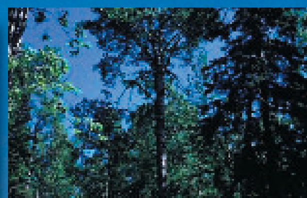
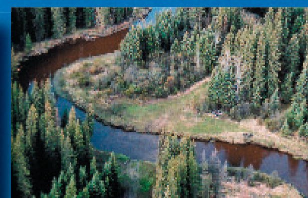
NORTHERN WATERSHED PROJECT

Project Report #3


CUMULATIVE EFFECTS:

Cumulative Effects of Watershed Disturbances on Stream Fish Communities in the Kakwa and Simonette River Basins, Alberta

The Northern Watershed Project is a collaborative research venture between the Alberta Conservation Association and the Alberta Research Council.

STREAM FISH MANAGEMENT:

CUMULATIVE EFFECTS OF WATERSHED DISTURBANCES ON STREAM FISH COMMUNITIES IN THE KAKWA AND SIMONETTE RIVER BASINS, ALBERTA

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EXECUTIVE SUMMARY

Study Focus

Developing management strategies that minimize the cumulative effects of human-induced disturbances on ecological systems is arguably the single largest challenge to sustainable resource management. In Alberta and elsewhere, rapid expansion of the forestry and oil and gas sectors, combined with conversion of forested lands to agriculture has raised concerns about the ecological sustainability of the boreal forest. The current lack of understanding of the cumulative effects of watershed disturbances on stream fish within Alberta's boreal forest is a central challenge to the management of stream fish assemblages.

We quantified the cumulative effects of watershed disturbances arising from industrial activities on stream fish communities in the Kakwa and Simonette River basins, Alberta, Canada using data collected between 1994 and 2001. Data on fish abundance, community composition and watershed attributes, including descriptors of disturbance, to address the following three focal questions:

1) Is the presence of fish, game fish and individual species affected by watershed disturbances?
2) Are species density and biomass affected by watershed disturbances?
3) Is fish community structure affected by watershed disturbances?

Summary of Study Findings

Overall patterns in fish communities

Analyses of fish communities from a total of 528 stream reaches revealed marked differences in fish communities between the Kakwa and Simonette River basins. The Kakwa watershed supports 9 species of fish comprising representatives from five familial groups whereas the Simonette River Basin supports 20 species of fish from nine Families.

In the Kakwa River Basin, bull trout and sculpin were the most frequently occurring species followed by Arctic grayling, mountain whitefish and rainbow trout. Longnose sucker, longnose dace, burbot and white sucker occurred at relatively few sites.

In the Simonette River Basin, sculpin, lake chub, bull trout, and white sucker were the most frequently occurring species followed by Arctic grayling, mountain whitefish, longnose sucker, northern redbelly dace, longnose dace, pearl dace, brook stickleback and redside shiner. Trout-perch, burbot, emerald shiner, finescale dace, flathead chub, northern pike and largescale sucker occurred at relatively few sites.

Based on density estimates, fish communities in the Kakwa were numerically dominated by sculpin, rainbow trout, bull trout and Arctic grayling. When combined, these four species accounted for the vast majority of all fish encountered. In contrast, northern redbelly dace, sculpin, lake chub, white sucker, brook stickleback and pearl dace were the predominant species in the Simonette River Basin. Overall density of fish in Kakwa watershed was about four-fold lower than that in the Simonette watershed.

Focal question 1: Is the presence of fish, game fish and individual species affected by watershed disturbances?

Logistic regression indicated that the presence of fish, game fish and individual species were moderately to highly predictable based on watershed area, stream width, elevation and to a lesser extent reach slope, size composition of the substratum and stream bank width.

At the stream reach scale, the occurrence of fish in both the Kakwa and Simonette River basins was strongly affected by stream size (i.e., order) and to a lesser extent watershed type.

With only two exceptions, the presence of fish, game fish and individual species were unrelated to watershed disturbances arising from the cumulative effects of industrial activities.

The two exceptions reflect the negative relations between bull trout presence and the: i) cumulative percent watershed disturbance in the Kakwa River Basin and ii) cumulative density of stream crossings in the Simonette River Basin.

Focal question 2: Is fish density and biomass affected by watershed disturbances?

Regression analyses generally showed that total fish density and density of the predominant species in the Kakwa watershed were primarily related to stream wetted width (i.e., the width of the water surface), elevation and reach slope. Fish density was generally highest in small streams or those at high elevation and decreased with increasing width or lower elevation. In general regression models explained relatively little of the overall variance in total density and density of the most abundant species and non-linear models did not typically explain appreciably more variance than linear models.

With some exceptions, our analyses generally showed that total fish density and biomass and density and biomass of the numerically dominant species and species groups were unrelated or poorly related to watershed disturbances attributes including harvesting, stream crossing attributes and their underlying attributes (e.g., percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines).

In the Kakwa River Basin, the notable exceptions to these findings were the positive relationships between: i) total fish density and percent watershed disturbance and, ii) density of sculpin and percent watershed disturbance.

In the Simonette watershed, the notable exceptions were a): the positive relationship between: i) total fish density and stream crossing density, ii) density of dace and stream crossing density, iii) total biomass and percent watershed disturbance, iv) biomass of sculpin and seismic line density, and iv) biomass of shiner and crossing density.

While we report some statistically significant relations between fish density and biomass and watershed disturbance attributes, the majority of these relations did not explain substantial amounts of variance in fish density or biomass.

Focal question 3: Is fish community structure affected by watershed disturbances?

We quantified the cumulative effects of watershed disturbances on fish community structure in the Kakwa and Simonette River basins using a reference condition approach. This approach evaluates the extent to which potentially impacted sites contain fish assemblages predicted by relationships between fish community structure and habitat variables derived from reference (i.e., least-impacted) sites. If the empirical model (discriminant function model) derived from the least-impacted sites also explains similar amounts of variance in fish assemblage membership in the potentially impacted sites then it is assumed that impacts are not detectable.

Kakwa River Basin

In the Kakwa River Basin, cluster analyses of 62 reference sites (i.e., least disturbed sites) using percent composition data identified three discrete fish assemblages. Assemblage 1 consisted primarily of bull trout, assemblage 2 was dominated by sculpin whereas mountain whitefish, and to a lesser extent rainbow trout, bull trout, Arctic grayling, and sculpin dominated assemblage 3.

Discriminant function analyses were used to determine linkages between the three fish assemblage types and habitat variables. Results of these analyses showed that stream wetted width, stream reach slope, site elevation and percent small gravel were moderately powerful discriminators among the three assemblage types and had an overall classification success of 71.0%.

Stream reaches supporting fish assemblage 1 were typically located at high elevations, were relatively narrow, with high reach slope and stream beds with low amounts of small gravel. Reaches supporting fish assemblage 2 were typically located at lower elevations, with broader stream channels, and higher amounts of small gravels. Stream reaches supporting fish assemblage 3 were typically located at low elevations, were relatively broad, with moderate reach slope and stream beds that contained low amounts of small gravel.

We evaluated larger patterns in fish communities initially by completing cluster analyses of least-impacted sites and potentially impacted sites. In addition to identifying the three fish assemblages from the least-impacted sites, clustering also identified two more assemblage types. The first new assemblage consisted primarily of sites dominated by Arctic grayling, and to lesser extent bull trout mountain whitefish and rainbow trout. The second new assemblage type was numerically dominated by primarily by rainbow trout which likely originated from fish stocked into the adjacent Musreau Lake.

We used discriminant function analyses to quantify how well fish communities at the potentially impacted sites (i.e., test sites where watershed disturbance ranged from 10 to 43%) were predictable based on the four watershed variables of stream wetted width, reach slope, elevation and percent small gravel within the substratum.

Our results showed that the four habitat variables of stream wetted width, reach slope, elevation and percent small gravel were poor predictors of fish assemblage type of the potentially impacted test sites and overall had a classification success of only 50%.

These results indicate that the discriminant function model developed from the least-impacted reference sites was a poor predictor of fish assemblage structure in the potentially impacted sites and that fish communities in the Kakwa River Basin are affected by the cumulative impacts of current levels of industrial activities.

Simonette River Basin

In the Simonette River Basin, cluster analyses of 106 reference (i.e., least impacted) sites identified five relatively discrete fish assemblages. Assemblage 1 consisted primarily of white sucker and to a lesser extent finescale dace, lake chub, trout-perch and redside shiner whereas assemblage 2 was dominated by lake chub and to a lesser extent white sucker, longnose sucker, sculpin and finescale dace. In contrast, northern redbelly dace, lake chub, pearl dace dominated assemblage 3. Fish communities comprising assemblages 4 and 5 were numerically dominated by mountain whitefish, pearl dace and Arctic grayling (assemblage 4) and sculpin (assemblage 5).

As for the Kakwa River Basin, discriminant function analyses were used to determine linkages between the five fish assemblages and habitat variables. Results of these analyses showed that stream wetted width, stream reach slope, site elevation and percent of fine materials in the substratum were moderately powerful discriminators among the five fish assemblages and had an overall classification success of 71.1%.

Stream reaches supporting fish assemblage 1 were located within moderately large, low elevation reaches within moderately large watersheds and characterized by low reach slope and with substrata dominated by fine sediments. Reaches supporting assemblage 2 were moderately small, low gradient systems located at intermediate elevations whereas assemblage 3, were typically located in small, low elevation reaches. Fish communities comprising assemblages 4 and 5 were located at high elevations with moderate reach slope but differed in stream size and the percent of fine materials within the stream bottom. Streams comprising assemblage 4 were typically large with a predominance of fine materials in the river bed compared with the small streams comprising assemblage 5 where fine sediments were relatively rare.

We used discriminant function analyses to quantify how well fish communities at the potentially impacted sites (i.e., test sites where watershed disturbance ranged from 21 to 61.8%) were predictable based on the four watershed variables of stream wetted width, reach slope, elevation and percent fine materials within the substratum.

Our results showed that the four habitat variables of wetted width, reach slope, elevation and percent fine materials within the substratum were moderately poor predictors of fish assemblage type and overall had a classification success of only 57.1%.

These results indicate that the discriminant function model developed from the least-impacted reference sites was a poor predictor of fish assemblage structure in the potentially impacted sites and that fish communities in the Simonette River Basin are affected by the cumulative impacts of current levels of industrial activities.

Management Implications and Recommendations

Towards an improved understanding of the cumulative effects of human-induced activities of stream fish communities

<u>Rationale</u>

The expansion of Alberta's forest industry since the mid-1980's combined with conversion of forest lands to agriculture and increased oil and gas activities has raised concerns about the ability of ecological sustainability of stream fish communities in northern Alberta. These industrial activities have the potential to affect stream fish communities by influencing the quantity and quality of habitat for stream fishes. Our results showed that current levels of industrial activity have detectable cumulative effects on stream fish communities in the Kakwa and Simonette River basins. Such effects were linked with forest harvesting and linear disturbances that intercept streams.

Recommendation

If management of the boreal forest is to be based on ecological considerations in addition to economic, social, political and cultural factors values, a more detailed understanding of the cumulative effects of multiple industrial activities on fish communities is required. Our pursuit of a knowledge base sufficient to understand the cumulative impacts of industrial activities on stream fish communities is in its infancy and this lack of knowledge challenges our ability to manage resources in a sustainable fashion. As a result we recommend an increased commitment to understand the cumulative effects of industrial activities on stream fish and other biotic communities in forested regions of Alberta.

Development and implementation of stream fish monitoring program

Rationale

In Alberta, fish communities and aquatic environments are protected under both provincial and federal legislation. For instance, consequences of watershed activities on aquatic environments and the biological diversity that they support are considered within Provincial Acts and Regulations (e.g., The *Water Act*, Timber harvest and planning operational Ground Rules [Anonymous 1994]). These provisions include operating ground rules for forest harvesting practices, and provincial codes of practice for: 1) watercourse crossings, 2) pipeline and telecommunication line crossings and 3) temporary diversion of water for hydrostatic testing. Operating ground rules define a set of forest harvesting practices including the protection of 30 and 60 m buffers adjacent to small and large permanent streams and practices that reduce inputs of organic matter into stream and avoidance of stream channels. Fish communities and aquatic habitats are also protected under habitat protection provisions (Section 35 [1]) of the *Federal Fisheries Act* which prohibits works or undertakings that result in the harmful alteration, disruption or destruction of fish habitat, while Section 35 [2], allows for authorization by the Minister, or under regulation, of harmful alteration, disruption or destruction of fish habitat. While these provisions are intended to protect stream fish populations, a scientifically rigorous program to monitor the consequences of industrial activities on stream fish communities does not exist.

Recommendation

The absence of effective monitoring programs compromises our ability to manage stream fish communities. Rigorous monitoring programs are required to: i) understand current trends in fish populations, ii) evaluate the ecological effects of anthropogenic and natural disturbances on stream fish communities and iii) evaluate the effectiveness of restoration measures, iv) to critically assess the effectiveness of current watershed management practices. As such, we recommend the allocation of resources to develop and implement a scientifically rigorous stream monitoring program in northern Alberta. This monitoring program should also include evaluations of focal watersheds, such as the Prairie Creek sub-basin, where ecological impacts have been detected and where information on recovery of fish communities is required.

The role of empirical modeling in stream fish management

Rationale

The use of empirical models to quantify the impacts of human activities on aquatic ecosystems has expanded rapidly over the last 15 years and biological assessments using multi-metric or multivariate approaches are now common place. These statistical methods represent important tools that can assist with the management of stream fish communities.

Recommendation

We suggest that the increased use of empirical modeling would assist with the management of stream fish communities by: i) providing techniques to understand large-scale patterns in fish communities, ii) gaining insights to potential cause-effect relationships, iii) evaluating environmental impacts, iv) quantifying temporal variance in fish communities, v) identifying fish community types and the environmental gradients that may drive them, and vi) monitoring the effectiveness of current management actions. As a result, we recommend the increased use of empirical and other modeling tools to assist with the management of stream fish communities.

1.0 GENERAL INTRODUCTION

1.1 Evaluating Cumulative Effects

Developing and implementing management strategies that minimize the cumulative effects of anthropogenic disturbances on plants and animals is the single largest obstacle to sustainable resource management.

Concerns related to the cumulative effects of anthropogenic disturbances on terrestrial and aquatic ecosystems has been a pervasive theme in landscape and conservation ecology within the last decade (Ziemer *et al.* 1991, Schindler 1998a, b, Schnackenberg and MacDonald 1998, MacDonald 2000, Pringle 2000). Understanding and developing management strategies that minimize cumulative effects of anthropogenic disturbances on ecological communities is arguably the single largest challenge to sustainable resource management. Cumulative effects can be broadly defined as those which result from the combined effects of multiple activities over time and space (MacDonald 2000). Within a spatial context, these activities can represent multiple events of the same type, such as multiple cut blocks within the same watershed, multiple events of different disturbance types such as forest harvest blocks and associated road networks combined with oil and gas exploration and distribution activities within the same watershed (i.e., deforestation to establish seismic lines and oil and gas pipelines). Irrespective of the event type, each activity is associated with an effect signature that changes through time (i.e., the temporal component of the cumulative effect). While the results of some activities may be detectable shortly after the activity, these relationships may take considerable time to become evident and time lags of several years between the event and the full biological response have been reported (Hartman *et al.* 1996, Fitzgerald *et al*. 1998). To date, the majority of cumulative effect studies focus on cumulative spatial effects in part because, cumulative effect studies are relatively new and quantifying cumulative effects through time is inherently difficult especially when: i) time sequences of disturbances are poorly quantified and ii) initial disturbances are followed by subsequent activities that preclude partitioning the impact of the first activity from subsequent activities.

Cumulative effects can be broadly defined as those which result from the combined effects of multiple activities over time and space.

1.2 Study Rationale: Industrial Expansion in Northern Alberta

The expansion of Alberta's forest industry since the mid-1980's combined with conversion of forest lands to agriculture and increased oil and gas activities has raised concerns about the ecological sustainability of northern ecosystems.

The expansion of Alberta's forest industry since the mid-1980's combined with conversion of forested lands to agriculture and increased oil and gas activities has raised concerns about the viability of stream fish communities in northern Alberta. These industrial activities potentially alter a myriad of interrelated stream and watershed attributes including, water quality (e.g., sediment and nutrient loadings), forest cover, and forest species composition that cumulatively can change the quantity and quality of habitat for stream fishes (e.g., Barton *et al.* 1985, Platts *et al.* 1989, Binns and Remmick 1994, Kreutzweiser and Capell 2001).

Empirical models can be used to quantify the cumulative effects of industrial activities on ecological communities.

The lack of empirical models describing the cumulative effects of watershed disturbances on stream fish within Alberta's boreal forest is a central challenge to the management of stream fish assemblages.

If management of the boreal forest is to be based on ecological considerations in addition to economic, social, political and cultural considerations, then an understanding of the cumulative effects of industrial activities on a multitude of biotic communities is required.

The lack of empirical models describing the cumulative effects of watershed disturbances on stream fish is a central challenge to the management of Alberta's boreal forest. In the absence of a process to evaluate both single impacts and cumulative impacts, watershed management planning is often a piece-meal exercise without a strong scientific basis. This paucity of information results in apparently conservative land use management prescriptions. However, the effectiveness of these apparently conservative actions is unknown and thus there is no evidence that these practices are protecting biological resources. Further, the lack of information also precludes forecasting the effects of changes in disturbance levels on ecosystems.

If management of the boreal forest is to be based on ecological, in addition to economic, social, political and cultural considerations, then an understanding of the cumulative effects of all industrial activities on a multitude of biotic communities is required. An understanding of the cumulative effects of watershed disturbance on fish communities would provide resource managers with an improved understanding of ecological risks by: 1) determining the extent to which current disturbance levels affect fish presence, abundance, and fish community types, 2) identifying the relative impact of different watershed disturbance types, and 3) provide environmental dose-response relationships that could be used in landscape simulation models.

1.3 Geological History of the Boreal Region and Fish Assemblages

In geological terms, boreal rivers and their fish assemblages are very young and probably developed within the last 10,000 years, when the Laurentide and Cordilleran ice sheets receded at the end of the Wisconsonian glacial period.

In geological terms, boreal rivers and their fish assemblages are very young and probably developed within the last 10,000 years, when the Laurentide and Cordilleran ice sheets receded at the end of the Wisconsonian glaciation (Steedman *et al.* 2003). This post-glacial waterscape was thought to have been strongly influenced by climate, regional topography, bedrock, and surficial deposits of tills and sediments. Approximately 5000-9000 ago, following glacial retreat, fish colonized the Boreal Plains and Cordillera as the continental glaciers melted to create a complex and dynamic drainage system. Fishes most likely recolonized the area from unglaciated refugia, primarily the Mississippi and Missouri, Columbia and the Yukon drainages (Crossman and McAllister 1986; Lindsey and McPhail 1986, Nelson and Paetz 1992). As a result, this young landscape supports a relatively simple fish community, the majority of which are considered to be ecological generalists, flexible in their trophic needs and abilities (e.g., Beaudoin *et al.* 2001).

In relative terms, fish communities in Alberta are depauperate consisting of 59 species including 51 native and eight introduced species.

In relative terms, fish communities in Alberta are depauperate, consisting of 59 species including 51 native and eight introduced species (Nelson and Paetz 1992). Bio-geographic patterns in stream fish assemblages and the extent to which they are affected by the cumulative effects of watershed disturbances within the provinces nine major watersheds, including that within the Kakwa and Simonette watersheds are poorly understood.

Stream fish communities are known to vary as a function of geological history, climate, landform and other watershed characteristics that affect stream habitats and define the matrix within which biological interactions occur.

Stream fish communities are known to vary as a function of geological history, climate, landform, watershed and riparian vegetation. These affect the physical and chemical characteristics of stream habitats and define the matrix within which biological interactions occur (Byorth and Magee 1998, Fausch *et al.* 1988, Rahel and Nibblelink 1999, Kilgour and Barton 1999, Paul and Post 2001). Fish assemblages are affected by a suite of local and regional processes and events that operate at several spatial and temporal scales to structure and organize fish assemblages (Tonn 1990). At the regional level, a body of evidence has shown that stream fishes can be distributed along environmental gradients often resulting in predictable distinct or semi-distinct fish assemblages (Hughes and Gammon 1987, Hughes *et al.* 1987, Jackson and Harvey 1989, Frenzel and Swanson 1996, Maret *et al.* 1997, Rahel and Nibblelink 1999). Spatial patterns in fish communities can be altered by single or interactive effects of watershed disturbance arising from activities within the forestry, oil and gas, and agricultural sectors.

Relatively few studies have evaluated the cumulative effects of multiple industrial sectors (e.g., forestry, oil and gas, agriculture) on aquatic ecosystems.

In an attempt to understand the effects of watershed disturbances on aquatic ecosystems, ecologists have developed empirical models to determine how industrial activities alter natural patterns in the occurrence and abundance of fish and other biota (e.g., St. Onge and Magnan 2000, Tonn *et al.* In Press). In the majority of cases, effects of watershed disturbances on aquatic ecosystems typically focus on: i) quantifying the effects of a focal stressor, or stressors, related to a specific industrial activity (e.g., forest harvesting) or ii) comparing the effects from a low number of disturbance types (e.g., forest harvesting versus natural wildfire) (Carignan *et al.* 2000, Garcia and Carignan 2000, McEachern *et al.* 2000, Scrimgeour *et al.* 2000, St-Onge and Magnan 2000). Relatively few studies attempt to quantify the cumulative effects of multiple industrial sectors (e.g., forestry, municipal, oil and gas, agriculture) or stressors on aquatic ecosystems (Chambers *et al.* 1997, Scrimgeour and Chambers 2000).

