sub-reaches within a each channel type, to determine if community patterns are consistent in the lateral and longitudinal dimensions.

All analyses were applied to two datasets, the first including all fish with appropriate size class divisions for those species showing an ontogenetic habitat shift, and the second isolating the smallest size class of each species to examine habitat associations of fish approximately 0-1 year in age. Juvenile fish typically have relatively narrow habitat requirements (Schiemer *et al.* 1991), and therefore may demonstrate more distinct habitat associations. Moreover, the habitat associations of fish species at this critical life stage are of interest. The upper size limit corresponding to juvenile fish was species-specific and based on biosystematic references cited above; the size limit was between 35 mm and 60 mm. Longnose dace was excluded from analyses of juvenile fish because small dace could pass through the mesh of the beach seine, thereby biasing catch rates.

6.3 Results

The percent representation of common species collected in beach seine hauls is shown in **Table 6-1**. Values are averaged across all sub-reaches, channel types, and habitats, for the purpose of presenting general patterns of relative abundance for species over the duration of the study. The sixteen species represented greater than 90% of all fish collected in beach seine hauls during all seasons of sampling except autumn 2001 and spring 2000. In autumn 2001, a higher than average number of very small "unidentifiable" fish was collected (711), along with over 100 small Catostomidae whose species identification (largescale or mountain sucker) was uncertain. The low (61%) total representation of the sixteen species in spring 2000 was due to an exceptional number of marine stickleback (3,909) that was encountered in dense schools in the Rosedale sub-reach.

Juvenile chinook salmon was captured most consistently of all fish species (69% of seine hauls, Chapter 3), and had the highest proportional representation, on average (21%), of all species in the gravel reach. Chinook was proportionately most abundant in winter months, but had a minimum 5% representation across all seasons of sampling. Other species belonging to the family Salmonidae were comparatively rare in the gravel reach, the exception being migrating chum salmon fry in late winter and early spring when they represented greater than 20% of fish in beach seine hauls. Pink salmon fry also were highly abundant in late winter and spring of 2000, but most fry passed through the mesh of the beach seine and therefore catch rates are unreliable.

Leopard dace was second only to chinook salmon in both its frequency of occurrence (66% of seine hauls, Chapter 3) and proportional representation in beach seine hauls. It consistently represented between 7% and 28% of the total catch and, numerically, was the most abundant species counted in the study (6405 fish). All other cyprinid species, including longnose dace, northern pikeminnow, peamouth chub, and redside shiner, had a high proportional representation in summer and autumn samples but were relatively rare during winter and spring. Mountain sucker showed a similar trend, being rare in beach seine hauls during winter and spring (< 1%) but consistently representing between 5% and 10% of all fish captured during summer and autumn. The proportional representation of mountain sucker was relatively consistent in all years of sampling, and a total of 2637 fish representing this blue-listed species were counted. Largescale sucker ranked fourth of all species in numerical abundance (5178) and occurred in 33% of all beach seine hauls (Chapter 3). However, it had low percent representation in all seasons except autumn of 2000 and 2001. During these seasons, significant numbers of fish were collected in a relatively small number of samples (e.g., >1500 largescale sucker counted in a single open nook at Queens Bar in August 2000).

Table 6-1. Average percent (%) representation of the sixteen most common fish species collected by beach seine in the gravel reach (1999-2001).

Fish Species (total counted)	**Summer**			**Autumn**			**Winter**		**Spring**	**Mean**
	1999	**2000**	**2001**	**1999**	**2000**	**2001**	**2000**	**2001**	**2000**	**-**
chinook salmon (6268)	**29.0**	**18.5**	**13.3**	**14.8**	**5.1**	**10.2**	**47.4**	**30.1**	**20.2**	**20.9**
sockeye salmon (201)	**2.6**	0.9	1.9	0.1	0.3	0.2	0	0.3	0.5	0.8
chum salmon (2990)	0	0	0	0	0	0	**23.2**	**32.1**	**25.3**	**8.9**
rainbow trout (105)	0.3	0.6	0.1	0.3	0.1	0.3	0	0	0.2	0.2
cutthroat trout (41)	0	0.2	0.1	0	0.1	0.04	3.2	0.3	0.1	0.4
mountain whitefish (579)	0	**2.1**	**3.0**	1.0	1.0	1.6	**3.7**	1.5	1.0	1.7
largescale sucker (5178)	0.1	**2.1**	**2.3**	1.4	**27.1**	**10.2**	0	**2.6**	0.3	**5.1**
mountain sucker (2637)	**8.6**	**5.0**	**4.5**	**7.7**	**7.7**	**5.4**	0.5	0	0.8	**4.5**
northern pikeminnow (1190)	1.4	**2.3**	1.7	**5.5**	**2.8**	1.8	0	0.4	0.1	1.8
redside shiner (4761)	**21.7**	**13.0**	**6.2**	**27.1**	**5.6**	**8.0**	0	0	1.9	**9.3**
peamouth chub (5539)	1.1	**8.2**	**5.8**	**12.4**	**15.9**	**19.3**	0	0.2	**2.2**	**7.2**
leopard dace (6405)	**25.8**	**21.8**	**27.5**	**17.5**	**12.3**	**10.2**	**16.8**	**24.8**	**6.5**	**18.1**
longnose dace (3017)	**5.6**	**12.8**	**24.3**	1.4	**7.5**	**13.0**	0.5	0.3	0.4	**7.3**
prickly sculpin (432)	1.0	1.4	1.5	1.1	0.7	0.5	1.6	0.4	1.1	1.0
coastrange sculpin (39)	0.1	0.04	0	0.04	0.03	0	1.1	0	0.3	0.2
threespine stickleback (2203)	**2.6**	**3.4**	0.2	**9.0**	**9.6**	**7.8**	0.5	1.2	0.5	**3.9**
Total (40,974)	**100**	**92.3**	**92.3**	**99.2**	**95.7**	**88.7**	**98.4**	**94.0**	**61.1**	**91.3**

Bold type indicates a sampling period in which the taxon represented >2%, on average, of all fish collected.
Season is defined by discharge (refer to Chapter 3).

6.3.1 Univariate Analysis of Community Metrics

Fish density varied significantly among habitat types ($p < 0.0001$, **Table 6-2**), with channel nook and open nook units supporting higher densities than bar head, bar tail, and flat bar edge units (**Figure 6-1**). Average density did not differ among channel types, but a significant interaction between habitats and channel types was found (**Table 6-2**). Results from Tukey's test indicated that average density in channel nooks was lower within summer channels than in the main and side channels.

The proportion of salmonids in beach seine hauls did not differ among habitat types, but was significantly higher in side channels compared to the main channel (**Table 6-2**). This was particularly notable for flat bar edges, bar tails, and channel nooks (**Figure 6-1**).

Both Shannon-Wiener's and Simpson's diversity were statistically similar among habitats and channel types (**Table 6-2**), but values were notably higher in flat bar edge and bar tail units of side channels, and lower in open nooks and channel nooks of summer channels (**Figure 6-1**). Diversity was most consistent among channel types within bar head and eddy pool units.

Pielou's evenness was similar among all habitats and channel types (**Table 6-2**) but, similar to measures of diversity, higher values were observed in flat bar edge and bar tail units of side channels, as well as in bar heads and eddy pools of summer channels (**Figure 6-1**).

Margalef's richness varied among channel types and habitats, but differences among groups were not significant at the critical level of $\alpha = 0.008$, adjusted by Bonferroni's correction for multiple contrasts. Average richness was highest in side channels and lowest in summer channels, and richness averaged almost 5 species per sample among habitats (**Figure 6-1**). In the main channel, bar head and eddy pool units had highest average richness, whereas richness was highest in flat bar edges, bar tails, and eddy pools of side channels. Richness was consistently high in eddy pools within all channel types.

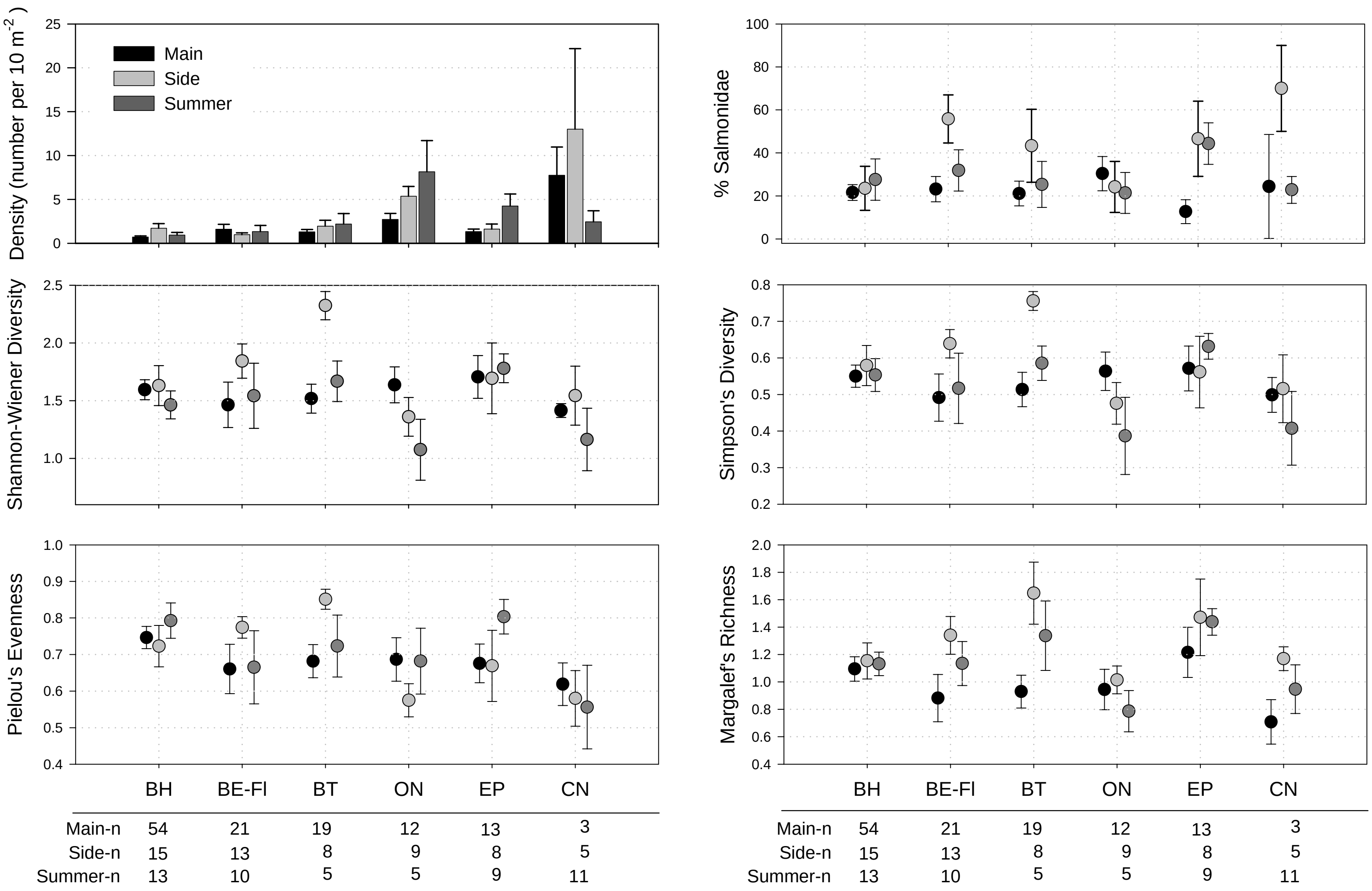


Figure 6-1. Community metrics (mean ± SE) characterizing fish samples collected by beach seine in main channel habitat units of the Rosedale sub-reach during autumn, 1999-2001. Two-letter habitat abbreviations are given in Chapter 4 (‘n’: sample size).

Table 6-2. Two-factor ANOVA examining differences in community metrics among habitats and channel types. Analysis based on data collected in autumn from the Rosedale sub-reach (1999-2001).

Variable	**SS**	***df***	**MS**	**F**	***p***
Density					
Channel	1.42	2	0.71	1.29	0.28
Habitat	31.46	5	6.29	11.48	<0.0001*
Channel*Habitat	15.26	10	1.53	2.78	0.003*
Error	117.81	215	0.55		
% Salmonidae					
Channel	2.60	2	1.30	6.28	0.002*
Habitat	1.38	5	0.28	1.33	0.25
Channel*Habitat	3.51	10	0.35	1.70	0.08
Error	44.48	215	0.21		
Shannon-Wiener Diversity					
Channel	2.71	2	1.35	2.99	0.05
Habitat	4.00	5	0.80	1.77	0.12
Channel*Habitat	5.28	10	0.53	1.17	0.32
Error	97.34	215	0.45		
Simpson's Diversity					
Channel	0.15	2	0.07	1.52	0.22
Habitat	0.35	5	0.07	1.43	0.21
Channel*Habitat	0.55	10	0.05	1.13	0.34
Error	10.42	215	0.05		
Pielou's Evennness					
Channel	0.02	2	0.009	0.17	0.84
Habitat	0.45	5	0.09	1.80	0.11
Channel*Habitat	0.47	10	0.05	0.93	0.51
Error	10.80	215	0.05		
Margalef's Richness					
Channel	2.07	2	1.03	3.54	0.03
Habitat	4.09	5	0.82	2.80	0.02
Channel*Habitat	2.74	10	0.27	0.94	0.50
Error	62.76	215	0.29		

* significant at the $\alpha = 0.008$ level (adjusted by Bonferroni's correction for multiple contrasts).

6.3.2 Habitat Associations of Fish Species

Chinook and sockeye salmon were captured only as juveniles by beach seine and no shift in habitat association related to size was detected. Chinook density was lowest in open nooks and highest in eddy pools, and differences in average density among “normal” hydraulic habitats (flat bar edge, bar tail, open nook) were apparent (**Figure 6-2**). Higher chinook density in eddy pools compared to bar heads, bar tails, and open nooks, was marginally insignificant ($p = 0.005$) at an adjusted critical level of $\alpha = 0.002$. Juvenile sockeye salmon captured in the gravel reach are considered “accidental” because the species *typically* rears in lakes before migrating to sea (some stocks such as Pitt River sockeye rear predominantly in rivers). Sockeye was captured only in channel nooks within the Rosedale sub-reach in autumn ($p < 0.0001$, **Figure 6-2**), and also in bays in other sub-reaches and during different seasons. Densities of rainbow trout and cutthroat trout were low, making it impractical to evaluate separate size classes for each species. Both species were absent from open nook and channel nook units, and cutthroat was absent from flat bar edge units as well. Rainbow trout density was relatively similar among bar head, flat bar edge, bar tail, and eddy pool units. Average cutthroat trout density was higher in eddy pools compared to bar heads and bar tails. Differences in density among habitats were not significant for either trout species. Mountain whitefish was equally common in bar head, flat bar edge, and open nook units, and most common in bar tails. Densities were statistically similar among habitat types.

Several resident species were associated with low velocity habitats over the range of sizes captured by beach seine. Largescale sucker <75 mm were observed strictly in open nooks and channel nooks (**Figure 6-3**) and habitat-specific differences in density were significant ($p = 0.001$). Larger sucker >75 mm were collected principally in channel nooks by beach seine, and also commonly in bays by gill net. Juvenile prickly sculpin <50 mm also occurred almost strictly in open nook and channel nook units (**Figure 6-3**), and average density in channel nooks was significantly higher than in other habitats ($p < 0.0001$). Larger prickly sculpin >50 mm were associated mainly with eddy pools, and the difference in eddy pool density compared with all other habitats was significant ($p < 0.0001$).

A size-related shift in habitat association was detected for northern pikeminnow, with densities of fish <100 mm highest in bar tails and channel nooks, and densities of fish >100 mm highest in eddy pools (**Figure 6-3**). Differences in density among habitats were not significant for smaller northern pikeminnow, and the higher density of fish >100 mm in eddy pools was marginally insignificant ($p = 0.009$). Peamouth chub and threespine stickleback were collected almost exclusively in channel nooks over the entire size range of fish in beach seine hauls (**Figure 6-3**). For each species, average density in channel nooks was significantly higher than all other habitats ($p < 0.0001$). Longnose dace typically is found in high velocity habitats (Mullen and Burton 1995) and, in Fraser River, riffles sampled by electro-shocking in 1999 hosted a high proportion of longnose dace. In beach seine hauls, density was highest in open nooks over the range of sizes observed (12-85 mm, **Figure 6-3**). Fish too small for accurate species identification were found almost exclusively in channel nooks in the Rosedale sub-reach (**Figure 6-3**), and in open nooks during other seasons and in other sub-reaches.

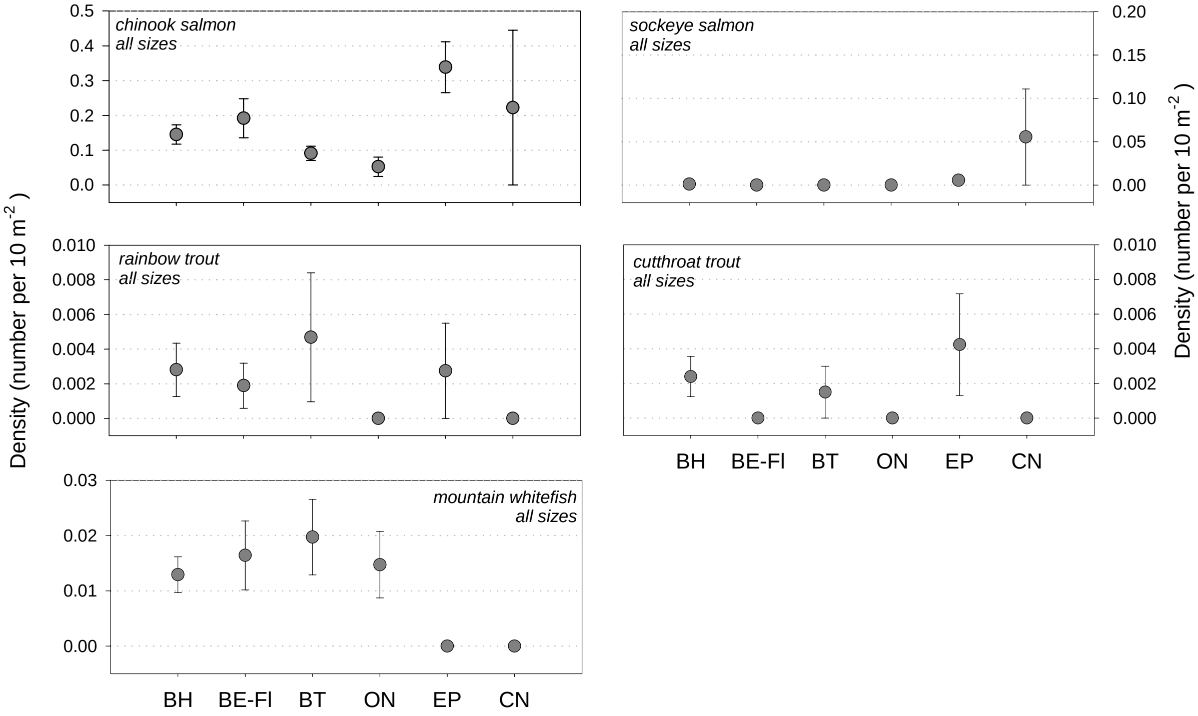


Figure 6-2. Average (± SE) density of salmonid species in the main channel of the Rosedale sub-reach, collected by beach seine in the autumn season. The x-axis approximately corresponds with a hydraulic gradient from high (left) to low velocity (right).

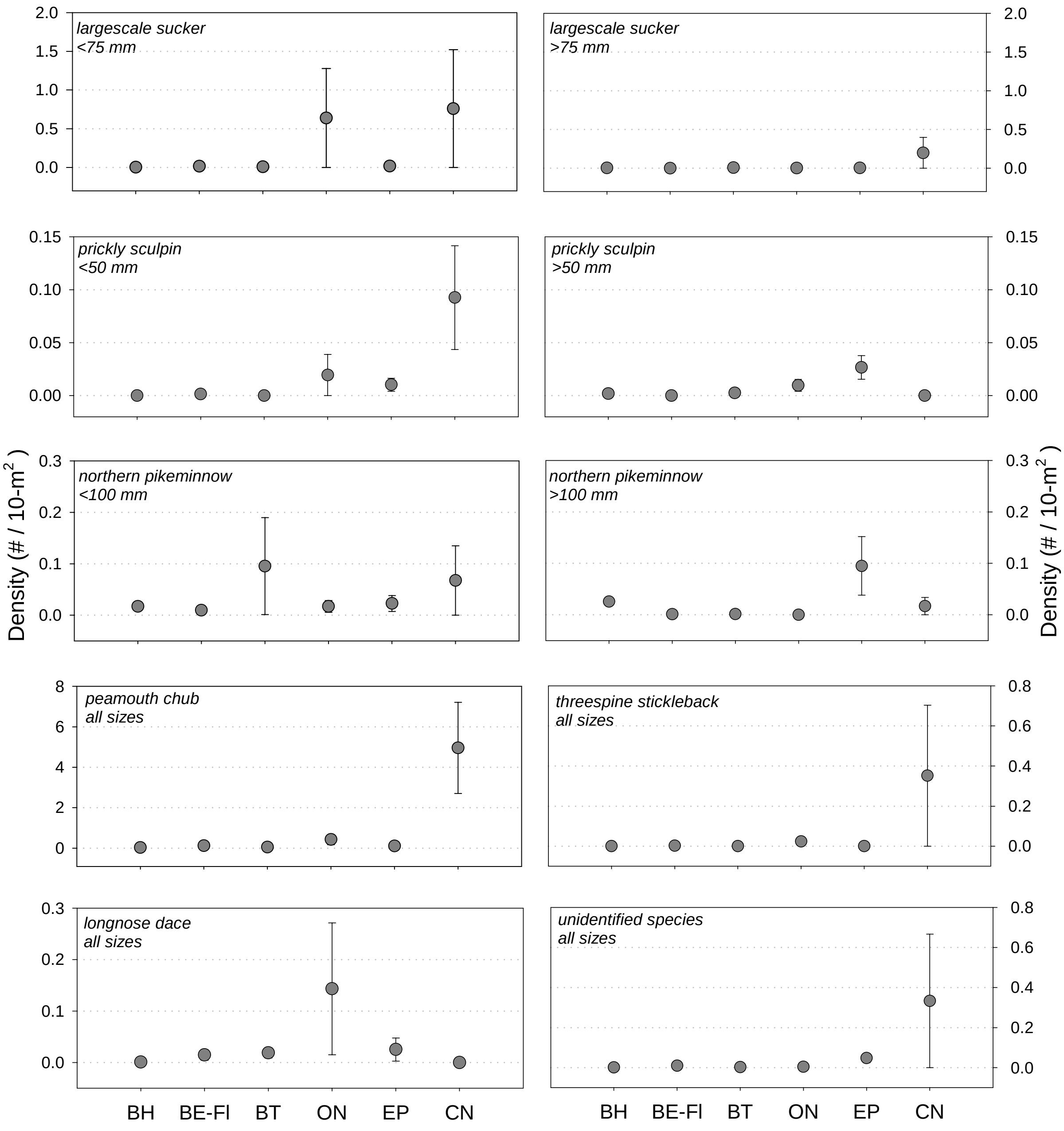


Figure 6-3. Average (± SE) density of fish species in the main channel of the Rosedale sub-reach, collected by beach seine in the autumn season. The x-axis approximately corresponds with a hydraulic gradient from high (left) to low velocity (right).

Mountain sucker was virtually absent from hydraulically "sheltered" habitats and open nooks (**Figure 6-4**). A shift in habitat association was detected at approximately 100 mm length, with highest densities of smaller fish in flat bar edge units, and larger mountain sucker found almost exclusively in bar tails. Differences in density among habitats were insignificant for both size groups of mountain sucker. Habitat associations of leopard dace also changed with size, and densities of fish >50 mm were highest in flat bar edge and open nook units (**Figure 6-4**). Larger leopard dace clearly differentiate among "normal" habitats based on the comparatively low density in bar tail units. Juvenile leopard dace <50 mm were most frequently associated with low velocity conditions and almost exclusively found in open nooks (**Figure 6-4**). Habitat related differences in density for both size classes of leopard dace were statistically insignificant. Densities of redside shiner were highest in bar tail, channel nook and eddy pool units (**Figure 6-4**) and differences in density among habitats were insignificant.

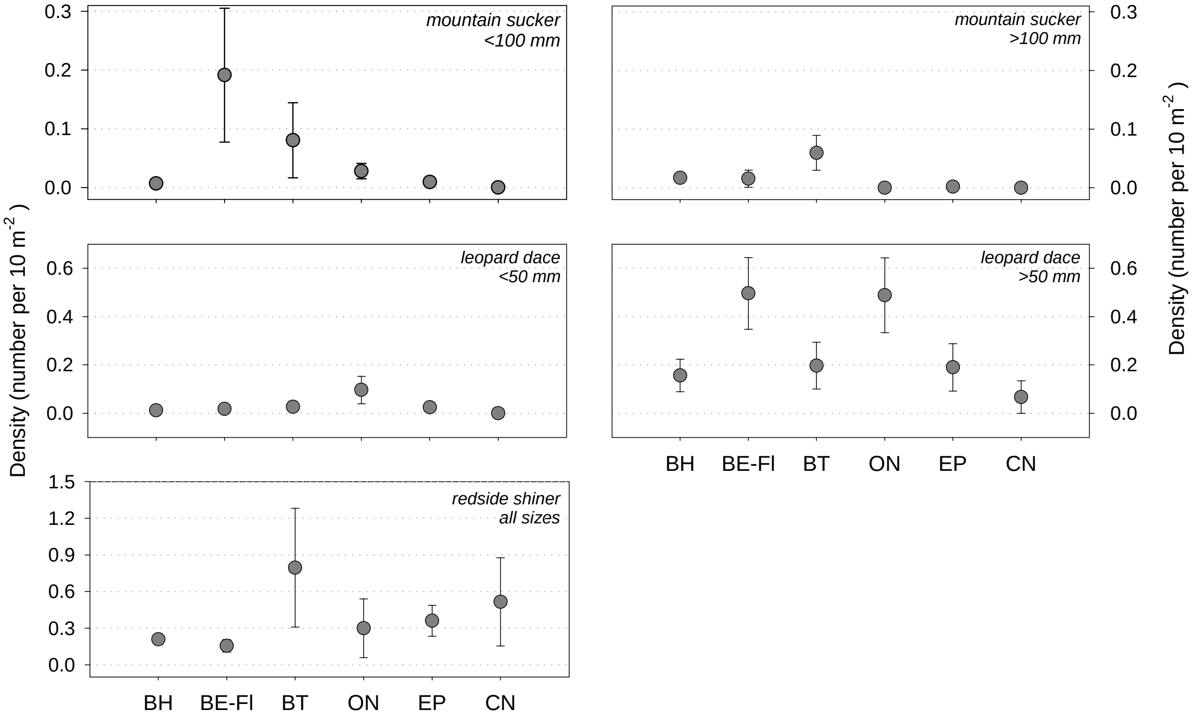


Figure 6-4. Average (± SE) density of fish species in the main channel of the Rosedale sub-reach, collected by beach seine in the autumn season. The x-axis approximately corresponds with a hydraulic gradient from high (left) to low velocity (right).

6.3.3 Reach-Scale Distribution of Chinook Salmon

The distribution of juvenile chinook salmon was of particular interest because no spawning takes place within the gravel reach, however, relatively high densities were encountered throughout the study. The Cheam sub-reach had comparatively higher main channel density (**Figure 6-5a**) but lower side channel density (**Figure 6-5b**) than downstream reaches. Within summer channels, density was comparable between the Rosedale and Cheam sub-reaches, but was lower in the Chilliwack sub-reach (**Figure 6-5c**). Differences among channels and sub-reaches were not significant (**Table 6-3**) based on two-factor ANOVA.

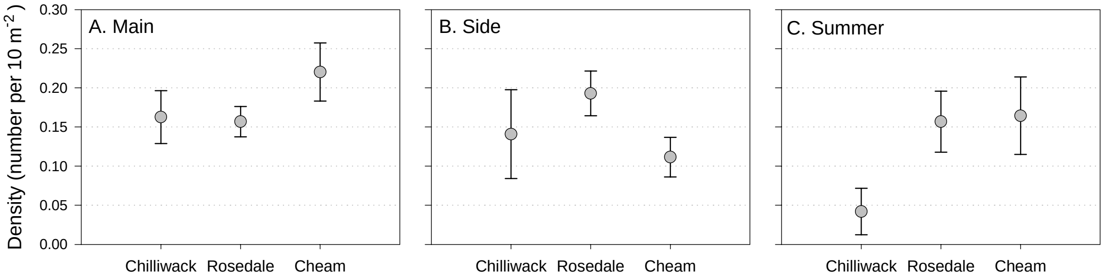


Figure 6-5. Comparison of juvenile chinook salmon density between sub-reaches. Data are grouped according to channel type in each plot (based on autumn data, 1999-2001).

Table 6-3. Two-factor ANOVA on log-transformed data comparing chinook salmon density among channel types and sub-reaches.

Variable	**SS**	***df***	**MS**	**F**	***p***
Channel	0.26	2	0.13	1.77	0.17
Sub-Reach	0.21	2	0.11	1.46	0.23
Channel*Sub-Reach	0.43	4	0.11	1.46	0.21
Error	37.22	509	0.07		

Data collected in autumn, 1999-2001. All habitat types pooled for analysis.

The majority of chinook salmon collected in the main channel during autumn months were spawned in tributaries of the middle Fraser Basin (**Table 6-4**), particularly the Stuart and Nechako Rivers (see **Appendix E**). The North Thompson system was the second largest producer of juvenile chinook that rear in the gravel reach, the majority originating from the Raft and Clearwater River populations. A higher proportion of North Thompson fish was collected in the Rosedale and Cheam sub-reaches, whereas Middle Fraser chinook were more common in the Chilliwack and Cheam sub-reaches.

Table 6-4. Results from DNA analysis* of juvenile chinook salmon collected in August 2000 from three sub-reaches. Values are the percent (%) of fish originating from upstream sub-basins of the Fraser Basin. Number of fish from each sub-reach is in parentheses.

Sub-Basin	**Chilliwack (78)**	**Rosedale (125)**	**Cheam (74)**	**Average**
Upper Fraser	4.6	1.8	2.8	3.0
Middle Fraser	82.6	72.4	77.9	77.7
Lower Fraser	0	0	0	0
North Thompson	10.9	21.0	17.3	16.4
South Thompson	0	2.3	0.5	0.9
Lower Thompson	1.9	2.5	1.5	2.0
Birkenhead	0	0	0	0

* Laboratory analysis based on a 50-stock Fraser baseline dataset with 13 microsatellite loci (**Appendix E**).

6.3.4 Multivariate Analysis of Community Structure

6.3.4.1 Habitat Associations – All Years Combined

Ordination by non-metric multi-dimensional scaling identified both annual and habitat-related patterns in fish community structure, but temporal and spatial patterns were muted by overall variability among samples. The stress value (0.23) indicates moderate distortion in the portrayal of similarity between samples; therefore the 3-D ordination was examined to assist in interpretation (**Appendix F**). Samples collected in 1999, a year of prolonged and above-average flooding, were clustered more tightly than samples from 2000 and 2001 (**Figure 6-6a**); samples collected in 2000 and 2001 generally overlapped in ordination space. Two-way ANOSIM indicated that the apparent dissimilarity between years was marginally insignificant (global-R, $p = 0.12$, **Table 6-5**).

Morphological habitat types showed relatively weak groupings in the MDS ordination, but a general hydraulic gradient separated habitats along the horizontal axis of **Figure 6-6b,** particularly bar head units. There was modest differentiation among habitat types representing the "normal" hydraulic class, with open nooks grouped separately from flat bar edge and bar tail habitats (more clearly shown in the 3-D plot). Channel nooks were isolated from all other habitat types, whereas eddy pools were plotted in an intermediate position between bar heads and "normal" habitats. The 3-D plot (**Appendix F**) revealed improved separation of eddy pool units from bar heads in the *z*-plane. Two-way ANOSIM indicated that open nooks and channel nooks each were dissimilar in community structure from all other habitat types (**Table 6-5**). Eddy pools also were dissimilar from flat bar edge units ($p = 0.04$), and differences between eddy pools and bar tails were marginally insignificant ($p = 0.15$). Eddy pools shared greatest similarity with bar heads, and the habitats are most often geographically associated with one another. The BIO-ENV procedure found that the combination of large cobble, sand/silt, bank angle, and velocity produced the largest rank correlation ($\rho = 0.39$) between the Bray-Curtis and Euclidean sample similarities. Therefore, these variables had highest correlation with fish community structure.

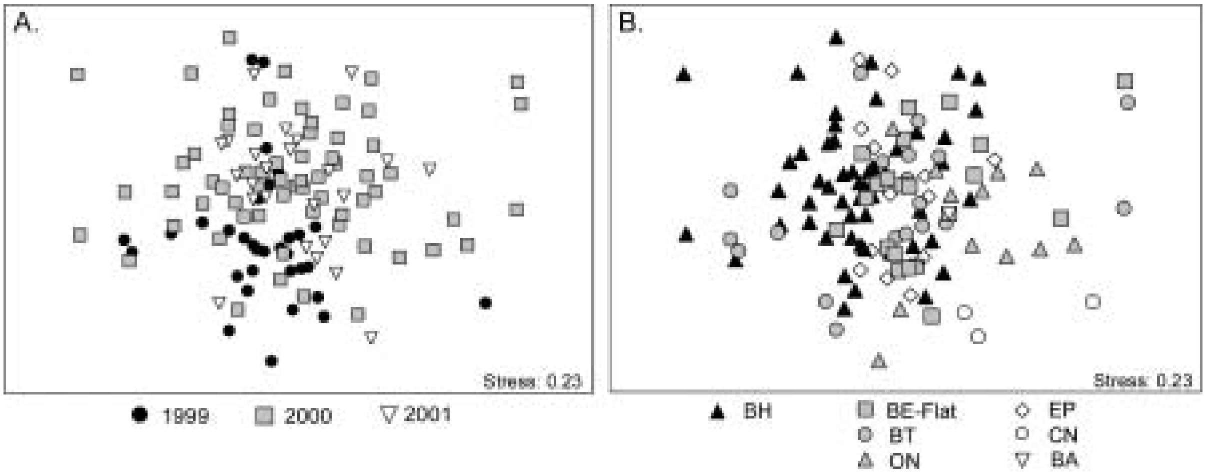


Figure 6-6. MDS ordination of square-root transformed fish densities in main channel samples collected in the Rosedale sub-reach during autumn months (1999-2001). Each plot shows the identical ordination with samples classified according to A) years, and B) morphological habitats. The shading of morphological habitats corresponds to hydraulic habitat classes.

The significant dissimilarity between members of the "sheltered" hydraulic class and among members of the "normal" hydraulic class implied that hydraulic habitat classes encompass significant within-group variability. This was supported by two-way ANOSIM results (**Table 6-5**), in which exposed and normal groups had distinct community structure (though being more similar in hydraulic character), but exposed and sheltered groups were found to have similar community structure (though very dissimilar hydraulic character).

Table 6-5. Two-way ANOSIM results comparing community patterns between years and habitats, based on main channel data collected in autumn from the Rosedale sub-reach, 1999-2001.

Comparison	**Possible Permutations**	**# Cases ≥ Observed**$^{+}$	***R*-Statistic**	***p*-value**
Differences between years averaged across habitat types				
Global			0.052	0.12
*Differences between morphologic habitat types averaged across years***				
Global			0.129	0.017*
Open Nook, Bar Head	>999	2	0.343	0.003*
Open Nook, Flat Bar Edge	>999	73	0.186	0.074*
Open Nook, Bar Tail	>999	76	0.146	0.077*
Open Nook, Eddy Pool	>999	1	0.541	0.002*
Open Nook, Channel Nook	105	1	0.927	0.01*
Channel Nook, Bar Head	>999	8	0.611	0.009*
Channel Nook, Flat Bar Edge	330	7	0.658	0.021*
Channel Nook, Bar Tail	320	14	0.560	0.044*
Channel Nook, Eddy Pool	120	4	0.591	0.033*
Eddy Pool, Flat Bar Edge	>999	43	0.141	0.044*
Eddy Pool, Bar Tail	>999	148	0.078	0.149
Differences between hydraulic habitat types averaged across years				
Global			0.134	0.001*
Exposed, Normal	>999	0	0.131	0.001*
Exposed, Sheltered	>999	175	0.081	0.18
Normal, Sheltered	>999	23	0.177	0.024*

+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.
* significant at the adjusted value of $\alpha = 0.1$.
** only contrasts with $p < 0.20$ are reported

The SIMPER procedure was used to identify species contributing most to the dissimilarity between morphological habitats. Only habitat pairs shown by ANOSIM to be dissimilar were examined, and bubble plots of species' densities are included in **Appendix F** to assist in visual interpretation of the results (refer to **Section 6.3.2** for untransformed abundance data). SIMPER analysis indicated that the dissimilarity of open nooks was largely attributed to high densities of largescale sucker <75 mm, leopard dace (all sizes), and peamouth chub (**Table 6-6**). For eddy pools, relatively high densities of chinook salmon and redside shiner and low densities of leopard dace and peamouth chub contributed to the dissimilarity with flat bar edge units. Lower densities of largescale sucker, redside shiner, and chinook salmon in flat bar edge units, and higher densities of leopard dace and mountain sucker <100 mm, all contributed to the dissimilarity of flat bar edges with eddy pools and channel nooks. Peamouth chub was found almost exclusively in channel nooks and contributed >27% to channel nook dissimilarity with all other habitats.

Table 6-6. Results of SIMPER analysis based on square-root transformed data indicating the contribution (%) made by species to the *dissimilarity* in community structure between habitats in the main channel of the Rosedale sub-reach (autumn data, 1999-2001).

Species	**ON/BH**	**ON/BE**	**ON/BT**	**ON/EP**	**ON/CN**	**BH/CN**	**BE/CN**	**BE/EP**	**BT/CN**	**EP/CN**
% Diss.	74.6	66.5	73.9	71.4	77.0	85.4	83.8	65.2	87.5	78.6
leopard dace >50	**+13.6**	**-13.8**	**+14.1**	**+13.7**	**+8.3**	-4.4	**+7.2**	**+15.2**	+3.7	+4.6
largescale sucker <75	**+13.3**	**+13.8**	**+13.2**	**+12.6**	**+12.1**	**-10.5**	**-9.8**	-3.9	**-9.9**	**-10.1**
redside shiner	**+12.5**	**+11.2**	**-15.2**	**-12.9**	**-9.1**	**-8.4**	**-7.7**	**-13.6**	**+9.9**	**-8.8**
peamouth chub	**+11.6**	**+12.5**	**+11.7**	**+11.0**	**-27.8**	**-36.0**	**-32.5**	**+9.4**	**-34.4**	**-33.2**
longnose dace	**+9.3**	-8.8	**+9.5**	+8.8	+5.8	+3.1	+3.4	-7.5	+3.5	+3.9
leopard dace <50	+8.8	**+8.9**	+8.6	+7.2	+5.2	+0.8	+3.5	**+7.9**	+3.3	+2.0
chinook salmon	-8.2	-8.8	-6.7	**-11.5**	-5.9	**-7.2**	-7.1	**-12.2**	**-6.5**	**+8.6**
threespine stickleback	+2.7	+2.8	+2.5	+2.3	**-7.8**	**-8.8**	**-8.1**	-0.8	**-8.3**	**-8.3**
mountain sucker<100	+3.4	-6.9	-4.6	+3.2	+1.8	+0.5	+3.7	+7.3	+1.5	+0.7
prickly sculpin<50	+1.1	+1.3	+1.1	+2.0	-4.9	-5.7	-5.1	-1.6	-5.4	-5.0

% Diss.: overall dissimilarity between habitat pairs, expressed as a percentage.
+/- indicates the direction of difference in species density between habitats, e.g., ON/BH: +13.6 indicates that leopard dace density was higher in open nooks and contributed 13.6% to the dissimilarity with bar heads.
Bolded text highlights the five species contributing most to dissimilarity *between pairs*.
Listed species contributed substantially to the dissimilarity between at least one habitat pair.

The identical analysis applied to juvenile fish produced results (not shown) similar to those based on all size ranges of fish, but with several differences worthy of mention. First, community structure in each year was dissimilar ($p < 0.001$), and species contributing most to between-year dissimilarity were (listed in order of importance) chinook salmon, northern pikeminnow, prickly sculpin, and leopard dace. Densities of chinook salmon were highest in 1999, and lowest in 2001. Densities of prickly sculpin followed the same year-to-year pattern. Northern pikeminnow were most abundant in 2000 whereas densities of juvenile leopard dace were highest in 1999. The fish assemblage associated with open nooks was dissimilar from all other habitat types and high densities of juvenile northern pikeminnow, prickly sculpin, and leopard dace relative to other habitats contributed most to the significant dissimilarity.

6.3.4.2 Habitat Associations –Year 2000

Patterns revealed by MDS ordination of fish data collected in autumn 2000 were consistent with those from the analysis of autumn samples in all years combined. The majority of bar heads were separated from open nooks, flat bar edges, and bar tails in ordination space, and eddy pools were plotted in an intermediate position between these groups (**Figure 6-7**). Several open nooks were clustered separately from other "normal" habitat units, and ANOSIM results showed that open nooks were dissimilar in community structure compared to most other habitats except flat bar edge (**Table 6-7**). The dissimilarity between open nooks and channel nooks was marginally insignificant ($p = 0.14$), though contrasts of all habitats with channel nooks are constrained by low sample size in 2000. Differences between bar heads and open nooks contributed to the significant dissimilarity between "exposed" and "normal" hydraulic habitats ($p < 0.001$, **Table 6-7**), and the similarity between eddy pools and bar heads was reflected in the non-significant contrast between "exposed" and "sheltered" hydraulic habitats ($p = 0.33$). Results from the BIO-ENV correlation analysis also were similar to the analysis of all years combined; the combination of large cobble, sand/silt, bank angle, and velocity yielded the highest correlation ($\rho = 0.43$) between the environmental and biological similarity matrices.

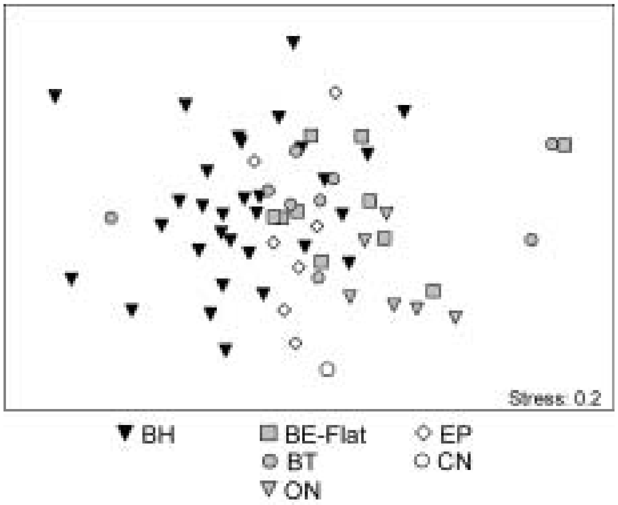


Figure 6-7. MDS ordination of square-root transformed fish densities in main channel samples collected in the Rosedale sub-reach during autumn in 2000. Samples are classified according to morphological habitats. The shading of morphological habitat units corresponds to hydraulic habitat classes.

One-way ANOSIM was used to evaluate differences in community structure among gravel bar sites in the Rosedale sub-reach. The global *R*-statistic was significant ($p = 0.09$, **Table 6-7**), and all pairwise contrasts with Hamilton Bar, located near the upstream end of the Rosedale sub-reach, revealed dissimilarity, including the contrast with nearby Big Bar (**Table 6-7**). Subsequent analysis by SIMPER indicated that the abundances of several species were higher along the main channel perimeter of Hamilton Bar compared to most other bars in the sub-reach. These species, listed in order of decreasing contribution to site dissimilarity, are chinook salmon, peamouth chub, redside shiner, mountain sucker >100 mm, and leopard dace >50 mm. Densities of largescale sucker <75 mm and leopard dace <50 mm were lower at Hamilton Bar, and also contributed to site dissimilarity.

Significant dissimilarity among morphological habitats prompted the use of SIMPER analysis to evaluate species-specific contributions to habitat-related dissimilarity. Results were similar to the previous analysis combining all years (**Table 6-6**), but several additional patterns are worthy of mention (refer to **Appendix F** for complete results). Higher densities of chinook salmon in eddy pools and channel nooks contributed to the dissimilarity of these habitats with bar head, flat bar edge, and bar tail units. Sockeye salmon, found almost exclusively in channel nooks, contributed to the dissimilarity of channel nooks with bar heads and flat bar edges.

Table 6-7. Two-way ANOSIM results comparing community patterns between habitats and sites, based on main channel data collected in autumn 2000 from the Rosedale sub-reach.

Comparison	Possible Permutations	# Cases ≥ Observed$^+$	*R*-Statistic	*p*-value
*Differences between morphologic habitat types***				
Global			0.129	0.017*
Open Nook, Bar Head	>999	7	0.354	0.008*
Open Nook, Bar Tail	>999	28	0.240	0.029*
Open Nook, Eddy Pool	>999	0	0.653	0.001*
Open Nook, Channel Nook	7	1	0.889	0.14
Channel Nook, Bar Head	34	4	0.482	0.12
Channel Nook, Flat Bar Edge	11	2	0.358	0.18
Eddy Pool, Bar Tail	>999	112	0.089	0.11
Differences between hydraulic habitat types				
Global			0.112	0.007*
Exposed, Normal	>999	0	0.132	0.001*
Exposed, Sheltered	>999	332	0.041	0.33
Normal, Sheltered	>999	141	0.114	0.14
*Differences between sites***				
Global			0.084	0.09*
Hamilton, Harrison	>999	91	0.109	0.09*
Hamilton, Foster	>999	16	0.114	0.02*
Hamilton, Carey	>999	33	0.108	0.03*
Hamilton, Big	>999	42	0.317	0.04*

* significant at the adjusted value of $\alpha = 0.1$.
+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.
** only pairwise contrasts with $p < 0.20$ are reported

The identical analysis applied to juvenile fish produced results similar to those based on all size ranges of fish, but with reduced sensitivity to detect differences among habitats and sites. ANOSIM revealed no significant dissimilarity in community structure between sites ($p = 0.22$) or hydraulic habitats ($p = 0.13$). Community structure differed among morphological habitats ($p = 0.004$) and pairwise contrasts indicated that open nooks were dissimilar from all other habitats ($p < 0.10$) except channel nooks. Two species collectively contributed greater than 60% to open nook dissimilarity with other habitats: northern pikeminnow and prickly sculpin <50 mm. Densities of each were highest in open nooks. Eddy pools showed significant dissimilarity from bar heads ($p = 0.07$), attributed mostly to higher densities of chinook salmon and prickly sculpin <50 mm in eddy pools.

6.3.4.3 Habitat Associations –Year 1999

Exploratory analyses were carried out on 1999 and 2001 data to confirm that results from 2000 were representative of annual patterns. Results from 2001 data matched well with those collected in 2000, and results from 1999 data were similar but with some differences worthy of mention. One-way ANOSIM comparing morphological habitat types revealed significant dissimilarity in all pairwise contrasts between bar heads, flat bar edge, and bar tails, which was not recorded in analyses from other years. Four species collectively contributed greater than 50% of the dissimilarity among these habitats: leopard dace >50 mm, chinook salmon, mountain sucker <100 mm, and redside shiner. Both leopard dace and mountain sucker had highest densities associated with flat bar edge units, redside shiner were substantially more common in bar tail units, and chinook salmon were equally common in bar head and flat bar edge units, but relatively uncommon in bar tails.

6.3.4.4 Differences Among Channel Types

Fish community structure was compared among channel types, and three separate analyses were run based on data collected from each of the sub-reaches. Analyses were restricted to data collected in autumn 2000 to eliminate between-year and seasonal variability that would possibly confound results.

Within the Cheam sub-reach, MDS revealed no distinct groupings by channel type, which was supported by ANOSIM results showing no significant dissimilarity between channels. However, sampling effort across channel types in the Cheam sub-reach was lower than in downstream sub-reaches.

MDS ordination of samples collected from the Rosedale sub-reach revealed loose groupings of samples by channel type, and results from ANOSIM indicated that community structure in the main channel was dissimilar from summer channels ($p = 0.005$, **Appendix F**). Comparisons of community structure between the main channel and side channels, and between side and summer channels were not significant ($p = 0.77$ and $p = 0.22$, respectively). Higher summer channel densities of largescale sucker (<75 mm), peamouth chub, and redside shiner compared to the main channel, and a higher main channel density of chinook salmon, all contributed substantially to the dissimilarity.

Morphological and hydraulic habitat types showed significant dissimilarity across channel types, but the morphological classification better represents differences in community structure because open nook units were dissimilar from all other habitat types, including flat bar edges and bar tails. Species contributing most to the distinctiveness of open nooks were largescale sucker (<75 mm), peamouth chub, and both size classes of leopard dace.

Within the Chilliwack sub-reach, MDS ordination showed relatively distinct groupings of samples classified by channel type (**Figure 6-8a**). The 3-D ordination also was examined (not shown) to assist in interpretation because the stress coefficient exceeded 0.2. Classification of samples by morphological habitat types revealed isolated clustering of open nook units with bays (**Figure 6-8b**), and weak separation of all samples according to hydraulic habitat classes. Compared with previous analyses, eddy pools showed greater dissimilarity with bar heads in the Chilliwack sub-reach, and flat bar edges and bar tails are plotted in closer proximity to bar heads in ordination space.

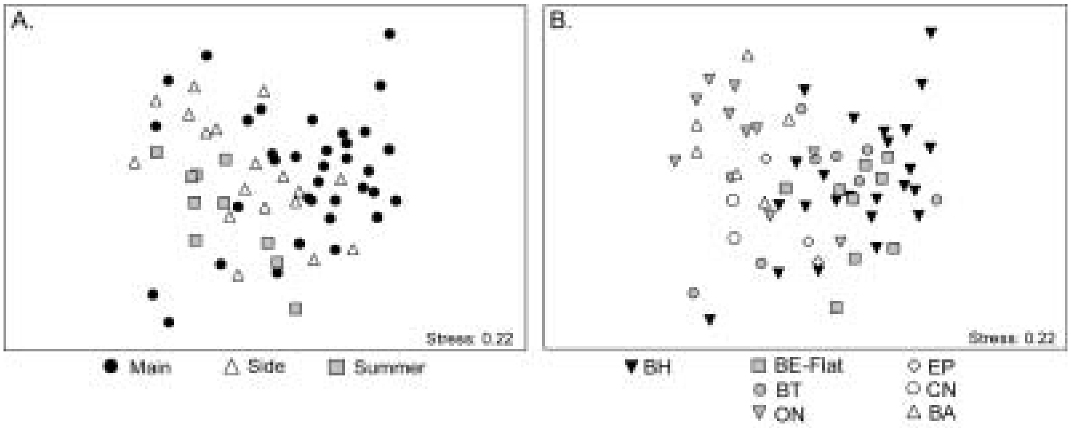


Figure 6-8. MDS ordination of square-root transformed fish densities in samples collected from the Chilliwack sub-reach in autumn 2000. Each plot shows the identical ordination with samples classified by A) channel type, and B) morphological habitats. The shading of morphological habitat units corresponds to hydraulic habitat classes.

Two-way ANOSIM results indicated that dissimilarity in fish community structure was significant between main and side channels (p = 0.04), but not summer channels (**Table 6-8**). The limited number of possible permutations to compare summer channels by two-way ANOSIM contributed to these results because one-way ANOSIM evaluating channel type differences irrespective of habitat groups showed significant dissimilarity amongst all channel types (p = 0.005). Contrasts between morphological habitats showed that both open nooks and bays were dissimilar from bar head and flat bar edge units (**Table 6-8**). All other pairwise combinations of habitat types were similar in community structure. BIO-ENV identified the combination of sand/silt and velocity as maximizing the environmental-species correlation (ρ=0.47)

Table 6-8. Results of 2-way ANOSIM tests comparing community patterns between channel types and habitat, based on data collected from the Chilliwack sub-reach in autumn 2000.

Comparison	Possible Permutations	# Cases ≥ Observed$^+$	*R*-Statistic	*p*-value
Differences between channel types averaged across morphologic habitat types				
Global			0.140	0.008*
Main, Side	>999	92	0.154	0.093*
Main, Summer	360	62	0.247	0.17
Side, Summer	6	1	1.0	0.17
*Differences between morphologic habitat types averaged across channel types***				
Global			0.193	0.002*
Open Nook, Bar Head	>999	54	0.299	0.055*
Open Nook, Flat Bar Edge	>999	0	0.643	<0.001*
Open Nook, Bay	10	2	0.185	0.20
Bay, Bar Head	>999	5	0.630	0.006*
Bay, Flat Bar Edge	40	1	1.0	0.003*
Bay, Bar Tail	660	134	0.153	0.20

* significant at the adjusted value of $\alpha = 0.1$.
\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.
** only pairwise contrasts with $p < 0.20$ are reported

Many species had highest densities in side channels, which contributed to its significant dissimilarity with the main channel. SIMPER identified threespine stickleback as contributing the most of all species to the dissimilarity between channel types (**Table 6-9**), with highest average density in side channels. Largescale sucker <75 mm and peamouth chub showed a similar pattern. For most species with highest densities in side channels, main channel density was higher than in summer channels. The exceptions were threespine stickleback and leopard dace <50 mm, whose summer channel densities were higher than in the main channel. Patterns revealed by SIMPER are supported by trends in species density shown in bubble plots (**Appendix F**).

Table 6-9. Results of SIMPER analysis (square-root transformed data) indicating the average density (untransformed, # 10-m^{-2}) of species that contributed most to the *dissimilarity* in community structure between channel types.

Species	**Average Density**			**% Contributed to Dissimilarity**		
	Main	**Side**	**Summer**	**M/Si**	**M/Su**	**Si/Su**
peamouth chub	0.37	0.80	0.17	**9.7**	**11.2**	**9.3**
longnose dace	0.14	0.35	0.11	**9.1**	7.8	7.1
largescale sucker <75	0.38	2.36	0.25	7.9	**11.6**	**11.2**
threespine stickleback	0.19	8.51	0.40	**21.6**	**12.5**	**21.8**
leopard dace <50	0.03	0.96	0.23	6.8	**9.6**	**9.5**
leopard dace >50	0.09	0.32	0.03	**8.1**	7.2	8.6
redside shiner	0.08	0.24	0.01	6.9	7.6	7.0
Mean Similarity	29.4	32.5	35.3	-	-	-
Mean Dissimilarity	-	-	-	73.6	76.2	71.0

Data are from the Chilliwack sub-reach collected in autumn months.
Bolded text highlights the four species contributing most to dissimilarity *between pairs*.

6.3.4.5 Differences Among Sub-Reaches

Fish community structure was compared among sub-reaches based on data collected in autumn 2000, and separate comparisons were made for each of the three channel types. Within the main channel, MDS ordination showed no grouping of samples by sub-reach ($p = 0.31$), but significant dissimilarity in community structure was detected both among morphological ($p = 0.02$) and hydraulic ($p < 0.001$) habitat types. The morphological classification better characterized fish community structure because ANOSIM results indicated that open nooks were dissimilar from bar tails, but similar to flat bar edge units. Bar heads were dissimilar from habitats of lower velocity, namely open nooks, channel nooks, and bays.

Ordination of side channel data revealed distinct clustering among sub-reaches (**Figure 6-9a**), and significant sub-reach dissimilarity was confirmed by two-way ANOSIM ($p < 0.001$, **Table 6-10**). The fish community of the Cheam sub-reach was most dissimilar to the Chilliwack sub-reach; hence, the degree of dissimilarity matched the longitudinal position of sub-reaches in Fraser River. Similar to the results of main channel data, the morphological classification better characterized fish community structure because ANOSIM results indicated dissimilarity among members of the "normal" hydraulic group, with open nooks dissimilar from both flat bar edge and bar tail units, and also from bar heads and eddy pools (**Table 6-10**). Morphological habitats showed poor grouping in ordination space, with the exception of open nooks that were mostly isolated from other habitat units (**Figure 6-9b**). Flat bar edge and bar tail units were similar in community structure ($p = 0.41$) and each was similar to bar heads ($p > 0.18$), whereas eddy pools were dissimilar from bar heads and bar tails. The BIO-ENV procedure yielded results identical to the analysis of Chilliwack sub-reach data collected in all channels: the combination of sand/silt and velocity maximized the correlation between environmental and species similarity matrices ($\rho = 0.38$).

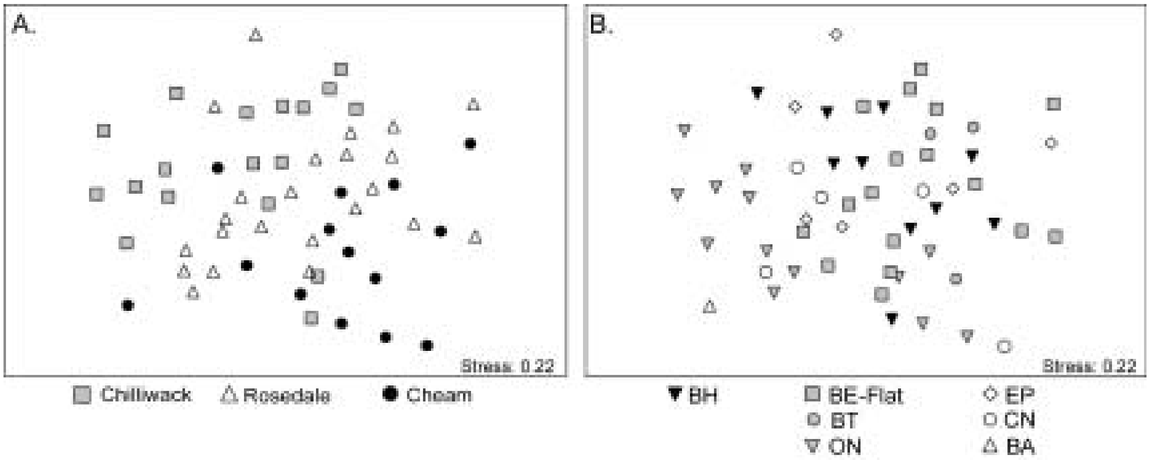


Figure 6-9. MDS ordination of square-root transformed fish densities in samples collected from side channels in autumn 2000. Each plot shows the identical ordination with samples classified by A) sub-reaches, and B) morphological habitats. The legend lists sub-reaches in order from downstream (left) to upstream (right). The shading of morphological habitat units corresponds to hydraulic habitat classes.

SIMPER analysis indicated that higher densities of several species in the Chilliwack sub-reach contributed substantially to between-reach dissimilarity, in particular threespine stickleback that was collected almost exclusively in the Chilliwack sub-reach (**Table 6-11**). Juvenile largescale sucker <75 mm, peamouth chub, redside shiner, and all sizes of leopard dace also were most abundant in side channels of the Chilliwack sub-reach, and several had higher densities in both the Chilliwack and Cheam sub-reaches compared with the Rosedale sub-reach. Of those species contributing substantially to between-reach dissimilarity, only chinook salmon had highest density in side channels of the Rosedale sub-reach. Longnose dace density increased between sub-reaches in an upstream direction and it was the only species contributing substantially to between-reach dissimilarity that was most abundant in the Cheam sub-reach.

Table 6-10. Results of 2-way ANOSIM tests comparing community patterns between sub-reaches and habitat types, based on data collected from side channels in autumn 2000.

Comparison	Possible Permutations	# Cases ≥ Observed[+]	*R*-Statistic	*p*-value
Differences between sub-reaches averaged across morphologic habitat types				
Global			0.405	<0.001*
Chilliwack, Rosedale	>999	0	0.442	<0.001*
Chilliwack, Cheam	>999	3	0.480	0.004*
Rosedale, Cheam	>999	24	0.277	0.025*
*Differences between morphologic habitat types averaged across sub-reaches***				
Global			0.349	<0.001*
Bar Head, Bar Tail	10	2	0.583	0.20
Open Nook, Bar Head	>999	3	0.476	0.004*
Open Nook, Flat Bar Edge	>999	0	0.460	0.001*
Open Nook, Bar Tail	40	3	0.713	0.075*
Open Nook, Eddy Pool	224	5	0.528	0.022*
Open Nook, Channel Nook	100	12	0.255	0.12
Flat Bar Edge, Bar Head	>999	182	0.104	0.18
Flat Bar Edge, Channel Nook	>999	166	0.210	0.17
Flat Bar Edge, Bay	5	1	0.500	0.20
Eddy Pool, Bar Head	56	2	0.590	0.036*
Eddy Pool, Flat Bar Edge	>999	132	0.124	0.13
Eddy Pool, Bar Tail	21	1	0.618	0.048*

* significant at the adjusted value of $\alpha = 0.1$.
+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.
** only significant contrasts are reported

Within summer channels, fish community structure showed significant dissimilarity between the Cheam sub-reach and downstream reaches ($p = 0.04$). Greatest dissimilarity was detected between the Cheam and Chilliwack reaches, and higher average abundances in the Chilliwack sub-reach of threespine stickleback, largescale sucker <75 mm, leopard dace <50 mm, and peamouth chub contributed most to between-reach dissimilarity. Differences between habitat types, averaged across sub-reaches, were detected as well. Based on the morphologic classification, most pairwise contrasts were significant except those of flat bar edge and bar tail units, and those of channel nooks with open nooks, bar tails, eddy pools, and bays.

Table 6-11. Results of SIMPER analysis (square-root transformed data) indicating density (untransformed, # 10-m^{-2}) averaged across all habitat types for species contributing most to the *dissimilarity* in community structure between sub-reaches.

Species	Average Density			% Contributed to Dissimilarity		
	Cheam	**Rosedale**	**Chilliwack**	**Cm/R**	**Cm/Ck**	**R/Ck**
peamouth chub	0.65	0.63	0.80	**14.7**	**9.7**	**10.5**
largescale sucker <75	1.63	0.44	2.36	**13.4**	**10.1**	**9.7**
longnose dace	0.56	0.45	0.35	**11.7**	7.9	**8.9**
leopard dace <50	0.34	0.23	0.96	**11.5**	**8.0**	8.6
chinook salmon	0.10	0.14	0.02	9.9	5.5	6.2
redside shiner	0.14	0.07	0.24	8.0	7.6	6.6
leopard dace >50	0.01	0.09	0.32	7.0	8.4	7.5
threespine stickleback	<0.01	0.01	8.51	<4	**21.3**	**20.3**
Mean Similarity	29.2	32.8	32.5	-	-	-
Mean Dissimilarity	-	-	-	70.5	77.0	72.3

Data are from side channels collected in autumn 2000.
Bolded text highlights the four species contributing most to the dissimilarity *between pairs*.

6.3.5 Summary of Results

Table 6-12 provides a summary of results from multivariate analyses of fish data collected in the gravel reach of Fraser River. Overall, many species showed relatively indiscriminate use of habitats, however, several species had specific habitat associations and contributed substantially to the ecological dissimilarity among habitats. The hydraulic habitat classification incorporated significant within-class dissimilarity and therefore did not accurately represent the spatial distribution of fish species. Members of the “normal” habitat class had significant dissimilarity in community structure, particularly open nooks, and members of the “sheltered” class were dissimilar as well. Eddy pools often shared greatest similarity with bar heads, and channel nooks were consistently dissimilar from most other habitat types.

Habitat types of the morphological classification had higher within-habitat community similarity and between-habitat dissimilarity, and are therefore considered to be most ecologically meaningful. The species assemblage associated with several habitat types was relatively distinct and consistently identified at a variety of temporal and spatial scales. Three habitat types, in particular, were differentiated consistently based on the composition and abundances of fish species: open nooks, channel nooks, and eddy pools. These habitats hosted high densities of many species, including relatively high proportions of juvenile fish that rear in the gravel reach. Open nooks were differentiated based on high densities of largescale sucker <75 mm, leopard dace <50 mm, and longnose dace. Peamouth chub were found almost exclusively in channel nooks, along with threespine stickleback, chinook salmon, and redside shiner. Eddy pools had even higher densities of chinook salmon and redside shiner compared with channel nooks. Flat bar edge and bar tail units had similar community structure, but were differentiated from other habitat types based on the presence of mountain sucker and larger size classes of leopard dace.

These habitat-related patterns in community structure were generally consistent across sub-reaches and among channel types. The influence of these larger-scale factors on overall community structure was detectable based on differences in density and proportional representation of particular fish species.

Table 6-12. Summary of results from multivariate analysis of the fish assemblage associated with habitat types, channel types, and sub-reaches in the gravel reach of Fraser River. All analyses are based on square-root transformed data.

Data Source	MDS stress	ANOSIM	SIMPER	BIO-ENV
ALL YEARS (3) <u>Rosedale sub-reach</u> Main channel Autumn only	*2-D*: 0.23 *3-D*: 0.16	<u>*Two-way tests: Year & Habitat*</u> *Years*: Global-*R*, $p = 0.12$ - 1999, 2000: $p = 0.007$* - 1999, 2001: $p = 0.001$* - 2000, 2001: $p = 0.97$ *Morphologic*: Global-*R*, $p = 0.02$* - Open Nook and Channel Nook dissimilar from all other habitat types - Eddy Pool dissimilar from Flat Bar Edge and Bar Tail *Hydraulic*: Global-*R*, $p < 0.001$* - significant variability within Normal and Sheltered groups	Relatively high densities of many species contributed to dissimilarity of open nooks and channel nooks; in particular, largescale sucker <75 mm, leopard dace, and peamouth chub. High densities of chinook salmon and redside shiner in eddy pools contributed to its dissimilarity with flat bar edges and bar tails. High density of mountain sucker <100 mm in flat bar edge units contributed to its dissimilarity with eddy pools, channel nooks, and open nooks.	large cobble, sand/silt, bank angle, velocity, correlated with species similarity ρ=0.39
YEAR 2000 <u>Rosedale sub-reach</u> Main channel Autumn only	*2-D*: 0.20 *3-D*: 0.15	<u>*One-way test: Habitats*</u> *Morphologic*: Global-*R*, $p = 0.08$* - Open Nook dissimilar from Bar Head, Bar Tail, Eddy Pool, Channel Nook - Channel Nook dissimilar from Bar Head, Flat Bar Edge - Eddy Pool dissimilar from Bar Tail *Hydraulic*: Global-*R*, $p = 0.007$* - significant variability within Normal group ** similar results based on 1999 data	Similar results to analysis of all years combined: relatively high densities of many species, particularly smaller size classes, in open nooks compared to all other habitats. Sockeye salmon found almost exclusively in channel nooks. Relatively high density of chinook salmon in eddy pools and channel nooks compared to bar heads, flat bar edges, and bar tails. Densities of mountain sucker and leopard dace were relatively high in flat bar edge units and contributed to habitat dissimilarity. ** similar patterns based on 1999 data	large cobble, sand/silt, bank angle, velocity, correlated with species similarity ρ=0.43

Data Source	MDS stress	ANOSIM	SIMPER	BIO-ENV
ALL CHANNELS 2000 Chilliwack sub-reach Autumn only	*2-D*: 0.22 *3-D*: 0.15	*Two-way tests: Channels & Habitats* *Channels:* Global-R, $p = 0.008$ • main, side: $p = 0.093$* • main, summer: $p = 0.17$ • side, summer: $p = 0.17$ *Morphologic*: Global-R, $p = 0.25$ • Open Nook dissimilar from Bar Head and Flat Bar Edge • Bay dissimilar from Bar Head and Flat Bar Edge *Hydraulic*: Global-R, $p = 0.07$* • significant variability within Normal group **Similar results based on Rosedale sub-reach ** No trend based on data from Cheam sub-reach	Threespine stickleback, largescale sucker <75 mm, leopard dace, and peamouth chub had highest densities in side channels. Leopard dace and threespine stickleback densities were higher in summer channels than in the main channel. High density of chinook salmon in the main channel differentiated it from secondary channels. Dissimilarity patterns between habitat types similar to results presented above.	sand/silt, velocity, correlated with species similarity ρ=0.47
ALL SUB-REACHES 2000 Side Channels Autumn only	*2-D*: 0.22 *3-D*: 0.15	*Two-way tests: Sub-Reaches & Habitats* *Reaches*: Global-R, $p < 0.001$* • all sub-reaches dissimilar *Morphologic*: Global-R, $p = 0.25$ • Open Nook dissimilar from all other habitat types except Channel Nook • Eddy Pool dissimilar from Bar Head and Bar Tail *Hydraulic*: Global-R, $p = 0.06$* • significant variability within Normal group **Cheam sub-reach dissimilar to downstream reaches based on summer channel data ** No trend based on main channel data	Threespine stickleback, largescale sucker <75 mm, leopard dace, peamouth chub, and redside shiner had highest densities in the Chilliwack sub-reach. Several species had lowest density in the Rosedale sub-reach, the exception being chinook salmon that had highest density in the Rosedale sub-reach. Longnose dace had highest density in the Cheam sub-reach. Dissimilarity patterns between habitat types similar to results presented above.	sand/silt, velocity, correlated with species similarity ρ=0.38

6.4 Discussion

6.4.1 Local-Scale Habitat Associations

Most species in the gravel reach were widely distributed, both at a local scale among habitat types and at the larger reach-scale among channel types and sub-reaches. Measures of species diversity, evenness, and richness also revealed high similarity among habitats and channel types. However, the relative abundances of individual species, when compared among habitats, indicated that, for most species, highest densities were associated with a limited number of habitat types. For some species such as peamouth chub, occurrence was restricted almost exclusively to a single habitat type (i.e., channel nook). For other species such as mountain sucker, densities were highest among multiple habitat types (i.e., flat bar edge and bar tail) that share a high degree of physical similarity. The specialized habitat associations of individual species contributed to the consistent dissimilarity of several habitat types at multiple spatial and temporal scales of examination. Three habitat types, in particular, presented fish with distinct functional opportunities for rearing based on highly dissimilar species assemblages (eddy pool, open nook, channel nook). Open nooks and channel nooks also hosted significantly higher fish densities than other habitats, including highest densities of juvenile fish 0 to 1 year in age, which highlights the ecological importance of these habitats in the gravel reach for fish during this critical life stage.

Fish community structure was correlated with both sedimentary and hydraulic variables and, not surprisingly, the morphological habitat classification showed the highest degree of congruence with the distribution and abundances of fish species. The hydraulic classification, proposed in Chapter 4 as a simplified alternative because it maximizes the physical distinctiveness between habitat groups, did not accurately characterize community patterns. The hydraulic class of "normal" habitats, consisting of open nooks, flat bar edges, and bar tails, pooled significant within-group variability because the fish assemblage associated with open nooks was dissimilar from that in all other alluvial habitats. Flat bar edge and bar tail units had relatively similar assemblages, but the degree of similarity was variable among analyses. The hydraulic grouping of "sheltered" habitats, consisting of eddy pools, channel nooks, and bays, also pooled within-group variability based on the variable dissimilarity of channel nooks and eddy pools.

Eddy pools typically form in association with bar heads in the lee of a riffle and, although offering hydraulically "sheltered" conditions, they are morphologically distinct from channel nooks and bays. Eddy pools also are considerably more coupled with the flow and adjacent bar head unit. The geographical association between eddy pools and bar heads may be advantageous because fish can exploit a range of physical conditions and food sources with minimal relocation. The significantly greater foraging opportunities provided by pools compared with shallow-water habitats has been demonstrated for cutthroat trout in a small, coastal stream in British Columbia (Rosenfeld and Boss 2001). This relation is expected to transcend spatial scales to Fraser River. Eddy pools in Fraser River should favour species that feed in the water column, in particular, because the back eddy flow concentrates suspended particulate material, including insects and zooplankton. This assertion is supported by the habitat-scale distribution of water-column feeding species, as defined by Zaroban *et al*. (1999), in the gravel reach. Chinook and sockeye salmon, cutthroat trout, redside shiner, peamouth chub, and northern pikeminnow are water-column feeders, and all but sockeye and peamouth showed a strong association with eddy pools.

Peamouth chub, sockeye salmon, and threespine stickleback were found almost exclusively in channel nooks, and contributed substantially to channel nook dissimilarity in the Rosedale sub-reach. Each species is common in lakes and slow-water environments in southern British Columbia (Scott and Crossman 1973), and their association with channel nooks is therefore not surprising. The sheltered flow environment of channel nooks also favours highest densities of fry too small for accurate species identification. For such fish, channel nooks may represent valuable "nursery" habitat where energetic costs are minimized and moderate water depths reduce the risk of detection by avian predation, but also restrict larger piscivorous fish that occupy deeper-water bays (Power 1987). However, the typically narrow morphology of channel nooks may represent a trade-off because the restricted boundaries of each unit increase the likelihood for species interactions (competition and predation). As well, channel nooks are prone to warm water temperatures in summer and early autumn, which may exclude some species, particularly salmonids.

The distinctive morphology of channel nooks has not been described previously for wandering, gravel-bed rivers. They are significantly smaller than back-channels, rarely with riparian influence and, unlike bays whose substrate is predominantly fine sediment, channel nooks often have a gravel-sand substrate reflective of the antecedent flow conditions under which they formed. Nevertheless, channel nooks and bays share a high degree of morphological similarity, differing mostly with respect to size, and this physical similarity may translate into ecological similarity in fish community structure. A comparison between channel nooks and bays was hindered by relatively low samples sizes for each habitat within the Rosedale sub-reach. The single analysis providing a reliable contrast was based on data from all channel types in the Chilliwack sub-reach. In this analysis, no significant physical dissimilarity between channel nooks and bays was found. Moreover, species most frequently associated with channel nooks (peamouth chub, threespine stickleback, sockeye salmon) were collected in bays by beach seine and gill net in other sub-reaches and seasons. The ecological similarity of channel nooks and bays remains uncertain, but it is speculated that bays support a wider range of fish sizes, as a consequence of the deeper water.

Open nooks represent the most ecologically distinct of all alluvial habitats, based on the consistency with which ANOSIM contrasts revealed significant dissimilarity with other habitats. Dissimilarity was expressed at the local scale among habitats in the Rosedale sub-reach, and was demonstrated consistently at a larger scale of examination, both extending laterally across channel types and longitudinally among sub-reaches. The scale-independence of open nook dissimilarity substantiates their ecological value as rearing habitat for resident fish. Their ecological dissimilarity, however, is not matched by a high degree of physical distinctiveness (Chapter 4). In fact, units easily could be overlooked in habitat assessment due to the relatively subtle visual differences that distinguish open nooks from flat bar edges. The fact that open nooks have not been described previously for gravel-bed rivers is likely more reflective of their subtle physical character than their uniqueness to Fraser River.