1.4 Study Objectives

Study objectives:

To quantify the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette watersheds.

The objectives of the present study were to describe the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette watersheds. Our general approach was based on: 1) defining the areal extent to which the study watersheds were altered by patch (e.g., forest harvest blocks, well sites) and linear disturbances (e.g., roads, railway lines, power lines, pipelines, seismic lines), and 2) quantifying the type (e.g., crossings by roads, pipelines, powerlines, seismic), number and adjacency of stream crossings. Using these data we developed empirical models that quantified the amount of variance in fish population and community characteristics that could be explained by disturbance attributes.

The cumulative effects of watershed disturbances on fish communities were evaluated by addressing the three focal questions:

1) Is the presence of fish, game fish and individual species affected by watershed disturbances?

2) Are species density and biomass affected by watershed disturbances?

3) Is fish community structure affected by watershed disturbances?

This approach allowed us to address the following three focal questions within a cumulative effects framework:

1) Is the presence of fish, game fish and individual species affected by watershed disturbances?

2) Are species density and biomass affected by watershed disturbances?

3) Is fish community structure affected by watershed disturbances?

We predicted that: 1) fish presence would be moderately to strongly predictable based on watershed attributes, 2) fish presence would not be strongly affected by watershed disturbances either based on single or cumulative disturbance attributes, 3) fish density and biomass would be affected by the cumulative watershed disturbance attributes, 4) cumulative effects of watershed disturbances would be more severe in the Simonette rather than in the Kakwa watershed, commensurate with differences in levels of industrial activity, 5) cumulative effects of watershed disturbance on fish would be species-specific.

We discuss the management implications of the results of our work in terms of completing ecological assessments and managing industrial activities.

Lastly, we discuss the management implications of our work and how our approach and results can assist with ecological assessments and planning of forest harvest blocks and road networks and their broader application to the management of stream fish communities.

2.0 STUDY AREA

2.1 Description

Fieldwork was completed in the Kakwa (drainage area = 3,475 km^2) and Simonette (drainage area = 5,222 km^2) watersheds located south east and south west of Grande Prairie, Alberta (Figure 1). The Kakwa and Simonette Rivers originate in the foothills of the Rocky Mountains of west central Alberta (Figure 2). The Kakwa River flows northeast across alpine, subalpine, and Boreal Cordillera ecoregions and converges with the Smoky River that drains into the Peace River. The Simonette River flows north across Subalpine and Upper and Lower Cordillera, and Mid Boreal Mixedwood ecoregions and converges with the Smoky River that drains into the Peace River.

Monthly flow of the Kakwa River at the Highway 40 gauging station between March and October averages 50 m^{-3} s^{-1} (1983-2000) with highest flows occurring in June after snowmelt (Mean monthly discharge between 1975-2000 = 122 m^{-3} s^{-1}) and lowest flows during November – February (Mean monthly discharge <5 m^{-3} s^{-1}) (Figure 3). Similarly, monthly flow of the Simonette River near Goodwin between March and October averages 36^{-3} s^{-1} (1970-2000) with highest flows occurring in May after snowmelt (Mean monthly discharge = 66 m^{-3} s^{-1}) and lowest flows in December - February (Mean monthly discharge <4 m^{-3} s^{-1}) (Figure 3).

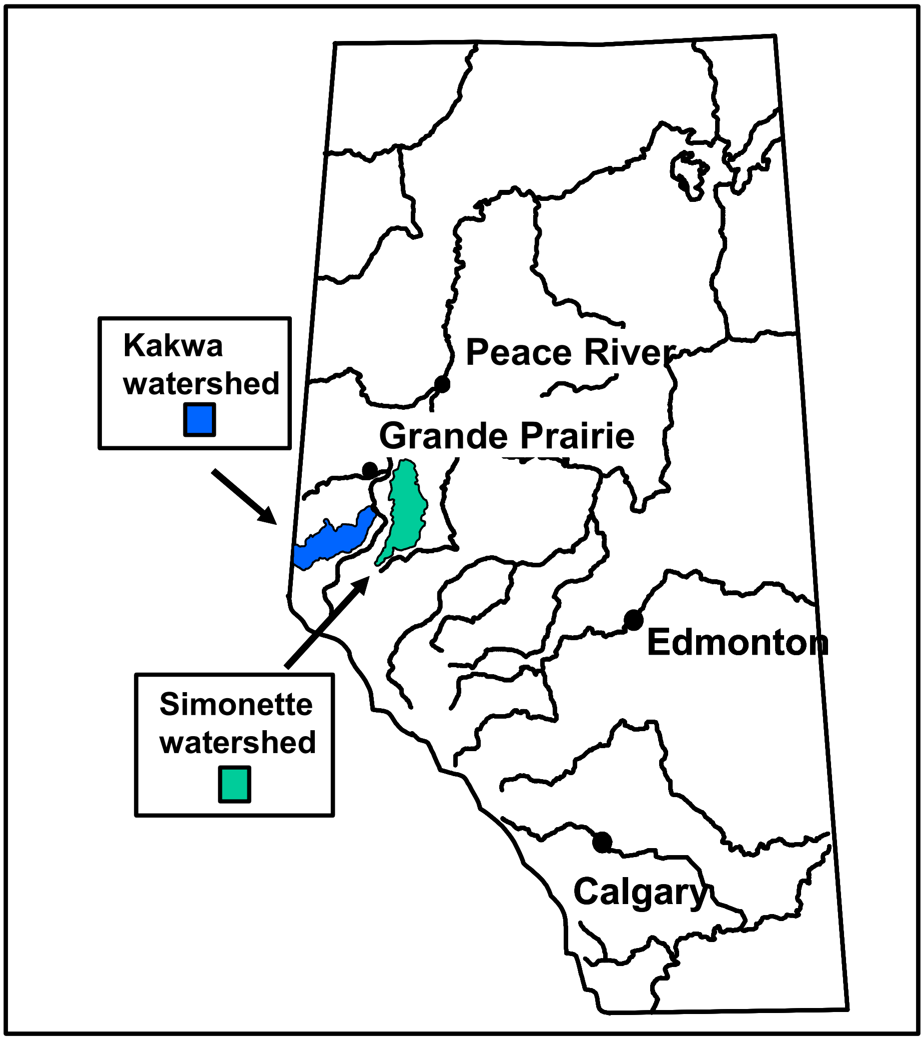

Figure 1. Locations of the Kakwa and Simonette River basins in west-central Alberta, Canada.

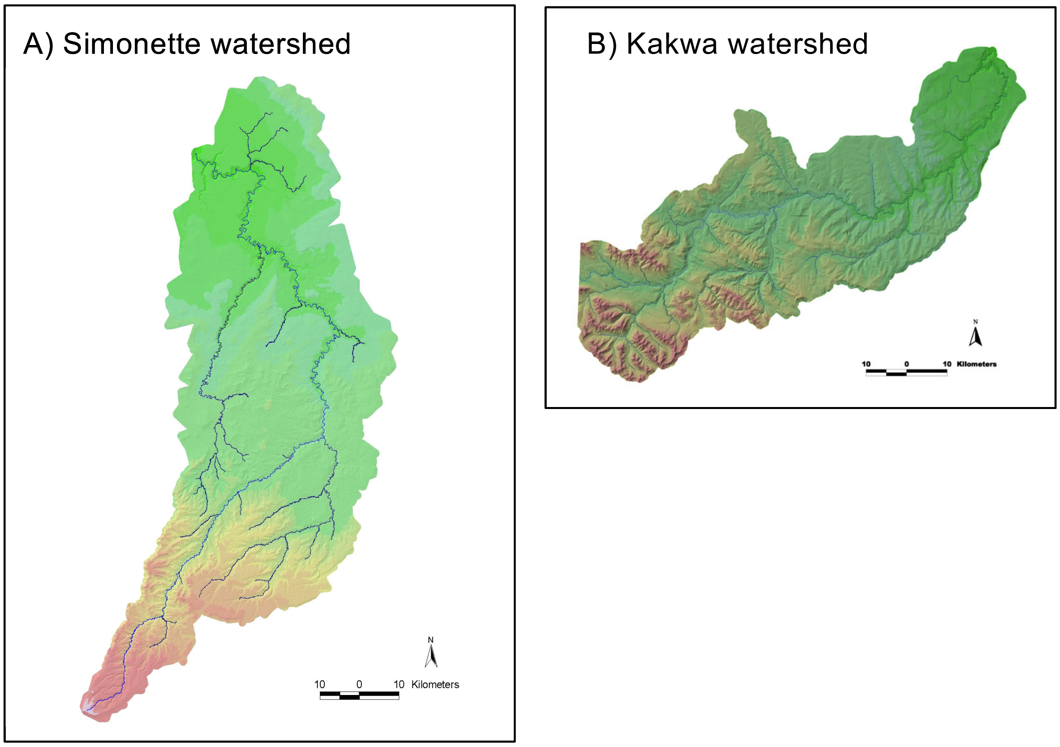

Figure 2. Digital elevation models and flow paths in the Simonette (A) and Kakwa (B) River basins, in north-central Alberta.

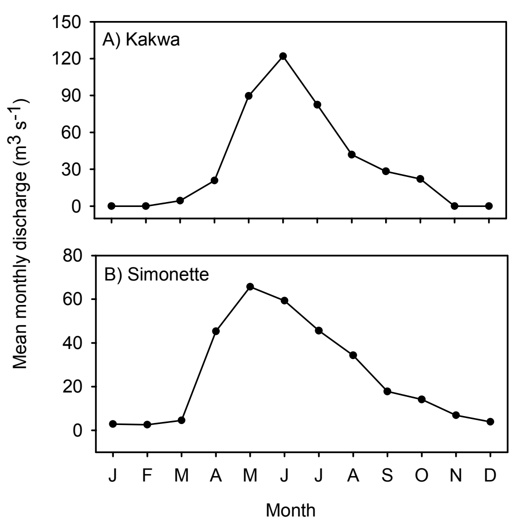

Figure 3. Mean monthly discharge in the Kakwa (A) and Simonette (B) River basins, Alberta. Data were compiled from Provincial records between 1975-2000 (Kakwa watershed) and 1970-2000 (Simonette watershed) and represent 16 to 32 monthly records during the open water period.

2.2 Ecoregions, Forest Cover and Soils

The Simonette watershed is encompassed primarily within the Lower Boreal Cordillera and to a lesser extent the Upper Boreal Cordilleran, Mid Boreal Mixedwood and Subalpine ecoregions.

The Simonette watershed is encompassed primarily within the Lower Boreal Cordillera and to a lesser extent the Upper Boreal Cordilleran, Mid Boreal Mixedwood and Subalpine ecoregions (Figure 4). In contrast, the majority of the Kakwa watershed is located within the Subalpine and Lower and Upper Cordilleran ecoregions (Figure 4).

The majority of the Kakwa watershed is located within the Subalpine and Lower and Upper Cordilleran ecoregions.

The Lower Boreal-Cordilleran is an ecotone between boreal and cordilleran climatic conditions and reflects the transition from deciduous boreal to coniferous cordilleran vegetation. Lodgepole pine (*Pinus contorta*) is the dominant species on well drained Gray luvisols while trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), paper birch (*Betula papyrifera*), lodgepole pine, white spruce (*Picea glauca*) black spruce (*Picea mariana*), and balsam fir (*Abies balsamea*) predominant on moderately well drained sites. Poorly drained sites are dominated by black spruce whereas a black spruce, white spruce and lodgepole pine complex is found on imperfectly drained sites.

Vegetation in the Upper Boreal Cordilleran ecoregion is dominated by closed canopy conifer forest.

Vegetation in the Upper Boreal Cordilleran Ecoregion is dominated by closed canopy conifer forest comprising lodgepole pine that restricts the growth of deciduous species of trembling aspen and balsam poplar located on Gray luvisols and Brunisols (Strong and Legatt 1992).

Upland sites in the Mid Boreal Mixedwood are dominated by trembling aspen and balsam poplar.

Upland sites in the Mid Boreal Mixedwood are dominated by trembling aspen and balsam poplar with black spruce, willow (*Salix* spp.) and sedge dominant on poorly drained depressions and in lowland sites. Climax species of white spruce (*Picea glauca*) and balsam fir are seldom dominant due to the frequent occurrence of fires. Soils in reference sites are dominated by gray luvisols and eutric brunisols.

The Subalpine ecoregion is dominated by coniferous forests.

The Subalpine ecoregion is dominated by coniferous forests with Engelmann spruce (*Picea engelmannii*) diagnostic of the region. Englemann spruce and subalpine fir (*Abies lasiocarpa*) are often codominant in the lower zone of the Subalpine ecoregion with open canopied dwarfed Engelmann spruce, subalpine fir, whitebark pine (*Pinus albicaulis*) and subalpine larch (*Larix lyallii)* dominant in the uppermost portion of this zone. Soils within the subalpine ecoregion are characteristically shallow, poorly developed and commonly consist of brunisols (Strong and Legatt 1992).

Within these ecoregions, mean summer temperatures are lowest in the Subalpine and highest in the Mid boreal mixedwood. In winter, mean temperatures are lowest in the Mid Boreal Mixedwood and highest in the Upper Boreal Cordilleran.

Within these ecoregions, mean summer temperatures are lowest in the Subalpine (9.4^{o}C) and highest in the Mid Boreal Mixedwood (13.5^{o}C). During the winter months, mean winter temperatures are lowest in the Mid Boreal Mixedwood (-13.2^{o}C) and highest in the Upper Boreal Cordilleran ($^{-}6^{o}$C) (Strong and Legatt 1992; Table 1). However, extreme winter temperatures can fall to -30^{o}C in all ecoregions. With the exception of the Subalpine ecoregion where 46% of all precipitation falls during May-August, the majority (60.5% to 64%) of annual precipitation in the Mid Boreal Mixedwood, Lower Boreal Cordilleran, Upper Boreal

Cordilleran falls during the summer period. Annual precipitation in the Subalpine and Upper cordilleran is about 25% higher than that in the Lower Boreal Cordillera and Mid Boreal Mixedwood ecoregions (Strong and Legatt 1992, Table 1).

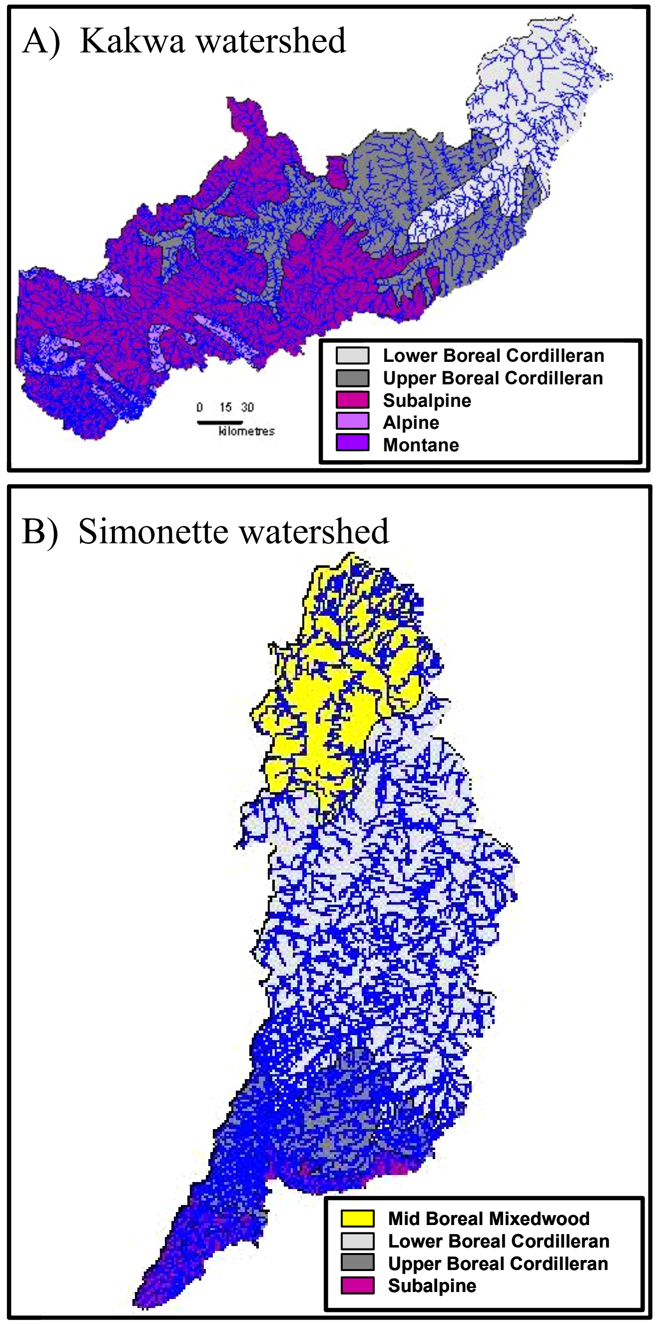

Figure 4.   Location of the ecoregions in the Kakwa (A) and Simonette (B) River basins, Alberta.

Table 1. Summary of summer and winter air temperatures and annual precipitation in the dominant ecoregions in the Simonette and Kakwa River basins, Alberta. Summer = May-August, Winter = November-February.

River basin	**Simonette**	**Kakwa**				
Dominant ecoregion within river basin	**Mid Boreal Mixedwood**	**Lower Boreal Cordilleran**	**Upper Boreal Cordilleran**	**Lower Boreal Cordilleran**	**Upper Boreal Cordilleran**	**Subalpine**
Air Temperatures						
Summer						
Mean	13.5	12.8	11.5	12.8	11.5	9.4
Mean minimum	7.3	6.9	5.9	6.9	5.9	3.8
Mean maximum	19.6	18.3	16.7	18.3	16.7	14.7
Winter						
Mean winter	-13.2	-7.8	-6.0	-7.8	-6.0	-8.9
Mean minimum	-18.6	-14.3	-12.5	-14.3	-12.5	-14.5
Mean maximum	-7.7	-2.1	0.5	-2.1	0.5	-2.8
Precipitation						
Summer	240	295	340	295	340	263
Winter	64	60	60	60	60	146
Annual	397	464	538	464	538	568

2.3 Fish Communities

Fish communities in the Simonette River Basin (twenty two species) are substantially more diverse than that in the Kakwa River Basin (10 species).

Analyses of historical databases indicate that twenty two species of fish from nine families have been recorded in the Simonette River Basin and ten species from five families in the Kakwa River Basin (Table 2). Cyprinid minnows are the dominant species group (eight species) in the Simonette watershed, followed by salmonids (four species) and suckers (three species). In contrast, salmonids (four species), suckers (two species) and cyprinids (two species) were the dominant groups in the Kakwa watershed. These watersheds support five (Kakwa) or six (Simonette) game species of Arctic grayling, bull trout, rainbow trout, mountain whitefish, northern pike and walleye.

Table 2. Common and scientific names of fish recorded in the Simonette and Kakwa River basins. Species presence: + = species present, - = species absent. * = game species.

Family	Common name	Occurrence Simonette	Kakwa
Cyprinidae			
Lake chub	*Couesius plumbeus* (Agassiz)	+	+
Flathead chub	*Platygobio gracilis* (Richardson)	+	-
Finescale dace	*Phoxinus neogaeus* Cope	+	-
Pearl dace	*Margariscus margarita* (Cope)	+	-
Longnose dace	*Rhinichthys cataractae* (Valienciennes)	+	+
Northern redbelly dace	*Phoxinus eos* (Cope)	+	-
Emerald shiner	*Notropis atherinoides* Rafinesque	+	-
Redside shiner	*Richardsonius balteatus* (Richardson)	+	-
Percopsidae			
Trout-perch	*Percopsis omiscomaycus* (Walbaum)	+	-
Gasterosteidae			
Brook stickleback	*Culaea inconstans* (Kirtland)	+	-
Percidae			
Walleye*	*Stizostedion vitreum vitreum* (Mitchill)	+	+
Salmonidae			
Arctic grayling*	*Thymallus arcticus* (Pallas)	+	+
Bull trout*	*Salvelinus confluentus* (Suckley)	+	+
Mountain whitefish*	*Prosopium williamsoni* (Girard)	+	+
Rainbow trout*	*Oncorhynchus mykiss* (Walbaum)	+	+
Esocidae			
Northern pike*	*Esox lucius* Linnaeus	+	-
Catostomidae			
Longnose sucker	*Catostomus catostomus* (Lacepède)	+	+
White sucker	*Catostomus commersoni* (Forster)	+	+
Largescale sucker	*Catostomus macrocheilus* Girard	+	-
Cottidae			
Slimy sculpin	*Cottus cognatus* Richardson	+	+
Spoonhead sculpin	*Cottus ricei* (Nelson)	+	-
Gadidae			
Burbot*	*Lota lota* (Linnaeus)	+	-

3.0 MATERIALS AND METHODS

Cumulative effects of human-induced watershed disturbances on fish communities were evaluated from a total of 413 sites in the Kakwa and 428 sites in the Simonette watersheds.