Juvenile fish density was particularly high in open nooks, based on analyses restricted to age 0 to 1 fish, and several species were shown to occupy open nooks as juveniles and move into other habitats as adults. Open nooks therefore also represent exceptional “nursery” habitat in the gravel reach and had high densities of species such as leopard dace and longnose dace that prefer low to moderate velocity conditions (Gee and Northcote 1963). These species are “benthic” species (Zaroban *et al.* 1999) that forage at the substrate-water interface and, for such species, open nooks may be favourable because energetic costs are modest and foraging opportunities are relatively high. The shallow water depth favours periphyton growth, which is an attractive food source both for aquatic invertebrates and benthic fish feeding on invertebrates and plant material. The stomach contents of leopard dace, juvenile largescale sucker, and juvenile northern pikeminnow, all of which were abundant in open nooks, contained a relatively high proportion of algae and plant material (**Appendix F**), which substantiates the functional value of open nooks for feeding. However, the potentially favourable foraging conditions for juvenile and benthic fish species represents a trade-off because the risk of avian predation in shallow water is likely high. Fish may counter-balance this risk in open nooks by moving frequently between deeper, adjacent habitats. This behavioural strategy is likely necessary due to the transient nature of open nooks that are strongly affected by changes in water level.

The ecological dissimilarity of habitats defined partially based on geographic position (bar heads, flat bar edges, bar tails) was less consistently expressed than for other habitats. Bar heads showed the greater dissimilarity of the three habitats, whereas flat bar edges and bar tails were somewhat overlapping in ecological character. The distinctiveness of bar heads may be related to the longitudinal gradient in velocity and sediment texture observed along the length of gravel bars, with bar heads having coarse substrate and typically higher flow exposure. But just as these sedimentary and hydraulic gradients are punctuated by local discontinuities related to bar topography, community structure also varied along this longitudinal gradient and many species were widely distributed among these habitats. Modest habitat partitioning was reflected in the upstream distribution of chinook salmon associated with bar heads, the mid-bar distribution of mountain sucker and leopard dace associated with flat bar edges, and the downstream association of redside shiner with bar tails.

6.4.2 Channel-Scale Differences in Community Structure

The expectation that community structure is highly dissimilar between main and secondary channels was confirmed for the Rosedale and Chilliwack sub-reaches, but not for the Cheam sub-reach, wherein sampling effort was lower. Higher densities of many species in side channels, including threespine stickleback, largescale sucker, leopard dace, and peamouth chub, contributed to the dissimilarity. These results are consistent with other studies in northern temperate rivers that have reported higher densities, enhanced growth, and recruitment within side channels and floodplain habitats (Sommer *et al.* 2001, Grift *et al.* 2003). However, comparisons with other studies are tenuous because side channels examined in most studies are within the floodplain zone and often more closely resemble lentic habitats, with soft-bottomed sediment and negligible velocity.

In the gravel reach, side channels are situated within the active channel zone and are coupled with gravel bar units. In some locations, they mark the transition between the active channel and young islands. However, they differ substantially from truly floodplain channels, the majority of which have been isolated from the main channel for over a century (Ellis *et al.* 2004). Hence, side channels do not constitute truly floodplain habitat and the fluvial processes that shape and maintain these channels more closely match those of the main channel. The range of habitats associated with gravel bars in side channels is similar to the main channel, but gravel bars are scaled in proportion to channel size; hence, habitat units typically are shorter in length and habitat diversity per unit area is greater. Compared to the main channel, side channels have greater connectivity with the riparian zone, which represents a source of drop-in terrestrial insects, shelter from overhanging vegetation, and increased habitat complexity due to large woody debris (Sedell and Froggatt 1984, Johnson and

Jennings 1998). Only at the most downstream end of side channels do habitat conditions exhibit a lentic quality, with a high proportion of fine sediment and low flow velocity (Ellis 2004).

Perhaps the most distinguishing feature of Fraser River side channels from floodplain and summer channels is their hydraulic character. Flow conveyance is reliable throughout much of the growing season for fish, yet flow forces are moderated compared to the main channel, even during flooding. Summer channels, in contrast, are typically more ephemeral in character, and substantially less stable (though exceptions exist, such as the summer channels dissecting Minto Island). The shallow water depth limits the size range of fish in summer channels, and the risk of becoming stranded on the declining limb of the hydrograph is comparably greater. Not surprisingly, autumn sampling found lowest densities of most species in summer channels.

The predictability and duration of flooding have been identified as key elements determining the likelihood for enhanced fish growth and recruitment in non-mainstem habitats of large rivers. This is the basic premise of the Flood Pulse Concept (FPC) proposed by Junk *et al.* (1989), which recognizes the flood pulse as governing biological productivity in tropical, floodplain rivers. The FPC is not directly applicable to northern temperate rivers (refer to Chapter 2), although the annual freshet in rivers such as Fraser represents a significant event. A surrogate to the FPC is needed for northern temperate rivers that recognizes the importance of annual freshet in ecological terms, *as a seasonal reoccupation of the secondary channel network, nearly always within the active channel zone, that prompts lateral migration by fish out of the main channel and presents opportunities for spawning and recruitment*. Results from several studies lend support to the significance of seasonal inundation within the active channel zone of temperate rivers for improved growth and foraging opportunities for fish (Gutreuter *et al.* 1999, Sommer *et al.* 2001, Grift *et al.* 2003); although see King *et al*. (2003).

In Fraser River, the annual freshet represents a predictable and long-lasting event, and one which resident populations are most likely accustomed. Moreover, freshet coincides with the period of spawning and rearing for most resident species (Carlander 1969, Scott and Crossman 1973). The reliable, yet sheltered flow environment of side channels makes it probable they represent flow refugia for fish to escape the hydraulic stress of freshet. Perrin *et al*. (2003a) identified side channels in the gravel reach as the primary spawning habitat for endangered white sturgeon, which spawn between June and August during freshet. Moreover, the relatively slow decline in water levels through summer and early autumn affords fish an extended opportunity to move between the main and secondary channels and optimize foraging opportunities. Such lateral habitat shifts have been reported for other large rivers (Galat *et al.* 1998, Humphries *et al.* 1999, Sommer *et al.* 2001, Hohausova *et al.* 2003, Grift *et al.* 2003). Comparative sampling in the main and secondary channels during high flow is necessary to substantiate these predictions for Fraser River.

6.4.3 Reach-Scale Differences in Community Structure

Differences in fish community structure were revealed at the largest spatial scale of sub-reaches, similar to the analysis of invertebrates in Chapter 5. Thus, congruence between habitat structure and fish community structure occurs at multiple spatial scales in the gravel reach, including the local, lateral, and longitudinal dimensions. Sub-reach differences in community structure were shown both within side channels and summer channels, and these analyses indicate that the Cheam sub-reach is generally dissimilar from downstream sub-reaches based on lower densities of several species and the absence of threespine stickleback. Sub-reach differences, however, were not found within the main channel. Differences in the physical characteristics of main channel habitat types between the Cheam sub-reach and downstream reaches (Chapter 4) therefore do not translate to ecological dissimilarity.

The lack of a longitudinal gradient in fish community structure within the main channel contradicts results from smaller systems. However, such studies examining a longitudinal gradient have contrasted fish assemblages over several orders of stream size (3rd to 5th orders, Peterson and Rabeni 2001b, 1st to 5th orders, Wright and Li 2002), across which substantial changes in channel morphology and habitat characteristics occur. In comparison, the Cheam and Chilliwack sub-reaches of Fraser River were separated by approximately 50 km (~80 main channel widths, **Table 4-1**), which represents a relatively short distance along a river. Major morphological and sedimentary gradients are observed over this distance in the main channel; however, differences between sub-reaches were apparently not sufficient to affect abundances and distributions of fish species at the reach-scale.

Sub-reach differences within side and summer channels reinforce the ecological distinctiveness of secondary channels in the gravel reach, and confirm the value of the hierarchical habitat classification, presented in Chapter 4, for examining fish community structure. Reach-scale differences are related to differences in density of several species, including chinook salmon. Of particular significance is that chinook salmon spawn in upstream tributaries of the Fraser Basin but significant numbers migrate into the gravel reach to rear (**Appendix E**); hence, the gravel reach represents highly desirable habitat.

Previous research documented the spatial distribution of over-wintering chinook salmon throughout the Fraser Basin (Zallen and Boyd 1986, Levings and Lauzier 1991) and several studies have investigated localized habitat use near Hope (Zallen and Boyd 1986, Beniston *et al.* 1986). However, this study was the first to characterize habitat use at a fine spatial scale and results indicate that the gravel reach is used heavily for rearing year-round. Stomach contents of chinook salmon collected during summer months included a range of aquatic and terrestrial insects (**Appendix G**), with nymph and adult chironomids making up the majority of food by volume in summer and winter months, and Ephemeroptera, Plecoptera, Trichoptera, and small fish also being common.

6.4.4 Summary

Physically distinct habitat units occurring around the perimeter of gravel bars present fish with distinct functional opportunities for feeding and rearing. Community structure was distinct within several habitat types, particularly open nooks, channel nooks, and eddy pools, and the ecological importance of these habitats was confirmed over multiple spatial scales. Despite the ecological dissimilarity of physically defined habitat types, most species in the gravel reach have widespread distributions, both at a local scale among habitat types and at the larger scale of channel types and sub-reaches. Most species are found at least occasionally in every habitat type, and the large variability in species habitat associations suggests that most habitat units within the gravel reach offer relatively desirable opportunities for rearing. Moreover, the variable habitat associations of most species are predicted to afford the fish community resilience to physical disturbance resulting in habitat change (Poff and Ward 1990). This prediction will be evaluated in Chapter 8.

The importance of side channels as rearing habitat for a wide range of species was demonstrated, and previous research has identified side channels as the primary spawning habitat for endangered white sturgeon in the gravel reach (Perrin *et al.* 2003a). The study by Perrin *et al.* (2003a) is significant because it is the first documentation of white sturgeon spawning in side channels. Despite significant loss of floodplain channels (“sloughs”) in the gravel reach since the Chilliwack dyke was constructed in 1903, side channel habitat within the Rosedale sub-reach has remained relatively stable (Ellis *et al.* 2004). However, side channels are under increasing threat from urban encroachment and infrastructure development within the floodplain and a loss in side channel habitat would have significant consequences for the ecosystem.

Chapter 7. Physical Habitat Changes Resulting From Disturbance: The Example Of Gravel Mining

7.1 Introduction

Previous chapters characterized the physical organization of the gravel reach of Fraser River in the framework of a hierarchical habitat classification (Chapter 4), and evaluated its ecological basis by determining the distinctiveness of invertebrate (Chapter 5) and fish (Chapter 6) assemblages associated with habitat types. The congruence between habitat types and invertebrate and fish assemblages provides a basis for predicting the ecological response to physical disturbance causing habitat change. In this chapter, short-term habitat changes are documented that resulted from an experimental gravel removal at Harrison Bar, in the Rosedale sub-reach. The gravel removal took place in February 2000. Systematic physical and biological monitoring began six months prior to the extraction and continued over two freshet cycles after mining. The purpose of this chapter is to describe the physical changes that resulted from gravel mining, while the following chapter evaluates the response of the invertebrate and fish community in the context of documented habitat changes.

7.1.1 Background

The tendency for sediment to accumulate as bars and islands in the gravel reach of Fraser River creates outstanding habitat for many fish species and aquatic organisms. But the sediment is important not only to the river's ecosystem; it is highly desirable for construction and industrial purposes. Road and highway construction, septic systems, and concrete are only a few uses for the valued material. Alluvial gravel from rivers is particularly desirable because of its high quality and ease of removal. In many systems, historical extraction volumes have greatly exceeded the natural rate of replenishment. This has happened in part because of a misperception that large volumes of gravel stored in the riverbed correspond with high rates of gravel influx (Kondolf *et al.* 2002). Alluvial gravel mining is now regulated in most industrialized countries, including Canada, although the demand for aggregate continues to grow.

Alluvial gravel mining can occur within the wetted channel by suction dredging, dragline, or clamshell excavation, or in dry sediment by pit mining from floodplain deposits or scalping of gravel bar deposits from within the channel at low flow. Gravel extraction by dry bar scalping is the focus of this chapter. Morphological studies characterizing the physical impacts of gravel mining are reasonably common (e.g., Lagasse 1986, James 1999). Collins and Dunne (1990) summarized observations from several rivers in the western United States with extensive mining activity. Gravel extraction from rivers in California has been reviewed in detail by Kondolf (1994, 1998a, 1998b). The physical impacts of river engineering and gravel extraction on European rivers (Sear and Archer 1998, Surian 1999) and several coastal streams in British Columbia (Sutek and Kellerhals 1989, reviewed by Church et al. 2001) have been described as well.

Far fewer studies have examined the ecological impacts of gravel extraction, despite growing concern that mining damages aquatic habitat. Direct effects of gravel mining arise from the removal of fish, eggs, and invertebrates by in-stream dredging (Perrin and Rempel 2002), mechanical disturbance to fish eggs and larvae within the streambed (Griffith and Andrews 1981), and possibly the removal of invertebrates living in the hyporheic zone (saturated sediment below or adjacent to the river) by dry bar scalping. The former two effects are minimized by current regulations restricting the timing of in-stream removals to prevent overlap with fish spawning and egg incubation. Hyporheic animals that live within interstitial spaces below the channel surface were once thought to be an inconsequential component of river ecosystems, but have been found in high densities up to 10 m below the channel surface and more than 2 km from the channel edge of a gravel-bed river in Montana (Stanford and Ward 1988). No studies have considered the loss of hyporheic animals due to gravel mining and the extent of a hyporheic zone in Fraser River is unknown.

Increased turbidity is an additional direct effect of gravel mining that may impact upon organisms, although the effect is usually short-lived. Turbidity arises either from in-stream excavation that disturbs the streambed and exposes finer subsurface sediment that is readily suspended in the flow (Rivier and Seguier 1985, Perrin and Rempel 2000b), or from dry bar scalping when high flows initially inundate scalped bar tops where the coarse surface sediment has been removed. Increased turbidity reduces light penetration and can thus limit algae and plant growth (Dokulil 1994), as well as smother benthic organisms (Wood and Armitage 1997). Some species of fish and invertebrates also are sensitive to elevated suspended sediment in the water (Berg and Northcote 1985, Wood and Armitage 1997, Birtwell 1999, Shaw and Richardson 2001), which may affect foraging efficiency, territorial behaviours, and gill flaring frequency.

The most substantial impacts of gravel mining on river ecosystems are likely transmitted through modifications to the physical habitat. Gravel mining directly alters channel morphology by the reduction in net sediment supply, though sometimes increasing sediment mobility locally (USACE 1982), and disruption of sediment transport processes (Collins and Dunne 1990, Kondolf 1997). Changes in morphology affect patterns of flow velocity, substrate texture, channel depths, and riparian cover, all of which affect the distribution and abundance of aquatic organisms. Species with particularly specific habitat requirements can disappear from a system where substantial modifications to the habitat have occurred (Ricciardi and Rasmussen 1999). Other species with a greater tolerance to habitat change may persist; however, recurrent alterations to the habitat will inevitably impact species composition and the productivity of river ecosystems (Benke 1990, Richter *et al.* 1997).

In the case of dry bar scalping, indirect ecological impacts also arise through the simplification of bar topography and reduction of shallow-water habitat available at high flow. This is because scalping typically targets areas of high-elevation bar top to maximize removal volumes. In the absence of scalping, bars in many gravel-bed channels like Fraser River build to the point where flow velocity remains low over their tops even in flood. These areas provide low velocity habitat during flooding, and favour sand deposition and the establishment of grasses and shrubs. Vegetation establishment contributes significantly to the stabilization of bar surfaces to form islands. The process of island formation is inevitably slow (several decades in Fraser River, Ham and Church 2002), and the ecological consequences of persistent bar top lowering and disrupted island development are possibly substantial.

Overall, there is evidence to conclude that ecological effects of gravel mining (positive or negative) are transmitted mostly by way of physical alterations to river habitat. However, the extent of physical habitat change necessary to elicit an ecological response remains unknown. As well, the degree and trajectory of ecological response remain uncertain. Most previous studies that have examined ecological effects have been conducted in channels with a prolonged history of gravel mining (e.g., Brown *et al.* 1998). In these studies, no data from true “control” sites, or from mined sites prior to extraction, are available with which to compare conditions at the removal site. As well, many studies have neglected to document the mining history (frequency and volumes removed) and sediment transport regime of the channel, making it difficult to draw conclusions and transfer results to other systems.

Ecological studies of sites with a prolonged history of gravel mining may provide insight into the chronic effects resulting from persistent extraction. However, there remains a significant gap in our knowledge of the immediate and short-term impacts on aquatic habitat and the pattern of response by organisms. To date, no known study has tracked a removal operation at a previously undisturbed site, with time series data of pre-removal and post-removal sampling to examine mining effects.

7.1.2 Objectives

The purpose of this chapter is to characterize the short-term physical changes due to gravel mining at Harrison Bar, Fraser River. Results presented herein are coupled with the following chapter, which examines the short-term ecological response to gravel mining in the context of documented physical changes. The removal operation involved scalping sediment from the surface of Harrison Bar at low flow in February 2000. The site had no prior history of gravel extraction. Physical and biological monitoring activities at the removal site and three chosen reference sites began in September 1999, prior to scalping, and continued through September 2001. Data collection was nested within the sampling framework for research presented in the three previous chapters.

The sampling design and monitoring activities were intended to meet the following objectives: 1) to determine the extent of immediate physical habitat change resulting from bar scalping; 2) to characterize the temporal response of the physical habitat to bar scalping over two freshet cycles; and 3) to characterize the temporal response of the invertebrate and fish community to habitat alterations over two freshet cycles. The former two objectives are satisfied in this chapter, and the latter objective is addressed in Chapter 8.

Three hypotheses were evaluated to meet chapter objectives. First, it was hypothesized that bar scalping caused topographic simplification and a reduction in the total area of high-elevation bar top habitat. Second, that topographic changes and the physical disturbance caused by scalping produced changes in surface sediment texture and the characteristics and availability of habitat types. Finally, that the temporal pattern of physical habitat response at the site was governed by the freshet cycle and dependent upon fluvial processes of sediment transport and deposition.

7.1.3 Local Context

For many decades, Fraser River has been exploited by the local aggregate industry as a source of high quality gravel for the Lower Mainland. At least 4.6 million m^3 of gravel have been mined since 1964 (Weatherly and Church 1999), and there likely have been many unrecorded removals, such as those discovered on historical air photographs (e.g., Queens Bar in 1949, Big Bar in 1974). The average annual removal volume of approximately 120,000 $m^3\ yr^{-1}$ is, by fortunate coincidence, less than the annual estimated volume of sediment recruitment to the gravel reach (see Chapter 2). Chapter 3 summarized the mining history of sites included in the sampling program for this thesis, and **Figure 7-1** illustrates the historical variation in total annual gravel extraction between Hope and Sumas Mountain. The majority of gravel mining from Fraser River has been by dry bar scalping in the main channel between January and March of a given year, including recent above-average removals in 1995, 1996, and 2004. Large removal volumes in the 1970's and early 1980's were achieved by a combination of dry bar scalping at multiple sites, and dragline excavation in Minto Channel (Weatherly and Church 1999). Gravel mining from within the main channel is now being investigated as an effective strategy to reduce the flood risk and, in certain circumstances, counter erosion of the river banks.

Government regulatory agencies recognize the potentially negative impacts to fish habitat and have become more restrictive in allowing gravel removal from Fraser River. However, concerns of bank erosion and flooding continue to increase and mining is viewed by some as an economically profitable strategy to mitigate against these threats. A temporary moratorium on gravel mining from Fraser River was in place for three years (1998-2001) to allow scientific studies to proceed and to draft a long-term gravel management plan. Although a complete moratorium was not achieved, as evident in **Figure 7-1**, extraction volumes during the period were substantially lower than the yearly average. Results presented in the following two chapters contributed directly to the gravel management plan (Fraser Basin Council 2002), and the Department of Fisheries and Oceans has implemented recommendations arising from this research for environmental monitoring of future in-stream removals in southern British Columbia. These recommendations are presented in Chapter 9.

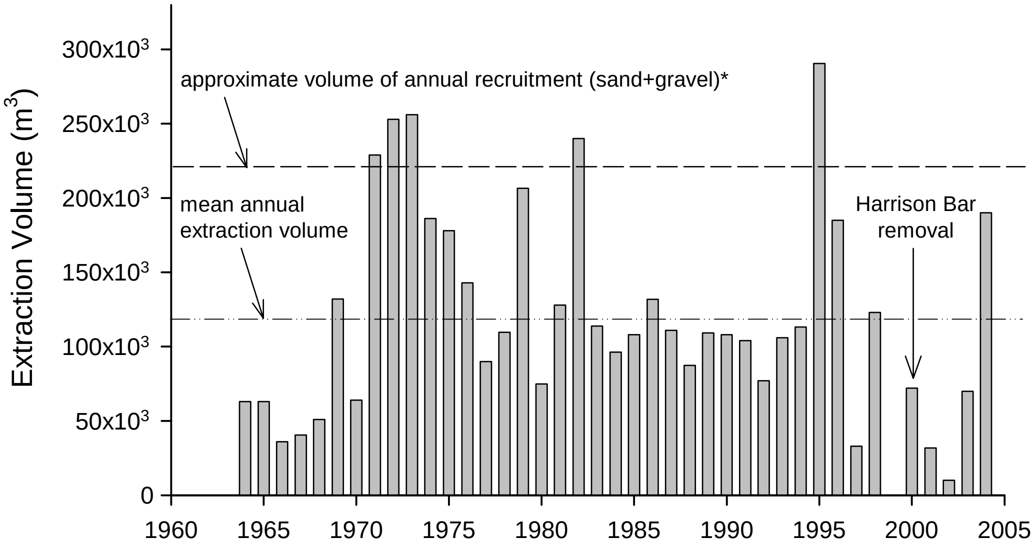


Figure 7-1. Annual volumetric totals of recorded gravel removals in Fraser River between Hope and Sumas Mountain (amended from Weatherly and Church 1999). *from Ham and Church 2003.

Herein, the term gravel extraction is used interchangeably with mining, and is defined as the removal of sediment (sand, gravel, cobble) for the purpose of aggregate recovery for profit, for river management, or both. In order to place the study results in context, three phases of system response to gravel extraction are proposed for Fraser River (**Table 7-1**). This study has examined the short-term impacts of bar scalping only. Immediate impacts occurring on the rising limb of the hydrograph, such as potentially elevated suspended sediment concentrations that may affect juvenile salmonid behaviour and foraging efficiency, were not assessed due to the size of the river and the diffuse behaviour of suspended sediment in Fraser River (Perrin and Rempel 2000b).

Table 7-1. Proposed phases of river system response to gravel extraction in Fraser River.

PHASE	DESCRIPTION
Immediate	*During removal activities and on the rising limb of the hydrograph* as the bar is first inundated. The surface sediment is loose and of small calibre, hence the rate of fine sediment transport is relatively high.
Short-Term	*Extending from first inundation over 2-3 subsequent flood cycles* as the bar sediment redevelops a coarse surface layer and the river has the opportunity to replenish sediment at the removal site. Bar-scale adjustments in topography occur and fluvial process begin to modify the site: bar elevation rebuilding to initiate island formation; sediment recruitment and fluvial processes redistributing material to recreate habitat units; secondary channels developing and incising across the bar; and the spectrum of useable habitat types becoming available over the range of discharges.
Long-Term	*Continuing over a prolonged cycle of freshets* as morphological and ecological adjustment take place. Fluvial processes (described above) continue to modify the site. Adjustments in channel form extending upstream and downstream from the removal site may occur.

7.2 Site Selection and Treatment

7.2.1 Site Selection

Several factors contributed to the selection of Harrison Bar for experimental bar scalping. The main scientific criterion for choosing a site was that it had no prior history of gravel extraction, which might otherwise confound patterns of response. Factors that contributed to final site selection, as well as regulatory approval, were that Harrison Bar is situated within a zone of perceived higher flood risk, it is a site of recent and major sediment deposition, and bedrock protecting neighbouring banks would limit the magnitude of morphological changes that might result.

Harrison Bar is situated on the south side of the main channel opposite the Harrison River confluence (**Figure 7-2**). At this location, the river is set against the north bank before making a 90° turn as flow impinges against Harrison Knob. The backwater effect induced by the addition of Harrison River flow and the sharp bend has contributed to significant gravel deposition and bed level change on Harrison Bar in recent decades (**Table 7-2**). Sediment deposition in the past 15 years has averaged 120,800 m^3 yr^{-1} in the vicinity of Harrison Bar.

Table 7-2. Sediment volume changes and bed level changes in the vicinity of Harrison Bar. Refer to **Figure 7-2** for cell locations.

Cell	1952 – 1984			1984 - 1999		
	Bed Level Change (m)*	Gravel (10^3 m^3)$^+$	Sand+Gravel (10^3 m^3)$^+$	Bed Level Change (m)*	Gravel (10^3 m^3)$^+$	Sand+Gravel (10^3 m^3)$^+$
31	-0.29	-283	-434	+1.63	+1,104	+1,713
32	-0.09	-106	-530	+0.69	+709	+1,082

* from Church *et al.* 2001.
\+ from Ham and Church 2003.

The area of lower Harrison Bar where scalping took place, referred herein as Har-S, is outlined in **Figure 7-2**. The upstream portion of Harrison Bar was used as a reference area (Har-R) against which changes in surface sediment texture at the scalped site were compared. Two additional gravel bars (Carey and Foster) were designated as reference sites for assessing bar scalping impacts on the invertebrate and fish community (Chapter 8). The upstream portion of Harrison Bar (Har-R), Carey Bar (Car-R), and the lower portion of Foster Bar (Fos-R) have no known history of gravel mining and their upstream proximity to the scalping site meant that they were physically similar with respect to channel morphology and gradient, sediment transport regime, and substrate texture. It should be noted, however, that approximately 170,000 m^3 of gravel was removed from the most upstream portion of Foster Bar in February 1995 by dry bar scalping. Three topographic surveys conducted after scalping (between 1995 and 2003) indicate that only 9% of the removal volume has replenished the site; a change in channel alignment at Carey Point, upstream of Foster Bar, is partially responsible for the lack of coarse sediment recruitment. While these factors may have affected lower Foster Bar (Fos-R), its selection as a reference site is defensible based on results in Chapter 5 (**Section 5.3.2.3**) and Chapter 6 (**Section 6.3.4.2**) indicating no difference in invertebrate and fish community structure, respectively, among reference sites.

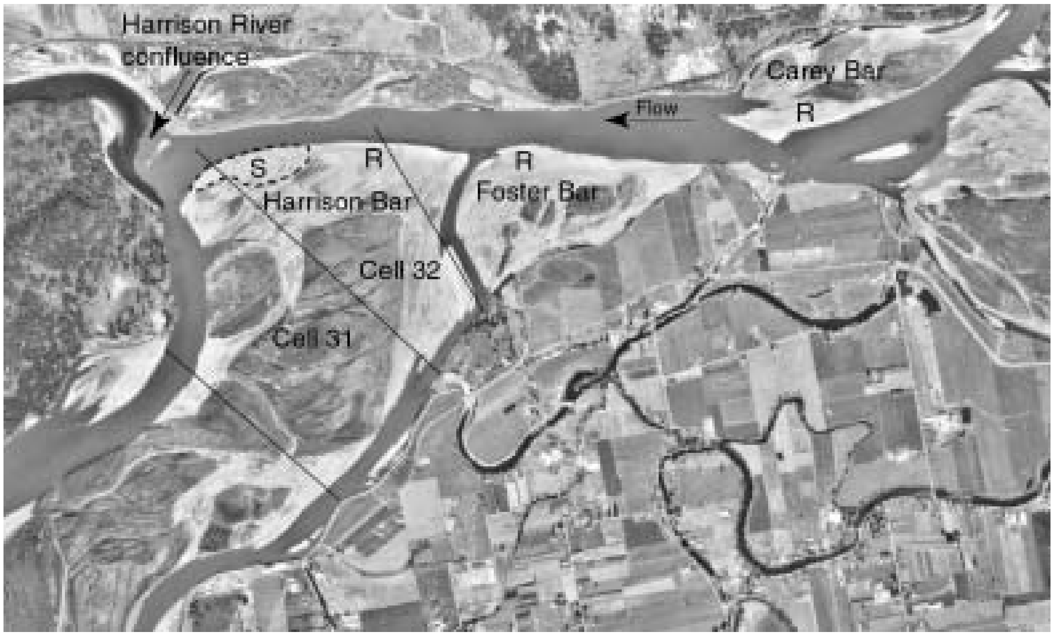


Figure 7-2. Segment of Lower Fraser River where the experimental gravel removal at Harrison Bar took place. The scalped site (S) and reference areas (R) are indicated. Photograph taken March 27, 1999.

7.2.2 Bar Scalping At Harrison Bar

Steelhead Aggregates Ltd. removed approximately 70,000 m^3 of sand and gravel from Harrison Bar between February 26 and March 17, 2000. A target removal volume of 100,000 m^3 was set to approximately equal the mean annual volume of aggregate previously removed from the gravel reach (**Figure 7-1**) and, for experimental purposes, to promote the likelihood of detecting an ecological response. However, insufficient time was available to complete the removal prior to the March 15th deadline imposed by DFO to minimize impacts to migrating salmon fry. Discharge at Hope remained stable during the operation and averaged 707 $m^3 s^{-1}$.

The removal site consisted of two areas, referred to as “A” (~200 x 400 m) and “B” (~100 x 200 m), separated by a narrow gravel berm (**Figure 7-3**). A 1-m buffer strip was left along the water edge in Area A whereas the downstream corner of Area B was scalped directly to water to ensure positive drainage. Haul trucks and a front-end loader were barged to Minto Island at the start of the operation, and vehicle access to the removal site was via a graded haul road from the stockpile area (**Figure 7-3**). Each truck had a haul capacity of approximately 10 m^3 and transported gravel from the removal site to the stockpile area. A large conveyor spanning Minto Channel was then used to transfer gravel from the stockpile to the Steelhead yard. The scalping operation followed a 24-hr schedule in order to maximize the removal volume by the March 15th deadline. After completion, the site was graded at a 2% slope towards the main channel to ensure positive drainage and that no depressions or low areas could strand fish. The haul road was left hard-packed due to the haste in completing the operation, rather than being scarified and loosened (standard procedure). Rempel and Church (2003) include further details and photographs of the operation. Aside from the haul road intersecting the inner corner of the reference area, Har-R remained unaltered by removal activities.

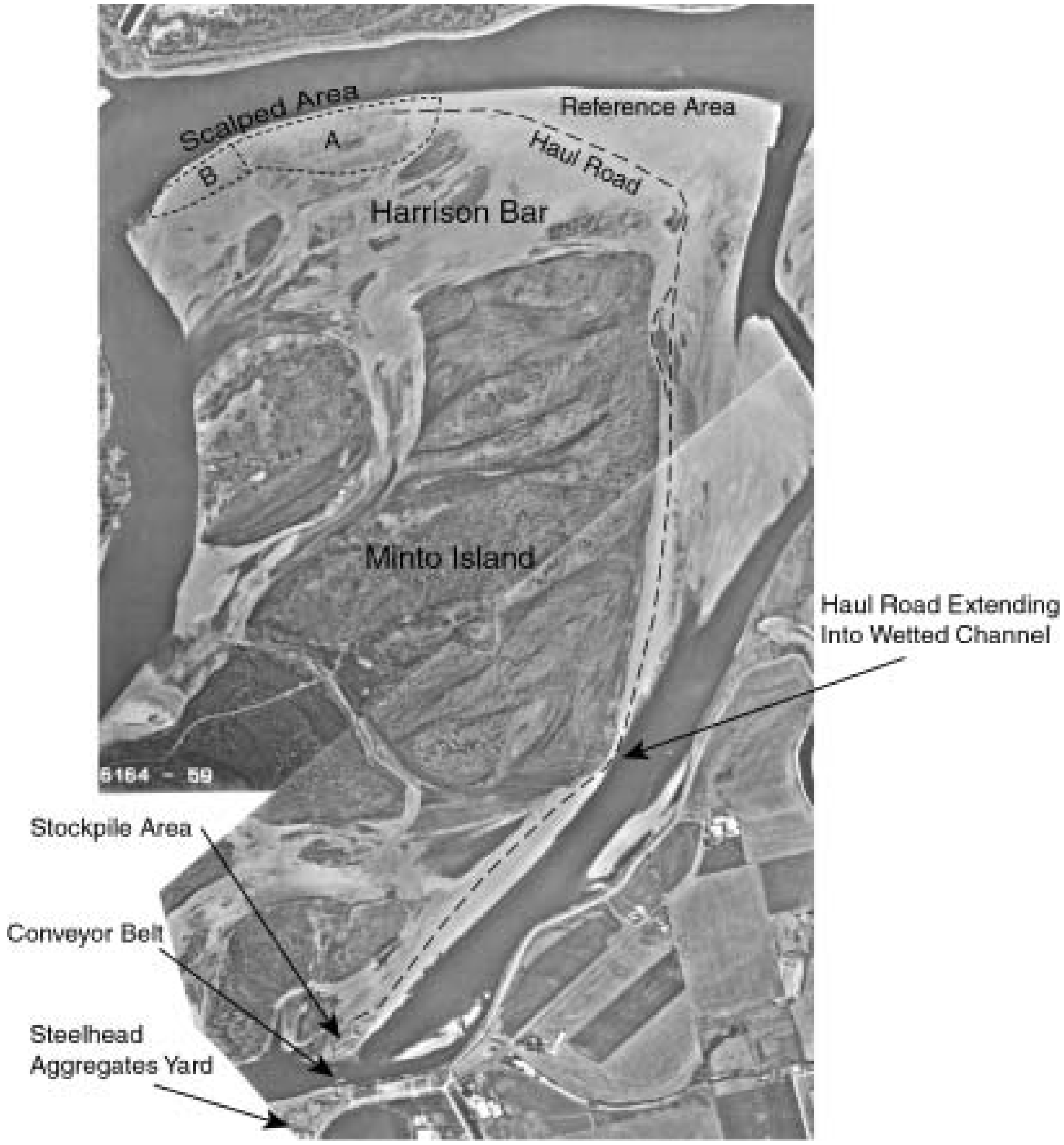


Figure 7-3. Minto Island showing the scalped area on Harrison Bar, haul access road, stockpile area, conveyor belt, and Steelhead Aggregates yard. Photograph taken March 10, 2000**.**

7.3 Methods

Only field methods specific to the assessment of gravel mining impacts at Harrison Bar, and therefore not described in Chapter 3, are presented below.

7.3.1 Sedimentology

7.3.1.1 Before Scalping

Major sedimentary units (areas of relatively uniform sediment texture) were delineated by ground observation in the removal and upstream reference areas on February 8, 2000. Surface sediment sampling was stratified within the units, following recommendations of Wolcott and Church (1991), to reduce the effect of spatial variability on estimates of grain size. Two such units were identified in each of the removal and reference areas from which samples were collected, and 2 additional samples were taken near the water edge in each area (**Figure 7-4**). Sampling followed the Wolman method (Wolman 1954), which involved measuring the B-axis of 400 stones picked at a fixed spacing along a line-transect. Stone spacing always exceeded the size of the largest stone observed within the sedimentary unit and was standardized at 75 cm for most Wolman samples. This spacing ensured that the size of successively sampled stones was independent. Stones were measured using templates of standard grain size categories (Wentworth classification, Church *et al.* 1987) to a minimum size of approximately 2 mm, from which cumulative frequency curves were constructed. Wolman samples also assessed the overall proportion of the surface area covered with sand. The frequency curve of the gravel-sized fraction was used to determine median grain size (D_{50}) and two distribution percentiles to indicate the size of the coarse (D_{95}) and fine (D_5) material present. **Table 7-3** summarizes sediment sampling effort over the duration of monitoring activities.

Table 7-3. Number of surface sediment samples collected at Harrison Bar.

Site	February 2000	April 2000	September 2000	September 2001
Mined Area	4 (W)	3 (W)	5 (W)	12 (P)
Reference Area	4 (W)	0	5 (W)	9 (P)

W: Wolman method, P: photo method.

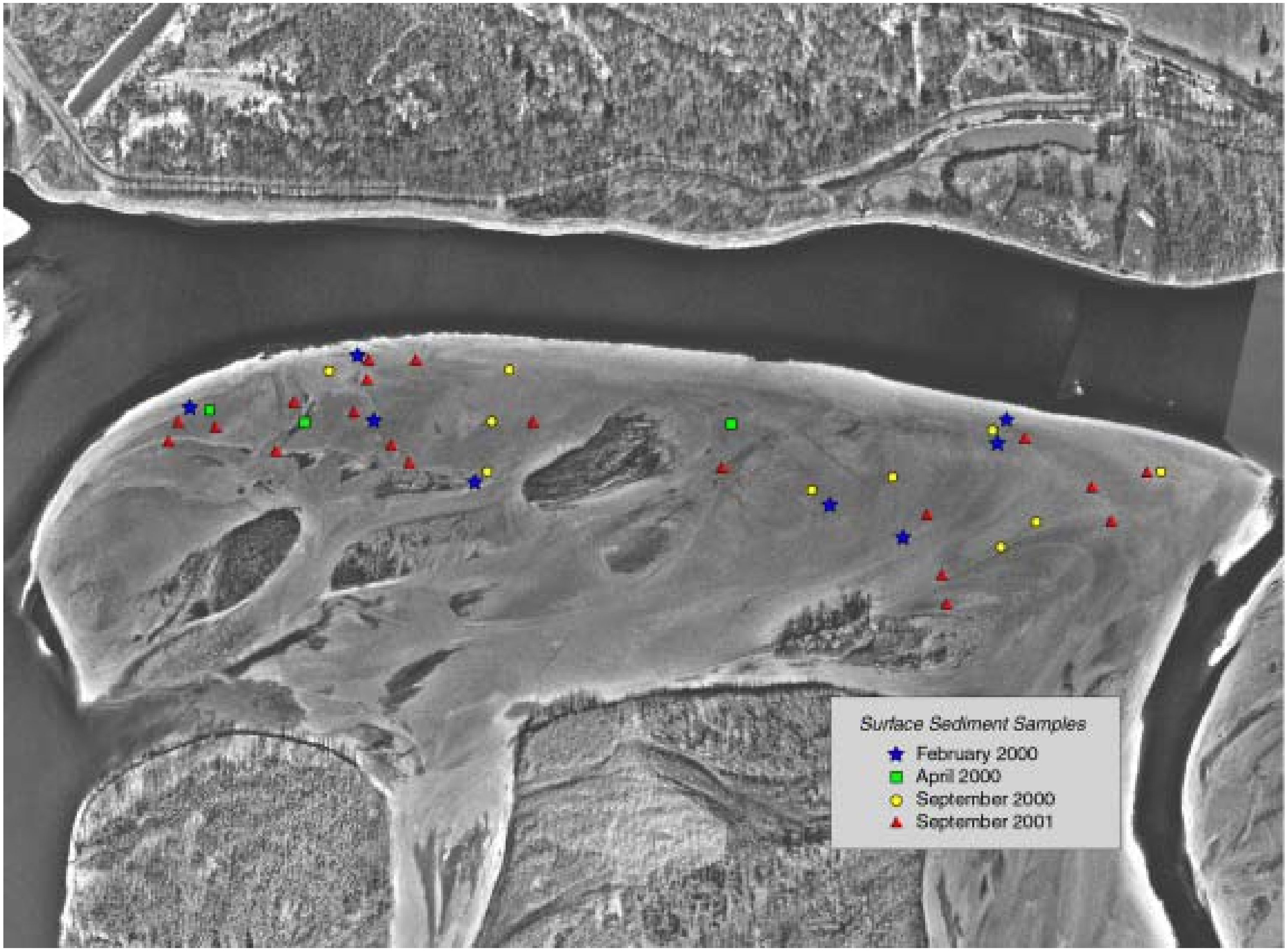


Figure 7-4. Surface sediment sampling locations at Harrison Bar. Photograph taken March 7, 2001.

7.3.1.2 After Scalping

Three Wolman samples were collected immediately after scalping in late March 2000 from the removal area (**Table 7-3**), including one along the road surface. The small number of samples was adequate because scalping had left a loose and uniformly graded matrix of surface and sub-surface material. The reference area was not sampled because no change to the site had occurred. After freshet in September 2000, five Wolman samples were collected from the removal area within identifiable sedimentary units, including one from the road surface. Five samples also were collected in a similar manner from the reference area (**Figure 7-4**).

Surface sediment sampling in September 2001 followed a photographic method, which has the advantage of allowing many samples to be obtained rapidly over large areas in the field, but the disadvantage of requiring laborious office time counting stones from photographs. The photographic method was calibrated for Fraser River by Dr. S. P. Rice (Geography, Loughborough University) and described in Church *et al*. (2000). Briefly, the technique is based on the inverse relation between the size of the stones that occur on a surface and the number of those stones present per unit area. The initial calibration data set consisted of 83 Wolman samples paired with vertical photographs of a 0.5 x 0.5 m quadrat laid down within the Wolman sampling grid. The number of visible stones in each photograph was counted and the plot of grain-size data (Wolman samples) against stone counts (photographs) yielded consistent relations between stone count (number m^{-2}) and the D_5, D_{50}, and D_{95}. Linear regression on log-transformed variates yielded calibration functions that are applicable to counts obtained from other sites in the gravel reach. For this study, 12 photographs were taken in the removal area and 9 photographs in the reference area.

7.3.2 Bar Topography

Tunbridge & Tunbridge Ltd. was retained to survey lower Harrison Bar on February 3, 2000, prior to scalping. The removal area was re-surveyed on March 20, 2000, immediately after scalping, to determine the volume of gravel removed. Tunbridge & Tunbridge repeated the survey over the lower bar on February 19, 2001 and October 5, 2001, and then again on March 28, 2003. The most recent survey was prompted by above-average flood discharge in 2002 and these data have been incorporated into the thesis, although other monitoring activities were completed in September 2001. The three post-removal surveys were intended to determine topographical changes and quantify sediment recruitment to the scalped area following subsequent freshet events. It was only during freshet that sediment transport occurred and any gravel replenishment to the removal site could take place (McLean *et al.* 1999).

7.3.3 Habitat Mapping

Habitat mapping was applied to Harrison Bar before and after gravel mining to examine differences in habitat availability. Mapping followed Level Three of the morphological habitat classification, described in Chapter 4, and used a combination of photo interpretation and ground surveys. The hydraulic habitat classification, introduced in Chapter 4, was not favoured for this analysis because of the loss of morphological information about habitat units. Three water levels were examined: low flow in winter and two levels of moderate discharge in summer. In all photographs, Harrison Bar was split into two halves of approximately equal perimeter length that corresponded with the upstream reference area and the downstream scalped area. No photographs depicting high discharge were available for comparison. However, topographic survey data were used to speculate on the change in habitat availability at high flow before and after scalping.

Low flow conditions were assessed using aerial photographs flown in March in 1999, 2000, and 2001 when discharge was approximately 700 $m^3 s^{-1}$. Habitat units were identified, counted, and the perimeter length measured based on geo-referenced photographs and follow-up ground surveys.

Oblique photographs depicting moderate summer discharge in 1995, 1999, 2000, and 2001 were taken from a fixed-wing aircraft flown approximately 1000 m above the ground. River discharge at the time of photography was similar in 1995 and 2000 (2680 $m^3 s^{-1}$ and 2844 $m^3 s^{-1}$, respectively), and in 1999 and 2001 (1950 $m^3 s^{-1}$ and 1580 $m^3 s^{-1}$). Each photo-pair represented comparable water levels before and after scalping and were used to classify habitat units. Mapping from oblique photographs was based only on counts of units around the bar perimeter because distortion in the photographs prevented perimeter lengths from being measured accurately. For all years except 1995, photo interpretation was followed by ground surveys to confirm the location and identity of habitat units. For 1995 photographs, habitat typing was carried out using photographs only. Although habitat units are most accurately identified by ground surveys, Rempel and Church (2002) demonstrated that units can be identified accurately from aerial and oblique photographs.

7.4 Data Analysis

7.4.1 Sedimentology

The grain size distribution of surface sediment was compared before and after scalping based on samples truncated to retain sizes >4 mm. This coarse sediment fraction was important to consider because it determines the ultimate stability of the bed at a particular place. The proportion of sand covering the surface was estimated as well, and was of interest from an ecological perspective because fine sediment affects primary production, modifies the architecture of macroinvertebrate habitat and, in part, determines the spawning quality of the bed for resident and anadromous fish.

The grain size distributions of Wolman samples collected in February 2000 were compared to samples from September 2000 (after scalping) by the Kolmogorov-Smirnov (K-S) goodness of fit test (Zar 1984). Samples from September 2001 were not included because they were collected by the photographic method and lacked size distribution information. The test is suited to continuous data grouped by size class and determines if a particular sample distribution differs from an expected distribution. In this case, the expected distribution for post-scalp samples was a match with pre-scalp data. Exploratory analysis found the critical value of the K-S test to be impractically sensitive for Wolman samples: two distributions with >5% difference between any grain size class were statistically unique. For a 400-stone Wolman sample grouped according to 11 size classes, this translated into a critical difference of <20 stones for any given class. Based on this criterion, even replicate samples collected simultaneously within a sedimentary unit were statistically different. An alternate critical value was chosen based on the maximum difference found between replicate Wolman samples collected within a homogeneous sedimentary unit. This value of 48 stones was chosen as the *adjusted* critical value for K-S comparisons of grain size distributions between Wolman samples in February and September 2000.

Single-factor analysis of variance (ANOVA) was used as a complement to the K-S test to compare several summary metrics derived from surface sediment samples. Comparisons were made among samples collected on one date before (February 2000) and two dates after scalping (September 2000, 2001). Separate analyses were run for each of two sedimentary units: the water edge and inner bar. Four parameters were examined after meeting assumptions of normality (Shapiro-Wilks test) and homogeneity of variance (Levene's test): arcsine-transformed proportion of sand, log-D_5, D_{50}, and D_{95}. The critical value was adjusted by the Bonferroni method because multiple significance tests were performed on the same data ($p = 0.0125$).

It should be noted that the exact locations sampled before freshet were not revisited in April 2000, September 2000 and September 2001. Rather, sediment sample sites in all months were chosen to correspond with distinguishable sedimentary units following methods of Wolcott and Church (1991). Using this strategy, sampling on all dates was adequate to characterize the surface sediment texture across Harrison Bar and sample sites were sufficiently close together for valid comparisons to be made between months.

7.4.2 Bar Topography

Tunbridge & Tunbridge (Ltd) provided surface elevation data from each of their five surveys. With these data, volumetric changes were estimated at Harrison Bar by producing topographic surface grids from each of the surveys using the TOPOGRID command in Arc/Info (5-m grid spacing). The TOPOGRID command was used previously to produce a realistic bed surface model for the entire gravel reach of Fraser River (Church *et al.* 2001). The TOPOGRID command is applied to an area designated by a *boundary* within which there must be a sufficient density of elevation measurements to produce a smooth surface. Hence, it was important that elevation data for each survey were well distributed within the boundary to minimize interpolation error. After each survey was converted to topographic surface grids, the CUTFILL command in Arc/Info was used to determine volumetric differences between two surfaces.

A complicating factor was that some surveys did not overlap the boundaries of the removal area along the waterline (**Figure 7-5**). These discrepancies were due to differences in river discharge between surveys and changes to the bar perimeter over time. Whereas the survey in March 2000 corresponded approximately with the removal boundary, surveys in October 2001 and March 2003 excluded a sizeable area around the lower corner. To overcome this problem, three boundaries were established within which volumetric comparisons between surveys were made (**Figure 7-6**). *Area A* and *Area B* boundaries approximately corresponded with the original removal boundary. The *Lower Bar* boundary represented the largest common area of all surveys. Only survey data from February 2000, March 2000, and February 2001 could be compared within *Area A* and *B* boundaries, whereas all survey dates could be compared within the *Lower Bar* boundary.

Finally, topographic changes at Harrison Bar were examined by calculating the relative proportion of bar surface area that was at or above a given elevation. These calculations were based on the grid surfaces. Hypsometric curves, which describe the relation between bar surface area and elevation, were produced from each surface grid to facilitate comparisons between surveys.

The topographic surface grid of Harrison Bar, before and after scalping, was related to river discharge in order to determine the minimum discharge at which the entire bar surface was completely submerged. This question addressed the hypothesis that lowering of bar surface elevation by scalping resulted in a loss of near-shore and bar top habitat during high flows. The relation between water surface elevation and discharge was established based on gauge readings from the Canadian Pacific Railway Bridge at Harrison Mills (corrected to geodetic datum) and discharge data at Hope.

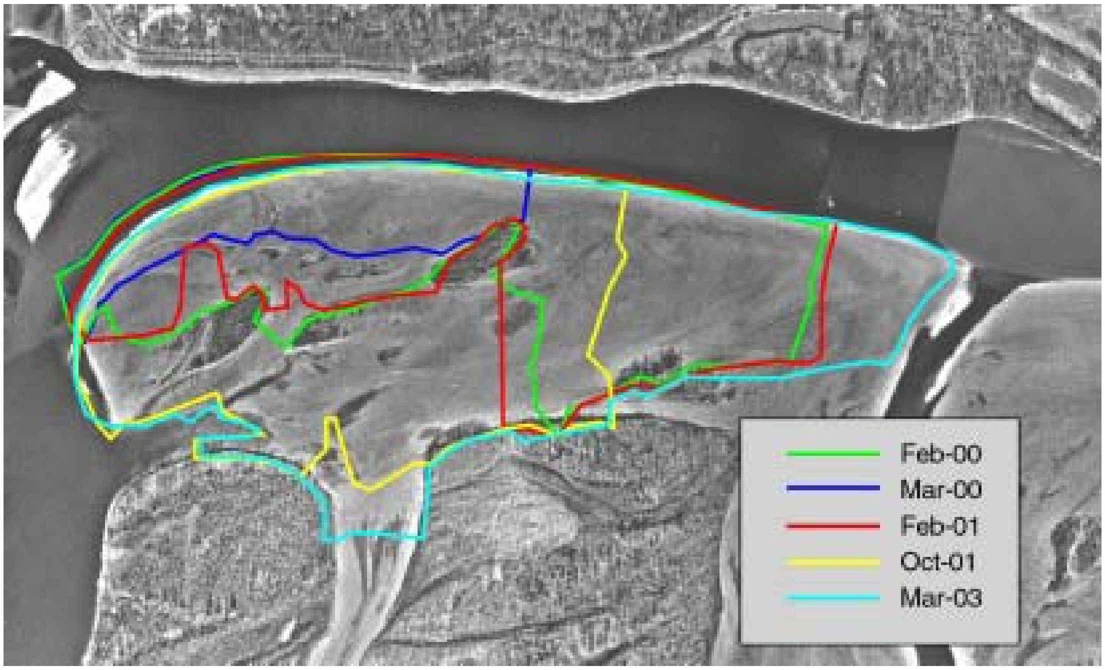


Figure 7-5. The approximate extent of surveys conducted by Tunbridge & Tunbridge at Harrison Bar on five dates before and after scalping. The survey in March 2000 corresponded approximately with the removal boundary. Photograph taken March 7, 2001.

Figure 7-6. The three boundaries within which calculations of volumetric change between survey dates were made. Photograph taken March 7, 2001.

7.4.3 Habitat Availability and Physical Characteristics

Habitat-specific analyses followed two lines of examination. First, habitat mapping was used to determine *if the availability of habitat types changed* at Harrison Bar after scalping. For stereo photographs depicting low flow conditions in March, the number and length of each habitat unit were determined based on a combination of air photograph interpretation and ground truthing. Habitat value was estimated as the total *count* of each habitat type multiplied by the mean density of juvenile fish captured in the habitat by beach seine over 3 years of sampling (1999 to 2001). An alternate measure of habitat value was calculated as the total *length* of each habitat type multiplied by mean density. Only fish data collected at low flow (November through April) were used in density estimates for March photographs. The same procedure was carried out for reference sites in order to provide a statistical reference of the natural change in habitat value at low flow between years.

Perspective distortion in oblique photographs taken in summer months prevented accurate length measurements of habitat units. Hence, only the count-based estimate of habitat value was calculated for each habitat type (number of units * mean fish density). Fish data collected during summer months (July through August) between 1999 and 2001 were used to estimate fish density for oblique photographs. The photographic record for reference sites was not adequate to make comparisons in order to provide a statistical reference of natural change in habitat value at moderate discharge. However, mapping at nearby Big Bar was carried out over a range of flows and in various years, and these data are considered as a surrogate reference for natural variance.

The second line of examination served to determine *if the physical characteristics of habitat units changed* at Har-S after scalping (difference between sampling periods) and if conditions were comparable with reference sites (difference between sites). The characteristics of flat bar edge habitat were compared between the scalped area and reference sites based on observations from beach seine sampling. Only flat bar edge habitat was sampled prior to scalping at Har-S and at all reference sites. It is worth noting that, prior to scalping, Harrison Bar had relatively monotonous topography and consisting almost entirely of flat bar edge habitat. Flat bar edge also was the dominant habitat per unit bar length in the gravel reach.

The multivariate nature of habitat attributes suited examination by principal components analysis (PCA) to summarize total variation in the physical data set and reduce the number of variables to a subset of linear, orthogonal axes representing the dominant physical gradients. These PC-axes then were used to examine habitat differences over time and between sites by considering the relation among all physical factors simultaneously. PCA has the advantage of considering all physical variables together, but the disadvantage of yielding multivariate PC-axes that are not as readily interpretable. PCA was applied to flat bar edge habitat data using a correlation matrix. Variables were first tested for normality (Shapiro-Wilks test) and homogeneity of variance (Levene's test), and the appropriate transformation applied. The following variables were included: bank angle (log-transformed), mean depth, maximum depth, mean velocity, maximum velocity, and the proportions of cobble, gravel and sand (each arcsine-transformed). PC-axes with an eigenvalue >1 were considered to represent dominant physical gradients.

Asymmetrical analysis of variance (Underwood 1991, Underwood 1992, Underwood 1993) was applied to each of the first three PC-axes (eigenvalue >1) to determine if the physical characteristics of flat bar edge units changed as a result of scalping at Harrison Bar. Separate analyses of each PC-axis were valid because the axes were orthogonal, and therefore independent of each other. The framework and mechanics of asymmetrical ANOVA are described in detail in Chapter 8, **Section 8.4.1**, in which the analysis is used almost exclusively to assess biological impacts due to scalping. The breakdown of the analysis is provided in **Appendix I**.

7.5 Results

7.5.1 Sedimentology

7.5.1.1 Pre-Scalping

Surface grain size prior to scalping exhibited a fining trend from the water edge toward the inside bar top, consistent with less powerful and less frequent current washing on the bar top. The moderately sloped beach face was coarser and had a low proportion of sand (< 3%). Absolute sediment size also declined downstream, chiefly in the D_{50} along the water edge. The median grain size on the inner bar top was similar between Har-S and the reference area, Har-R. Average sand coverage was relatively high on both the scalping (11%) and reference (17%) surfaces, but was locally variable, reflecting the transient nature of sand deposition and persistence in the gravel reach.

7.5.1.2 Post-Scalping

After the removal and prior to the 2000 freshet, the scalped surface had a higher proportion of sand (average cover 32%) and the grain size distribution was notably finer. Average D_{95} dropped from 92 mm to 39 mm at the water edge, and average D_{50} dropped from 29 mm to 13 mm (**Figure 7-7**). This fining reflects disruption of the coarse surface layer to expose finer materials beneath.

After freshet in September 2000, sand cover in both the reference and scalped areas was substantially reduced (**Figure 7-7**). The reduction in sand was observed both along the water edge and the inner bar top. Median gravel size in the reference area was relatively consistent with previous sampling dates whereas the scalped area coarsened, particularly along the water edge. Surface sediment texture within Har-S also was comparable to reference sites in September 2000 (Carey Bar: D_{50} = 26 mm, D_{95} = 49 mm). However, the coarsest fractions appear not to have been replenished along the water edge of Har-S after one freshet (**Figure 7-7**), where D_{95} after freshet (64 mm) was less than before scalping (92 mm) along the water edge.

The freshet of 2001 produced only modest changes to surface sediment texture at Harrison Bar. The percent sand cover increased in the upper removal area whereas the reference areas showed little change. The gravel grain size distributions in both areas were highly similar as well. Median grain size at Har-R was relatively unchanged and Har-S showed some coarsening, with average values along the water edge increasing from 26 mm to 35 mm, and within the inner bar increasing from 18 mm to 23 mm (**Figure 7-7**).

Comparing grain size data collected in February 2000 prior to scalping (**Figure 7-7**), the median grain size in September 2001 was slightly higher at Har-S both at the water edge and inner bar. The coarsest size fraction (D_{95}) was less than the average pre-scalping value at the water edge (82 mm versus 92 mm), but an even larger reduction over this period was observed at Har-R (100 mm versus 60 mm). A similar pattern of change in the reference and scalped areas suggests that sediment recruitment through two below-average freshets may be responsible for the outcome. Sequential photographs of the removal site before and after bar scalping reflect the overall change in surface sediment texture at the scalped site post-freshet (**Figure 7-8**).

Sediment texture along the access road did not recover to pre-scalping conditions after two freshet events. It was left hard-packed after bar scalping and obvious crushing of the coarse sediment had occurred. The surface had low sand content (<1%) and grain sizes were fine (D_{50} = 6 mm, D_{95} = 25 mm). Sediment texture was consistent along the entire road surface post-scalping. The freshets of 2000 and 2001 deposited sediment in isolated patches over the road and in these areas the hardened surface appeared to have broken up. But other areas remained compact and were blanketed with sand. These areas could be discerned from air photographs taken in March 2001 and, to a lesser degree, from oblique photographs taken in August 2001.

The Kolmogorov-Smirnov (K-S) test comparing grain size distributions of Wolman samples collected in February (before scalping) and September 2000 (after scalping and flooding) indicated that the maximum difference of samples along the water edge was 110 stones, and differences from two sets of samples collected from the inner bar had differences of 84 and 68 stones (critical value was 48 stones). Thus, the modified K-S test found a significant difference in grain size distributions before and after one freshet post-scalping along the water edge and inner bar.

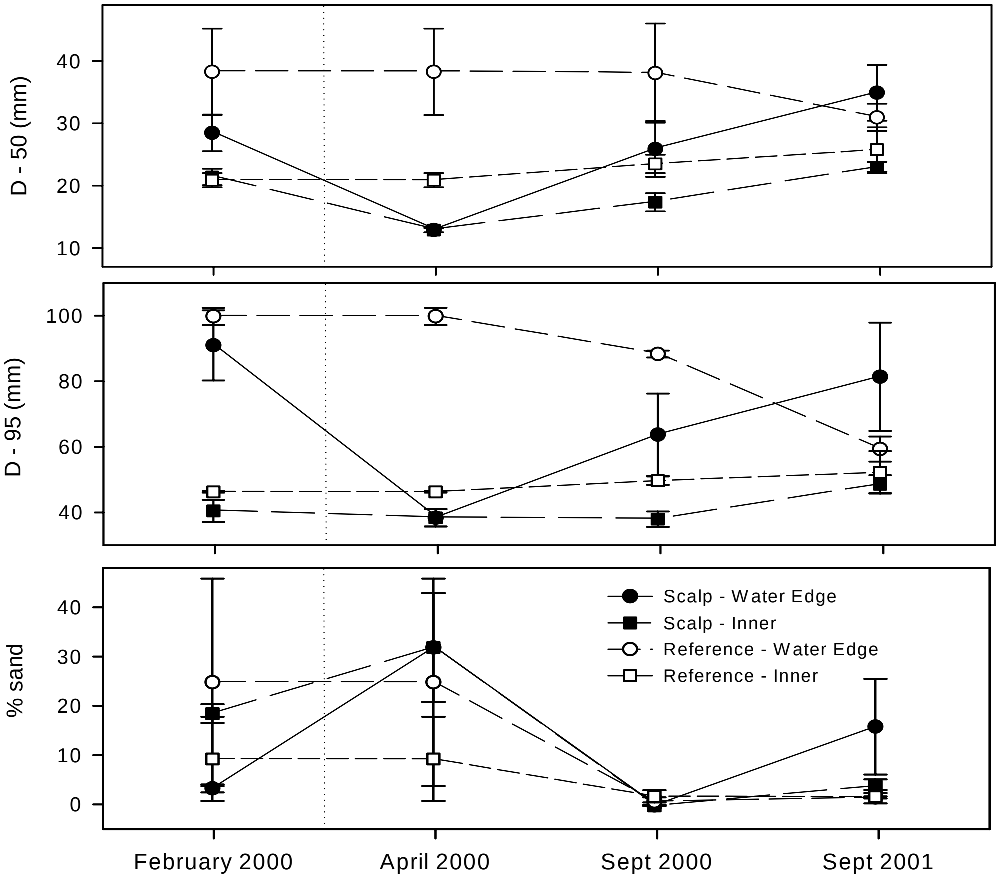


Figure 7-7. Surface sediment characteristics (mean ± SE) in the scalped and reference areas of Harrison Bar before and after bar scalping. The vertical dotted line indicates the timing of scalping.

Results of one-factor ANOVA examining differences in summary metrics between sampling dates were more conservative than the K-S test (**Table 7-4**). This analysis included Wolman samples and surface samples collected in September 2001 by the photographic method. Only the proportion of sand in samples collected from the inner bar differed after scalping, with higher sand content before scalping than on both dates after scalping.

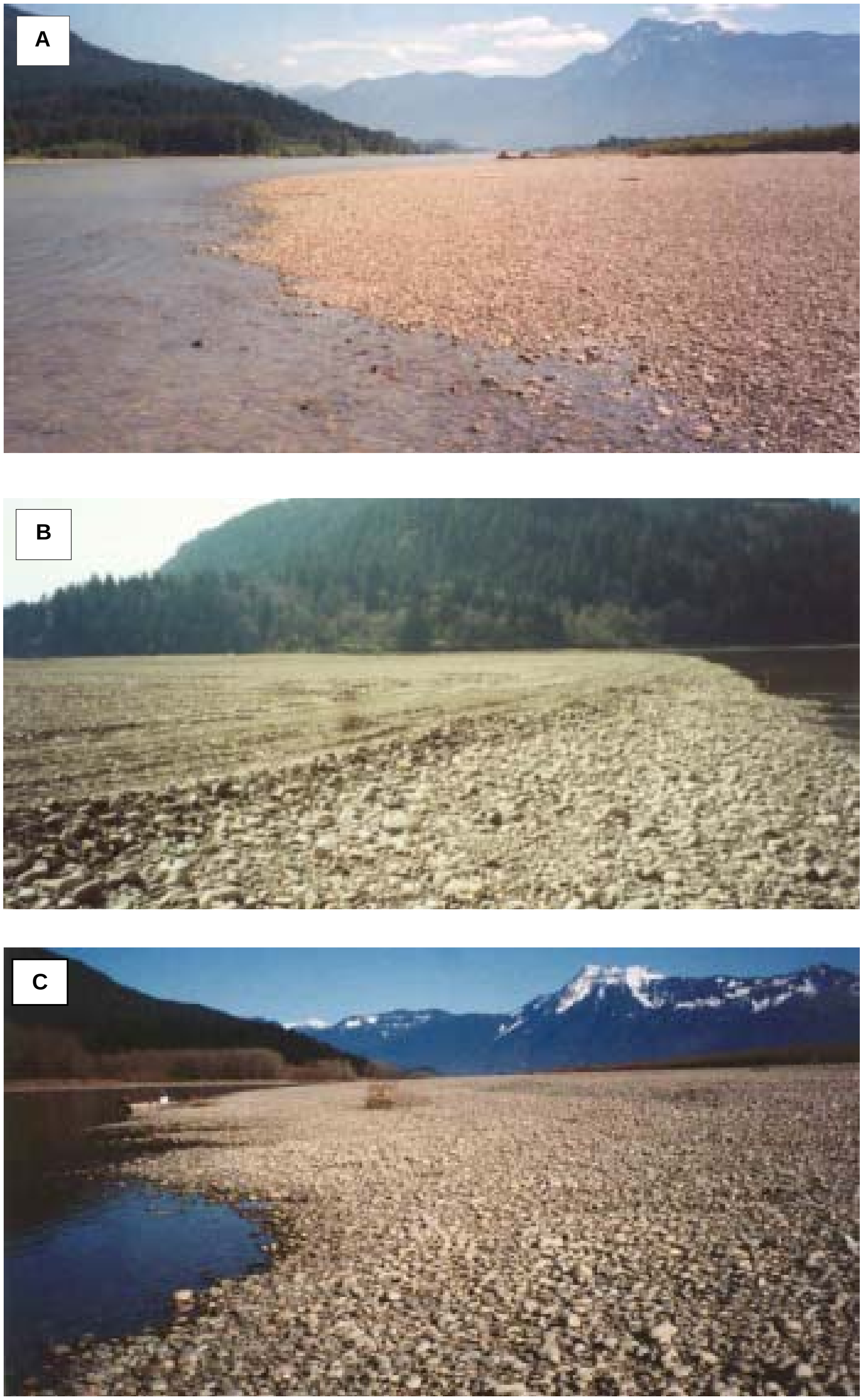


Figure 7-8. Lower Harrison Bar A) before scalping (upstream view August 17, 1999), B) post-scalping (downstream view March 26, 2000), and C) after one freshet post-scalping (upstream view March 7, 2001).

Table 7-4. Results of single-factor ANOVA contrasting surface sediment texture over three dates (February 2000, September 2000, September 2001).

Parameter	**df**	**MS**	**F**	***p***
Water Edge				
Proportion Sand	2 (3)	0.04	0.83	0.52
D_5	2 (3)	0.2	2.74	0.21
D_{50}	2 (3)	43.3	0.75	0.54
D_{95}	2 (3)	384.3	0.53	0.64
Inner Bar				
Proportion Sand	2 (3)	0.1	33.31	0.009*
D_5	2 (3)	0.02	3.07	0.19
D_{50}	2 (3)	19.3	1.67	0.32
D_{95}	2 (3)	33.5	0.83	0.52

* denotes a significant difference at a critical value of $\alpha = 0.0125$, corrected by Bonferroni's method for multiple contrasts.

7.5.2 Bar Topography

The *Lower Bar Boundary* of Harrison Bar, as delineated in **Figure 7-6**, had a relatively simple morphology prior to mining and consisted of two flat, open areas that dipped moderately steeply over coarse gravel beaches into the main channel (**Figure 7-9**). These open areas were separated from the vegetated islands in the centre of the bar by a summer channel containing complex chute and lobe features. Average surface elevation was 8.36 m and maximum elevation was 11.64 m (**Table 7-5**). Morphology in the reference area (Har-R) was equally simple: a large flat-topped area dipping gently to the side channel on its left (south) and more steeply to the main channel on its right (north).

Scalping removed the moderately steep beach face from the upper scalped area (Area A), producing a low gradient slope (1-2%) running without interruption from the low water edge to a high point in front of the central islands. Within removal Area B, scalping left a low-lying basin separated from the upstream scalped area by a low berm. Area B joined the main channel opposite the Harrison River confluence as a trough-shaped tongue. Removal volumes in Area A and B were 49,484 m^3 and 19,586 m^3, respectively, and totalled 69,070 m^3. The maximum vertical depth of extraction was approximately 2 m (along the inner boundary of removal area A), and average surface elevation was reduced by over 1 m (**Table 7-5**).

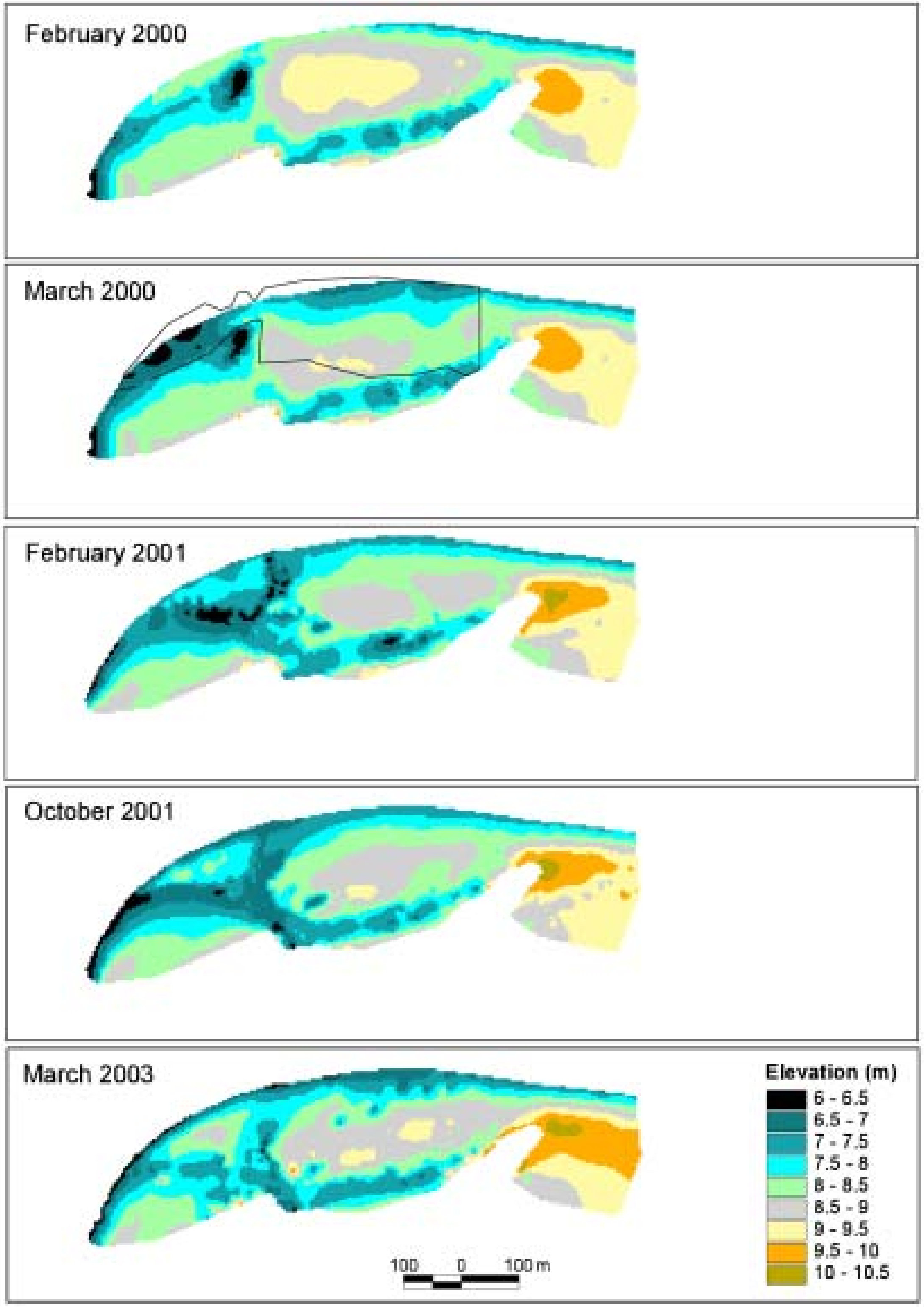


Figure 7-9. Topographic images portraying surface elevation classes at Harrison Bar. The images are based on repeated surveys of Harrison Bar before (February 2000), immediately after (March 2000), and on 3 dates following freshet events post-scalping. The perimeter outlined in March 2000 delineates the scalp boundary.

Table 7-5. Surface elevation (m) characteristics of lower Harrison Bar (total area: 247,825 m^2). Scalping took place immediately following the survey in February 2000. River discharge (m^3 s^{-1}) was estimated at Hope.

Date	Discharge	Mean	Maximum	Minimum	SD
Feb-00	888	8.36	11.64	5.67	0.71
Mar-00	733	8.14	11.64	5.48	0.75
Mar-00*	733	7.16	9.71	5.48	0.98
Feb-01	521	8.12	10.89	6.05	0.78
Oct-01	1320	8.13	11.93	5.76	0.81
Mar-03	900	8.25	11.99	5.60	0.81

*determined within removal area boundary only (Area A and Area B; 91,645 m^2).

The modest freshet of 2000 produced negligible volumetric change in Area A (715 m^3 erosion) and deposited 3838 m^3 of sediment within Area B (**Table 7-6**). This material was seen as exposed gravel bar surrounded by a relatively deep summer channel intersecting the lower corner of the bar (**Figure 7-9**). Over the entire lower bar, a net loss of 6635 m^3 was recorded after the 2000 freshet. This loss is reflected in the average and maximum bar surface elevation (**Table 7-5**).

A negligible volumetric change was calculated over the lower bar after the even lower 2001 freshet (1676 m^3 net erosion, **Table 7-6**), however, some topographic changes were observed within the removal area. There was deposition both in Area A (where maximum surface elevation increased to >9 m) and Area B (where maximum elevation increased to >8 m), including some infilling of the summer channel (**Figure 7-9**).

Topographic changes over lower Harrison Bar were most notable following the large freshet of 2002, which deposited a net sediment volume of 27,630 m^3 (**Table 7-6**). Sediment deposition occurred across the inner and middle scalped area, as well as the lower area. Erosion was noted along the downstream corner on the apex of the bend.

Comparing the *Lower Bar Boundary* of Harrison Bar between February 2000 and March 2003, there was a net loss of 42,913 m^3 of sediment. Given a scalped volume of 62,232 m^3 within the lower bar boundary, 31% of the removed volume appears to have been replenished after 3 freshets. All of this material was deposited during the most recent and largest freshet. Whereas scalping lowered mean bar surface elevation by 22 cm, sediment deposition by the 2002 freshet restored average bar surface elevation to within 9 cm of the pre-scalped surface (**Table 7-5**).

Table 7-6. Volumetric (m^3) comparisons between surveys conducted at Harrison Bar. Scalping occurred within areas A and B following the February 2000 survey. ne: not estimated.

Survey Comparisons	**Area A (71,975 m^2)**	**Area B (19,670 m^2)**	**Lower Bar* (247,825 m^2)**
Feb-00 vs Mar-00	*fill*: 0 **net loss: -49,484**	*fill*: 0 **net loss: -19,586**	*loss*: -63,881 *fill*: +1,648 **net loss: -62,232**
Feb-00 vs Feb-01	*loss*: -50,697 *fill*: +498 **net loss: -50,199**	*loss*:-15,797 *fill*: +49 **net loss: -15,748**	*loss*: -89,200 *fill*: +20,3328 **net loss: -68,868**
Mar-00 vs Feb-01 (2000 freshet)	*loss*: -14,233 *fill*: 13,519 **net loss: -715**	*loss*:-3,791 *fill*: +7,629 **net fill: +3,838**	*loss*: -47,476 *fill*: +40,840 **net loss: -6,635**
Feb-01 vs Oct-01 (2001 freshet)	ne	ne	*loss*: -28,414 *fill*: +26,737 **net loss: -1,676**
Oct-01 vs Mar-03 (2002 freshet)	ne	ne	*loss*: -23,348 *fill*: +50,978 **net fill: +27,630**
Feb-00 vs Mar-03	ne	ne	*loss*: -81,317 *fill*: +38,400 **net loss: -42,913**

*Inconsistencies arise among values because the "Lower Bar" excludes a small portion of Areas A and B, which was not consistently surveyed. See Figure 11.