Cumulative effects of watershed disturbances on fish communities were evaluated using data on fish communities and watershed attributes from a total of 413 and 428 sites in the Kakwa and Simonette River basins, respectively. In both watersheds, information on fish communities and instream habitats were derived from historical databases held by the Government of Alberta (i.e., the Fisheries Management Information System) and that collected as part of the Cooperative Fisheries Inventory Program (Hvenegaard 1998) and the Northern Watershed Project.

For each river basin, fish and instream habitat data were combined with landscape and watershed-scale characteristics derived primarily from queries of Geographic Information Systems (GIS) databases that described watershed morphometry and forest cover.

3.1 Data Screening

Our primary interest was to quantify the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette River Basins.

A data screening process was completed to exclude stream reaches that were highly adjacent to other sites.

Our primary interest was to quantify the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette River basins. For both watersheds, we completed a data screening process to remove stream reaches that were highly adjacent to other sites. Stream sites were excluded from analyses when they occurred within the same stream reach (i.e., have the same stream order) and were located within 3 km of an adjacent site. When clusters of two or more sites were identified, sites were either preferentially removed based on the completeness of habitat variables or randomly selected and removed when adjacent sites contained similar amounts of information on habitat conditions.

3.2 Watershed Characteristics

Predictors of fish communities were divided into the three types of: i) watershed characteristics, ii) stream reach and riparian attributes and iii) disturbance attributes.

Predictors of fish communities were divided into the three types of: i) watershed characteristics, ii) stream reach and riparian attributes and iii) potential disturbance attributes (Table 3). The majority of watershed–scale variables were quantified using databases created within the Provincial Government's Geographic Information System (GIS) (i.e., Base Features). Some of the watershed attributes were defined within the watershed contributing flow to the study site (i.e., contributing watershed) as well as that within watershed defined by the stream order (Figure 5).

Table 3. Environmental variables used to explain spatial and temporal variation in stream fish communities in the Kakwa and Simonette River basins.

Variable	Abbreviation	Units
Watershed		
Latitude [a]	LAT	Degrees
Longitude [a]	LONG	Degrees
Elevation [a]	ELEV	Meters above sea level
Ecoregion [a]	ECOR	3 categories
Watershed & stream order [b]	W-ORDER	6 categories
Contributing watershed [b]	CW-AREA	km^2
Forest	FOREST	Percent
Conifer [c]	%CONI	Percent
Deciduous [d]	%DECI	Percent
Conifer & deciduous [e]	%CON-DEC	Percent
Deciduous & conifer [f]	%DEC-CON	Percent
Productive forest [g]	PROD-F	Percent
Non-productive forest [g]	NPROD-F	Percent
Non-classified forest [g]	NONC-F	Percent
Standing water [g]	ST-WATER	Percent
Contributing watershed slope [h]	CW-SLOPE	Degrees
Stream reach		
Latitude [a]	LAT	Degrees
Longitude [a]	LONG	Degrees
Elevation [a]	ELEV	Meters above sea level
Stream reach slope [i]	RSLOPE	Degrees
Adjacency to fourth order stream [j]	AD-FOUR	Kilometers
Adjacency to fifth order stream [j]	AD-FIVE	Kilometers
Adjacency to sixth order stream [j]	AD-SIX	Kilometers
Riparian conifer – 100 [k,l]	%RIPC-100	Percent
Riparian deciduous –100 [k,l]	%RIPD-100	Percent
Reach conifer & deciduous -100 [n]	%RIPCD-100	Percent
Riparian deciduous & conifer –100 [k,l]	%RIPDC-100	Percent
Stream reach		
Riparian productive forest - 100 m [k,l]	%RIP-PR-100	Percent
Riparian non-productive forest –100 m [k,l]	%RIP-NPR-100	Percent
Riparian non-classified forest –100 m [k,l]	%RIP-NONC-100	Percent

Table 3 continued

Variable	Abbreviation	Units
Riparian slope – 100m [m]	RIP-SLP-100	Percent
Riparian conifer – 300 m [k, n]	%RIPC-200	Percent
Riparian deciduous – 300 m [k, n]	%RIPD-200	Percent
Riparian conifer & deciduous - 300 m [k, n]	%RIPCD-200	Percent
Riparian deciduous & conifer - 300 m [k, n]	%RIPDC-200	Percent
Riparian productive forest – 300 m [k, n]	%RIP-PR-200	Percent
Riparian non-productive forest - 300 m [k, n]	%RIP-NPR-200	Percent
Riparian non-classified forest - 300 m [k, n]	%RIP-NONC-200	Percent
Riparian slope – 300 m [m]	RIP-SLP-200	Percent
Riparian conifer – 500 m [k,o]	%RIPC-300	Percent
Riparian deciduous – 500 m [k,o]	%RIPD-300	Percent
Riparian conifer & deciduous - 500 m [k,o]	%RIPCD-300	Percent
Riparian deciduous & conifer - 500 m [m, k,o]	%RIPDC-300	Percent
Riparian productive forest - 500 m [k,o]	%RIP-PR-300	Percent
Riparian non-productive forest - 500 m [k,o]	%RIP-NPR-300	Percent
Riparian non-classified forest - 500 m [k,o]	%RIP-NONC-300	Percent
Riparian slope – 500 m [m]	RSLOPE-300	Percent
Discharge [p]	DISC	m^3/s
Fines [q]	%FINE	Percent
Gravel [q]	%GRVL	Percent
Cobble [q]	%COBB	Percent
Boulder [q]	%BOLD	Percent
Woody debris volumes [r]	WOOD-V	$m^3/100\ m^2$
Bankfull width [s]	BFULL	m
Temperature [t]	TEMP	Degrees
Turbidity [u]	TURB	NTU
Dissolved oxygen [v]	DO	mg/L
Water depth	DEPTH	(cm)
<u>Potential disturbance indicators</u>		
Percent harvested in contributing watershed [a b k]	% HARVEST-CA	Percent
Percent harvested in entire watershed [a b k]	% HARVEST-WA	Percent
Percent harvested in last ten years in contributing watershed [a b k]	% HARVEST-10-CA	Percent
Percent harvested in last ten years in entire watershed [a b k]	% HARVEST-10-WA	Percent
Density of well sites in contributing watershed [a b]	WELL-DEN-CA	No. km^2
Density of well sites in entire watershed [a b]	WELL-DEN-WA	No. km^2
Density of defined facilities in contributing watershed [a b]	FAC-DEN-CA	No. km^2

Table 3 continued

Variable	Abbreviation	Units
Density of defined facilities in entire watershed [a b]	FAC-DEN-WA	No. km^2
Percent of contributing watershed as defined roads [a b]	% ROADS-CA	Percent
Percent of entire watershed as defined roads in contributing watershed [a b]	% ROADS-WA	Percent
Percent of contributing watershed as-paved roads in contributing watershed [a b]	% PAVED-CA	Percent
Percent of entire watershed as paved roads in entire watershed [a b]	% PAVED-WA	Percent
Percent of contributing watershed as non-paved roads in contributing watershed [a b]	% NON-PAVED-CA	Percent
Percent of entire watershed as paved roads in entire watershed [a b]	% NON-PAVED-WA	Percent
Percent of contributing watershed as seismic lines (cut lines) in contributing watershed [a b]	%SEISMIC-CA	Percent
Percent of entire watershed as seismic lines (cut lines) in entire watershed [a b]	%SEISMIC-WA	Percent
Percent of contributing watershed as pipelines in contributing watershed [a b]	%PIPELINES-CA	Percent
Percent of entire watershed as pipelines in entire watershed [a b]	%PIPELINES-WA	Percent
Percent of contributing watershed as power lines in contributing watershed [a b]	%POWERLINES-CA	Percent
Percent of entire watershed as power lines in entire watershed [a b]	%POWERLINES-WA	Percent
Density of all stream crossings in the contributing watershed [a b]	CROSS-DEN-CA	No. km^2
Density of stream crossings in the entire watershed [a b]	CROSS-DEN-WA	No. km^2
Number of all road crossings within 10 km downstream of the study site [a b]	ROAD-10	No. km
Number of all road crossings within 20 km downstream of the study site [a b]	ROAD-20	No. km
Number of all road crossings within 40 km downstream of the study site [a b]	ROAD-40	No. km
Number of all road crossings within 80 km downstream of the study site [a b]	ROAD-80	No. km
Number of all paved road crossings within 10 km downstream of the study site [a b]	PAVED-10	No. km
Number of all paved road crossings within 20 km downstream of the study site [a b]	PAVED-20	No. km

Table 3 continued

Variable	**Abbreviation**	**Units**
Number of all paved road crossings within 40 km downstream of the study site [a b]	PAVED-40	No. km
Number of all paved road crossings within 80 km downstream of the study site [a b]	PAVED-80	No. km
Number of all non-paved road crossings within 10 km downstream of the study site [a b]	NON-PAVED-10	No. km
Number of all non-paved road crossings within 20 km downstream of the study site [a b]	NON-PAVED-20	No. km
Number of all non-paved road crossings within 40 km downstream of the study site [a b]	NON-PAVED-40	No. km
Number of all non-paved road crossings within 80 km downstream of the study site [a b]	NON-PAVED-80	No. km
Number of all seismic line crossings within 10 km downstream of the study site [a b]	SEISMIC-10	No. km
Number of all non-paved road crossings within 20 km downstream of the study site [a b]	SEISMIC-20	No. km
Number of all non-paved road crossings within 40 km downstream of the study site [a b]	SEISMIC-40	No. km
Number of all non-paved road crossings within 80 km downstream of the study site [a b]	SEISMIC-80	No. km
Number of all pipeline crossings within 10 km downstream of the study site [a b]	PIPELINES-10	No. km
Number of all pipeline crossings within 20 km downstream of the study site [a b]	PIPELINES-20	No. km
Number of all pipeline crossings within 40 km downstream of the study site [a b]	PIPELINES-40	No. km
Number of all pipeline crossings within 80 km downstream of the study site [a b]	PIPELINES-80	No. km
Number of all power line crossings within 10 km downstream of the study site [a b]	POWERLINES-10	No. km
Number of all power line crossings within 20 km downstream of the study site [a b]	POWERLINES-20	No. km

Table 3 continued

Variable	Abbreviation	Units
Number of all power line crossings within 40 km downstream of the study site [a b]	POWERLINES-40	No. km
Number of all power line crossings within 80 km downstream of the study site [a b]	POWERLINES-80	No. km
Number of all railway line crossings within 10 km downstream of the study site [a b]	RAILWAY-10	No. km
Number of all railway line crossings within 20 km downstream of the study site [a b]	RAILWAY-20	No. km
Number of all railway line crossings within 40 km downstream of the study site [a b]	RAILWAY-40	No. km
Number of all railway line crossings within 80 km downstream of the study site [a b]	RAILWAY-80	No. km
Number of all bridge crossings within 10 km downstream of the study site [a b]	BRIDGES-10	No. km
Number of all bridge crossings within 20 km downstream of the study site [a b]	BRIDGES-20	No. km
Number of all bridge crossings within 40 km downstream of the study site [a b]	BRIDGES-40	No. km
Number of all bridge crossings within 80 km downstream of the study site [a]	BRIDGES-80	No. km
Number of all culverts crossings within 10 km downstream of the study site [a b]	CULVERT-10	No. km
Number of all culverts crossings within 20 km downstream of the study site [a b w]	CULVERT-20	No. km
Number of all culverts crossings within 40 km downstream of the study site [a b w]	CULVERT-40	No. km
Number of all culverts crossings within 80 km downstream of the study site [a b w]	CULVERT-80	No. km
Number of culverts crossings with extreme potential for stream fragmentation within 10 km downstream of the study site [a b w]	CULV-EXT-10	No. km
Number of culverts crossings with extreme potential for stream fragmentation within 20 km downstream of the study site [a b w]	CULV-EXT-20	No. km
Number of culverts crossings with extreme potential for stream fragmentation within 40 km downstream of the study site [a b w]	CULV- EXT-40	No. km

Table 3 continued

Variable	Abbreviation	Units
Number of culverts crossings with extreme potential for stream fragmentation within 80 km downstream of the study site [a b w]	CULV- EXT-80	No. km
Number of culverts crossings with high potential for stream fragmentation within 10 km downstream of the study site [a b w]	CULV- HIGH-10	No. km
Number of culverts crossings with high potential for stream fragmentation within 20 km downstream of the study site [a b w]	CULV- HIGH-20	No. km
Number of culverts crossings with high potential for stream fragmentation within 40 km downstream of the study site [a b w]	CULV- HIGH-40	No. km
Number of culverts crossings with high potential for stream fragmentation within 80 km downstream of the study site [a b w]	CULV- HIGH-80	No. km
Number of culverts crossings with moderate potential for stream fragmentation within 10 km downstream of the study site [a b w]	CULV- MOD-10	No. km
Number of culverts crossings with moderate potential for stream fragmentation within 20 km downstream of the study site [a b w]	CULV- MOD-20	No. km
Number of culverts crossings with moderate potential for stream fragmentation within 40 km downstream of the study site [a b w]	CULV- MOD-40	No. km
Number of culverts crossings with moderate potential for stream fragmentation within 80 km downstream of the study site [a b w]	CULV- MOD-80	No. km

[a] Obtained from Geographic Information Systems (GIS) based on 1:50,000 topographic maps.
[b] Hydrologically corrected digital elevation model (DEM) based on 1:50,000 scale topographic maps.
[c] Conifer as the dominant overstory tree species. Data from digital Phase III and Alberta Vegetation Inventory (AVI) databases.
[d] Deciduous as the dominant overstory species within the contributing watershed. Data as described above.
[e] Conifer as the dominant overstory with deciduous understory species within the contributing watershed. Data as described above.
[f] Deciduous as dominant overstory with conifer as the dominant understory species within the contributing watershed. Data as described above.
[g] Data from digital Phase III and Alberta Vegetation Inventory (AVI) databases.
[h] Median watershed slope derived from the DEM.

Table 3 continued

Variable	Abbreviation	Units

[i] Slope (rise over run) calculated from points located 100 m upstream and 100 meters downstream of the sampling site using the 1:50,000 scale DEM.
[j] Distance from the sampling point along the stream channel to the nearest fourth, fifth and sixth order stream. Derived from single line hydrography GIS layer.
[k] Digital Phase III and AVI.
[l] Defined within a 100 m radius of the study point (i.e., Area = 3.14 ha).
[m] Median slope based on 1000 randomly selected pixels (2 x 2 m) within the 100, 300 or 500 m of the study point.
[n] Defined within 300 m of the study point (i.e., Area = 28.13 ha).
[o] Defined within 500 m of the study point (i.e., Area = 78.53 ha).
[p] Typically based on three transects across the stream channel with velocity measured at 0.4 depth.
[q] Percent silt and sand (<4.7 mm), gravel (4.8 - 76 mm), cobble (76.1 – 304.7 mm) and boulder (>304 mm) estimated visually at three, equidistant locations along each of three to five transects across the stream channel.
[r] Based on the density and volumes of 20 to 50 pieces of woody debris within the stream channel.
[s] Typically based on three estimates at the study site (± 0.1 to 1 m).
[t] Handheld mercury thermometer (± 1°C) or a Multiline P4 portable dissolved oxygen and water temperature meter fitted with a OxiCal 325 probe (+0.1 °C).
[u] Halltech Global water turbidity meter (+1 Nephlometeric Turbidity Unit).
[v] WTW Multiline P4 portable dissolved oxygen meter fitted with a OxiCal 325 probe.
[w] Calculated for the Simonette River Basin only using results from a road crossing inventory study.
[x] Culvert fragmentation potential descriptions were based on the distance from the bottom of the downstream opening of the culvert to the stream water surface, also defined as hang-height. Qualitative culvert fragmentation potential codes: extreme = > 30 cm hang height, high = 10 to 30 cm hang height, moderate = 2 to 10 cm hang height.

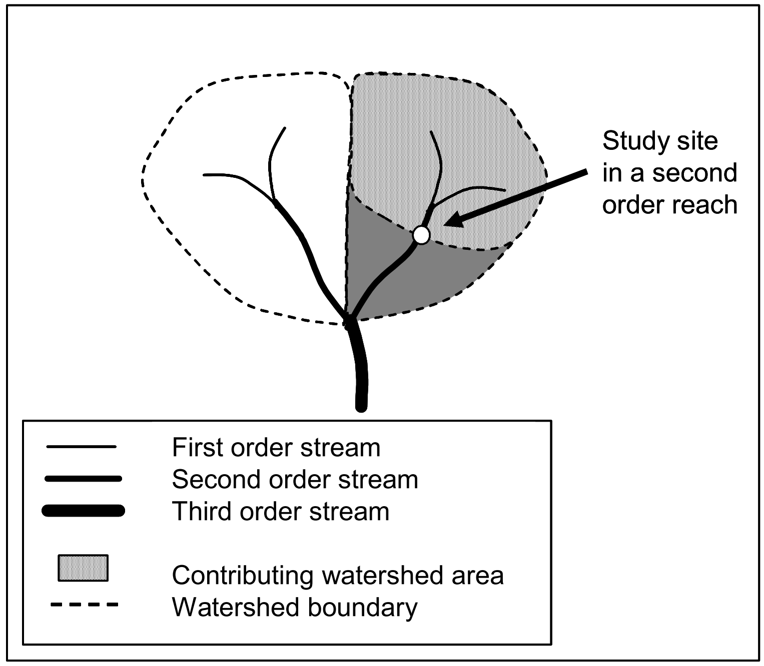

Figure 5. Conceptual model discriminating between watershed areas. Defined using the stream reach and the study site (i.e., contributing watershed) for a hypothetical second order stream. The contributing watershed (light gray) is a portion of the overall area delimited by the second order stream (dark gray).

The Base Features database was used to describe the physical location of study watersheds and to define ecoregion types.

The Base Features database was used to describe the physical location of study watersheds (i.e., longitude, latitude, elevation [meters above sea level {m.a.s.l.}]) and to define ecoregion type. Watershed areas, contributing watershed areas, and stream order were determined using a hydrologically-corrected digital elevation model (DEM) comprising 25 x 25 m cells interpreted from lower resolution imagery (i.e., 100 x 100 m cells). The elevation model has a vertical resolution of about 3 m. The DEM was also used to calculate stream order (Strahler 1957), median watershed slopes and median slopes within each of the three riparian distance zones extending from the centroid of the study reach (i.e., 100, 300, and 500 m radii from the sampling site).

Forest cover attributes were derived from digital Phase III and Alberta Vegetation Inventory databases.

Forest cover attributes were derived from digital Phase III and Alberta Vegetation Inventory databases (AVI) databases developed from 1:15,000 aerial photography combined with extensive ground-truthing (AVI) (Anonymous 1991) and 1: 20,000 scanned aerial photography (Phase III). Each identifiable stand on the forested land-base was attributed to identify dominant overstory tree species and forest species composition (% based on canopy closure). The stand resolution of AVI has been estimated at ~0.4 ha (Joy 1996).

Forest cover types, based on the dominant overstory canopy closure, were separated into the four classes of: 1) conifer, 2) deciduous, 3) conifer with a deciduous understory, 4) deciduous with a conifer understory, 5) potentially productive, 6) productive, non-forested and 7) unproductive, non-forested and 8) non-classified. The Base Features data base was also used to describe overall watershed slope (mean slope, upper 25% quartile slope, lower 75% quartile slope) and site aspect. We quantified the percentage of each watershed that was forested by combining the stands identified as conifer, deciduous, conifer with a deciduous understory and deciduous with a conifer understory.

3.3 Stream Reach Characteristics

Physical characteristics of stream reaches were determined by completing field surveys and queries of GIS databases.

Physical characteristics of stream reaches (average stream reach = 195 m [Kakwa] and 225 m [Simonette]) were determined during field surveys completed between 1995 and 2001 combined with queries of GIS databases. The location (± 5 to 25 m) of study sites (i.e., stream reaches) was determined in the field using a Garmin handheld (Model 12 XL) Global Positioning System. These coordinates were entered into a GIS data base. GIS tools and relevant spatial layers were used to estimate site elevation, reach slope, adjacency of study reaches to fourth, fifth and sixth order streams, and adjacent forest cover types. Relations between fish communities and adjacent forest cover and slope were determined by delineating three circular zones around each study location: Zone 1 = 3.14 ha (radius = 100 m), Zone 2 = 28.13 ha (radius = 300 m), Zone 3 = 78.53 ha (radius = 500 m). We quantified forest cover attributes and median slope within each of the areas as described previously for watershed-scale attributes (Table 3).

Physical characteristics of stream reaches included water depth, wetted and bankfull width, reach slope, substratum size composition, discharge and volume of woody debris.

We established 3 (1999-2001) or 5 (1995-1997) transects across the stream channel within each study reach to quantify mean depth, substratum composition, bankfull width, wetted width and in stream discharge (1999-2001). Transects were located at equal distances from the most downstream location of the study reach to the most upstream position. In 1995-1997 stream discharge was estimated using the float method (Buchanan and Somers 1969) whereas in 1999-2001 discharge was based on measurements of water depth and velocity (at 0.4 times depth) at three to five distance intervals along two or three transects across the stream channel. In 1995-1997 water depth was measured at 3 locations along each of the 5 transects within the study reach.

Size composition of the materials on the river bottom was determined by visually estimating the percent cover of silt and sand, gravel, cobble and boulder.

Size composition of the substratum was determined by visually estimating the percent cover of silt and sand (< 2 mm [fines]), small gravel (2-16 mm), large gravels (17-64 mm), cobble (65– 256 mm) and boulder (>256 mm) within three 1 m^2 areas, locations along five (1995-1997) or three transects (1999-2001) across the stream channel. Wetted and bankfull widths (m) were recorded along each transect.

In 2000 and 2001 we determined the volume of woody debris at 100 study reaches in the Simonette and 104 sites in the Kakwa River basins. At each site, we counted and measured all pieces of woody material that exceeded 20 cm. For each piece, we quantified the length and mean diameter (as the average of the diameter at each end) and converted these dimensions to volume estimates (V) using the equation:

$$V = \pi \times (d/2)^2 \times L$$

Where d is the mean diameter and L is the length of the woody debris. Volumes of woody debris in each of the four sub-reaches were converted into mean woody debris volumes ($m^3/100\ m^2$) by dividing the volume of woody debris by the area sampled.

Stream reach slope was calculated as the rise over run of points located 100 m upstream and 100 meters downstream of the sampling site using the 1:50,000 scale DEM. Size composition of the substratum, mean water depth, bankfull width and water temperature were measured at all sites whereas woody debris volumes, water temperature, water turbidity and dissolved oxygen were measured at the majority of sites (Table 3).

Streams were defined primarily using Strahler stream classes.

Forest practice codes in Alberta identify five classes of stream based largely on width, water flow period, and channel development (Table 4). For several analyses we defined relationships between fish communities and stream using Alberta watercourse classifications and Strahler stream orders as descriptors of stream size (Strahler 1957, Anonymous 1994).

3.4 Watershed Disturbance Characteristics

The cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette watersheds were evaluated using two data types:

i) Patch and linear disturbances (e.g., forest harvest blocks, seismic lines, roads, pipelines)

ii) Stream crossing disturbances (e.g., all stream crossings and those arising from roads, seismic lines, pipelines transmission lines).

The cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette watersheds were evaluated using two approaches. First, we quantified the total area of study watersheds disturbed by calculating areas set back to early successional stages or converted, either permanently or non-permanently to non-forested land cover types. Second, we quantified stream crossings by roads, seismic lines, power lines, pipelines and railway lines.

Specifically, we quantified the area of watersheds disturbed by forest harvesting and the development of oil and gas well sites, various industrial facilities, roads, seismic lines, power lines, pipelines and railway lines. Using GIS tools and relevant data bases, we quantified the total (i.e., cumulative) area of watersheds disturbed by all of these activities. To allow comparisons among watersheds, disturbance areas were converted from areas to percentages by dividing the areas disturbed by the watershed area. Thus, a hypothetical 400 ha watershed comprising a 30 ha forest harvest block combined with a road network and seismic lines occupying a total of 10 ha represents a disturbance percentage of 10%.

Table 4. Government of Alberta watercourse classification codes for flowing water bodies. Modified from Anonymous (1994). [a] Probable Strahler stream order classes derived by the authors.

Classification	Physical description	Water flow period	Channel development	Strahler order [a]
Water source areas	Saturated soils and seepages.	All year.	N/A	1st
Ephemeral	Often vegetated draws.	Only during or after rainfall.	Minimal, often vegetated or snow melt.	1st
Intermittent	Small stream channels. Springs are often source areas outside of spring runoff and heavy rainfall.	Wet seasons or storms. Dry up during drought.	Distinct channel. Usually not vegetated Width up to 0.5 m.	1st
Small permanent	Permanent streams. Often within small valleys with bench and floodplain development.	All year but may freeze in winter.	Banks and channel well defined. Channel widths of 0.5 to 5 m.	2nd to 4th
Large permanent	Major streams or rivers. Well defined floodplains. Valley width usually exceeds 400 m.	All year.	Non-vegetated channel. Channel width > 5m.	3rd to 7th

We calculated the cumulative impacts of all stream crossings by quantifying the number and, to a lesser extent, the type of crossing resulting from roads, seismic lines, power lines, pipelines and railway lines. In many cases, we expressed disturbance areas as the proportion of individual watersheds as linear (e.g., seismic lines, roads) and patch disturbances (e.g., forest harvest blocks, well sites). These disturbance measures were expressed in the two forms of: i) contributing watershed (i.e., the area of the watershed immediately upstream of the study site) and ii) stream-order watershed (i.e., the area of the watershed delineated by the entire stream order). We also defined linear disturbances like road crossings at various distances downstream of study sites.

3.5 Fish Communities

Data from electroshocking was used to describe: 1) fish presence-absence 2) total and species –specific density and biomass and 3) composition of fish communities.

Fish communities in the Kakwa and Simonette watersheds were described by electroshocking stream reaches (Kakwa: average reach = 195 m, range in reach lengths = 40 to 450 m; Simonette: average reach = 225 m, range in reach lengths = 25 to 663 m) with a backpack (i.e., first to fourth order sites) or a boat mounted Smith-Root electroshocker (i.e., several of the fourth and fifth order stream reaches).

Because our primary objective was to describe large-scale patterns in fish community structure, we used a single pass rather a multi-pass approach, to describe fish communities because it allowed us to sample greater numbers of sites.

Because our primary objective was to describe large-scale patterns in fish community structure, we used a single pass, rather than a multi-pass approach to describe fish communities, as it allowed us to sample greater numbers of sites. While the single pass approach underestimates fish density by about 30 to 40%, it provides an abundance estimate that is strongly related to overall density (e.g., Kruse *et al.* 1998, Mitro and Zale 2000) and is commonly used in community-based studies (e.g., Kilgour and Barton 1999).

Fish captured during electroshocking were identified, measured and released into the stream reach from which they were captured earlier.

Fish captured during electroshocking were identified using taxonomic keys in Nelson and Paetz (1992) or Scott and Crossman (1998) and, when possible, measured and released after the reach had been shocked. In several cases, shocked fish were not retrieved when the current swept them past capture nets or they became entangled in interstitial spaces or woody debris. Non-captured fish (< 2%) that were identified to species with confidence were included in total density and species density estimates. Data from electroshocking was used to describe: 1) fish presence and 2) total faunal density and density of individual species after calculating the total area shocked (i.e., total reach distance times mean wetted width (i.e., width of the water surface perpendicular to the direction of flow). Because fish densities in the Kakwa and Simonette basins are relatively low, we converted density and biomass measures per m^2 to per 100 m^2.

We also estimated total and species-specific fish biomass by either weighing fish (± 0.1 to 1 gm) in the field or converting fish lengths measured in the field to fish biomass after creating wet weight (g) - length (mm) regressions (Figures 6 and 7). Power functions ($y = ax^b$) where y is fish mass (gm) and x is fish length (mm) explained between 86.6% to 98.4 % of the variance in fish weight and due to the range of fish lengths used provided moderate to highly powerful predictors of fish biomass (Table 5).

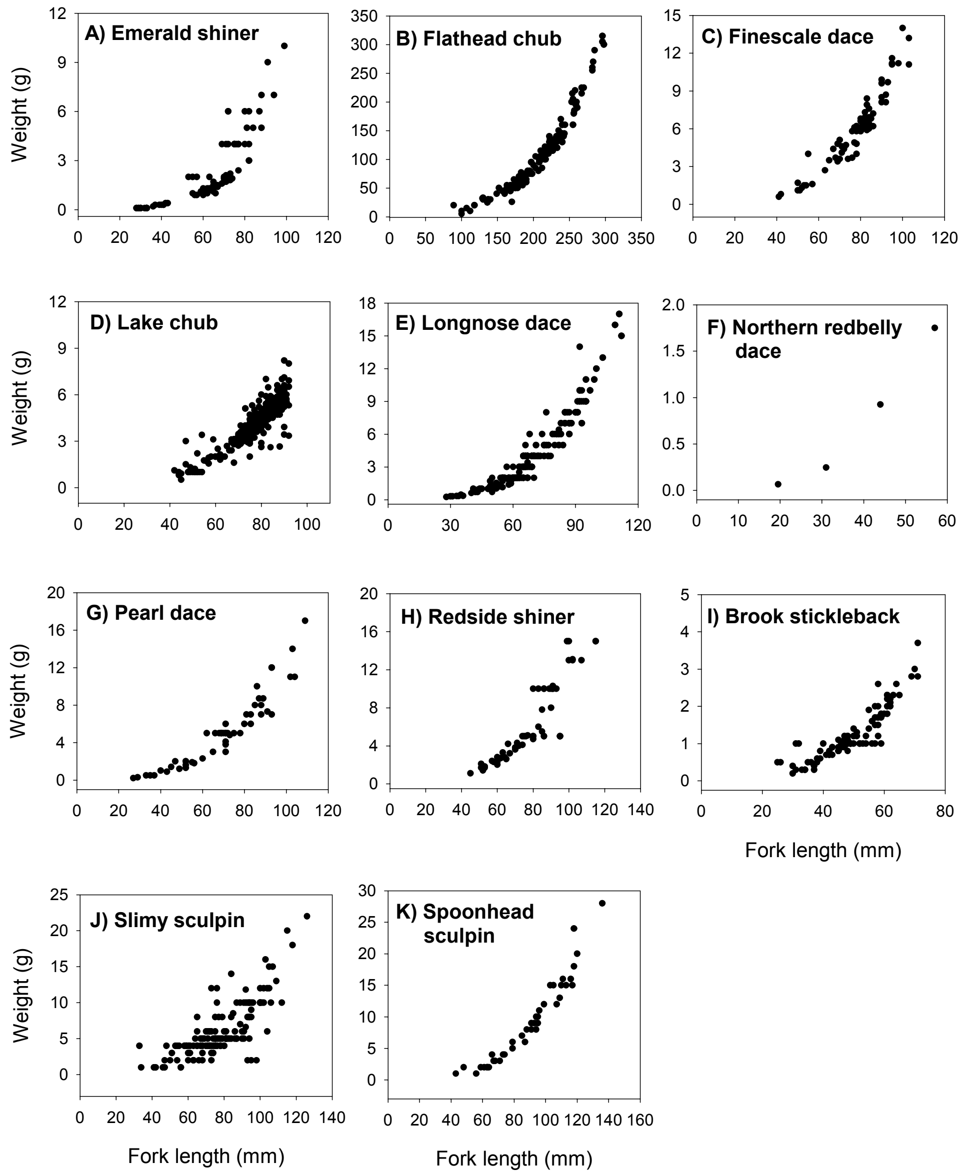

Figure 6. Relationships between wet weight (g) and total length (mm) of 11 species of stream fish from the Kakwa and Simonette River basins, Alberta. Data reflect that collected in the present study combined with historical data obtained from the Alberta Conservation Association, Government of Alberta (Fisheries Management Information System Database), Golder Associates (2000) and Carlander (1969) (northern redbelly dace after correcting for total versus fork lengths).

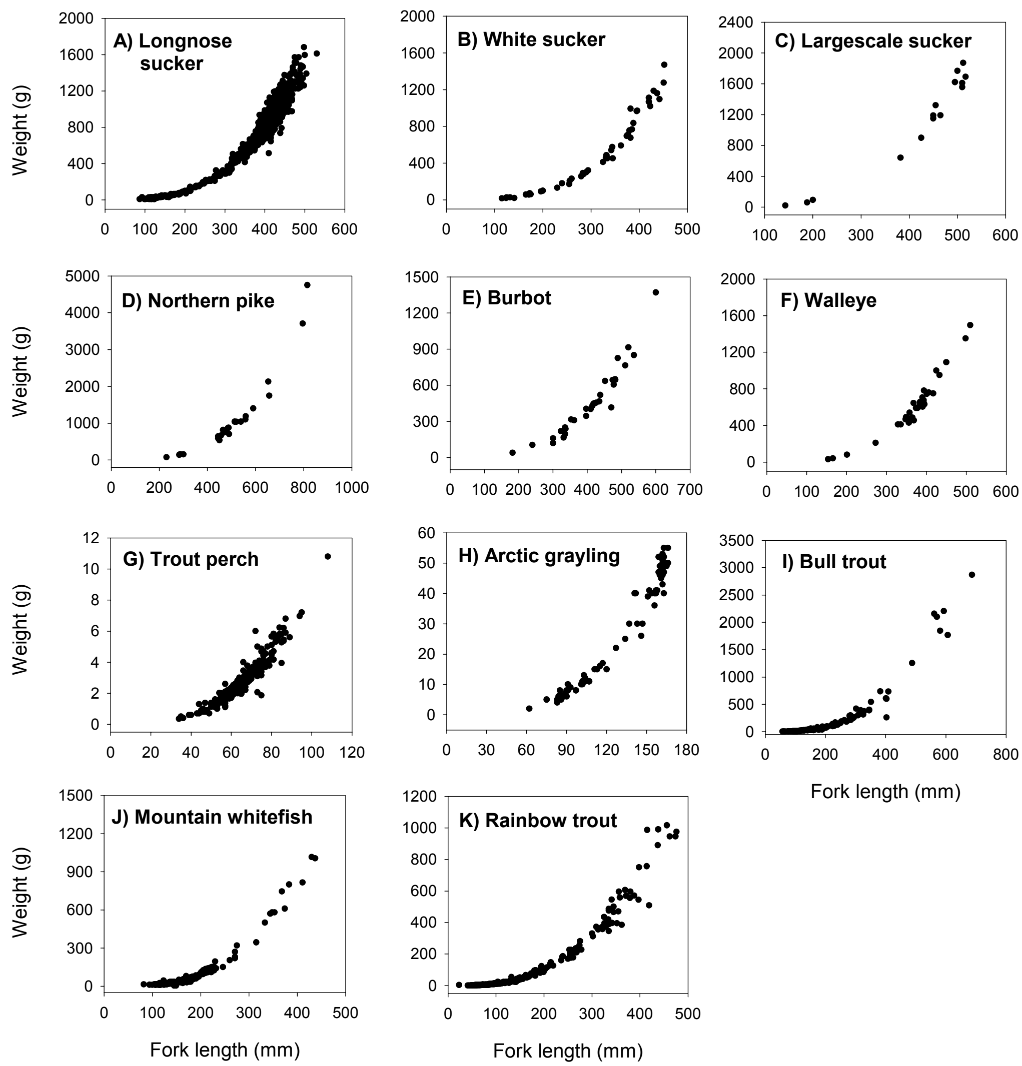

Figure 7. Relationships between wet weight (g) and total length (mm) of 11 species of stream fish from the Kakwa and Simonette River basins, Alberta. Data reflect that collected in the present study combined with historical data obtained from the Alberta Conservation Association, Government of Alberta (Fisheries Management Information System Database) and Golder Associates (2000).

Table 5. Summary of power functions ($y = ax^b$) between weight (g) and total length (mm) of 22 species of fish from streams in the Kakwa and Simonette River basins, Alberta. Historical data were obtained from the Government of Alberta, Fisheries Management Information System Database, Golder Associates (2000), data collected during the present study, and Carlander (1969) (northern redbelly dace after correcting for total versus fork lengths). a = intercept, b = exponent of fish length. R^2 = coefficient of determination.

Species	**A**	**b**	**R^2**
Emerald shiner	0.000000066	4.0996	0.84
Flathead chub	0.000002962	3.2408	0.98
Finescale dace	0.000008472	3.0765	0.93
Lake chub	0.000010906	2.9406	0.88
Longnose dace	0.000003639	3.2540	0.95
Northern redbelly dace	0.000204517	2.2135	0.98
Pearl dace	0.000024974	2.8354	0.87
Redside shiner	0.000032345	2.7699	0.73
Brook stickleback	0.000022351	2.7720	0.86
Slimy sculpin	0.000092035	2.5257	0.66
Spoonhead sculpin	0.000006547	3.1189	0.95
Longnose sucker	0.000018831	2.9297	0.93
White sucker	0.000001783	3.3441	0.98
Largescale sucker	0.000005134	3.1474	0.98
Northern pike	0.000000742	3.3554	0.98
Burbot	0.000003315	3.0958	0.97
Walleye	0.000007177	3.0742	0.97
Trout-perch	0.000027550	2.7474	0.88
Arctic grayling	0.000011967	2.9897	0.97
Bull trout	0.000028924	2.8281	0.97
Mountain whitefish	0.000006141	3.1224	0.98
Rainbow trout	0.000071814	2.6798	0.98

3.6 Statistical Analyses

3.6.1 Refining databases: the rationale for watershed-specific assessments of cumulative effects

We evaluated whether to merge data from the two watersheds by comparing selected population and community-level characteristics of fish communities in the Kakwa with that in the Simonette.

Focus and rationale. We evaluated whether to merge data from the two watersheds by comparing selected population and community-level characteristics of fish communities in the Kakwa and Simonette watersheds. These comparisons included species richness, frequency of occurrence, total and species-specific density and biomass, relative abundance and hierarchical clustering to identify similarities in fish communities among sites. To avoid potential confounding effects of differences in watershed disturbance and fish occurrence among sites, population and community structure comparisons were restricted to sites that contained fish and where watershed disturbances were <15% of study basins. These two criteria resulted in analyses of 88 and 73 sites from the Simonette and Kakwa watersheds, respectively.

Similarities in fish community structure in the Kakwa and Simonette watersheds were completed using hierarchical clustering of fish density data.

While overall comparisons of species richness, frequency of occurrence, density and biomass were completed qualitatively, we also completed hierarchical clustering to compare community structure among sites. Similarities in fish community structure in the Kakwa and Simonette watersheds were completed using hierarchical clustering of fish density data. Because rare taxonomic groups often provide little information to multivariate analyses and tend to add noise to the data, we combined several relatively rare species into lower taxonomic groups. Combinations within familial groups formed the following taxonomic categories: 1) chub (lake chub, flathead chub), 2) dace (pearl dace, finescale dace, longnose dace, northern redbelly dace), 3)shiner (emerald shiner and redside shiner) and, 4) sucker (white sucker, longnose sucker, largescale sucker). To avoid potential problems with the misidentification of slimy sculpin and spoonhead sculpin, these species were also combined into a single taxonomic group (i.e., sculpin). Burbot, northern pike and walleye were excluded from analyses because of low abundance and occurrence. As a result, clustering was completed using the following 11 taxonomic groups: chub, dace, shiner, trout-perch, brook stickleback, Arctic grayling, mountain whitefish, bull trout, rainbow trout, sucker and sculpin.

Fish abundance data were classified using the Bray-Curtis association measure and clustered using the unweighted pair-group method using arithmetic averages, an agglomerative hierarchical fusion method.

Fish abundance data were classified using the Bray-Curtis association measure and clustered using the unweighted pair-group method using arithmetic averages (UPGMA), an agglomerative hierarchical fusion method. The number of groups was determined by examining group structure to ensure that groups contained meaningful numbers of sites.

Focal question 1:

Is the presence of fish, game fish and individual species affected by watershed disturbances?

3.6.2 Focal question 1: Is the presence of fish, game fish and individual species affected by watershed disturbances?

The extent to which the presence of fish, game fish and selected individual species could be predicted from watershed attributes was determined using logistic regression.

The extent to which the presence of fish, game fish and selected individual species could be predicted from watershed attributes was determined using logistic regression. Logistic regression defines the relationship between the logarithm of the odds of a response event (i.e. log(p/(1-p)), where p is the probability of response event) and explanatory variables using maximum mean likelihood estimates (Schlotzhauer 1993). When applied to fish presence data, the probability of the response is the number of streams in which fish are present divided by the total number of streams sampled (i.e. the proportion of streams with fish present). The predictive power of logistic regression was based on the statistical significance of the model, AIC and -2LL statistics, and classification success using a jackknife process to quantify cross-validation (CTABLE option in Proc Logistic) to reduce classification bias. The jackknife procedure calculates classification success after sequentially removing one observation and is based on N (number of sites) permutations.

Models predicting the presence of fish were also derived for at the landscape and stream order scales.

Logistic regression was used to predict the presence of fish, game fish, and individual species at both the landscape and multiple watershed -scales of: 1) watersheds <70 $km^2$and using Strahler stream order, 2) 1^{st} and 2^{nd} order watersheds and 3) 3^{rd} and 4^{th} order watersheds. At the landscape level, the majority of variables describing stream size (i.e., watershed area, bankfull width, wetted width, discharge, depth) were correlated (Pearson correlation coefficients, $P<0.05$) with instream characteristics (e.g., substratum size composition, stream slope). Thus, for initial descriptive purposes, we developed landscape models using watershed area and a reduced set of poorly correlated habitat and disturbance attributes (Pearson correlation coefficients ≤ 0.5). The objective of these models was to gain an overall understanding of the predictability of fish occurrence at the landscape scale.

Models derived for moderately small watersheds (<70 km^2) and for 1st & 2nd order streams and 3rd & 4th order streams provide the opportunity to quantify the predictive power of the larger set of variables (including many of the disturbance attributes) on fish presence. Many of the disturbance descriptors related to watershed area at the landscape levels are not strongly related to watershed area within stream order classes.

Prior to completing logistic regressions, correlation analyses were completed on all stream and watershed variables to eliminate highly correlated potential predictors and to reduce the large number of predictive variables into a smaller set of variables. Logistic regression included main factor effects and first order interactions of main effects. Lastly, we did not develop logistic regression models where comparisons were compromised due to low sample sizes or poor representation of sites that contained fish or where fish were absent.

Focal question 2:

Is fish density and biomass affected by watershed disturbances?