Bar scalping resulted in a shift in the elevation profile of the *Lower Bar Boundary* of Harrison Bar (**Figure 7-10a**). Approximately 24% (or 58,925 m^2) of the lower bar area was >9 m elevation before scalping, which declined to 14% (or 35,300 m^2) after scalping and after one freshet in September 2000 (**Table 7-7**). Deposition resulting from flooding in 2001 and 2002 increased the bar area >9 m to 18%, or 44,750 m^2. This increase is reflected in a positive shift of the elevation profile towards the pre-scalped state (**Figure 7-10a**). The proportion of bar area >8 m elevation was 71% prior to scalping, 4% immediately after scalping, and 57% after the freshet of 2002. Based on the relation between water surface elevation at Harrison Bar and discharge at Hope (**Figure 7-11**), the entire bar surface surveyed in 2003 would be inundated at a discharge of 10,760 m^3 s^{-1}. The pre-scalp surface would have been inundated completely at 10,200 m^3 s^{-1}. With consideration to the elevation-area relation, 24% of the lower bar area remained exposed at flows of 4960 m3 s^{-1} whereas 18% of the post-scalp surface in 2002 was exposed at this flow.

Despite no long-term reduction in maximum surface elevation, there remained a net loss of bar area >8.5 m elevation (**Figure 7-10b**), which becomes inundated at flows exceeding 3960 $m^3 s^{-1}$. Such flows include the period of hatching and rearing for fish in the gravel reach. The reduction in area of the 9-9.5 m elevation class was particularly notable.

Table 7-7. Bar area (m^2) and percentage (%) of the total area at greater than three surface elevations for the *Lower Bar Boundary* of Harrison Bar (247,825 m^2 total area).

Date	**>8 m**		**>8.5 m**		**>9 m**	
	m^2	**%**	**m^2**	**%**	**m^2**	**%**
Feb-00 (pre-scalping)	175,725	71	111,750	45	58,925	24
Mar-00 (post-scalping)	9,775	4	1,625	1	0	0
Feb-01 (2000 freshet)	135,900	55	80,225	32	35,300	14
Oct-01 (2001 freshet)	136,950	55	80,975	33	38,250	15
Mar-03 (2002 freshet)	141,250	57	94,250	38	44,750	18

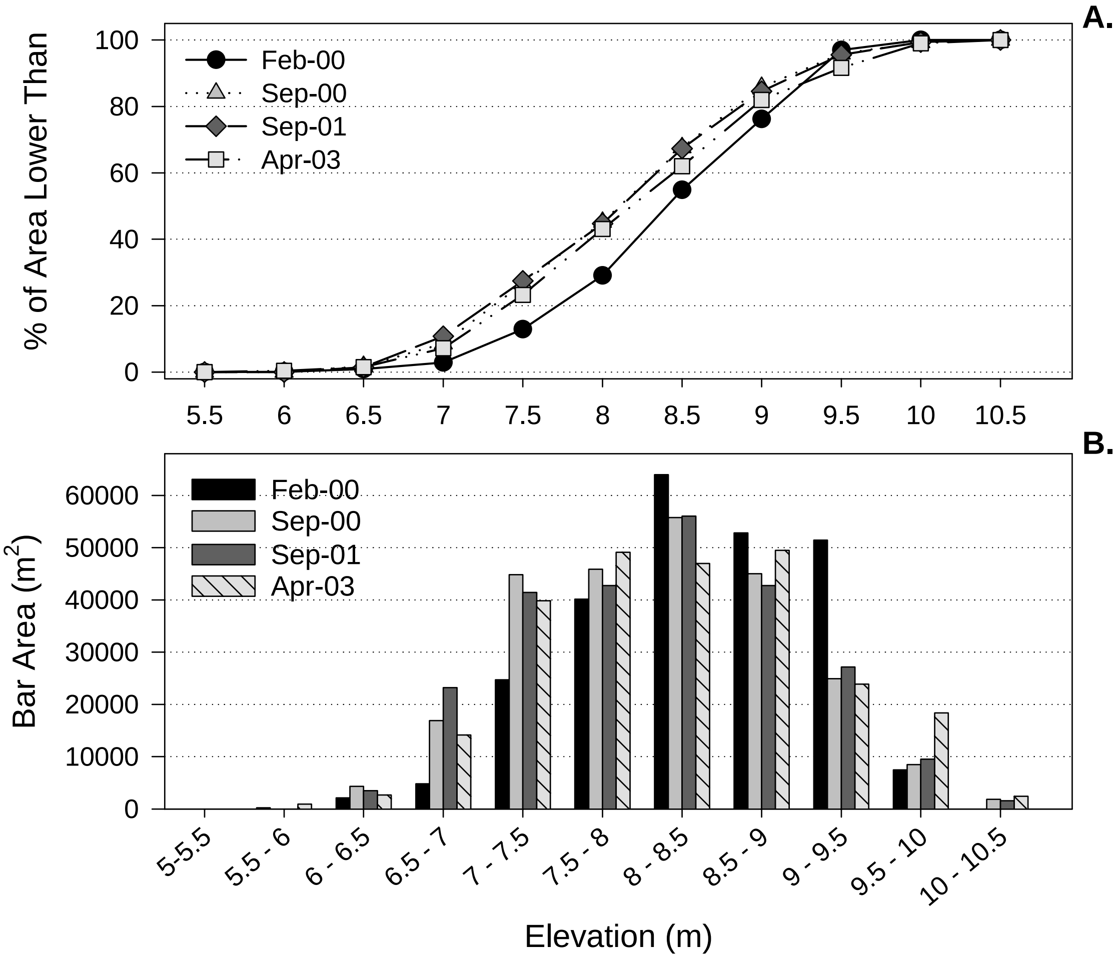


Figure 7-10. Area-elevation relation within the *Lower Ba7r Boundary* of Harrison Bar, based on topographic surface modeling before scalping (February 2000) and after three freshets.

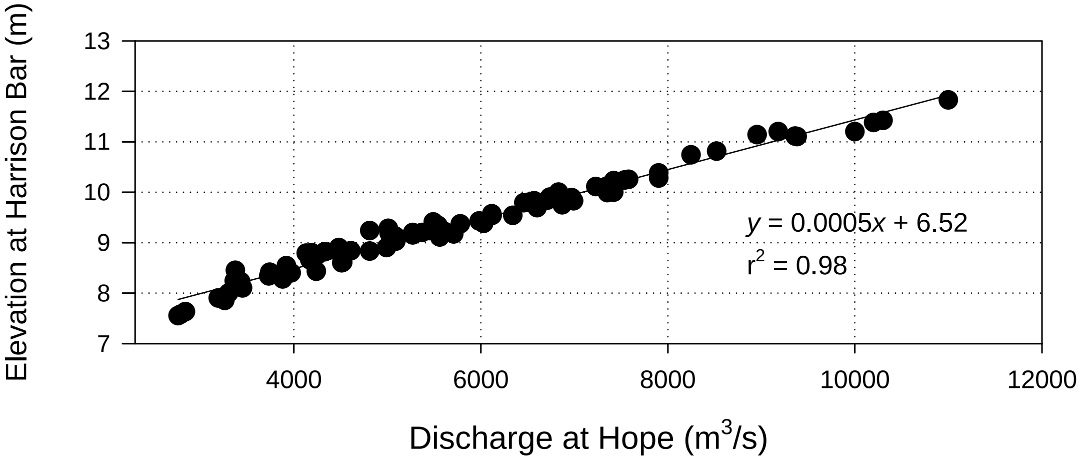


Figure 7-11. Discharge-elevation relation at Harrison Bar based on gauge data from the CPR Bridge at the mouth of Harrison River (1995 – 2002).

7.5.3 Habitat Availability and Physical Characteristics

7.5.3.1 Photographic Mapping

The availability of habitat units at low flow (500-860 $m^3 s^{-1}$ discharge) remained similar in the reference area of Harrison Bar between 1999 and 2001 (**Figure 7-12**). Within the scalped area, only minor changes were observed in habitat availability between 1999 and 2000. Air photographs taken in March 2001 after one freshet since scalping revealed a higher number (6 units) of habitat units around lower Harrison Bar. Two distinct open nooks developed at the midpoint of the scalped area. Riffle, flat bar edge, and bar tail units were observed around the downstream corner.

Habitat value, *estimated as the total number of units representing each habitat type multiplied by average fish density*, remained virtually unchanged in the reference area between 1999 and 2001. In contrast, habitat value within the scalp area was lower in 2000 (0.26) than in 1999 (0.51, **Table 7-8**). Gravel mining was not responsible for this change because the bar flank was unmodified by scalping in the 2000 photograph. In March 2001, after one freshet post-scalping, habitat value within the scalped area increased to 0.73. Substituting total counts with total lengths of habitat types produced similar results. The direction of change in habitat value is noteworthy, given the natural variation in habitat availability at reference sites over the same period.

Table 7-8. Habitat value in the reference and scalped areas of Harrison Bar. Values in parentheses indicate the mean ± SE of all reference sites. Photographs in each year were taken in March when discharge was less than 900 $m^3 s^{-1}$.

Habitat	**March 1999** *(pre-scalp)*		**March 2000** *(pre-scalp)*		**March 2001** *(post-scalp)*	
	Har-R	**Har-S**	**Har-R**	**Har-S**	**Har-R**	**Har-S**
Bay	0	0	0	0	0	0
Flat Bar Edge	.08	.08	.08	.08	.08	.16
Bar Head	.13	0	.13	0	.13	0
Bar Tail	0	0	0	0	0	.10
Cut Bank	0	0	0	0	0	0
Channel Nook	0	.43	0	0	0	0
Eddy Pool	0	0	0	0	0	0
Open Nook	0	0	0	.18	0	.35
Riffle	.12	0	.13	0	0	.12
Total	**0.33** (0.74 ± 0.2)	**0.51**	**0.33** (0.28 ± 0.2)	**0.26**	**0.21** (0.32 ± 0.2)	**0.73**

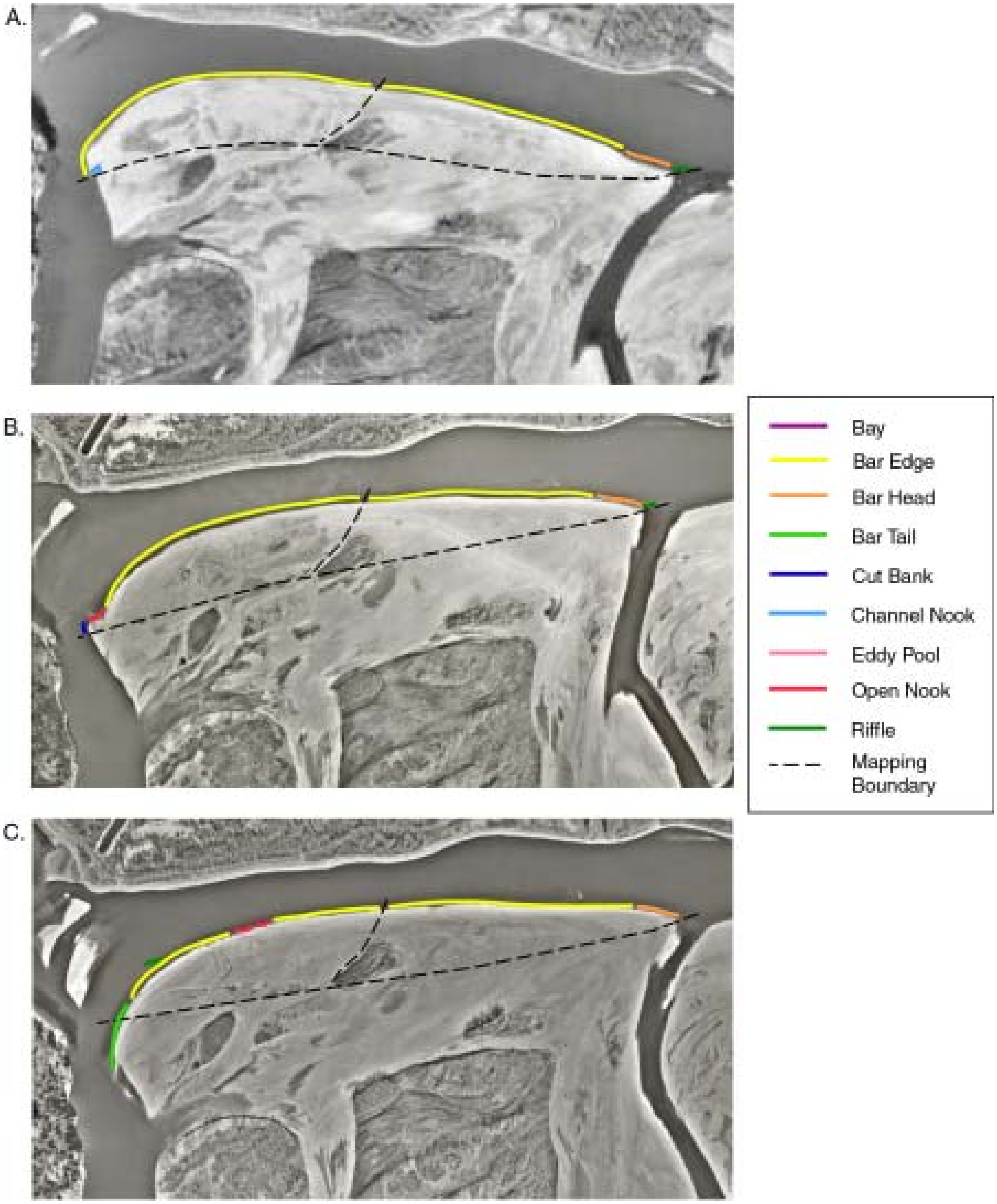


Figure 7-12. Habitat units around Harrison Bar on (A) March 27, 1999 (discharge: 860 m^3 s^{-1}); (B) March 10, 2000 (discharge: 677 m^3 s-1); and (C) March 7, 2001 (discharge: 502 m^3 s^{-1}).

Based on oblique photographs, habitat availability in the reference area showed little change between August 1995 and 2000 (**Figure 7-13**). Within the scalped area, the number of habitat units increased from 6 to 15 units. The increase resulted mostly from a new summer channel that intersected the lower corner of the removal area, making available flat bar edge, channel nook, eddy pool, and open nook habitats. Habitat value in the scalped area increased substantially as a result (**Table 7-9**). The channel-enriched habitat availability at intermediate flows and was likely the consequence of lowering the bar top by scalping and shaping the lower removal area as a trough.

A comparison of photographs taken before and after gravel mining in September 1999 and 2001 found a minor reduction in habitat value in the reference area and a low number of habitat units overall (**Figure 7-14**). The same trend was observed at Big Bar, a surrogate reference site for which photographs are available. Within the scalped area there were 4 units before and 6 units after mining. Habitat value in the scalped area increased from 0.84 to 1.70, mostly due to the presence of open nook. It is worth noting that discharge at the time of photography in 2001 was almost 400 $m^3 s^{-1}$ lower than in 1999. On-site sampling 5 days before the photography when discharge was 1780 $m^3 s^{-1}$ (and more comparable to the 1999 photograph) found the summer channel conveying flow and offering a high number and variety of habitats for juvenile fish. Hence, habitat availability post-scalping has been under-represented in this photographic comparison.

Table 7-9. Habitat value in the reference and scalped areas of Harrison Bar. Discharge in each pair of years was approximately equal.

Habitat	**August 1995** *(pre-scalp)*		**August 2000** *(post-scalp)*		**September 1999** *(pre-scalp)*		**September 2001** *(post-scalp)*	
	Har-R	**Har-S**	**Har-R**	**Har-S**	**Har-R**	**Har-S**	**Har-R**	**Har-S**
Bay	0	.38	0	0	0	0	0	0
Flat Bar Edge	.10	.10	.10	.59	.10	.10	.10	.20
Bar Head	.09	0	.09	.09	0	0	.09	0
Bar Tail	0	.16	0	.16	0	.16	0	.16
Cut Bank	0	0	0	0	0	0	0	0
Channel Nook	0	0	0	.90	0	.45	0	.45
Eddy Pool	.29	0	0	.29	0	0	0	0
Open Nook	0	1.52	.76	2.28	.76	0	0	.76
Riffle	.13	.13	.13	.13	0	.13	0	.13
Total	**0.61**	**2.30**	**1.08**	**4.45**	**0.86**	**0.84**	**0.19**	**1.70**

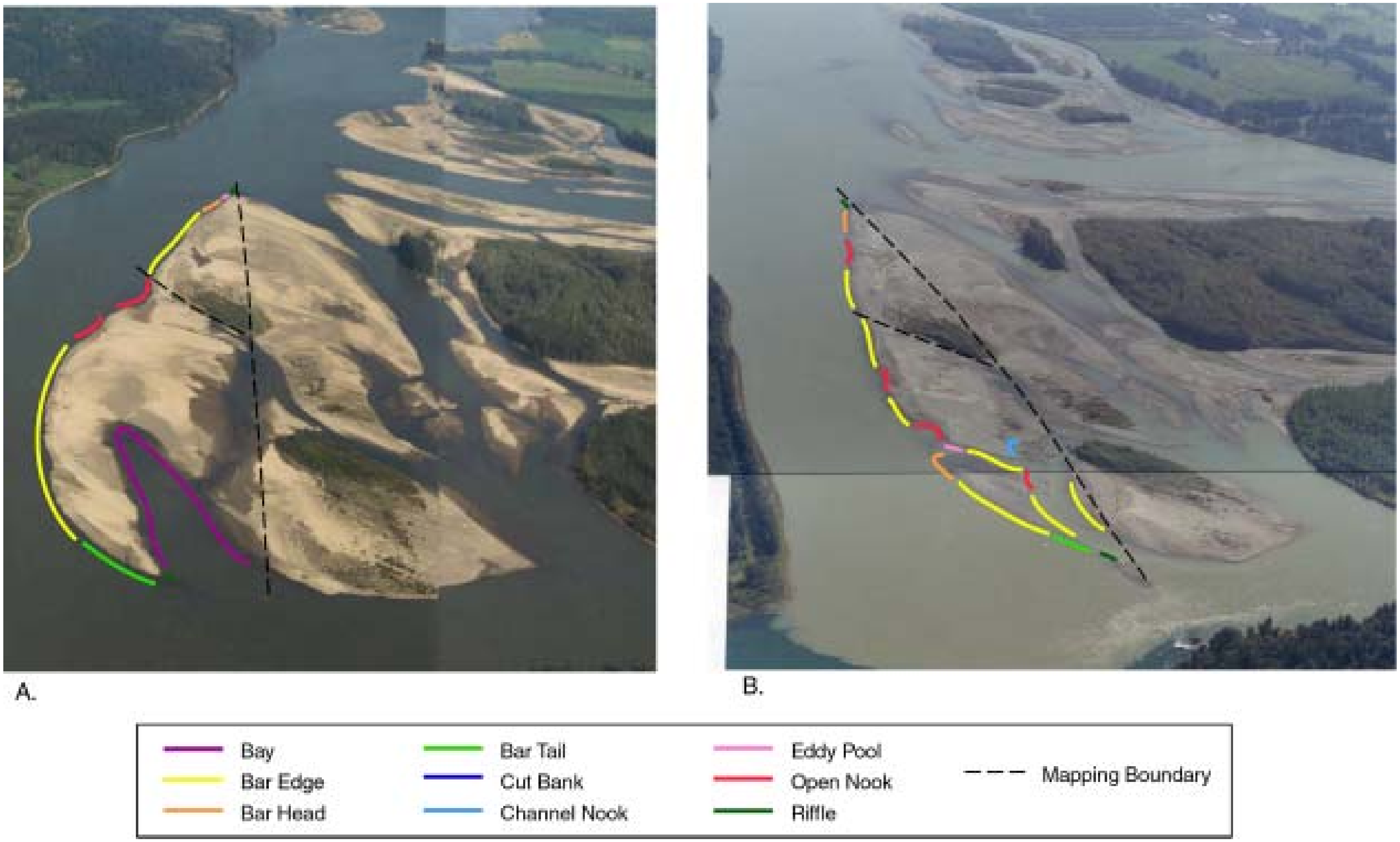


Figure 7-13. Habitat units around Harrison Bar on A) August 30, 1995 (discharge: 2680 $m^3 s^{-1}$) and B) August 21, 2000 (discharge: 2844 $m^3 s^{-1}$). One freshet event had occurred since scalping in February 2000. Photos courtesy of Dr. V. Galay.

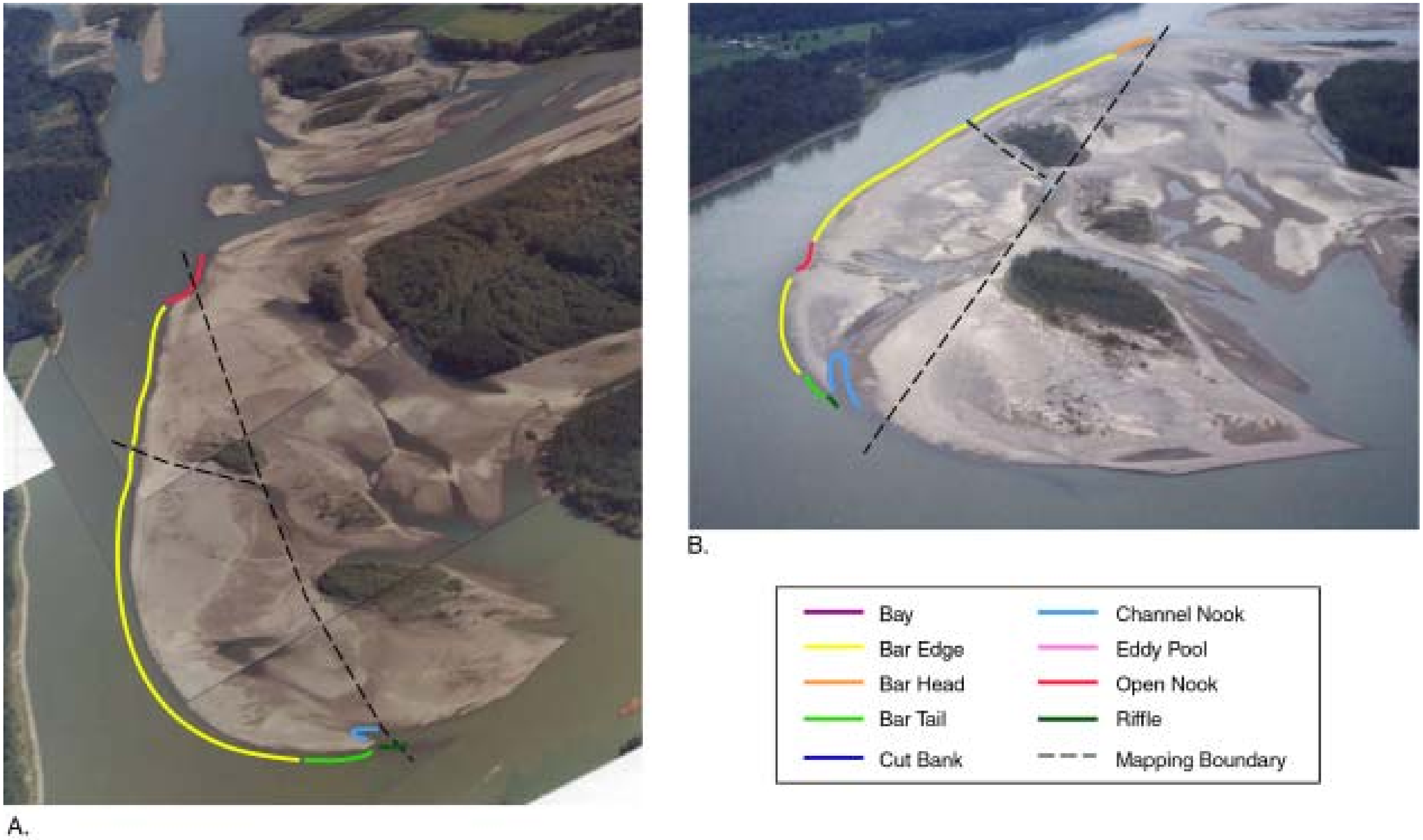


Figure 7-14. Habitat units around Harrison Bar on A) September 27, 1999 (discharge: 1950 $m^3 s^{-1}$) and B) September 20, 2001 (discharge: 1580 $m^3 s^{-1}$). Two freshet events had occurred between the gravel removal and photography in 2001. Photo (A) courtesy of Dr. V. Galay.

7.5.3.2 Physical Contrasts Between Habitat Types

Principal components analysis (PCA) explained 90.5% of the total variation of flat bar edge habitat characteristics in the first three PC axes. PC-1 explained 44.3% of the variation and represented a hydraulic gradient of increasing water depth, bank angle, and velocity. PC-2 accounted for 25.6% of the variation and was correlated with cobble and gravel to represent a gradient of coarse sediment. PC-3 explained 20.6% of the variation and was correlated with sand, representing a gradient of fine sediment. The loading of each physical variable on the PC-axes is given in **Table 7-10**. Asymmetrical ANOVA showed a significant short-term impact along the hydraulic gradient represented by PC-1 (**Table 7-11**), and graphical examination suggested that the timing of impact was coincident with scalping (**Figure 7-15a**). However, the limited number of sampling episodes prior to mining constrained the power of the test to confirm that the impact was due to scalping. (Refer to **Appendix I** for the complete analysis.) Sedimentary gradients represented by PC-2 and PC-3 showed no change related to scalping, consistent with results from univariate analysis of surface sediment texture.

Table 7-10. Factor loadings from principal components analysis of flat bar edge habitat units. Variables significantly correlated with PC-axes are highlighted in bold.

Variable	**PC-1**	**PC-2**	**PC-3**
Cobble	-0.63	**0.71**	-0.26
Gravel	0.29	**-0.88**	-0.35
Sand	0.55	0.08	**0.78**
Bank Angle	**-0.81**	-0.31	0.44
Average Depth	**-0.82**	-0.27	0.42
Average Velocity	**-0.73**	-0.26	-0.27
Eigenvalue	2.66	1.53	1.24
% Variance Explained	44.3	25.6	20.6

Table 7-11. Results of the asymmetrical ANOVA examining impacts of scalping on the physical characteristics of bar edge habitat units. Refer to Chapter 8, **Section 8.4.1**, for further details on the analysis.

Variable	Do reference locations have variable short-term trends after scalping?	Does inclusion of scalped site affect short-term temporal trend?	Do reference locations vary in difference from before to after scalping?	Does scalping affect differences before or after?	Conclusion	p-value
Hydraulic Gradient (PC-1)	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001
Coarse Sediment Gradient (PC-2)	No	No	No	No	No impact detected	0.72
Fine Sediment Gradient (PC-3)	No	No	No	No	No impact detected	0.48

Figure 7-15a shows a seasonal shift in PC-1 at Har-S relative to reference sites from shallow and lower velocity conditions in summer months to deeper and faster flowing water in winter. The shift was observed over three summers of sampling but only in April and May 2000 did PC-1 scores for Har-S fall outside the range of conditions observed at the reference sites (based on 95% confidence intervals). April and May 2000 corresponded to the period when flooding began to inundate the scalped site. The difference in hydraulic conditions between reference sites and Har-S was less in August and September 2000 on the declining limb of the hydrograph, and PC-1 values in September 2001 were similar to those in September 1999 at reference sites and Har-S. Values of PC-2 and PC-3 were similar at Har-S and the reference sites on all dates before and after scalping (**Figure 7-15b, c**).

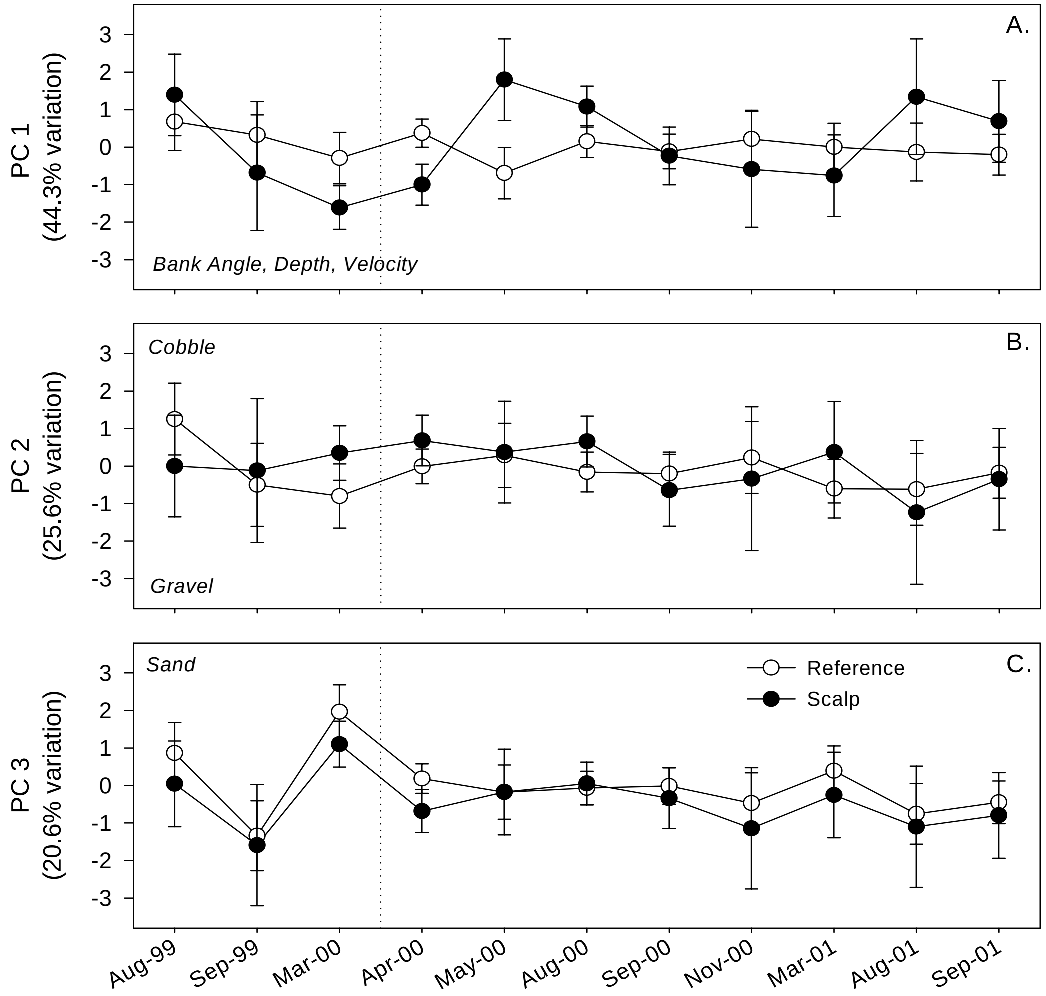


Figure 7-15. Mean factor scores (± 95% confidence interval) for reference sites and the scalped site, derived from principal components analysis of flat bar edge habitat characteristics. The proportion of variation explained by each PC-axis is given and the variables most highly correlated with each axis are listed in italics. The vertical dotted line marks the timing of bar scalping.

7.6 Discussion

7.6.1 *Bar-Scale Physical Changes*

Immediate physical changes to Harrison Bar as a result of gravel mining were substantial. Within the scalped boundary, the proportion of bar area at greater than 8 m elevation was reduced from 71% to 4% and the area >9 m elevation was reduced from 24% to 0%. The scalped area was graded to a slope of approximately 2% and left topographically simple, consistent with expectations. The evenly graded, homogeneous area of unconsolidated fine gravel and sand replaced a pre-existing coarse and relatively stable bar surface. However, the dramatic change in surface sediment texture due to scalping was relatively short-lived. Flooding in spring 2000 transformed the loose and sandy substrate into a moderately coarse surface with negligible sand cover. Sand cover in the scalped and reference areas of Harrison Bar was similar (~1%) after flooding, which indicated that a significant amount of sand was entrained from across the entire bar surface. Flooding in 2001 produced additional surface coarsening within the removal area, particularly along the water edge where median grain size increased from 29 mm before scalping to 35 mm after scalping and after two freshets. The coarsest fraction (D_{95}) increased in size along the water edge after scalping, but to a lesser extent: the grain size after two freshets remained lower than prior to scalping.

Topographical changes after scalping of lower Harrison Bar occurred concurrently with sedimentary changes as a result of flooding. Flood discharge in 2000 and 2001 was below average and produced negligible volumetric change (net erosion), both over the entire lower bar and locally within the removal area. However, there was a relatively large and balanced exchange of sediment (deposition and erosion) over the lower bar in each freshet that resulted in topographical changes and minor rebuilding of high bar habitat. It remains uncertain whether or not this amount of sediment exchange is typical of gravel bars in Fraser River or if the destabilized bar surface after scalping was more easily entrained. Other studies have documented increased sediment entrainment locally after gravel mining (USACE 1982).

Topographical changes after two freshets included an increase in maximum bar surface elevation but a slight decrease in average bar elevation, deposition of an isolated gravel bar at the lower corner, and erosion of a summer channel that flowed diagonally across the lower bar. The channel had irregular geometry with high habitat diversity, and established a flow connection between the main channel and inner side channel through until November 2000 (discharge

>1500 $m^3 s^{-1}$). The new channel increased the amount of wetted area available for fish and was host to a variety of habitat types including eddy pool, open nook, and flat bar edge. Fresh sediment deposited by the 2001 freshet cut off flow into the channel after September 15, 2001 (discharge >1800 $m^3 s^{-1}$) and in 2002, the channel carried flow through August but was cut off prior to September 15 (discharge >2000 $m^3 s^{-1}$). Although the channel appears to be infilling, it provided diverse aquatic habitat in the short-term after scalping as the bar was changing toward its new configuration.

Of particular significance was the increased availability of channel nook and open nook units, which were shown in Chapter 6 to host high densities and distinct species assemblages of fish. The increase was not statistically significant, but the pattern of habitat diversification after scalping was consistent for three water levels examined by photography. Prior to scalping, the bar surface was an expansive, flat area of simple topography. Particularly at low flow, units of the most common habitat type, flat bar edge, were large (>700 m in length) and other habitat types were generally rare. Flat bar edge remained the most common habitat type at low flow one year after scalping, however, units were shorter in length and spaced between open nooks. Habitat response was therefore contrary to expectations that scalping would cause topographic simplification.

Flood discharge in 2002 was above average and deposited 27,630 m^3 of sediment over lower Harrison Bar, approximately 31% of the scalped volume. The bar surface was transformed, by way of flooding, from a simple and homogeneous surface into an irregular surface of variable elevation that offered greater topographical variability in comparison with pre-scalp conditions. Average bar surface elevation increased to within 11 cm of the average prior to scalping, and maximum bar elevation exceeded pre-scalp conditions by 35 cm in 2003. However, the proportion of bar area >9 m remained 6% less than before scalping, which becomes inundated at flows >4000 $m^3 s^{-1}$ and generally occurrs in May through August during the period when newly hatched fish are rearing in the gravel reach.

Multiple freshet events were necessary for surface sediment recovery and topographical diversification on Harrison Bar after scalping and a freshet of above-average discharge was required for notable sediment recruitment to the lower bar. Given the substantially greater bed material transport at higher flows, this condition is apt to be general along the river. However, such events may become less frequent over the next century, in light of recent climate change projections (Morrison *et al.* 2002). Morrison *et al.* (2002) predict a modest (5%) increase in mean annual flow for Fraser River, but a decrease of about 18% in the average peak discharge, or 1600 $m^3 s^{-1}$, concurrent with an increase in average summer water temperature by 1.9 ^oC. These projections have wide-ranging implications both for fisheries and gravel management, given that bar growth and island development are dependent upon high flow events, and that these processes are interrupted by bar top scalping.

7.6.2 Summary

As hypothesized, the freshet cycle and processes of sediment transport and deposition governed the temporal response of the physical habitat at Harrison Bar after scalping. Two freshets of below-average peak discharge resulted in substantial reworking of surface sediment and adjustments in topography across lower Harrison Bar. However, an above average flood exceeding 10,000 $m^3 s^{-1}$ peak discharge was necessary for sediment recruitment and rebuilding of high bar habitat. After two modest freshets and one large event, the proportion of bar area >9 m in elevation and inundated by flows exceeding 5000 $m^3 s^{-1}$ remained 6% less than prior to scalping. Repeated topographic surveys were necessary to detect this impact, which may directly affect fish because the amount of shallow water habitat during freshet was reduced. Interestingly, a comparison of average and maximum bar surface elevation from before to after scalping found negligible differences. Only by comparing the elevation-to-area relation and then relating it to the specific range of flows over which fish would be affected was the impact to fish habitat identified.

The exchange of sediment by erosion and deposition over the bar surface appeared to assist in site recovery because the scalped surface of Harrison Bar was uniformly graded to a 2% slope but developed topographical variability after only one freshet event. Topographical change continued over subsequent freshets and the transformation highlighted the critical role of sediment transport throughout the gravel reach in habitat creation and maintenance. Even in the absence of scalping, gravel bars undergo changes in sediment texture and configuration on an annual basis, creating alternating zones of sediment deposition and erosion. Gravel deposits divert the flow around them and the fact that these deposits shift in space causes episodic lateral instability that is important for maintaining a diverse array of channel networks and habitats. Gravel deposition also is important, at a local scale, for maintaining fish habitat of high quality by producing topographic irregularities across a bar surface and by episodically reworking and cleaning the substrate. Such fluvial processes and the relative physical instability of habitats characterize the “natural” state for the gravel reach to which resident populations of aquatic organisms are accustomed.

Chapter 8. Ecological Response To Disturbance By Gravel Mining

8.1 Introduction

The habitat associations of resident invertebrate and fish species have been characterized for the gravel reach of Fraser River. Chapter 5 showed that the distribution of invertebrates is correlated with velocity; however, the species assemblage is only modestly distinct within discrete habitat units defined by morphological or hydraulic attributes, even at a fine temporal scale of examination. Results from Chapter 6 indicate that fish community structure is correlated with both sedimentary and hydraulic attributes, and that fish assemblages associated with several habitat types are modestly distinct and identified at multiple spatial scales of examination. However, many fish species show relatively indiscriminate use of habitat types, having widespread distributions and relatively high abundances throughout the gravel reach.

The modest congruence between the physical habitat and the spatial distribution and abundances of invertebrate and fish species provides the basis for predicting ecological response to habitat change. A significant response, expressed as a change in the composition and/or relative abundances of species, is expected for communities in which there is a high degree of habitat specialization and the species assemblage within habitat types is consistently predictable (Poff and Ward 1990, Death 1995). Conversely, a community is expected to show resilience to habitat change if the assemblage of species and relative abundances are variable over time and space. Resilience to disturbance is also expected of communities in variable or spatially heterogeneous environments (Palmer and Poff 1997). These expectations for community-level response to habitat disturbance are evaluated in this chapter, with habitat disturbance exemplified as a gravel extraction by bar scalping at Harrison Bar.

Gravel mining in February 2000 at Harrison Bar produced substantial physical changes to surface sediment texture and bar topography. The scalped area was left more topographically simple than prior to scalping, and the stable cobble-gravel surface was replaced by a loose mix of gravel and sand. However, most physical changes were relatively short-lived. Sediment transport over the bar surface, mediated by a single flood event, renewed surface sediment texture and two freshets of below-average discharge produced topographical complexity and resulted in improved habitat diversity around the bar perimeter. However, an above-average flood event in 2002 was necessary for sediment replenishment to Harrison Bar. Topographical changes documented over several freshets highlight the role of sediment transport in habitat development, and suggest that the nature of physical changes at Harrison Bar did not differ significantly from what gravel bars undergo annually. On this basis, the physical changes at Harrison Bar may fall within the range that resident populations of aquatic organisms can tolerate.

8.1.1 Objectives

The objective of this chapter is to examine the temporal response of the invertebrate and fish community to bar scalping over two freshet cycles, and to determine if the nature of the response was detectable and significant. The ecological response is evaluated in the context of habitat changes documented in Chapter 7, because of the expectation that ecological impacts are mostly mediated through bar-scale morphological changes and habitat-scale alterations. Moreover, the magnitude of ecological response to habitat change is expected to be governed both by the severity of habitat change and by the strength of the association between species' assemblages and habitat attributes.

It is hypothesized that physical disturbance by scalping and the resulting habitat changes cause a reduction in the abundance and diversity of invertebrates and fish, but that the detectable ecological response is relatively short-lived, consistent with the short duration and extent of documented physical changes. Furthermore, ecological recovery at Harrison Bar, by way of restored abundances and diversity of invertebrates and fish to pre-scalping levels, is expected to coincide with freshet, which was responsible for the rapid recovery of surface sediment texture and topographical complexity.

8.2 Sampling Design

8.2.1 Conceptual Approach

The conceptual approach to biological sampling followed a BACI-design (<u>B</u>efore-<u>A</u>fter-<u>C</u>ontrol-<u>I</u>mpact; Stewart-Oaten *et al*. 1986) whereby measurements are collected several times before and after an impact takes place from a control and an impacted site. BACI is a commonly used acronym, however, its original authors admit that the term "reference site" is usually more appropriate than "control site" (Stewart-Oaten and Bence 2001). Herein, the term <u>reference site</u> is used because conditions at a site remained subject to seasonal modification by river processes. The

BACI-design was modified for this study to include three reference sites because of the spatial variability of the system, which also was more favourable for analysis of fish and invertebrate data (described in **Section 8.4.1**). Selecting reference sites that resembled as closely as possible the physical characteristics at the scalped site was an important consideration. The three chosen reference sites introduced in Chapter 7 (Har-R, Fos-R, and Car-R) are believed to have met this criterion because of their proximity to Har-S and their similarity with respect to channel morphology, gradient, sediment transport regime, and substrate texture. Being situated upstream, each reference site also was unaffected by possible changes at Har-S after the removal.

The BACI design was first introduced as a solution to the problem of assessing the environmental effects of an unreplicated disturbance, such as gravel mining, where the location is not randomly assigned (Green 1979, Stewart-Oaten *et al.* 1986). This situation poses difficult statistical problems, identified by Hurlbert (1984), when the main goal is to determine whether the state of the impacted site differs significantly from what it would have been in the absence of the disturbance. Ideally, such a study would proceed as an experiment with a number of replicate sites (i.e., gravel bars), each randomly assigned to one of two treatments (i.e., scalping or no scalping), and then applying standard statistical analyses. However, this ideal situation rarely occurs. More often, and in the case of scalping at Harrison Bar, the location is not randomly determined and replication is not feasible.

Because the state of Harrison Bar in the absence of mining could not be observed post-scalping, an estimate was needed of the what that state would have been to compare with the observed condition. The BACI approach accomplished this by collecting samples at both the scalped site and nearby reference sites simultaneously (as nearly as possible). Replication was achieved by collecting the samples from all sites on a number of dates both before and after scalping. Differences among the reference and scalped sites prior to mining were taken to be an estimate of the difference expected in the period after mining had the removal not occurred. This design allows for natural differences between the reference and scalped locations, and for changes during the before and after periods that influence all sites in the same way (e.g., differences in discharge or water temperature).

Figure 8-1 illustrates the approach with a simple example. In case A, mean density is greater in the control area than in the impacted area and the average difference between impact and control does not change significantly from before to after (bottom panel), indicating that there has been no detectable impact. Case B illustrates a situation in which the disturbance has reduced density at the impacted site, leading to an increase in the difference from before to after the impact.

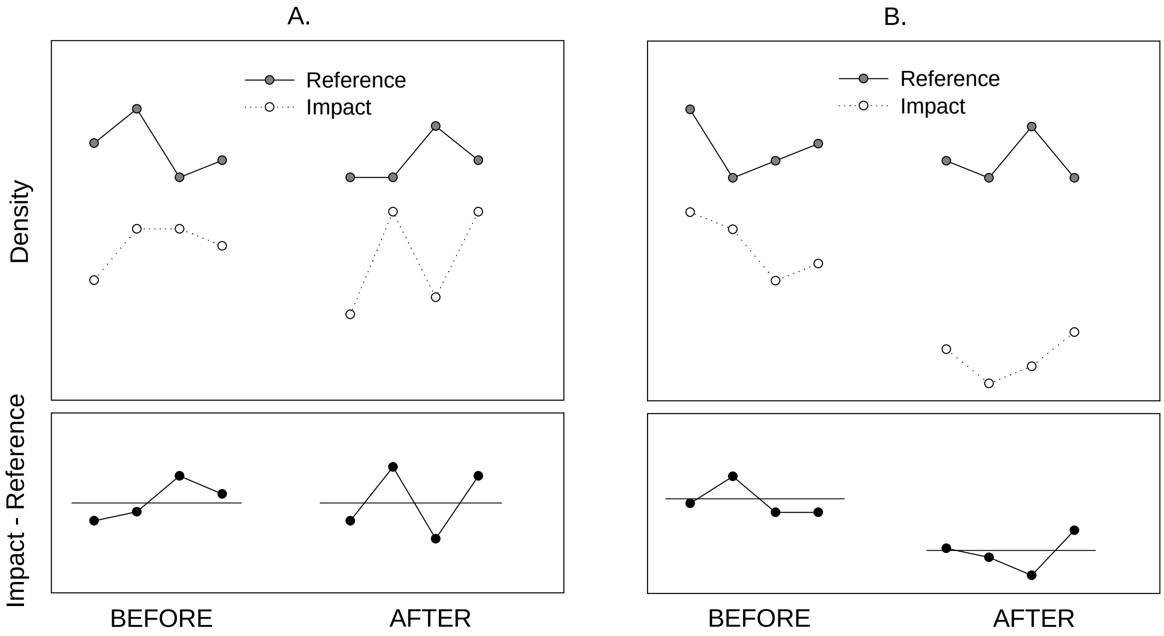


Figure 8-1. Hypothetical data collected according to the BACI design where average density is greater in the reference area than in the impact area. (A) The average difference in abundance between impact and reference does not change significantly from before to after (bottom panel), indicating that there has been no impact. (B) Disturbance has reduced density at the impact site, causing a change in the difference from before to after (adapted from Stewart-Oaten *et al*. 1992).

The example presented in **Figure 8-1** also illustrates a critical weakness of the BACI design: it provides no means to estimate the variability of the possible outcomes at the impact or reference sites, so that it becomes impossible to ascertain what a truly significant change at the impact site might be. On this basis, the design has been criticized. A solution to the problem was presented by Underwood (1991, 1992, 1994) as an extension of the BACI design, in which variance is dealt with by introducing multiple reference sites. This strategy leads to an asymmetrical analysis of variance in which sources of environmental variance are assessed from the observations at the reference sites. The solution introduced by Underwood was followed for analysis in this study, and details are given in **Section 8.4.1**.

8.2.2 Timing and Hydrology

Pre-removal sampling for benthic invertebrates and fish began in August/September 1999, somewhat fortuitously, as part of the research presented in previous chapters examining the physical and ecological organization of the gravel reach. At this time, the removal was not anticipated and consequently sampling effort at some sites was incomplete for the design requirements of the study. Only limited data were gathered from upper Harrison Bar and Foster Bar. Also, sampling was not stratified equally among available habitat types. Sampling took place on the declining limb of the hydrograph between 5260 m^3 s^{-1} and 1800 m^3 s^{-1}.

The Harrison Bar experimental removal was approved in late January 2000 and systematic monitoring of the scalp and reference sites began in February 2000, immediately prior to scalping. Sampling was repeated over 18 months following the removal, beginning in April 2000 when the rising water level began inundating the scalped area. Monthly sampling was scheduled originally but the sampling frequency was scaled back after May 2000 to be more cost effective. The revised schedule had an irregular sampling frequency and was intended to coincide with the timing of key life cycle stages of invertebrates and juvenile fish (**Table 8-1**). Sampling in January, March, and November targeted invertebrate larvae that typically mature through the winter and emerge as terrestrial adults by late March. Sampling in July through September of 2000 and 2001 targeted juvenile fish rearing in near-shore habitats as well as newly hatched invertebrate larvae. Summer and autumn fish sampling post-scalping occurred during flows of 5710 m^3 s^{-1} and 2880 m^3 $s^{-}1$ in 2000, and 5320 m^3 s^{-1} and 1780 m^3 s^{-1} in 2001. In total, invertebrate sampling occurred twice before and eight times after scalping. Sampling for juvenile fish occurred over three periods before and eight periods after scalping.

Table 8-1. Sampling schedule for juvenile fish (F) and benthic invertebrates (B). The shaded cell marks the timing of bar scalping.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1999								F	F, B			
2000		F	B	F, B	F, B			F, B	F, B		F, B	
2001	B	F	B					F	F, B			

Fish and invertebrate sampling in some months occurred at lower elevations than the limit of bar scalping (e.g., January through March). These samples were important for characterizing the overall time-series response to mining and to assist in the interpretation of patterns detected in other months. These data also were useful because the bar-scale response to mining was uncertain; it was possible that the impacts of scalping might extend laterally and cause the bar to erode, especially given the lateral instability of wandering rivers. It is recognized, however, that because summer sampling for fish and benthic invertebrates was carried out within the actual removal boundary, these samples may more likely reveal an impact due to bar scalping. Hence, they are given greater emphasis in data interpretation. It must also be recognized that no fish sampling took place at peak discharge (>5700 $m^3 s^{-1}$) due to safety and sampling constraints, but that this represents the most hydraulically stressful period for fish when the effects of reduced high-elevation bar habitat would be most severe. Topographical data presented in Chapter 7 allows for speculation of this impact, which is discussed later in the chapter.

8.3 Methods

Biological sampling for invertebrates and fish was intended to test the hypothesis that habitat changes caused by gravel mining significantly altered community structure as well as species abundance. Sampling methods were identical with those described in Chapter 3.

8.4 Data Analysis

Hypotheses presented in **Section 8.1.1** were tested using asymmetrical analysis of variance (Underwood 1991, Underwood 1992, Underwood 1994). The method is an extension of standard analysis of variance (ANOVA) developed to detect environmental impacts, and the procedure is described briefly below. The practical mechanics of the analysis are presented in Underwood (1993) and summarized briefly in **Appendix H**. Detailed breakdowns of all analyses are provided in **Appendix J** through **Appendix N**.

8.4.1 Asymmetrical Analysis of Variance

Asymmetrical ANOVA is an extension of the BACI design that requires repeated sampling at multiple reference locations in order to achieve spatial replication. Including multiple reference sites establishes a measure of sampling variance lacking in simple BACI designs, so that observed changes may be tested rigorously for significance. The design is "asymmetrical" because only the reference condition is replicated. Sampling on multiple occasions before and after scalping achieves temporal replication. According to the analysis, an impact is defined as some difference (negative or positive) in the *change* of mean abundance (or other parameter of interest) at the scalped site from before to after scalping compared with the change from before to after at the reference sites (see **Figure 8-1**). Thus, there must be a statistical interaction in the difference between the scalped and reference locations from before to after the disturbance.

The detection of an impact is complex because it may show up in different ways depending on the spatial and temporal consistency of the data being measured. Underwood (1993) provides a flowchart for proceeding through a set of statistical tests to address whether or not an impact has occurred (**Figure 8-2**). The answer to the question at each branch of the flowchart determines the sources of variation and degrees of freedom used to calculate an F-value in the ANOVA.

If, among reference sites, there is a significant temporal interaction after scalping (A, **Figure 8-2**), the test to detect a different temporal pattern at Har-S will not be very sensitive (few degrees of freedom in the denominator). However, this condition reflects the fact that there are large natural variations over time from one reference site to another. Accordingly, a specific impact would have to be large for it to push the system beyond its capacity to recover because, as Underwood (1993) points out, populations in a naturally variable environment likely have resilience and can recover from insignificant disturbance.

When scalping is found to cause a short-term interaction between Har-S and the reference sites (A2 or B2, **Figure 8-2**), the conclusion is that the temporal trend at Har-S was outside that found naturally at the reference sites. For this interpretation to be realistic, there must be no corresponding change in the interactions from before to after scalping among the reference sites. Otherwise, the evidence must be interpreted to mean that there has been a widespread change that affected all sites. It must also follow that the change in this temporal interaction between Har-S and the reference sites was coincident with the timing of scalping. It is important to note that the number of observations made prior to the disturbance greatly improves the sensitivity of this analysis. A low number of sampling episodes before scalping, particularly when variance is large, will afford the analysis few degrees of freedom and prevent it from definitively showing that the impact was coincident with scalping, leading to indeterminate results. In this study, fish were sampled over three periods prior to scalping and benthic invertebrates sampled on two dates prior to scalping; although additional pre-scalp sampling episodes were desirable, the imposed planning timeline and available resources determined the limits of the sampling program. The analysis is as rigorous as the data allow.

When short-term temporal interactions among reference sites are small and a short-term interaction is not detected between Har-S and the reference sites (B1, **Figure 8-2**), a more rigorous test for impact (more degrees of freedom in the denominator) is available. An impact is then evaluated at the larger time-scale of Before versus After (C and D, **Figure 8-2**). For an impact to be detected, there must be an interaction in the difference between Har-S and the reference sites before scalping compared with the difference after scalping. Otherwise, the final conclusion is that scalping did not produce a detectable impact.

Underwood (1993) raised the issue of serial correlation between sampling periods because, in analysis of variance, serial effects should be explicitly incorporated into the analysis or eliminated. Seasonal effects were incorporated in the analysis as a time-factor, but the possibility remains for persistence to influence fish samples taken in adjacent months at certain stages in the sampling program (e.g., August and September 2000). These samples collected in succession were intended to search for specific discontinuities in the data and to identify specific potential effects of central interest; hence, all these data have been retained in the analysis.

A critical value of $\alpha = 0.05$ was chosen for asymmetrical analyses. An α-value greater than 0.05 is sometimes chosen for impact assessment because failing to detect a real impact may have severe economic and social consequences that are possibly non-recoverable (Underwood 1993). However, the probability of committing a Type I error and rejecting the null hypothesis even when it is true (i.e., falsely detecting an impact) is increased when carrying out multiple analyses on the same dataset. The Bonferroni correction was applied in previous chapters to overcome the risk of Type I error inflation. The choice in this chapter of $\alpha = 0.05$ represents a compromise between inflating the risk of a Type I error by performing multiple analyses on the same dataset, and wanting to ensure that any real impact was detected.

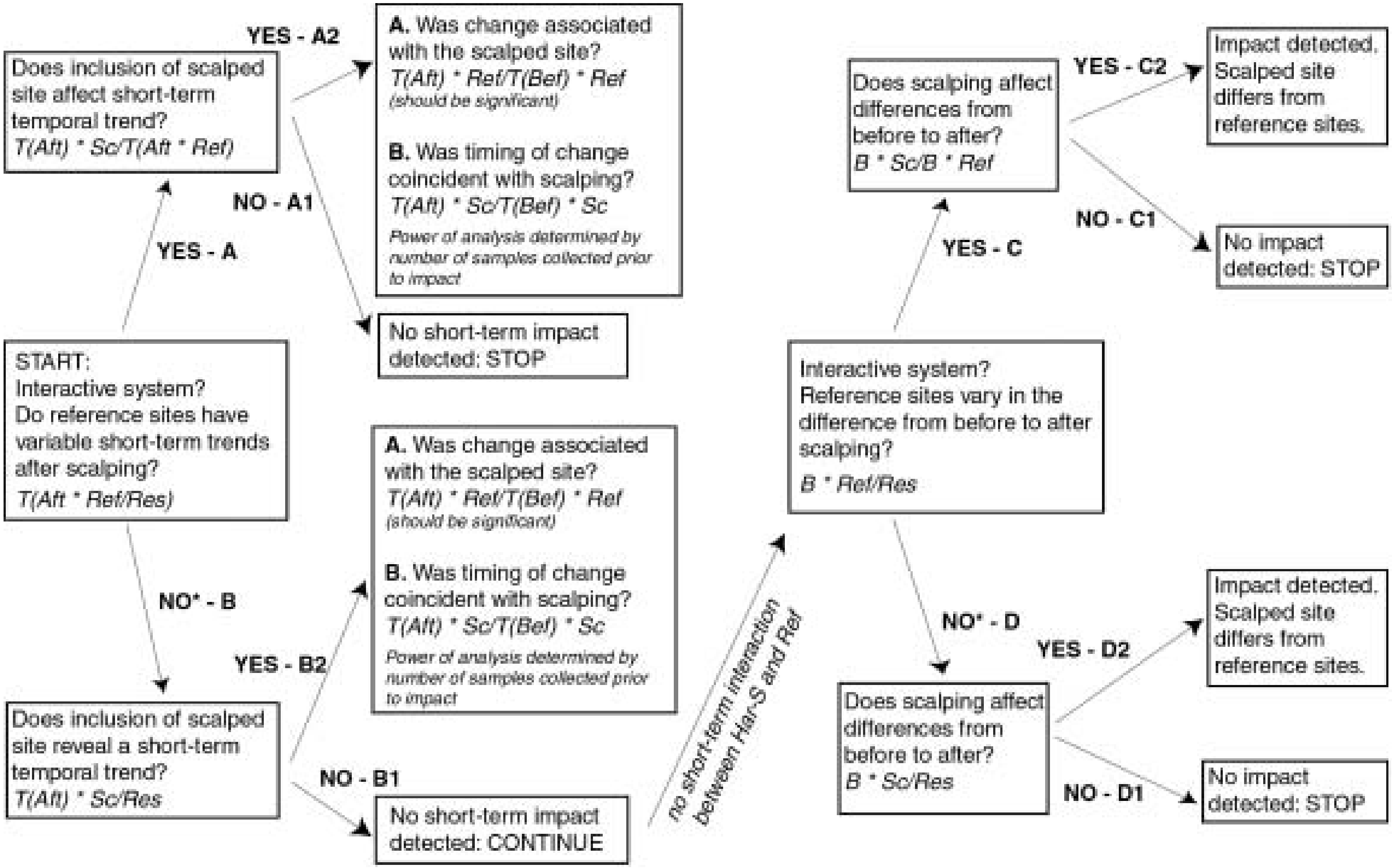


Figure 8-2. Sequence of questions for asymmetrical ANOVA to detect an impact (from Table 6, Underwood 1993). The answer to each question determines the sources of variation and degrees of freedom used to calculate the F-value. (Sources of variation for F-value calculation are as follows. Sc: Scalp site. Ref: Reference sites. Res: Residual. Bef: Before. Aft: After. T: Time.) *If "NO", the residual source of variation (error term) is used as the denominator in F-value calculations and sensitivity of the analysis is higher.

8.4.1.1 Power Analysis

A challenge for studies examining environmental impacts where results may influence management decisions is the need to quantify the power of the analysis to *detect* an impact. A null-result (i.e., a non-significant impact) can be generated from a small or non-effect, but may also occur when the power of the analysis to detect a real impact is very low. This is referred to as a Type II error. The *power* of a statistical test is defined as its capacity to reject, when appropriate, a null hypothesis (Underwood 1993). This is the complement of a Type II error; therefore, power is defined as (1 - Probability of Type II error).

Standard ANOVA has a straightforward method to determine power between group means given a specified significance level, sample size, and estimate of variance (see Zar 1984). Underwood (1993) has developed a similar method for power analysis to complement asymmetrical ANOVA, which was applied here in order to determine how likely the test was to detect a difference when no impact was detected. (Calculation of power is not relevant when a significant impact is detected; Peterman 1990). A breakdown of power calculations is presented in the appropriate appendices (**Appendix J** through **Appendix N**) and the mechanics of the calculation are described in **Appendix H**. Power analysis produces a number between 0 (no power) and 1 (maximum power) that represents the probability of detecting an impact when it actually occurs.

8.4.2 Benthic Invertebrates

The experimental removal was carried out over a large area of lower Harrison Bar encompassing multiple habitat types, which implied that the appropriate spatial unit for examining scalping impacts was the gravel bar. However, results presented in previous chapters demonstrated that gravel bars are an agglomeration of smaller habitat units that differ in physical and ecological character. Proceeding to analyze data grouped at the bar scale would pool this habitat-specific variability, thereby inflating the error variance and possibly reducing statistical power to detect a significant impact. However, considering only a sub-set of the data grouped by habitat type would reduce sample size and not directly address impacts at the bar-scale.

This dilemma was resolved by carrying out identical statistical analyses at the bar scale (i.e., all habitat types pooled) and habitat unit scale (i.e., flat bar edge habitat). Habitat-specific comparisons between sites before and after scalping examined <u>flat bar edge</u> only, because it was present at all sites on all dates before and after scalping. The analyses provided a useful contrast because analyses based on all habitat types pooled had a larger sample size, which afforded the analysis a higher number of degrees of freedom (i.e., power) from which to detect an impact. However, analyses conducted at the habitat-unit scale may be more sensitive because the variance due to habitat differences was accounted for rather than being incorporated into the error term of the analysis.

Comparisons of invertebrate data at the habitat- and bar-scale were made using a combination of (1) graphical examination and (2) asymmetrical ANOVA. The approach was chosen over multivariate techniques (e.g., MDS) due to the statistical rigor and quantitative capacity to detect an impact. Comparisons were based on several community metrics presented in Chapter 5 to assess differences in the invertebrate community between Harrison Bar and the reference sites before and after scalping: total density, % EPT (percent of animals belonging to the orders Ephemeroptera, Plecoptera, Trichoptera), taxon richness, EPT richness, Simpson's diversity, and Simpson's evenness. Margalef's richness and Shannon-Weiner's diversity were excluded because results from previous chapters indicated that each is generally redundant to that of taxon richness and Simpsons' diversity, respectively. Pielou's evenness was replaced by Simpsons' evenness because of its direct relation to Simpson's index. Simpson's evenness was calculated as:

$$E = {}^{1}\!/_{D} * s$$

where D is Simpson's Index and s is the number of unique species (taxa).

Variables were first transformed to meet assumptions of normality (Shapiro-Wilks test) and homogeneity of variance (Levene's test). Total density was log-transformed, % EPT was arcsine-transformed, and the remaining metrics of richness, diversity, and evenness did not require transformation. Graphical presentations of the data depict normal-transformed variables to reflect the scale on which statistical analyses were based.

There were two cases of missing flat bar edge data over the course of sampling: Fos-R in May 2000; and Car-R in September 2000. One case was missing for the bar-scale analysis, Fos-R in May 2000. Missing data create an unbalanced design that poses a statistical challenge for analysis. Following Underwood's (1997) recommendation, missing data were replaced with "dummy values" using the mean of the other reference sites on that date. These values generated no variance and contributed nothing to the sum of squares. The residual degrees of freedom were reduced by the number of dummy values to compensate for the missing values in all analyses.

Asymmetrical ANOVA also was used to examine population-level metrics by comparing densities of common taxonomic groups (representing greater than 1% of the total invertebrates collected) between sites before and after scalping. In most cases, family-level groupings were chosen because most families were represented by one common genus and several authors have found the family-level of identification sufficient for bio-monitoring purposes and for detecting site impairment due to physical, toxicological or biological disturbance (Bournaud *et al.* 1996, Reece *et al.* 2001). Seven groups met the abundance criterion: Baetidae, Heptageniidae, Ephemerellidae, Capniidae, Chironomidae, nematodes, and oligochaetes. Samples from all habitat types were included and invertebrate densities were log-transformed prior to analysis to meet assumptions of normality (Shapiro-Wilks test) and homogeneity of variance (Levene's test). A secondary goal for this analysis was to gain insight into patterns of recolonization by invertebrates of a scalped gravel bar.

8.4.3 Fish

Analyses of fish data were carried out at the bar scale (i.e., all habitat types pooled) and habitat unit scale, similar to benthic invertebrates. Habitat-specific comparisons between sites examined <u>flat bar edge</u> only because no other habitat type was represented at all sites during all periods of sampling. The removal site, in particular, consisted almost exclusively of flat bar edge habitat prior to scalping.

Several community metrics presented in Chapter 6 were calculated to assess differences in the fish community between Harrison Bar and the reference sites before and after scalping: total density, % Salmonidae, species richness, Simpson's diversity, and Simpson's evenness. Comparisons of bar-scale and flat bar edge samples were made based on a combination of (1) graphical examination and (2) ANOVA. Prior to analysis, each variable was tested for normality (Shapiro-Wilks test) and homogeneity of variance (Levene's test). To meet these conditions, total density was log-transformed, % Salmonidae was arcsine-transformed, and remaining metrics of richness, diversity, and evenness did not require transformation. Graphs depict normal-transformed variables to reflect the scale on which statistical analyses were based.

There were six cases of missing flat bar edge data over the course of sampling: Car-R August 1999; Car-R and Fos-R February 2000; Fos-R September 2000; Car-R November 2000; and Car-R August 2001. There were three missing observations for the bar-scale analysis: Car-R and Fos-R February 2000; and Car-R August 2001. Following Underwood (1997), missing data were replaced with "dummy values" using the mean of the other reference sites on that date. The residual degrees of freedom were reduced by the total number of dummy values to compensate for the missing values.

8.5 Results

8.5.1 Benthic Invertebrates

Asymmetrical ANOVA to examine scalping impacts on benthic invertebrates yielded identical results for all variables analyzed at the bar-scale and at the scale of flat bar edge habitat units. However, statistical power to detect an impact was higher for habitat-scale analyses. Graphical data and statistical results based on the habitat-scale only are presented below. Detailed reporting of ANOVA results is in **Appendix J** (whole bar), and **Appendix K** (flat bar edge units only).

8.5.1.1 Habitat-Scale Examination: Flat Bar Edge Units

A total of 85,704 benthic invertebrates were collected at Har-S and three reference sites between September 1999 and September 2001 (11 sampling episodes). Invertebrate density varied according to season at all sites, with March samples containing more than 4 times the density of animals as in August and September (**Figure 8-3**). Year to year variability was evident as well: average density in September 2000 was 4 times lower than in September 1999 and 9 times lower than in September 2001. A sudden rise in discharge in early September 2000 (refer to Chapter 3) may be the cause of low average density in these samples, in which case, August 2000 samples may be more representative of late summer conditions and more appropriately compared with September samples from other years.

Average density at Har-S was similar to that of reference sites prior to scalping and in April 2000, immediately after scalping. However, density was lower in May and August 2000. This change in the relation between scalped and reference sites is reflected in the top panel of **Figure 8-3**, comparing mean density at Har-S to the average of all reference sites on each date. Samples in May and August 2000 were collected within the removal boundary and from sediment previously disturbed by scalping. On all sampling dates after August 2000, density at Har-S was higher than the average of reference sites, particularly in January 2001. Notably high density at Har-S in January 2001 consisted of large numbers of Chironomidae and the mayfly *Baetis* sp. Asymmetrical ANOVA detected a short-term change in density at Har-S (**Table 8-2**), but the test lacked statistical power to determine if the significant change was coincidental with scalping or occurred in January 2001, because of the small number of sampling episodes before scalping.

The average percent of invertebrates belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) was highest in samples from Har-S before scalping and values remained higher than reference sites immediately after scalping in April 2000 (**Figure 8-4**). Har-S had the highest April value of all sites (53%), and samples contained a high proportion of the mayfly *Ameletus* sp. (this was typical of samples from all sites in April 2000). In contrast, the % EPT at Har-S in May 2000 was lower than at all reference sites and asymmetrical ANOVA detected a significant impact (**Table 8-2**). The test lacked statistical power to confirm if the significant change was coincidental with scalping (due to the small number of samples collected pre-scalping), but graphical examination indicates that in all months after May 2000, the difference between reference sites and Har-S was relatively small (upper panel, **Figure 8-4**). The % EPT at Har-S in September 2001 was lower than before scalping in 1999 but fell within the range observed at reference sites. Winter samples collected in March 2000 had a similar % EPT at all sites, ranging between 4% at Har-R and 19% at Har-S. After scalping, the average % EPT at Har-S in March 2001 was higher than at most reference sites (15%) and comparable to samples collected in March 2000 (19%).

Table 8-2. Results of the asymmetrical ANOVA examining impacts of scalping on the benthic invertebrate community at Harrison Bar (flat bar edge units only).

Variable	**Do reference locations have variable short-term trends after scalping?**	**Does inclusion of scalped site affect short-term temporal trend?**	**Do reference locations vary in difference from before to after scalping?**	**Does scalping affect differences before or after?**	**Conclusion**	**p-value**	**Power**
Invertebrate Density	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001	-
% EPT	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001	-
Taxa Richness	Yes	No	-	-	No short-term impact detected	0.39	0.54
# EPT taxa	Yes	No	-	-	No short-term impact detected	0.21	0.67
Simpson's Diversity	Yes	No	-	-	No short-term impact detected	0.35	0.62
Simpson's Evenness	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001	-

Details of analysis in **Appendix K**.

Taxonomic richness followed a seasonal pattern similar to density, with the highest number of unique taxa in winter samples collected between November and March (**Figure 8-5**). Richness at Har-S was consistent between March 2000 and 2001, averaging approximately 13 taxa, and values were higher than the average at most reference sites (upper panel, **Figure 8-5**). Over three sampling episodes after scalping, Har-S had lower richness than reference sites but this difference was not statistically significant (**Table 8-2**) and no significant impact due to scalping was detected. Power to detect an impact was 0.54 compared with 0.23 when all habitat types were pooled for analysis. Comparing richness in summer months among years, Har-S was higher than reference sites in 1999, comparable after scalping in 2000, and lower than reference sites in 2001.

The number of EPT taxa in samples followed temporal patterns similar to that of taxonomic richness at all sites (**Figure 8-6**). EPT richness was highest in winter months and Har-S had higher values than the average of reference sites both before and after scalping. A notable reduction in the relative number of EPT at Har-S in May and August 2000 samples was not statistically significant and no impact due to scalping was detected (**Table 8-2**). The power to detect an impact was high (0.67), and was approximately double the power when all habitat types were pooled for analysis (0.34).

A reduction in Simpson's diversity between September 1999 and March 2000 was observed at all sites, but while all sites increased in diversity between March and April 2000, the increase was smallest at Har-S (**Figure 8-7**). Between May and November 2000, diversity at Har-S was higher than the average of reference sites (upper panel, **Figure 8-7**). Diversity was lowest at Har-S in January 2001, when densities of Chironomidae and Baetidae were high. No impact due to bar scalping was detected based on the analysis of diversity, and statistical power (0.62) was relatively high (**Table 8-2**).

Simpson's evenness was highly similar among reference sites and Har-S prior to scalping and immediately after scalping in April 2000 (**Figure 8-8**). Values were higher at Har-S in two subsequent months of sampling, and similar to reference sites in September and November 2000. The greatest difference in evenness between the scalped and reference sites was in January 2001, when Chironomidae and Baetidae were dominant in samples. The numerical dominance of the chironomid Orthocladiinae contributed to low evenness values in March of both years. Asymmetrical ANOVA revealed a short-term impact at Har-S (**Table 8-2**), but similar to the analysis of density, the test lacked statistical power to determine if the significant change was coincidental with scalping or occurred in January 2001, because of the small number of sampling episodes before scalping.