3.6.3 Focal question 2: Is fish density and biomass affected by watershed disturbances?

Relationships between total and species-specific density and biomass were evaluated using linear and non-linear regression models. Simple linear regression and multiple regression were used to test for relationships between: i) total and species-specific density and biomass estimates and ii) instream habitat variables and watersheds attributes including disturbance characteristics. For multiple regression analyses, we used the forward selection option after completing correlation analyses to exclude highly correlated variables (SAS 1987). Prior to analyses, proportion data (e.g., reach slope) were arc-sine transformed, whereas density and other environmental variables were transformed as $\log_{10}(x+1)$ so that data were normally distributed. To ensure that linear models were appropriate, we examined plots of residuals and compared model fit using polynomial and other functions (e.g., power functions).

When fish abundance or biomass were clearly non-linearly related to watershed and disturbance attributes we quantified the amount of variance explained by these variables with polynomial and power functions using the PROC NLIN procedure in SAS (SAS 1987). The fit of polynomial functions was evaluated using model significance (P), coefficient of determination and examination of residual plots. Non-linear regression provides the opportunity to identity disturbance thresholds, similar to dose-response curves, where low levels of disturbance, or doses, do not translate into detectable effects. In contrast, effects are detectable when disturbance levels exceed threshold levels.

Focal question 3:

Is the structure of fish communities affected by watershed disturbances?

The reference-condition approach consists of four steps that establish and test for relationships between fish communities and watershed attributes.

In the Northern Watershed Project these steps include:

i) collection of data on fish communities and habitat variables from the Kakwa and Simonette watersheds

ii) classification of reference (least-impacted) sites using clustering methods based on their taxonomic composition

iii) development of a predictive model for reference sites using discriminant function analysis with selected habitat characteristics

and

iv) application of the predictive model developed from test sites to the suite of test (potentially impacted) sites.

3.6.4 Focal question 3: Is the structure of fish communities affected by watershed disturbances?

<u>The reference condition approach</u>

The effects of watershed disturbances on fish community structure were evaluated using a reference conditions approach where the structure of assemblages at a group of potentially impacted sites (i.e., test sites) is compared to assemblages from a set of least-impacted sites (Reynoldson *et al.* 1997, 2001). If environmental variables explain a substantial part of the variation in assemblage structure at reference sites, empirical models can be constructed to predict the structure of assemblages at potentially disturbed sites. Thus, deviations from the predicted assemblage can be used to assess effects of the disturbance. If the assemblages at the suite of test sites fall within similar environmental conditions as that defined for minimally impacted sites, the group of test sites are considered to be non-impacted.

By identifying assemblage types among the reference sites, via multivariate community analysis techniques (Legendre and Legendre 1998), the assemblage-level prediction can be refined, making the approach more sensitive (Bailey *et al.* 1998). Such an approach is used in aquatic biomonitoring programs (Wright *et al*. 1984, Parsons and Norris 1996, Reynoldson *et al.* 2001), including those focused on assessments of macroinvertebrate and more recently fish communities (Joy and Death 2000, Tonn *et al.* 2003).

Reynoldson *et al.* (1997) described the reference condition approach to characterize the biological conditions of a region and how to compare test sites with reference sites. When applied to the Kakwa and Simonette watersheds the steps are: i) collection of data on fish communities and habitat variables (i.e., watershed and instream variables) at a range of reference sites (described above), ii) classification of reference sites using clustering methods based on their taxonomic composition (described below), iii) development of a predictive model for reference sites using discriminant function analysis with selected habitat characteristics (described below), and iv) comparison of test sites with one of the references site groups (described below).

Classification techniques. Classification methods were used to identify and group the structure of fish communities in the Kakwa and Simonette watersheds. Because rare taxonomic groups have been shown to provide little information to multivariate analyses and tend to add noise to the data, we combined several relatively rare species into lower taxonomic groups and excluded some species from analyses. Thus, we combined several species within familial groups to form the following taxonomic groupings: 1) chub (lake chub, flathead chub), 2) dace (pearl dace, finescale dace, longnose dace, northern redbelly dace), 3) shiner (emerald shiner and redside shiner), 4) sucker (white sucker, longnose sucker, largescale sucker). To avoid potential problems with the misidentification, slimy sculpin and spoonhead sculpin were combined into a single taxonomic group (i.e., sculpin). Burbot, northern pike and walleye were excluded from analyses because of low abundance and occurrence. As a result, clustering was completed using 11 taxonomic groups as follows: chub, dace, shiner, trout-perch, brook stickleback, Arctic grayling, mountain whitefish, bull trout, rainbow trout, sucker and sculpin.

Fish abundance data were classified using the Bray-Curtis association measure and clustered using the unweighted pair-group method using arithmetic averages (UPGMA), an agglomerative hierarchical fusion method. The number of groups was determined by examining group structure to ensure that groups contained meaningful numbers of sites.

Identifying fish community types. Following the identification of fish assemblages in the two watersheds through clustering, we used multiple discriminant function analysis (DFA) to predict assemblage type based on instream and watershed characteristics. While the DFA identified six predictors of community structure (i.e., mean percent gravel, mean water depth, mean bankfull width, forest cover, reach slope, and reach elevation), we restricted our DFA models to variables that are robust to watershed disturbances and easily measured (e.g., mean bankfull width, forest cover, reach slope, elevation).

A multiple discriminant function analysis was used to identify variables that best separated fish communities at individual sites into predefined groups formed by classification of the fish community data. We initially completed a forward selection discriminant function analysis (entry and removal criteria = 0.10) to identify habitat variables using the PROC STEPDISC function in SAS (SAS 1997). Based on this selection process, a discriminant function analysis (PROC DISCRIM, SAS 1997) was completed to generate discriminant functions and scores and to predict the probability of group membership. The accuracy of predictions of the discriminant function analyses was determined from overall group and error rate classification success (SAS 1997).

Statistical analyses were typically performed using SAS (SAS 1987), CANOCO (ter Braak and Šmilauer 1998) and PC-ORD (PC-ORD 2000) software with an alpha of 0.05. Entry and removal criteria for multiple regression and discriminant function analyses were typically set at 0.10 and 0.15. Higher removal criteria were often used in logistic regressions to establish relationships between the presence of fish using forest cover attributes.

4.0 RESULTS

4.1 Data Screening and Watershed Characteristics

Following the data screening process, the cumulative effects of watershed disturbances on fish communities were evaluated using 215 and 313 sites in the Kakwa and Simonette watersheds, respectively.

Our data screening process resulted in the removal of 63 and 110 sites from the Simonette and Kakwa watersheds, respectively. Thus, our evaluations of the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette watersheds were completed using data on fish communities and watershed attributes from 215 sites in the Kakwa watershed and 313 sites in the Simonette watershed (Table 6).

In the Kakwa River Basin data on fish communities and instream habitats were derived from historical databases (i.e. Government of Alberta, Fisheries Management Information System, N = 52 sites sampled between 1996-1999) and that collected as part of the Cooperative Fisheries Inventory Program (N = 59 sites sampled between 1994-1998) (Hvenegaard 1998) and the Northern Watershed Project (104 stream sites sampled in 1999-2001). Similarly, in the Simonette River Basin, data on fish communities and instream habitats were derived from historical databases (i.e. Government of Alberta, Fisheries Management Information System, N = 6 sites sampled in 1994 and 1998) and that collected as part of the Cooperative Fisheries Inventory Program (N = 196 sites sampled between 1994-1997) (Hvenegaard 1998) and the Northern Watershed Project (111 stream sites sampled in 2000-2001).

For both watersheds, the majority of sampling effort was directed to stream reaches in small 1st to 4th order streams.

In general, sampling sites were located moderately well throughout both watersheds.

For both watersheds, the majority of sampling effort was allocated to describing fish communities in 1st to 4th order stream reaches with relatively fewer sites in 5th and 6th order reaches (Tables 6 and 7). With the exception of the western most region of the Kakwa watershed, sampling sites were distributed moderately well throughout both watersheds (Figure 8).

Table 6. Summary of stream fish and habitat surveys completed in the Kakwa and Simonette River basins, 1995 to 2001. Data collected between 1995 and 1997 were provided by the Co-operative Fisheries Inventory Program (Hvenegaard 1998) whereas surveys between 1999-2001 were completed as part of the Northern Watershed Project.

Watershed	**Year**	**Period**	**Number of sites**
A) Kakwa			
	1994	19 Oct – 22 Oct	5
	1995	15 July – 26 Sept	22
	1996	28 July – 29 Aug	13
	1997	29 May – 26 Sept	38
	1998	27 May – 27 Sept	25
	1999	27 May – 20 Aug	8
	2000	15 Aug – 18 Aug	53
	2001	14 Aug – 16 Aug	51
		Total	215

	Year	Period	Number of sites
B) Simonette			
	1994	20 July – 29 Oct	38
	1995	8 June – 4 Oct	43
	1996	25 June – 23 Sept.	54
	1997	2 June – 3 Oct	66
	1998	11 Sept	1
	2000	7 July – 18 July	53
	2001	9 July – 12 July	58
		Total	313

Table 7. Total number of first to sixth order stream reaches in the Kakwa (A) and Simonette (B) River basins sampled between 1994 to 2001. Data collected between 1994 and 1999 were completed by the Co-operative Fisheries Inventory Program (Hvenegaard 1998) or are historical records derived from the Provincial Governments Fisheries Management Information System. Surveys in 2000 and 2001 were completed by the Northern Watershed Project.

	Strahler stream order						
Year	**First**	**Second**	**Third**	**Fourth**	**Fifth**	**Sixth**	**Total**
A) Kakwa							
1994	0	0	4	0	1	0	5
1995	1	3	9	6	3	0	22
1996	1	0	8	4	0	0	13
1997	5	11	10	9	2	1	38
1998	1	4	4	7	6	3	25
1999	0	0	0	4	3	1	8
2000	6	12	17	17	1	0	53
2001	12	13	14	8	4	0	51
Total	26	43	66	55	20	5	215
B) Simonette							
1994	5	8	12	7	4	2	38
1995	5	16	10	12	0	0	43
1996	12	12	16	11	3	0	54
1997	9	17	28	10	2	0	66
1998	0	0	1	0	0	0	1
1999	0	0	0	0	0	0	0
2000	5	13	17	9	5	4	53
2001	13	19	17	4	3	2	58
Total	49	85	101	53	17	8	313

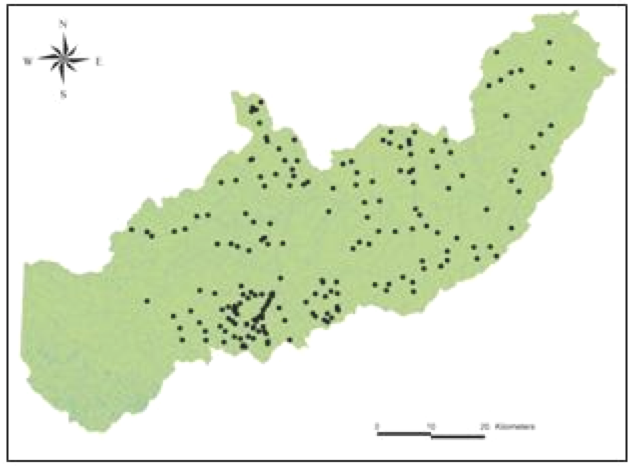

Figure 8. Location of sampling sites (black circles) in the Kakwa and Simonette River basins, Alberta.

4.2 Stream Crossing Inventory

Our GIS queries indicated a total of 572 intercepts between roads and streams (i.e., road crossings on streams) in the Simonette watershed.

Based on measurements of culvert hang-height (i.e., the vertical distance from bottom of the culvert outlet to the water surface) we identified culverts as potentially imposing extreme, high and moderate risk of habitat fragmentation.

Our GIS queries indicated a total of 572 intercepts between roads and streams (i.e., road crossings on streams) in the Simonette watershed. Due to poor access, descriptions of road crossings including the crossing structure (e.g., bridge or culvert) were completed at a total of 477 road-stream intercepts. Based on measurements of culvert hang-height (i.e., the vertical distance from the bottom of the culvert outlet to the water surface) we identified culverts as imposing extreme (hang height: > 30 cm), high (hang height: 10 to 30 cm) or moderate (hang height: 2 to 10 cm) risk of fragmentation. However, our initial analyses indicated that few (<10) culverts within 40 km downstream of all sampling sites were classified as the potentially imposing extreme, high or moderate risk of fragmentation. Consequently, we were unable to test for effects of culvert hang height of fish presence, abundance, and community structure. Additional results of the stream crossing, including geo-referenced locations and descriptions of crossings are provided in Tchir *et al.* (2001).

4.3 Overall Patterns in Fish Communities

4.3.1 Fish community structure in the Kakwa and Simonette River basins

Species richness and frequency of occurrence

Analyses of fish communities from 215 sites in the Kakwa and 313 sites in the Simonette river basins revealed marked differences in species richness between watersheds (Table 8). Species richness in the Simonette watershed (20 species from nine families) was more than two-fold higher than that in the Kakwa watershed (9 species of fish, from five familial groups).

Bull trout, sculpin, Arctic grayling and mountain whitefish were the most frequently occurring species at the study sites in the Kakwa River Basin.

In contrast, sculpin, lake chub, bull trout and white sucker were the most frequently occurring species at study sites in the Simonette watershed.

In the Kakwa River, bull trout and sculpin were the most frequently occurring species (frequency of occurrence = 40.5% and 34.4%, respectively) followed by Arctic grayling, mountain whitefish and rainbow trout each of which occurred at between 11.6% and 15.8% of all sites (Table 9). Longnose dace, longnose sucker, white sucker and burbot were encountered less frequently and occurred at fewer that 5% of all sites (i.e., <10 sites) (Table 8). In contrast, sculpin (45.7%), lake chub (26.2%), bull trout (22.4%), white sucker (18.9%) were the most frequently occurring species in the Simonette watershed with northern redbelly dace, Arctic grayling, mountain whitefish and longnose sucker occurring at between 10.9% to 14.7% of all sites sampled (Table 9).

Table 8. Occurrence of fish species reported from the Simonette (313 sites) and Kakwa River (215 sites) basins, 1994- 2001. Species presence: + = species present, - = species absent. * = game species. Due to potential errors in identification, slimy and spoonhead sculpin were collapsed into one taxonomic group.

Family	Simonette	Kakwa
Cyprinidae		
Lake chub	+	-
Flathead chub	+	-
Finescale dace	+	-
Pearl dace	+	-
Longnose dace	+	+
Northern redbelly dace	+	-
Emerald shiner	+	-
Redside shiner	+	-
Percopsidae		
Trout-perch	+	-
Gasterosteidae		
Brook stickleback	+	-
Percidae		
Walleye*	+	-
Salmonidae		
Arctic grayling*	+	+
Bull trout*	+	+
Mountain whitefish*	+	+
Rainbow trout*	-	+
Esocidae		
Northern pike*	+	-
Catostomidae		
Longnose sucker	+	+
White sucker	+	+
Largescale sucker	+	-
Cottidae		
Sculpin	+	+
Gadidae		
Burbot*	+	+

Percent composition and density

Based on density estimates, fish communities in the Kakwa were numerically dominated by sculpin (percent composition = 50.6%), rainbow trout (24.1%), bull trout (16.8%) and Arctic grayling (5.2%). When combined, these four species accounted for 96.7% of all fish collected. In contrast, northern redbelly dace (31.43%), sculpin (23.1%), lake chub (15.8%), white sucker (8.7%), brook stickleback (6.1%) and pearl dace (6.4%) were the predominant species in the Simonette and accounted for 91.5% of all fish sampled (Table 9).

Overall density of fish in Kakwa watershed (Mean = 1.85 individuals/ 100 m^2) was about four-fold lower than that in the Simonette watershed (Mean = 7.32 individuals 100 m^2). Densities of sculpin, rainbow trout, and bull trout exceeded 0.1 individuals 100 m^2. In contrast, eight species of fish in the Simonette (northern redbelly dace, sculpin, lake chub, white sucker, brook stickleback, pearl dace, finescale dace, longnose sucker and pearl dace exceeded densities of 0.1 individuals / 100 m^2 (Table 9).

4.4 Overall Patterns in Watershed Disturbances

Overall levels of watershed disturbances were highly variable between the Kakwa and Simonette River Basins.

Overall levels of watershed disturbance were highly variable both within and among watersheds (Table 10). However, the level of watershed disturbance arising from industrial activity in the Simonette watershed (18.7%) was almost twice that in the Kakwa watershed (10.2%). In relative terms, the majority of watershed disturbances in both the Kakwa and Simonette watersheds results from forest harvesting (Kakwa = 84.1%, Simonette = 84%) rather than roads (Kakwa = 3.8%, Simonette = 3.3%), pipelines (Kakwa = 1.1%, Simonette = 1.2%) and seismic lines (Kakwa = 9.5%, Simonette = 9.3%) (data calculated from overall means; Table 10).

The overall average level of watershed disturbance in 1st to 5th order watersheds in the Simonette (ca. 19%) is about twice that in the Kakwa river basin (ca. 10%).

For both the Kakwa and Simonette River basins, density of roads within 1st to 5th order watersheds was highly variable, ranged from 0.07 to 0.39 km / km^2 and on average was almost 2-fold higher in the Simonette (0.33 km / km^2) than in the Kakwa (0.17 km / km^2). Density of seismic lines and oil and gas well sites was also higher in the Simonette compared to the Kakwa watershed. In contrast, density of pipelines in the Kakwa exceeded that in the Simonette (Table 10).

On an areal basis, the vast majority of human-induced watershed disturbance (>80%) resulted from forest harvesting which exceeded that due to seismic lines, roads and pipelines.

Table 9. Frequency of occurrence (%), percent composition (%) and mean (±1 SE) density (number / 100 m^2) of fish from the Kakwa and Simonette River basins, 1994-2001. Data are overall averages from 215 (Kakwa) and 313 sites (Simonette) from first to sixth order stream reaches. Percent composition data were calculated from density estimates (i.e., numbers of fish / 100 m^2).

Common name	Frequency of occurrence		Percent composition		Density	
	Kakwa	Simonette	Kakwa	Simonette	Kakwa	Simonette
Cyprinidae						
Lake chub	0	26.20	0	15.83	0	1.159±0.217
Flathead chub	0	0.96	0	0.0001	0	0.000007±0.000004
Finescale dace	0	2.88	0	1.54	0	0.113±0.070
Pearl dace	0	9.27	0	6.43	0	0.471±0.142
Longnose dace	1.40	9.58	0.05	0.67	0.001±0.001	0.049±0.013
Northern redbelly dace	0	10.86	0	31.43	0	2.301±0.706
Emerald shiner	0	2.88	0	1.33	0	0.097±0.053
Redside shiner	0	6.07	0	0.78	0	0.057±0.045
Percopsidae						
Trout-perch	0	3.83	0	0.53	0	0.039±0.023
Gasterosteidae						
Brook stickleback	0	6.09	0	6.05	0	0.443±0.339
Percidae						
Walleye	0	0	0	0.0001	0	0.00001±0.00001
Salmonidae						
Arctic grayling	15.81	14.70	5.18	0.29	0.096±0.0001	0.021±0.007
Mountain whitefish	15.81	13.42	2.27	0.36	0.042±0.015	0.026±0.008
Bull trout	40.47	22.36	16.80	1.24	0.311±0.0750	0.091±0.023
Rainbow trout	11.63	0	24.10	0	0.447±0.388	0
Esocidae						
Northern pike	0	0.64	0	0.0014	0	0.0001±0.0001

Table 9 continued

Table 9 continued.

Common name	**Frequency of occurrence**		**Percent composition**		**Density**	
	Kakwa	**Simonette**	**Kakwa**	**Simonette**	**Kakwa**	**Simonette**
Catostomidae						
Longnose sucker	2.79	12.78	0.32	1.64	0.006±0.0031	0.120±0.033
White sucker	0.47	18.85	0.32	8.70	0.006±0.006	0.637±0.253
Largescale sucker	0	0.32	0	0.01	0	0.0001±0.0001
Cottidae						
Sculpin species	34.42	45.69	50.6	23.09	0.937±0.233	1.690±0.271
Gadidae						
Burbot	0.93	3.39	0.05	0.08	0.001±0.0009	0.006±0.002
Total density					1.85±0.45	7.32±1.13

Table 10. Summary of selected disturbance attributes (mean±1SD) from 1st to 6th order study reaches from the Kakwa and Simonette River basins. %HARV = % harvested, %RECENT-HARV = % harvested within last ten years. Disturbance attributes expressed as densities = Number / km^2. Total number of watersheds and numbers within each watershed order are provided in parentheses. Density attributes: road density = lineal km of road / km^2, seismic density =lineal km of seismic lines / km^2, pipeline density = lineal km of seismic lines / km^2, density of wells = number / km^2, density of stream crossings in the watershed = total number of stream crossings / km^2.