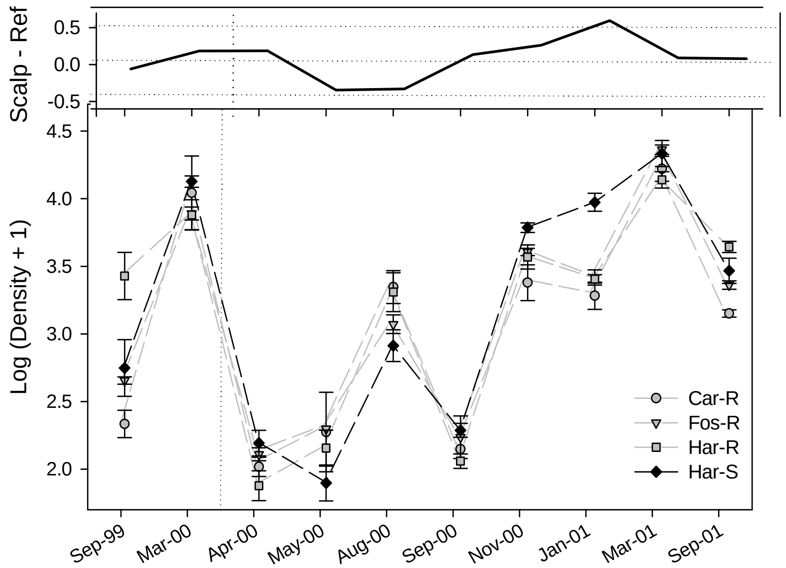


Figure 8-3. Average (± SE) density of benthic invertebrates collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

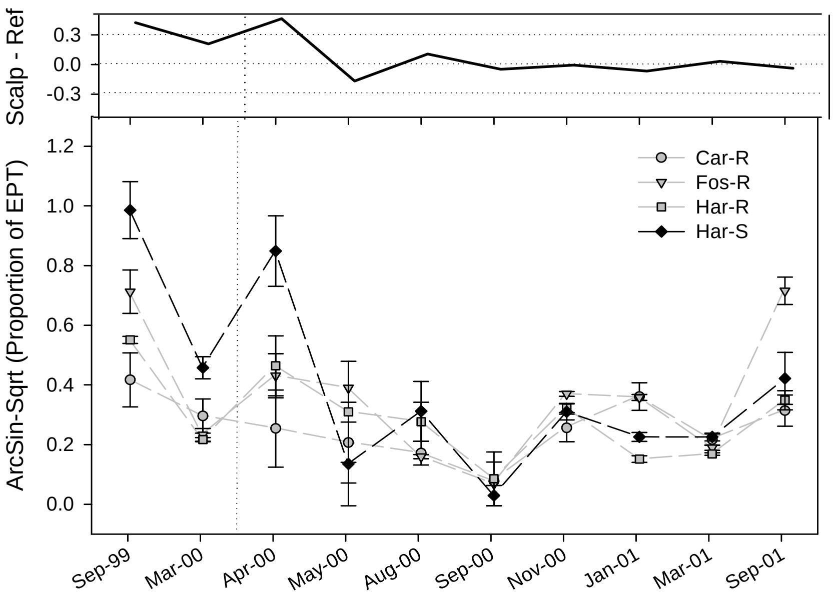


Figure 8-4. Average (± SE) proportion of Ephemeroptera, Plecoptera, and Trichoptera collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

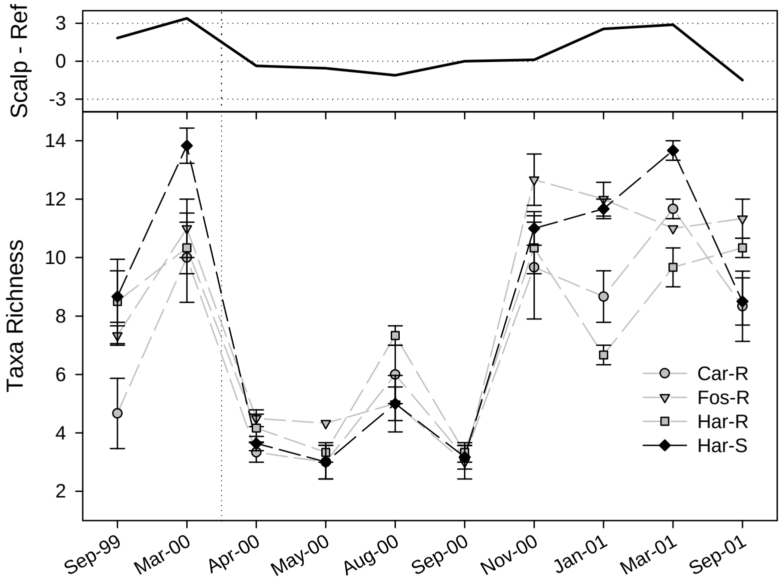


Figure 8-5. Average (± SE) taxon richness in samples collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

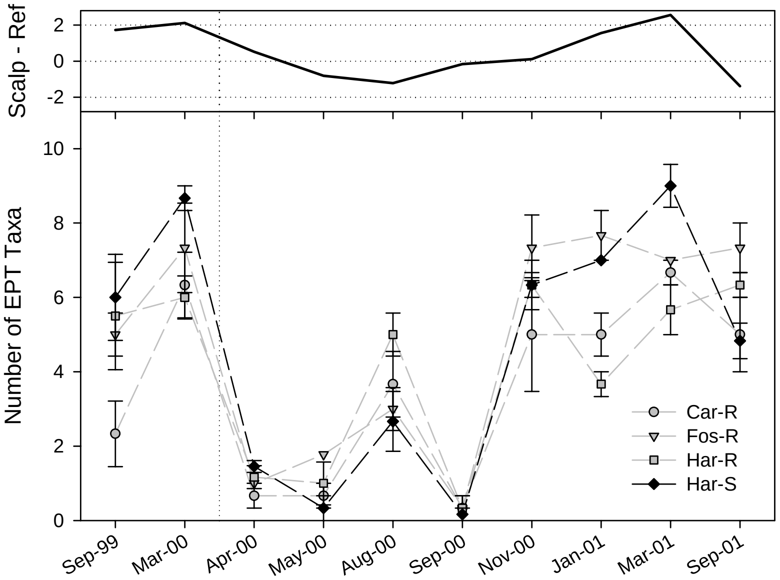


Figure 8-6. Average (± SE) number of taxa belonging to the Orders Ephemeroptera, Plecoptera, and Trichoptera in samples collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

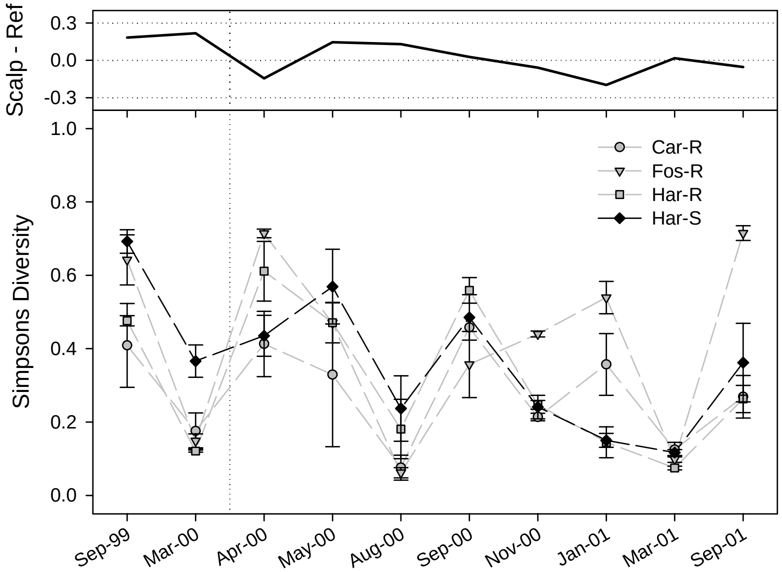


Figure 8-7. Average (± SE) Simpson's diversity of samples collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

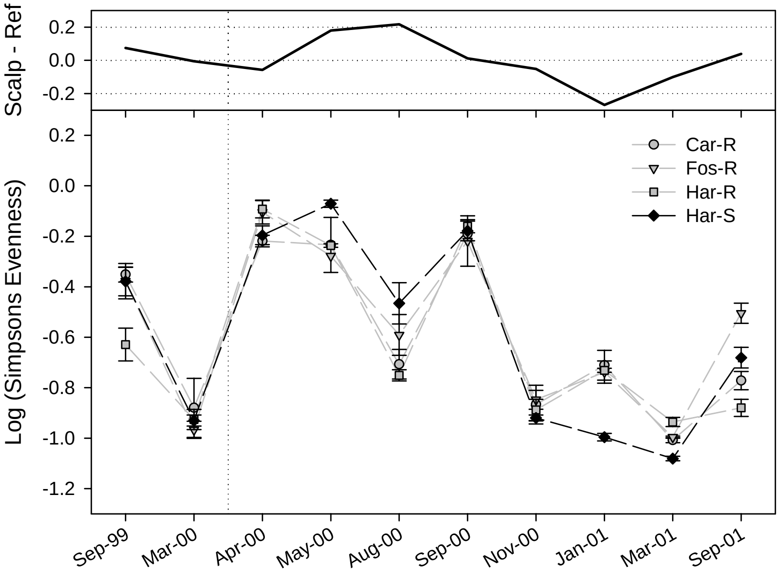


Figure 8-8. Average (± SE) Simpson's evenness of samples collected in flat bar edge habitat on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

8.5.1.2 Population-Level Examination

Seven taxon groups each represented greater than 1% of the total invertebrates collected. Of these, the mayfly Baetidae showed strong seasonal trends in abundance that reflected life history events. Abundance was highest in March of each year and lowest in spring after aquatic larvae had emerged (**Figure 8-9**). Abundance was also low in September 2000, but the decline between August and September 2000 was possibly the result of the sudden rise in discharge in early September that may have either flushed invertebrates from the near-shore zone of gravel bars or not allowed sufficient time for invertebrates to colonize the newly flooded area.

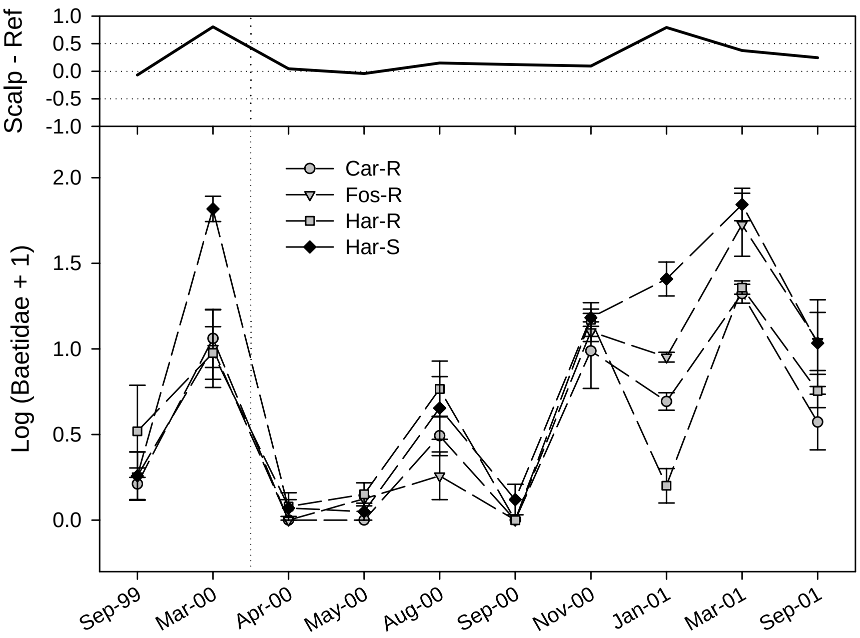


Figure 8-9. Average (± SE) number of Baetidae mayflies in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

In all months after scalping, Baetidae abundance at Har-S was approximately equal to or higher than the average at all reference sites. No notable change in abundance at Har-S was observed between March 2000 and 2001; however, Har-S had a substantially higher abundance of Baetidae than most other reference sites in 2001. Whereas abundance declined between November 2000 and March 2001 at all reference sites, it increased at Har-S. Asymmetrical ANOVA detected a significant change in Baetidae density at Har-S relative to the reference sites (**Table 8-3**), and examination of the graph suggests that the change occurred in January 2001, albeit an increase in abundance that occurred almost a year after scalping.

Table 8-3. Results of asymmetrical ANOVA examining impacts of scalping on the abundances of common benthic invertebrate families.

Variable	**Do reference locations have variable short-term trends after scalping?**	**Does inclusion of scalped site affect short-term temporal trend?**	**Do reference locations vary in difference from before to after scalping?**	**Does scalping affect differences before or after?**	**Conclusion**	**p-value**	**Power**
Baetidae	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	0.03	-
Chironomidae	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001	-
Capniidae	No	No	No	No	No impact detected	0.71	0.07
Heptageniidae	Yes	No	-	-	No short-term impact detected	0.29	0.79
Ephemerellidae	Yes	No	-	-	No short-term impact detected	0.98	0.15
Nematoda	Yes	No	-	-	No short-term impact detected	0.74	0.18
Oligochaeta	Yes	No	-	-	No short-term impact detected	0.40	0.81

Details of analysis in **Appendix I**.

Chironomids belonging to the sub-family Orthocladiinae were the most numerous taxon group in most samples on all dates, consistent with results in Chapter 5, and dramatic shifts in abundance reflected life history events (e.g., emergence between March and April 2000). Prior to scalping, abundance at Har-S was lower than average in September 1999 and higher than average in March 2000 (**Figure 8-10**). Har-S had lower than average abundance immediately after scalping and higher than average abundance in all months after August 2000, particularly January 2001. A significant short-term interaction between Har-S and the reference sites was detected (**Table 8-3**), but the test lacked statistical power to determine if the significant change was coincidental with scalping or occurred in January 2001.

The stonefly Capniidae was the only common taxon for which the temporal trend at reference sites was consistent after scalping and no short-term interaction between Har-S and the reference sites was found (**Table 8-3**). This allowed a more rigorous examination of an impact due to bar scalping at the larger time-scale of Before versus After (refer to **Figure 8-2**). No significant change in the abundance of Capniidae was detected from before to after bar scalping, although statistical power was very low (**Table 8-3**). Large seasonal differences in abundance were apparent and these differences were relatively consistent among sites (**Figure 8-11**). Whereas abundance at Har-S was lower than at all reference sites in September 1999 prior to scalping, it was higher than reference sites in September 2001. Har-S also maintained a higher than average number of Capniidae in March of both years.

The mayfly family Heptageniidae was mostly represented by *Rhithrogena* sp. and *Cinygmula* sp., and family abundance was variable at all sites among sampling dates (**Figure 8-12**). Whereas Har-S had higher relative abundance prior to scalping, Heptageniidae were less common at Har-S in spring and summer immediately after scalping. Abundance at all sites, however, was low during these months. No significant short-term impact at Har-S was detected with high statistical power (0.79, **Table 8-3**), and abundance was higher at Har-S than the average of reference sites in samples collected in November 2000 and all later sampling dates.

Abundances of Ephemerellidae (mostly *Ephemerella* sp.) were similar to those of Heptageniidae during the study. A comparison between September sampling events showed lowest abundance in 2000 and comparable values at most sites in 1999 and 2001 (**Figure 8-13**). Ephemerellidae abundance was highly variable among sites on all sampling dates in 2001. The group was absent from the majority of samples in April 2000 and no significant short-term impact at Har-S was detected (**Table 8-3**).

Nematodes were common at all sites in winter months, but virtually absent during summer (**Figure 8-14**). All sites had highly similar abundances among dates in the first six episodes of sampling, before and after scalping. However, there was variability among all sites during winter 2000/01. Abundances at Har-S were similar to that of reference sites on all dates throughout the study except January 2001, when abundance was lower than the average value of reference sites. No impact on nematode abundance due to scalping was detected (**Table 8-3**).

The abundance of Oligochaeta was highly variable among sites on most dates and no strong seasonal trend was apparent (**Figure 8-15**). Har-S had below average abundance in September 1999 but abundance was similar to reference sites in March 2000, prior to scalping. Abundance remained lower than at reference sites immediately after scalping, but was higher than at reference sites in the fall and winter of 2000. On no date after scalping did oligochaete abundance at Har-S fall outside the range of values at reference sites and power to detect an impact was high (**Table 8-3**).

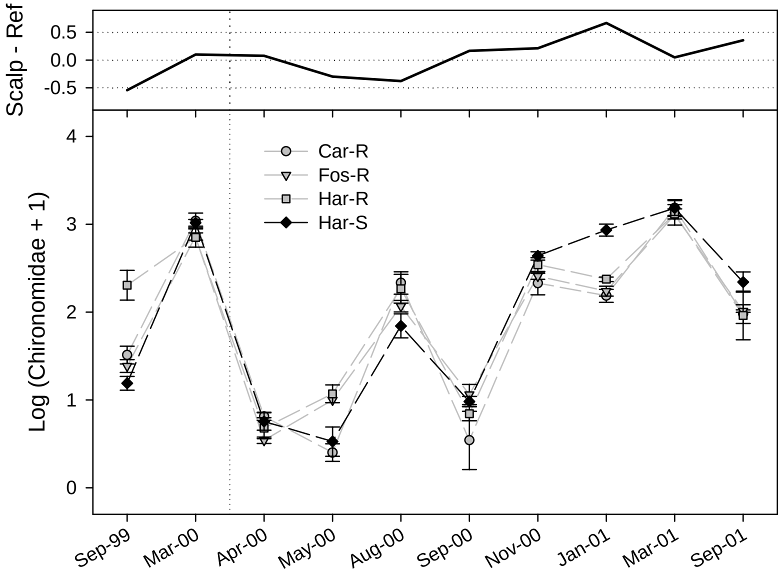


Figure 8-10. Average (± SE) number of Chironomidae midges in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

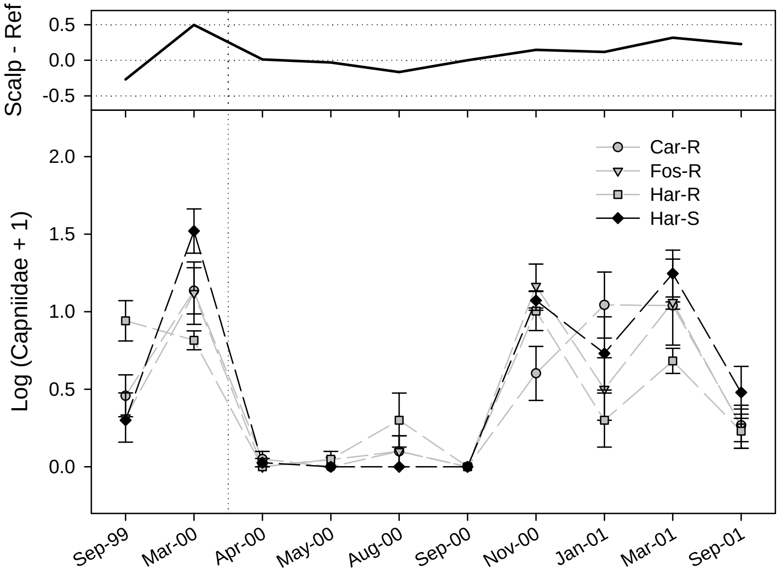


Figure 8-11. Average (± SE) number of Capniidae stoneflies in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

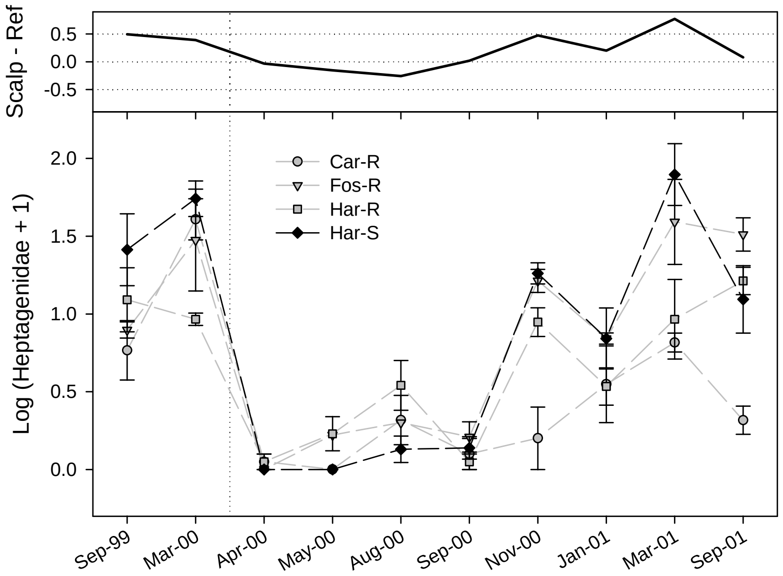


Figure 8-12. Average (± SE) number of Heptageniidae mayflies in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

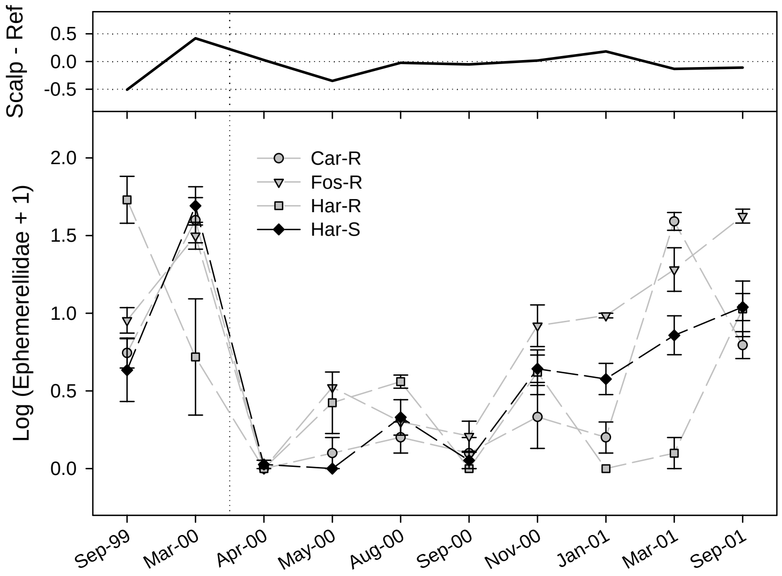


Figure 8-13. Average (± SE) number of Ephemerellidae mayflies in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

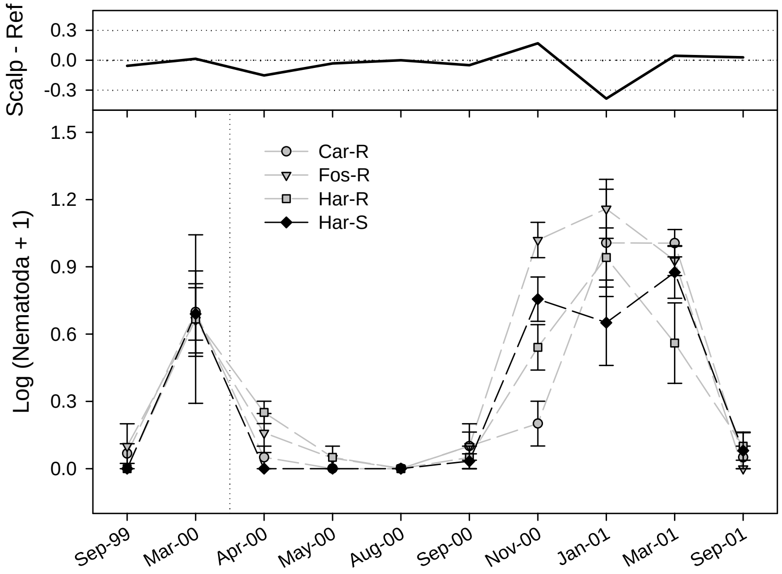


Figure 8-14. Average (± SE) number of nematodes in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

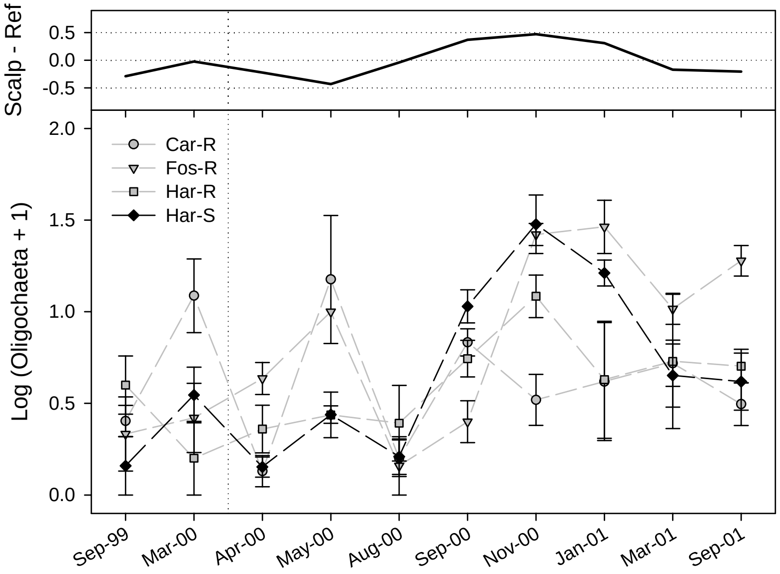


Figure 8-15. Average (± SE) number of oligochaetes in samples collected on 2 dates before and 8 dates after scalping at Har-S. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols reflect the uncertain temporal trend between dates. Upper panel values are the average of reference site values subtracted from the scalped site value on each date.

8.5.2 Fish

Asymmetrical ANOVA to examine scalping impacts on fish yielded similar results for analyses at the bar-scale and the scale of flat bar edge units. In contrast to invertebrate analyses, however, statistical power was higher for bar-scale analyses. Graphical data and statistical results from bar-scale analyses are presented below, and comment is made for the two cases when habitat-scale results differed. Complete ANOVA breakdowns are provided in **Appendix M** (whole bar), and **Appendix N** (flat bar edge units only).

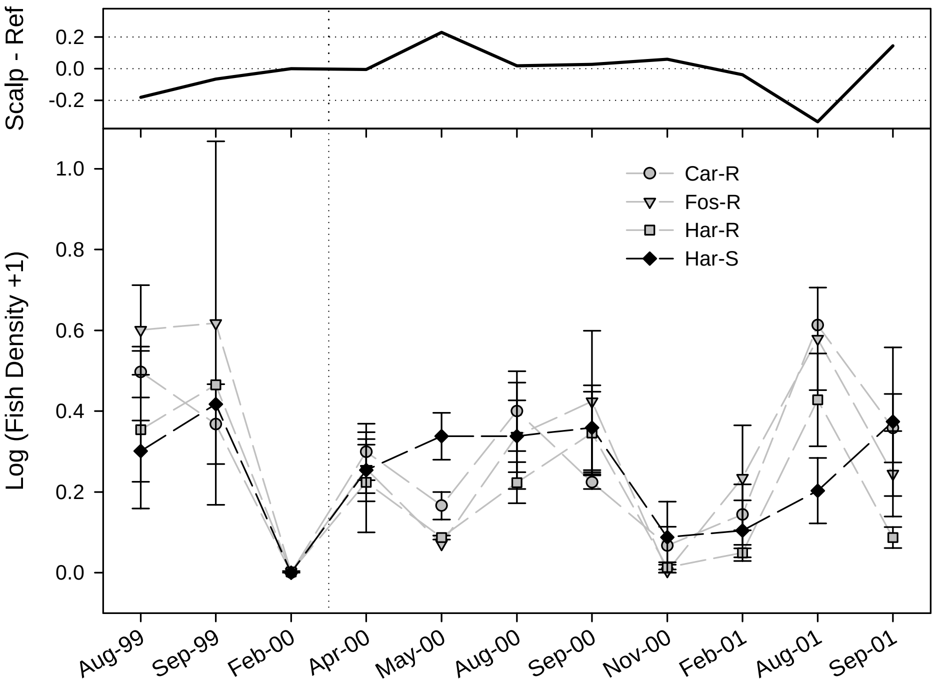
8.5.2.1 Bar-Scale Examination

A total of 12,094 fish were captured at Har-S and the reference sites by beach seine between August 1999 and September 2001. The number of beach seines varied among months (**Figure 8-16**) due to a variety of factors including sampling constraints and safety. Limited sampling took place during winter because few fish were encountered, and limited sampling also took place during peak freshet (May, June) when high velocities and water levels created dangerous conditions for sampling.

Fish density was variable among sites on most sampling dates throughout the study but sites followed a similar seasonal trend, with highest fish density between April and September and lowest density in winter months (**Figure 8-16**). In all months after scalping except February and August 2001, density at Har-S was equal to or greater than the average density at reference sites (upper panel **Figure 8-16**) and no significant impact due to scalping was detected (**Table 8-4**). The identical result was obtained by analysis of flat bar edge habitat. Fish density was most variable among sites in summer 1999 when unusually large numbers of redside shiner and leopard dace occupying open nook and bar edge habitats were collected at Fos-R. The increase in density from February to April 2000 was partly attributed to chum salmon fry migrating downstream. The stomachs of chum salmon were always full and consisted mostly of adult and larval chironomids, zooplankton, and mayflies (**Appendix G**). These observations indicated that chum fry were using habitat in the gravel reach for feeding and justified the species' inclusion in data tabulations.

The proportion of fish belonging to the family Salmonidae was highly variable among sites during all periods of sampling by beach seine. Har-S had consistently lower salmon representation than reference sites before scalping, but a higher than average value in May 2000 (**Figure 8-17**). Virtually all these fish were chinook salmon occupying bar edge and bay habitat within the former removal area. The proportion of salmon species was low at Har-S relative to all reference sites during summer and autumn sampling in 2000, but showed a sharp increase again in February 2001, when chum salmon and mountain whitefish were collected. After one year post-scalping in 2001, the proportion of salmon species at Har-S was similar to reference sites both in summer and winter. No impact due to scalping was detected (**Table 8-4**), although the power to detect an impact due to scalping was low. The analysis of flat bar edge units showed a significant change at Har-S ($p = 0.01$), but statistical power was insufficient to confirm that the change was coincident with scalping.

Species richness at Har-S was similar to that of reference sites in most months of sampling and averaged approximately 4 species per beach seine (**Figure 8-18**). After scalping, values at Har-S and reference sites were similar during all periods except September 2000 and August 2001, when richness was below average. Richness was extremely low in winter months in comparison with spring and summer, during which period values were relatively consistent among sites and months. It followed that power to detect an impact was moderate and no impact due to bar scalping was found (**Table 8-4**), consistent with the analysis of flat bar edge units.



6	2	7	12	4	15	9	2	5	5	3	Har-S
14	11	5	27	5	29	21	6	18	13	22	Reference Sites

Figure 8-16. Average (± SE) fish density (number per 10-m^2) in all habitats during 3 sampling periods before and 8 sampling periods after scalping at Harrison Bar. The number of beach seines in each month is indicated. Vertical dotted line denotes the timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of the reference site values on each date.

Table 8-4. Results of the asymmetrical ANOVA examining impacts of gravel mining on juvenile fish metrics at the bar scale (all habitats combined).

Variable	Do reference locations have variable short-term trends after scalping?	Does scalping affect short-term temporal trend?	Do reference locations vary in difference from before to after scalping?	Does scalping affect differences before or after?	Conclusion	p-value	Power
Total Density	No	No	No	No	No impact detected	0.37	0.18
Salmonid Index	No	No	No	No	No impact detected	0.27	0.08
Species Richness	No	No	No	No	No impact detected	0.68	0.37
Simpson's Diversity	No	No	No	No	No impact detected	0.49	0.26
Simpson's Evenness	No	Yes	-	-	*Short-term impact detected at Har-S but uncertain if coincident with scalping*	<0.001	-

Details of analysis in **Appendix M**.

Simpson's diversity showed a temporal pattern similar to species richness. Seasonal differences between winter and spring/summer were dramatic and were observed consistently at all sites (**Figure 8-19**). Diversity was lower at Har-S than reference sites in summer 1999 before scalping and remained lower immediately after scalping in April and May 2000. However, samples in all periods after August 2000 had similar diversity at the scalped site compared with reference sites. No impact due to bar scalping was detected by bar-scale analysis (**Table 8-4**) and examination of flat bar edge units.

Simpson's evenness did not show a strong seasonal trend, unlike most other metrics evaluated. Average evenness at Har-S was highly variable both before and after scalping in relation to the mean value observed at reference sites (upper panel, **Figure 8-20**). Evenness was higher at Har-S in August 1999, 2000, and 2001 compared with reference sites, but varied considerably in the intervening periods. A significant impact due to bar scalping was detected by asymmetrical ANOVA but the small number of sampling episodes before scalping gave the analysis inadequate power to confirm that the timing of change at Har-S was coincident with scalping (**Table 8-4**). In the analysis of flat bar edge units, no impact was detected ($p = 1.0$, power = 0.07).

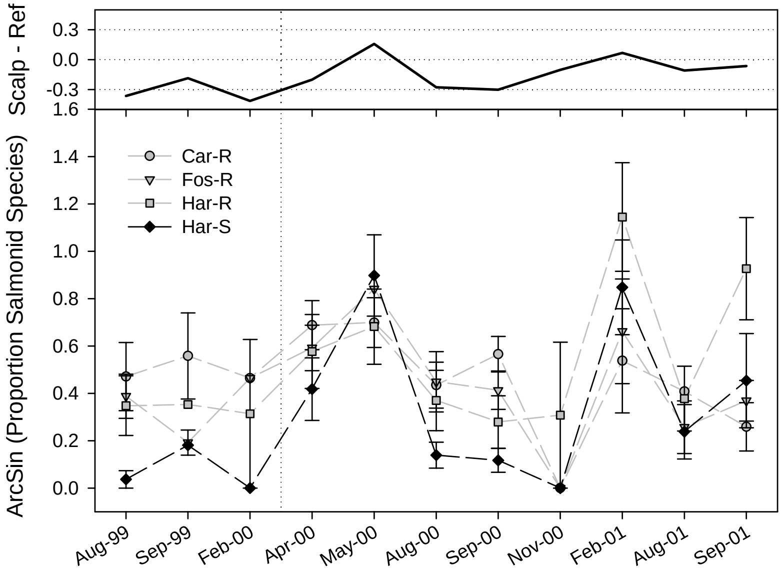


Figure 8-17. Average (± SE) proportion of salmonid species in all habitats during 3 sampling periods before and 8 sampling periods after scalping at Harrison Bar. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

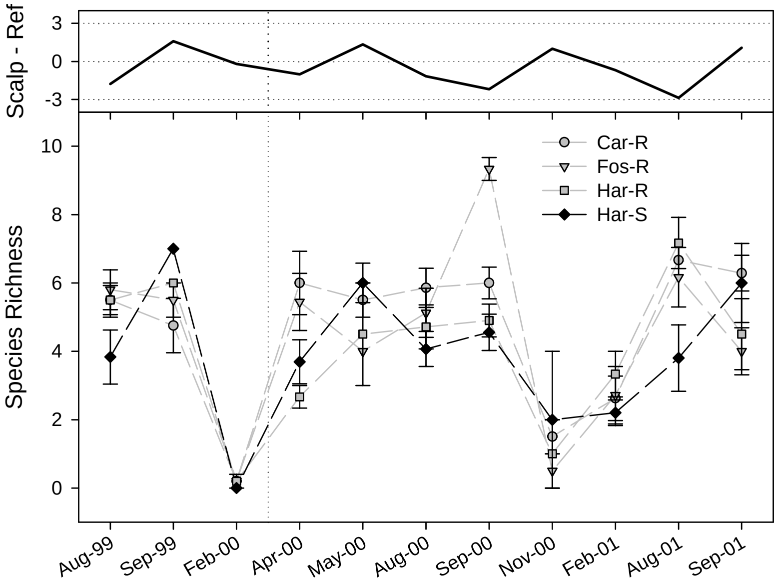


Figure 8-18. Average (± SE) number of unique species in all habitats during 3 sampling periods before and 8 sampling periods after scalping at Harrison Bar. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

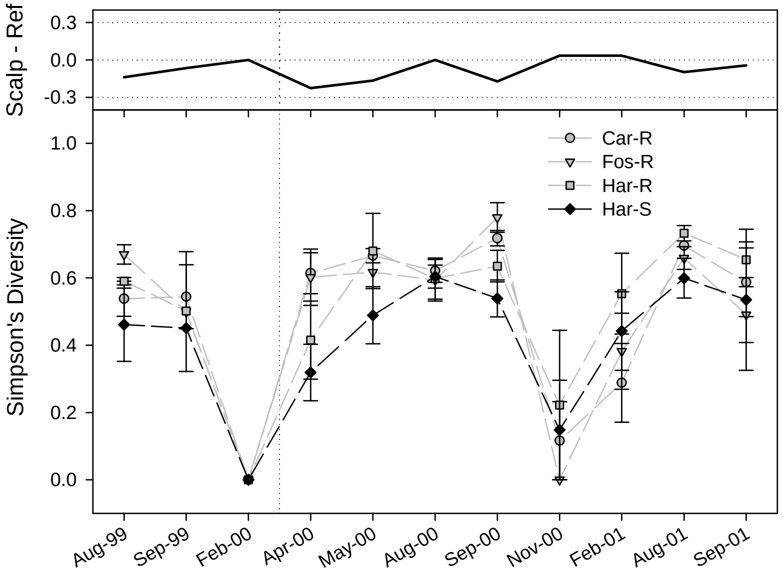


Figure 8-19. Average (± SE) Simpson's diversity in all habitats during 3 sampling periods before and 8 sampling periods after scalping at Harrison Bar. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

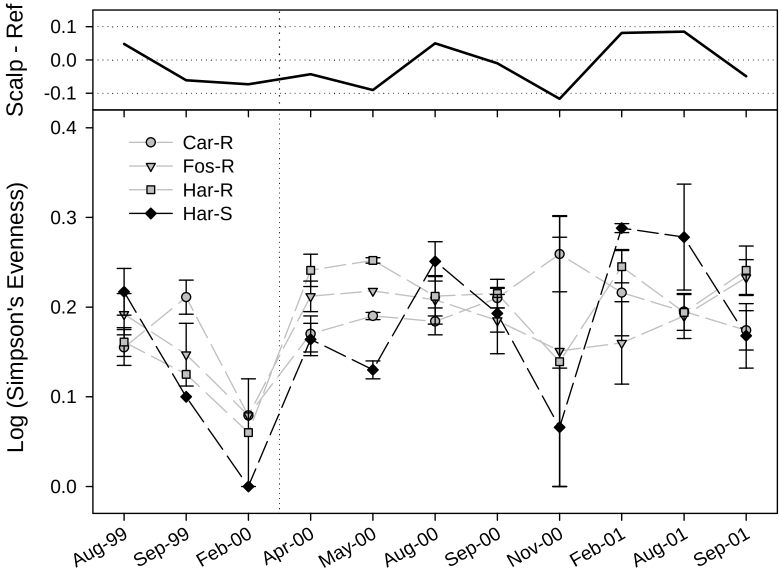


Figure 8-20. Average (± SE) Simpson's evenness in all habitats during 3 sampling periods before and 8 sampling periods after scalping at Harrison Bar. Vertical dotted line denotes timing of scalping. Hatched lines joining symbols connect successive sampling episodes at a site and should not be interpreted to represent a trend in the inter-sampling period. Upper panel values are the scalped site value minus the average of reference site values on each date.

8.6 Discussion

8.6.1 Response by Benthic Invertebrates

Benthic invertebrates recolonized lower Harrison Bar immediately after scalping as water inundated the site with the onset of freshet. Samples collected in April 2000 had above average density and included a high proportion of mayfly, stonefly, and caddisfly (EPT) nymphs. Several of these taxa (e.g., *Ameletus* sp.) are known to be highly mobile and have good swimming ability (Mackay 1992). These behavioural tactics are practical for survival in the gravel reach, where water levels shift over several hundred meters across the surface of gravel bars during freshet. The rapid colonization with the onset of flooding is consistent with the community-scale response to seasonal flooding documented previously in the gravel reach by Rempel *et al*. (1999), where invertebrates were shown to migrate laterally across gravel bars with the rise and fall of water levels. Just as *Ameletus* sp. was found almost exclusively in April 2000 samples in this study, it was collected in the study by Rempel *et al*. (1999) only in April.

In two subsequent sampling episodes after April 2000, invertebrate density at the scalped site was lower than reference sites. Taxon and EPT richness were lower at Har-S during this period as well. These samples from May and August 2000 were collected within the scalped boundary and it was during this period of high discharge that the bar surface underwent the most significant physical change in sediment texture. The rate of sediment transport across Harrison Bar was likely higher due to the loose substrate framework left by scalping, and these conditions may have deterred settlement by some taxa or crushed them. As documented in Chapter 7, surface sediment texture recovered after a single freshet to conditions similar to those before scalping. Just as sediment texture had recovered by September 2000, invertebrate samples collected from the scalped site in September 2000 and all months thereafter had higher density than the average of all reference sites.

The proportion of mayflies, stoneflies, and caddisflies was higher at Harrison Bar than reference sites prior to scalping, but the average value declined sharply between April and May 2000. The reduction was statistically significant and indicated a short-term impact due to scalping at Harrison Bar. However, the reduction was short-lived and the proportion of EPT at Har-S was higher than reference sites in August 2000, and higher or similar to reference sites in all months thereafter. Hence, the change due to scalping at Har-S was significant but short in duration.

Taxon richness, the number of EPT taxa, and species diversity each showed variable trends after scalping at Harrison Bar and no short-term impact was detected. Statistical power was relatively high for these analyses both at the bar-scale and within flat bar edge habitat units. The sensitivity of the analyses to detect an impact, however, was reduced because of the temporal variability observed at reference sites. Metrics for which a short-term impact was detected, in fact, had coherent patterns of variance relative to those metrics showing no detectable impact. This level of variance characterizes natural condition for Fraser River, and Underwood (1993) indicated that a significant temporal interaction among reference sites after scalping indicates that an impact would have to be large to be ecologically "important". Otherwise, it simply falls within the range naturally encountered by populations and is therefore within the community's capacity to recover. The fact that virtually all community parameters recorded values at Har-S were similar to or higher than those at reference sites after August 2000 is suggestive of the system's capacity to recover from a modest one-time removal of gravel from lower Harrison Bar.

The temporal response in abundance of common invertebrate taxa after scalping was variable and a significant short-lived impact due to gravel mining was detected in the abundances of 2 of the 7 taxa. The impact was positive for Baetidae and negative for Chironomidae. The response of Baetidae, being more abundant at the scalped site than the average of reference sites on all dates before and after scalping, may be reflective of the taxon's swimming abilities and tendency to colonize new surfaces (Mackay 1992). In contrast, chironomid midges have relatively poor swimming ability and the colonization of new sites may occur by drifting from upstream sources. Chironomids (mostly Orthocladiinae) were the most abundant taxon at all sites in the gravel reach, and showed a significant reduction in abundance immediately after scalping in May and August, although abundance was higher at the scalped site in all subsequent months of sampling relative to reference sites.

Recolonization of the scalped site by invertebrates depended on an abundant source population existing upstream. These animals would have arrived at Harrison Bar by a combination of passive and active drift as flooding inundated the site (Mackay 1992, Rader 1997). Drift sampling in the main channel of the gravel reach in September 2000 found high taxon richness (averaging 27 taxa in 12 samples) and moderate volumetric density (2.7 animals m^{-3}; L. Rempel, unpublished data). Drift samples consisted of both aquatic nymphs and terrestrial insects that would have dropped in from riparian areas. The amount that a single or multiple scalping operations may reduce the source population of invertebrates in the gravel reach is uncertain, but likely only a very large and widespread disturbance in the gravel reach might reduce the source population significantly.

8.6.2 Response By Fish

Fish density, along with several other metrics characterizing the fish community, showed no detectable impact at lower Harrison Bar as a result of scalping over the range of flows sampled (<5700 $m^3 s^{-1}$). The statistical analyses were relatively sensitive because reference sites varied in a consistent manner over time. Density was lower in the scalped area prior to scalping relative to reference sites, but was higher in 7 of 8 periods of sampling post-scalping. This pattern was consistent regardless of whether the spatial scale of examination was the lower bar (all habitat types pooled) or flat bar edge units only.

At the bar-scale, only Simpson's evenness showed a significant change at Har-S relative to the time-course pattern at reference sites. Values at Har-S were highly variable after scalping, in some months, such as August 2000, being higher than at all reference sites, and in other months, such as May and November 2000, being lower than at reference sites. The analysis of flat bar edge habitat units showed no significant change in evenness, but rather a significant change at Har-S in the proportion of fish belonging to the family Salmonidae. Average values before scalping were lower at the scalped site than all reference sites, but were higher at the scalped site in May 2000 and February 2001, after scalping. The higher representation in May 2000 consisted of chinook salmon, whereas chum salmon fry were significantly more common at the scalped site in February 2001.

8.6.3 High-Bar Habitat Availability For Fish

The potential impacts on fish of reduced high-elevation bar habitat and overall topographical changes are discussed here, although the sampling program did not comprehensively assess these effects. The lowering of bar surface elevation and reduction in high bar habitat due to scalping directly reduced the amount of shallow water habitat available at flows exceeding 4000 $m^3 s^{-1}$. Areas >8.5 m elevation before scalping would now have deeper water and higher velocity during summer months, which includes the period when fish are rearing in the gravel reach. Such conditions are not favourable for most species of juvenile fish and, consequently, *the availability of shallow, low velocity habitat over this range of flow was reduced by scalping*. Habitat value in the gravel reach naturally decreases in summer months as discharge increases (**Figure 8-21**), due to the reduction in perimeter length of islands and gravel bars over the period when abundances of juvenile fish are high. The additional reduction in high elevation bar area as a consequence of scalping is particularly significant because it would have further reduced habitat value at Harrison Bar during a period when it is already limiting.

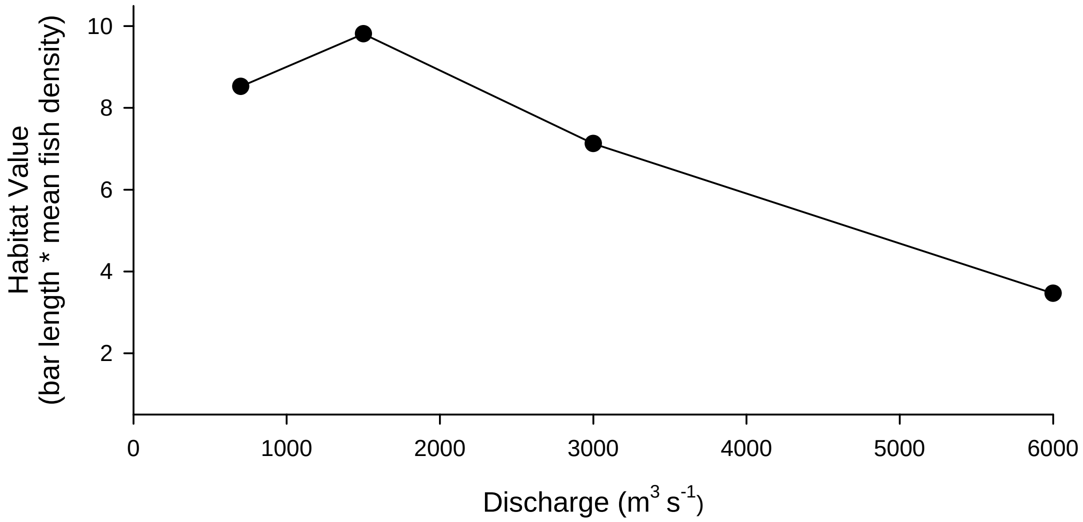


Figure 8-21. The relation between habitat value (averaged over all habitat types) and discharge, based on the average relation for three bars in the gravel reach.

This study did not specifically evaluate the importance of high-elevation bar top as habitat for fish, which remains an area for future study and is important to quantify for assessing the consequences of habitat loss (Minns 1997). Areas of high bar top and low lying-vegetation create flow resistance during flooding and may represent favourable fish habitat during freshet. In this context, the reduction in high-bar habitat by scalping would be particularly consequential to juvenile fish during flooding.

However, fish species may have alternative strategies in response to seasonal flooding in Fraser River that are worthy of further investigation. In tropical rivers, seasonal flooding elicits a lateral migration of fish from the main channel into off-channel habitats of reduced hydraulic stress and elevated productivity (Graaf 2003). Several contemporary studies of northern floodplain rivers have documented lateral shifts in distribution (Sommer *et al.* 2001, Hohausova *et al.* 2003, King *et al.* 2003), with side channels becoming increasingly important as fish habitat during predictable, high flow events. Flood predictability and duration are noted as important factors determining the importance of side channels as spawning and rearing habitat, and as refugia from flooding (Galat *et al.* 1998, King *et al.* 2003). In Fraser River, side channel fish density was higher than main channel density in early autumn (Chapter 6) and, compared with the main channel, side channels offer both reduced hydraulic stress and more extensive areas of overhanging vegetation for shelter and as a source of terrestrial insects. Gill net and minnow trap sampling in side channels consistently yielded high catches during freshet, but analogous data from main channel areas and from bar top habitat are not available for comparison.

At flows less than 3000 $m^3 s^{-1}$, habitat diversity was higher at Harrison Bar after scalping due to the greater topographic variability. Moreover, the availability of ecologically significant habitat types such as open nooks, which host highest densities of juvenile fish, increased. Prior to scalping, the bar surface was an expansive, flat area of simple topography. Flat bar edge remained the most common habitat type at low flow one year after scalping, however, units were shorter in length and spaced between open nooks. From the perspective of a fish, smaller and more diverse habitat units offer a wider choice in the range of habitat conditions available and support a greater number of species. This situation is energetically favourable because animals can disperse over short distances and select from a range of physical conditions and food sources (Rosenfeld and Boss 2001). It is also favourable for sustaining populations over the long term because reaches of river with high habitat diversity can support multiple life stages of animals with variable habitat requirements (Galat and Zweimüller 2001). In contrast, long-distance dispersal over extensive areas is necessary in a more simplified reach of river with few habitat types available (Nicolas and Pont 1997). Such a reach, ultimately, may lack the capacity to support some species of fish and high fish densities (Ward et al. 2002).

8.6.4 Statistical Power

Statistical power is a well-established concept (Zar 1984) that is being applied increasingly in studies in which the results have important implications for the management of fisheries or other natural resources (Peterman 1990). Power analysis is most appropriately carried out before initiating a study to determine the necessary sampling effort for a desired level of power, given knowledge of the system's natural variance. Similarly, it can be used to solve for the minimum detectable effect size when a feasible sample size and natural variance are known. These applications of power analysis are useful in the planning of studies to ensure that resources are allocated most appropriately for data collection. The estimates of variance yielded by this study may be useful in this regard for future studies in the gravel reach of Fraser River.

In the case of invertebrate sampling, power to detect an impact due to bar scalping was relatively high, and higher when flat bar edge units were isolated from riffles, bar heads, and bar tails. Isolating bar edge units appeared to reduce habitat-specific variance that was otherwise included in the error term of the analysis, thereby increasing power to detect a change. This is in contrast to the analysis of fish data in which power was greatest when all habitat types were pooled and sample size was large (242 versus 124 samples in flat bar edge units). Overall, however, the statistical power of fish analyses was low and indicated a large amount of variance in the data, both in space and time, relative to the amount of sampling effort. Natural variability inherent to fish distributions may reduce statistical power to detect significant change without very great sampling effort. Greater sampling replication (larger number of beach seines at a site during each sampling episode) and a greater number of sampling episodes *before* gravel mining are expected to yield improved estimates of the natural variance (Resh 1979) and thereby increase power to detect an impact.

The ability of a statistical test to detect an impact depends on several factors: the effect size (or magnitude of the impact), the natural variability of the data, and the number of independent sampling events (Osenberg *et al.* 1994). A severe impact is always more readily detectable, but the sensitivity of an analysis to detect an impact can be improved by increasing sample size, either in the number of replicates per sample (which increases precision of the estimated mean value), or the number of control sites, or the number of sampling episodes (Underwood 1993). Determining the most effective sampling strategy that returns the most discriminating results will depend on the temporal and spatial structure of variance in the data.

Large variability both in space and time characterized the data collected for this study and exploratory analyses were carried out to evaluate different sampling strategies and determine which returned the most discriminating results. The invertebrate metric taxon richness was chosen, and analyses consisted of, first, simulating an increased effect size in April 2000 immediately after scalping by systematically reducing taxon richness; secondly, adding a fourth reference site to the analysis (Calamity Bar was included in all sampling activities); and lastly, eliminating several sampling episodes after scalping. Collectively, these analyses suggested that when large-scale (i.e., bar to bar) spatial variability is great, the addition of another reference site does not improve resolution greatly. An additional reference site will improve power, however, when the effect size is very large. Also, the addition of one or two sampling episodes may not greatly increase resolution when there is high temporal variability. In such cases, it is more efficient to increase sample replication at each site in order to improve as much as possible the estimates of mean values, hence improve the ability of the analysis to discriminate among them. In this study, fewer post-scalp sampling periods and an increase in sample replication during each period may have improved overall statistical power. Most importantly, a greater number of sampling episodes prior to scalping would have increased resolution and improved estimates of the natural variance, thereby increasing statistical power to detect an impact. Of course, such insights can only be drawn once knowledge of the system's natural variance has been gained.

Chapter 9. Conclusions

9.1 Introduction

This thesis presents a hierarchical habitat classification developed specifically for the gravel reach of Fraser River, but intended for general use in wandering gravel-bed rivers, as a tool for scientific study and river management. The classification has a morphological basis and characterizes the physical organization of the gravel reach at three spatial scales, the finest of which is intended to specify a scale that is ecologically relevant to invertebrates and fish. Results demonstrate that the assemblages of invertebrates and fish associated with habitat types are moderately distinct and differentiated along a hydraulic gradient corresponding to velocity. Several fish species have specialized associations with one or a limited number of habitats, and fish community structure is particularly distinct within open nooks, channel nooks, and eddy pools. However, the congruence between habitat structure and the structure of aquatic communities is weakened by large spatial and temporal variability in the distribution and abundances of many species. This variability is attributed to the spatial heterogeneity that characterizes the gravel reach, the pronounced hydrological cycle that redistributes sediment and reconfigures habitat on an annual basis, the various life cycles of resident species, and the widespread distributions and flexible habitat associations of many species. These factors also are predicted to afford the invertebrate and fish community resilience to physical disturbance and, consistent with this expectation, gravel extraction at Harrison Bar elicited a modest ecological response from which the local community recovered within one freshet cycle.

9.2 Hierarchical Habitat Classification

The hierarchical habitat classification presented herein is proposed as a model of physical and ecological organization in the gravel reach, with application in strategic planning, operational management, biophysical inventory, and the design of scientific studies. It also is intended to serve as a general model for large gravel-bed rivers in northern temperate regions, although further work is required to confirm its applicability to other gravel-bed rivers with a wandering morphology. General observations from smaller coastal rivers in British Columbia such as the wandering reaches of Lillooett and Squamish Rivers reveal similar physical habitat structure to the gravel reach of Fraser River. Moreover, sedimentary features such as gravel sheets and sediment lobes, as well as their associated habitat types, are common in Tagliamento River, northeast Italy (Dr. P. Huggenberger, Geology, University of Basel, pers. comm.; van der Nat *et al.* 2003).

At the highest level of the classification, trends in channel gradient, river-bed sediment size, and sediment transport regime differentiate sub-reaches. Sedimentation patterns and gradational tendency lend each sub-reach a distinctive channel form and determine the physical complexity of each sub-reach. As well, these factors likely contribute to the significant dissimilarity in invertebrate and fish community structure among sub-reaches, which qualifies sub-reaches as units of ecological as well as physical significance in the hierarchical habitat classification. Sub-reaches are suitable units for strategic management planning because the physical character will remain unchanged for many decades as a consequence of the large volume of stored sediment that would require redistribution to effect morphological change.

Gravel bars nested within sub-reaches constitute the intermediate level of habitat classification, and represent nuclei of morphological complexity and habitat diversity in the gravel reach. Localized erosion and deposition during individual flood events modify gravel bars whereas vegetation over areas of highest elevation promotes bar stability and reflects the overall longevity of gravel bars in the river. The size of bars, and the fact that they are the largest organizational units within which the full range of local habitats may be found, makes them suitable for operational management and appropriate as planning units for scientific studies of river sedimentation and ecology.

Habitat units associated with gravel bars and river banks represent the finest level of the classification, and the spatial scale of primary focus in this study. Alluvial habitat units are classified with reasonable predictive accuracy based on sedimentary and hydraulic characteristics, and the distinctiveness of associated fish assemblages confirms that the spatial scale of habitat units is ecologically relevant. The moderate association between habitat structure and the distribution and abundances of fish species qualifies habitat types as key elements in biophysical inventory and assessment in the gravel reach, and as the appropriate units among which to stratify sampling effort for fisheries research.

The proposed alternative “hydraulic” model for habitat classification, which pools 7 morphological habitats into 3 hydraulic classes (exposed, normal, sheltered), has practical advantages due to its simplicity. However, the simplified hydraulic classification incorporates considerable physical heterogeneity with a major loss of information about the morphological complexity of the reach. This has disadvantages from the perspective of habitat assessment, but the classification is also ineffective because fish species discriminate among morphological habitat types that are pooled in the hydraulic classification. Hence, the morphological classification of habitat units is favoured for its physical and ecological significance.

Study results demonstrate that local-scale habitat characteristics differ at the larger scale of sub-reaches and channel types; however, the influence of large-scale processes on local-scale habitat structure remains an area for future research. A simple starting point would be a comparison of habitat availability and diversity among sub-reaches, because significant distinctions in channel gradient, sediment texture, and sediment depositional tendency are apparent. Are there more slower gradient and sandy habitats in the Chilliwack sub-reach, and more riffles and steep bar edges in the Cheam sub-reach? Is habitat unit size and density (number of habitat units per unit bar area) related to the gradational state of the sub-reach, with highly aggrading sub-reaches having smaller and more diverse habitat units than a degrading sub-reach? Such possible differences in habitat distribution and frequency between sub-reaches would have implications for the assemblages and relative abundances of species that each sub-reach may support. The comparison of habitat availability between sub-reaches should include several water levels to determine the relative abundance of habitat types at varying stage because any change in abundance would affect the relative use of other habitats by organisms. These recommended analyses may be carried out conveniently using aerial and oblique photographs because habitat units can be identified from photographs with reasonable accuracy (Rempel and Church 2002).

9.2.1 Proposed Revisions to the Classification

Overall, the hierarchical habitat classification successfully characterized the physical organization in the gravel reach; however, several classes had consistently low predictive accuracy. The habitat type with lowest predictive accuracy was bar tail, which also was found to have the most variable morphological expression during field surveys. The poor physical discrimination of bar tails was not surprising because of the geographical basis and stage dependency of unit definition. At a larger scale, habitat units in secondary channels had low predictive accuracy, possibly because different sedimentation processes and a shorter annual inundation period lend secondary channels slightly different habitat attributes (though not a distinct suite of habitat types).

Misclassifications of habitat types are attributed to the overall variability in physical characteristics of the habitats themselves, as opposed to misidentification during field sampling. This physical variability likely translated to variability in the associated invertebrate and fish assemblages. Hence, final refinement of the habitat classification is proposed in order to maximize between-group distinctiveness and predictability (**Figure 9-1**). These proposed refinements arise from results in preceding chapters, as well as observations made over three years of field sampling. Further study is required to evaluate the merits of the proposed revisions.

Refinement to the definition of bar tails is intended to reflect the stage-dependency of bar tail attributes. The revised definition identifies “morphological” bar tail units corresponding to an extended riffle or depositional gravel lobe, generally only visible at low and moderate discharge, and consisting of well-sorted gravel sediment, moderate velocity, and a flat bank angle. “Geographical” bar tail units occurring at higher flows and defined by position relative to the exposed gravel bar in actuality represent mid-bar habitat because the “morphological” bar tail is submerged. According to the refined classification, such units would be reclassified as either flat bar edge or steep bar edge, depending on bank angle.

At the bar-scale, secondary channels may be sub-divided according to stability and inundation period, and an additional channel class added to represent mature, narrow channels with heavily vegetated banks. Side channels, all of which are considered “stable”, are split according to “perennial” (e.g., Minto Island, Spring Bar, Gill Island) and “seasonal” flow (e.g., Carey Bar, Foster Bar), in recognition that the duration of flow conveyance may influence the associated fish and invertebrate community. Summer channels, all of which have seasonal flow conveyance, are split according to “stable” and “young” morphology. Examples of stable summer channels are those dissecting Queens Bar, Harrison Bar, and Herrling Island. Young summer channels typically dissect bar top habitat and have poorly defined bank lines. Additionally, a fourth class of channels, “anabranches”, defines slough-like channels that are highly stable and dissect large islands and young floodplain surfaces (e.g., Maria Slough, Greyell Slough, Herrling Island back-channel, and several channels dissecting Minto Island). The term “anabranch” refers to narrow channels associated with large, stable islands, which persist for decades or centuries and support well-established vegetation (Knighton 1998). Anabranches represent the class of channel that has suffered the greatest loss this past century due to dyking (Ellis *et al.* 2004).

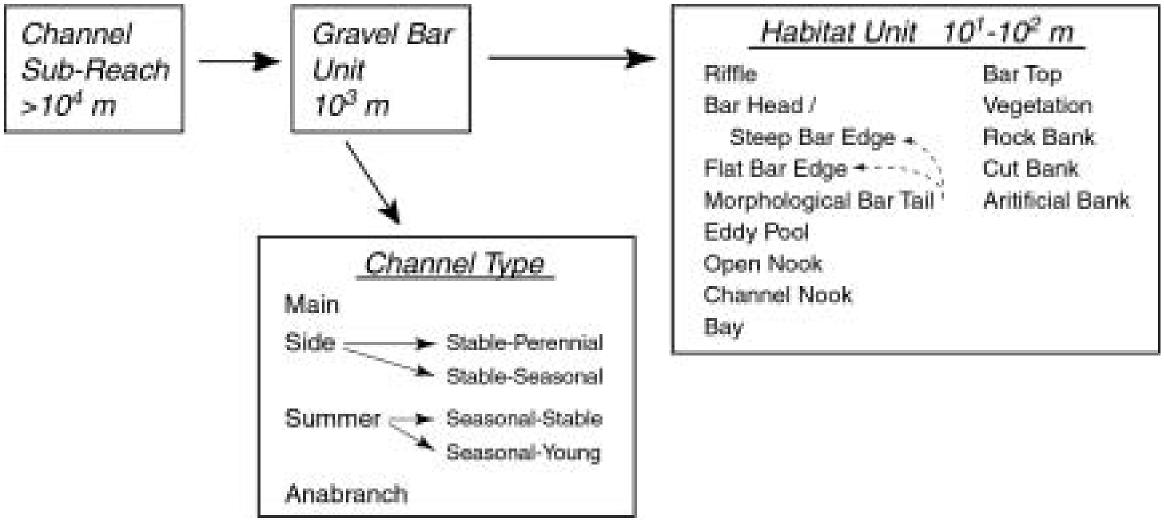


Figure 9-1. Revised hierarchical habitat classification for the gravel reach of Fraser River. Refer to text for details.

9.2.2 Ecological Significance of Side Channels

A wandering morphology, with multiple channels separated by gravel bars and large islands, describes the gravel reach of Fraser River. Wandering rivers are characterized by an extensive network of perennial, seasonal, and anabranch channels, reflecting the lateral instability of the system and sometimes indicating locations where the main channel once flowed. Whereas the network of anabranch channels in Fraser River has been mostly cut-off from the active channel zone for more than a century, the amount of perennial and seasonal side channel habitat remains relatively unchanged over the period of photographic record (Ellis *et al.* 2004).

Side channels host high species diversity and densities of fish during summer and autumn, consistent with other studies in northern temperate rivers that have reported higher densities, enhanced growth, and recruitment within side channels and floodplain habitats (Sommer *et al.* 2001, Grift *et al.* 2003). Moreover, side channels in the gravel reach are the primary spawning habitat for endangered white sturgeon, which spawn between June and August during freshet (Perrin *et al.* 2003a). Compared to the main channel, side channels have greater connectivity with the riparian zone, which represents a source of drop-in terrestrial insects, shelter from overhanging vegetation, and increased habitat complexity due to large woody debris. Perhaps most importantly, flow conveyance in side channels is reliable throughout much of the growing season for fish, yet flow forces are moderated compared to the main channel even during flooding (Ellis 2004).

The collective evidence of seasonally high fish densities and diversity in side channels, the primary use by endangered white sturgeon for spawning in the gravel reach, and the potential function of side channels as a flow refuge and for enhanced fish recruitment, suggests that the loss of side channels would have severe ecological consequences. It is therefore proposed that side channels be considered a "keystone habitat" in the gravel reach. The term "keystone predator" was introduced by Paine (1969) based on research in the rocky intertidal zone in reference to a species whose density would control the densities of other predators and consumers. The term is now used more broadly in reference to a species whose removal is expected to result in a dramatic change to the ecosystem. In a habitat-context, side channels likely serve a keystone function in the gravel reach, in the sense that a reduction in side channel habitat would result in significant ecosystem change. Ellis (2004) provided a detailed characterization of the hydraulic geometry and sedimentary characteristics of several side channels in the Rosedale and Chilliwack sub-reaches. Rich opportunities exist for further research to characterize the ecological significance of side channels.

9.3 The Habitat Template of the Gravel Reach

The gravel reach of Fraser River is characterized by substantial spatial heterogeneity both at a local scale and at a reach scale in the lateral and longitudinal dimensions. This heterogeneity is coupled with temporal variability in the distribution and availability of habitat units with fluctuating water levels and in the persistence of habitat units over repeated freshet events. The pronounced temporal pattern of habitat availability and persistence, relative to the specific life cycle requirements of resident populations, are proposed as constituting a physical habitat template (cf. Southwood 1977) that constrains the types of "ecological strategies" that are successful over the long-term. Ecological strategies may be physiological, behavioural, and/or related to life history characteristics, and the habitat template acts as a bottleneck or "filter", selecting a subset of the regionally available species that possess strategies favourable to the prevailing habitat conditions (Poff 1997).

For environments characterized by spatial heterogeneity and temporal variability such as the gravel reach, favoured species are those with flexible habitat associations, opportunistic life history characteristics, and good dispersal ability (Poff and Ward 1990). The invertebrate and fish community in the gravel reach appears to possess these traits and, in this context, the moderate congruence between physical habitat structure and the structure of the aquatic community is explained. The habitat template for the gravel reach acts as a filter to select species with flexible habitat associations and strong dispersal ability because the availability and distribution of habitat units change over the hydrograph, and also change on an annual basis in response to localized erosion and deposition during freshet. Species with highly specialized habitat requirements, or weak dispersal ability, may not persist under the filtering effect of the habitat template that characterizes the gravel reach. However, habitat complexity at multiple spatial scales likely enables a relatively broad species assemblage to persist (Lancaster and Belyea 1997). A practical implication for environments characterized by large spatial and temporal variability is that habitat classification may only be moderately useful as a predictive tool for biophysical inventory and fisheries research.

Just as the habitat template for the gravel reach favours species with flexible habitat associations and strong dispersal ability, it also is credited with building ecological resilience to physical disturbance. (Here, physical disturbance is defined, according to Sousa (1984), as an event or regime that demonstrably affects the fitness (e.g., by damaging or killing) of individuals, or populations, or communities.) Hence, an inverse relation is predicted between the strength of species' habitat associations and resilience to disturbance. The gravel reach is subjected to a relatively continuous and moderate level of natural disturbance, with approximately 200,000 $m^3 yr^{-1}$ of material recruited to the gravel reach downstream of Agassiz during freshet, and an additional 1 million $m^3 yr^{-1}$ of material redistributed locally (Ham and Church 2003). Given the natural disturbance regime, coupled with high spatial heterogeneity, a successful strategy for resident populations includes being widely distributed with relatively low habitat specialization. Physical disturbance that does not exceed the built-in resilience of the community is not expected to produce a significant or prolonged response (Poff and Ward 1990, Underwood 1993). Accordingly, disturbance and physical habitat change resulting from bar scalping at Harrison Bar apparently did not exceed the tolerance threshold for those species studied; hence, the conclusion of a relatively short-lived and modest ecological response.

The timing of bar scalping is believed to have contributed to the modest response because the period of most significant habitat change coincided with the period of greatest natural disruption to which resident populations are accustomed. The physical effects of intensive suction and dragline extraction, documented by Church and Weatherly (1998) in Minto Channel, are notably different. The ecological response to physical disturbance occurring during a period offset from freshet remains uncertain. Moreover, the critical question of what types of intervention (i.e., river training measures) or disturbance regime (i.e., timing, intensity, frequency) may exceed the built-in resistance of the community and therefore not be tolerated by resident populations, remains unanswered.

The ecosystem of the gravel reach is under increasing risk from a variety of pressures, including fishing, urban encroachment, and channel training by bank revetment, as well as gravel mining. The cumulative effects of these pressures are very difficult to estimate and the resilience of the ecosystem likely has a disturbance threshold beyond which it can no longer recover. This threshold is unknown and the point at which the threshold is exceeded may not be readily detectable. Given the influence of habitat structure on aquatic communities, geomorphological indicators may be an effective way by which to monitor the state of the ecosystem. Summary metrics such as active channel width, depth, and sinuosity are useful indicators of channel form. Also, parameters characterizing the variance in select environmental parameters (i.e., using variance as a dependent variable in analysis) may be particularly sensitive indicators for ecosystem stress (Palmer *et al.* 1997), given that spatial and temporal variability represent the habitat template governing community structure in the gravel reach.

The spatial heterogeneity that characterizes the gravel reach depends upon intact fluvial processes of sediment transport and lateral instability. Sedimentation features associated with processes of gravel bar development create habitat units of differing physical character and a range of channel sizes that together provide varied combinations of velocity, depth, and substrate to support the productive ecosystem. The sequence of sediment erosion and deposition around bars also maintains fish habitat of high quality and supports benthic invertebrate production because the sediment is episodically reworked and cleaned. The seasonal recruitment of gravel and cobble sediment from upstream, and local redistribution and lateral exchange within the gravel reach, also produce topographically complex habitat features that support a range of fish species and life stages. Preserving habitat complexity therefore depends on at least modest sedimentation and lateral instability within the reach, which appear incompatible with current proposals for increased bank revetment to counter bank erosion and increased gravel extraction for flood management.

9.4 Spatial and Temporal Variance Structure

While affording aquatic communities resilience to physical disturbance, an environment characterized by large spatial and temporal variability has practical implications for scientific study. Simply put, large and spatially heterogeneous environments are costly to study because substantial sampling effort is necessary for even moderately precise characterization and metric estimates. Large river research therefore depends on generous funding relative to comparable studies in smaller systems, and typically will be contingent upon endorsement at a political level with the research contributing directly to management and policy issues. Such was the case for this study on Fraser

River. This situation is not unfavourable, but implies that current management issues and the political mandate are likely to direct scientific inquiry in large river research. That is, research on large systems such as Fraser River is apt chiefly to be "mandated research" (research for specific applied purposes).

The most effective sampling strategy for large river research (i.e., the strategy that returns the most discriminating results for a given level of effort) depends on the temporal and spatial structure of variance in the data. Invertebrate and fish data were characterized by a large amount of variance, both in space and time, and in this situation the greatest gain in metric precision will come from increasing sample replication (Downing 1979). Exploratory analyses summarized in Chapter 8, **Section 8.6.4** support this recommendation, demonstrating that when spatial or temporal variability are great, increasing the number of specific sample locations or sampling episodes may not improve precision as much as increased sample replication.

The temporal variability characterized in this study highlights the need to specify an appropriate sampling schedule for scientific research or biomonitoring programs because of the sensitivity of invertebrate and fish abundance to both time of year and antecedent flow conditions. The timing of sampling events should be dictated by the purpose and goals of the study. For the purpose of monitoring a planned intervention such as gravel extraction, the sampling schedule must allow sufficient time in advance of the treatment to assure a reasonably symmetric temporal design and adequate characterization of pre-intervention spatial and temporal variance.

Winter is recommended as the season most suited to invertebrate sampling for the purpose of characterizing community structure and taxonomic inventory. However, fluctuations in flow during this relatively stable hydrologic period have the potential to influence community structure and the overall numerical abundance of organisms. The timing of sampling events also influences the appropriate gear for data collection, as the effectiveness of most equipment for biological and environmental sampling is limited by flow conditions.

Overall, future studies attempting to discriminate among conditions in the gravel reach require strategic planning due to the large natural variability that characterizes the system. A successful study requires a design that discriminates spatial and temporal effects, particularly seasonal patterns, with sufficient sample replication as dictated by the observed variability, and with deliberate selection and targeting of indicator metrics that are expected to be most sensitive or informative to meet the study goals. A clearly defined conceptual model, within which a framework of testable hypotheses is generated, also is a necessary precursor to successful research in large and variable systems.

Indicator metrics providing the best discrimination and sensitivity in this study were total density, evenness (Simpson or Pielou), richness (Margalef or unweighted), diversity (Simpson or Shannon-Wiener), and the proportion of specific species groups (salmonids for fish, and Ephemeroptera, Plecoptera, Trichoptera for invertebrates). Examining a combination of metrics was worthwhile in order to provide assurance of a thorough characterization of the community, and possibly identify sampling inconsistencies.

9.5 Recommendations For Gravel Management

This study initially was structured for examination of gravel mining impacts in order to contribute to the gravel management plan for Fraser River (Fraser Basin Council 2002). Although the scope of research expanded significantly beyond this single purpose, as evident in this thesis, the original research goal remained a priority (2003). Management recommendations for the planning of future gravel removals in Fraser River are summarized here.

1. Complex bar topography is essential for making available a variety of habitats for fish at all water levels on the rising and declining limb of the flood hydrograph. Strictly targeting areas of highest elevation to maximize removal depth is not recommended because these areas provide shallow zones of reduced hydraulic stress at high discharge and they contribute to island development and bar stabilization (Ham and Church 2002). *Site selection and planning for future removals should give due consideration to the availability of high bar habitat as well as local patterns of sediment transport and the likelihood of gravel replenishment to the site.*
2. The observed relatively short-lived ecological response to bar scalping is not surprising given that approximately 200,000 m^3 yr^{-1} of material is recruited to the gravel reach downstream of Agassiz, and an additional 1 million m^3 yr^{-1} of material is redistributed locally (Ham and Church 2003). Within the vicinity of Harrison Bar, gravel deposition in the past 15 years has averaged between 65,000 m^3 yr^{-1} (based on gravel transport estimates; **Figure 19** of Church *et al*. 2001) and 120,800 m^3 yr^{-1} (based on volumetric calculations for gravel only; **Table 6** of Ham and Church 2003). Relative to these reported volumes and in comparison to prior removals at nearby sites (Weatherly and Church 1999), the removal of 70,000 m^3 from lower Harrison Bar was relatively modest. *In general, the removal volume at a site should not exceed the best estimate of local gravel deposition in a year of mean annual flood discharge (derived from volumetric or sediment*

transport estimates). This is to ensure that physical changes to a site fall within the range of change that might be observed at a non-removal site in a large freshet.

3. Very few gravel bars have escaped gravel removal and the value of these sites for future monitoring studies will increase as the pressure for bar scalping continues to grow. *Preserving multiple undisturbed bars as reference sites is absolutely necessary if the impacts of future removals are to be investigated following statistically rigorous methods.*

4. There is potential for this study to be used in conjunction with future removal studies as a time block in a repeated measures analytical framework. Each removal study would serve as an independent replicate for the analysis, allowing conclusions to be generalized beyond the site-scale to a larger area of the gravel reach. Such a design is more rigorous than asymmetrical ANOVA because the treatment effect is replicated. Measures of variances estimated in this study will be useful in the early planning stages of future studies to determine the necessary sampling effort for a desired level of statistical power. *Further studies are necessary to determine if the patterns of physical and ecological response at Harrison Bar are characteristic of removals at other sites.*

5. The statistical rigor to detect impacts due to bar scalping was limited by the short time period between permit approval and the start of extraction activities. It was only fortuitous that sampling had been carried out at Harrison Bar and the three reference sites in August and September 1999, prior to bar scalping, as part of this thesis research. Typically, candidate sites for gravel extraction are identified only months or weeks before scalping is to begin. Time required for review by regulatory agencies is often several weeks and, together, these factors result in an inadequate period for pre-removal data collection. *The planning and approval processes must be modified to allow for a sufficient number of pre-removal sampling episodes if the desired outcome of monitoring is a rigorous statistical analysis of gravel mining impacts.*

References

Aadland, L.P. 1993. Stream habitat types: their fish assemblages and relationship to flow. *North American Journal of Fisheries Management* **13**:790-806.

Abell, R. 2002. Conservation biology for the biodiversity crisis: a freshwater follow-up. *Conservation Biology* **16**:1435-1437.

Ashworth, P.J. and R.I. Ferguson. 1989. Size-selective entrainment of bed load in gravel bed streams. *Water Resources Research* **25**:627-634.

Bailey, R.C., M.G. Kennedy, M.Z. Dervish, and R.M. Taylor. 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.

Bain, M.B., T.C. Hughes, and K.K. Arend. 1999. Trends in methods for assessing freshwater habitats. *Fisheries* **24**:16-21.

Bayley, P.B. 1991. The flood pulse advantage and the restoration of river-floodplain systems. *Regulated Rivers: Research & Management* **6**:75-86.

Bayley, P.B. and D.C. Dowling. 1993. The effect of habitat in biasing fish abundance and species rishness estimates when using various sampling methods in streams. *Polskie Archiwum Hydrobiologii* **40**:5-14.

Beecher, H.A., J.P. Carleton, and T.H. Johnson. 1995. Utility of depth and velocity preferences for predicting steelhead parr distribution at different flows. *Transactions of the American Fisheries Society* **124**:935-938.

Beechie, T.J., E. Beamer, and L. Wasserman. 1994. Estimating coho salmon rearing habitat and smolt production losses in a large river basin, and implications for habitat restoration. *North American Journal of Fisheries Management* **14**:797-811.

Benedict, A.H., K.J. Hall, and F.A. Koch. 1973. A preliminary water quality survey of the Lower Fraser River system. *Report No*. 2. Westwater Research Centre, Vancouver, British Columbia.

Beniston, R.J., D.B. Lister, and G.J. Naito. 1986. Juvenile salmonid rearing studies, Fraser River, 1984-1986, CNR Yale subdvision mile 40.1-94.5. Report prepared for CN Rail by D.B. Lister & Associates Ltd., Clearbrook, British Columbia.

Benke, A.C. 1990. A perspective on America's vanishing streams. *Journal of the North American Benthological Society* **9**:77-88.

Benke, A.C. 2001. Importance of flood regime to invertebrate habitat in an unregulated river-floodplain ecosystem. *Journal of the North American Benthological Society* **20**:225-240.