Disturbance attribute	**Kakwa** Overall (N=215)	1st (N =26)	2nd (N=43)	3rd (N=66)	4th (N=55)	5th (N=20)	**Simonette** Overall (N = 313)	1st (N = 49)	2nd (N = 85)	3rd (N = 101)	4th (N =53)	5th (N = 17)
DENSITY OF LINEAR DISTURBANCES												
Density of roads	0.17±0.21	0.19±0.27	0.23±0.28	0.17±0.18	0.14±.0.15	0.07±0.07	0.33±0.32	0.39±0.52	0.33±0.35	0.33±0.23	0.30±0.18	0.31±0.17
Density of seismic lines	1.64±1.03	1.93±1.39	1.56±0.93	1.52±0.72	1.86±1.26	1.40±0.86	2.94±1.11	2.93±1.34	2.82±1.18	2.98±1.05	2.84±0.84	3.43±0.89
Density of pipelines	0.10±0.21	0.03±0.11	0.93±0.14	0.10±0.19	0.05±0.12	0.01±0.03	0.03±0.10	0.05±0.18	0.03±0.07	0.04±0.08	0.04±0.07	44±44
Density of wells	0.13±0.23	0.15±0.31	0.27±0.32	0.14±0.19	0.07±0.11	0.02±0.03	0.21±0.36	0.21±0.52	0.21±0.32	0.24±0.34	0.22±0.28	0.01±0.2
AREAL MEASURES OF DISTURBANCES												
% DISTURBED-WA	10.19±13.50	14.9±16.4	13.6±16.7	10.9±12.9	7.94±11.0	2.92±4.56	18.73±15.07	16.76±16.31	17.47±16.89	19.22±15.63	24.81±10.18	17.16±12.15
% HARVESTED-WA	8.57±012.78	13.2±15.7	11.5±15.9	9.3±12.1	6.42±10.5	1.90±4.31	15.74±15.04	13.43±15.81	14.68±16.86	16.20±15.71	22.04±10.56	14.16±12.21
% ROADS	0.39±0.51	0.37±44	0.71±0.10	0.41±0.49	0.32±0.35	0.15±0.16	0.62±0.63	0.74±1.03	0.61±0.69	0.63±0.48	0.62±0.35	0.51±0.31
% PIPELINES	0.11±0.28	1.14±0.82	0.30±0.45	0.12±0.26	0.05±0.17	0.004±0.01	0.22±.0.42	0.35±0.63	0.18±0.41	0.21±0.37	0.13±0.25	0.002±0.002
% SEISMIC	0.97±0.60	1.14±0.82	0.92±0.55	0.90±0.42	1.11±0.74	0.83±0.051	1.74±0.65	1.72±0.78	1.67±0.70	1.75±0.62	1.67±0.49	2.02±0.52
STREAM CROSSING DISTURBANCES												
Total crossing density	1.23±0.95	1.13±1.39	1.16±0.95	1.12±0.69	1.51±1.05	1.20±0.72	6.02±76.41	28.88±191.2	1.69±1.53	1.68±1.05	1.68±1.05	1.58±0.96
Total number of stream crossings within 40 km downstream of site	11.75±10.28	13.8±9.5	14.3±11.2	11.8±11.3	10.0±9.24	9.50±8.69	18.81±12.42	18.32±12.91	20.19±12.40	18.79±12.91	19.42±12.31	14.12±9.39
Number of stream crossings by roads within 40 km downstream of site	1.34±1.92	1.0±1.2	1.19±1.42	0.98±1.35	2.15±2.98	1.25±0.91	1.48±1.34	1.38±1.38	1.29±1.29	1.67±1.40	1.34±1.31	1.76±1.09
Number of stream crossings by seismic lines within 40 km downstream of site	9.67±10.03	12.0±8.8	12.0±11.2	9.95±11.11	7.38±8.75	7.95±8.46	14.64±11.44	14.00±11.61	16.23±11.88	14.58±11.67	15.28±11.22	9.76±8.14
Number of culverts crossings within 40 km downstream of site	-	-	-	-	-	-	0.19±0.56	0.22±0.91	0.14±0.38	0.30±0.59	0.09±0.30	0.18±0.39

4.5 Refining Databases: The Rationale for Watershed-Specific Assessments of Cumulative Effects

Hierarchical cluster analyses of fish density identified five relatively discrete fish assemblages in the Kakwa and Simonette River basins.

The majority of the five fish assemblage types were dominated by sites either exclusively or predominantly from one watershed.

Based on fish density data, hierarchical cluster analyses identified five relatively discrete assemblages in the Simonette and Kakwa watersheds (Figure 9). Assemblage 6 contained only six sites and from a statistical analysis perspective was not a meaningful assemblage type. With the exception of assemblage 5 where fish communities were comprised of relatively equal numbers of sites from the Kakwa (56% of all sites) and Simonette (44%) watersheds, fish communities from assemblages 1 to 4 were dominated by sites from one watershed. For example, assemblage 1 was comprised of fish communities solely from the Simonette (37 of 37 sites) watershed. Fish communities from the Simonette also comprised the majority of sites from assemblage 4 (27 of the 36 sites) whereas fish communities from the Kakwa were dominant in assemblage 2 (13 of 18 sites; 72%) and 3 (90%) (Figure 9).

Assemblage 1 was dominated primarily by chub (percent composition = 33.3%) and dace (23.5%), with shiner, sucker and brook stickleback also prominent (10.8 to 14.9 %). Total density in assemblage 1 was high with a mean total density of 17.8 individuals /100 m^2 despite the absence of mountain whitefish and rainbow trout. In contrast, fish assemblage 2 was dominated by communities from the Kakwa watershed with mountain whitefish (33.2%), Arctic grayling (23.6%), bull trout (19.4%) and to a lesser extent sculpin (13.9%) and rainbow trout (7.6%) being numerically dominant. Mean total fish density in assemblage 2 was low (1.3 individuals /100 m^2) and brook stickleback, chub, shiner and trout-perch were absent. Bull trout was the numerically dominant species in assemblage 3 and accounted for 94.1% of all individuals. Arctic grayling (2.2%) and sculpin were also present but numerically rare. Assemblage 3 contained only bull trout, Arctic grayling, and sculpin with none of the remaining 8 taxonomic groups present (i.e., brook stickleback, chub, dace mountain whitefish, rainbow trout, shiner, sucker and trout-perch). Mean total fish density in assemblage 3 was low (1.4 individuals /100 m^2). Assemblage 4 was dominated by sculpin (83.1%) with bull trout 7.8%) and dace (3.1%) also predominant. Brook stickleback, chub, mountain whitefish and sucker were also present but not numerically abundant. Rainbow trout, shiner and trout-perch were absent from assemblage 4 and mean total density was moderately high (8.3 individuals /100 m^2). Assemblage 5 was dominated by sculpin (73.8%) and bull trout (15.0%) and to a lesser extent mountain whitefish (4.9%) and rainbow trout (3.5%). Arctic grayling, sucker and trout-perch were also present but accounted for <3% of all fish collected. Mean total fish density was very low (0.94 individuals /100 m^2).

Marked differences in fish communities from the Kakwa and Simonette watersheds suggest that assessments of cumulative effects of watershed disturbances should be completed separately for each river basin.

These data suggest marked differences in fish communities from minimally disturbed (i.e., references) sites in the Kakwa and Simonette watersheds and that assessment of cumulative effects of watershed disturbances should be completed separately for each basin.

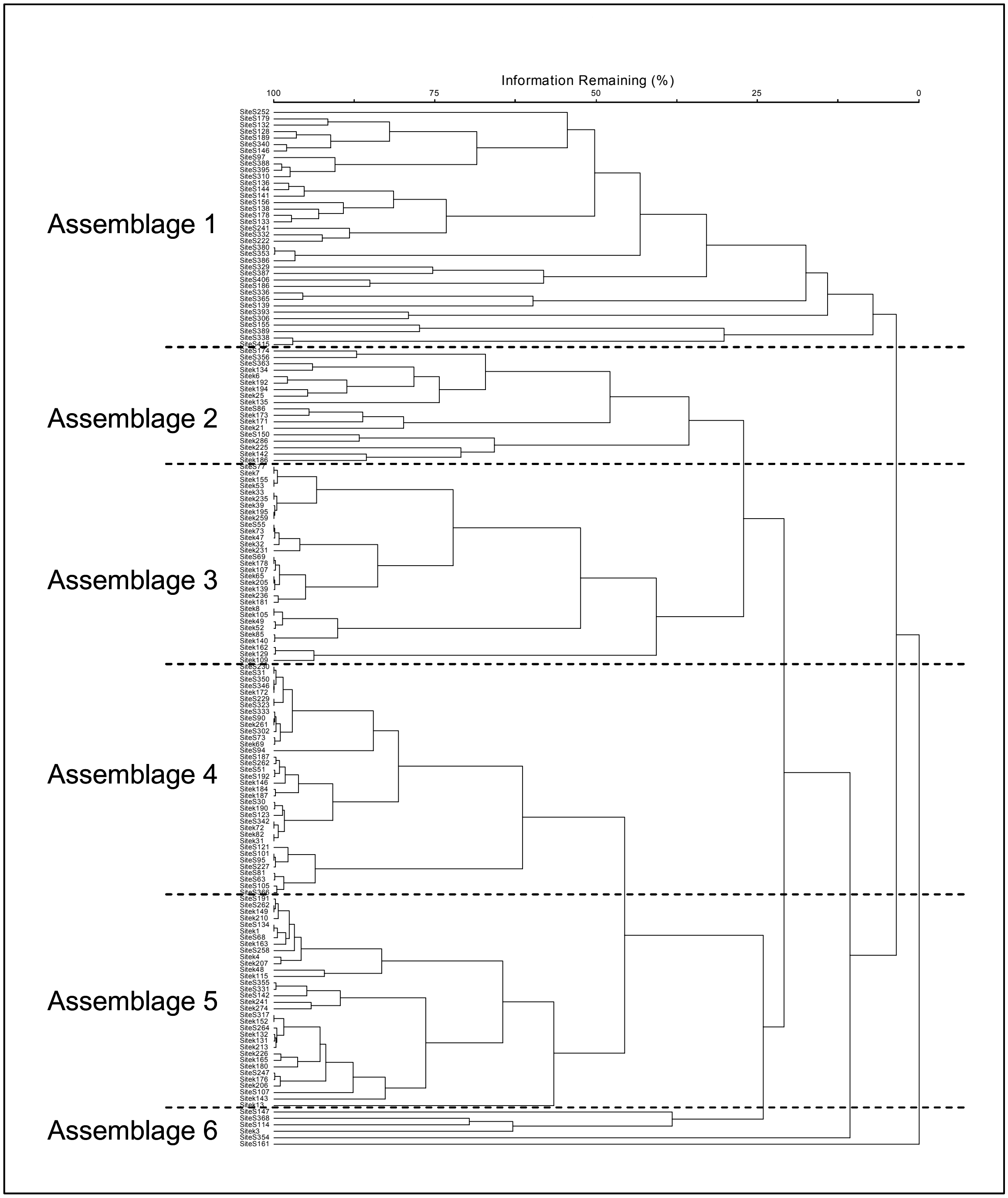

Figure 9. Agglomerative hierarchical cluster analyses of fish communities from the Kakwa and Simonette River basins. Analyses were completed with the Bray-Curtis association measure and the Unweighted pair-group method using arithmetic averages of fish density data. The analysis identified five relatively discrete fish assemblages.

Focal Question 1:

Is the presence of fish, game fish and individual species affected by watershed disturbances?

4.6 Focal Question 1: Is the Presence of Fish, Game Fish and Individual Species Affected by Watershed Disturbances?

4.6.1 General patterns in fish occurrence

The presence of fish was strongly affected by stream size.

The occurrence of fish was strongly affected by stream size (i.e., order) and to a lesser extent watershed type (i.e., the Kakwa and Simonette) (Figures 10 & 11). In the Kakwa watershed, fish occurred relatively infrequently at sites located in first (ca. 15%) and second order watersheds (ca. 35%) but were typically found in third (73%), fourth (96%) and fifth & sixth order reaches (96%) (Figure 11). In contrast, fish were more prevalent in first (ca. 31%) and second order (ca. 59%) reaches in the Simonette but were typically present in larger stream reaches (79% to 83%) (Figure 11). In both watersheds, fish occurrence typically increased with stream order although for some species, percent frequency of occurrence was highest in fourth rather than in 5 & 6 order reaches (Figure 11).

4.6.2 Landscape models

At the landscape level, the presence of fish, game, and individual species were highly predictable based on watershed and instream habitat variables.

Industrial activities including forest harvesting, seismic lines, pipelines and other oil and gas infrastructure did not have detectable effects on fish occurrence.

With one exception, fish presence in the Kakwa River Basin was unrelated to watershed disturbance.

The notable exception was the statistically significant and negative relation between bull trout presence and percent watershed harvested.

At the landscape level, logistic regression indicated that the presence of fish, game fish and individual species were moderately to highly predictable (jack knife classification success = 67.8 to 92.3%), overall average = 81%). In the majority of cases (17 of the 19 models) watershed area and elevation were the main factors predicting fish occurrence followed by reach slope, stream bankfull width, and to a lesser extent size composition of the substrata were less powerful predictors of fish presence (Table 11, Figures 12 and 13). With one noteworthy exception, analyses completed at the landscape-scale showed that disturbance attributes including percentages of the watershed harvested, dedicated to roads, seismic land pipelines, and powerlines were not statistically significant predictors of fish occurrence, even when entry and removal criteria were dropped to 0.15. Lastly, density of road crossings and numbers of roads crossings immediately downstream of study sites were not statistically significant predictors of fish occurrence (Table 11).

In the Kakwa watershed, the probability of sites containing fish, game fish, mountain whitefish and sculpin increased with watershed area whereas the occurrence of Arctic grayling and rainbow trout generally decreased with elevation (Table 11, Figure 12). The notable exception to our general finding of minimal impacts of watershed disturbance on fish presence results from the statistically significant and negative relationship between bull trout presence and percent watershed harvested in the Kakwa River Basin (Figure 12 D). Bull trout presence was also negatively related to percent watershed disturbed largely because forest harvesting accounted for the majority of watershed disturbances. Additional analyses showed that the presence of bull trout was unrelated to watershed disturbance when the proportion of the watershed harvested was deducted from watershed disturbance calculations. The negative relation between bull trout presence and harvesting is strongly influenced by data from the Prairie Creek sub-basin.

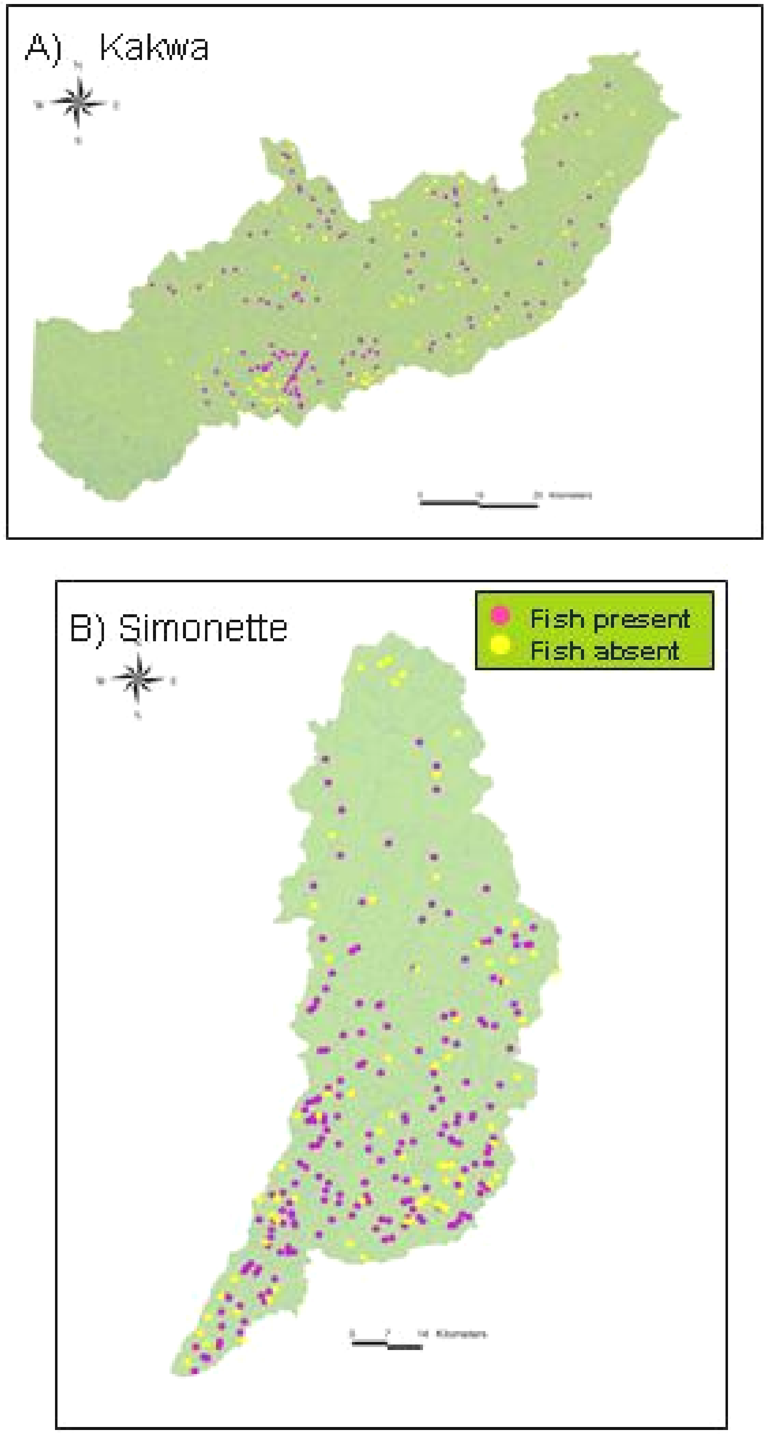

Figure 10. Location of sites in A) Kakwa and B) Simonette River basins supporting fish and those where fish were not detected.

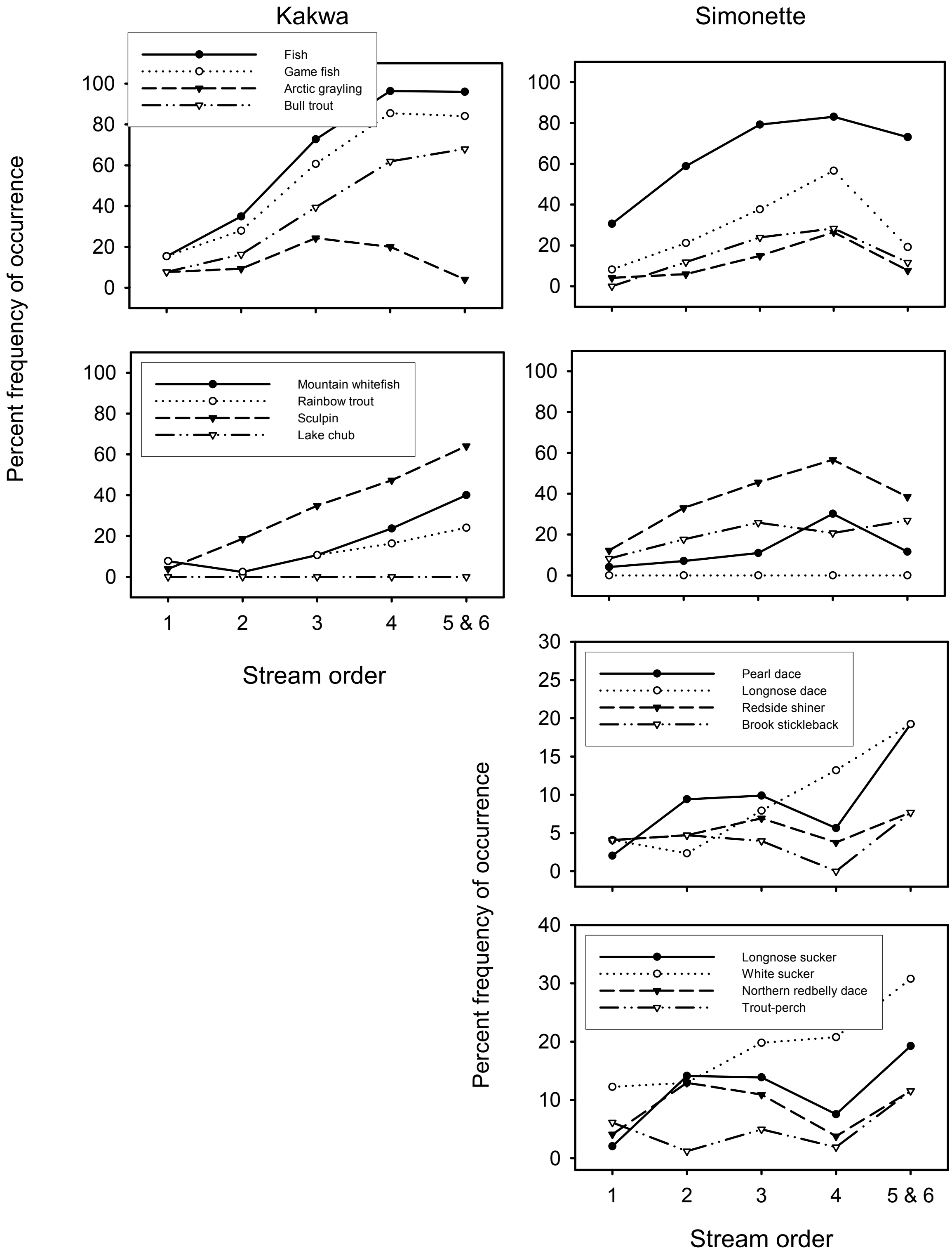

Figure 11. Frequency of occurrence of fish, game fish, and frequently occurring fish species groups in first to fifth and sixth order streams in the Kakwa and Simonette River basins.

Table 11. Summary of landscape-scale logistic regression models predicting the presence of fish (logit) based on watershed and stream characteristics in the Kakwa (A) and Simonette (B) River basins. Analyses were restricted to commonly encountered species and species groups (i.e., sculpin). Fish = all species, Game fish = Arctic grayling, bull trout, burbot, northern pike, walleye. Classification success was calculated using jack knife procedure with N (number of study sites) permutations. Significance levels for intercepts and main factors are shown within brackets: NS = non-statistically significant model (P>0.05), * = P <0.05, ** = P < 0.01. P value derived from the Wald statistic.

Species group	Logistic regression equation	N	P	Classification success
A) Kakwa				
Fish	Y = -0.87 (*) + 0.09 (**) WATERSHED AREA	215	<0.0001	81.9
Game fish	Y = -0.07 (NS) + 0.006 (**) WATERSHED AREA	215	0.017	69.8
Arctic grayling	Y = 3.11 (*) - 0.004 (**) ELEV	215	<0.0001	83.3
Bull trout	Y = 1.21 (**) – 4.34 (**) asin PERCENT WATERSHED HARVESTED - 1.22 log WATERSHED AREA	215	<0.0001	75.3
Mountain whitefish	Y = 2.05 (**) + 0.0006 (**) WATERSHED AREA	215	<0.0001	84.2
Rainbow trout	Y = 1.73 (NS) - 0.0027 (***) ELEVATION	215	0.01	88.4
Sculpin	Y = -1.62 (**) + 0.013 (**) WATERSHED AREA + 0.021 (*) PERCENT LARGE GRAVEL	199	0.0002	67.8
B) Simonette				
Fish	Y = -1.008 (**) +1.48 (**) log WATERSHED AREA	313	<0.0001	77.0
Game fish	Y = -33.37 (**) + 10.32 (**) log ELEVATION + 1.09 (**) log WATERSHED AREA	313	<0.001	73.2
Arctic grayling	Y = -3.01 (**) + 0.72 (**) log WATERSHED AREA	313	<0.001	87.9
Bull trout	Y = -32.6 (**) + 10.32 (**) log ELEVATION	313	<0.0001	80.8
Lake chub	Y = 39.62 (**) - 14.14 (**) log ELEVATION + 0.15 (**) arcsin percent BOULDER	313	<0.0001	80.1
Mountain whitefish	Y = -27.12 (**) + 1.29 (**) log WATERSHED AREA	313	<0.0001	87.9
Pearl dace	Y = 40.62 (**) - 14.66 (**) log ELEVATION	313	<0.0001	90.4
Longnose dace	Y = 26.23 (**) - 9.75 (**) log ELEVATION	313	<0.0001	92.3

Table 11 continued.

Species group	Logistic regression equation	N	P	Classification success
Northern redbelly dace	Y = 67.82 (**) - 23.42 (**) log ELEVATION - 1.27 (**) log WATERSHED AREA	313	<0.0001	90.7
Sculpin	Y = 25.87 (**) – 12.24 (**) log SLOPE + 8.58 (**) log ELEVATION + 2.70 (**) arcsin PERCENT LARGE GRAVEL	311	<0.0001	70.7
Longnose sucker	Y = 34.84 (**) – 12.80 (**) log ELEVATION + 3.03 (**) BOULDER	312	<0.0001	87.5
White sucker	Y = 37.79 (**) – 13.34 (**) log ELEVATION	313	<0.0001	79.9

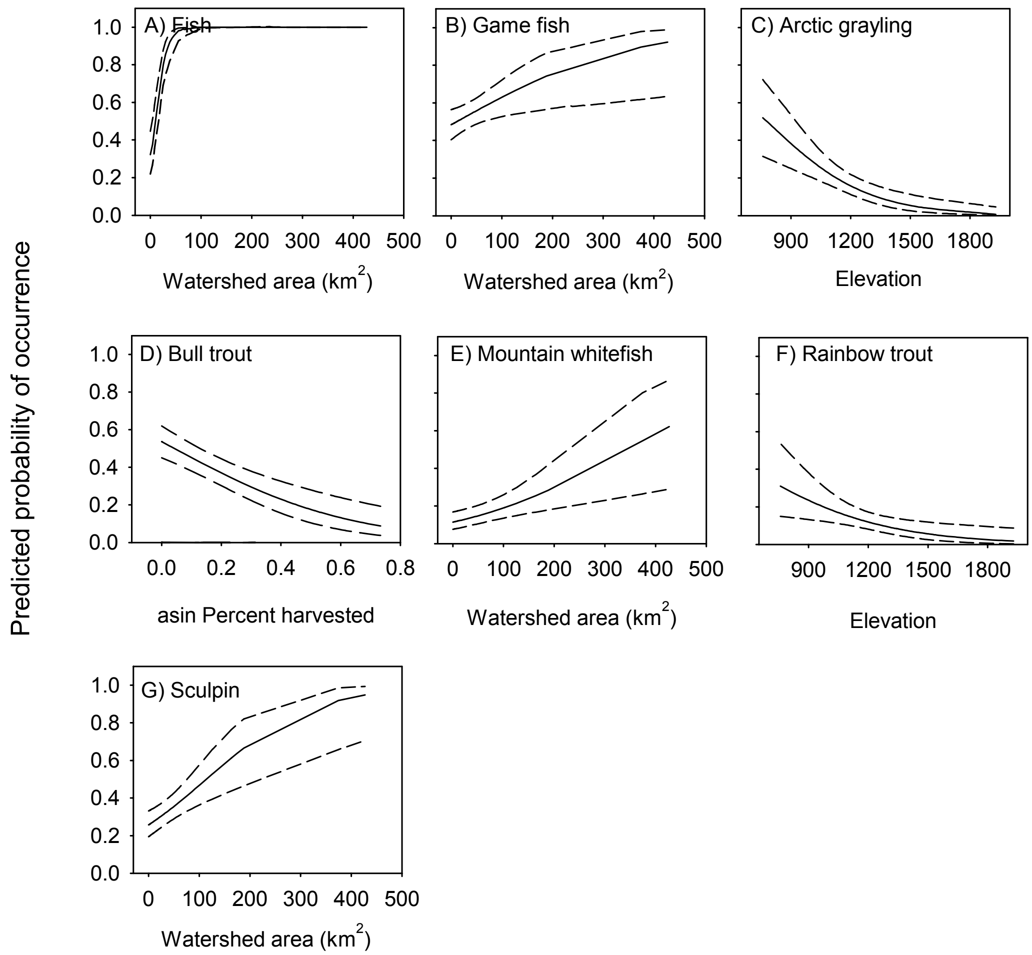

Figure 12. Predicted probability of occurrence of fish, game fish and selected species from the Kakwa River Basin, Alberta. Solid line is the predicted probability with dashed lines representing upper and lower 95% confidence limits. Models represent results from landscape-scale analyses using data from the majority of study sites.

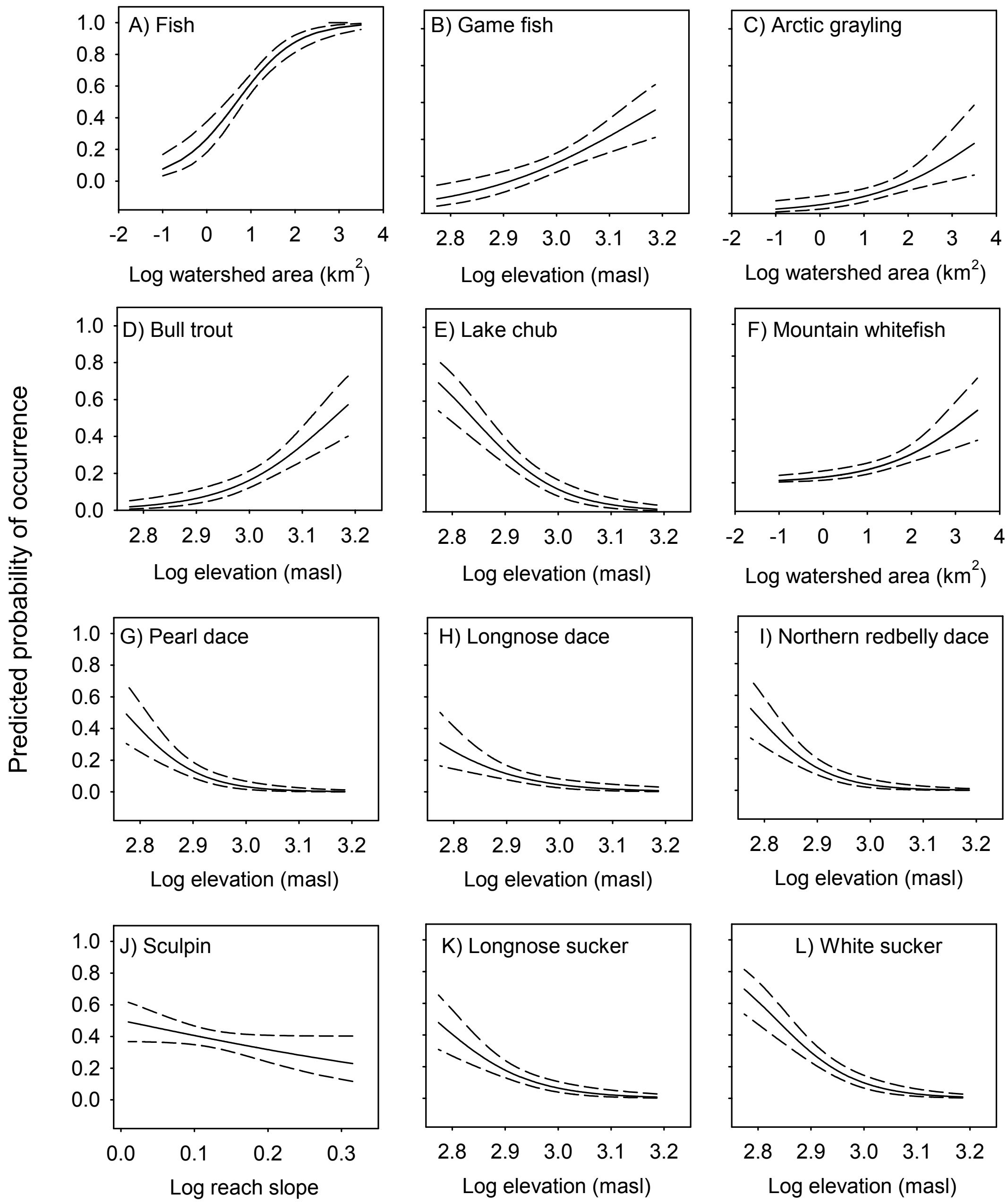

Figure 13. Predicted probability of occurrence of fish, game fish and selected species from the Simonette River Basin, Alberta. Solid line is the predicted probability with dashed lines representing upper and lower 95% confidence limits. Models represent results from landscape-scale analyses using data from the majority of study sites.

In the Simonette watershed, the occurrence of fish, game fish, bull trout, Arctic grayling and mountain whitefish increased with either watershed area or elevation. In contrast, the majority of other species decreased with increasing elevation or slope of the stream reach (Table 11, Figure 13).

4.6.3 Watershed scale and reach scale models

The majority of study sites in the Kakwa (85% of all study watersheds) and the Simonette (78%) watersheds were located within relatively small watersheds of <70 km^2 (Figure 14). Because one of our primary interests was to evaluate the cumulative effects of watershed disturbances at multiple scales, we tested for significant effects of watershed disturbance at both the stream scale (1st and 2nd order combined and 3rd and 4th order reaches combined) and for all sites within watersheds of <70 km^2.

Logistic regression analyses showed that in the majority of cases, fish presence was moderately to highly predictable based on a small suite of recurring variables of watershed area, elevation, reach slope and the size composition of the substratum.

Logistic regression models derived for 1st & 2nd, 3rd & 4th order reaches and sites within watersheds of < 70 km^2 showed that in the majority of cases, fish presence was moderately to highly predictable (classification success 62 to 90%) based on a small suite of recurring variables including watershed area, elevation, reach slope, and the size composition of the substratum (Table 12). With one exception, our initial analyses provided little evidence that the presence of fish was affected by watershed disturbance attributes. In fact, watershed disturbance attributes were identified as statistically significant ($P<0.05$) predictors of the presence of fish, game fish and individual species and species groups in only one of the 52 models initially derived from the Kakwa (16 logistic models) and Simonette River (36 logistic models) basins based on an entry and removal criteria of 0.05.

In the vast majority of cases, indicators of watershed disturbance were non-statistically significant predictors of fish occurrence.

Relaxation of model entry criteria to 0.15 did however result in entry of some disturbance indicators (i.e., percent watershed disturbance) for several of species of dace, shiner and sculpin. However, in these instances disturbance attributes appeared as secondary factors and their inclusion to logistic models did not greatly improve model fit or classification success.

Additional analyses showed that the presence of bull trout was significantly and negatively related to the number of stream crossings located with 40 km downstream of the sampling site.

Our analyses showed that the presence of bull trout in the Simonette watershed was significantly and positively related to elevation. We quantified the ability of disturbance attributes to explain the presence of bull trout at moderately high elevations by performing logistic regression analyses using data from sites located above 1000 meters above sea level (m.a.s.l.). Initial analyses showed that only 5 of the 62 sites (i.e., 8.1%) below this elevation (<1000 m.a.s.l.) supported bull trout. Results from this analysis showed that the presence of bull trout was significantly and negatively related to the number of stream crossings within 40 km downstream (Table 12, Additional analyses, Bull trout). Thus, when combined with results of logistic regression analyses from the Kakwa River Basin, our analyses identified significant relations between bull trout presence and: i) percent watershed harvested (Kakwa River Basin) and ii) stream crossing density (Simonette River Basin).

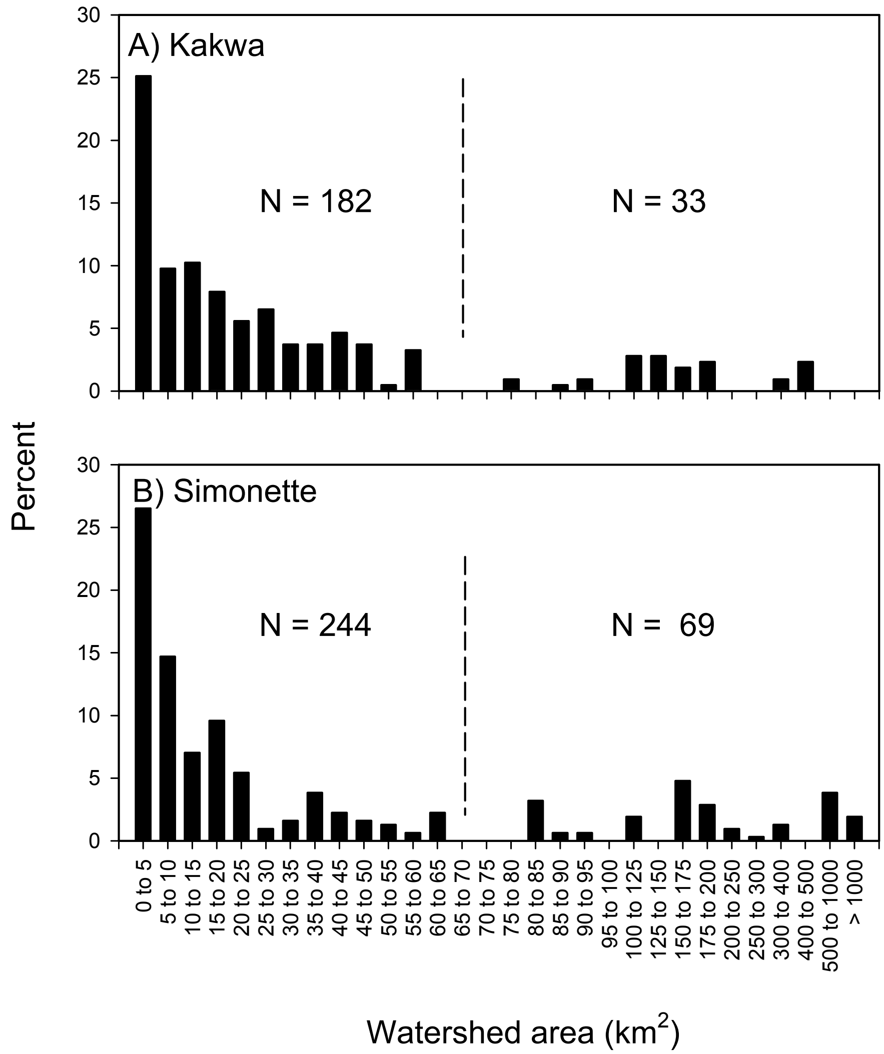

Figure 14. Frequency distributions of watershed areas of study sites in the Kakwa (A) and Simonette (B) River basins, Alberta. Dashed line represents a break in the frequency distribution at 70 km^2.

Table 12. Summary of logistic regression models predicting the presence of fish (logit) in 1) moderate sized-watersheds (i.e., < 70 km^2) and 2) 1st & 2nd and 3) 3rd & 4th order reaches in the Kakwa (A) and Simonette (B) River basins, Alberta. Analyses were restricted to commonly encountered species or species groups. Game fish = Arctic grayling, bull trout, mountain whitefish, rainbow trout, northern pike, walleye; Sucker = white sucker and longnose sucker; Minnows = lake chub, flathead chub, finescale dace, pearl dace, longnose dace, northern redbelly dace, emerald shiner and northern pikeminnow. NS = not significant ($p > 0.05$). Classification success was calculated using a jack knife procedure with N (number of study sites) permutations. Significance levels for intercepts and main factors are shown within brackets. Low occurrences of fish presence or fish absence precluded analyses of all species groups. – insufficient data for model generation. Model abbreviations: Elevation (meters above sea level), temperature = instantaneous water temperature, bankfull width = stream bankfull width (m), gravel = mean proportion gravel, conifer-300 = proportion of conifer encompassed within 300 m of the study site, conifer = proportion of conifer in watershed, UTM-N= Universal Transverse Mercator Northing. Watershed disturbance attributes were not statistically significant ($P < 0.05$) of the presence of fish, game fish, species and species groups in the Kakwa and Simonette River basins. † = highly unbalanced models.