Berg, L. and T.G. Northcote. 1985. Changes in territorial, gill-flaring, and feeding behaviour in juvenile coho salmon (*Oncorhynchus kisutch*) following short-term pulses of suspended sediment. *Canadian Journal of Fisheries and Aquatic Sciences* **42**:1410-1417.

Birtwell, I.K. 1999. The effects of sediment on fish and their habitat. *Report No*. 99/139. Canadian Stock Assessment Secretariat Research Document.

Bisson, P.A., J.L. Nielsen, R.A. Palmason, and L.E. Grove. 1981. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. *In* Acquisition and utilization of aquatic habitat inventory information. *Edited by* N.B. Armantrout. American Fisheries Society, Western Division, Portland, OR. pp. 62-73.

Bland, R.G. and Jacques, H.E. 1978. How to Know the Insects. Wm. C. Brown Co., Iowa.

Blinn, D.W., J.P. Shannon, L.E. Stevens, and J.P. Carder. 1995. Consequences of fluctuating discharge for lotic communities. *Journal of the North American Benthological Society* **14**:233-248.

Boniface, C. 1985. Vegetation succession on mid-channel bars of the Fraser River, British Columbia. M.Sc. thesis, Department of Geography, Simon Fraser University.

Borror, D.J. and Delong, D.M. 1964. An Introduction to the Study of Insects. Holt, Rinehart & Winston Inc., New York.

Boulton, A.J., C.G. Peterson, N.B. Grimm, and S.G. Fisher. 1992. Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. *Ecology* **73**:2192-2207.

Bournaud, M., B. Cellot, P. Richoux, and A. Berrahou. 1996. Macroinvertebrate community structure and environmental characteristics along a large river: congruity of patterns for identification to species or family. *Journal of the North American Benthological Society* **15**:232-253.

Bravard, J.P., C. Amoros, and G. Pautou. 1986. Impact of civil engineering works on the successions of communities in a fluvial system. *Oikos* **47**:92-111.

Brown, A.V., M.M. Lyttle, and K.B. Brown. 1998. Impacts of gravel mining on gravel bed streams. *Transactions of the American Fisheries Society* **127**:979-994.

Brown, T.J., Whitehouse, T.R., and Levings, C.D. 1989. Beach seine data from the Fraser River at the North Arm, Main Arm, and Agassiz during 1987-88. *Canadian Data Report of Fisheries and Aquatic Sciences* No. **737**.

Brunet, R.C., G. Pinay, F. Gazelle, and L. Roques. 1994. Role of the floodplain and riparian zone in suspended matter and nitrogen retention in the Adour River, southwest France. *Regulated Rivers: Research & Management* **9**:55-63.

Buffagni, A., G.A. Crosa, D.M. Harper, and J.L. Kemp. 2000. Using macroinvertebrate species assemblages to identify river channel habitat units: an application of the functional habitats concept to a large, unpolluted Italian river (River Ticino, northern Italy). *Hydrobiologia* **435**:213-225.

Carlander, K.D. 1969. Handbook of Freshwater Fishery Biology: Volume One. The Iowa State University Press, Ames, Iowa.

Caron, F. and A. Talbot. 1993. Re-evaluation of habitat classification criteria for juvenile salmon. *Canadian Special Publication of Fisheries and Aquatic Science* **118**:139-148.

Casselman, J.M., T. Penczak, L. Carl, R.H.K. Mann, J. Holcik, and W.A. Woitowich. 1990. An evaluation of fish sampling methodologies for large river systems. *Polskie Archiwum Hydrobiologii* **37**:521-551.

Church, M. 1990. Fraser River in central British Columbia. *In* Surface-Water Hydrology, The Geology of North America. *Edited by* M.G. Wolman and H.C. Riggs. Geological Society of America, Boulder, Colorado. pp. 282-287.

Church, M. 1992. Channel morphology and typology. *In* The Rivers Handbook. *Edited by* P. Calow. Blackwell Scientific Publishers, Oxford, UK. pp. 126-143.

Church, M. and D.G. Ham. 2004. Atlas of the alluvial gravel-bed reach of Fraser River in the Lower Mainland showing channel changes in the period 1912-1999. Department of Geography, The University of British Columbia, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/.

Church, M., D.G. Ham, and H. Weatherly. 2001. Gravel Management in Lower Fraser River. Department of Geography, The University of British Columbia, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/report2001.pdf.

Church, M. and D.G. McLean. 1994. Sedimentation in Lower Fraser River, British Columbia: Implications for Management. *In* The Variability of Large Alluvial Rivers. *Edited by* S.A. Schumm. American Society of Civil Engineers, New York. pp. 221-241.

Church, M., L.L. Rempel, and S.P. Rice. 2000. Morphological and habitat classification of the lower Fraser River gravel-bed reach. Report prepared by Department of Geography, The University of British Columbia for Fraser Basin Council, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/DFO_report.pdf.

Church, M. and O. Slaymaker. 1989. Disequilibrium of Holocene sediment yield in glaciated British Columbia. *Nature* **337**:452-454.

Church, M. and H. Weatherly. 1998. Historical changes in Minto Channel during the twentieth century. Report prepared by Department of Geography, The University of British Columbia for District of Chilliwack, Chilliwack, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/minto.pdf.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**:117-143.

Clarke, K.R. and M. Ainsworth. 1993. A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* **92**:205-219.

Clarke, K.R. and Gorley, R.N. 2001. PRIMER v5: User Manual/Tutorial. Plymouth Marine Laboratory, Plymouth, UK.

Clarke, K.R. and Warwick, R.M. 1994. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation. Plymouth Marine Laboratory, Plymouth, UK.

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

Collins, B. and Dunne, T. 1990. Fluvial Geomorphology and River-Gravel Mining: A Guide for Planners, Case Studies Included. California Department of Conservation, Sacramento, CA.

Culp, J.M., S.J. Walde, and R.W. Davies. 1983. Relative importance of substrate particle size and detritus to stream benthic macroinvertebrate microdistribution. *Canadian Journal of Fisheries and Aquatic Sciences* **40**:1568-1574.

Davies, N.M., R.H. Norris, and M.C. Thoms. 2000. Prediction and assessment of local stream habitat features using large-scale catchment characteristics. *Freshwater Biology* **45**:343-369.

Death, R.G. 1995. Spatial patterns in benthic invertebrate community structure: products of habitat stability or are they habitat specific? *Freshwater Biology* **33**:455-467.

Death, R.G. and M.J. Winterbourn. 1995. Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. *Ecology* **76**:1446-1460.

Desloges, J.R. and M. Church. 1989. Wandering gravel-bed rivers: Canadian landform examples. *Canadian Geographer* **33**:360-364.

Dillon, W.R. and Goldstein, M. 1984. Multivariate Analysis: Methods and Applications. John Wiley & Sons, New York.

Dokulil, M.T. 1994. Environmental control of phytoplankton productivity in turbulent turbid systems. *Hydrobiologia* **289**:65-72.

Downing, J.A. 1979. Aggregation, transformation, and the design of benthos sampling programs. *Journal of the Fisheries Research Board of Canada* **36**:1454-1463.

Dymond, P.F. 1998. Seasonal changes of benthic macroinvertebrate communities in southwestern British Columbia. M.Sc. thesis, Department of Zoology, The University of British Columbia.

Dynesius, M. and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. *Science* **266**:753-762.

Ellis, E.R. 2004. Hydraulic geometry of secondary channels of lower Fraser River, British Columbia, from Acoustic Doppler Velocimetry. M.Sc. thesis, Department of Geography, The University of British Columbia.

Ellis, E.R., M. Church, and M.L. Rosenau. 2004. Characterization of four floodplain side channels of the lower Fraser River. Report prepared by Department of Geography, The University of British Columbia for Habitat Conservation Trust Fund, Victoria, British Columbia. http://www.geog.ubc.ca/fraserriver/publications.html.

Fisheries and Oceans Canada. 1985. Inventory of dams in the Fraser River Basin. Water Use Unit, Habitat Management Division, Vancouver, British Columbia.

Fisheries and Oceans Canada. 1999. Eulachon. *Report No*. B6-06. DFO Science Stock Status Report, Vancouver, British Columbia.

Fitzpatrick Jr., J.F. 1983. How To Know the Freshwater Crustacea. Wm. C. Brown Co., Iowa.

Fraser Basin Council. 2002. Fraser River management plan: Hope to Mission. Proactive strategy for 2001/2002. Vancouver, British Columbia.

Fraser River Action Plan. 1995. Fraser River Pink. *Report No*. HQ-0429. Fisheries and Oceans Canada, Vancouver, British Columbia.

Fremling, C.R., J.L. Rasmussen, R.E. Sparks, S.P. Cobb, C.F. Bryan, and T.O. Claflin. 1989. Mississippi River fisheries: a case history. *Canadian Special Publication of Fisheries and Aquatic Science* **106**:309-351.

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management* **10**:199-214.

Galat, D.L., L.H. Fredrickson, D.D. Humburg, K.J. Bataille, J.R. Bodie, J. Dohrenwend, G.T. Gelwicks, J.E. Havel, D.L. Helmers, J.B. Hooker, J.R. Jones, M.F. Knowlton, J. Kubisiak, J. Mazourek, A.C. McColpin, R.B. Renken, and R.D. Semlitsch. 1998. Flooding to restore connectivity of regulated, large-river wetlands. *BioScience* **48**:721-733.

Gard, M. and E. Ballard. 2003. Applications of new technologies to instream flow studies in large rivers. *North American Journal of Fisheries Management* **23**:1114-1125.

Gee, J.H. and T.G. Northcote. 1963. Comparative ecology of two sympatric species of dace (Rhinichthys) in the Fraser River system, British Columbia. *Journal of the Fisheries Research Board of Canada* **20**:105-118.

Gomez, B., R.L. Naff, and D.W. Hubbell. 1989. Temporal variations in bedload transport rates associated with the migration of bedforms. *Earth Surface Processes and Landforms* **14**:135-156.

Gore, J.A. and F.D.J. Shields. 1995. Can large rivers be restored? *BioScience* **45**:142-152.

Graaf, G.D. 2003. The flood pulse and growth of floodplain fish in Bangladesh. *Fisheries Management and Ecology* **10**:241-247.

Graf, W.L. 2001. Damage control: restoring the physical integrity of America's rivers. *Annals of the Association of American Geographers* **91**:1-27.

Green, R.H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. Wiley, Chichester, England.

Gregory, R.S. and C.D. Levings. 1998. Turbidity reduces predation on migrating juvenile Pacific Salmon. *Transactions of the American Fisheries Society* **127**:275-285.

Gregory, S.V., D.W. Hulse, D.H. Landers, and E. Whitelaw. 1998. Integration of biophysical and socioeconomic patterns in riparian restoration of large rivers. *Hydrology in a Changing Environment* **1**:231-247.

Griffith, J.S. and D.A. Andrews. 1981. Effects of a small suction dredge on fishes and aquatic invertebrates in Idaho streams. *North American Journal of Fisheries Management* **1**:21-28.

Grift, R.E., A.D. Buijse, W.L.T. Van Densen, M.A.M. Machiels, J. Kranenbarg, J.G.P. Klein Bretleler, and J.J.G.M. Backx. 2003. Suitable habitats for 0-group fish in rehabilitated floodplains along the lower River Rhine. *River Research and Applications* **19**:353-374.

Growns, I.O. and J.A. Davis. 1994. Longitudinal changes in near-bed flows and macroinvertebrate communities in a Western Australian stream. *Journal of the North American Benthological Society* **13**:417-438.

Grubaugh, J.W. and R.V. Anderson. 1989. Upper Mississippi River: seasonal and floodplain forest influences on organic matter transport. *Hydrobiologia* **174**:235-244.

Gutreuter, S., A.D. Bartels, K. Irons, and M.B. Sandheinrich. 1999. Evaluation of the flood-pulse concept based on statistical models of growth of selected fishes of the Upper Mississippi River system. *Canadian Journal of Fisheries and Aquatic Sciences* **56**:2282-2291.

Hall, K.J., F.A. Koch, and I. Yesaki. 1974. Further investigations into water quality conditions in the Lower Fraser River system. *Report No*. 4. Westwater Research Centre, Vancouver, British Columbia.

Ham, D.G. and M. Church. 2002. Channel island and active channel stability in the lower Fraser River gravel reach. Department of Geography, The University of British Columbia, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/channelreport2002.pdf.

Ham, D.G. and M. Church. 2003. The sediment budget in the gravel-bed reach of Fraser River: 2003 revision. Department of Geography, The University of British Columbia, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/publications.html.

Hankin, D.G. and G.H. Reeves. 1988. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. *Canadian Journal of Fisheries and Aquatic Sciences* **45**:834-844.

Harvey, M.D., R.A. Mussetter, and E.J. Wick. 1993. A physical process-biological response model for spawning habitat formation for the endangered Colorado Squawfish. *Rivers* **4**:114-131.

Hawkins, C.P. 1981. Longitudinal and seasonal changes in functional organization of macroinvertebrate communities in four Oregon streams. *Ecology* **62**:387-397.

Hawkins, C.P., J.N. Hogue, L.M. Decker, and J.W. Feminella. 1997. Channel morphology, water temperature, and assemblage structure of stream insects. *Journal of the North American Benthological Society* **16**:728-749.

Hawkins, C.P., J.L. Kershner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A. McCullough, C.K. Overton, G.H. Reeves, R.J. Steedman, and M.K. Young. 1993. A hierarchical approach to classifying stream habitat features. *Fisheries* **18**:3-12.

Hawkins, C.P., R.H. Norris, J. Gerritsen, R.M. Hughes, S.K. Jackson, R.K. Johnson, and R.J. Stevenson. 2000. Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendations. *Journal of the North American Benthological Society* **19**:541-556.

Healey, M.C. 1994. Effects of dams and dikes on fish habitat in two Canadian river deltas. *In* Global Wetlands: Old World and New. *Edited by* W.J. Mitsch. Elsevier, Amsterdam. pp. 385-398.

Healey, M.C. 1997. Prospects for sustainability: integrative approaches to sustaining the ecosystem function of the lower Fraser Basin. Institute for Resources and Environment, The University of British Columbia, Vancouver, British Columbia.

Healey, M.C. 1998. Life history of chinook salmon. *In* Pacific salmon life histories. *Edited by* C. Groot and L. Margolis. UBC Press, Vancouver, British Columbia. pp. 311-394.

Healey, M.C. and J.S. Richardson. 1996. Changes in the productivity base and fish populations of the lower Fraser River (Canada) associated with historical changes in human occupation. *Archiv für Hydrobiologie Supplement* **113**:279-290.

Heard, W.R. 1991. Life history of pink salmon. *In* Pacific Salmon Life Histories. *Edited by* C. Groot and L. Margolis. UBC Press, Vancouver, British Columbia.

Heino, J., T. Muotka, H. Mykra, R. Paavola, H. Hamalainen, and E. Koskenniemi. 2003. Defining macroinvertebrate assemblage types in headwater streams: implications for bioassessment and conservation. *Ecological Applications* **13**:842-852.

Hohausova, E., G.H. Copp, and P. Jankovsky. 2003. Movement of fish between a river and its backwater: diel activity and relation to environmental gradients. *Ecology of Freshwater Fish* **12**:107-117.

Holland-Bartels, L.E. and M.R. Dewey. 1997. The influence of seine capture efficiency on fish abundance estimates in the upper Mississippi River. *Journal of Freshwater Ecology* **12**:101-111.

Humphries, P., A.J. King, and J.D. Koehn. 1999. Fish, flows and flood plains: links between freshwater fishes and their environment in the Murray-Darling River system, Australia. *Environmental Biology of Fishes* **56**:129-151.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Monographs* **54**:187-211.

Hynes, H.B.N. 1970. The Ecology of Running Waters. University of Toronto Press, Toronto, Ontario.

Hynes, H.B.N. 1989. Keynote Address. *Canadian Special Publication of Fisheries and Aquatic Science* **106**:5-10.

Inoue, M. and M. Nunokawa. 2002. Effects of longitudinal variations in stream habitat structure on fish abundance: an analysis based on subunit-scale habitat classification. *Freshwater Biology* **47**:1594-1607.

James, A.G. 1999. Time and the persistence of alluvium: river engineering, fluvial geomorphology, and mining sediment in California. *Geomorphology* **31**:265-290.

Johnson, B.L. and C.A. Jennings. 1998. Habitat associations of small fishes around islands in the upper Mississippi River. *North American Journal of Fisheries Management* **18**:327-336.

Jowett, I.G. 1993. A method for objectively identifying pool, run, and riffle habitats from physical measurements. *New Zealand Journal of Marine and Freshwater Research* **27**:241-248.

Jungwirth, M., S. Muhar, and S. Schmutz. 2002. Re-establishing and assessing ecological integrity in riverine landscapes. *Freshwater Biology* **47**:867-887.

Junk, W.J. 1984. Ecology of the varzea, floodplain of Amazonian whitewater rivers. *In* The Amazon. *Edited by* H. Sioli. Dr. W. Junk, Dordrecht, The Netherlands. pp. 215-244.

Junk, W.J., P.B. Bayley, and R.E. Sparks. 1989. The flood pulse concept in river-floodplain systems. *In* Proceedings of the International Large River Symposium. *Edited by* D.P. Dodge. Canadian Special Publication of Fisheries and Aquatic Sciences **106**. pp. 110-127.

Kellerhals, R. and M. Church. 1989. The morphology of large rivers: characterization and management. *In* Proceedings of the International Large River Symposium. *Edited by* D.P. Dodge. Canadian Special Publication of Fisheries and Aquatic Sciences **106**. pp. 31-48.

Kemp, J.L., D.M. Harper, and G.A. Crosa. 1999. Use of 'functional habitats' to link ecology with morphology and hydrology in river rehabilitation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **9**:159-178.

King, A.J., P. Humphries, and P.S. Lake. 2003. Fish recruitment on floodplains: the roles of patterns of flooding and life history characteristics. *Canadian Journal of Fisheries and Aquatic Sciences* **60**:773-786.

Knighton, D. 1998. Fluvial Forms and Processes. Oxford University Press Inc., New York, New York.

Kondolf, G.M. 1994. Geomorphic and environmental effects of instream gravel mining. *Landscape and Urban Planning* **28**:225-243.

Kondolf, G.M. 1995. Geomorphological stream channel classification in aquatic habitat restoration: uses and limitations. *Aquatic Conservation: Marine and Freshwater Ecosystems* **5**:127-141.

Kondolf, G.M. 1997. Hungry water: effects of dams and gravel mining on river channels. *Environmental Management* **2**:533-551.

Kondolf, G.M. 1998a. Environmental effects of aggregate extraction from river channels and floodplains. *In* Aggregate Resources: A Global Perspective. *Edited by* P.T. Bobrowsky. A.A. Balkema, Brookfield. pp. 113-129.

Kondolf, G.M. 1998b. Lessons learned from river restoration projects in California. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**:39-52.

Kondolf, G.M., M. Smeltzer, and L. Kimball. 2002. Freshwater gravel mining and dredging issues. Washington Departments of Fish and Wildlife, Ecology, and Transportation, Washington. http://www.wdfw.wa.gov/hab/ahg/freshdrg.pdf.

Krebs, C.J. 1998. Ecological Methodology. Benjamin/Cummings, California, USA.

Kruskal, J.B. 1964. Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis. *Psychometrika* **29**:1-27.

Lagasse, P.F. 1986. River response to dredging. *Journal of Waterway, Port, Coastal and Ocean Engineering* **112**:1-14.

Lancaster, J. and L.R. Belyea. 1997. Nested hierarchies and scale-dependence of mechanisms of flow refugium use. *Journal of the North American Benthological Society* **16**:221-238.

Leland, H.V. and S.V. Ford. 1998. Benthic invertebrate distributions in the San Joaquin River, California, in relation to physical and chemical factors. *Canadian Journal of Fisheries and Aquatic Sciences* **55**:1051-1067.

Leopold, L.B., Wolman, M.G., and Miller, J.P. 1964. Fluvial Processes in Geomorphology. W.H. Freeman and Co., San Francisco, USA.

Levings, C.D. and R.B. Lauzier. 1991. Extensive use of the Fraser River basin as winter habitat by juvenile chinook salmon (*Oncorhynchus tshawytscha*). *Canadian Journal of Zoology* **69**:1759-1767.

Lobb, M.D.I. and D.J. Orth. 1991. Habitat use by an assemblage of fish in a large warmwater stream. *Transactions of the American Fisheries Society* **120**:65-78.

Mackay, R.J. 1992. Colonization of lotic macroinvertebrates: a review of processes and patterns. *Canadian Journal of Fisheries and Aquatic Sciences* **49**:617-628.

Maddock, I. 1999. The importance of physical habitat assessment for evaluating river health. *Freshwater Biology* **41**:373-391.

Manly, B.F.J. 1986. Multivariate Statistical Methods: A Primer. Chapman & Hall, London, England.

Mantua, N.L., S.R. Hare, Y. Zhang, J.M. Wallace, and R.C. Francis. 1997. A Pacific interdecadal climate oscillation with impacts on salmon production. *Bulletin of the American Meteorological Society* **78**:1069-1079.

McCormick, F.H., D.V. Peck, and D.P. Larsen. 2000. Comparison of geographic classification schemes for Mid-Atlantic stream fish assemblages. *Journal of the North American Benthological Society* **19**:385-404.

McIntosh, R.P. 1995. H. A. Gleason's "individualistic community concept" and theory of animal communities: a continuing controversy. *Biological Reviews* **70**:317-357.

McLean, D.G. and M. Church. 1999. Sediment transport along lower Fraser River. 2. Estimates based on the long-term gravel budget. *Water Resources Research* **35**:2549-2559.

McLean, D.G., M. Church, and B. Tassone. 1999. Sediment transport along lower Fraser River. 1. Measurements and hydraulic computations. *Water Resources Research* **35**:2533-2548.

McPhail, J.D. and Carveth, R. 1993. A Foundation for Conservation: The Nature and Origin of the Freshwater Fish Fauna of British Columbia. Queen's Printer for British Columbia, Victoria, British Columbia.

McPhail, J.D. and Carveth, R. 1994. Field Key to the Freshwater Fishes of British Columbia. Resource Inventory Committee, Province of British Columbia, Victoria, British Columbia.

McPhail, J.D. and Lindsey, C.C. 1970. Freshwater fishes of northwestern Canada and Alaska. *Bulletin of the Fisheries Resource Board of Canada* No. **173**.

Meidinger, D. and Pojar, J. 1991. Ecosystems of British Columbia. Research Branch, Ministry of Forests, Victoria, British Columbia.

Merritt, R.W. and Cummins, K.W. 1996. An Introduction To the Aquatic Insects of North America. Kendall Hunt, Dubuque, Iowa.

Minchin, P.R. 1987. An evaluation of the relative robustness of techniques for ecological ordination. *Vegetatio* **69**:89-107.

Minns, C.K. 1997. Quantifying "no net loss" of productivity of fish habitats. *Canadian Journal of Fisheries and Aquatic Sciences* **54**:2463-2473.

Morrison, J., M.C. Quick, and M.G.G. Foreman. 2002. Climate change in the Fraser River watershed: flow and temperature projections. *Journal of Hydrology* **263**:230-244.

Mullen, D.M. and T.M. Burton. 1995. Size-related habitat use by longnose dace (*Rhinichthys cataractae*). *American Midland Naturalist* **133**:177-183.

Murphy, B.R. and Willis, D.W. 1996. Fisheries Techniques. American Fisheries Society, Maryland.

Neill, C.R. 1969. River bed transport related to meander migration rates. *Journal of Waterway Harbors Coastal Engineering Division American Society of Civil Engineering* **95**:783-786.

Neill, C.R. 1973. Hydraulic and morphologic characteristics of Athabasca River near Fort Assiniboine- the anatomy of a wandering gravel bed river. *Report No*. REH/73/8. Alberta Research Council, Highways and River Engineering Division, Edmonton, Alberta.

Newson, M.D. and C.L. Newson. 2000. Geomorphology, ecology and river channel habitat: mesoscale approaches to basin-scale challenges. *Progress in Physical Geography* **24**:195-217.

Northcote, T.G., G.L. Ennis, and M.H. Anderson. 1975. Periphytic and planktonic algae of the Lower Fraser River in relation to water quality conditions. *Report No*. **8**. Westwater Research Centre, Vancouver, British Columbia.

Northcote, T.G., N.T. Johnston, and K. Tsumura. 1976. Benthic, epibenthic and drift fauna of the Lower Fraser River. *Report No*. 11. Westwater Research Centre, Vancouver, British Columbia.

Northcote, T.G., N.T. Johnston, and K. Tsumura. 1978a. A regional comparison of species distribution, abundance, size and other characteristics of Lower Fraser River fishes. *Report No*. 14. Westwater Research Centre, Vancouver, British Columbia.

Northcote, T.G., N.T. Johnston, and K. Tsumura. 1978b. Feeding relationships and food web structure of Lower Fraser River fishes. *Report No*. 16. Westwater Research Centre, Vancouver, British Columbia.

Northcote, T.G. and P.A. Larkin. 1989. The Fraser River: A major salmonine production system. *In* Proceedings of the International Large River Symposium. *Edited by* D.P. Dodge. Canadian Special Publication of Fisheries and Aquatic Science **106**. pp. 172-204.

Olden, J.D., D.A. Jackson, and P.R. Peres-Neto. 2002. Predictive models of fish species distributions: a note on proper validation and chance predictions. *Transactions of the American Fisheries Society* **131**:329-336.

Osenberg, C.W., R.J. Schmitt, S.J. Holbrook, K.E. Abu-Saba, and A.R. Flegal. 1994. Detection of environmental impacts: natural variability, effect size, and power analysis. *Ecological Applications* **4**:16-30.

Oswood, M.W., J.B. Reynolds, J.G. Irons, and A.M. Milner. 2000. Distributions of freshwater fishes in ecoregions and hydroregions of Alaska. *Journal of the North American Benthological Society* **19**:405-418.

Paine, R.T. 1969. A note on trophic complexity and community stability. *American Naturalist* **103**:91-93.

Palmer, M.A., A.E. Bely, and K.E. Berg. 1992. Response of invertebrates to lotic disturbance: a test of the hyporheic refuge hypothesis. *Oecologia* **89**:182-194.

Palmer, M.A., C.C. Hakenkamp, and K. Nelson-Baker. 1997. Ecological heterogeneity in streams: why variance matters. *Journal of the North American Benthological Society* **16**:189-202.

Palmer, M.A. and N.L. Poff. 1997. The influence of environmental heterogeneity on patterns and processes in streams. *Journal of the North American Benthological Society* **16**:169-173.

Parsley, M.J. and L.G. Beckman. 1994. White sturgeon spawning and rearing habitat in the lower Columbia River. *North American Journal of Fisheries Management* **14**:812-827.

Parsley, M.J., L.G. Beckman, and G.T. McCabe Jr. 1993. Spawning and rearing habitat use by white sturgeons in the Columbia River downstream of McNary Dam. *Transactions of the American Fisheries Society*217-227.

Parsley, M.J., D.E. Palmer, and R.W. Burkhardt. 1989. Variation in capture efficiency of a beach seine for small fishes. *North American Journal of Fisheries Management* **9**:239-244.

Parsons, M., M.C. Thoms, and R.H. Norris. 2003. Scales of macroinvertebrate distribution in relation to the hierarchical organization of river systems. *Journal of the North American Benthological Society* **22**:105-122.

Perrin, C.J. and L.L. Rempel. 2000a. Eulachon (*Thaleichthys pacificus*) spawning activity in the Fraser River at Barnston Island, 2000. Report prepared by Limnotek Research and Development for Fraser River Port Authority, Richmond, British Columbia.

Perrin, C.J. and L.L. Rempel. 2000b. Observations of turbidity and biological communities in the Fraser River during operation of an hydraulic clamshell dredge. Report prepared by Limnotek Research and Development for Catherwood Towing Ltd., Mission, British Columbia.

Perrin, C.J. and L.L. Rempel. 2002. The effect of transfer pit operations on the aquatic community in the Fraser River at Barnston Island, 2000. Report prepared by Limnotek Research and Development for Fraser River Port Authority, Richmond, British Columbia.

Perrin, C.J., L.L. Rempel, and M.L. Rosenau. 2003a. White sturgeon spawning habitat in an unregulated river: Fraser River, Canada. *Transactions of the American Fisheries Society* **132**:154-165.

Perrin, C.J., J.L. Taylor, and T.B. Stables. 2003b. Effects of dredging and transfer pit operations on the aquatic community in the Lower Fraser River near Barnston Island. Report prepared by Limnotek Research and Development for Fraser River Port Authority, Richmond, British Columbia.

Peterman, R.M. 1990. Statistical power analysis can improve fisheries research and management. *Canadian Journal of Fisheries and Aquatic Sciences* **47**:2-15.

Peterson, J.T. and C.F. Rabeni. 2001a. Evaluating the physical characteristics of channel units in an Ozark stream. *Transactions of the American Fisheries Society* **130**:898-910.

Peterson, J.T. and C.F. Rabeni. 2001b. The relation of fish assemblages to channel units in an Ozark stream. *Transactions of the American Fisheries Society* **130**:911-926.

PGL Organix. 1996. Inventory of wood residues in the lower Fraser Basin and on eastern Vancouver Island. *Report No*. DOE/FRAP 1996-05. Environment Canada, Fraser River Action Plan, Vancouver, British Columbia.

Pitlick, J. and M.M. Van Steeter. 1998. Geomorphology and endangered fish habitats of the upper Colorado River 2. Linking sediment transport to habitat maintenance. *Water Resources Research* **34**:303-320.

Poff, N.L. 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. *Journal of the North American Benthological Society* **16**:391-409.

Poff, N.L. and J.D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. *Ecology* **76**:606-627.

Poff, N.L. and J.V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. *Canadian Journal of Fisheries and Aquatic Sciences* **46**:1805-1818.

Poff, N.L. and J.V. Ward. 1990. Physical habitat templet of lotic systems: recovery in the context of historical pattern of spatiotemporal heterogeneity. *Environmental Management* **14**:629-645.

Pomeroy, W.M. and C.D. Levings. 1980. Association and feeding relationships between *Eogammarus confervicolus* (Amphipoda, Gammaridae) and benthic algae on Sturgeon and Roberts Banks, Fraser River estuary. *Canadian Journal of Fisheries and Aquatic Sciences* **37**:1-10.

Poole, G.C., C.A. Frissell, and S.C. Ralph. 1997. In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. *Journal of the American Water Resources Association* **33**:879-896.

Popov, I.V. 1962. Application of morphological analysis to the evaluation of the general channel deformations of the River Ob. *Soviet Hydrology* **3**:267-324.

Power, M.E. 1987. Predator avoidance by grazing fishes in temperate and tropical streams: importance of stream depth and prey size. *In* Predation: direct and indirect effects on aquatic communities. *Edited by* W.C. Kerfoot and A. Shi. University Press of New England, Hanover, New Hampshire. pp. 262-284.

Power, M.E. 2001. Controls on food webs in gravel bedded rivers: the importance of the gravel-bed habitat to trophic dynamics. *In* Gravel-Bed Rivers. *Edited by* M.P. Mosely. New Zealand Hydrological Society Inc., Wellington. pp. 405-421.

Power, M.E., W.E. Dietrich, and J.C. Finlay. 1996. Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. *Environmental Management* **20**:887-895.

Power, M.E., R.J. Stout, C.E. Cushing, P.P. Harper, F.R. Hauer, W.J. Matthews, P.B. Moyle, B. Statzner, and I.R. Wais De Badgen. 1988. Biotic and abiotic controls in river and stream communities. *Journal of the North American Benthological Society* **7**:456-479.

Pringle, C.M., R.J. Naiman, G. Bretschko, J.R. Karr, M.W. Oswood, J.R. Webster, R.L. Welcomme, and M.J. Winterbourn. 1988. Patch dynamics in lotic systems: the stream as a mosaic. *Journal of the North American Benthological Society* **7**:503-524.

Quinn, J.M. and C.W. Hickey. 1994. Hydraulic parameters and benthic invertebrate distributions in two gravel-bed New Zealand rivers. *Freshwater Biology* **32**:489-500.

R.L.&L. 2000. Fraser River white sturgeon monitoring program - comprehensive report (1995 to 1999). Report prepared by Rivers, Lakes and Lands for BC Ministry of Environment, Lands and Parks, Surrey, British Columbia.

Rabeni, C.F., K.E. Doisy, and D.L. Galat. 2002. Testing the biological basis of a stream habitat classification using benthic invertebrates. *Ecological Applications* **122**:782-796.

Rabeni, C.F. and R.B. Jacobson. 1993. The importance of fluvial hydraulics to fish-habitat restoration in low-gradient alluvial streams. *Freshwater Biology* **29**:211-220.

Rader, R.B. 1997. A functional classification of the drift: traits that influence invertebrate availability to salmonids. *Canadian Journal of Fisheries and Aquatic Sciences* **54**:1211-1234.

Randall, R.G. and C.K. Minns. 2000. Use of fish production per unit biomass ratios for measuring the productive capacity of fish habitats. *Canadian Journal of Fisheries and Aquatic Sciences* **57**:1657-1667.

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

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

Reeves, G.H., L.E. Benda, K.M. Burnett, P.A. Bisson, and J.R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. *American Fisheries Society Symposium* **17**:334-349.

Rempel, L.L. and M. Church. 2002. Morphological and habitat classification of the lower Fraser River gravel-bed reach: confirmation and testing. Report prepared by Department of Geography, The University of British Columbia for Fraser Basin Council and Department of Fisheries and Oceans, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/DFO_report2.pdf.

Rempel, L.L. and M. Church. 2003. The Harrison Bar gravel removal experiment: final report. Department of Geography, The University of British Columbia, Vancouver, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/harris_0903.pdf.

Rempel, L.L., J.S. Richardson, and M.C. Healey. 1999. Flow refugia for benthic macroinvertebrates during flooding of a large river. *Journal of the North American Benthological Society* **18**:34-48.

Rempel, L.L., J.S. Richardson, and M.C. Healey. 2000. Macroinvertebrate community structure along gradients of hydraulic and sedimentary conditions in a large gravel-bed river. *Freshwater Biology* **45**:57-73.

Rempel, L.L. and D.G. Smith. 1998. Postglacial fish dispersal from the Mississippi refuge to the Mackenzie River basin. *Canadian Journal of Fisheries and Aquatic Sciences* **55**:893-899.

Resh, V.H. 1979. Sampling variability and life history features: basic consideration in the design of aquatic insect studies. *Journal of the Fisheries Research Board of Canada* **36** :290-311.

Ricciardi, A. and J.B. Rasmussen. 1999. Extinction rates of North American freshwater fauna. *Conservation Biology* **13**:1220-1222.

Rice, S.P. and M.T. Greenwood. 2001. Macroinvertebrate community changes at coarse sediment recruitment points along two gravel bed rivers. *Water Resources Research* **37**:2793-2803.

Rice, S.P., M.T. Greenwood, and C.B. Joyce. 2001. Tributaries, sediment sources, and the longitudinal ogranisation of macroinvertebrate fauna along river systems. *Canadian Journal of Fisheries and Aquatic Sciences* **58**:824-840.

Richardson, J.S. 1991. Seasonal food limitation of detritivores in a montane stream: an experimental test. *Ecology* **72**:873-887.

Richardson, J.S., T.J. Lissimore, M.C. Healey, and T.G. Northcote. 2000. Fish communities of the lower Fraser River (Canada) and a 21-year contrast. *Environmental Biology of Fishes* **59**:125-140.

Richter, B.D., D.P. Braun, M.A. Mendelson, and L.L. Master. 1997. Threats to imperiled freshwater fauna. *Conservation Biology* **11**:1081-1093.

Rivier, B. and J. Seguier. 1985. Physical and biological effects of gravel extraction in river beds. *In* Habitat modification and freshwater fisheries. *Edited by* J.S. Alabaster. Butterworths, London, UK. pp. 131-146.

Roper, B.B. and D.L. Scarnecchia. 1995. Observer variability in classifying habitat types in stream surveys. *North American Journal of Fisheries Management* **15**:49-53.

Rosenau, M.L. and M. Angelo. 2000. Sand and gravel management and fish-habitat protection in British Columbia salmon and steelhead streams. *Report No*. 2000/3. Pacific Fisheries Resource Conservation Council, Vancouver, British Columbia.

Rosenfeld, J. and S.M. Boss. 2001. Fitness consequences of habitat use for juvenile cutthroat trout: energetic costs and benefits in pools and riffles. *Canadian Journal of Fisheries and Aquatic Sciences* **58**:585-593.

Rosenfeld, J., M. Porter, and E. Parkinson. 2000. Habitat factors affecting the abundance and distribution of juvenile cutthroat trout (Oncorhynchus clarki) and coho salmon (Oncorhynchus kisutch). *Canadian Journal of Fisheries and Aquatic Sciences* **57**:766-774.

Salo, E.O. 1998. Life history of chum salmon (*Oncorhynchus keta*). *In* Pacific salmon life histories. *Edited by* C. Groot and L. Margolis. UBC Press, Vancouver, British Columbia. pp. 231-310.

Scarsbrook, M.R. and C.R. Townsend. 1993. Stream community structure in relation to spatial and temporal variation: a habitat templet study of two contrasting New Zealand streams. *Freshwater Biology* **29**:395-410.

Schiemer, F., T. Spindler, H. Wintersberger, A. Schneider, and A. Chovanec. 1991. Fish fry associations: important indicators for the ecological status of large rivers. *Verhandlungen der Internationalen Vereinigung für Limnologie* **24**:2497-2500.

Schindler, D.W. 1987. Detecting ecosystem responses to anthropogenic stress. *Canadian Journal of Fisheries and Aquatic Sciences* **44 (Suppl. 1)**:6-25.

Scott, W.B. and Crossman, E.J. 1973. Freshwater fishes of Canada. *Fisheries Research Board of Canada Bulletin* No. **184**.

Scullion, J., C.A. Parish, N. Morgan, and R.W. Edwards. 1982. Comparison of benthic macroinvertebrate fauna and substratum composition in riffles and pools in the impounded River Elan and the unregulated River Wye, mid-Wales. *Freshwater Biology* **12**:579-595.

Sear, D.A. and D. Archer. 1998. Effects of gravel extraction on stability of gravel-bed rivers: the Wooler Water, Northumberland, U.K. *In* Gravel-bed rivers in the environment. *Edited by* R.L. Beschta, P.D. Komar, and J.B. Bradley. Water Resources Publications, Highlands Ranch, Colorado. pp. 415-432.

Sedell, J.R. and J.L. Froggatt. 1984. Importance of streamside forests to large rivers: The isolation of the Willamette River, Oregon, U.S.A., from its floodplain by snagging and streamside forest removal. *Verhandlungen der Internationalen Vereinigung für Limnologie* **22**:1828-1834.

Sedell, J.R., J.E. Richey, and F.J. Swanson. 1989. The river continuum concept: a basis for the expected ecosystem behavior of very large rivers? *In* Proceedings of the International Large River Symposium. *Edited by* D.P. Dodge. Canadian Special Publication of Fisheries and Aquatic Sciences **106**. pp. 49-55.

Shaw, A. and J.S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* **58**:2213-2221.

Shepard, R.N. 1962. The analysis of proximities: multidimensional scaling with an unknown distance function. *Psychometrika* **27**:125-140.

Shipley, B. and P.A. Keddy. 1987. The individualistic and community-unit concepts as falsifiable hypotheses. *Vegetatio* **69**:47-55.

Smith, C.T. 2002. Molecular genetic markers and the conservation of anadromous fishes at broad and local scales: coho salmon (*Oncorhynchus kisutch*) and white sturgeon (*Acipenser transmontanus*) as case studies. Ph.D. thesis, Department of Biology, University of Victoria.

Smith, N.D. 1974. Sedimentology and bar formation in the upper Kicking Horse River, a braided meltwater system. *Journal of Geology* **82**:205-223.

Smock, L.A., J.E. Gladden, J.L. Riekenberg, L.C. Smith, and C.R. Black. 1992. Lotic macroinvertebrate production in three dimensions: channel surface, hyporheic, and floodplain environments. *Ecology* **73**:876-886.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham, and W.J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* **58**:325-333.

Sousa, W.P. 1984. The role of disturbance in natural communities. *Annual Review of Ecology and Systematics* **15**:353-391.

Southwood, T.R.E. 1977. Habitat, the templet for ecological strategies? *Journal of Animal Ecology* **46**:337-367.

Southwood, T.R.E. 1988. Tactics, strategies and templets. *Oikos* **52**:3-18.

Sparks, R.E. 1995. Need for ecosystem management of large rivers and their floodplains. *BioScience* **45**:168-182.

Stables, T.B. and C.J. Perrin. 2002. Video and bathymetric imaging of substrata in the Fraser River at Rosedale with reference to spawning habitat used by pink salmon (*Oncorhynchus gorbuscha*). Report prepared by Shuksan Fisheries Consulting and Limnotek Research and Development for BC Ministry of Water, Land and Air Protection, Surrey, British Columbia.

Stalnaker, C.B., R.T. Milhous, and K.D. Bovee. 1989. Hydrology and hydraulics applied to fishery management in large rivers. *Canadian Special Publication of Fisheries and Aquatic Science* **106**:13-30.

Stanford, J.A. and J.V. Ward. 1988. The hyporheic habitat of river ecosystems. *Nature* **335**:64-66.

Stevens, L.E., J.C. Schmidt, T.J. Ayers, and B.T. Brown. 1995. Flow regulation, geomorphology, and Colorado River marsh development in the Grand Canyon, Arizona. *Ecological Applications* **5**:1025-1039.

Stewart-Oaten, A. and J.R. Bence. 2001. Temporal and spatial variation in environmental impact assessment. *Ecological Monographs* **71**:305-339.

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker. 1986. Environmental Impact Assessment: "Pseudoreplication" in time? *Ecology* **67**:929-940.

Stewart, K.W. and Stark, B.P. 1988. Nymphs of North America Stonefly Genera (Plecoptera). Entomological Society of America, Maryland.

Surian, N. 1999. Channel changes due to river regulation: the case of the Piave River, Italy. *Earth Surface Processes and Landforms* **24**:1135-1151.

Sutek and Kellerhals. 1989. Assessing gravel supply and removal in fisheries streams. Report prepared by Sutek Services and Kellerhals Engineering for Canada Department of Fisheries and Oceans and British Columbia Ministry of Environment, Vancouver, British Columbia.

Sweka, J.A. and K.J. Hartman. 2001. Influence of Turbidity on Brook Trout Reactive Distance and Foraging Success. *Transactions of the American Fisheries Society* **130**:138-146.

Sylte, T. and C. Fischenich. 2002. Techniques for measuring substrate embeddedness. *Report No*. ERDC TN-EMRRP-SR-36. US Army Engineer Research and Development Center, Vicksburg, Missouri. http://www.wes.army.mil/el/emrrp/pdf/sr36.pdf.

Ter Braak, C.J.F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. *Ecology* **67**:1167-1179.

Thomson, J.R., M.P. Taylor, K.A. Fryirs, and G.J. Brierley. 2001. A geomorphological framework for river characterization and habitat assessment. *Aquatic Conservation: Marine and Freshwater Ecosystems* **11**:373-389.

Thorp, J.H. 1992. Linkage between islands and benthos in the Ohio River, with implications for riverine management. *Canadian Journal of Fisheries and Aquatic Sciences* **49**:1873-1882.

Thorp, J.H., A.R. Black, K.H. Haag, and J.D. Wehr. 1994. Zooplankton assemblages in the Ohio River: seasonal, tributary, and navigation dam effects. *Canadian Journal of Fisheries and Aquatic Sciences* **51**:1634-1643.

Thorp, J.H. and M.D. Delong. 1994. The riverine productivity model: a heuristic view of carbon sources and organic processing in large river ecosystems. *Oikos* **70**:305-308.

Tipper, H.W. 1971. Glacial geomorphology and Pleistocene history of central British Columbia. *Bulletin of the Geological Survey of Canada* No. **196**.

Townsend, C.R. 1989. The patch dynamics concept of stream community ecology. *Journal of the North American Benthological Society* **8**:36-50.

Trainor, K. and M. Church. 2003. Quantifying variability in stream channel morphology. *Geomorphology* **39**:1248-1259.

Underwood, A.J. 1991. Beyond BACI: Experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research* **42**:569-587.

Underwood, A.J. 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology* **161**:145-178.

Underwood, A.J. 1993. The mechanics of spatially replicated sampling programmes to detect environmental impacts in a variable world. *Australian Journal of Ecology* **18**:99-116.

Underwood, A.J. 1994. On beyond BACI: Sampling designs that might reliably detect environmental disturbances. *Ecological Applications* **4**:3-15.

Underwood, A.J. 1997. Experiments in Ecology: Their Logical Design and Interpretation Using Analysis of Variance. Cambridge University Press, Cambridge, England.

USACE. 1982. Report on the impacts of commercial dredging on the fishery of the lower Kansas River. *Report No*. DACW 41-79-C-0075. U.S. Army Corps of Engineers, Kansas City, Missouri.

van der Nat, D., K. Tockner, P.J. Edwards, J.V. Ward, and A.M. Gurnell. 2003. Habitat change in braided flood plains (Tagliamento, NE-Italy). *Freshwater Biology* **48**:1799-1812.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* **37**:130-137.

Waite, I.R., A.T. Herlihy, D.P. Larsen, N.S. Urquhart, and D.J. Klemm. 2004. The effects of macroinvertebrate taxonomic resolution in large landscape bioassessments: an example from the Mid-Atlantic Highlands, U.S.A. *Freshwater Biology* **49**:474-489.

Walters, D.M., D.S. Leigh, M.C. Freeman, B.J. Freeman, and C.M. Pringle. 2003. Geomorphology and fish assemblages in a Piedmont river basin, U.S.A. *Freshwater Biology* **48**:1950-1970.

Ward, J.V. 1989. The four-dimensional nature of lotic ecosystems. *Journal of the North American Benthological Society* **8**:2-8.

Ward, J.V. and J.A. Stanford. 1989. Riverine ecosystems: The influence of man on catchment dynamics and fish ecology. *In* Proceedings of the International Large River Symposium. *Edited by* D.P. Dodge. Canadian Special Publication of Fisheries and Aquatic Sciences **106**. pp. 56-64.

Ward, J.V., K. Tockner, and F. Schiemer. 1999. Biodiversity of floodplain river ecosystems: ecotones and connectivity. *Regulated Rivers: Research & Management* **15**:125-139.

Warner, B.G., R.W. Mathewes, and J.J. Clague. 1982. Ice-free conditions on the Queen Charlotte Islands, British Columbia, at the height of late Wisconsin glaciation. *Science* **218**:675-677.

Weatherly, H. and M. Church. 1999. Gravel extraction inventory for Lower Fraser River Mission to Hope - 1964 to 1998. Report prepared by Department of Geography, The University of British Columbia for District of Chilliwack, Chilliwack, British Columbia. http://www.geog.ubc.ca/fraserriver/reports/gravelextraction.pdf.

Welcomme, R.L. 1979. Fishery Management in Large Rivers. Food and Agriculture Organization of the United Nations, Rome, Italy.

Welcomme, R.L. 1995. Relationships between fisheries and the integrity of river systems. *Regulated Rivers: Research & Management* **11**:121-136.

Wetmore, S.H., R.J. Mackay, and R.W. Newbury. 1990. Characterization of the hydraulic habitat of Brachycentrus occidentalis, a filter-feeding caddisfly. *Journal of the North American Benthological Society* **9**:157-169.

Whiting, P.J. and J.B. Bradley. 1993. A process-based classification system for headwater streams. *Earth Surface Processes and Landforms* **18**:603-612.

Wiggins, G.B. 1996. Larvae of the North America Caddisfly Genera (Tricoptera). University of Toronto Press, Toronto, Ontario.

Wilcock, P.R. and B.W. McArdell. 1993. Surface-based factional transport rates: mobilization thresholds and partial transport of a sand-gravel sediment. *Water Resources Research* **29**:1297-1312.

Wiley, M.J., S.L. Kohler, and P.W. Seelbach. 1997. Reconciling landscape and local views of aquatic communities: lessons from Michigan trout streams. *Freshwater Biology* **37**:133-148.

Williams, L.G. and C. Scott. 1962. Principal diatoms of major waterways of the United States. *Limnology and Oceanography* **7**:365-379.

Wolcott, J.F. and M. Church. 1991. Strategies for sampling spatially heterogeneous phenomena: the example of river gravels. *Journal of Sedimentary Petrology* **61**:534-543.

Wolman, M.G. 1954. A method of sampling coarse river-bed material. *American Geophysical Union Transactions* **35**:951-956.

Wolman, M.G. and J.P. Miller. 1960. Magnitude and frequency of forces in geomorphic processes. *Journal of Geology* **68**:64-74.

Wood, P.J. and P.D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. *Environmental Management* **21**:203-217.

Wright, K.K. and J.L. Li. 2002. From continua to patches: examining stream community structure over large environmental gradients. *Canadian Journal of Fisheries and Aquatic Sciences* **59**:1404-1417.

Wydoski, R.G. and R.S. Wydoski. 2002. Age, growth, and reproduction of mountain suckers in Lost Creek Reservoir, Utah. *Transactions of the American Fisheries Society* **131**:320-328.

Zallen, M. and J. Boyd. 1986. Investigations of habitat preferences by juvenile chinook salmon in the Fraser River mainstem - Floods-Hope area. *Report No*. 86-ESL-179. Report prepared by ESL Environmental Sciences Ltd. for British Columbia Ministry of Transportation and Highways, Design and Surveys Branch, Vancouver, British Columbia.

Zar, J.H. 1984. Biostatistical Analysis. Prentic Hall, New Jersey.

Zaroban, D.W., M.P. Mulvey, T.R. Maret, R.M. Hughes, and G.D. Merritt. 1999. Classification of species attributes for Pacific Northwest freshwater fishes. *Northwest Science* **73**:81-93.

Appendix A

Biomass Determination by Length-Weight Regression

List of Tables

List of Figures

Appendix A. Biomass Determination by Length-Weight Regression

Of the 54,684 fish collected by various methods over three years of sampling, 26,771 (49%) were measured for fork length and 11,533 (21%) were weighed to the nearest 1/10th gram. Only a fraction of all fish were measured and weighed in order to minimize the stress to captured fish and to maximize time efficiency in the field. Weather conditions, particularly wind, also limited the number of fish weighed because the scale became unreliable. All fish belonging to the family Salmonidae, and a minimum of 15 fish of each non-salmonid species, were measured and weighed. The 15 fish of each non-salmonid species were selected as being representative of the range of sizes captured and the sub-sampled size distribution was then extrapolated to the total catch for each species.

In addition to length measurements, the biomass of each fish needed to be determined. Fish weight is related to length and a log-log linear regression provides an accurate model from which to estimate weight (Murphy and Willis 1996). The regression model is species-specific and the large number of fish from Fraser River for which weight and length were known was ideal for precise model estimation. Sampling near-shore by beach seine targeted young age classes; hence, length-weight models provided best estimates for these size classes. The regression model for some species may differ between adult and juvenile fish.

The regression model is based on a log-log transformation and back-transformation of the weight data (Y-value) to original units (grams) is necessary. This back-transformation requires a bias correction because the predicted Y-value by back-transformation is the *geometric* mean whereas the predicted value of log-Y is the *arithmetic* mean of the conditional distribution of log-Y at log-X. The geometric mean is always lower than the arithmetic mean. Whereas bias correction in linear regression commonly is applied in sediment transport (e.g., Ferguson 1986) and statistical research (e.g., Duan 1983), it appears to have been ignored in most fisheries research.

Three methods are available for bias correction: the Quasi-Maximum Likelihood Estimator (QMLE); the Smearing Estimator (SMEAR); and the Minimum Variance Unbiased Estimator (MVUE). In general, QMLE is applied only when the distribution of residuals is normal and SMEAR (also referred to as Duan Estimator, after Duan 1983) is used when a non-normal error distribution occurs. MVUE is computationally involved and uncommonly applied (Cohn and Gilroy 1991). The SMEAR method (Duan 1983) was used to estimate fish weight for Fraser River fish because residuals had a non-normal distribution.

Following the SMEAR method, back-transformed and bias-corrected fish weight (W_{smear}) for each species was calculated as:

$$W_{smear} = W_{reg} \times \frac{\sum_{i=1}^{N} e^{res_i}}{N}$$

where W_{reg} is the predicted fish weight by linear regression, res_i are the residuals from the regression model, and N is number of fish in the species group. The method performs nearly as well as its parametric equivalents when residuals are normally distributed and the method has the advantage that it can be generalized to transformations other than the logarithm (Cohn and Gilroy 1991).

Log-log regression was applied to 17 species for which sufficient length and weight data for juvenile fish were available. Comparisons with previously published information on length-weight relations were made for those fish species with available data (Northcote *et al.* 1978). Although Richardson *et al*. (2000) presented updated length-weight regression slopes for several species, the analysis pooled data from downstream sampling locations in the estuary, making comparisons inappropriate. Nevertheless, previously published length-weight relations did not include a bias-correction factor.

The SMEAR method of weight estimation was evaluated for all fish with actual weight measurements to determine if weight estimates were improved with the SMEAR correction: estimated fish weight was subtracted from the actual weight, and the difference was divided by the actual fish weight to proportionately weight the error according to fish size. The SMEAR method was then evaluated by comparing the average weighted error for all fish of a given species to the average weighted error without bias-correction.

As a secondary analysis, separate species-specific regression slopes were calculated for each season and channel sub-reach to determine if length-weight relations differed in time or space. Only ten species were collected in sufficient numbers across each level of season and sub-reach for the analysis. Slopes were compared separately by season and sub-reach following methods of covariance analysis for multiple slope comparisons (Zar 1984). Tukey’s test was applied to significantly different slopes to elucidate differences. When slopes were similar by season or segment, the elevations of the slopes were compared (Zar 1984).

Linear regression results are presented in **Table A-1** for the seventeen fish species with sufficient length and weight data available. Twelve of 17 species had r^2 values greater than 0.90 and 4 species for which the r^2 was low had either a low sample size or narrow weight range because of a short residence time in Fraser River. Anadromous chum salmon and marine stickleback, in particular, occur in Fraser River only briefly for spawning. Longnose dace had a low r^2 despite its large sample size, possibly because the difference in size between fry and adults is relatively small compared with most other species. As well, the precision of weight measurements (1/10th gram) for longnose dace fry, which typically weigh less than 1.5 g, may have been inadequate.

Previously published length-weight regression slopes for four species in the gravel reach (Northcote *et al.* 1978) were matched well with those derived in this study. The slopes for peamouth chub and prickly sculpin fell within the range observed by Northcote *et al*. (1978), whereas largescale sucker had a slightly higher slope and northern pikeminnow a slightly lower slope than previously published data.

The SMEAR bias-correction method provided an improved estimate of fish weight for 11 of 17 species when compared with the actual weight of fish. The weight of six species was estimated with slightly more accuracy when no bias-correction was applied (**Table A-2**). The accuracy of weight estimates was weakly related to sample size as those species with low error rates often had a large sample size (CHI, MTS, RSS). **Figure A-1** presents the relation between the actual and predicted weights for each fish species using the SMEAR method. Of those 5 species with r^2 values less than 0.90, the weights of 3 were best estimated using the SMEAR method and weights of 2 species were most accurate with no bias-correction. These two species, chum salmon and marine stickleback, are anadromous species that have a brief in-river residence time and consequently a narrow weight range is represented in each freshwater cohort (**Figure A-1**). Overall, the SMEAR bias-correction provided the most statistically defensible and accurate estimates of fish weight. Hence, biomass estimation for those fish lacking weight measurements in this study was made by SMEAR-corrected regression.

Table A-1. Length-weight regression summary and the SMEAR bias-correction factor for common fish species in the gravel reach.

Species*	Size (mm)	N	r^2	Intercept	Slope	$\sum_{i=1}^{N} e^{res_i}$	SMEAR
CHI	30-120	2961	0.94	-5.594	3.343	2969.6	1.003
CHU	29-49	165	0.40	-5.346	3.093	166.7	1.011
CRS	33-81	137	0.75	-5.491	3.283	137.8	1.006
CUT	32-325	20	0.99	-4.886	2.957	20.1	1.003
LED	23-130	1716	0.93	-5.186	3.110	1724.1	1.005
LGS	29-365	382	0.99	-5.083	3.063$^+$	383.1	1.003
LND	21-81	556	0.70	-4.793	2.877	561.5	1.010
MSB	41-76	63	0.51	-2.826	1.800	63.1	1.002
MTS	30-258	1134	0.99	-4.912	2.993	1135.9	1.002
MWF	31-310	368	0.99	-4.961	2.970	368.8	1.002
NPM	30-340	741	0.99	-4.949	2.983$^+$	742.3	1.002
PEA	23-235	786	0.99	-5.058	3.036$^+$	789.1	1.004
PRS	22-198	297	0.98	-5.365	3.210$^+$	298.2	1.004
RBT	29-314	70	0.99	-4.74	2.872	70.3	1.004
RSS	25-160	1962	0.97	-5.04	3.065	1967.0	1.003
SOC	33-133	103	0.90	-5.409	3.198	103.5	1.005
TSS	19-60	60	0.68	-4.448	2.623	60.5	1.008

*Three-letter species codes listed in Chapter 3.

+ Range of slopes determined by Northcote *et al.* (1978) for the gravel reach: LGS, 3.003-3.020; NPM, 3.083-3.251; PEA, 3.021-3.067; PRS, 3.134-3.219.

Table A-2. Evaluation of the SMEAR method for bias correction of predicted fish weight. The best estimate for each species is indicated by bold-faced type.

Species	**N**	**[Actual Weight – Predicted Weight] / Actual Weight (%)***		**Difference (No Correction – SMEAR)**
		No Correction	**SMEAR**	
CHI	2961	11.55	**11.45**	0.10
CHU	165	**29.75**	30.60	-0.85
CRS	137	18.67	**17.62**	1.05
CUT	20	10.95	**10.10**	0.85
LED	1716	**14.69**	15.26	-0.57
LGS	382	11.09	**10.99**	0.10
LND	556	24.68	**24.67**	0.01
MSB	63	**10.91**	11.05	-0.14
MTS	1134	**8.84**	8.92	-0.08
MWF	368	10.43	**10.42**	0.01
NPM	741	**9.24**	9.39	-0.15
PEA	786	13.86	**13.85**	0.01
PRS	297	13.63	**13.48**	0.15
RBT	70	**12.32**	12.61	-0.29
RSS	1962	10.99	**10.88**	0.10
SOC	103	17.34	**15.86**	1.48
TSS	60	25.33	**25.23**	0.10

* values are the average of all fish for which weight was known

A secondary analysis of spatial and seasonal differences in the length-weight relation of common species was conducted by comparing separately derived regression slopes for each of the channel sub-reaches (Cheam, Rosedale, Chilliwack) and three seasons (spring, summer, fall). An insufficient number of fish were captured during winter months for analysis. The regression slopes did not differ among sub-reaches for 6 of 10 species, and the slope elevations were similar for 5 of these species (**Table A-3**). Only leopard dace had similar slopes among sub-reaches but a significant difference in slope elevation for the Cheam Segment. Chinook salmon showed a difference in slope among all sub-reaches, whereas the slope for mountain sucker and prickly sculpin differed only between the Chilliwack and Rosedale sub-reaches. Northern pikeminnow in the Cheam sub-reaches had a length-weight slope that differed from both downstream sub-reaches.

Table A-3. Covariance analysis to determine if the slope of length-weight regressions differed among *sub-reaches* for common fish species in the gravel reach.

Species	Slope			Elevation		
	F	**p**	**Contrast***	**F**	**p**	**Contrast***
CHI	99.0	<0.0005	all differ			
LED	0.5	0.25	all similar	5.7	<0.005	Ck R Cm
LGS	1.1	0.25	all similar	1.0	0.25	all similar
LND	1.7	0.1	all similar	0.1	0.25	all similar
MTS	5.5	<0.005	R Cm Ck			
MWF	1.8	0.1	all similar	1.2	0.25	all similar
NPM	4.4	<0.025	Ck R Cm			
PEA	0.1	0.25	all similar	1.6	0.1	all similar
PRS	4.1	<0.025	R Cm Ck			
RSS	2.5	0.08	all similar	3.0	0.05	all similar

Cm: Cheam Segment; R: Rosedale Segment; Ck: Chilliwack Segment.
*underscore indicates which contrasts are statistically similar based on Tukey's Test.

Only longnose dace had a similar slope and similar slope elevation in all seasons. Six of 10 species had different length-weight slopes between seasons (**Table A-4**), and summer differed from fall in all cases. Summer also differed from spring in 3 of 6 cases. Of the four species with similar slopes between seasons, three (peamouth chub, longnose dace, mountain whitefish) also had similar slopes among channel sub-reaches.

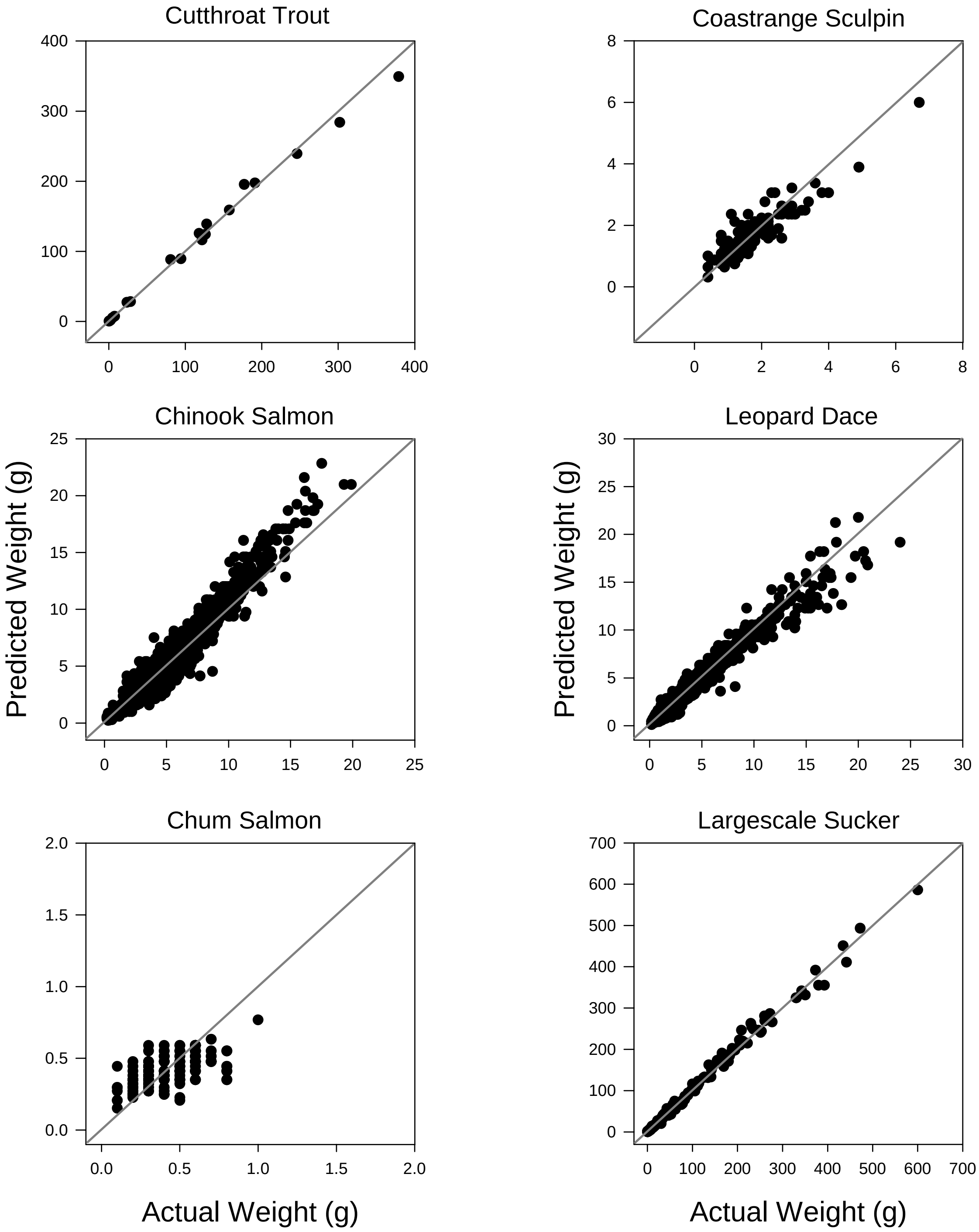


Figure A-1. Actual and predicted fish weight based on linear regression and the SMEAR method of bias-correction.

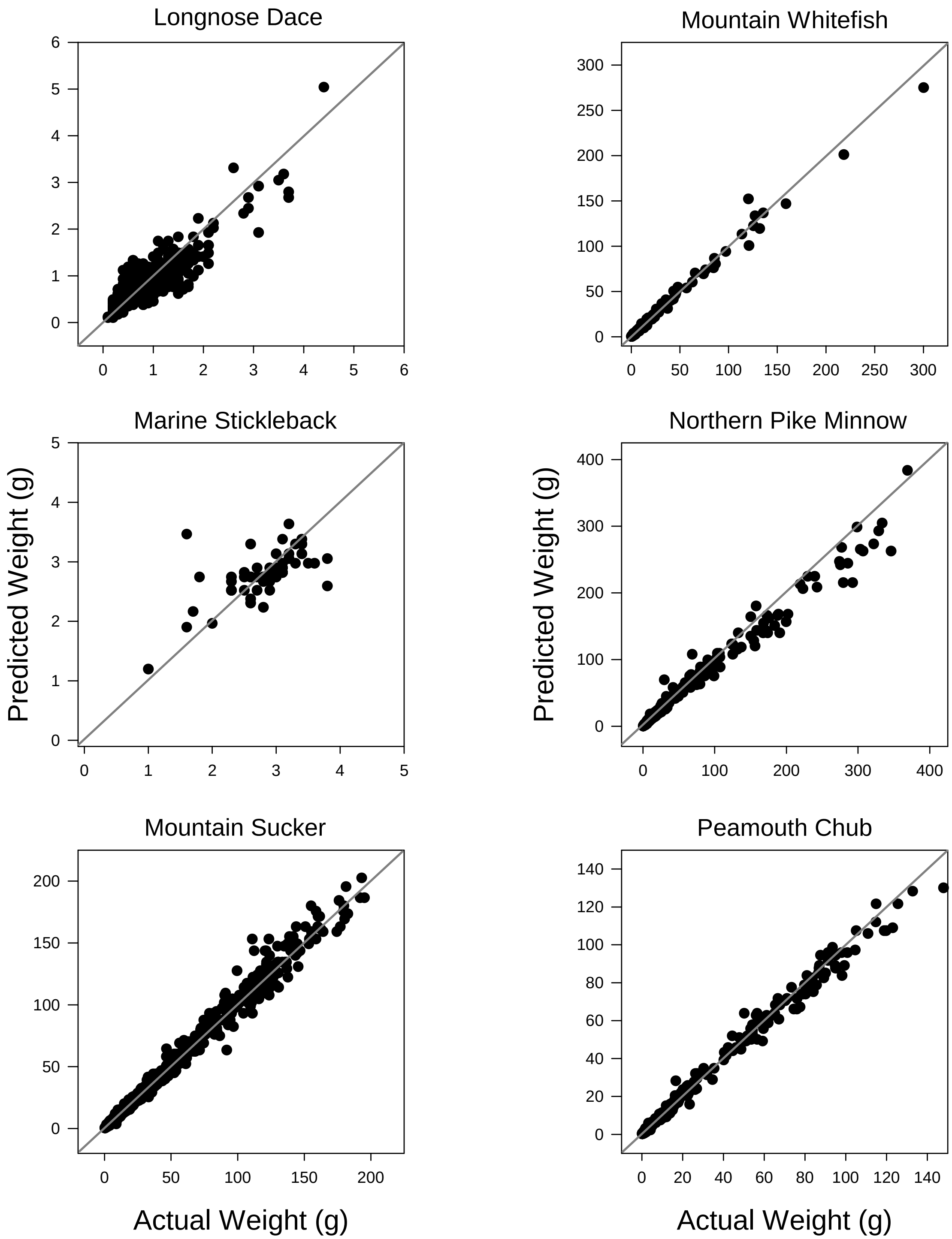


Figure A-1. continued.

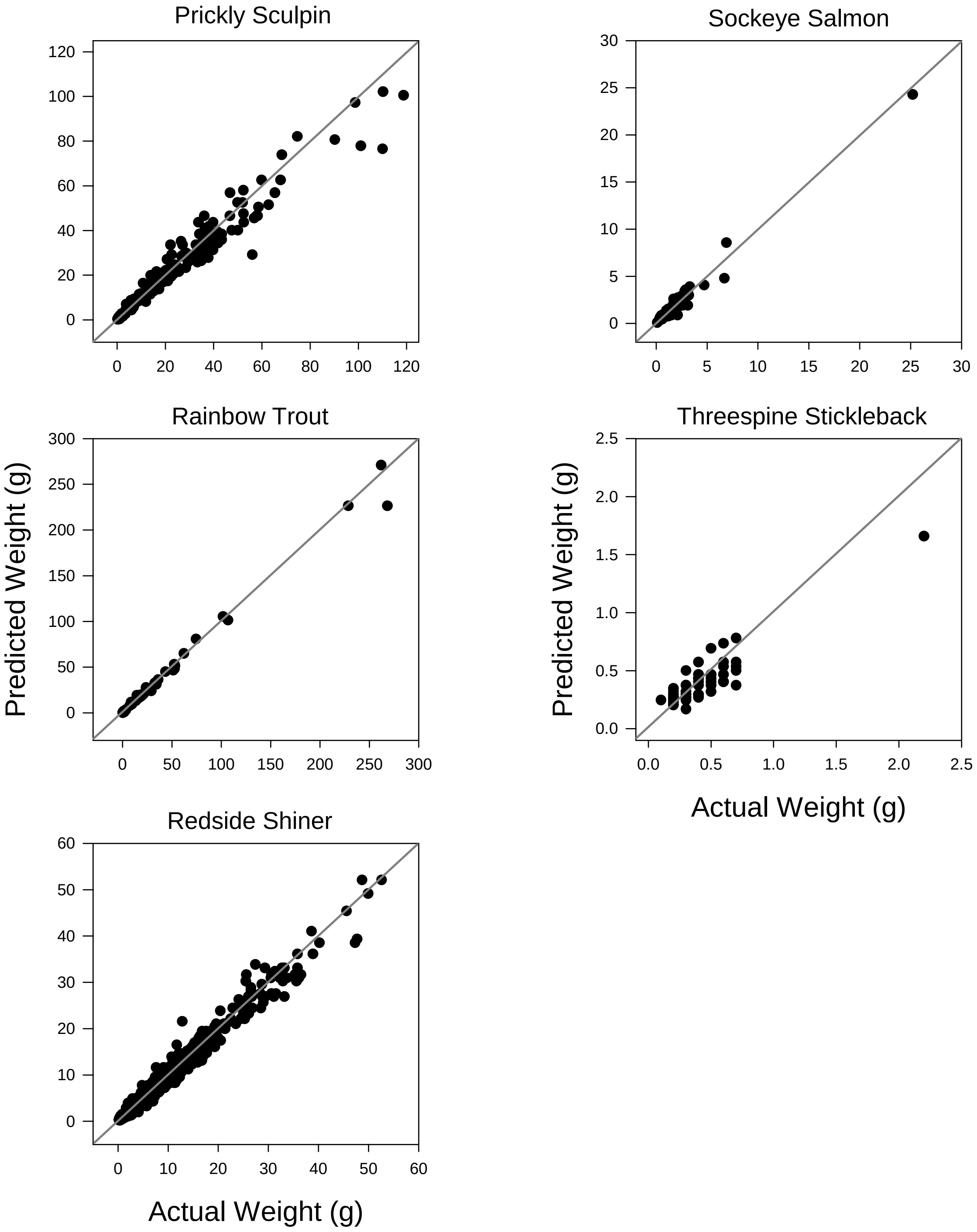


Figure A-1. continued.

Table A-4. Covariance analysis to determine if the slope of length-weight regressions differed among *seasons* for common fish species in the gravel reach.

Species	**Slope**			**Elevation**		
	F	**p**	**Contrast***	**F**	**p**	**Contrast***
CHI	231.8	<0.0005	all differ			
LED	22.1	<0.0005	Sp Fa Su			
LGS	4.7	<0.01	Sp Fa Su			
LND	3.1	0.05	all similar	1.4	0.25	all similar
MTS	0.2	0.25	all similar	10.7	<0.0005	Sp Fa Su
MWF	2.2	0.1	all similar	3.8	<0.025	Sp Su Fa
NPM	6.6	<0.0025	Su Sp Fa			
PEA	1.0	0.25	all similar	0.1	0.25	all similar
PRS	7.5	<0.001	Su Sp Fa			
RSS	15.1	<0.0005	Su Sp Fa			

Sp: Spring; Su: Summer; Fa: Fall.
*underscore indicates which contrasts are statistically similar based on Tukey's Test.

Appendix B:

Day and Night Fish Sampling in Summer and Winter Seasons

List of Figures

Appendix B. Day and Night Fish Sampling in Summer and Winter Seasons

The interactive effects of water turbidity and daylight on fish catch were examined by paired sampling during day and night periods in summer (high turbidity) and winter (low turbidity). "Night" time corresponded to post-sunset darkness hours. For each pair of day and night samples, the sampling location was identical so that differences in fish catch could not be attributed to environmental differences. Sampling was carried out on several occasions between winter 2000 and summer 2001 using three different methods (beach seine, gillnet, minnow trap), and was limited to sites where night sampling was feasible. For beach seine sampling, night site access was by vehicle except at Calamity Bar in August 2001 when the field crew camped on the bar and sampled through the night. For both gillnet and minnow trap sampling, night sets were begun near dusk using a boat and the gear was left fishing overnight and retrieved the following morning. No night gillnetting was authorized during summer months in order to minimize fish mortality.

Beach seine sampling methods are described in Chapter 3. Catch data are presented in **Figures B-1** though **B-5**.

Gillnetting was restricted to habitats of deep, standing water away from the main channel to minimize the risk of intercepting migratory salmon. Nets were set at the water surface and were clearly marked with floats and permit identification while left fishing in the river. Daytime sets averaged 2 hr in duration whereas nighttime sets averaged 18 hr (winter months only). Fish were then removed from the net as carefully as possible to minimize injury, and immediately transferred to holding buckets containing fresh river water for recovery and processing. Winter gill net catch data are presented in **Figure B-6**.

Minnow traps were used extensively during winter months and traps often were set where both beach seining and gillnetting were not feasible, such as surrounding large woody debris accumulations and along densely vegetated island banks. Traps (baited with salmon roe) were clearly marked with floats and anchored with lead weight to the bottom while fishing. Fish were removed from the trap and immediately transferred to holding buckets containing fresh river water for processing. Daytime sets averaged 5 hr, however most traps were left overnight and set duration averaged 19 hr. Daytime catch rates are considered more reliable because bait "attractiveness" decreased exponentially with time, which biased night catch rates. In many cases, the bait was consumed or washed out of the trap during sets of extended duration. Average catch rates in summer and winter are presented in **Figure B-7**.

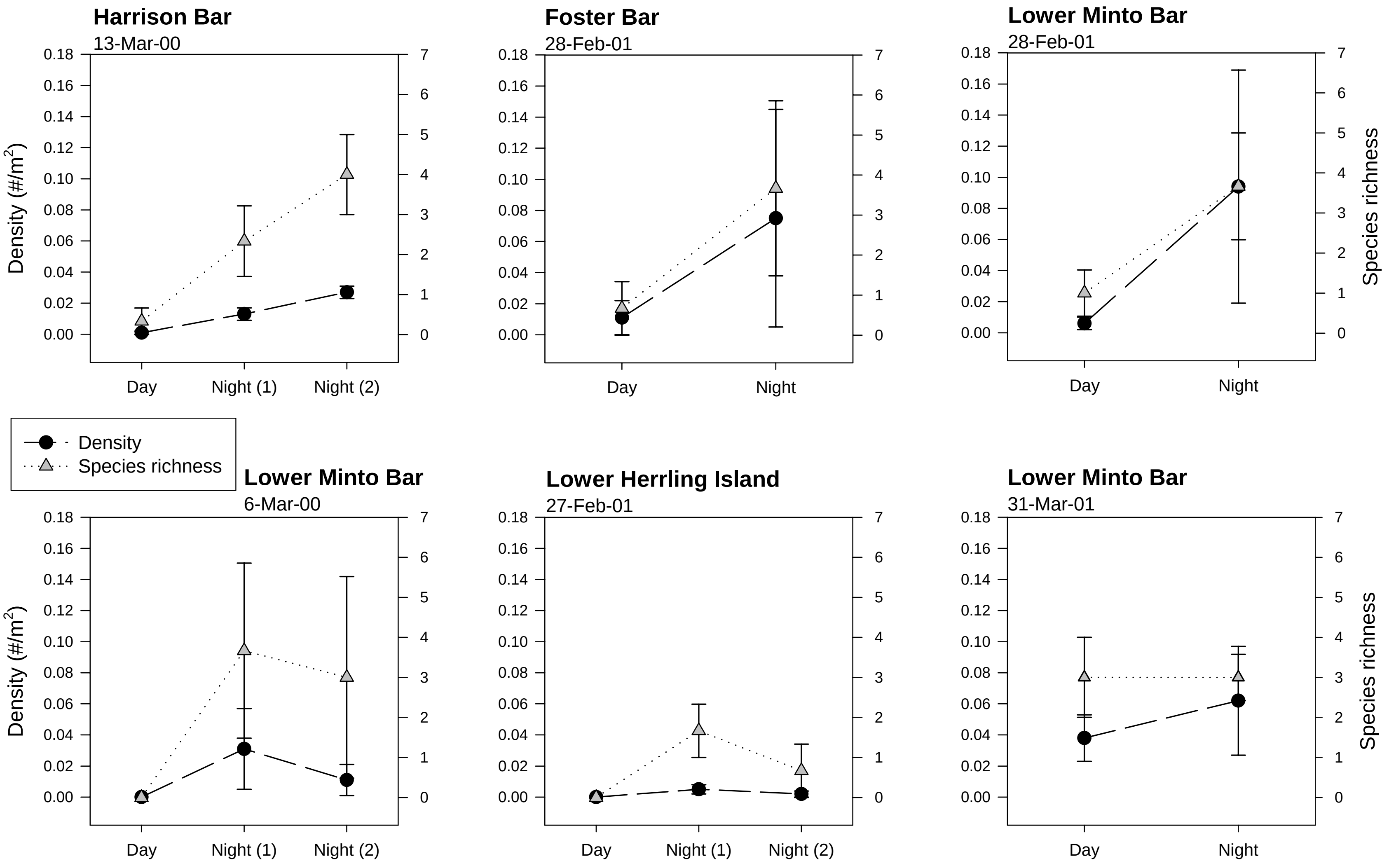


Figure B-1. Fish density and species richness in *beach seine* hauls carried out on six dates in winter when water clarity was high.

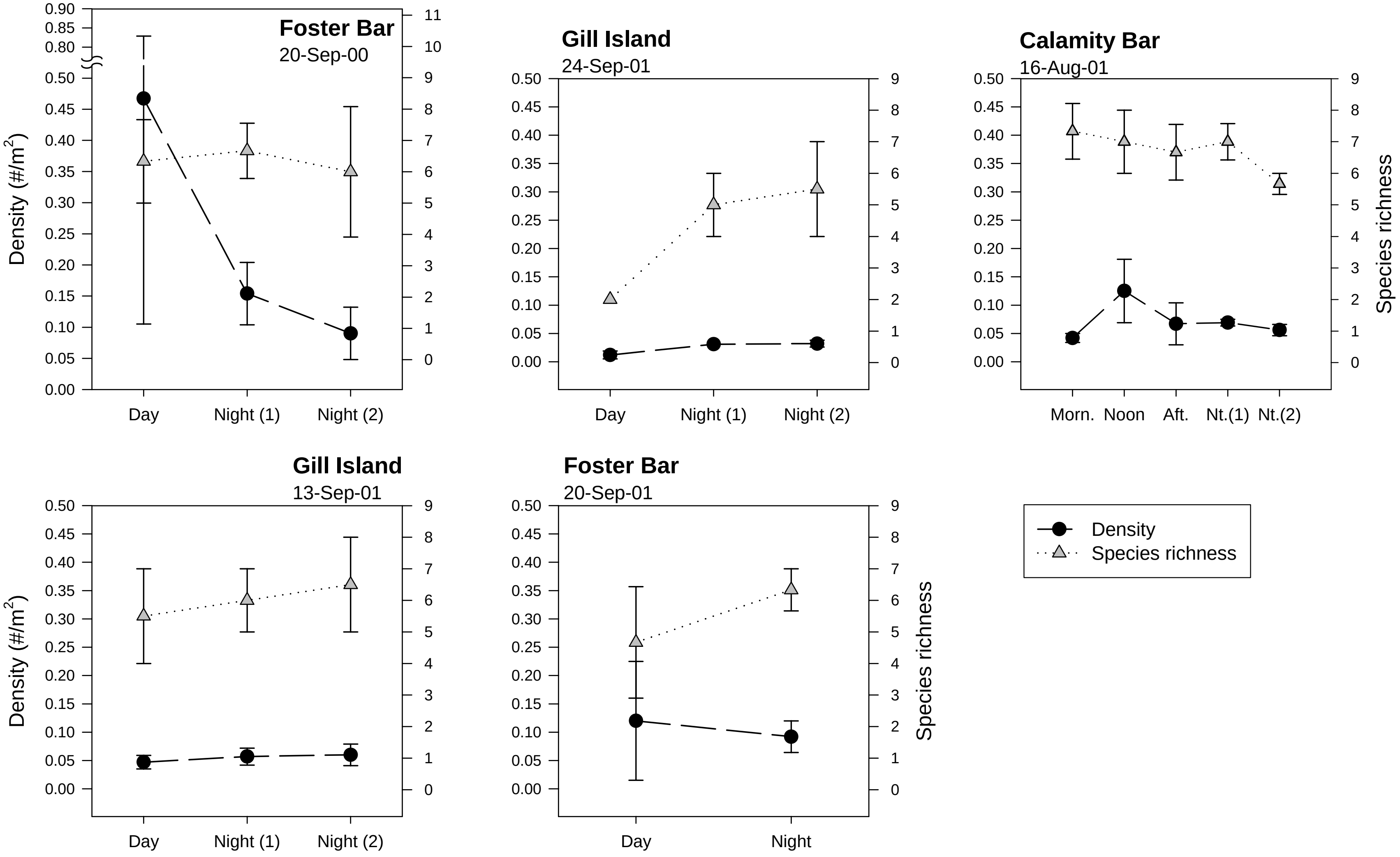


Figure B-2. Fish density and species richness in *beach seine* hauls carried out on five dates in summer when water clarity due to turbidity was low.

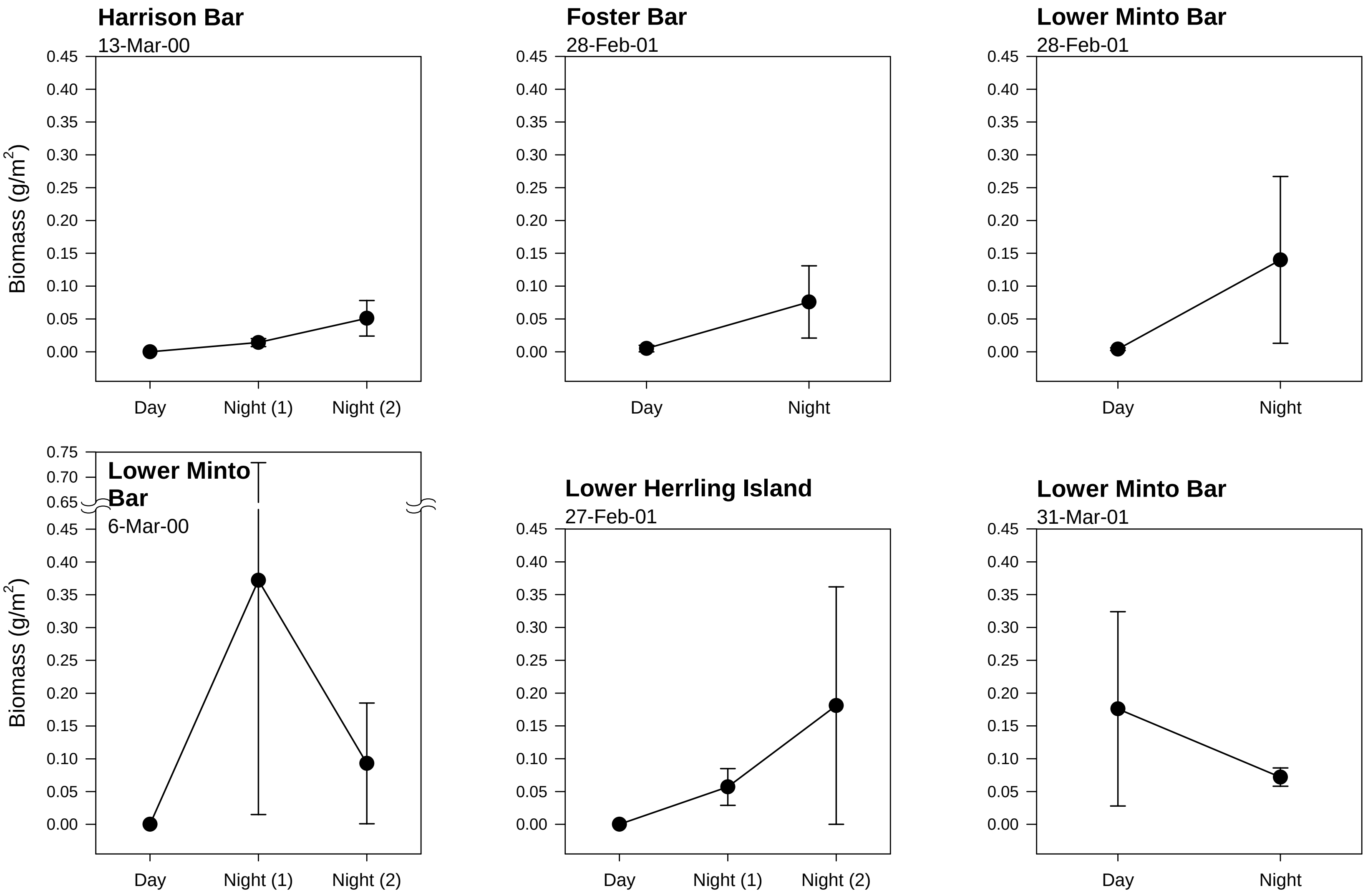


Figure B-3. Fish biomass in *beach seine* hauls that were carried out on six dates in <u>winter</u> when water clarity was high.

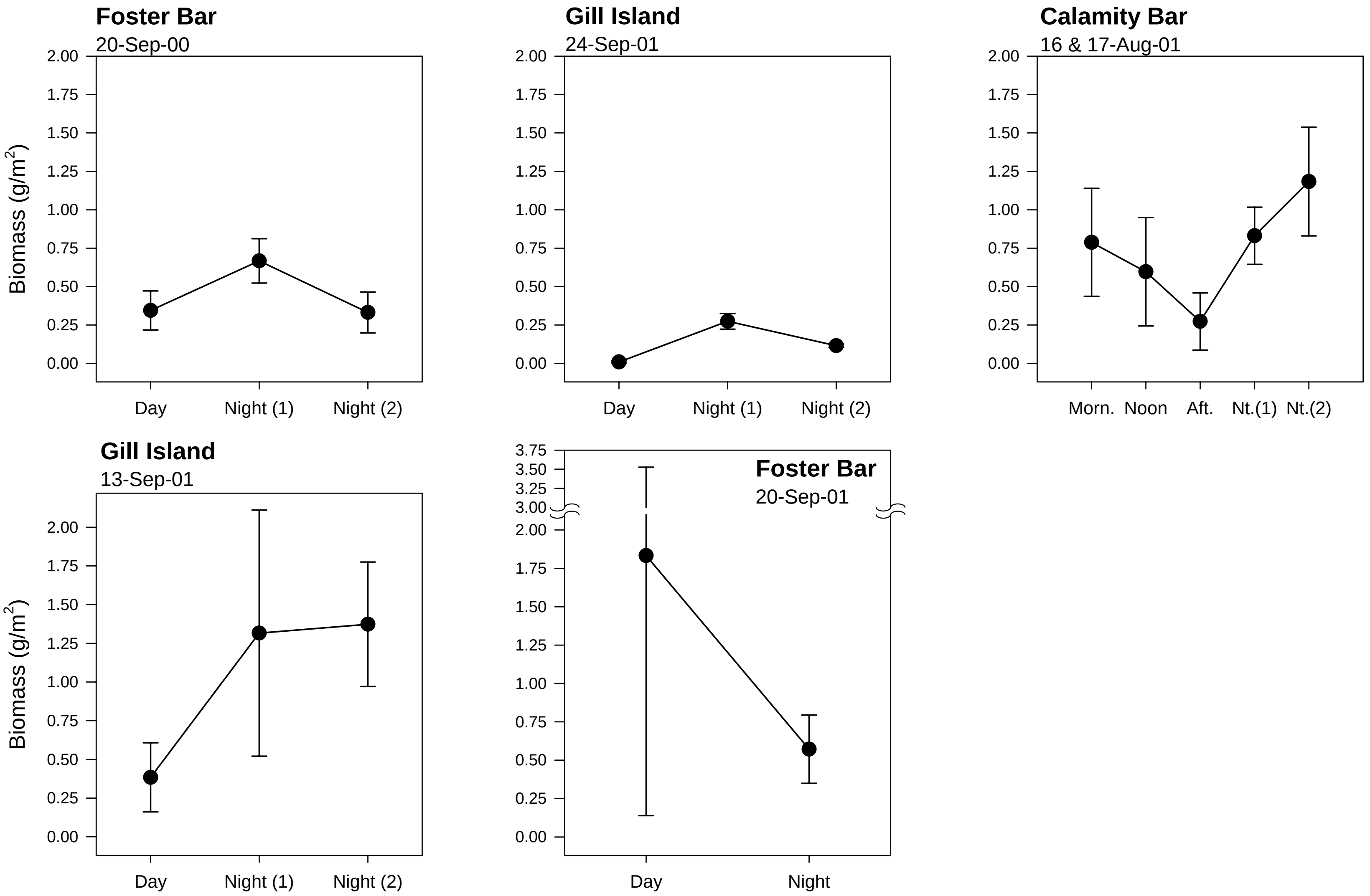


Figure B-4. Fish biomass in *beach seine* hauls that were carried out on five dates in summer when water clarity due to turbidity was low.

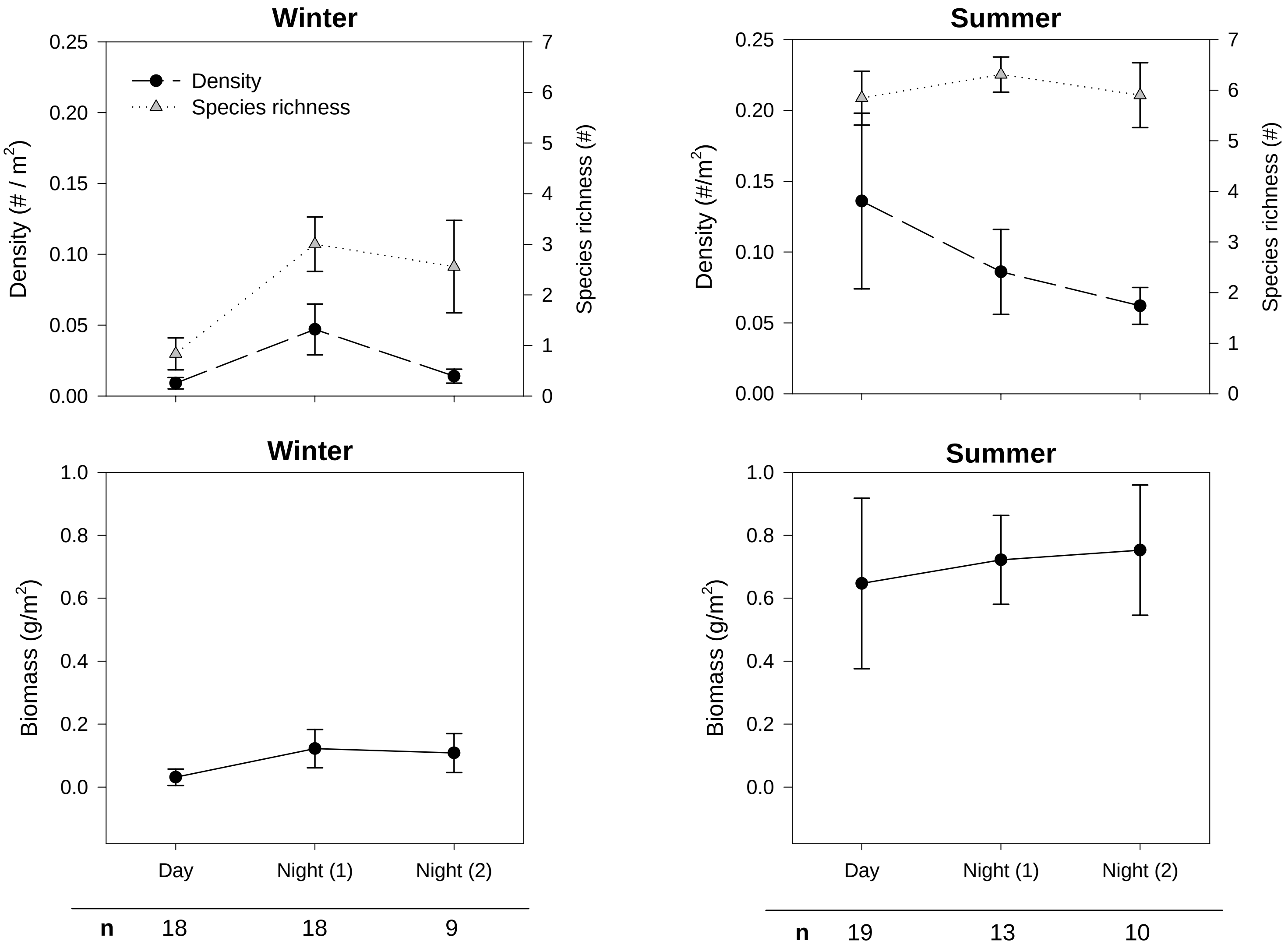


Figure B-5. Comparison of average density, species richness, and fish catch biomass in *beach seine* hauls during <u>winter</u> and <u>summer</u>.

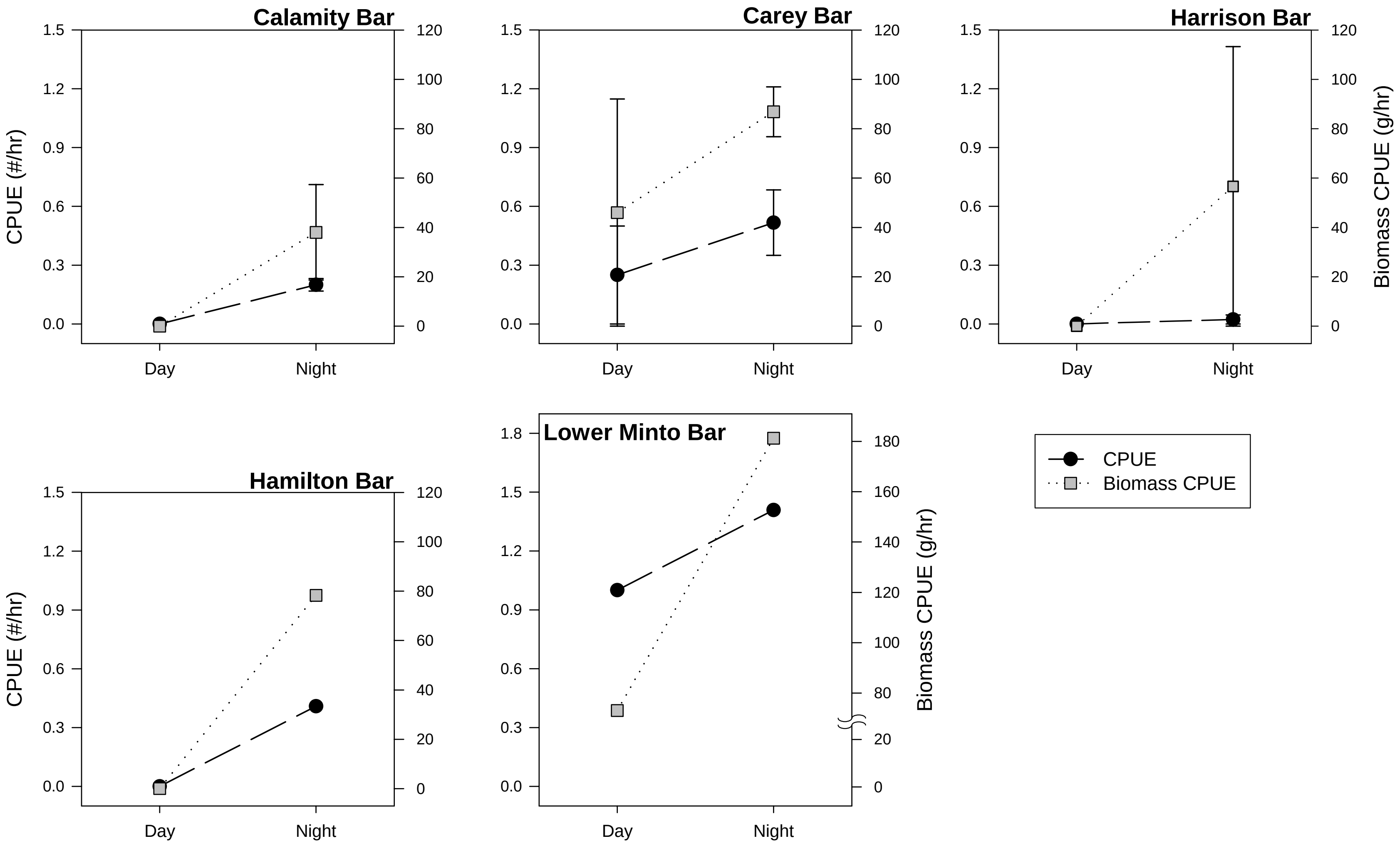


Figure B-6. Fish density and biomass in *gill net* sets that occurred on six dates in winter when water clarity was high.

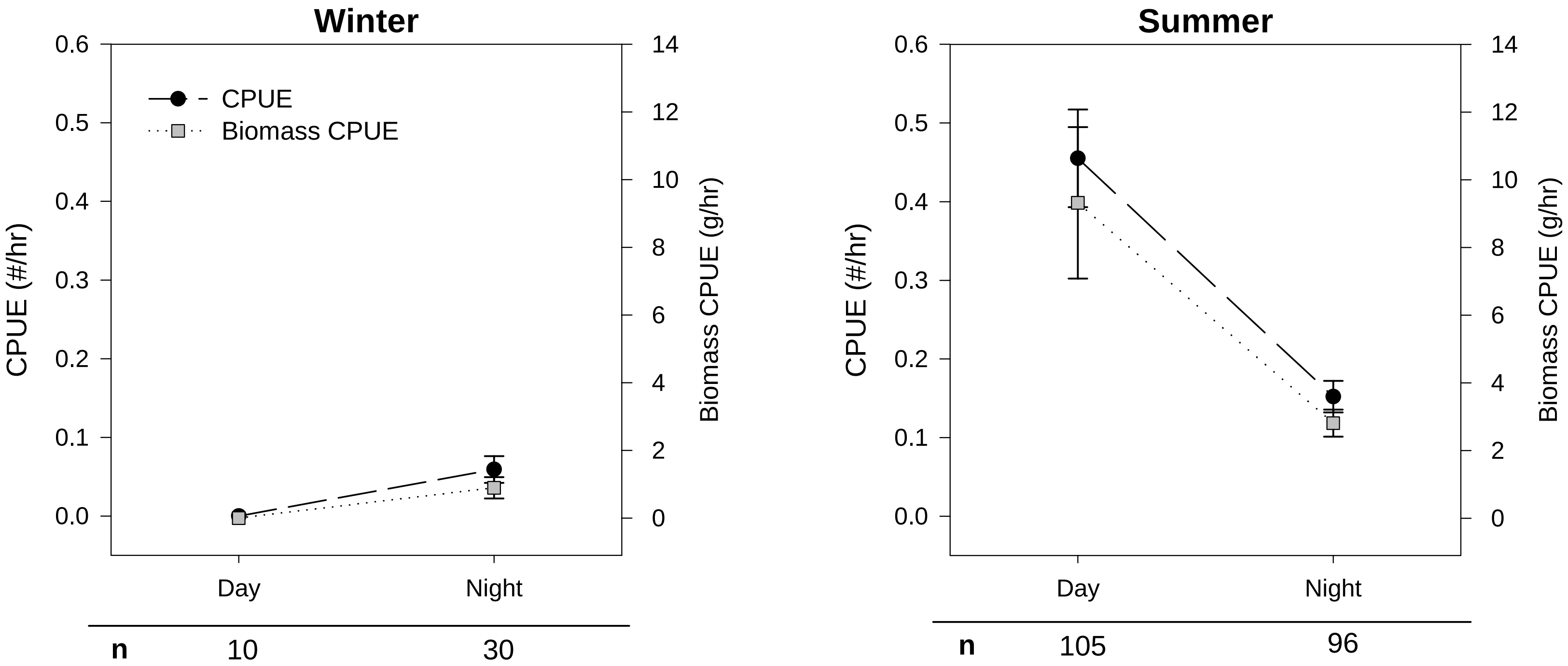


Figure B-7. Comparison of the average catch rate and fish biomass in *minnow trap* sets during <u>winter</u> and <u>summer</u>.

Appendix C:

Supplementary Results From Multivariate Analyses of Invertebrate Data

List of Tables

List of Figures

Appendix C. Supplementary Results From Multivariate Analyses of Invertebrate Data

Table C-1. Results of SIMPER analysis (square-root transformed data) indicating average abundance (untransformed) of taxa contributing most to the *dissimilarity* in community composition between sampling episodes. Data are from the main channel of the Rosedale sub-reach collected in winter months (square-root transformation applied to abundance data to derive dissimilarity contributions).

Taxon	**Average Abundance**				**% Contributed to Dissimilarity**					
	Nov-00	**Jan-01**	**Mar-00**	**Mar-01**	**N/J**	**N/M0**	**N/M1**	**J/M0**	**J/M1**	**M/M**
***s.f.* Orthocladiinae**	237.4	174.7	1029	1366	**10.3**	**31.5**	**41.4**	**32.4**	**41.1**	**21.5**
***s.f.* Chironominae**	60.3	11.4	17.2	109.0	**16.9**	**8.2**	**7.2**	**4.0**	**10.9**	**14.4**
***Baetis* sp.**	13.4	4.2	29.0	51.2	**7.0**	4.1	**4.9**	5.6	**7.6**	**6.7**
***Ephemerella* sp.**	5.5	3.1	41.1	21.4	5.6	**7.5**	**5.0**	**8.6**	**5.4**	6.1
***Rhithrogena* sp.**	2.1	0.6	29.6	24.9	<4	**6.7**	4.5	**7.4**	5.1	**6.9**
Cinygmula sp.	6.5	3.4	19.1	18.5	<4	4.0	3.8	3.9	<3	4.8
Capnia sp.	12.5	5.3	24.1	16.2	6.3	3.9	3.7	4.7	<3	4.7
Naididae	14.5	14.1	7.2	8.3	**6.5**	3.8	<3	3.3	<3	3.6
Taenionema sp.	7.5	3.3	11.8	13.6	4.2	2.6	4.5	<3	3.3	4.8
Hemerodromia sp.	1.7	3.9	4.0	6.2	5.8	<2.5	<3.5	3.1	3.3	<3
Mean Similarity	78.6	73.0	76.0	72.0	-	-	-	-	-	-
Mean Dissimilarity	-	-	-	-	29.4	38.3	38.3	47.0	47.5	29.0

Bolded text highlights the four taxa contributing most to dissimilarity *between pairs*.

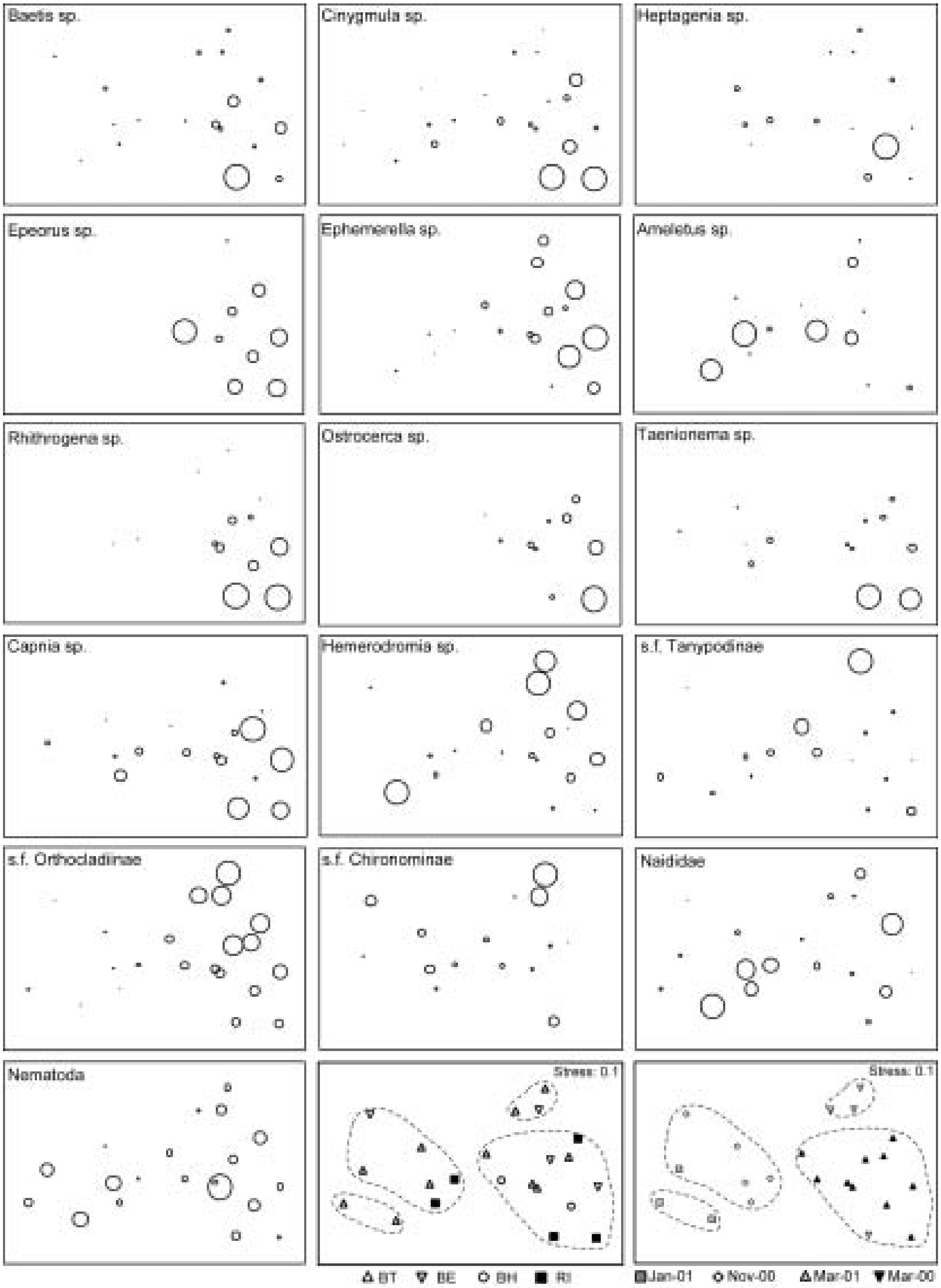


Figure C-1. Bubble plots corresponding to the untransformed abundances of select invertebrate taxa that contributed significantly to the dissimilarity between months. Plots are overlaid on the MDS ordination of samples collected from the main channel of the Rosedale sub-reach in winter months. Symbols are sized proportionately to individual taxon abundance.

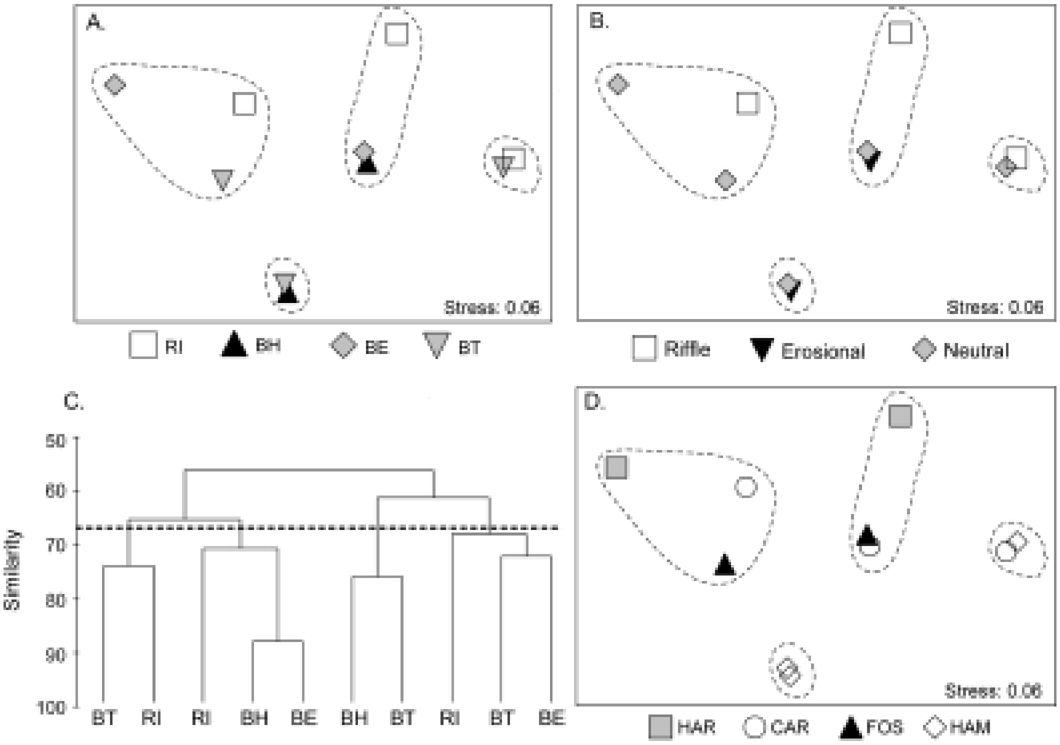


Figure C-2. MDS ordination of main channel samples (square-root transformed) collected in September 1999 from the Rosedale sub-reach. The legend lists sites from downstream (left) to upstream (right).

Table C-2. Results of 1-way ANOSIM comparing sub-reaches based on main channel samples collected in September 1999.

Comparison	**Possible Permutations**	**# Cases ≥ Observed**$^{+}$	***R*-Statistic**	***p*-value**
Global			0.15	0.04*
Chilliwack, Rosedale	>999	267	0.06	0.27
Chilliwack, Cheam	462	150	0.03	0.33
Rosedale, Cheam	>999	8	0.32	0.009*

* α=0.1, adjusted for the exploratory nature of the analysis.

\+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

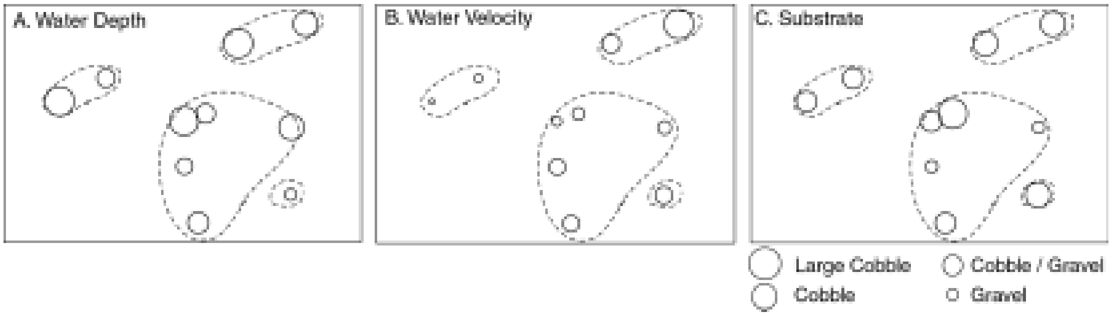


Figure C-3. Bubble plots representing untransformed values of water depth, velocity, and substrate size that correspond to invertebrate samples plotted in ordination space. Substrate symbols are sized according to size class categories. Depth and velocity were measured on a continuous scale and symbols are sized proportionately to the individual variables. Samples were collected in March 2000 from the main channel of the Rosedale sub-reach.

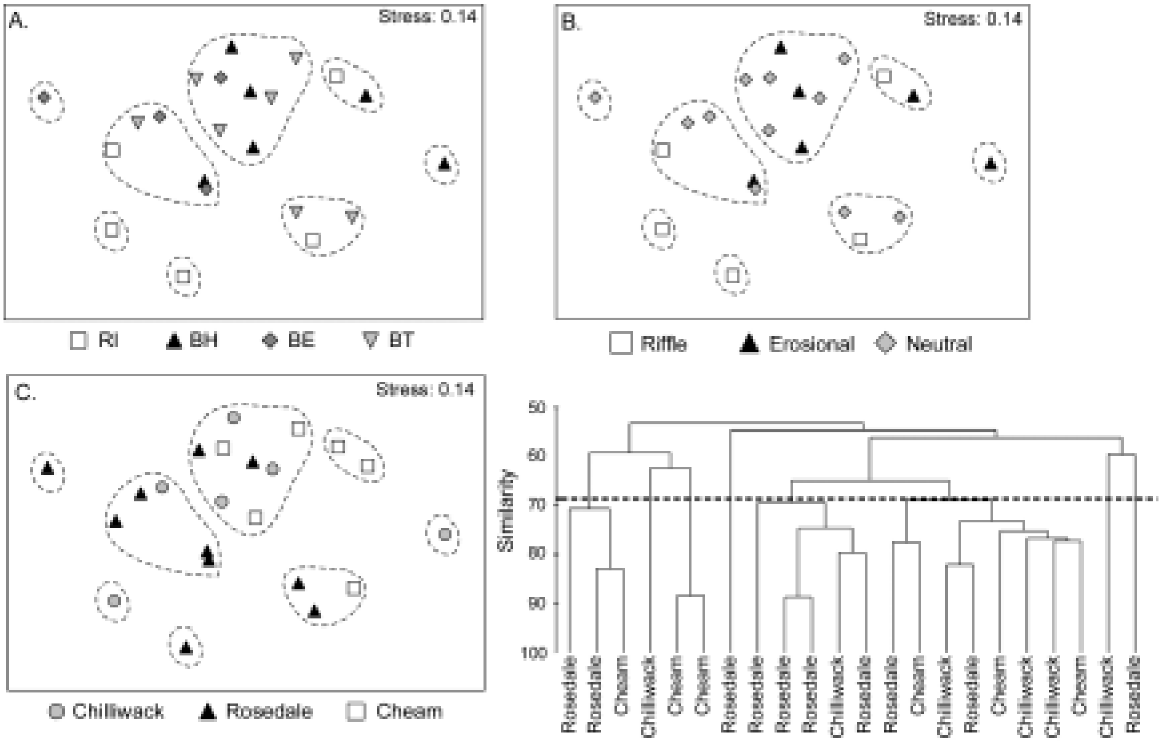


Figure C-4. MDS ordination of invertebrate samples (square-root transformed) collected in September 1999 from the main channel of all sub-reaches. Legend lists sub-reaches from downstream (left) to upstream (right).

Table C-3. Results of SIMPER analysis indicating average abundance (untransformed) of taxa contributing most to the *dissimilarity* in community composition between sampling. Data are from the main channel of the Chilliwack, Rosedale, and Cheam sub-reaches and collected in September 1999 (square-root transformation applied to abundance data to derive dissimilarity contributions).

Taxon	**Average Abundance**			**% Contributed to Dissimilarity**		
	Chilliwack	**Rosedale**	**Cheam**	**Ck/R**	**Ck/Cm**	**R/Cm**
***s.f.* Orthocladiinae**	46.9	65.3	43.4	**22.5**	**17.0**	**24.1**
***Ephemerella* sp.**	5.1	13.1	1.6	**13.3**	**12.9**	**13.3**
***Rhithrogena* sp.**	2.6	9.8	0.6	**14.5**	**9.7**	**16.8**
Naididae	1.5	1.7	2.1	4.8	**9.5**	7.5
***Hydropsyche* sp.**	1.8	3.8	0	**9.3**	5.5	**8.5**
Simulium sp.	5.9	0	0	6.4	7.8	-
s.f. Tanypodinae	1.2	1.5	0.7	5.8	7.4	6.4
s.f. Chironominae	0.1	0	0.1	0.8	1.5	0.6
Baetis sp.	0.3	1.2	0.6	5.7	6.1	5.2
Dicranota sp.	0.4	0.7	0.1	3.6	4.1	4.6
Nematoda	0.3	0.2	0.2	3.1	3.9	2.7
Mean Similarity	57.9	62.0	64.6	-	-	-
Mean Dissimilarity	-	-	-	40.5	39.2	43.4

Bolded text highlights the four taxa contributing most to dissimilarity between pairs.

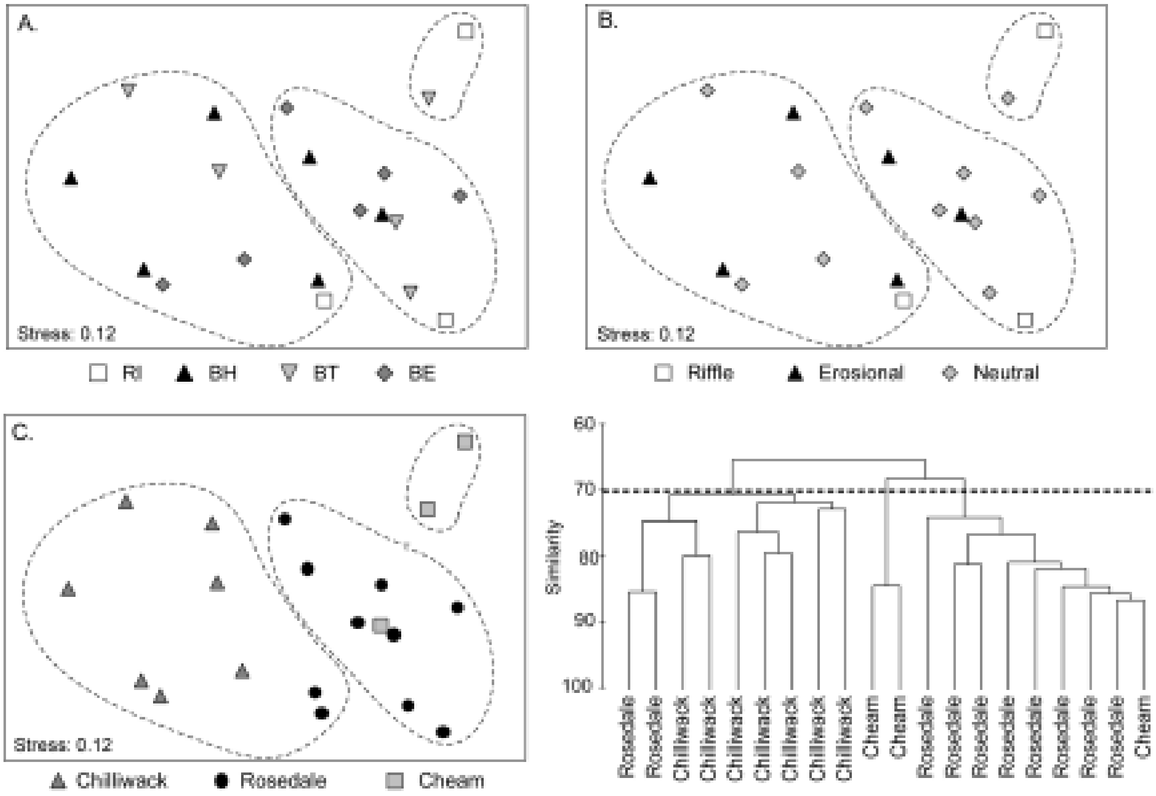


Figure C-5. MDS ordination of main channel samples (square-root transformed) collected in March 2000 from all sub-reaches. Legend lists sub-reaches from downstream (left) to upstream (right).

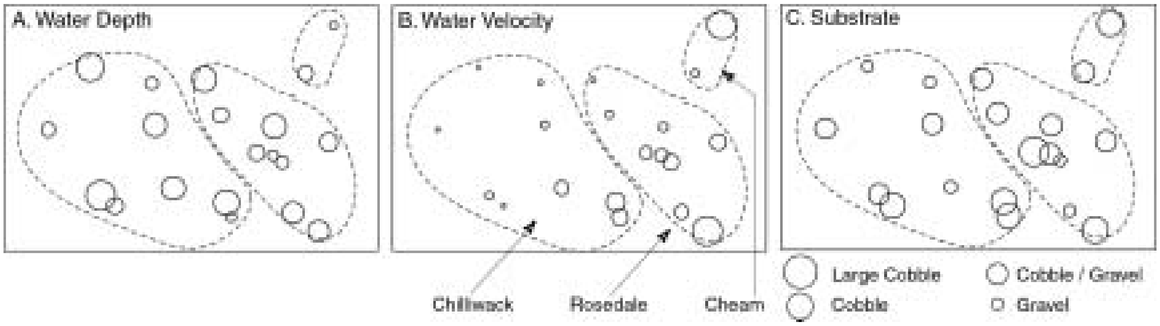


Figure C-6. Bubble plots representing values of water depth, velocity, and substrate size that correspond to invertebrate samples plotted in ordination space (refer to **Figure C-5**). Substrate symbols are scaled according to size categories. Depth and velocity symbols are scaled proportionately to a continuous scale of measurement. Dotted outlines indicate sampling episodes clustered at 70% similarity. Samples were collected in September 1999 from the main channel in all sub-reaches.

Table C-4. Results of SIMPER analysis indicating average abundance (untransformed) of taxa contributing most to the *dissimilarity* in community composition between sub-reaches. Data are from the main channel of the Chilliwack, Rosedale, and Cheam sub-reaches and collected in March 2000 (square-root transformation applied to abundance data to derive dissimilarity contributions).

Taxon	**Average Abundance**			**% Contributed to Dissimilarity**		
	Chilliwack	**Rosedale**	**Cheam**	**Ck/R**	**Ck/Cm**	**R/Cm**
***s.f.* Orthocladiinae**	573.2	1028.6	867.0	**14.5**	**10.0**	**13.4**
***s.f.* Chironominae**	41.9	17.2	6.0	**6.5**	7.1	**6.3**
***Ameletus* sp.**	49.3	0.6	1.6	**9.9**	**9.1**	2.2
***Ephemerella* sp.**	79.7	41.1	7.3	5.7	**11.0**	**10.4**
***Rhithrogena* sp.**	9.6	29.6	10.4	5.7	3.4	**7.0**
Naididae	40.1	7.2	1.2	**7.0**	**9.6**	4.8
Cinygmula sp.	6.5	19.1	16.8	3.6	3.8	5.3
Capnia sp.	4.5	24.1	14.4	4.9	3.2	4.1
Taenionema sp.	1.3	11.8	11.2	4.1	4.5	2.9
Hydropsyche sp.	12.0	7.3	0.8	4.1	3.9	3.5
Ostrocerca sp.	1.3	7.0	1.9	3.2	<2	3.9
Baetis sp.	11.1	29.0	20.2	4.6	3.0	5.1
Tubificidae	15.7	3.4	0.4	4.5	5.8	3.1
Hemerodromia sp.	7.8	4.0	1.2	2.0	3.4	3.3
s.f. Tanypodinae	12.1	4.0	1.3	2.9	4.8	3.9
Mean Similarity	72.3	76.0	75.6	-	-	-
Mean Dissimilarity	-	-	-	34.3	40.0	27.7

Bolded text highlights the four taxa contributing most to dissimilarity between pairs.

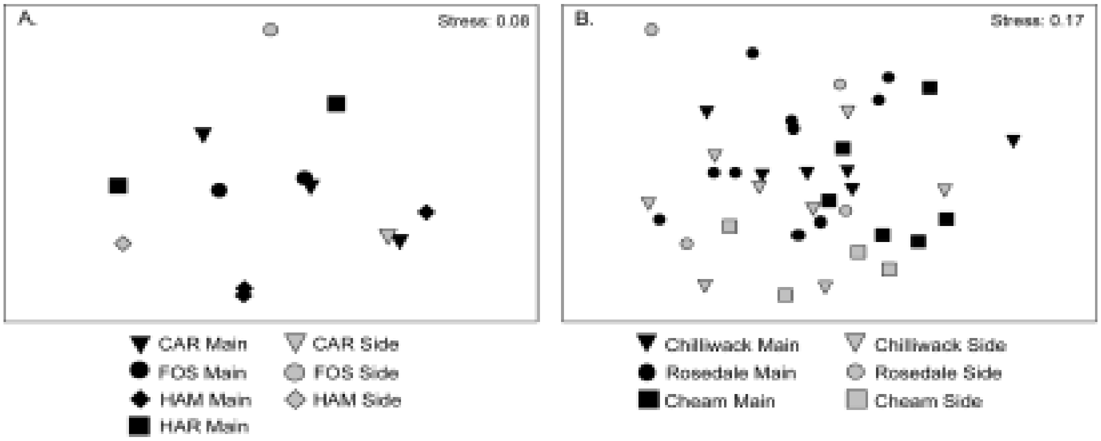


Figure C-7. MDS ordination of samples (square-root transformed) collected from in September 1999 from the main channel and side channels of the Rosedale sub-reach (A), and from all sub-reaches combined (B).

Appendix D

Length-Frequency Plots of Fish Species

Appendix D:

Length-Frequency Plots of Fish Species

List of Figures

Appendix D. Length-Frequency Plots of Fish Species

Length-frequency plots are presented for all species with a well-represented range of size classes. Of the 25 species captured during this study, 15 species met this criterion and 10 species did not warrant length-frequency plots: brassy minnow, bridgelip sucker, bull char, coastrange sculpin, coho salmon, Dolly Varden, lamprey species, marine stickleback, pink salmon, and white sturgeon. Only those fish collected by beach seine were included in plotted data because the seine mesh size was comparatively non-selective relative to both the gill net and minnow trap.

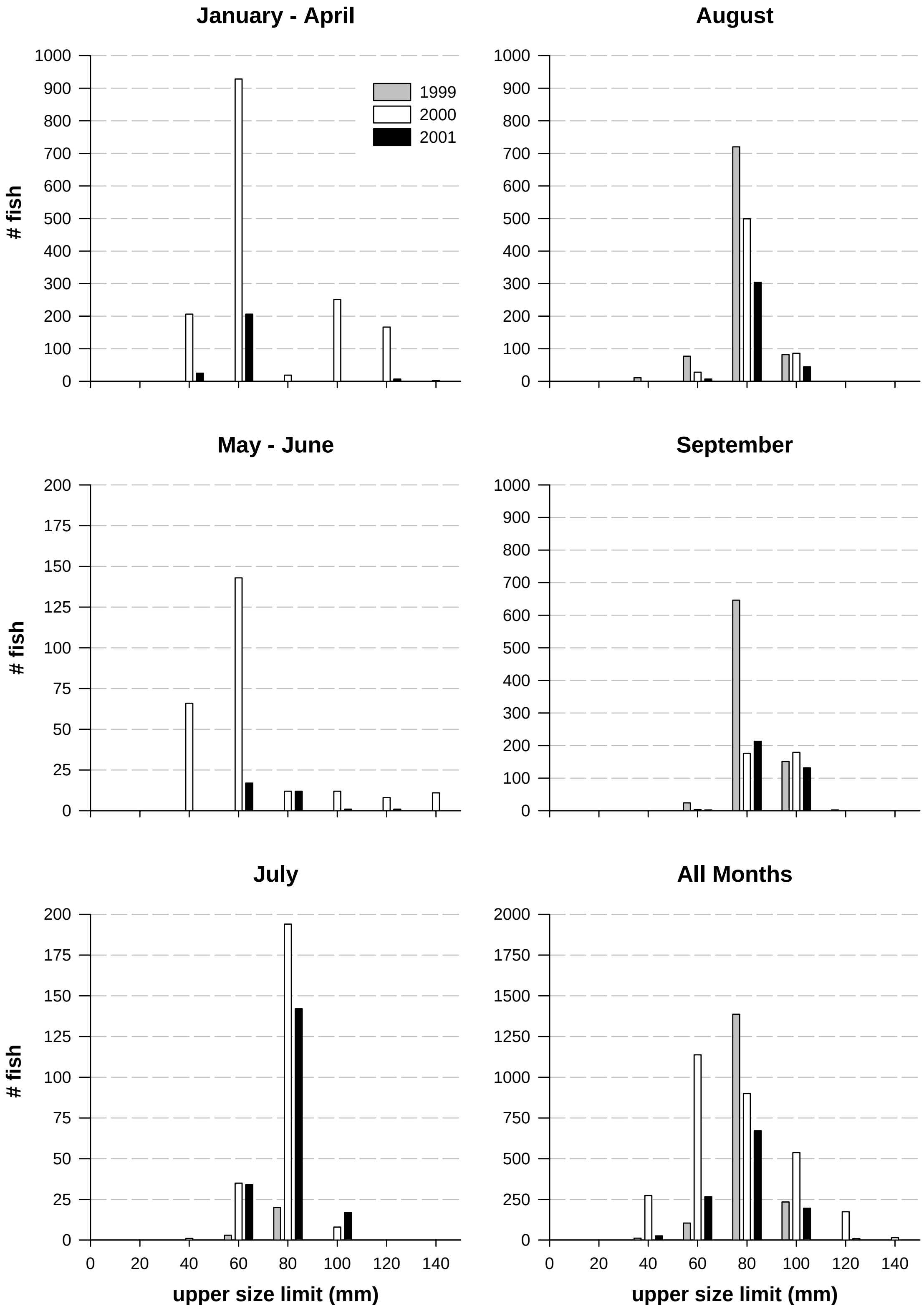


Figure D-1. Size distribution of chinook salmon (*Oncorhynchus tshawytscha*) collected by beach seine, 1999 to 2001.

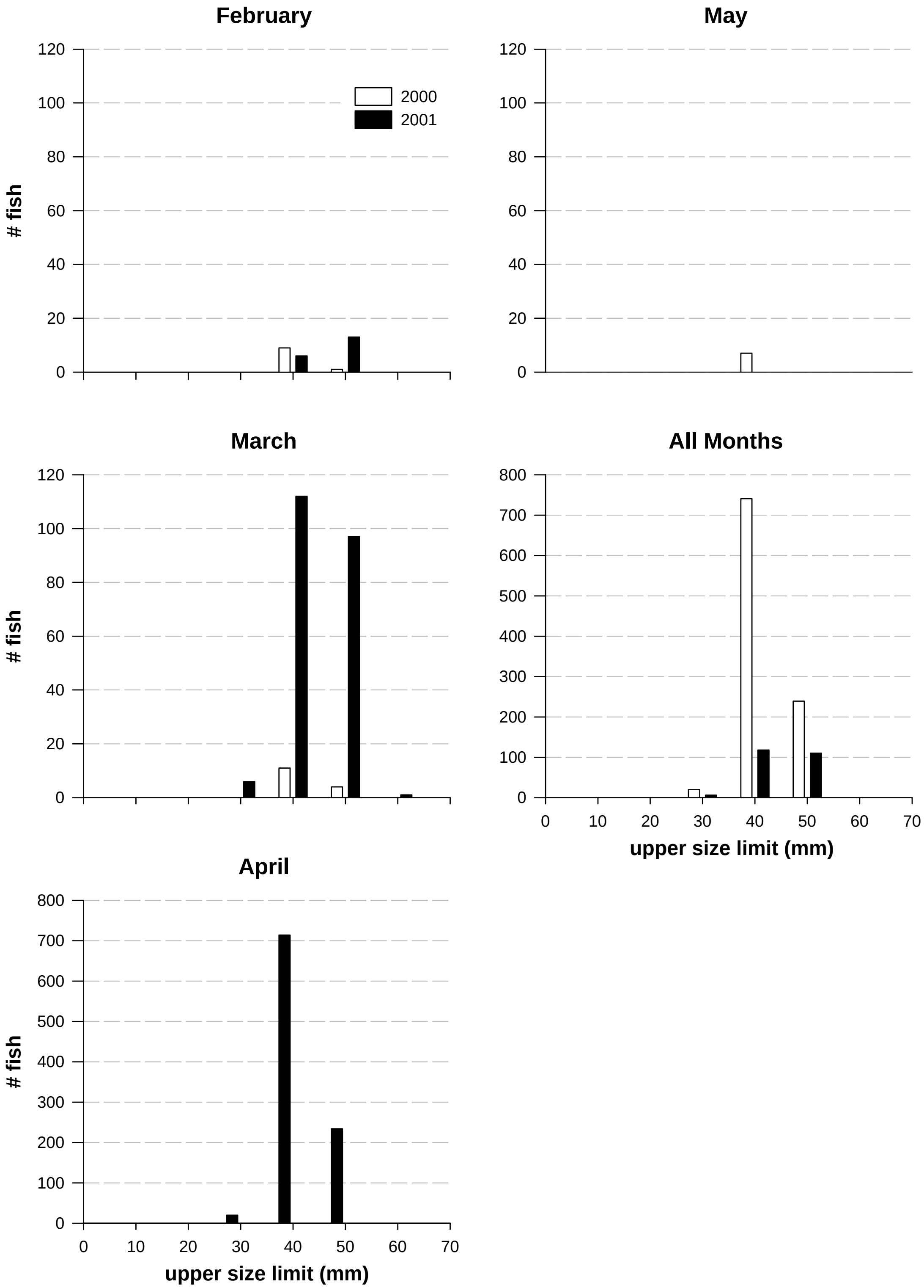


Figure D-2. Size distribution of chum salmon (*Oncorhynchus keta*) collected by beach seine, 1999 to 2001.

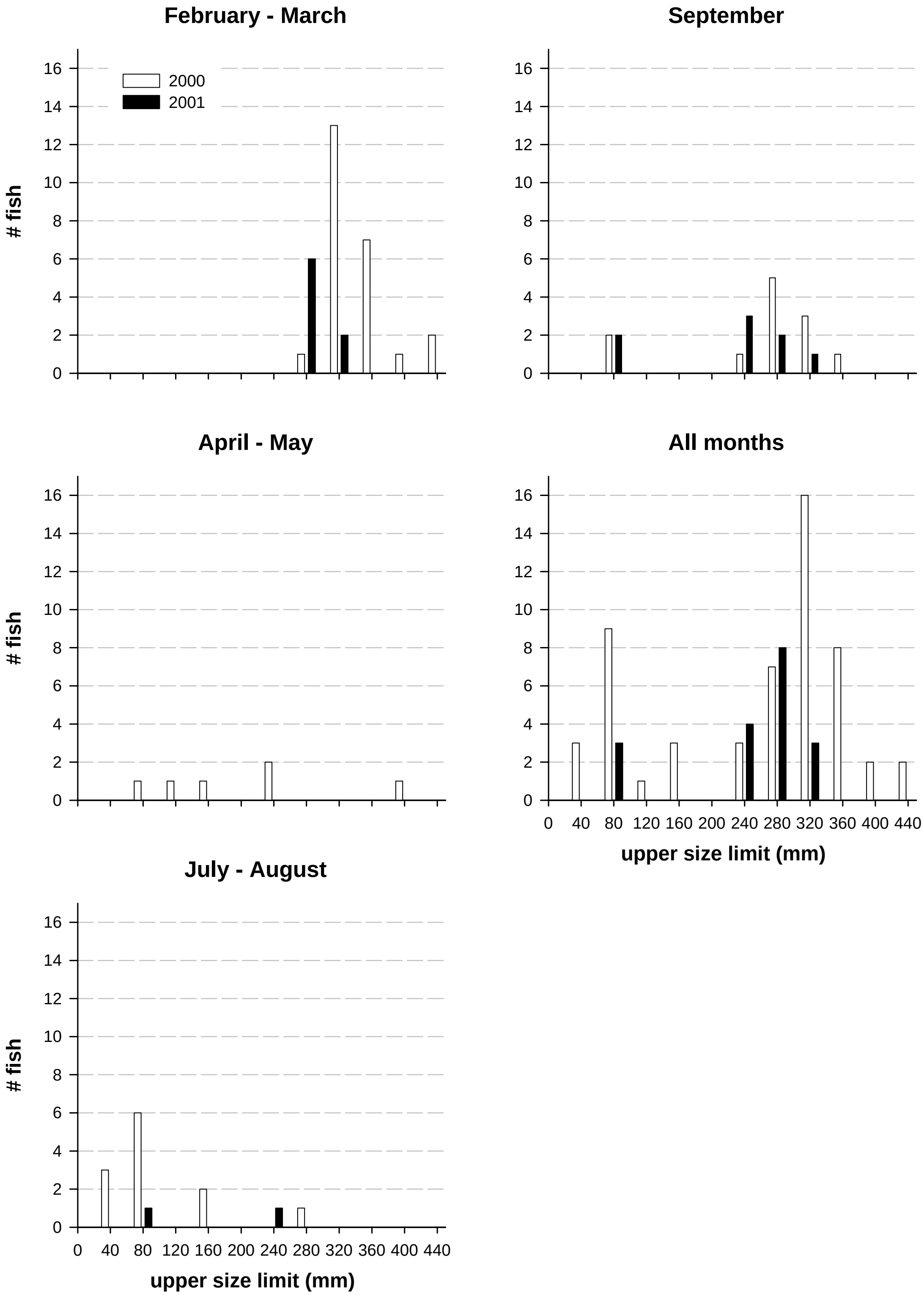


Figure D-3. Size distribution of cutthroat trout (*Oncorhynchus clarki*) collected by beach seine, 1999 to 2001.

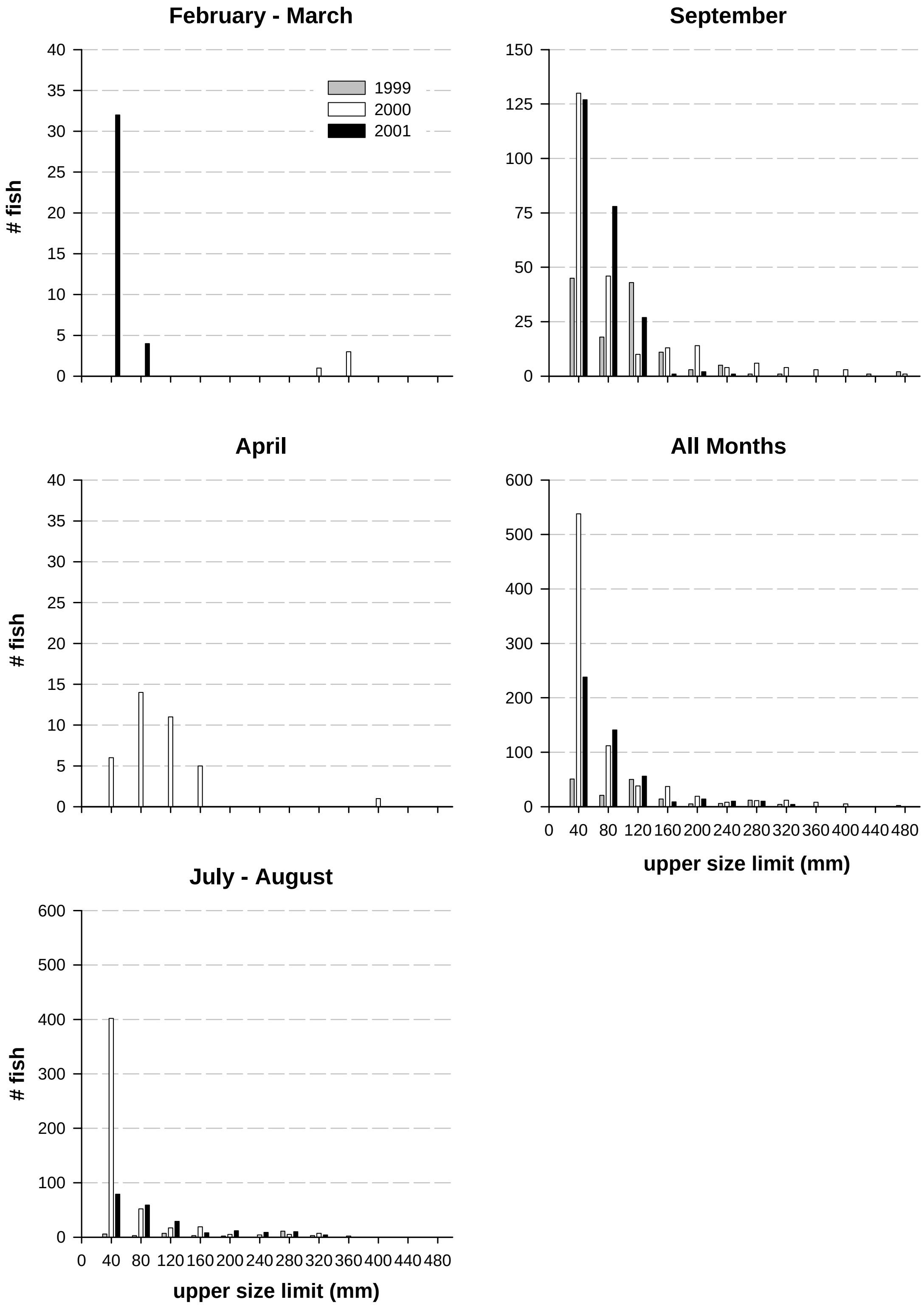


Figure D-4. Size distribution of largescale sucker (*Catostomus macrocheilus*) collected by beach seine, 1999 to 2001.

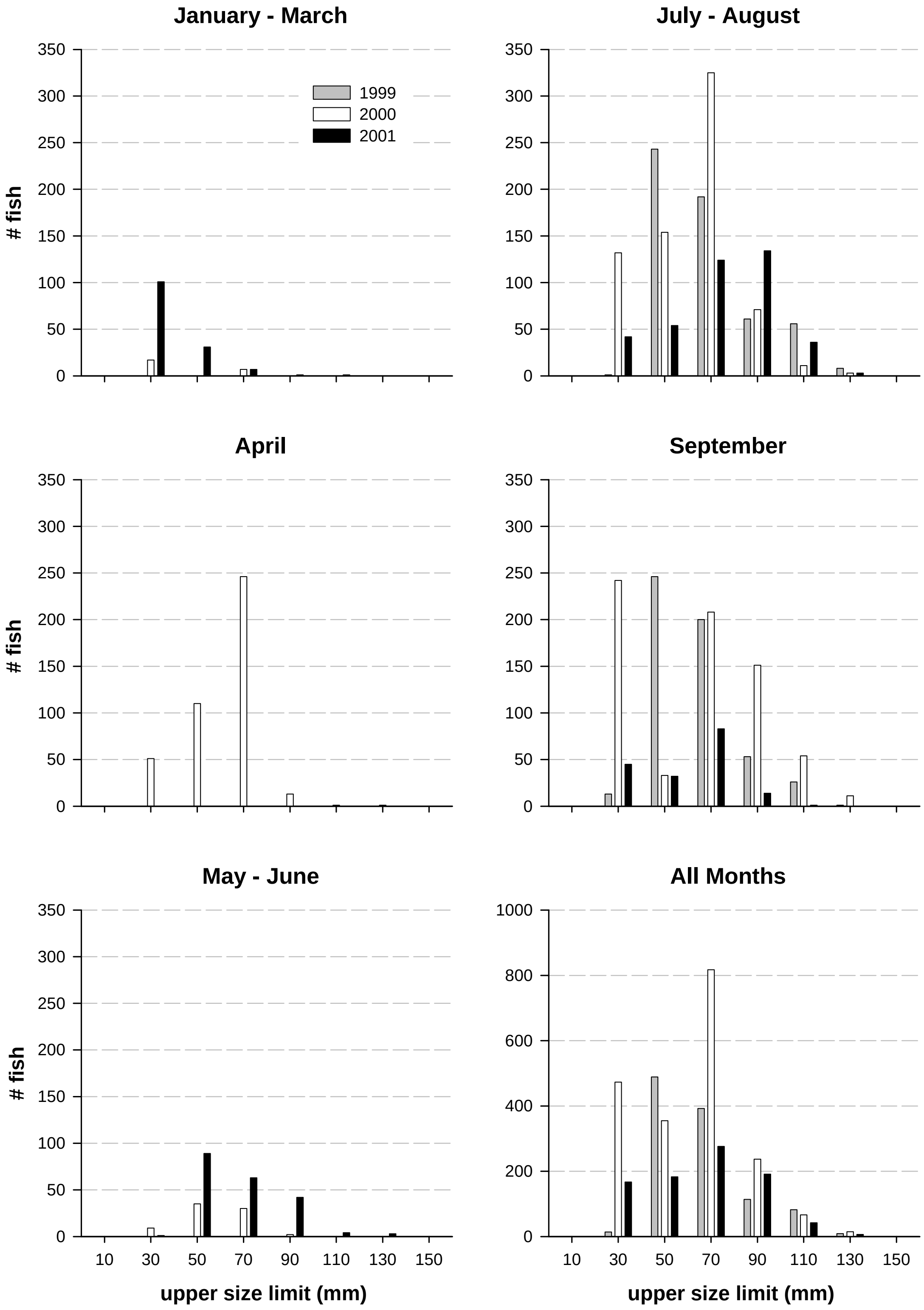


Figure D-5. Size distribution of leopard dace (*Rhinichthys falcatus*) collected by beach seine, 1999 to 2001.

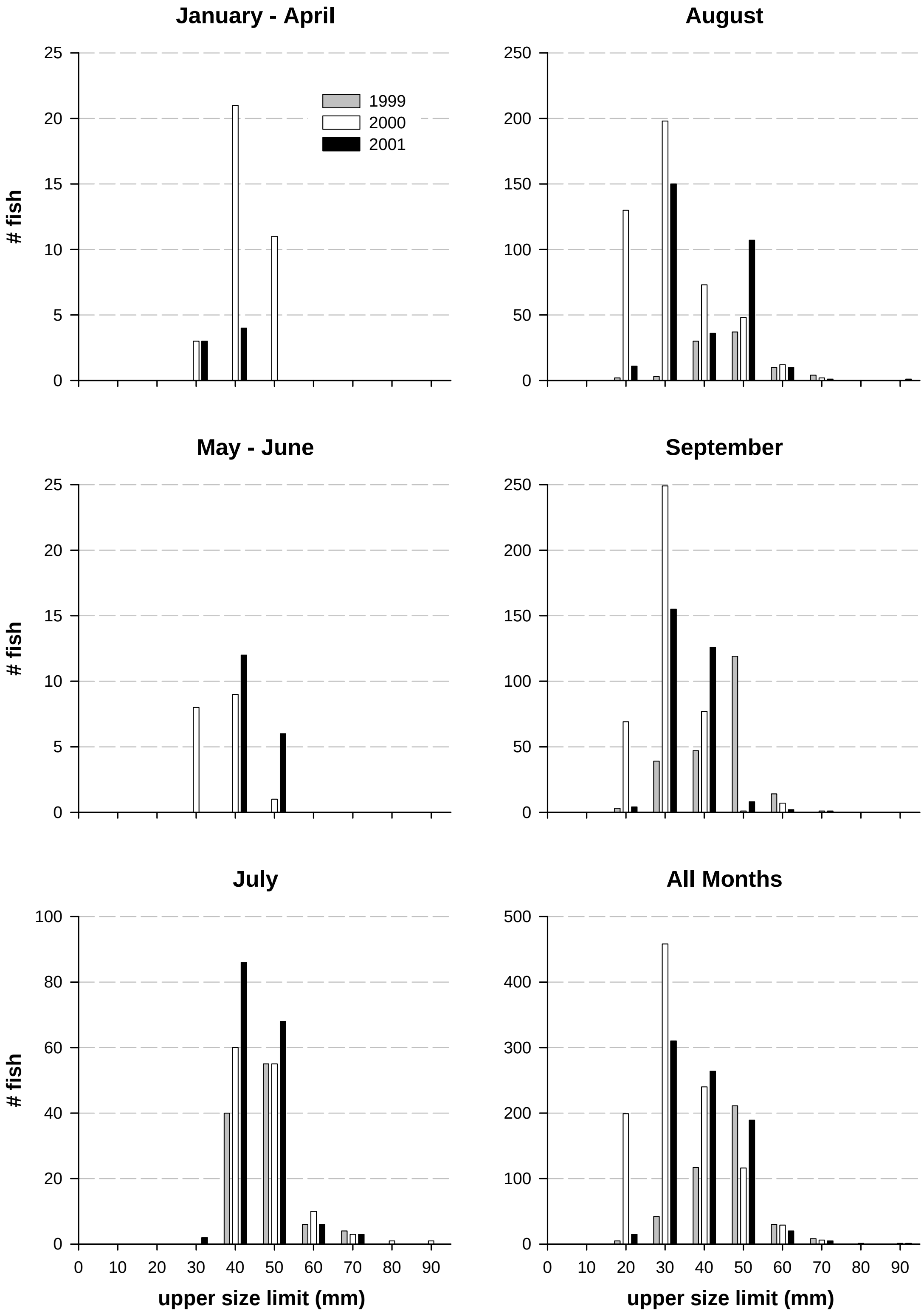


Figure D-6. Size distribution of longnose dace (*Rhinichthys cataractae*) collected by beach seine, 1999 to 2001.

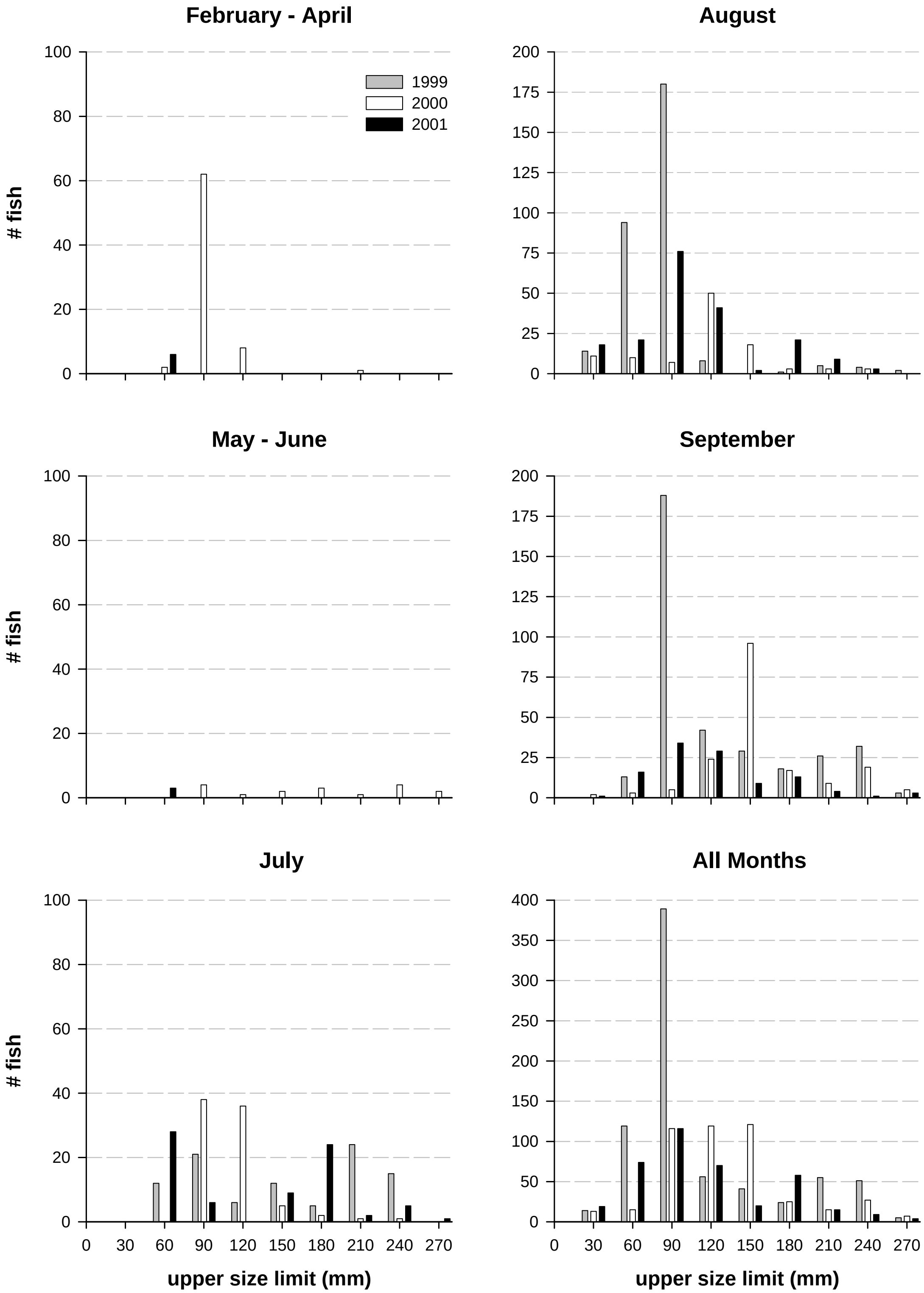


Figure D-7. Size distribution of mountain sucker (*Catostomus platyrhynchus*) collected by beach seine, 1999 to 2001.

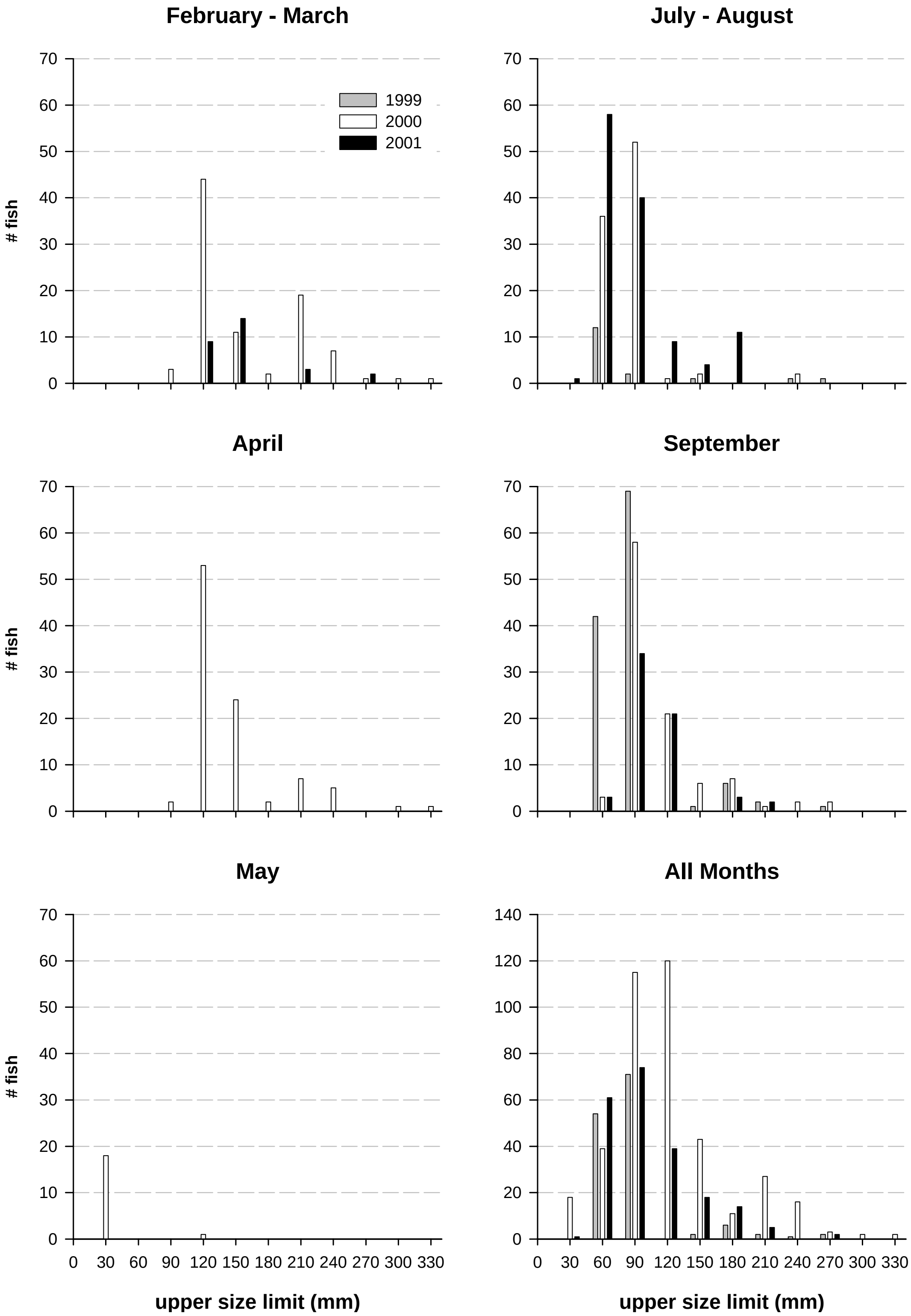


Figure D-8. Size distribution of mountain whitefish (*Prosopium williamsoni*) collected by beach seine, 1999 to 2001.

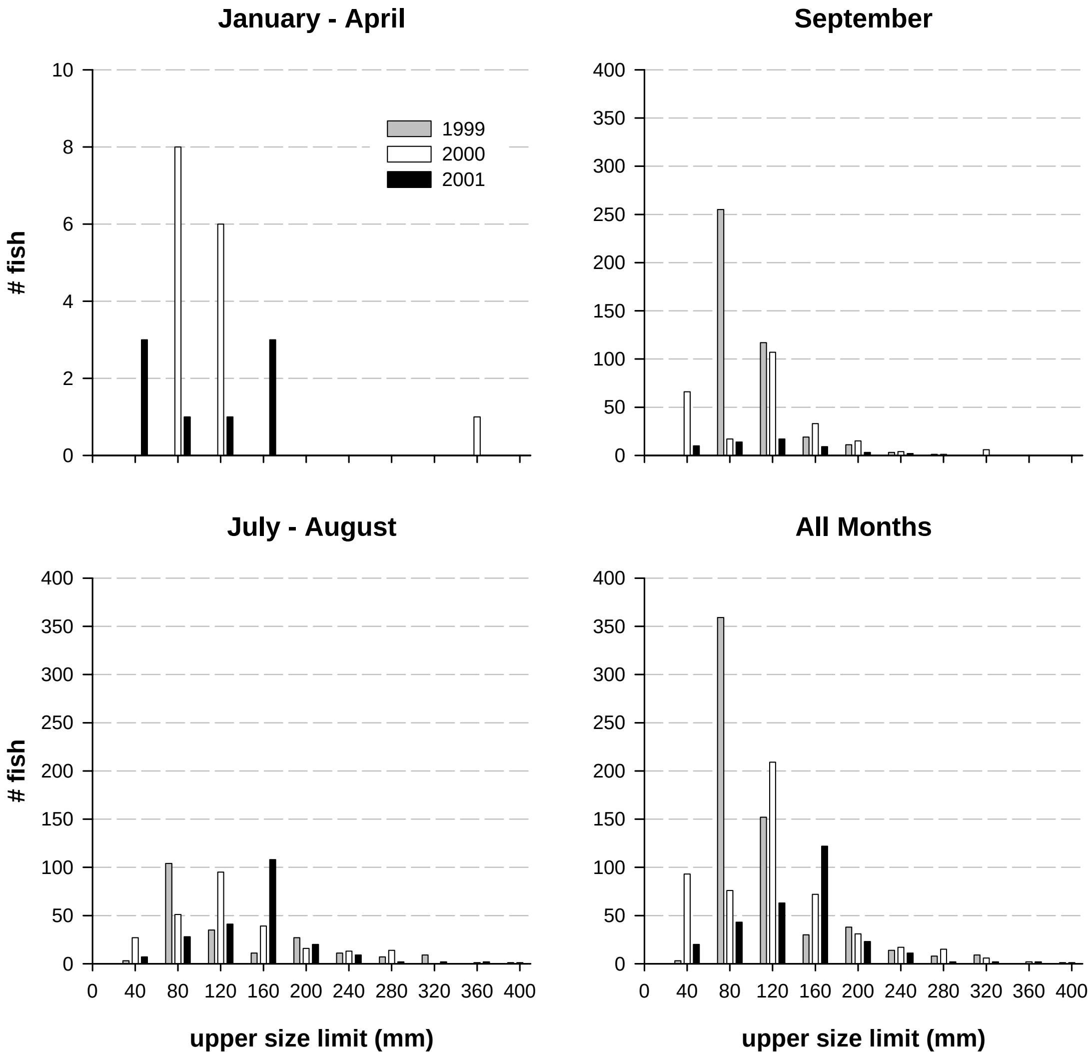


Figure D-9. Size distribution of northern pikeminnow (*Ptychocheilus oregonensis*) collected by beach seine, 1999 to 2001.

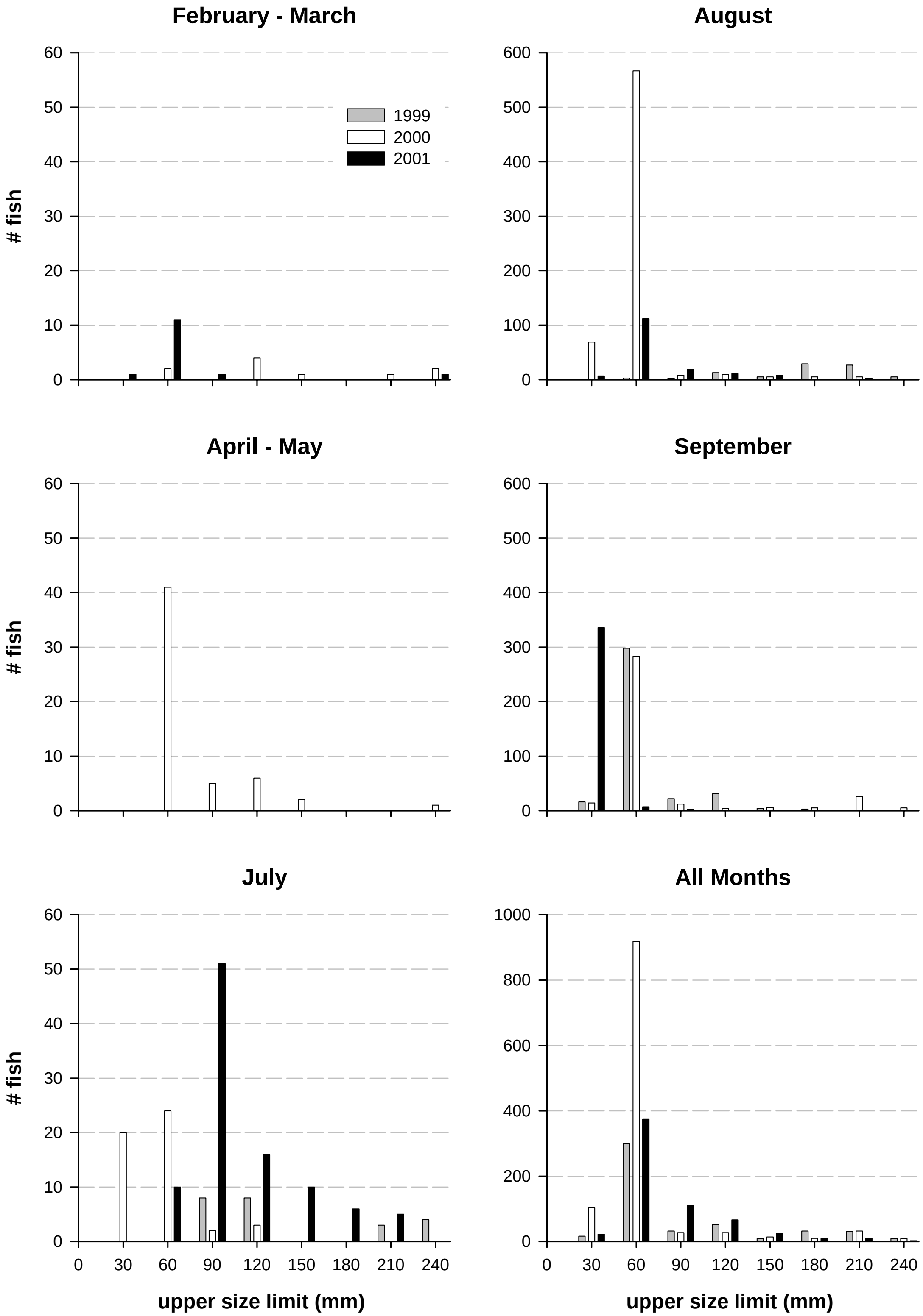


Figure D-10. Size distribution of peamouth chub (*Mylocheilus caurinus*) collected by beach seine, 1999 to 2001.

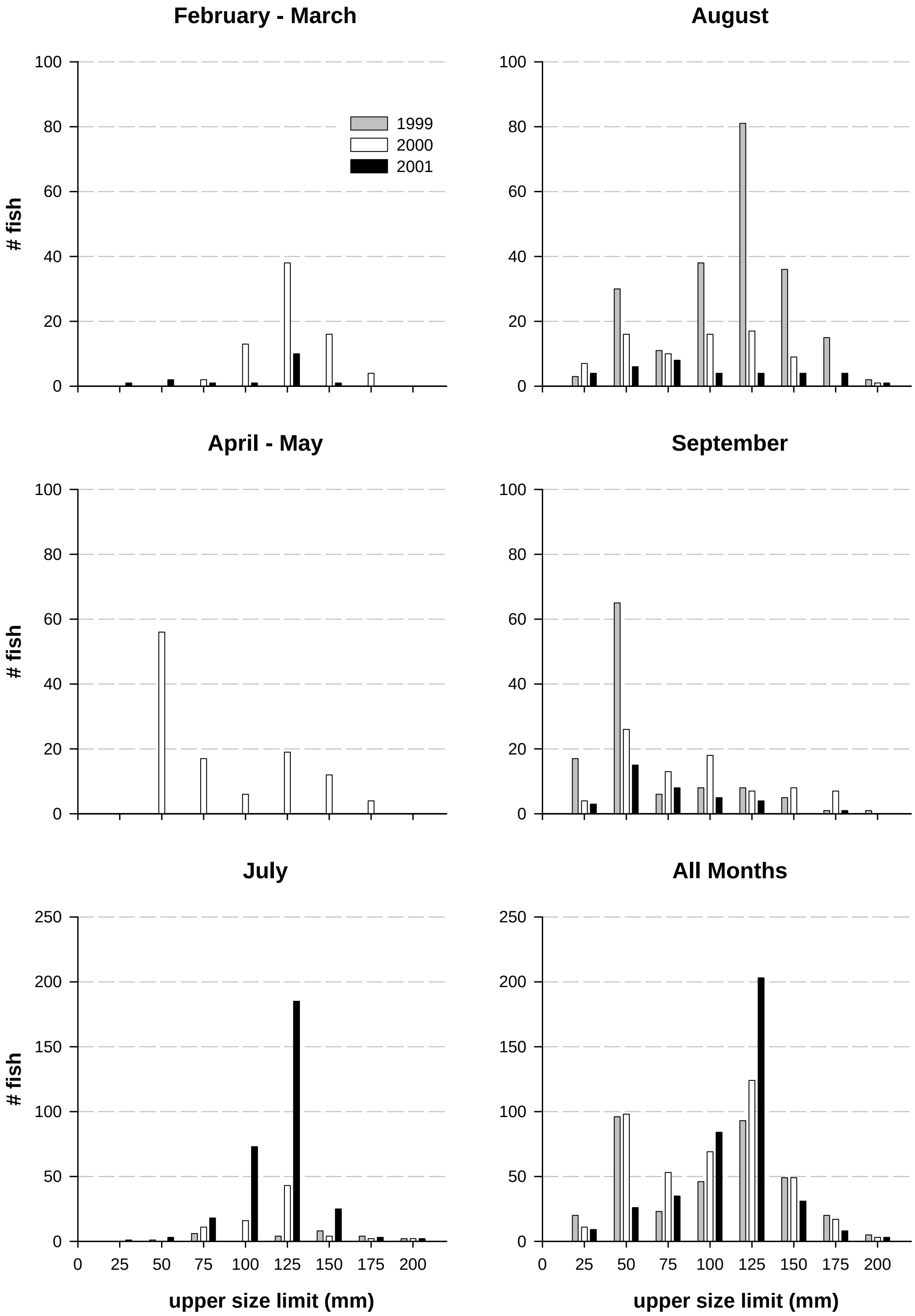


Figure D-11. Size distribution of prickly sculpin (*Cottus asper*) collected by beach seine, 1999 to 2001.

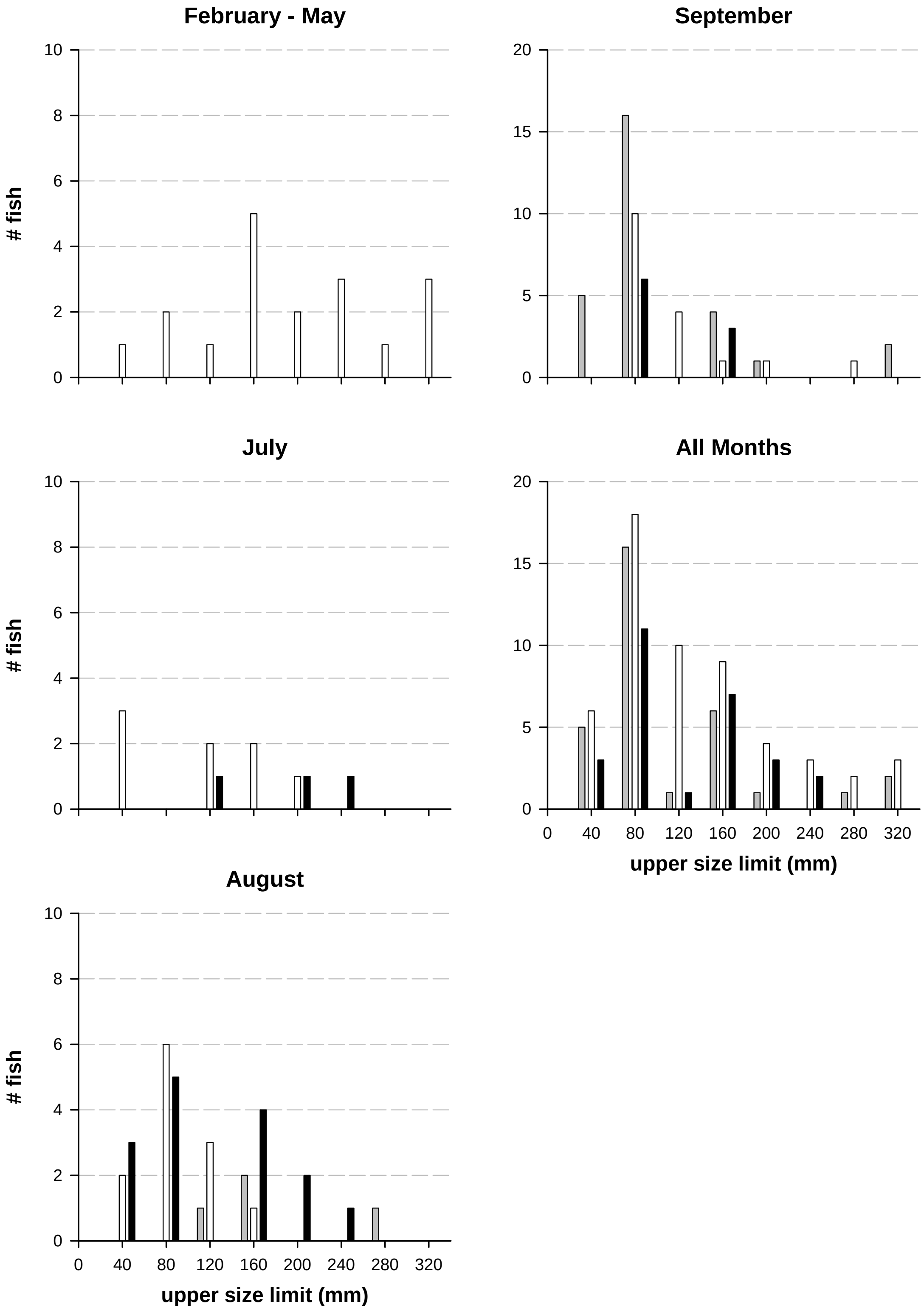


Figure D-12. Size distribution of rainbow trout (*Oncorhynchus mykiss*) collected by beach seine, 1999 to 2001.

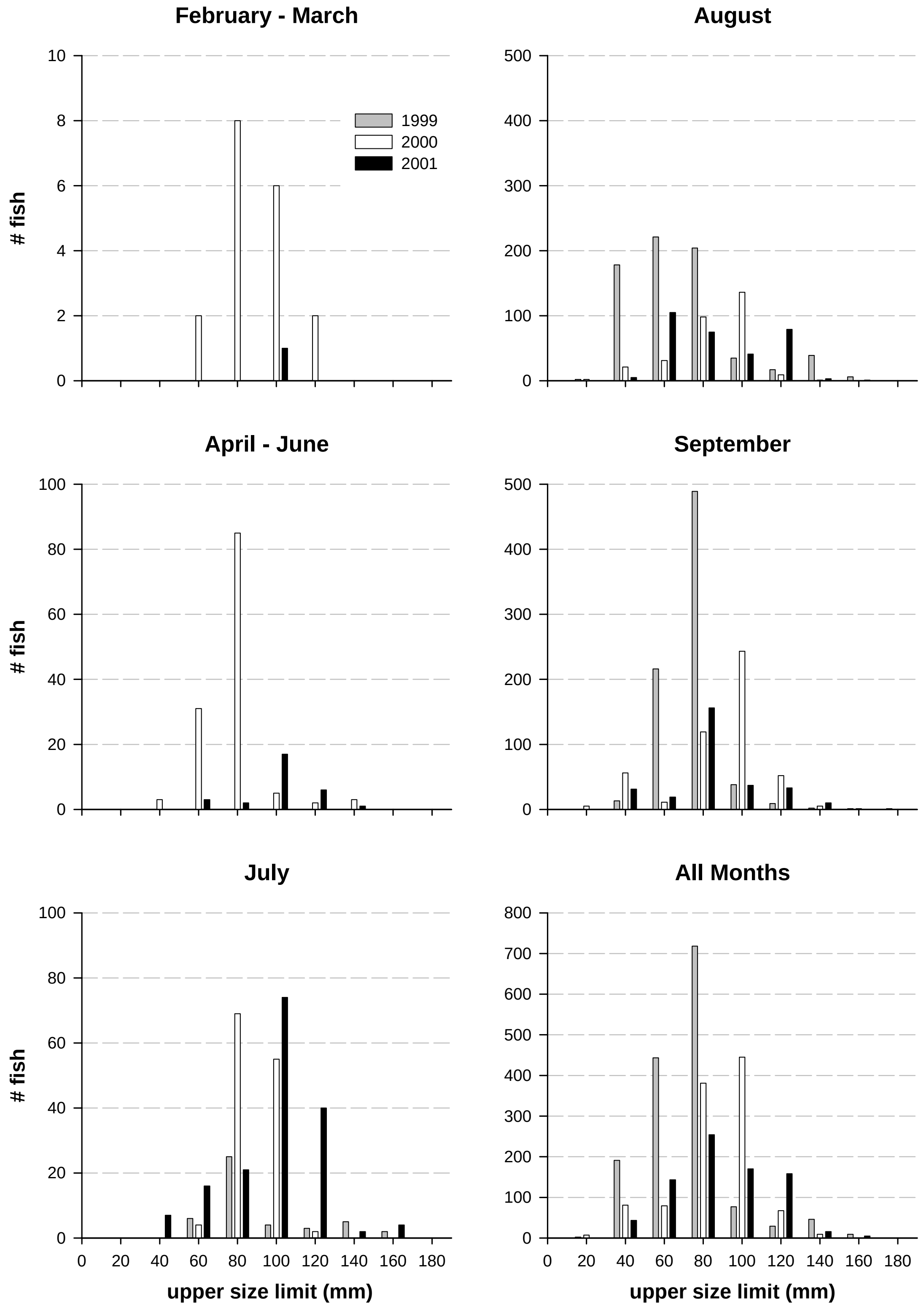


Figure D-13. Size distribution of redside shiner (*Richardsonius balteatus*) collected by beach seine, 1999 to 2001.

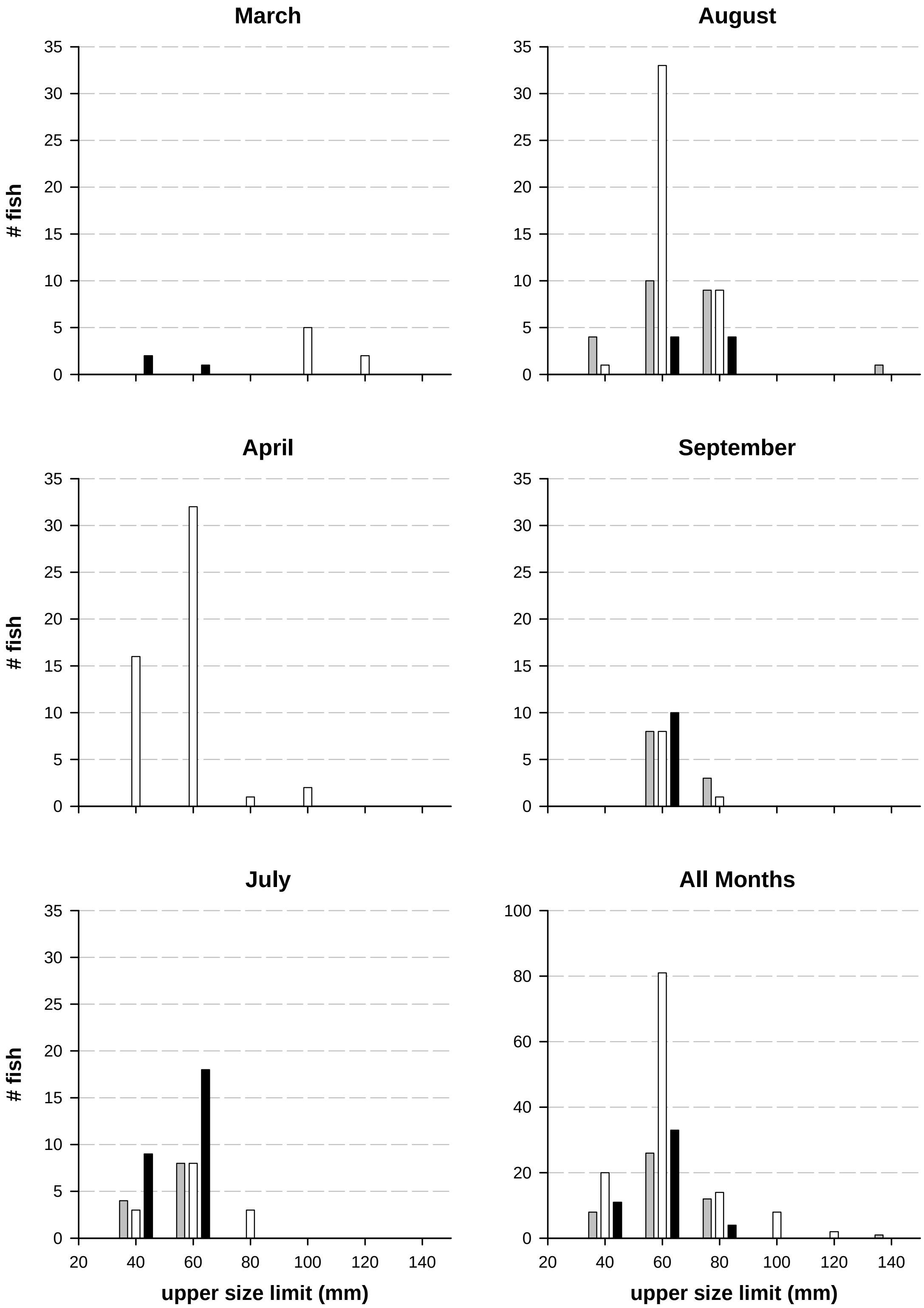


Figure D-14. Size distribution of sockeye salmon (*Oncorhynchus nerka*) collected by beach seine, 1999 to 2001.

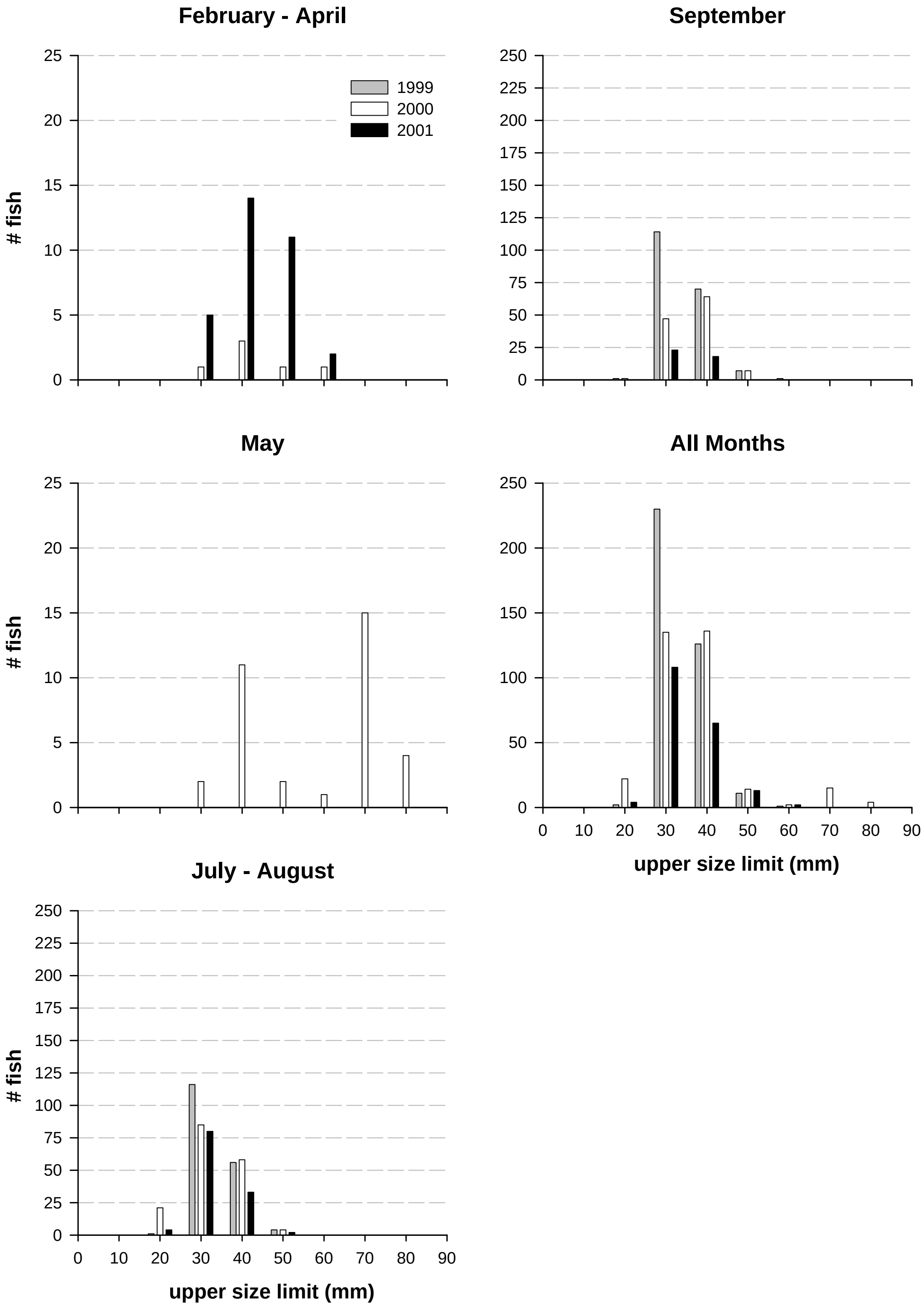


Figure D-15. Size distribution of threespine stickleback (*Gasterosteus aculeatus*) collected by beach seine, 1999 to 2001.

Appendix E:

DNA Analysis of Juvenile Chinook Salmon

Table E-1. The percentage (%) of fish spawned in upstream tributaries of the Fraser Basin that were collected in August 2000 in each of three sub-reaches of the gravel reach. DNA analysis was carried out by Mr. J. Irvine, Fraser River Chinook/Coho Program Head, Pacific Biological Station, Nanaimo, British Columbia, V9R 5K6. The analysis was based on a 50-stock Fraser baseline dataset with 13 microsatellite loci.

Natal Tributary	Chilliwack	Rosedale	Cheam
Sample Size	78	125	74
Harrison	0.0	0.0	0.0
Birkenhead	0.0	0.0	0.0
W_Chilliwack	0.0	0.0	0.0
Stave	0.0	0.0	0.0
Bonaparte	0.0	0.0	0.0
Coldwater	0.0	0.0	0.0
Deadman	0.0	0.9	0.0
Nicola	1.9	0.0	1.5
Spius	0.0	1.7	0.0
Blackwater	0.0	0.0	0.0
Bridge	3.5	3.8	2.7
Chilcotin mix	1.6	1.8	2.1
Chilko	0.1	0.7	4.0
Cottonwood	0.0	0.0	0.0
Elkin	2.2	1.6	0.0
Endako	0.0	0.0	0.0
Horsefly	0.0	0.0	0.0
L.Chilcotin	0.0	0.0	2.7
Portage	0.0	0.0	0.0
Quesnel	6.1	6.6	0.8
Taseko	0.0	0.0	4.0
U. Chilcotin	0.0	0.0	0.0
Clearwater (Thompson)	4.3	6.9	12.1
Finn	0.0	0.0	0.0
Louis	0.0	0.0	0.0

Natal Tributary	**Chilliwack**	**Rosedale**	**Cheam**
Mahood	0.0	0.0	0.0
Raft	6.6	14.1	5.2
Eagle	0.0	0.0	0.0
L. Shuswap	0.0	0.3	0.0
U. Adams Transp	0.0	0.0	0.0
Little River	0.0	0.0	0.0
Lower Adams	0.0	0.0	0.5
M. Shuswap	0.0	0.0	0.0
Salmon River @ SA	0.0	0.0	0.0
South Thompson	0.0	2.0	0.0
Bowron	0.0	0.0	0.0
Dome	0.0	0.0	0.0
TeteJeune	0.0	0.0	0.0
Willow	0.0	0.0	0.0
Fontoniko	0.0	0.0	0.0
Goat	0.0	0.0	1.4
Holmes	3.7	0.0	0.0
Horsey	0.0	0.0	0.0
Indianpoint	0.0	0.0	0.0
MacGregor	1.0	0.8	0.0
Nechako	28.2	29.7	15.7
Salmon River @ PG	0.0	1.0	0.0
Slim	0.0	0.0	0.0
Stuart	40.9	28.3	46.0
Swift	0.0	0.0	1.4
TOTAL	100.0	100.0	100.0

Appendix F:

Supplementary Results From Multivariate Analyses of Fish Data

List of Tables

List of Figures

Appendix F – Supplementary Results From Multivariate Analyses of Fish Data

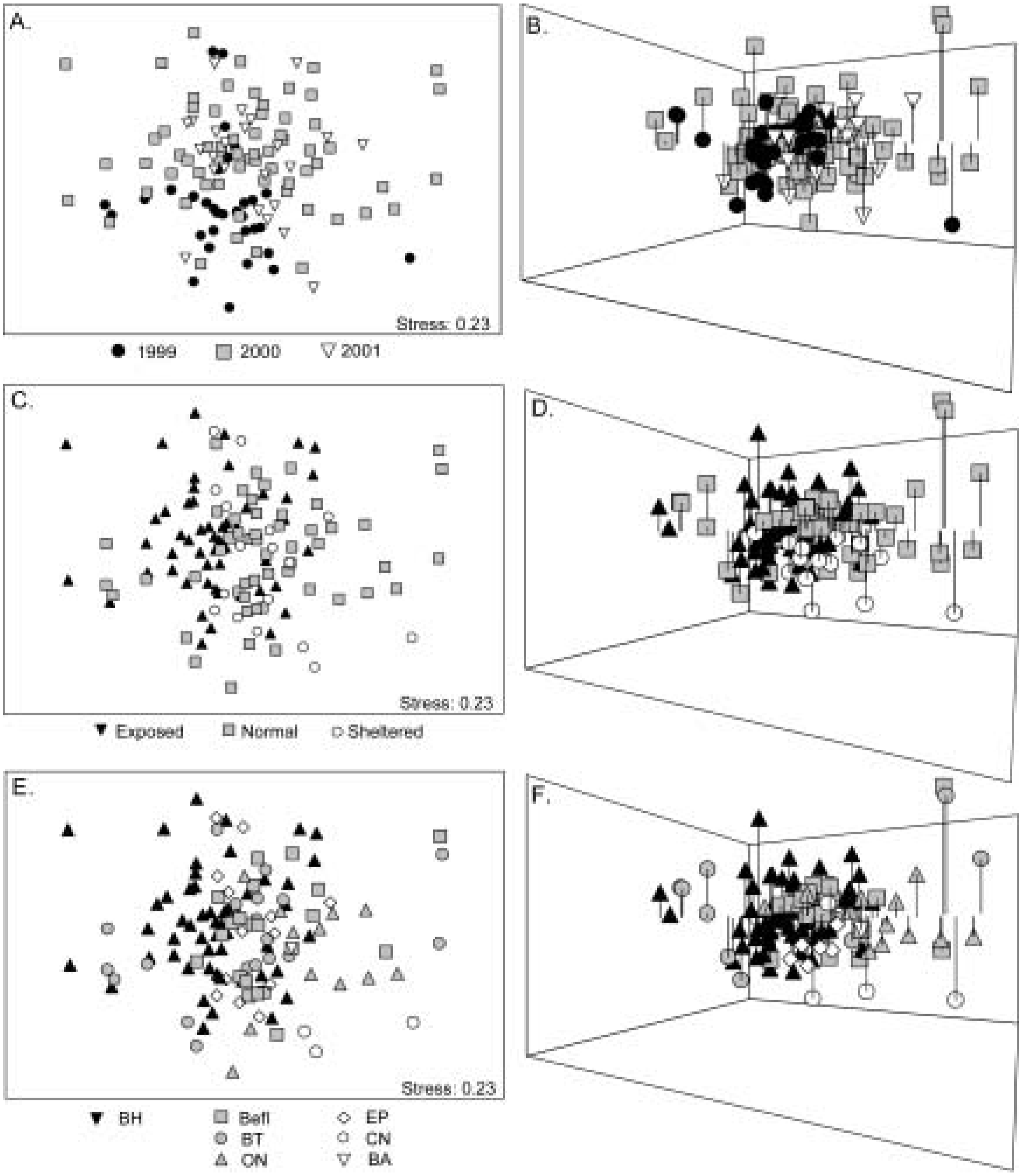


Figure F-1. MDS ordination of square-root transformed fish densities in main channel samples collected in the Rosedale sub-reach in autumn months (1999-2001). Each plot shows the identical ordination with samples classified according to A) years, B) morphological habitats, and C) hydraulic habitats.

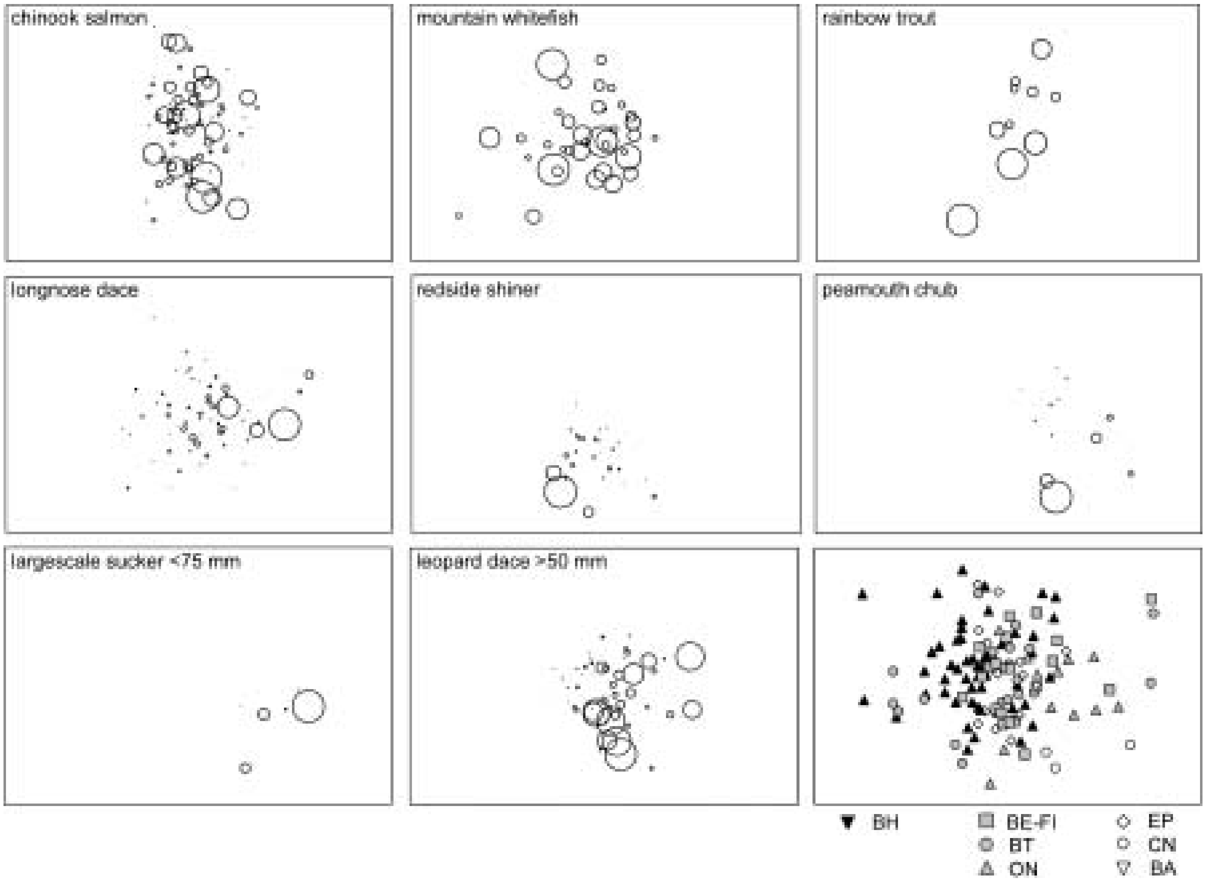


Figure F-2. Bubble plots corresponding to the untransformed densities of select fish species that contributed to the dissimilarity in community structure between habitats. Plots are overlaid on the MDS ordination of samples collected in autumn from the main channel of the Rosedale sub-reach (1999-2001). Symbols are sized proportionately to individual species' densities.

Table F-1. Results of SIMPER analysis based on square-root transformed data indicating the contribution (%) made by species to the *dissimilarity* in community structure between habitats in the main channel of the Rosedale sub-reach (autumn data, 2000 only).

Species	**ON/BH**	**ON/BT**	**ON/EP**	**ON/CN***	**BH/CN***	**BE/CN***	**BT/EP**
%Dissimilarity	77.8	72.2	72.9	75.9	81.3	77.1	68.9
leopard dace >50	**+11.7**	**+11.9**	**+11.6**	**+9.0**	+4.0	+6.9	**-11.2**
largescale sucker <75	**+25.5**	**+25.4**	**+22.0**	**+17.2**	+2.5	+2.5	+4.4
redside shiner	**-6.3**	-3.9	**-8.4**	**-9.1**	**-10.7**	**-10.7**	**-11.9**
peamouth chub	**+19.2**	**+19.9**	**+14.5**	**-22.7**	**-43.1**	**-37.1**	**+8.9**
longnose dace	+5.5	**-6.7**	-5.0	+3.8	+4.4	+4.4	**+8.5**
leopard dace <50	**+9.3**	**+8.0**	+8.1	+6.4	+2.0	+2.0	+8.0
chinook salmon	-5.9	-6.4	**-11.4**	**-12.9**	**-13.7**	**-11.5**	**-15.1**
mountain sucker >100	-1.3	-5.2	-0.4	-	+1.9	+1.9	+6.5
prickly sculpin <50	0	-	-1.1	-7.3	**-9.1**	**-8.9**	-1.5
sockeye salmon	-0.5	-	-0.6	-7.3	**-8.7**	**-8.5**	-0.8

+/- indicates the direction of difference in species density between habitats, e.g., ON/BH: +11.7 indicates that leopard dace density was higher in open nooks and contributed 11.7% to the dissimilarity with bar heads.
Bolded text highlights the five species contributing most to dissimilarity *between pairs*.
Listed species contributed substantially to the dissimilarity between at least one pairwise contrast.

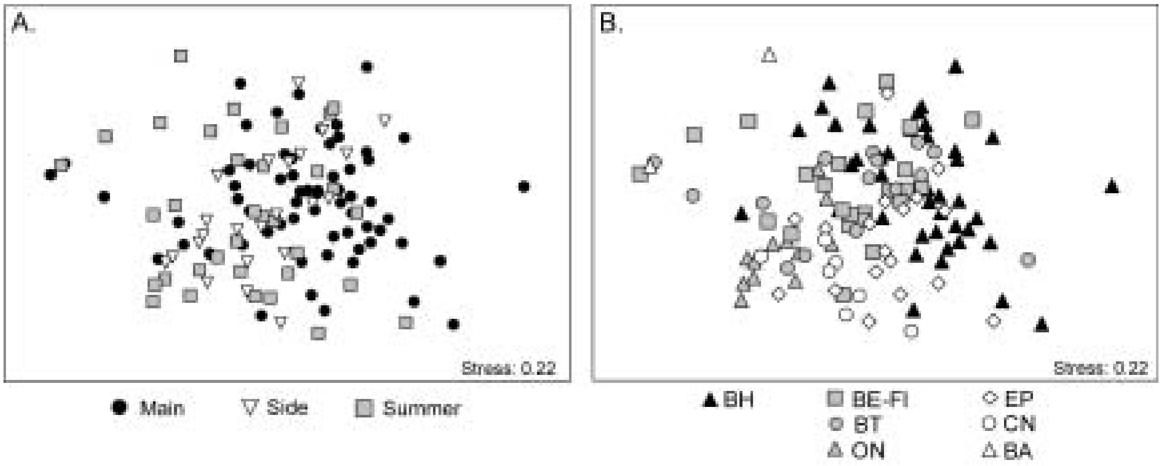


Figure F-3. MDS ordination of fish data collected from three channel types in autumn 2000 from the Rosedale sub-reach. Each plot shows the identical ordination with samples classified by A) channel types, and B) morphologic habitat types. The shading of morphological habitat units generally distinguishes hydraulic habitat groups.

Table F-2. Results of 2-way ANOSIM tests comparing community patterns between channel types and habitat types, based on data collected from the Rosedale sub-reach in autumn 2000.

Comparison	Possible Permutations	# Cases ≥ Observed[+]	*R*-Statistic	*p*-value
Differences between channel types averaged across morphologic habitat types				
Global			0.125	0.066*
Main, Side	>999	766	-0.108	0.77
Main, Summer	>999	4	0.252	0.005
Side, Summer	>999	223	0.052	0.22
*Differences between morphologic habitat types averaged across channel types***				
Global			0.124	0.024*
Open Nook, Bar Head	>999	4	0.370	0.005*
Open Nook, Flat Bar Edge	>999	2	0.272	0.003*
Open Nook, Bar Tail	>999	5	0.303	0.006*
Open Nook, Eddy Pool	>999	1	0.583	0.002*
Open Nook, Channel Nook	>999	21	0.476	0.022*
Channel Nook, Bar Head	>999	35	0.459	0.036*
Eddy Pool, Flat Bar Edge	>999	60	0.097	0.061*
Eddy Pool, Bar Tail	>999	4	0.214	0.005*

* significant at the adjusted value of α=0.1.

+ number of permutated cases where the simulated *R*-value is greater than the actual *R*-statistic.

** only significant contrasts are reported

Table F-3. Results of SIMPER analysis (square-root transformed data) indicating the average density (untransformed, # 10-m^{-2}) of species that contributed most to the *dissimilarity* in community structure between channel types.

Species	**Average Density**			**% Contributed to Dissimilarity**		
	Main	**Side**	**Summer**	**M/Si**	**M/Su**	**Si/Su**
peamouth chub	0.21	0.63	0.45	**14.2**	**13.5**	**14.1**
longnose dace	0.07	0.45	0.14	**10.7**	9.3	**11.2**
largescale sucker <75	0.19	0.44	0.77	**10.6**	**13.8**	**13.9**
chinook salmon	0.14	0.14	0.13	**10.4**	**11.3**	10.1
leopard dace <50	0.04	0.23	0.49	9.8	8.1	**11.0**
leopard dace >50	0.15	0.09	0.14	9.3	9.8	7.7
redside shiner	0.11	0.07	0.19	8.4	**10.1**	8.1
Mean Similarity	30.4	32.8	24.8	-	-	-
Mean Dissimilarity	-	-	-	69.8	75.0	71.5

Data are from the Rosedale sub-reach collected in autumn months.
Bolded text highlights the four taxa contributing most to dissimilarity *between pairs*.

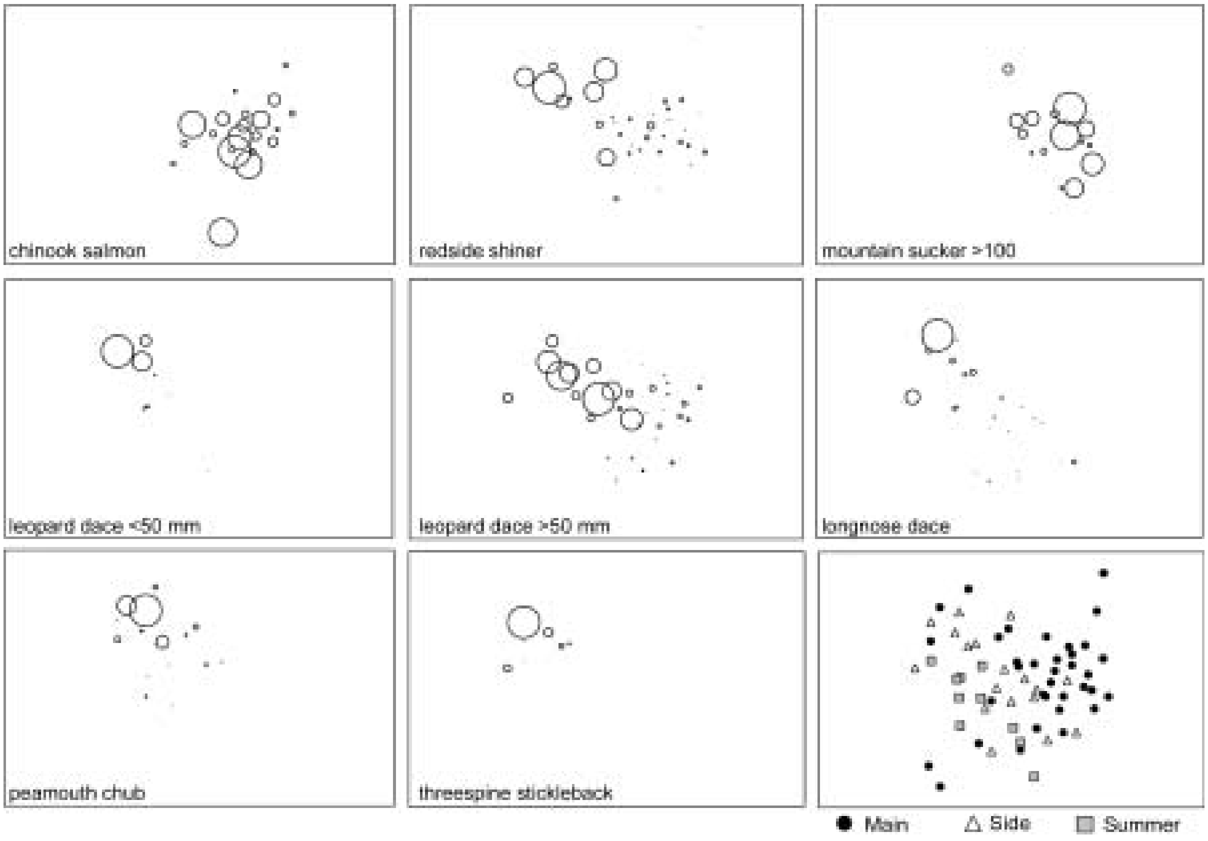


Figure F-4. Bubble plots corresponding to the untransformed densities of select fish species that contributed to the dissimilarity in community structure between channel types. Plots are overlaid on the MDS ordination of samples collected in autumn from the Chilliwack sub-reach in 2000. Symbols are sized proportionately to individual species' densities.

Appendix G. Stomach Contents Of Fish

Approximately 2% (490 fish in 1999, 619 in 2000, and 70 in 2001) of fish representing most species were sacrificed to examine stomach contents and determine diet composition. These fish were overdosed with anesthetic (MS-222), followed by a sharp blow to the spinal cord. Specimens were preserved in 10% formalin for later processing at the DFO Laboratory at Cultus Lake.

In the lab, each fish was lightly dried and weighed with the stomach intact and then weighed again after stomach removal. Visible parasites were removed from the fish and their net weight subtracted from the total weight. Following methods of Murphy and Willis (1996), the esophagus and stomach cavity were removed from all fish; no material was collected below the pyloric caeca (where the intestine leaves the stomach). Stomach contents were washed into a petri dish and distinguishable prey material picked with a dissecting microscope (20x lens). For animal prey, identification and counts were based primarily on head capsules but included major body parts resistant to digestion. Sources listed in Chapter 3 were used for taxonomic identification to the lowest possible level. Algae, seeds, and other plant material were identified and counted as well.

A volumetric estimation of prey items in each stomach was made in addition to count-based estimates. First, the stomach contents of a fish were separated according to major prey classes and each group was placed on plasticized graphing paper (2 mm grid). The vertical height of each group was held constant (approximately one body thickness) and the horizontal spread was estimated as the number of cells covered on the paper. All groups then were placed together in a graduated 12-ml vial and centrifuged for 18 minutes to obtain the total prey volume. The volume of each prey group was estimated by back-calculation based on the ratio of total number of squares covered and total prey volume.

Values in the following tables are the average volume of each prey type in stomachs grouped by species and year, expressed as a percentage of the total stomach volume.

Table G-1. Mean percent (% total stomach volume +/- SE) of prey items for fish species (family Salmonidae). Mean stomach weight (% body weight +/- SE) and sample size are given. Summer data were collected June-September, winter data February-April. Shaded cells highlight dominant prey items. Numbers in parentheses indicate the number of fish with stomachs containing the prey item when n < 3

Prey Item	**Chinook Salmon**				**Chum Salmon**	**Sockeye Salmon**				**Cutthroat Trout**		**Rainbow Trout**				**Mountain Whitefish**			
	S 1999	**S 2000**	**S 2001**	***W 2000***	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***	**S 2000**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***
Sample Size	**149**	**298**	**14**	***62***	***34***	**9**	**11**	**2**	***11***	**1**	***10***	**12**	**1**	**2**	***2***	**27**	**10**	**6**	***31***
% Body Weight as Stomach	6.78 ± 0.18	7.52 ± 0.13	10.62 ± 0.66	*10.37 ± 0.53*	*15.96 ± 0.77*	10.73 ± 0.95	9.60 ± 0.83	13.45 ± 0.41	*6.79 ± 0.93*	17.65	*3.57 ± 1.03*	7.64 ± 1.14	6.91	14.46 ± 4.17	*4.46 ± 1.03*	6.57 ± 0.85	6.34 ± 0.53	9.51 ± 0.66	*4.36 ± 0.29*
Sand	0	0	0	*0*	*0*	0	0	0	*0*	0	*0*	0	0	0	*0*	0	(1)	(1)	*6.96 ± 2.48*
***Ephemeroptera* (A)**	1.14 ± 0.67	21.32 ± 7.13	0	*(2)*	*(1)*	(1)	0	0	*0*	0	*0*	0	0	0	*0*	(1)	0	0	*0*
***Ephemeroptera* (N)**	9.32 ± 1.56	21.98 2.13	20.31 ± 5.24	*6.47 ± 1.83*	*9.12 ± 3.22*	7.78 ± 6.5	25.65 ± 4.90	0	*2.05 ± 1.04*	0	*(2)*	5.99 ± 4.27	(1)	(2)	*0*	26.33 ± 7.32	41.45 ± 11.29	30.53 ± 14.66	*1.94 ± 0.99*
***Plecoptera* (A)**	0	0	0	*8.38 ± 3.05*	*(1)*	0	0	0	*(2)*	0	*0*	0	0	0	*0*	0	0	0	*0*
***Plecoptera* (N)**	1.76 ± 0.86	15.74 2.87	3.10 ± 1.67	*19.37 ± 3.81*	*(2)*	0	0	0	*0.50 ± 0.26*	0	*(2)*	0	0	0	*(1)*	0	0	0	*2.55 ± 2.29*
***Trichoptera* (A)**	1.87 ± 0.99	31.31 ± 3.35	25.60 ± 8.31	*0*	*0*	(2)	(1)	0	*0*	0	*0*	0	0	0	*0*	0	0	0	*0*
***Trichoptera* (N)**	1.43 ± 0.52	9.91 ± 2.81	6.30 ± 0.90	*(1)*	*(2)*	(1)	0	0	*0*	(1)	*(2)*	(1)	(1)	0	*0*	3.81 ± 1.92	0	(2)	*0.52 ± 0.35*
***Diptera* (A)**	11.30 ± 1.81	17.27 ± 1.95	5.53 ± 0.88	*1.04 ± 0.45*	*2.56 ± 1.10*	24.11 ± 12.34	16.67 ± 0.67	0	*(2)*	0	*(1)*	3.22 ± 2.22	(1)	(2)	*0*	0.20 ± 0.11	0	0	*0*
***Diptera* (N)**	0.76 ± 0.31	10.91 ± 5.09	(2)	*(1)*	*0*	(2)	0	0	*0*	0	*0*	0	0	0	*0*	4.66 ± 2.66	0	0	*0*
***Chironomidae* (A)**	22.82 ± 2.65	45.68 ± 2.30	56.83 ± 6.64	*4.93 ± 2.10*	*19.21 ± 5.99*	20.11 ± 10.63	44.71 ± 12.09	(1)	*(1)*	(1)	*(2)*	8.93 ± 4.84	(1)	0	*(1)*	3.31 ± 1.66	(1)	0	*0*
***Chironomidae* (N)**	7.12 ± 1.60	32.39 ± 2.81	(2)	*34.36 ± 4.79*	*39.91 ± 7.27*	19.89 ± 10.83	43.02 ± 28.74	(2)	*64.67 ± 12.74*	(1)	*3.47 ± 2.51*	2.03 ± 1.29	0	(2)	*0*	28.70 ± 6.87	56.40 ± 11.19	72.89 ± 13.37	*74.73 ± 5.10*
***Hymenoptera* (A)**	10.38 ± 1.95	18.67 ± 3.52	7.58 ± 2.18	*0.38 ± 0.22*	*0*	(1)	(1)	0	*0*	0	*0*	(2)	0	0	*0*	4.41 ± 2.48	0	0	*0*
Other Aquatic Inverts	0.65 ± 0.26	8.20 ± 2.07	(1)	*0.92 ± 0.78*	*(2)*	0	0	0	*0*	0	*(1)*	0	0	0	*0*	3.91 ± 3.70	0	0	*(1)*
Terrestrial Inverts	7.00 ± 1.20	14.68 ± 1.44	11.51 ± 4.86	*2.86 ± 1.09*	*1.12 ± 0.57*	5.67 ± 3.47	36.35 15.83	0	*0.69 ± 0.47*	0	*0*	24.32 ± 10.10	0	(1)	*0*	0.87 ± 0.85	0	0	*0*
Fish Eggs	(2)	(1)	0	*0*	*0*	0	0	0	*0*	0	*0*	0	0	(1)	*0*	(2)	0	0	*0*
Fish Parts	1.24 ± 0.76	37.82 ± 8.75	0	*14.86 ± 4.38*	*0*	0	0	0	*0*	0	*0*	0	0	0	*0*	3.02 ± 2.15	0	0	*0*
Invertebrate Parts	5.23 ± 1.72	45.03 ± 28.86	0	*0*	*0*	0	0	0	*0*	0	*0*	0	0	0	*0*	(2)	0	0	*0*
Invertebrate (A)	1.10 ± 0.62	(1)	0	*0*	*0*	(1)	0	0	*0*	0	*0*	(1)	0	0	*0*	(2)	0	0	*0*
Invertebrate (P)	(1)	0	0	*0*	*0*	0	0	0	*0*	0	*0*	0	0	0	*0*	0	0	0	*0*
Invertebrate (N)	3.48 ± 1.29	0	0	*(1)*	*0*	0	(1)	0	*0*	0	*0*	(2)	0	0	*0*	0	0	0	*(1)*
Gastropoda	0	0	0	*0*	*0*	0	0	0	*0*	0	*0*	0	0	0	*0*	(1)	0	0	*0*
Algae	0.11 ± 0.07	8.17 ± 3.49	(1)	*0*	*0*	0	0	0	*0*	0	*0*	(1)	0	0	*0*	(1)	13.53 ± 9.25	0	*0.97 ± 0.59*
Plant Seeds	1.30 ± 0.50	7.67 ± 2.35	(1)	*(1)*	*0*	0	0	0	*(1)*	0	*0*	0	0	0	*0*	(1)	0	0	*0*
Plant Material	1.36 ± 0.72	5.20 ± 1.23	0	*0.62 ± 0.46*	*0*	0	0	0	*0*	0	*14.38 ± 9.38*	(1)	0	0	*0*	5.59 ± 3.75	(1)	(1)	*7.71 ± 2.69*
Zooplankton	(1)	0	0	*(1)*	*24.38 ± 7.12*	0	0	0	*0.26 ± 0.18*	0	*0*	0	0	0	*0*	0	0	0	*(2)*
Unidentified	0.62 ± 0.31	11.66 ± 6.43	4.63 ± 1.75	*(2)*	*0*	0	(1)	0	*0*	0	*(1)*	0	0	0	*0*	0	0	0	*(1)*

A = Adult
N= Nymph
P= Pupae

Table G-2. Mean percent (% total stomach volume +/- SE) of prey items for fish species (family Cyprinidae). Mean stomach weight (% body weight +/- SE) and sample size are given. Summer data were collected June-September, winter data February-April. Shaded cells highlight dominant prey items for each case. Numbers in parentheses indicate the number of fish with stomachs containing the prey item when n < 3.

Prey Item	**Leopard Dace**				**Longnose Dace**				**Redside Shiner**				**Northern Pike Minnow**			**Peamouth Chub**			
	S 1999	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	**S 1999**	**S 2000**	**S 2001**	***W 2000***
Sample Size	**19**	**11**	**3**	***10***	**15**	**3**	**1**	***1***	**105**	**23**	**20**	***8***	**46**	**24**	**10**	**55**	**7**	**5**	***17***
% Body Weight as Stomach	8.48 ± 0.98	9.71 ± 0.52	14.72 ± 0.48	*13.06 ± 1.47*	6.87 ± 1.09	7.65 ± 0.57	5.12	*14.00*	7.29 ± 0.23	9.83 ± 0.90	9.41 ± 0.51	*6.07 ± 0.64*	5.36 ± 0.34	7.09 ± 0.49	7.20 ± 0.81	6.00 ± 0.63	6.05 ± 0.66	12.54 ± 1.57	*8.80 ± 1.07*
Sand	(1)	69.45 ± 17.57	0	*0*	0	0	0	*0*	0	0	0	*0*	0	(2)	0	0	(2)	(1)	*(2)*
***Ephemeroptera* (A)**	(1)	0	0	*0*	0	0	0	*0*	2.06 ± 1.05	0	0	*0*	0	(1)	0	0	(1)	0	*0*
***Ephemeroptera* (N)**	(2)	(1)	6.11 ± 2.77	*0*	(1)	0	0	*(1)*	4.45 ± 1.37	21.83 ± 13.57	28.35 ± 13.70	*(2)*	7.42 ± 3.45	29.69 ± 10.08	32.29 ± 26.84	4.82 ± 2.22	0	0	*0*
***Plecoptera* (A)**	0	0	0	*0*	0	0	0	*0*	(1)	0	0	*0*	0	0	0	0	0	0	*0*
***Plecoptera* (N)**	0	0	0	*0*	0	0	0	*0*	1.41 ± 0.97	(1)	(1)	*0*	0	4.98 ± 2.53	0	0	0	0	*0*
***Trichoptera* (A)**	(2)	0	0	*0*	0	0	(1)	*0*	(2)	16.33 ± 6.17	73.38 ± 14.52	*0*	(1)	(2)	0	0	0	0	*(2)*
***Trichoptera* (N)**	(2)	(2)	(1)	*(1)*	0	0	0	*0*	1.35 ± 0.59	(1)	(2)	*0*	2.84 ± 1.62	17.67 ± 9.67	(2)	1.60 ± 0.87	0	0	*0*
***Diptera* (A)**	(2)	0	0	*(1)*	0	0	0	*0*	2.60 ± 0.93	18.06 ± 11.90	(2)	*(1)*	(1)	(1)	(1)	(1)	0	0	*0*
***Diptera* (N)**	0	0	0	*0*	(1)	0	0	*0*	0.74 ± 0.41	0	0	*0*	(1)	(1)	0	(1)	0	0	*(1)*
***Chironomidae* (A)**	0	(1)	0	*0*	(1)	0	0	*0*	11.28 ± 2.54	15.16 ± 10.15	17.22 ± 16.39	*0*	0.10 ± 0.07	0	(1)	(2)	0	0	*(1)*
***Chironomidae* (N)**	8.00 ± 3.40	52.12 ± 16.00	0	*57.68 ± 13.41*	7.80 ± 6.66	0	0	*(1)*	5.57 ± 1.64	43.92 ± 16.97	37.32 ± 29.32	*73.13 ± 12.60*	6.71 ± 3.19	0	(2)	2.39 ± 1.06	(1)	(2)	*58.00 ± 10.57*
***Hymenoptera* (A)**	0	0	0	*0*	0	0	0	*0*	12.49 ± 2.66	69.66 ± 6.60	(2)	*0*	0	0	0	(2)	0	0	*0*
Other Aquatic Inverts	(1)	0	0	*0*	0	0	0	*0*	1.69 ± 0.68	(1)	0	*0*	4.00 ± 2.46	0	(2)	2.04 ± 1.22	0	0	*(1)*
Terrestrial Inverts	(1)	(1)	0	*0*	0	0	0	*0*	10.27 ± 2.23	25.70 ± 7.73	9.47 ± 4.68	*0*	(2)	0	0	5.34 ± 2.30	0	0	*0*
Fish Eggs	0	0	0	*0*	0	0	0	*0*	0	0	0	*0*	(1)	54.02 ± 20.75	0	0	0	0	*0*
Fish Parts	0	0	0	*0*	(1)	0	0	*0*	(2)	0	(1)	*(2)*	10.11 ± 4.31	(2)	(1)	0	0	0	*0*
Invertebrate Parts	0	0	0	*0*	(1)	0	0	*0*	6.81 ± 2.14	0	0	*0*	0.72 ± 0.65	0	0	1.92 ± 1.35	0	0	*0*
Invertebrate (A)	0	0	0	*0*	0	0	0	*0*	2.45 ± 1.33	0	(1)	*0*	(1)	0	0	(1)	0	0	*0*
Invertebrate (P)	0	0	0	*0*	(1)	0	0	*0*	0	0	0	*0*	0	0	0	0	0	0	*0*
Invertebrate (N)	(2)	0	0	*0*	(1)	0	0	*0*	2.92 ± 1.29	0	0	*0*	2.65 ± 2.19	0	0	2.89 ± 1.95	0	0	*0*
Gastropoda	0	0	0	*0*	0	0	0	*0*	5.82 ± 1.95	0	0	*0*	(2)	0	0	25.11 ± 5.28	0	0	*0*
Algae	48.47 ± 10.30	46.54 ± 26.77	89.34 ± 5.62	*27.02 ± 12.33*	9.41 ± 5.95	0	0	*0*	10.48 ± 2.22	9.34 ± 5.97	71.91 ± 17.96	*7.25 ± 6.17*	7.28 ± 3.45	(1)	(2)	18.13 ± 4.92	(2)	28.99 ± 20.99	*7.76 ± 4.82*
Plant Seeds	0	0	0	*0*	0	0	0	*0*	2.86 ± 1.07	0	67.41 ± 11.75	*0*	12.51 ± 4.67	0	63.74 ± 18.10	2.62 ± 1.00	0	0	*0*
Plant Material	1.32 ± 1.02	0	(1)	*0*	13.20 ± 8.08	0	0	*0*	2.95 ± 0.80	41.52 ± 16.80	0	*(1)*	2.00 ± 1.56	16.31 ± 11.93	0	4.11 ± 2.08	(1)	0	*0*
Zooplankton	0	0	0	*0*	0	0	0	*0*	(1)	(1)	0	*0*	0	0	0	0	0	0	*0*
Unidentified	0	0	(1)	*0*	0	0	0	*0*	0	0	(2)	*(2)*	0	55.73 ± 17.62	(2)	2.85 ± 1.89	0	0	*0*

A = Adult
N= Nymph
P= Pupae

Table G-3. Mean percent (% total stomach volume +/- SE) of prey items for fish species (family Catostomidae and Cottidae). Mean stomach weight (% body weight +/- SE) and sample size are given. Summer data were collected June-September, winter data February-Apr
Shaded cells highlight dominant prey items for each case. Numbers in parentheses indicate the number of fish with stomachs containing the prey item when n < 3.

Prey Item	**Mountain Sucker**				**Largescale Sucker**				**Coastrange Sculpin**			**Prickly Sculpin**			
	S 1999	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***	**S 1999**	**S 2000**	***W 2000***	**S 1999**	**S 2000**	**S 2001**	***W 2000***
Sample Size	**24**	**2**	**4**	***2***	**14**	**4**	**2**	***3***	**7**	**3**	***22***	**7**	**1**	**1**	***4***
% Body Weight as Stomach	9.54 ± 2.07	4.67 ± 3.51	16.68 ± 1.02	*7.06 ± 0.46*	8.31 ± 1.18	5.75 ± 1.39	11.46 ± 0.87	*3.54 ± 1.18*	9.27 ± 1.97	9.76 ± 1.80	*8.69 ± 0.49*	6.03 ± 1.05	11.76	8.34	*4.77 ± 1.34*
Sand	(1)	0	0	*0*	3.93 ± 2.68	0	0	*0*	0	0	*(1)*	0	0	0	*0*
***Ephemeroptera* (A)**	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
***Ephemeroptera* (N)**	(1)	0	0	*0*	(1)	(1)	0	*(2)*	(2)	(1)	*12.95 ± 5.37*	(2)	0	0	*37.19 ± 13.62*
***Plecoptera* (A)**	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
***Plecoptera* (N)**	0	0	0	*0*	0	0	(1)	*0*	0	0	*1.55 ± 0.93*	0	0	0	*(2)*
***Trichoptera* (A)**	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
***Trichoptera* (N)**	(2)	0	0	*0*	0	(1)	0	*0*	27.00 ± 14.69	(1)	*0*	(1)	0	(1)	*11.05 ± 7.01*
***Diptera* (A)**	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
***Diptera* (N)**	(1)	0	0	*0*	(2)	0	0	*0*	25.29 ± 14.57	0	*0*	0	0	0	*(1)*
***Chironomidae* (A)**	(1)	0	0	*0*	(1)	0	0	*0*	0	0	*0*	0	0	0	*0*
***Chironomidae* (N)**	0.19 ± 0.07	0	0	*0*	12.16 ± 7.52	54.10 ± 6.74	0	*12.33 ± 7.88*	74.57 ± 14.69	(2)	*84.91 ± 5.28*	(2)	(1)	0	*10.23 ± 7.08*
***Hymenoptera* (A)**	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
Other Aquatic Inverts	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
Terrestrial Inverts	0	0	0	*0*	(1)	0	0	*0*	0	0	*0*	(1)	0	0	*0*
Fish Eggs	0	0	0	*0*	0	0	0	*0*	0	0	*0*	(1)	0	0	*0*
Fish Parts	0	0	0	*0*	0	0	0	*0*	0	0	*0*	(1)	0	0	*(1)*
Invertebrate Parts	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
Invertebrate (A)	0	0	0	*0*	(2)	0	0	*0*	0	0	*0*	(1)	0	0	*0*
Invertebrate (P)	0	0	0	*0*	0	0	0	*0*	0	0	*0*	0	0	0	*0*
Invertebrate (N)	0	0	0	*0*	0	0	0	*0*	0	0	*(1)*	(1)	0	0	*0*
Gastropoda	0	0	0	*0*	(1)	0	0	*0*	0	0	*0*	(2)	0	0	*0*
Algae	82.69 ± 7.71	(2)	100.00	*100.0*	68.22 ± 11.21	42.00 ± 8.41	(2)	*54.00 ± 27.50*	0	0	*0*	0	0	0	*0*
Plant Seeds	0	0	0	*0*	(1)	0	0	*0*	0	0	*0*	(1)	0	(1)	*(1)*
Plant Material	11.95 ± 6.41	0	0	*0*	(2)	(1)	0	*0*	(1)	0	*(2)*	(1)	0	(1)	*11.60 ± 5.44*
Zooplankton	0	0	0	*0*	(1)	0	0	*(1)*	0	0	*0*	0	0	0	*0*
Unidentified	0	0	0	*0*	0	0	0	*0*	0	(1)	*0*	0	0	0	*(1)*

Fish were collected June-September (S) or February-April (W).
A = Adult
N= Nymph
P= Pupae

Appendix H. Asymmetrical ANOVA – Mechanics and Power Calculation

1.0 Mechanics

Underwood (1993) describes in thorough detail the mechanics of asymmetrical ANOVA, which is carried out by running four separate ANOVA procedures using any common statistical software. The procedure is described briefly below. Refer to appendices I through N to examine its application.

Three independent variables are involved in the analysis:

1. Before/After (B) – fixed categorical variable
2. Times – random categorical variable
3. Locations – fixed or random categorical variable

First (Table A in appendices), analyze all data as a fully orthogonal analysis of Before/After, Locations, and Times nested within Before/After [T(B)]. This analysis does not differentiate between the impacted and reference sites.

Second (Table B in appendices), analyze data as a three-factor analysis of all data from the reference locations only. The identical model as for analysis #1 is used, only selecting for reference sites.

Third (Table C in appendices), analyze data from all sites as a two-factor analysis and exclude sampling periods after the impact. Factor B (Before/After) is not included and Times is included as a non-nested factor.

Fourth (Table D in appendices), repeat analysis #3 but include only the reference sites.

From these four analyses, the entire asymmetrical ANOVA can be calculated by subtractions and additions of the component terms. The algebra is indicated in the appendices. Once the Source of Variation table is completed, use the flow chart in Chapter 8 (Figure 8-2) and proceed through a set of questions and statistical tests to address whether or not an impact has occurred. The answer to the question at each branch of the flowchart determines the sources of variation and degrees of freedom used to calculate an F-value. Refer to Chapter 8 (Section 8.4.1) for further details.

2.0 Power Calculation

The power of a statistical analysis is its capacity to detect a difference between groups when a difference actually exists. The calculation of power for asymmetrical ANOVA is straightforward, using values derived from the Source of Variation table and from the central distribution of *F* that is in common use.

First, determine $1+n\theta$ = Mean Square [T(Aft) x Sc] / Mean Square [Residual]

Second, calcuate $F_{alt} = F_{crit} / (1+n\theta)$. F_{crit} is based on the degrees of freedom of $(1+n\theta)$. df_1 for MS [T(Aft) x Sc]; df_2 for MS [Residual].

Third, estimate power based on F_{alt} using the distribution function of F: $[(F_{alt}), df_1, df_2]$

Appendix I

ANOVA Results of Habitat Characteristics – Bar Edge Units

PC AXIS 1 (Hydraulic Gradient)

A. All Data

Source of Variance	SS	df	MS	Term
Bef/Aft - B	0.109	1	0.109	a1
Times(B)	10.180	9	1.131	a2
Locations	2.980	3	0.993	a3
B * L	2.400	3	0.800	a4
T(B) * L	43.725	27	1.619	a5
Residual	42.029	80	0.525	a6
Total	101.423	123		a7
B. Reference Locations on All Dates				
Bef/Aft - B	0.928	1	0.928	
Times(B)	7.272	9	0.808	
Locations	2.954	2	1.477	b1
B * L	0.044	2	0.022	b2
T(B) * L	11.900	18	0.661	b3
Residual	29.221	53	0.551	
Total	52.319	85		
C. All Locations Before Impact				
Times(B)	7.240	2	3.620	
Locations	2.106	3	0.702	
B * L	6.296	6	1.049	c1
Residual	2.332	13	0.179	
Total	17.974	24		
D. Reference Locations Before Impact				
Times(B)	2.053	2	1.027	
Locations	1.149	2	0.575	
B * L	1.685	4	0.421	d1
Residual	1.472	6	0.245	
Total	6.359	14		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.109	1	0.109
T(B)	a2	10.180	9	1.131
Location	a3	2.980	3	0.993
Impact vs Ref	a3-b1	0.026	1	0.026
Among Refs	b1	2.954	2	1.477
B * Location	a4	2.400	3	0.800
B * Impact	a4-b2	2.356	1	2.356
B * Ref	b2	0.044	2	0.022
T(B) * L	a5	43.725	27	1.619
T(Bef) * Location	c1	6.296	6	1.049
T(Bef) * Impact	c1-d1	4.611	2	2.306
T(Bef) * Ref	d1	1.685	4	0.421
T(Aft) * Location	a5-c1	37.429	21	1.782
T(Aft) * Impact	a5-c1-b3+d1	27.214	7	3.888
T(Aft) * Ref	b3-d1	10.215	14	0.730
Residual	a6	42.029	74	0.568
Total	a7	101.423	123	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	1.285	1.83	0.24
NO…				
2. Does scalping affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	6.845	2.14	p<0.0005
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,4	1.732	5.87	0.32
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,2	1.686	39.4	0.42
NO… change was not coincident with impact				

PC AXIS 2 (Coarse Sediment Gradient)

A. All Data

Source of Variance	SS	df	MS	Term
Bef/Aft - B	5.024	1	5.024	a1
Times(B)	30.489	9	3.388	a2
Locations	8.621	3	2.874	a3
B * L	3.399	3	1.133	a4
T(B) * L	13.377	27	0.495	a5
Residual	44.880	80	0.561	a6
Total	105.790	123		a7
B. Reference Locations on All Dates				
Bef/Aft - B	4.390	1	4.390	
Times(B)	23.538	9	2.615	
Locations	4.701	2	2.351	b1
B * L	2.930	2	1.465	b2
T(B) * L	10.484	18	0.582	b3
Residual	35.061	53	0.662	
Total	81.104	85		
C. All Locations Before Impact				
Times(B)	25.072	2	12.536	
Locations	6.567	3	2.189	
B * L	5.157	6	0.860	c1
Residual	8.742	13	0.672	
Total	45.538	24		
D. Reference Locations Before Impact				
Times(B)	19.136	2	9.568	
Locations	4.273	2	2.137	
B * L	5.006	4	1.252	d1
Residual	4.583	6	0.764	
Total	32.998	14		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	5.024	1	5.024
T(B)	a2	30.489	9	3.388
Location	a3	8.621	3	2.874
Impact vs Ref	a3-b1	3.920	1	3.920
Among Refs	b1	4.701	2	2.351
B * Location	a4	3.399	3	1.133
B * Impact	a4-b2	0.469	1	0.469
B * Ref	b2	2.930	2	1.465
T(B) * L	a5	13.377	27	0.495
T(Bef) * Location	c1	5.157	6	0.860
T(Bef) * Impact	c1-d1	0.151	2	0.075
T(Bef) * Ref	d1	5.006	4	1.252
T(Aft) * Location	a5-c1	8.220	21	0.391
T(Aft) * Impact	a5-c1-b3+d1	2.742	7	0.392
T(Aft) * Ref	b3-d1	5.478	14	0.391
Residual	a6	44.880	74	0.606
Total	a7	105.790	123	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	0.645	1.83	0.82
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	0.646	2.14	0.72
NO… no short-term impact detected				
3. Do reference sites vary in the difference from before to after?				
B * Ref / Residual	2,74	2.416	3.11	0.1
NO…				
4. Does impact affect difference from before to after impact?				
B * Imp / Residual	1,74	0.773	3.96	0.38
NO…no impact detected				

Power Analysis

1+n0 = MS[T(Aft) * Impact] / MS[Residual]	0.646
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	3.310
Power (based on F-distribution)	0.004

PC AXIS 3 (Fine Sediment Gradient)

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.194	1	0.194	a1
Times(B)	13.764	9	1.529	a2
Locations	1.803	3	0.601	a3
B * L	0.230	3	0.077	a4
T(B) * L	21.796	27	0.807	a5
Residual	76.002	80	0.950	a6
Total	113.789	123		a7
B. Reference Locations on All Dates				
Bef/Aft - B	0.126	1	0.126	
Times(B)	13.386	9	1.487	
Locations	1.623	2	0.812	b1
B * L	0.222	2	0.111	b2
T(B) * L	10.366	18	0.576	b3
Residual	29.022	53	0.548	
Total	54.745	85		
C. All Locations Before Impact				
Times(B)	8.056	2	4.028	
Locations	1.023	3	0.341	
B * L	6.617	6	1.103	c1
Residual	2.404	13	0.185	
Total	18.100	24		
D. Reference Locations Before Impact				
Times(B)	10.357	2	5.179	
Locations	0.940	2	0.470	
B * L	1.938	4	0.485	d1
Residual	1.117	6	0.186	
Total	14.352	14		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	0.939
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	2.277
Power (based on F-distribution)	0.034

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.194	1	0.194
T(B)	a2	13.764	9	1.529
Location	a3	1.803	3	0.601
Impact vs Ref	a3-b1	0.180	1	0.180
Among Refs	b1	1.623	2	0.812
B * Location	a4	0.230	3	0.077
B * Impact	a4-b2	0.008	1	0.008
B * Ref	b2	0.222	2	0.111
T(B) * L	a5	21.796	27	0.807
T(Bef) * Location	c1	6.617	6	1.103
T(Bef) * Impact	c1-d1	4.679	2	2.340
T(Bef) * Ref	d1	1.938	4	0.485
T(Aft) * Location	a5-c1	15.179	21	0.723
T(Aft) * Impact	a5-c1-b3+d1	6.751	7	0.964
T(Aft) * Ref	b3-d1	8.428	14	0.602
Residual	a6	76.002	74	1.027
Total	a7	113.789	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	0.586	1.82	0.87
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	0.939	2.13	0.48
NO… no short-term impact detected				
3. Do reference sites vary in the difference from before to after?				
B * Ref / Residual	2,74	0.108	3.11	0.9
NO…				
4. Does impact affect difference from before to after impact?				
B * Imp / Residual	1,74	0.008	3.96	0.93
NO…no impact detected				

Appendix J

Invertebrate ANOVA Results – Whole Bar Unit

INVERTEBRATE DENSITY

A. All Data				
Source	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	4.391	1	4.391	a1
Times(B)	79.129	8	9.891	a2
Locations	0.301	3	0.100	a3
B * L	0.469	3	0.156	a4
T(B) * L	3.798	24	0.158	a5
Residual	8.346	157	0.053	a6
Total	96.434	196		a7
B. Exclude Impact Site				
Bef/Aft - B	3.816	1	3.816	
Times(B)	50.615	8	6.327	
Locations	0.258	2	0.129	b1
B * L	0.404	2	0.202	b2
T(B) * L	1.713	16	0.107	b3
Residual	5.828	105	0.056	
Total	62.634	134		
C. All Locations Before Impact				
Times(B)	10.507	1	10.507	
Locations	0.391	3	0.130	
B * L	1.033	3	0.344	c1
Residual	2.253	34	0.066	
Total	14.184	41		
D. Reference Locations Before Impact				
Times(B)	6.915	1	6.915	
Locations	0.391	2	0.196	
B * L	0.979	2	0.490	d1
Residual	1.954	27	0.072	
Total	10.239	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	4.391	1	4.391
T(B)	a2	79.129	8	9.891
Location	a3	0.301	3	0.100
Impact vs Ref	a3-b1	0.043	1	0.043
Among Refs	b1	0.258	2	0.129
B * Location	a4	0.469	3	0.156
B * Impact	a4-b2	0.065	1	0.065
B * Ref	b2	0.404	2	0.202
T(B) * L	a5	3.798	24	0.158
T(Bef) * Location	c1	1.033	3	0.344
T(Bef) * Impact	c1-d1	0.054	1	0.054
T(Bef) * Ref	d1	0.979	2	0.490
T(Aft) * Location	a5-c1	2.765	21	0.132
T(Aft) * Impact	a5-c1-b3+d	2.031	7	0.290
T(Aft) * Ref	b3-d1	0.734	14	0.052
Residual	a6	8.346	156	0.054
Total	a7	96.434	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	0.980	1.75	0.48
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	5.423	2.075	<0.00001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	0.107	39.4	0.99
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	5.373	948	0.32
NO… change was not coincident with impact				

PROPORTION REPRESENTED BY EPHEMEROPTERA, PLECOPTERA, TRICOPTERA

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	1.035	1	1.035	a1
Times(B)	3.515	8	0.439	a2
Locations	0.558	3	0.186	a3
B * L	0.306	3	0.102	a4
T(B) * L	1.834	24	0.076	a5
Residual	6.355	157	0.040	a6
Total	13.603	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.387	1	0.387	
Times(B)	1.899	8	0.237	
Locations	0.193	2	0.097	b1
B * L	0.065	2	0.033	b2
T(B) * L	0.695	16	0.043	b3
Residual	3.587	105	0.034	
Total	6.826	134		
C. All Locations Before Impact				
Times(B)	1.013	1	1.013	
Locations	0.473	3	0.158	
B * L	0.127	3	0.042	c1
Residual	0.720	34	0.021	
Total	2.333	41		
D. Reference Locations Before Impact				
Times(B)	0.528	1	0.528	
Locations	0.110	2	0.055	
B * L	0.016	2	0.008	d1
Residual	0.625	27	0.023	
Total	1.279	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	1.035	1	1.035
T(B)	a2	3.515	8	0.439
Location	a3	0.558	3	0.186
Impact vs Ref	a3-b1	0.365	1	0.365
Among Refs	b1	0.193	2	0.097
B * Location	a4	0.306	3	0.102
B * Impact	a4-b2	0.241	1	0.241
B * Ref	b2	0.065	2	0.033
T(B) * L	a5	1.834	24	0.076
T(Bef) * Location	c1	0.127	3	0.042
T(Bef) * Impact	c1-d1	0.111	1	0.111
T(Bef) * Ref	d1	0.016	2	0.008
T(Aft) * Location	a5-c1	1.707	21	0.081
T(Aft) * Impact	a5-c1-b3+d	1.028	7	0.147
T(Aft) * Ref	b3-d1	0.679	14	0.049
Residual	a6	6.355	156	0.041
Total	a7	13.603	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	1.191	1.75	0.29
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	3.605	2.075	0.001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	6.063	39.4	0.150
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	1.323	948	0.59
NO… change was not coincident with impact				

TAXON RICHNESS

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	234.244	1	234.244	a1
Times(B)	1637.361	8	204.670	a2
Locations	26.372	3	8.791	a3
B * L	13.871	3	4.624	a4
T(B) * L	133.918	24	5.580	a5
Residual	481.823	157	3.069	a6
Total	2527.589	196		a7
B. Exclude Impact Site				
Bef/Aft - B	138.747	1	138.747	
Times(B)	1044.639	8	130.580	
Locations	15.640	2	7.820	b1
B * L	7.077	2	3.539	b2
T(B) * L	100.763	16	6.298	b3
Residual	375.056	105	3.572	
Total	1681.922	134		
C. All Locations Before Impact				
Times(B)	185.570	1	185.570	
Locations	12.841	3	4.280	
B * L	19.885	3	6.628	c1
Residual	223.056	34	6.560	
Total	441.352	41		
D. Reference Locations Before Impact				
Times(B)	132.302	1	132.302	
Locations	2.589	2	1.295	
B * L	19.881	2	9.941	d1
Residual	207.556	27	7.687	
Total	362.328	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	234.244	1	234.244
T(B)	a2	1637.361	8	204.670
Location	a3	26.372	3	8.791
Impact vs Ref	a3-b1	10.732	1	10.732
Among Refs	b1	15.640	2	7.820
B * Location	a4	13.871	3	4.624
B * Impact	a4-b2	6.794	1	6.794
B * Ref	b2	7.077	2	3.539
T(B) * L	a5	133.918	24	5.580
T(Bef) * Location	c1	19.885	3	6.628
T(Bef) * Impact	c1-d1	0.004	1	0.004
T(Bef) * Ref	d1	19.881	2	9.941
T(Aft) * Location	a5-c1	114.033	21	5.430
T(Aft) * Impact	a5-c1-b3+d1	33.151	7	4.736
T(Aft) * Ref	b3-d1	80.882	14	5.777
Residual	a6	481.823	156	3.089
Total	a7	2527.589	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	1.871	1.75	0.033
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	0.820	2.76	
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.533
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	1.350
Power (based on F-distribution)	0.23

NUMBER OF TAXA REPRESENTED BY EPHEMEROPTERA, PLECOPTERA, TRICOPTERA

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	174.386	1	174.386	a1
Times(B)	899.942	8	112.493	a2
Locations	22.739	3	7.580	a3
B * L	8.000	3	2.667	a4
T(B) * L	92.102	24	3.838	a5
Residual	298.449	157	1.901	a6
Total	1495.618	196		a7
B. Exclude Impact Site				
Bef/Aft - B	102.327	1	102.327	
Times(B)	585.328	8	73.166	
Locations	16.499	2	8.250	b1
B * L	2.174	2	1.087	b2
T(B) * L	67.694	16	4.231	b3
Residual	236.333	105	2.251	
Total	1010.355	134		
C. All Locations Before Impact				
Times(B)	58.368	1	58.368	
Locations	11.785	3	3.928	
B * L	12.485	3	4.162	c1
Residual	133.167	34	3.917	
Total	215.805	41		
D. Reference Locations Before Impact				
Times(B)	44.180	1	44.180	
Locations	4.346	2	2.173	
B * L	12.221	2	6.111	d1
Residual	121.833	27	4.512	
Total	182.580	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	174.386	1	174.386
T(B)	a2	899.942	8	112.493
Location	a3	22.739	3	7.580
Impact vs Ref	a3-b1	6.240	1	6.240
Among Refs	b1	16.499	2	8.250
B * Location	a4	8.000	3	2.667
B * Impact	a4-b2	5.826	1	5.826
B * Ref	b2	2.174	2	1.087
T(B) * L	a5	92.102	24	3.838
T(Bef) * Location	c1	12.485	3	4.162
T(Bef) * Impact	c1-d1	0.264	1	0.264
T(Bef) * Ref	d1	12.221	2	6.111
T(Aft) * Location	a5-c1	79.617	21	3.791
T(Aft) * Impact	a5-c1-b3+d1	24.144	7	3.449
T(Aft) * Ref	b3-d1	55.473	14	3.962
Residual	a6	298.449	156	1.913
Total	a7	1495.618	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	2.071	1.75	0.016
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	0.870	2.76	0.55
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.803
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	1.148
Power (based on F-distribution)	0.336

SIMPSON'S DIVERSITY

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.121	1	0.121	a1
Times(B)	3.353	8	0.419	a2
Locations	0.383	3	0.128	a3
B * L	0.209	3	0.070	a4
T(B) * L	1.170	24	0.049	a5
Residual	3.749	157	0.024	a6
Total	8.985	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.020	1	0.020	
Times(B)	2.725	8	0.341	
Locations	0.313	2	0.157	b1
B * L	0.137	2	0.069	b2
T(B) * L	0.793	16	0.050	b3
Residual	2.145	105	0.020	
Total	6.133	134		
C. All Locations Before Impact				
Times(B)	0.834	1	0.834	
Locations	0.209	3	0.070	
B * L	0.011	3	0.004	c1
Residual	0.660	34	0.019	
Total	1.714	41		
D. Reference Locations Before Impact				
Times(B)	0.621	1	0.621	
Locations	0.126	2	0.063	
B * L	0.009	2	0.005	d1
Residual	0.595	27	0.022	
Total	1.351	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.121	1	0.121
T(B)	a2	3.353	8	0.419
Location	a3	0.383	3	0.128
Impact vs Ref	a3-b1	0.070	1	0.070
Among Refs	b1	0.313	2	0.157
B * Location	a4	0.209	3	0.070
B * Impact	a4-b2	0.072	1	0.072
B * Ref	b2	0.137	2	0.069
T(B) * L	a5	1.170	24	0.049
T(Bef) * Location	c1	0.011	3	0.004
T(Bef) * Impact	c1-d1	0.002	1	0.002
T(Bef) * Ref	d1	0.009	2	0.005
T(Aft) * Location	a5-c1	1.159	21	0.055
T(Aft) * Impact	a5-c1-b3+d1	0.375	7	0.054
T(Aft) * Ref	b3-d1	0.784	14	0.056
Residual	a6	3.749	156	0.024
Total	a7	8.985	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	2.330	1.75	0.006
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	0.957	2.76	0.5
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	2.229
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	0.929
Power (based on F-distribution)	0.486

SIMPSON'S EVENNESS

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.489	1	0.489	a1
Times(B)	14.441	8	1.805	a2
Locations	0.098	3	0.033	a3
B * L	0.059	3	0.020	a4
T(B) * L	0.770	24	0.032	a5
Residual	2.393	157	0.015	a6
Total	18.250	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.429	1	0.429	
Times(B)	9.509	8	1.189	
Locations	0.089	2	0.045	b1
B * L	0.054	2	0.027	b2
T(B) * L	0.274	16	0.017	b3
Residual	1.640	105	0.016	
Total	11.995	134		
C. All Locations Before Impact				
Times(B)	1.874	1	1.874	
Locations	0.077	3	0.026	
B * L	0.068	3	0.023	c1
Residual	0.406	34	0.012	
Total	2.425	41		
D. Reference Locations Before Impact				
Times(B)	1.276	1	1.276	
Locations	0.070	2	0.035	
B * L	0.060	2	0.030	d1
Residual	0.372	27	0.014	
Total	1.778	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.489	1	0.489
T(B)	a2	14.441	8	1.805
Location	a3	0.098	3	0.033
Impact vs Ref	a3-b1	0.009	1	0.009
Among Refs	b1	0.089	2	0.045
B * Location	a4	0.059	3	0.020
B * Impact	a4-b2	0.005	1	0.005
B * Ref	b2	0.054	2	0.027
T(B) * L	a5	0.770	24	0.032
T(Bef) * Location	c1	0.068	3	0.023
T(Bef) * Impact	c1-d1	0.008	1	0.008
T(Bef) * Ref	d1	0.060	2	0.030
T(Aft) * Location	a5-c1	0.702	21	0.033
T(Aft) * Impact	a5-c1-b3+d1	0.488	7	0.070
T(Aft) * Ref	b3-d1	0.214	14	0.015
Residual	a6	2.393	156	0.015
Total	a7	18.250	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	0.996	1.83	0.46
NO…				
2. Does scalping affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	4.545	2.14	0.008
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	0.510	39.4	0.82
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	8.714	948	0.26
NO… change was not coincident with impact				

Appendix K

Invertebrate ANOVA Results – Bar Edge Units

INVERTEBRATE DENSITY

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	2.647	1	2.647	a1
Times(B)	71.848	8	8.981	a2
Locations	0.693	3	0.231	a3
B * L	0.512	3	0.171	a4
T(B) * L	3.766	24	0.157	a5
Residual	4.756	115	0.041	a6
Total	84.222	154		a7
B. Exclude Impact Site				
Bef/Aft - B	1.962	1	1.962	
Times(B)	45.796	8	5.725	
Locations	0.603	2	0.302	b1
B * L	0.509	2	0.255	b2
T(B) * L	1.979	16	0.124	b3
Residual	2.447	66	0.037	
Total	53.296	95		
C. All Locations Before Impact				
Times(B)	9.738	1	9.738	
Locations	0.690	3	0.230	
B * L	1.252	3	0.417	c1
Residual	1.118	22	0.051	
Total	12.798	29		
D. Reference Locations Before Impact				
Times(B)	6.346	1	6.346	
Locations	0.671	2	0.336	
B * L	1.193	2	0.597	d1
Residual	0.819	15	0.055	
Total	9.029	20		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	2.647	1	2.647
T(B)	a2	71.848	8	8.981
Location	a3	0.693	3	0.231
Impact vs Ref	a3-b1	0.090	1	0.090
Among Refs	b1	0.603	2	0.302
B * Location	a4	0.512	3	0.171
B * Impact	a4-b2	0.003	1	0.003
B * Ref	b2	0.509	2	0.255
T(B) * L	a5	3.766	24	0.157
T(Bef) * Location	c1	1.252	3	0.417
T(Bef) * Impact	c1-d1	0.059	1	0.059
T(Bef) * Ref	d1	1.193	2	0.597
T(Aft) * Location	a5-c1	2.514	21	0.120
T(Aft) * Impact	a5-c1-b3+d1	1.728	7	0.247
T(Aft) * Ref	b3-d1	0.786	14	0.056
Residual	a6	4.756	113	0.042
Total	a7	84.222	154	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	1.334	1.75	0.20
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,113	5.865	2.075	<0.00001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	0.094	39.4	0.99
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	4.184	948	0.36
NO… change was not coincident with impact				

PROPORTION REPRESENTED BY EPHEMEROPTERA, PLECOPTERA, TRICOPTERA

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.997	1	0.997	a1
Times(B)	2.594	8	0.324	a2
Locations	0.826	3	0.275	a3
B * L	0.270	3	0.090	a4
T(B) * L	1.710	24	0.071	a5
Residual	3.760	115	0.033	a6
Total	10.157	154		a7
B. Exclude Impact Site				
Bef/Aft - B	0.381	1	0.381	
Times(B)	1.221	8	0.153	
Locations	0.297	2	0.149	b1
B * L	0.043	2	0.022	b2
T(B) * L	0.483	16	0.030	b3
Residual	1.221	66	0.019	
Total	3.646	95		
C. All Locations Before Impact				
Times(B)	0.701	1	0.701	
Locations	0.609	3	0.203	
B * L	0.146	3	0.049	c1
Residual	0.421	22	0.019	
Total	1.877	29		
D. Reference Locations Before Impact				
Times(B)	0.308	1	0.308	
Locations	0.181	2	0.091	
B * L	0.040	2	0.020	d1
Residual	0.325	15	0.022	
Total	0.854	20		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.997	1	0.997
T(B)	a2	2.594	8	0.324
Location	a3	0.826	3	0.275
Impact vs Ref	a3-b1	0.529	1	0.529
Among Refs	b1	0.297	2	0.149
B * Location	a4	0.270	3	0.090
B * Impact	a4-b2	0.227	1	0.227
B * Ref	b2	0.043	2	0.022
T(B) * L	a5	1.710	24	0.071
T(Bef) * Location	c1	0.146	3	0.049
T(Bef) * Impact	c1-d1	0.106	1	0.106
T(Bef) * Ref	d1	0.040	2	0.020
T(Aft) * Location	a5-c1	1.564	21	0.074
T(Aft) * Impact	a5-c1-b3+d1	1.121	7	0.160
T(Aft) * Ref	b3-d1	0.443	14	0.032
Residual	a6	3.760	113	0.033
Total	a7	10.157	154	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	0.951	1.75	0.510
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,113	4.813	2.075	<0.00001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	1.582	39.4	0.45
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	1.511	948	0.56
NO… change was not coincident with impact				

TAXON RICHNESS

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	120.293	1	120.293	a1
Times(B)	1336.507	8	167.063	a2
Locations	68.836	3	22.945	a3
B * L	32.078	3	10.693	a4
T(B) * L	126.535	24	5.272	a5
Residual	267.379	115	2.325	a6
Total	1951.628	154		a7
B. Exclude Impact Site				
Bef/Aft - B	50.135	1	50.135	
Times(B)	820.065	8	102.508	
Locations	46.310	2	23.155	b1
B * L	10.698	2	5.349	b2
T(B) * L	85.735	16	5.358	b3
Residual	168.000	66	2.545	
Total	1180.943	95		
C. All Locations Before Impact				
Times(B)	141.440	1	141.440	
Locations	55.699	3	18.566	
B * L	15.930	3	5.310	c1
Residual	110.833	22	5.038	
Total	323.902	29		
D. Reference Locations Before Impact				
Times(B)	92.182	1	92.182	
Locations	29.896	2	14.948	
B * L	15.263	2	7.632	d1
Residual	95.333	15	6.356	
Total	232.674	20		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	120.293	1	120.293
T(B)	a2	1336.507	8	167.063
Location	a3	68.836	3	22.945
Impact vs Ref	a3-b1	22.526	1	22.526
Among Refs	b1	46.310	2	23.155
B * Location	a4	32.078	3	10.693
B * Impact	a4-b2	21.380	1	21.380
B * Ref	b2	10.698	2	5.349
T(B) * L	a5	126.535	24	5.272
T(Bef) * Location	c1	15.930	3	5.310
T(Bef) * Impact	c1-d1	0.667	1	0.667
T(Bef) * Ref	d1	15.263	2	7.632
T(Aft) * Location	a5-c1	110.605	21	5.267
T(Aft) * Impact	a5-c1-b3+d1	40.133	7	5.733
T(Aft) * Ref	b3-d1	70.472	14	5.034
Residual	a6	267.379	113	2.366
Total	a7	1951.628	154	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	2.127	1.782	0.015
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	1.139	2.76	0.39
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	2.423
Degrees of Freedom	7,113
Falt = Fcrit / (1+n0) = 2.09 / (1+n0)	0.863
Power (based on F-distribution)	0.538

NUMBER OF TAXA REPRESENTED BY EPHEMEROPTERA, PLECOPTERA, TRICOPTERA

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	109.014	1	109.014	a1
Times(B)	771.526	8	96.441	a2
Locations	45.659	3	15.220	a3
B * L	18.461	3	6.154	a4
T(B) * L	81.805	24	3.409	a5
Residual	168.061	115	1.461	a6
Total	1194.526	154		a7
B. Exclude Impact Site				
Bef/Aft - B	50.653	1	50.653	
Times(B)	488.736	8	61.092	
Locations	33.915	2	16.958	b1
B * L	5.869	2	2.935	b2
T(B) * L	50.551	16	3.159	b3
Residual	114.667	66	1.737	
Total	744.391	95		
C. All Locations Before Impact				
Times(B)	50.298	1	50.298	
Locations	35.604	3	11.868	
B * L	11.628	3	3.876	c1
Residual	70.500	22	3.205	
Total	168.030	29		
D. Reference Locations Before Impact				
Times(B)	36.379	1	36.379	
Locations	21.429	2	10.715	
B * L	11.596	2	5.798	d1
Residual	59.167	15	3.944	
Total	128.571	20		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	109.014	1	109.014
T(B)	a2	771.526	8	96.441
Location	a3	45.659	3	15.220
Impact vs Ref	a3-b1	11.744	1	11.744
Among Refs	b1	33.915	2	16.958
B * Location	a4	18.461	3	6.154
B * Impact	a4-b2	12.592	1	12.592
B * Ref	b2	5.869	2	2.935
T(B) * L	a5	81.805	24	3.409
T(Bef) * Location	c1	11.628	3	3.876
T(Bef) * Impact	c1-d1	0.032	1	0.032
T(Bef) * Ref	d1	11.596	2	5.798
T(Aft) * Location	a5-c1	70.177	21	3.342
T(Aft) * Impact	a5-c1-b3+d1	31.222	7	4.460
T(Aft) * Ref	b3-d1	38.955	14	2.783
Residual	a6	168.061	113	1.487
Total	a7	1194.526	154	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	1.871	1.782	0.037
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	1.603	2.76	0.21
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	2.999
Degrees of Freedom	7,113
Falt = Fcrit / (1+n0) = 2.09 / (1+n0)	0.697
Power (based on F-distribution)	0.674

SIMPSON'S DIVERSITY

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.075	1	0.075	a1
Times(B)	2.863	8	0.358	a2
Locations	0.502	3	0.167	a3
B * L	0.137	3	0.046	a4
T(B) * L	1.198	24	0.050	a5
Residual	2.633	115	0.023	a6
Total	7.408	154		a7
B. Exclude Impact Site				
Bef/Aft - B	0.005	1	0.005	
Times(B)	2.318	8	0.290	
Locations	0.379	2	0.190	b1
B * L	0.020	2	0.010	b2
T(B) * L	0.747	16	0.047	b3
Residual	1.060	66	0.016	
Total	4.529	95		
C. All Locations Before Impact				
Times(B)	0.638	1	0.638	
Locations	0.318	3	0.106	
B * L	0.013	3	0.004	c1
Residual	0.427	22	0.019	
Total	1.396	29		
D. Reference Locations Before Impact				
Times(B)	0.436	1	0.436	
Locations	0.179	2	0.090	
B * L	0.012	2	0.006	d1
Residual	0.362	15	0.024	
Total	0.989	20		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.075	1	0.075
T(B)	a2	2.863	8	0.358
Location	a3	0.502	3	0.167
Impact vs Ref	a3-b1	0.123	1	0.123
Among Refs	b1	0.379	2	0.190
B * Location	a4	0.137	3	0.046
B * Impact	a4-b2	0.117	1	0.117
B * Ref	b2	0.020	2	0.010
T(B) * L	a5	1.198	24	0.050
T(Bef) * Location	c1	0.013	3	0.004
T(Bef) * Impact	c1-d1	0.001	1	0.001
T(Bef) * Ref	d1	0.012	2	0.006
T(Aft) * Location	a5-c1	1.185	21	0.056
T(Aft) * Impact	a5-c1-b3+d1	0.450	7	0.064
T(Aft) * Ref	b3-d1	0.735	14	0.053
Residual	a6	2.633	113	0.023
Total	a7	7.408	154	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	2.253	1.782	0.008
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	1.224	2.76	0.35
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	2.759
Degrees of Freedom	7,113
Falt = Fcrit / (1+n0) = 2.09 / (1+n0)	0.758
Power (based on F-distribution)	0.624

SIMPSON'S EVENNESS

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.248	1	0.248	a1
Times(B)	12.449	8	1.556	a2
Locations	0.112	3	0.037	a3
B * L	0.066	3	0.022	a4
T(B) * L	0.767	24	0.032	a5
Residual	1.314	115	0.011	a6
Total	14.956	154		a7
B. Exclude Impact Site				
Bef/Aft - B	0.178	1	0.178	
Times(B)	8.052	8	1.007	
Locations	0.106	2	0.053	b1
B * L	0.066	2	0.033	b2
T(B) * L	0.334	16	0.021	b3
Residual	0.586	66	0.009	
Total	9.322	95		
C. All Locations Before Impact				
Times(B)	1.533	1	1.533	
Locations	0.097	3	0.032	
B * L	0.066	3	0.022	c1
Residual	0.254	22	0.012	
Total	1.950	29		
D. Reference Locations Before Impact				
Times(B)	0.981	1	0.981	
Locations	0.095	2	0.048	
B * L	0.053	2	0.027	d1
Residual	0.220	15	0.015	
Total	1.349	20		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.248	1	0.248
T(B)	a2	12.449	8	1.556
Location	a3	0.112	3	0.037
Impact vs Ref	a3-b1	0.006	1	0.006
Among Refs	b1	0.106	2	0.053
B * Location	a4	0.066	3	0.022
B * Impact	a4-b2	0.000	1	0.000
B * Ref	b2	0.066	2	0.033
T(B) * L	a5	0.767	24	0.032
T(Bef) * Location	c1	0.066	3	0.022
T(Bef) * Impact	c1-d1	0.013	1	0.013
T(Bef) * Ref	d1	0.053	2	0.027
T(Aft) * Location	a5-c1	0.701	21	0.033
T(Aft) * Impact	a5-c1-b3+d1	0.420	7	0.060
T(Aft) * Ref	b3-d1	0.281	14	0.020
Residual	a6	1.314	113	0.012
Total	a7	14.956	154	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,113	1.726	1.75	0.06
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,113	5.160	2.075	<0.00001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	0.757	39.4	0.70
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	4.615	948	0.34
NO… change was not coincident with impact				

Appendix L

Invertebrate ANOVA Results – Common Family Groups

BAETIDAE ABUNDANCE

A. All Data				
Source Of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.489	1	0.489	a1
Times(B)	46.166	8	5.771	a2
Locations	2.090	3	0.697	a3
B * L	0.245	3	0.082	a4
T(B) * L	4.552	24	0.190	a5
Residual	13.144	157	0.084	a6
Total	66.686	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.195	1	0.195	
Times(B)	26.888	8	3.361	
Locations	0.126	2	0.063	b1
B * L	0.137	2	0.069	b2
T(B) * L	2.111	16	0.132	b3
Residual	8.631	105	0.082	
Total	38.088	134		
C. All Locations Before Impact				
Times(B)	6.889	1	6.889	
Locations	1.000	3	0.333	
B * L	1.218	3	0.406	c1
Residual	5.185	34	0.153	
Total	14.292	41		
D. Reference Locations Before Impact				
Times(B)	3.024	1	3.024	
Locations	0.063	2	0.032	
B * L	0.172	2	0.086	d1
Residual	4.905	27	0.182	
Total	8.164	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.489	1	0.489
T(B)	a2	46.166	8	5.771
Location	a3	2.090	3	0.697
Impact vs Ref	a3-b1	1.964	1	1.964
Among Refs	b1	0.126	2	0.063
B * Location	a4	0.245	3	0.082
B * Impact	a4-b2	0.108	1	0.108
B * Ref	b2	0.137	2	0.069
T(B) * L	a5	4.552	24	0.190
T(Bef) * Location	c1	1.218	3	0.406
T(Bef) * Impact	c1-d1	1.046	1	1.046
T(Bef) * Ref	d1	0.172	2	0.086
T(Aft) * Location	a5-c1	3.334	21	0.159
T(Aft) * Impact	a5-c1-b3+d1	1.395	7	0.199
T(Aft) * Ref	b3-d1	1.939	14	0.139
Residual	a6	13.144	156	0.084
Total	a7	66.686	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	1.644	1.75	0.073
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	2.365	2.075	0.03
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	1.610	39.4	0.45
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	0.191	948	0.94
NO… change was not coincident with impact				

HEPTAGENIIDAE ABUNDANCE

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	12.404	1	12.404	a1
Times(B)	31.283	8	3.910	a2
Locations	2.766	3	0.922	a3
B * L	1.363	3	0.454	a4
T(B) * L	8.154	24	0.340	a5
Residual	17.188	157	0.109	a6
Total	73.158	196		a7
B. Exclude Impact Site				
Bef/Aft - B	7.300	1	7.300	
Times(B)	16.524	8	2.066	
Locations	0.787	2	0.394	b1
B * L	1.068	2	0.534	b2
T(B) * L	5.191	16	0.324	b3
Residual	12.925	105	0.123	
Total	43.795	134		
C. All Locations Before Impact				
Times(B)	1.395	1	1.395	
Locations	1.249	3	0.416	
B * L	1.165	3	0.388	c1
Residual	8.081	34	0.238	
Total	11.89	41		
D. Reference Locations Before Impact				
Times(B)	1.208	1	1.208	
Locations	0.122	2	0.061	
B * L	1.051	2	0.526	d1
Residual	7.373	27	0.273	
Total	9.754	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	12.404	1	12.404
T(B)	a2	31.283	8	3.910
Location	a3	2.766	3	0.922
Impact vs Ref	a3-b1	1.979	1	1.979
Among Refs	b1	0.787	2	0.394
B * Location	a4	1.363	3	0.454
B * Impact	a4-b2	0.295	1	0.295
B * Ref	b2	1.068	2	0.534
T(B) * L	a5	8.154	24	0.340
T(Bef) * Location	c1	1.165	3	0.388
T(Bef) * Impact	c1-d1	0.114	1	0.114
T(Bef) * Ref	d1	1.051	2	0.526
T(Aft) * Location	a5-c1	6.989	21	0.333
T(Aft) * Impact	a5-c1-b3+d1	2.849	7	0.407
T(Aft) * Ref	b3-d1	4.140	14	0.296
Residual	a6	17.188	156	0.110
Total	a7	73.158	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	2.684	1.75	0.002
YES…				
2. Does scalping affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	1.376	2.76	0.29
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	3.694
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	0.560
Power (based on F-distribution)	0.787

EPHEMERELLIDAE ABUNDANCE

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	13.772	1	13.772	a1
Times(B)	21.961	8	2.745	a2
Locations	0.679	3	0.226	a3
B * L	0.465	3	0.155	a4
T(B) * L	11.102	24	0.463	a5
Residual	10.758	157	0.069	a6
Total	58.737	196		a7
B. Exclude Impact Site				
Bef/Aft - B	10.217	1	10.217	
Times(B)	14.655	8	1.832	
Locations	0.643	2	0.322	b1
B * L	0.464	2	0.232	b2
T(B) * L	9.826	16	0.614	b3
Residual	8.402	105	0.080	
Total	44.207	134		
C. All Locations Before Impact				
Times(B)	1.107	1	1.107	
Locations	0.030	3	0.010	
B * L	4.582	3	1.527	c1
Residual	4.110	34	0.121	
Total	9.829	41		
D. Reference Locations Before Impact				
Times(B)	0.109	1	0.109	
Locations	0.023	2	0.012	
B * L	3.949	2	1.975	d1
Residual	3.412	27	0.126	
Total	7.493	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	13.772	1	13.772
T(B)	a2	21.961	8	2.745
Location	a3	0.679	3	0.226
Impact vs Ref	a3-b1	0.036	1	0.036
Among Refs	b1	0.643	2	0.322
B * Location	a4	0.465	3	0.155
B * Impact	a4-b2	0.001	1	0.001
B * Ref	b2	0.464	2	0.232
T(B) * L	a5	11.102	24	0.463
T(Bef) * Location	c1	4.582	3	1.527
T(Bef) * Impact	c1-d1	0.633	1	0.633
T(Bef) * Ref	d1	3.949	2	1.975
T(Aft) * Location	a5-c1	6.520	21	0.310
T(Aft) * Impact	a5-c1-b3+d1	0.643	7	0.092
T(Aft) * Ref	b3-d1	5.877	14	0.420
Residual	a6	10.758	156	0.069
Total	a7	58.737	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	6.087	1.75	<0.0005
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	0.219	2.76	0.98
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.332
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	1.554
Power (based on F-distribution)	0.153

CAPNIIDAE ABUNDANCE

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	5.220	1	5.220	a1
Times(B)	25.844	8	3.231	a2
Locations	0.214	3	0.071	a3
B * L	0.163	3	0.054	a4
T(B) * L	4.031	24	0.168	a5
Residual	12.618	157	0.080	a6
Total	48.090	196		a7
B. Exclude Impact Site				
Bef/Aft - B	3.745	1	3.745	
Times(B)	14.685	8	1.836	
Locations	0.029	2	0.015	b1
B * L	0.152	2	0.076	b2
T(B) * L	2.720	16	0.170	b3
Residual	9.955	105	0.095	
Total	31.286	134		
C. All Locations Before Impact				
Times(B)	3.504	1	3.504	
Locations	0.177	3	0.059	
B * L	1.583	3	0.528	c1
Residual	5.012	34	0.147	
Total	10.276	41		
D. Reference Locations Before Impact				
Times(B)	1.319	1	1.319	
Locations	0.088	2	0.044	
B * L	0.883	2	0.442	d1
Residual	4.396	27	0.163	
Total	6.686	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	5.220	1	5.220
T(B)	a2	25.844	8	3.231
Location	a3	0.214	3	0.071
Impact vs Ref	a3-b1	0.185	1	0.185
Among Refs	b1	0.029	2	0.015
B * Location	a4	0.163	3	0.054
B * Impact	a4-b2	0.011	1	0.011
B * Ref	b2	0.152	2	0.076
T(B) * L	a5	4.031	24	0.168
T(Bef) * Location	c1	1.583	3	0.528
T(Bef) * Impact	c1-d1	0.700	1	0.700
T(Bef) * Ref	d1	0.883	2	0.442
T(Aft) * Location	a5-c1	2.448	21	0.117
T(Aft) * Impact	a5-c1-b3+d1	0.611	7	0.087
T(Aft) * Ref	b3-d1	1.837	14	0.131
Residual	a6	12.618	156	0.081
Total	a7	48.090	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	1.622	1.75	0.08
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	1.079	2.075	0.43
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,156	0.940	3.05	0.39
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,156	0.136	3.9	0.71
NO impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.079
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	1.918
Power (based on F-distribution)	0.072

CHIRONOMIDAE ABUNDANCE

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	5.440	1	5.440	a1
Times(B)	122.079	8	15.260	a2
Locations	0.897	3	0.299	a3
B * L	1.021	3	0.340	a4
T(B) * L	5.965	24	0.249	a5
Residual	11.040	157	0.070	a6
Total	146.442	196		a7
B. Exclude Impact Site				
Bef/Aft - B	5.689	1	5.689	
Times(B)	77.639	8	9.705	
Locations	0.869	2	0.435	b1
B * L	0.474	2	0.237	b2
T(B) * L	2.884	16	0.180	b3
Residual	7.549	105	0.072	
Total	95.104	134		
C. All Locations Before Impact				
Times(B)	15.260	1	15.260	
Locations	0.955	3	0.318	
B * L	1.532	3	0.511	c1
Residual	1.711	34	0.050	
Total	19.458	41		
D. Reference Locations Before Impact				
Times(B)	9.043	1	9.043	
Locations	0.724	2	0.362	
B * L	1.138	2	0.569	d1
Residual	1.633	27	0.060	
Total	12.538	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	5.440	1	5.440
T(B)	a2	122.079	8	15.260
Location	a3	0.897	3	0.299
Impact vs Ref	a3-b1	0.028	1	0.028
Among Refs	b1	0.869	2	0.435
B * Location	a4	1.021	3	0.340
B * Impact	a4-b2	0.547	1	0.547
B * Ref	b2	0.474	2	0.237
T(B) * L	a5	5.965	24	0.249
T(Bef) * Location	c1	1.532	3	0.511
T(Bef) * Impact	c1-d1	0.394	1	0.394
T(Bef) * Ref	d1	1.138	2	0.569
T(Aft) * Location	a5-c1	4.433	21	0.211
T(Aft) * Impact	a5-c1-b3+d1	2.687	7	0.384
T(Aft) * Ref	b3-d1	1.746	14	0.125
Residual	a6	11.040	156	0.071
Total	a7	146.442	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	1.762	1.75	0.05
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,156	5.424	2.075	<0.0001
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,2	0.219	39.4	0.97
NO… change was associated with Impact site				
3B. Was timing of change coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,1	0.974	948	0.660
NO… change was not coincident with impact				

NEMATODA ABUNDANCE

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.018	1	0.018	a1
Times(B)	20.527	8	2.566	a2
Locations	0.111	3	0.037	a3
B * L	0.065	3	0.022	a4
T(B) * L	2.187	24	0.091	a5
Residual	8.029	157	0.051	a6
Total	30.937	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.007	1	0.007	
Times(B)	14.821	8	1.853	
Locations	0.089	2	0.045	b1
B * L	0.064	2	0.032	b2
T(B) * L	1.678	16	0.105	b3
Residual	6.434	105	0.061	
Total	23.093	134		
C. All Locations Before Impact				
Times(B)	3.444	1	3.444	
Locations	0.017	3	0.006	
B * L	0.018	3	0.006	c1
Residual	4.663	34	0.137	
Total	8.142	41		
D. Reference Locations Before Impact				
Times(B)	2.493	1	2.493	
Locations	0.012	2	0.006	
B * L	0.011	2	0.006	d1
Residual	4.254	27	0.158	
Total	6.77	32		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.018	1	0.018
T(B)	a2	20.527	8	2.566
Location	a3	0.111	3	0.037
Impact vs Ref	a3-b1	0.022	1	0.022
Among Refs	b1	0.089	2	0.045
B * Location	a4	0.065	3	0.022
B * Impact	a4-b2	0.001	1	0.001
B * Ref	b2	0.064	2	0.032
T(B) * L	a5	2.187	24	0.091
T(Bef) * Location	c1	0.02	3	0.006
T(Bef) * Impact	c1-d1	0.01	1	0.007
T(Bef) * Ref	d1	0.01	2	0.006
T(Aft) * Location	a5-c1	2.169	21	0.103
T(Aft) * Impact	a5-c1-b3+d1	0.502	7	0.072
T(Aft) * Ref	b3-d1	1.667	14	0.119
Residual	a6	8.029	156	0.051
Total	a7	30.9	196	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	2.314	1.75	0.006
YES…				
2. Does scalping affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	0.602	2.76	0.74
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.393
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	1.486
Power (based on F-distribution)	0.176

OLIGOCHAETA ABUNDANCE

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	1.658	1	1.658	a1
Times(B)	10.970	8	1.371	a2
Locations	0.490	3	0.163	a3
B * L	2.349	3	0.783	a4
T(B) * L	9.474	24	0.395	a5
Residual	16.621	157	0.106	a6
Total	41.562	196		a7
B. Exclude Impact Site				
Bef/Aft - B	0.861	1	0.861	
Times(B)	5.667	8	0.708	
Locations	0.324	2	0.162	b1
B * L	2.045	2	1.023	b2
T(B) * L	6.594	16	0.412	b3
Residual	12.109	105	0.115	
Total	27.600	134		
C. All Locations Before Impact				
Times(B)	0.305	1	0.305	
Locations	1.371	3	0.457	
B * L	1.481	3	0.494	c1
Residual	5.988	34	0.176	
Total	9.145	41		
D. Reference Locations Before Impact				
Times(B)	0.100	1	0.100	
Locations	1.014	2	0.507	
B * L	1.476	2	0.738	d1
Residual	5.140	27	0.190	
Total	7.730	32		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	1.658	1	1.658
T(B)	a2	10.970	8	1.371
Location	a3	0.490	3	0.163
Impact vs Ref	a3-b1	0.166	1	0.166
Among Refs	b1	0.324	2	0.162
B * Location	a4	2.349	3	0.783
B * Impact	a4-b2	0.304	1	0.304
B * Ref	b2	2.045	2	1.023
T(B) * L	a5	9.474	24	0.395
T(Bef) * Location	c1	1.481	3	0.494
T(Bef) * Impact	c1-d1	0.005	1	0.005
T(Bef) * Ref	d1	1.476	2	0.738
T(Aft) * Location	a5-c1	7.993	21	0.381
T(Aft) * Impact	a5-c1-b3+d1	2.875	7	0.411
T(Aft) * Ref	b3-d1	5.118	14	0.366
Residual	a6	16.621	156	0.107
Total	a7	41.562	196	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,156	3.431	1.75	<0.0005
YES…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / T(Aft) * Ref	7,14	1.123	2.76	0.40
NO… no short-term impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	3.855
Degrees of Freedom	7,156
Falt = Fcrit / (1+n0) = 2.07 / (1+n0)	0.537
Power (based on F-distribution)	0.805

Appendix M

Fish ANOVA Results – Whole Bar Unit

FISH DENSITY

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.019	1	0.019	a1
Times(B)	0.884	9	0.098	a2
Locations	0.184	3	0.061	a3
B * L	0.113	3	0.038	a4
T(B) * L	1.231	27	0.046	a5
Residual	14.965	198	0.076	a6
Total	17.396	241		a7
B. Exclude Impact Site				
Bef/Aft - B	0.056	1	0.056	
Times(B)	0.732	9	0.081	
Locations	0.171	2	0.086	b1
B * L	0.050	2	0.025	b2
T(B) * L	0.464	18	0.026	b3
Residual	9.304	139	0.067	
Total	10.777	171		
C. All Locations Before Impact				
Times(B)	0.292	2	0.146	
Locations	0.158	3	0.053	
B * L	0.121	6	0.020	c1
Residual	1.359	33	0.041	
Total	1.930	44		
D. Reference Locations Before Impact				
Times(B)	0.251	2	0.126	
Locations	0.118	2	0.059	
B * L	0.101	4	0.025	d1
Residual	1.311	22	0.060	
Total	1.781	30		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.391
Degrees of Freedom	7,195
Falt = Fcrit / (1+n0) = 2.06 / (1+n0)	1.481
Power (based on F-distribution)	0.176

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.019	1	0.019
T(B)	a2	0.884	9	0.098
Location	a3	0.184	3	0.061
Impact vs Ref	a3-b1	0.013	1	0.013
Among Refs	b1	0.171	2	0.086
B * Location	a4	0.113	3	0.038
B * Impact	a4-b2	0.063	1	0.063
B * Ref	b2	0.050	2	0.025
T(B) * L	a5	1.231	27	0.046
T(Bef) * Location	c1	0.121	6	0.020
T(Bef) * Impact	c1-d1	0.020	2	0.010
T(Bef) * Ref	d1	0.101	4	0.025
T(Aft) * Location	a5-c1	1.110	21	0.053
T(Aft) * Impact	a5-c1-b3+d1	0.747	7	0.107
T(Aft) * Ref	b3-d1	0.363	14	0.026
Residual	a6	14.965	195	0.077
Total	a7	17.396	241	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,195	0.338	1.93	0.98
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,195	1.391	2.35	0.21
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,195	0.326	3.76	0.72
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,195	0.821	5.1	0.37
NO…no impact detected				

PERCENT REPRESENTATION BY SALMON

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.610	1	0.610	a1
Times(B)	5.758	9	0.640	a2
Locations	0.713	3	0.238	a3
B * L	0.327	3	0.109	a4
T(B) * L	3.661	27	0.136	a5
Residual	25.510	198	0.129	a6
Total	36.579	241		a7
B. Exclude Impact Site				
Bef/Aft - B	0.234	1	0.234	
Times(B)	3.268	9	0.363	
Locations	0.077	2	0.039	b1
B * L	0.166	2	0.083	b2
T(B) * L	2.630	18	0.146	b3
Residual	20.304	139	0.146	
Total	26.679	171		
C. All Locations Before Impact				
Times(B)	0.032	2	0.016	
Locations	0.511	3	0.170	
B * L	0.115	6	0.019	c1
Residual	4.709	33	0.143	
Total	5.367	44		
D. Reference Locations Before Impact				
Times(B)	0.075	2	0.038	
Locations	0.023	2	0.012	
B * L	0.101	4	0.025	d1
Residual	4.668	22	0.212	
Total	4.867	30		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.111
Degrees of Freedom	7,195
Falt = Fcrit / (1+n0) = 2.06 / (1+n0)	1.855
Power (based on F-distribution)	0.079

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.610	1	0.610
T(B)	a2	5.758	9	0.640
Location	a3	0.713	3	0.238
Impact vs Ref	a3-b1	0.636	1	0.636
Among Refs	b1	0.077	2	0.039
B * Location	a4	0.327	3	0.109
B * Impact	a4-b2	0.161	1	0.161
B * Ref	b2	0.166	2	0.083
T(B) * L	a5	3.661	27	0.136
T(Bef) * Location	c1	0.115	6	0.019
T(Bef) * Impact	c1-d1	0.014	2	0.007
T(Bef) * Ref	d1	0.101	4	0.025
T(Aft) * Location	a5-c1	3.546	21	0.169
T(Aft) * Impact	a5-c1-b3+d1	1.017	7	0.145
T(Aft) * Ref	b3-d1	2.529	14	0.181
Residual	a6	25.510	195	0.131
Total	a7	36.579	241	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,195	1.381	1.93	0.16
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,195	1.111	2.35	0.35
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,195	0.634	3.76	0.53
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,195	1.231	5.1	0.27
NO…no impact detected				

SPECIES RICHNESS

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	12.077	1	12.077	a1
Times(B)	393.173	9	43.686	a2
Locations	2.368	3	0.789	a3
B * L	5.316	3	1.772	a4
T(B) * L	154.425	27	5.719	a5
Residual	733.625	198	3.705	a6
Total	1300.984	241		a7
B. Exclude Impact Site				
Bef/Aft - B	10.637	1	10.637	
Times(B)	323.943	9	35.994	
Locations	0.84	2	0.420	b1
B * L	4.693	2	2.347	b2
T(B) * L	92.76	18	5.153	b3
Residual	536.806	139	3.862	
Total	969.679	171		
C. All Locations Before Impact				
Times(B)	141.771	2	70.886	
Locations	0.682	3	0.227	
B * L	13.004	6	2.167	c1
Residual	68.433	33	2.074	
Total	223.89	44		
D. Reference Locations Before Impact				
Times(B)	92.801	2	46.401	
Locations	0.629	2	0.315	
B * L	1.101	4	0.275	d1
Residual	49.6	22	2.255	
Total	144.131	30		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.890
Degrees of Freedom	7,195
Falt = Fcrit / (1+n0) = 2.06 / (1+n0)	1.090
Power (based on F-distribution)	0.371

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	12.077	1	12.077
T(B)	a2	393.173	9	43.686
Location	a3	2.368	3	0.789
Impact vs Ref	a3-b1	1.528	1	1.528
Among Refs	b1	0.840	2	0.420
B * Location	a4	5.316	3	1.772
B * Impact	a4-b2	0.623	1	0.623
B * Ref	b2	4.693	2	2.347
T(B) * L	a5	154.425	27	5.719
T(Bef) * Location	c1	13.004	6	2.167
T(Bef) * Impact	c1-d1	11.903	2	5.952
T(Bef) * Ref	d1	1.101	4	0.275
T(Aft) * Location	a5-c1	141.421	21	6.734
T(Aft) * Impact	a5-c1-b3+d1	49.762	7	7.109
T(Aft) * Ref	b3-d1	91.659	14	6.547
Residual	a6	733.625	195	3.762
Total	a7	1300.984	241	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,195	1.740	1.93	0.06
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,195	1.890	2.35	0.07
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,195	0.624	3.76	0.53
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,195	0.166	5.1	0.68
NO…no impact detected				

SIMPSON'S DIVERSITY

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.065	1	0.065	a1
Times(B)	1.122	9	0.125	a2
Locations	0.042	3	0.014	a3
B * L	0.032	3	0.011	a4
T(B) * L	1.894	27	0.070	a5
Residual	9.826	198	0.050	a6
Total	12.981	241		a7
B. Exclude Impact Site				
Bef/Aft - B	0.020	1	0.020	
Times(B)	0.690	9	0.077	
Locations	0.018	2	0.009	b1
B * L	0.008	2	0.004	b2
T(B) * L	1.143	18	0.064	b3
Residual	6.963	139	0.050	
Total	8.842	171		
C. All Locations Before Impact				
Times(B)	0.497	2	0.249	
Locations	0.003	3	0.001	
B * L	0.219	6	0.037	c1
Residual	1.819	33	0.055	
Total	2.538	44		
D. Reference Locations Before Impact				
Times(B)	0.197	2	0.099	
Locations	0.003	2	0.002	
B * L	0.035	4	0.009	d1
Residual	1.464	22	0.067	
Total	1.699	30		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.607
Degrees of Freedom	7,195
Falt = Fcrit / (1+n0) = 2.06 / (1+n0)	1.282
Power (based on F-distribution)	0.261

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.065	1	0.065
T(B)	a2	1.122	9	0.125
Location	a3	0.042	3	0.014
Impact vs Ref	a3-b1	0.024	1	0.024
Among Refs	b1	0.018	2	0.009
B * Location	a4	0.032	3	0.011
B * Impact	a4-b2	0.024	1	0.024
B * Ref	b2	0.008	2	0.004
T(B) * L	a5	1.894	27	0.070
T(Bef) * Location	c1	0.219	6	0.037
T(Bef) * Impact	c1-d1	0.184	2	0.092
T(Bef) * Ref	d1	0.035	4	0.009
T(Aft) * Location	a5-c1	1.675	21	0.080
T(Aft) * Impact	a5-c1-b3+d1	0.567	7	0.081
T(Aft) * Ref	b3-d1	1.108	14	0.079
Residual	a6	9.826	195	0.050
Total	a7	12.981	241	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,195	1.571	1.93	0.08
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,195	1.607	2.35	0.13
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,195	0.079	3.76	0.92
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,195	0.476	5.1	0.49
NO…no impact detected				

SIMPSON'S EVENNESS

Source of Variance	SS	df	MS	Term
A. All Data				
Bef/Aft - B	0.117	1	0.117	a1
Times(B)	0.126	9	0.014	a2
Locations	0.006	3	0.002	a3
B * L	0.007	3	0.002	a4
T(B) * L	0.255	27	0.009	a5
Residual	1.042	198	0.005	a6
Total	1.553	241		a7
B. Exclude Impact Site				
Bef/Aft - B	0.077	1	0.077	
Times(B)	0.045	9	0.005	
Locations	0.000	2	0.000	b1
B * L	0.006	2	0.003	b2
T(B) * L	0.083	18	0.005	b3
Residual	0.755	139	0.005	
Total	0.966	171		
C. All Locations Before Impact				
Times(B)	0.088	2	0.044	
Locations	0.005	3	0.002	
B * L	0.043	6	0.007	c1
Residual	0.138	33	0.004	
Total	0.274	44		
D. Reference Locations Before Impact				
Times(B)	0.037	2	0.019	
Locations	0.002	2	0.001	
B * L	0.014	4	0.004	d1
Residual	0.117	22	0.005	
Total	0.170	30		

Source of Variation	Term	SS	df	MS
B (bef-aft)	a1	0.117	1	0.117
T(B)	a2	0.126	9	0.014
Location	a3	0.006	3	0.002
Impact vs Ref	a3-b1	0.006	1	0.006
Among Refs	b1	0.000	2	0.000
B * Location	a4	0.007	3	0.002
B * Impact	a4-b2	0.001	1	0.001
B * Ref	b2	0.006	2	0.003
T(B) * L	a5	0.255	27	0.009
T(Bef) * Location	c1	0.043	6	0.007
T(Bef) * Impact	c1-d1	0.029	2	0.015
T(Bef) * Ref	d1	0.014	4	0.004
T(Aft) * Location	a5-c1	0.212	21	0.010
T(Aft) * Impact	a5-c1-b3+d1	0.143	7	0.020
T(Aft) * Ref	b3-d1	0.069	14	0.005
Residual	a6	1.042	195	0.005
Total	a7	1.553	241	

	df	F	Fcrit	p
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,195	0.922	1.93	0.52
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,195	3.823	2.075	<0.001
YES… short-term impact detected				
3A. Are changes associated with impact site?				
T(Aft) * Ref / T(Bef) * Ref	14,4	1.408	39.4	0.40
YES… changes are associated with scalped site				
3B. Timing of change was coincident with impact?				
T(Aft) * Imp / T(Bef) * Imp	7,2	1.409	948	0.48
NO… timing of change was not coincident with impact				

Appendix N

Fish ANOVA Results – Bar Edge Units

FISH DENSITY

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.070	1	0.070	a1
Times(B)	0.297	9	0.033	a2
Locations	0.008	3	0.003	a3
B * L	0.013	3	0.004	a4
T(B) * L	0.137	27	0.005	a5
Residual	0.779	80	0.010	a6
Total	1.304	123		a7
B. Exclude Impact Site				
Bef/Aft - B	0.060	1	0.060	
Times(B)	0.202	9	0.022	
Locations	0.008	2	0.004	b1
B * L	0.011	2	0.006	b2
T(B) * L	0.066	18	0.004	b3
Residual	0.428	53	0.008	
Total	0.775	85		
C. All Locations Before Impact				
Times(B)	0.172	2	0.086	
Locations	0.012	3	0.004	
B * L	0.054	6	0.009	c1
Residual	0.040	13	0.003	
Total	0.278	24		
D. Reference Locations Before Impact				
Times(B)	0.116	2	0.058	
Locations	0.012	2	0.006	
B * L	0.054	4	0.014	d1
Residual	0.025	6	0.004	
Total	0.207	14		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.070	1	0.070
T(B)	a2	0.297	9	0.033
Location	a3	0.008	3	0.003
Impact vs Ref	a3-b1	0.000	1	0.000
Among Refs	b1	0.008	2	0.004
B * Location	a4	0.013	3	0.004
B * Impact	a4-b2	0.002	1	0.002
B * Ref	b2	0.011	2	0.006
T(B) * L	a5	0.137	27	0.005
T(Bef) * Location	c1	0.054	6	0.009
T(Bef) * Impact	c1-d1	0.000	2	0.000
T(Bef) * Ref	d1	0.054	4	0.014
T(Aft) * Location	a5-c1	0.083	21	0.004
T(Aft) * Impact	a5-c1-b3+d1	0.071	7	0.010
T(Aft) * Ref	b3-d1	0.012	14	*0.001*
Residual	a6	0.779	74	0.011
Total	a7	1.304	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	0.081	1.93	0.99
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	0.964	2.35	0.46
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,74	0.522	3.76	0.60
NO…				
4. Does scalping affect differences from before to after?				
B * Imp / Residual	1,74	0.190	5.1	0.66
NO…no impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	0.964
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	2.219
Power (based on F-distribution)	0.042

PROPORTION REPRESENTED BY SALMON

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.808	1	0.808	a1
Times(B)	1.289	9	0.143	a2
Locations	0.173	3	0.058	a3
B * L	0.488	3	0.163	a4
T(B) * L	5.306	27	0.197	a5
Residual	12.981	80	0.162	a6
Total	21.045	123		a7
B. Exclude Impact Site				
Bef/Aft - B	1.109	1	1.109	
Times(B)	1.335	9	0.148	
Locations	0.081	2	0.041	b1
B * L	0.049	2	0.025	b2
T(B) * L	1.699	18	0.094	b3
Residual	10.32	53	0.195	
Total	14.593	85		
C. All Locations Before Impact				
Times(B)	0.202	2	0.101	
Locations	0.364	3	0.121	
B * L	0.302	6	0.050	c1
Residual	2.165	13	0.167	
Total	3.033	24		
D. Reference Locations Before Impact				
Times(B)	0.275	2	0.138	
Locations	0.061	2	0.031	
B * L	0.071	4	0.018	d1
Residual	1.44	6	0.240	
Total	1.847	14		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.808	1	0.808
T(B)	a2	1.289	9	0.143
Location	a3	0.173	3	0.058
Impact vs Ref	a3-b1	0.092	1	0.092
Among Refs	b1	0.081	2	0.041
B * Location	a4	0.488	3	0.163
B * Impact	a4-b2	0.439	1	0.439
B * Ref	b2	0.049	2	0.025
T(B) * L	a5	5.306	27	0.197
T(Bef) * Location	c1	0.302	6	0.050
T(Bef) * Impact	c1-d1	0.231	2	0.116
T(Bef) * Ref	d1	0.071	4	0.018
T(Aft) * Location	a5-c1	5.004	21	0.238
T(Aft) * Impact	a5-c1-b3+d1	3.376	7	0.482
T(Aft) * Ref	b3-d1	1.628	14	0.116
Residual	a6	12.981	74	0.175
Total	a7	21.045	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after scalping?				
T(Aft) * Ref / Residual	14,74	0.663	1.93	0.80
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	2.749	2.35	0.01
YES… short-term impact detected				
3A. Was change observed at reference sites?				
T(Aft) * Ref / T(Bef) * Ref	14,4	6.551	39.4	0.04
YES… changes are associated with scalped site				
3B. Was timing of change coincident with scalping?				
T(Aft) * Imp / T(Bef) * Imp	7,2	4.176	948	0.21
NO… change was not coincident with scalping				

SPECIES RICHNESS

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	2.089	1	2.089	a1
Times(B)	291.578	9	32.398	a2
Locations	4.31	3	1.437	a3
B * L	1.124	3	0.375	a4
T(B) * L	62.743	27	2.324	a5
Residual	223.395	80	2.792	a6
Total	585.239	123		a7
B. Exclude Impact Site				
Bef/Aft - B	2.474	1	2.474	
Times(B)	219.268	9	24.363	
Locations	2.713	2	1.357	b1
B * L	0.561	2	0.281	b2
T(B) * L	36.63	18	2.035	b3
Residual	137.895	53	2.602	
Total	399.541	85		
C. All Locations Before Impact				
Times(B)	132.235	2	66.118	
Locations	0.334	3	0.111	
B * L	1.954	6	0.326	c1
Residual	6.3	13	0.485	
Total	140.823	24		
D. Reference Locations Before Impact				
Times(B)	85.253	2	42.627	
Locations	0.249	2	0.125	
B * L	0.473	4	0.118	d1
Residual	1.8	6	0.300	
Total	87.775	14		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	2.089	1	2.089
T(B)	a2	291.578	9	32.398
Location	a3	4.310	3	1.437
Impact vs Ref	a3-b1	1.597	1	1.597
Among Refs	b1	2.713	2	1.357
B * Location	a4	1.124	3	0.375
B * Impact	a4-b2	0.563	1	0.563
B * Ref	b2	0.561	2	0.281
T(B) * L	a5	62.743	27	2.324
T(Bef) * Location	c1	1.954	6	0.326
T(Bef) * Impact	c1-d1	1.481	2	0.741
T(Bef) * Ref	d1	0.473	4	0.118
T(Aft) * Location	a5-c1	60.789	21	2.895
T(Aft) * Impact	a5-c1-b3+d1	24.632	7	3.519
T(Aft) * Ref	b3-d1	36.157	14	2.583
Residual	a6	223.395	74	3.019
Total	a7	585.239	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	0.856	1.93	0.61
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	1.166	2.35	0.33
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,74	0.093	3.76	0.91
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,74	0.186	5.1	0.67
NO…no impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.166
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	1.834
Power (based on F-distribution)	0.093

SIMPSON'S DIVERSITY

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.356	1	0.356	a1
Times(B)	3.253	9	0.361	a2
Locations	0.179	3	0.060	a3
B * L	0.021	3	0.007	a4
T(B) * L	0.513	27	0.019	a5
Residual	2.8	80	0.035	a6
Total	7.122	123		a7
B. Exclude Impact Site				
Bef/Aft - B	0.234	1	0.234	
Times(B)	2.37	9	0.263	
Locations	0.032	2	0.016	b1
B * L	0.02	2	0.010	b2
T(B) * L	0.3	18	0.017	b3
Residual	1.692	53	0.032	
Total	4.648	85		
C. All Locations Before Impact				
Times(B)	1.346	2	0.673	
Locations	0.073	3	0.024	
B * L	0.064	6	0.011	c1
Residual	0.004	13	0.000	
Total	1.487	24		
D. Reference Locations Before Impact				
Times(B)	1.021	2	0.511	
Locations	0.018	2	0.009	
B * L	0.022	4	0.006	d1
Residual	0.002	6	0.000	
Total	1.063	14		

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.356	1	0.356
T(B)	a2	3.253	9	0.361
Location	a3	0.179	3	0.060
Impact vs Ref	a3-b1	0.147	1	0.147
Among Refs	b1	0.032	2	0.016
B * Location	a4	0.021	3	0.007
B * Impact	a4-b2	0.001	1	0.001
B * Ref	b2	0.020	2	0.010
T(B) * L	a5	0.513	27	0.019
T(Bef) * Location	c1	0.064	6	0.011
T(Bef) * Impact	c1-d1	0.042	2	0.021
T(Bef) * Ref	d1	0.022	4	0.006
T(Aft) * Location	a5-c1	0.449	21	0.021
T(Aft) * Impact	a5-c1-b3+d1	0.171	7	0.024
T(Aft) * Ref	b3-d1	0.278	14	0.020
Residual	a6	2.800	74	0.038
Total	a7	7.122	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	0.525	1.93	0.91
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	0.646	2.35	0.72
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,74	0.264	3.76	0.77
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,74	0.026	5.1	0.87
NO…no impact detected				

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	0.646
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	3.312
Power (based on F-distribution)	0.004

SIMPSON'S EVENNESS

A. All Data				
Source of Variance	**SS**	**df**	**MS**	**Term**
Bef/Aft - B	0.000	1	0.000	a1
Times(B)	0.070	9	0.008	a2
Locations	0.010	3	0.003	a3
B * L	0.005	3	0.002	a4
T(B) * L	0.214	27	0.008	a5
Residual	0.464	80	0.006	a6
Total	0.763	123		a7
B. Exclude Impact Site				
Bef/Aft - B	0.000	1	0.000	
Times(B)	0.079	9	0.009	
Locations	0.010	2	0.005	b1
B * L	0.005	2	0.003	b2
T(B) * L	0.160	18	0.009	b3
Residual	0.346	53	0.007	
Total	0.600	85		
C. All Locations Before Impact				
Times(B)	0.001	2	0.001	
Locations	0.001	3	0.000	
B * L	0.015	6	0.003	c1
Residual	0.042	13	0.003	
Total	0.059	24		
D. Reference Locations Before Impact				
Times(B)	0.001	2	0.001	
Locations	0.001	2	0.001	
B * L	0.009	4	0.002	d1
Residual	0.010	6	0.002	
Total	0.021	14		

Power Analysis	
1+n0 = MS[T(Aft) * Impact] / MS[Residual]	1.094
Degrees of Freedom	7,74
Falt = Fcrit / (1+n0) = 2.14 / (1+n0)	1.955
Power (based on F-distribution)	0.073

Source of Variation	**Term**	**SS**	**df**	**MS**
B (bef-aft)	a1	0.000	1	0.000
T(B)	a2	0.070	9	0.008
Location	a3	0.010	3	0.003
Impact vs Ref	a3-b1	0.000	1	0.000
Among Refs	b1	0.010	2	0.005
B * Location	a4	0.005	3	0.002
B * Impact	a4-b2	0.000	1	0.000
B * Ref	b2	0.005	2	0.003
T(B) * L	a5	0.214	27	0.008
T(Bef) * Location	c1	0.015	6	0.003
T(Bef) * Impact	c1-d1	0.006	2	0.003
T(Bef) * Ref	d1	0.009	4	0.002
T(Aft) * Location	a5-c1	0.199	21	0.009
T(Aft) * Impact	a5-c1-b3+d1	0.048	7	0.007
T(Aft) * Ref	b3-d1	0.151	14	0.011
Residual	a6	0.464	74	0.006
Total	a7	0.763	123	

	df	**F**	**Fcrit**	**p**
1. Do reference sites have variable short-term trends after impact?				
T(Aft) * Ref / Residual	14,74	1.720	1.93	0.07
NO…				
2. Does impact affect short-term temporal trend?				
T(Aft) * Imp / Residual	7,74	1.094	2.35	0.38
NO… no short-term impact detected				
3. Do reference sites vary in difference from before to after impact?				
B * Ref / Residual	2,74	0.399	3.76	0.67
NO…				
4. Does impact affect differences from before to after?				
B * Imp / Residual	1,74	0.000	5.1	1.00
NO…no impact detected				