Species group	Logistic equation	Number Total	Present	Absent	P	Classification success (%)
A) Kakwa						
1) Watershed models						
Fish	Y = -1.2661 (**) + 0.13 (**) WATERSHED AREA	180	110	70	<0.0001	78.3
Game fish	Y = -1.44 (**) + 0.05 (**) WATERSHED AREA + 2.17 (*) PERCENT BOULDER	170	81	89	<0.0001	64.7
Arctic grayling †	Y = 2.00 (NS) – 0.004 (**) ELEVATION + 3.20 (**) BOULDER	170	26	144	0.0006	84.7
Bull trout	Y = 1.37 (*) – 5.25 (**) asin PERCENT WATERSHED HARVESTED – 10.93 (**) asin SLOPE – 2.1 (**) asin PERCENT BOULDER	171	61	110	<0.0001	74.9
Mountain whitefish †	Y = -2.92 (**) + 0.038 (**) WATERSHED AREA	180	20	160	0.0053	88.9
Rainbow trout †	Y = 0.74 (NS) + 5.66 (**) PERCENT BOULDER – 0.005 (**) ELEVATION + 0.04 (*) WATERSHED AREA	170	16	154	0.0003	90.0
Sculpin †	Y = -2.58 (**) + 0.053 (**) WATERSHED AREA + 2.58 (*) PERCENT LARGE GRAVEL	160	48	122	<0.0001	73.5
2) First & second order reaches						
Fish	Y = -1.66 (**) + 0.13 (*) WATERSHED AREA	69	19	50	0.028	76.8
Game fish †	Y = -1.99 (**) + 0.144 (*) WATERSHED AREA	69	16	53	0.02	81.2

Arctic grayling	-					
Bull trout	-					
Mountain whitefish	-					
Rainbow trout	-					
Sculpin	-					
<u>3) Third & Fourth order reaches</u>						
Fish †	Y = -0.23 (NS) + 0.083 (**) WATERSHED AREA	121	101	20	0.0009	83.5
Game fish †	Y = 1.079 (**) – 23.25 (**) SLOPE + 2.65 PERCENT BOULDER	115	82	33	0.006	75.7
Arctic grayling†	Y = 2.807 + 4.42 (**) PERCENT BOULDER – 0.004 (*) ELEVATION – 36.63 (*) SLOPE	115	27	88	0.0007	80.0
Bull trout	Y = 2.21 (NS) – 5.88 (**) asin PERCENT HARVESTED – 7.80 (*) asin SLOPE	120	60	60	0.0001	71.9
Mountain whitefish	Y = -2.058 (**) + 0.011 (*) WATERSHED AREA	121	20	101	0.046	81.6
Rainbow trout†	Y = 1.055 (NS) + 3.99 (**) PERCENT BOULDER – 0.003 (*) ELEVATION	115	15	100	0.004	85.2
Sculpin	Y = 0.578 (NS) – 37.85 (**) SLOPE	121	49	72	0.0009	62.0
B) Simonette						
<u>1) Watershed models</u>						
Fish	Y =0.088 (NS) + 0.077 (**) WATERSHED AREA - 38.35 (**) SLOPE + 3.19 (*) PERCENT BOULDER	242	147	95	<0.0001	78.9
Game fish	Y = -6.15 (**) + 0.004 (**) ELEVATION + 0.046 (**) WATERSHED AREA	244	59	185	<0.0001	75.8
Arctic grayling †	Y = -3.17 (**) + 0.046 WATERSHED AREA	244	24	220	<0.0001	90.2
Bull trout †	Y= -5.77 (**) + 0.004 ELEVATION	244	42	202	<0.0001	80.7
Lake chub †	Y = 5.50 (**) -0.008 (**) ELEVATION + 4.26 (**) BOULDER	243	43	200	<0.0001	82.3
Mountain whitefish †	Y = -6.52 (**) + 0.043 (**) WATERSHED AREA + 0.003 (**) ELEVATION	244	19	225	<0.0001	91.8

Pearl dace†	Y = 8.53 (**) – 0.123 (**) ELEVATION	244	21	223	<0.0001	91.0
Longnose dace	-					
Northern redbelly dace †	Y = 9.07 (**) -0.012 (**) ELEVATION	244	27	217	<0.0001	89.3
Sculpin	Y = -4.66 (**) + 0.003 (**) ELEVATION + 3.12 (**) PERCENT LARGE GRAVEL + 0.03 WATERSHED AREA – 38.33 (**) SLOPE	242	90	152	<0.0001	67.8
Longnose sucker †	Y = 5.14 (**) - 0.01 (**) ELEVATION + 0.05 (**) WATERSHED AREA + 42.71 (*) SLOPE	243	20	223	<0.0001	91.8
White sucker †	Y = 5.46 (**) – 0.007 (**) ELEVATION – 64.66 (**) SLOPE + 3.00 (*) BOULDER	242	38	204	<0.0001	82.2
<u>2) First & second order reaches</u>						
Fish	Y = 0.88 (*) – 55.35 (**) SLOPE + 3.65 (*) PERCENT BOULDER	134	63	71	<0.0001	67.9
Game fish	-					
Arctic grayling	-					
Bull trout †	Y = -6.04 (**) - 0.004 (**) ELEVATION	135	13	122	<0.009	90.4
Lake chub †	Y = 5.44 (**) - 0.01 (**) ELEVATION + 6.42 (*) PERCENT BOULDER	134	19	115	0.0002	83.6
Mountain whitefish	-					
Pearl dace	-					
Longnose dace	-					
Northern redbelly dace	-					
Sculpin †	Y = -4.51 (**) + 4.41 (**) PERCENT GRAVEL + 0.002 (**) ELEVATION – 40.7 (*) PERCENT LARGE GRAVEL	134	34	100	0.0004	75.4
Longnose sucker †	Y = 3.55 (*) – 0.007 (**) ELEVATION	135	13	122	0.0018	90.4
White sucker †	Y = 6.27 (**) – 0.009 (**) ELEVATION	135	17	118	0.0002	87.4
<u>3) Third & fourth order reaches</u>						
Fish †	Y = -1.72 (**) – 52.18 (**) SLOPE + 5.16 (*) BOULDER	152	122	30	0.0055	82.9
Game fish	Y = -4.73 (**) + 0.004 (**) ELEVATION	153	61	92	<0.0001	68.0

Arctic grayling †	Y = -1.96 (**) + 0.068 (**) WETTED WIDTH	152	29	123	0.031	81.9
Bull trout †	Y = -7.72 (**) + 0.005 (**) ELEVATION + 3.94 (**) PERCENT LARGE GRAVEL	153	39	114	0.0001	75.8
Lake chub	Y = 5.60 (**) – 0.007 (**) ELEVATION	152	37	116	<0.0001	75.2
Mountain whitefish †	Y = -4.38 (**) 0.003 (**) ELEVATION – 79.74 SLOPE + 3.68 PERCENT BOULDER	152	27	125	0.0076	78.9
Pearl dace †	Y = 6.42 (**) – 0.01 (**) ELEVATION	153	13	140	<0.0001	88.9
Longnose dace †	Y = 3.77 (*) - 0.006 (**) ELEVATION	153	15	138	0.0005	90.2
Northern redbelly † dace	Y = 5.72 (**) - 0.009 (**) ELEVATION	153	13	140	0.0002	88.9
Sculpin	Y = -3.47 (**) - 0.003 (**) ELEVATION + 2.04 (*) PERCENT LARGE GRAVEL	153	76	77	0.0005	67.1
Longnose sucker †	Y = 3.22 (*) - 0.006 (**) ELEVATION + 4.35 (*) PERCENT BOULDER	153	18	135	0.0002	87.6
White sucker †	Y = 5.74 (**) - 0.008 (**) ELEVATION	153	31	122	<0.0001	80.4
4) Additional analyses						
High elevation sites						
Bull trout	Y = 1.56 (NS) -2.60 (**) log crossing density + 0.70 (*) log watershed area	151	46	105	<0.0001	69.5

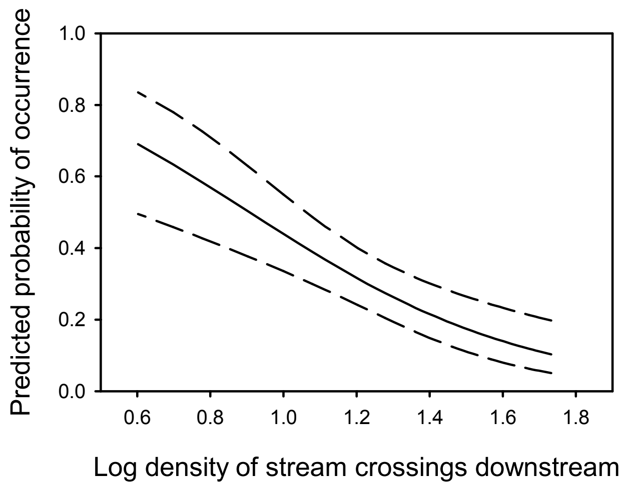

Figure 15. Relationship between the predicted probability of occurrence of bull trout and density of stream crossings downstream of study reaches from moderately high elevation streams in the Simonette River Basin. Solid line is the predicted probability with dashed lines representing upper and lower 95% confidence lines.

Focal Question 2:

Are fish density and biomass affected by watershed disturbances?

4.7 Focal Question 2: Are Fish Density and Biomass Affected by Watershed Disturbances?

4.7.1 Kakwa watershed

Relations between fish density and watershed attributes

In the Kakwa watershed fish densities were primarily related to stream width, elevation and reach slope.

Regression models showed that total fish density and density of the predominant species in the Kakwa watershed were primarily related to stream wetted width (i.e., the width of the water surface), elevation and reach slope. Fish density was generally highest in small streams or those at high elevation and decreased with increasing width or lower elevation (Figure 15). In general regression models explained relatively little to moderate amounts of the overall variance in total density and density of the most abundant species (Overall range in r^2= 0.14 to 51%) and non-linear models did not typically explain appreciably more variance than linear models (Table 13).

In the majority of cases, total fish density and density of the predominant fish species were unrelated to watershed disturbances including harvest blocks, road networks, pipe lines and seismic lines.

In the Kakwa River Basin, the notable exceptions to these findings were the positive relationships between: i) total fish density and percent watershed disturbance and, ii) density of sculpin and percent watershed disturbance.

While the multiple regression predicting total density and density of sculpin identified percent watershed disturbance as a statistically significant variable, it explained very little of the overall amount of variance in total fish density.

With the two exceptions, our analyses also showed that watershed disturbance, stream crossing attributes and their underlying attributes (e.g., percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines) did not explain significant amounts ($P>0.10$) of variance in fish density. However, total fish density was significantly ($P < 0.10$) and positively related to percent watershed disturbed and resulted from the positive relationships between density of sculpin and watershed disturbance (Figure 15 C, Table 13) . While the multiple regression predicting total density and density of sculpin identified percent watershed disturbance as a statistically significant variable, it explained very little of the overall amount of variance in total fish density (i.e.<5%).

Regression analyses completed solely using areal measures of watershed disturbance indicated that the significant effect of total watershed disturbance arises from the percent of watersheds disturbed by harvest blocks and not areas disturbed by roads, seismic lines, pipe lines and well sites. Species-level regression models showed that the positive relation between total fish density and the percent watershed disturbed arises from the positive relation between density of sculpin and percent watershed disturbance (Figure 15 N). The positive relation between density of sculpin and total watershed disturbance also arises from the percent of watersheds disturbed by harvest blocks and not areas disturbed by roads, seismic lines, pipe lines and well sites.

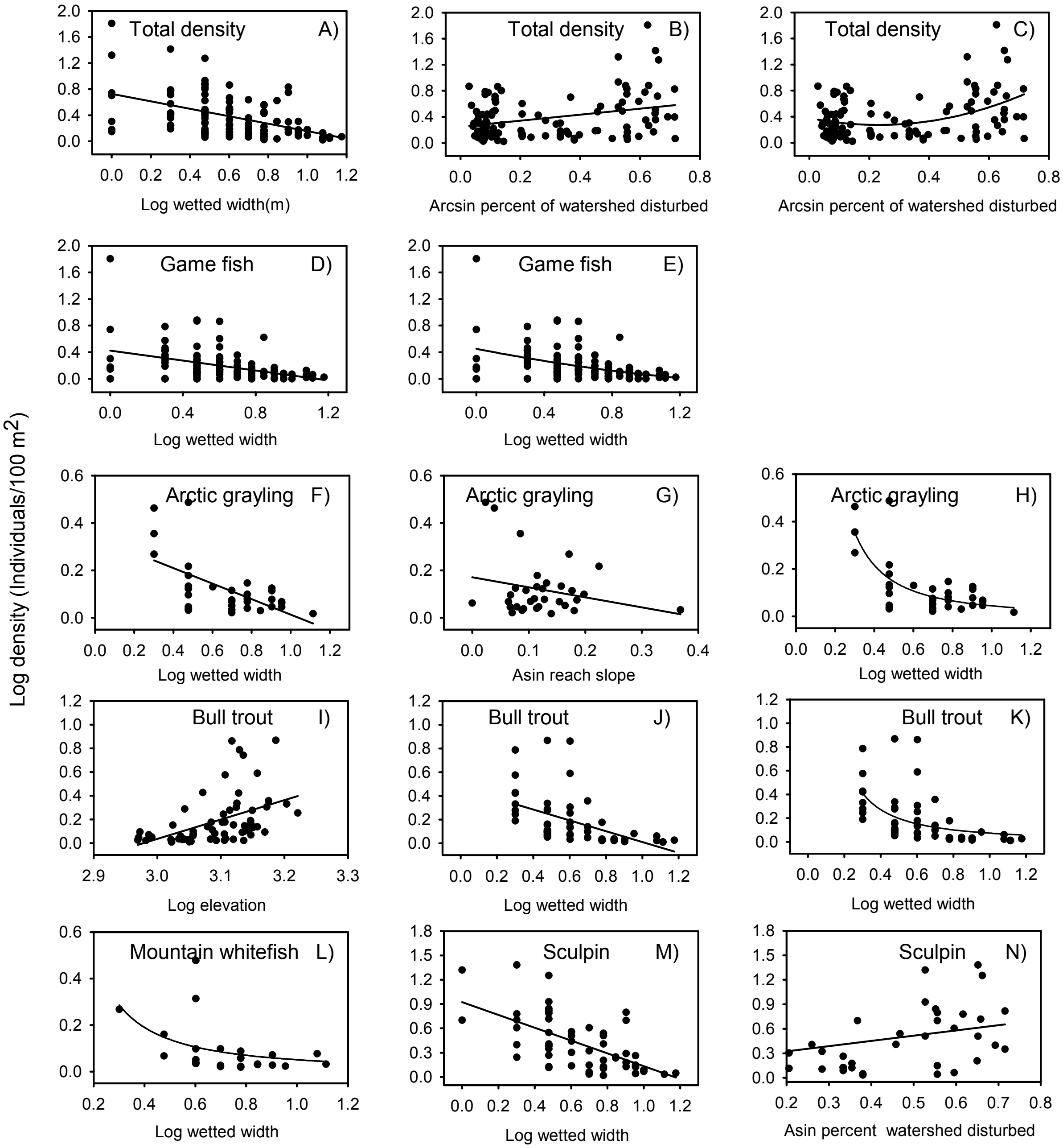

Figure 16. Regression models describing relations between total and species-specific fish density and watershed attributes in the Kakwa River Basin, Alberta. Data transformation abbreviations: asin = arc-sin squareroot transformed, log = $\log_{10}$ (x+1). Equations include linear and non-linear regressions (i.e., power functions and polynomial equations). Regression equations are provided in Table 13. Wetted width = width of the stream water surface.

Relations between fish biomass and watershed attributes

Total fish biomass and biomass of the numerically dominant species and species groups were primarily related to indicators of stream size.

Regression models showed that total fish biomass and biomass of the predominant species were also primarily related to stream size (i.e., the width of the water surface) and the size composition of the river bed (Figure 17, Table 13). Total fish biomass and biomass of bull trout and sculpin was highest in small streams and decreased with increasing stream width. Biomass of game fish, bull trout and sculpin were positively related to the percent of boulders within the substratum.

Watershed disturbance attributes explained relatively little of the overall variance in total biomass and biomass of the predominant species.

As described for fish density, watershed attributes explained relatively little of the overall variance in total biomass and biomass of the predominant species (Overall range in R^2= 0.08 to 50%) and non-linear models did not typically explain appreciably more variance than linear models (Table 13).

Our analyses also showed that watershed disturbances, stream crossing attributes and their component metrics (e.g., percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines) did not explain significant amounts (P>0.10) of variance in fish biomass in the Kakwa River Basin (Figure 17).

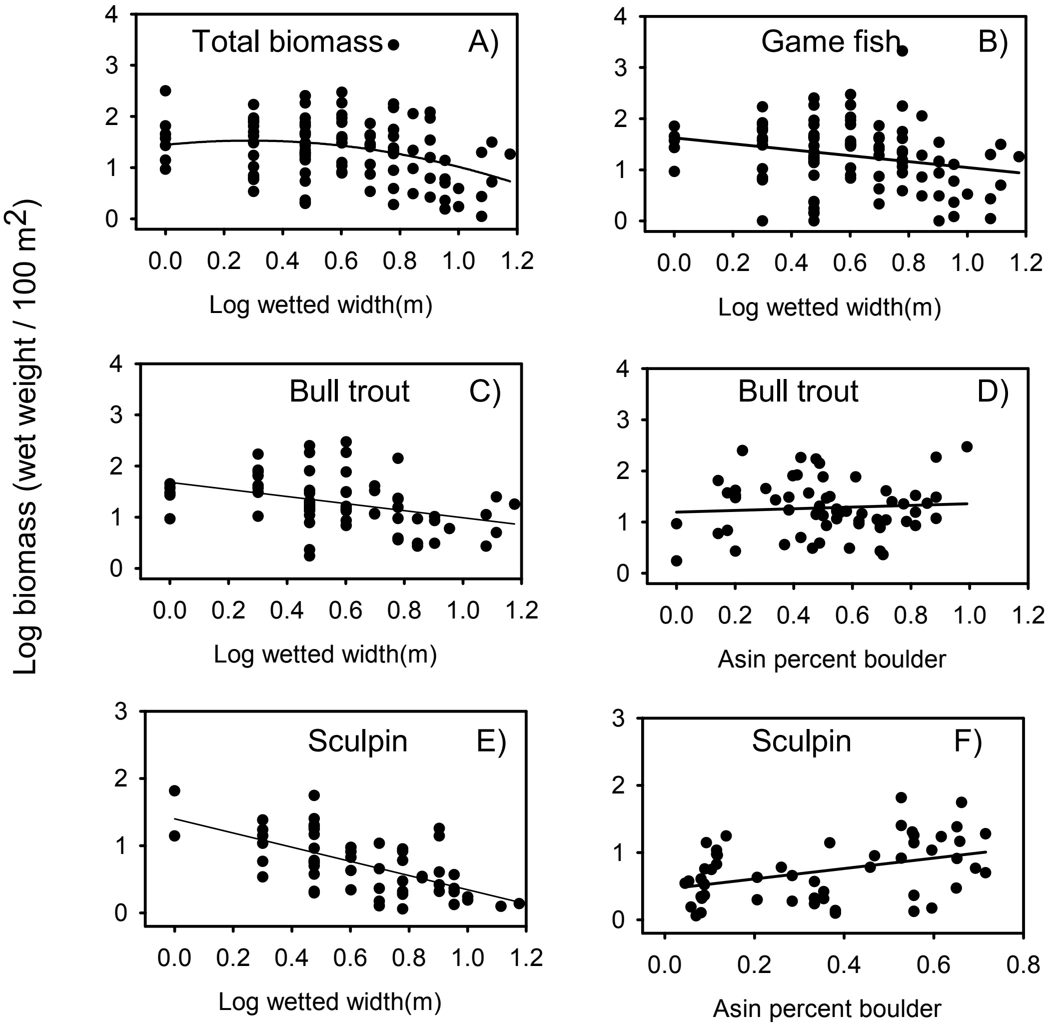

Figure 17. Regression models describing relations between total and species-specific fish biomass and watershed attributes in the Kakwa River Basin, Alberta. Data transformation abbreviations: asin = arc-sin squareroot transformed, log = $\log_{10}(x+1)$. Equations include linear and non-linear regressions (i.e., power functions and polynomial equations). Regression equations are provided in Table 13. Wetted width = width of the stream water surface.

Table 13. Summary of regression models describing relations between total and species-specific fish density and biomass and watershed attributes in the Kakwa River Basin. Data transformation abbreviations: asin = arc-sin squareroot transformed, lg = $\log_{10}$, lg1 = $\log_{10}(x+1)$. Equations include linear and non-linear regressions (i.e., power functions and polynomial equations). Regression equation abbreviations: F = F statistic, d.f. = model degrees of freedom, r^2= adjusted r^2. Regression model forms: linear regression model: Y = a+bx, multiple linear regression: Y=a + bx +cx, non-linear power function: Y = ax^b , - = figure not shown. † = No significant model.

	Model	F	d.f	r^2	P	Figure
Density						
Total	Multiple linear regression:					
	Y = 0 0.48 – 0.05 WETTED + 0.51 asin PERCENT WATERSHED DISTURBED	24.4	2,105	0.30	<0.0001	15A, 15B
	Non linear quadratic:					
	Y = 0.75 – 0.11 asin PERCENT WATERSHED DISTURBED + 1.99 asin PERCENT WATERSHED DISTURBED^2	14.3	2, 105	0.14	<0.001	15C
Game fish	Simple linear regression:					
	Y = 0.41 – 0.37 log WETTED WIDTH	20.2	1, 106	0.15	<0.0001	15D
	Non linear quadratic model:					
	Y = 0.45 – 050 log WETTED WIDTH + 0.11 log WETTED WIDTH^2	10.14	2, 105	0.15	<0.0001	15E
Arctic grayling	Multiple linear regression:					
	Y = 1.47 – 1.27 lg WETTED WIDTH – 1.96 as REACH SLOPE	10.0	2,30	0.40	0.0005	15F, G
	Non linear power function:					
	Y = 0.044 * log wetted width$^{-1.73}$	34.3	1,31	0.51	0.0015	15H
Bull trout	Y = -3.32 +1.18 lg ELEVATION – 0.27 lg WETTED WIDTH	9.63	2,55	0.30	<0.0001	15I, J
	Non linear power function:					
	Y = 0.07 x lg WETTED WIDTH$^{-1.44}$	20.8	1,55	0.26	<0.0001	15K
	Non-linear quadratics					
	Y = 20.54 – 14.85 lg ELEVATION + 2.67 ELEVATION^2	8.2	2,55	0.19	0.0007	-
Mountain whitefish	Y = 0.05 x lg WETTED WIDTH$^{-1.44}$	6.4	1,22	0.19	0.02	15L
Rainbow trout	†					
Sculpin	Multiple linear regression;					
	Y = 0.70 – 0.73 lg WETTED WIDTH - + 0.53 asin PERCENT WATERSHED DISTURBED	11.6	2,50	0.48	<0.0001	15M, N

Table 15 – continued	Model	F	d.f	r^2	P	Figure
Biomass						
Total fish density	Non-linear quadratic: $Y = 1.44 + 0.58$ lg WETTED WIDTH - 1.01 lg WETTED WIDTH^2	5.4	2,105	0.08	0.0059	16A
Game fish	Linear multiple regression: $Y = 1.55 –$ lg WETTED WIDTH + 0.53 asin PERCENT BOULDER	6.15	2,91	0.12	0.0031	16B, C
Arctic grayling	†					
Bull trout	$Y = 1.48 – 0.91$ lg WETTED WIDTH + 0.64 asin PERCENT BOULDER	7.71	2,59	0.18	0.0011	16 D, E
Mountain whitefish	†					
Rainbow trout	†					
Sculpin	$Y = 1.14 – 0.98$ lg WETTED WIDTH + 0.63 asin PERCENT WATERSHED DISTURBED	25.15	2,51	0.50	<0.0001	16 F, G

4.7.2 Simonette watershed

Relations between fish density and watershed attributes

Total fish density and density of the predominant species were primarily related to stream width, elevation and reach slope.

Regression models showed that total fish density and density of the predominant species in the Simonette watershed were primarily related to variables describing stream size (i.e., wetted width, bankfull width, mean water depth), elevation and size composition of material on the river bottom (Figure 18). Fish density was generally highest in high elevation streams or small streams (i.e., low wetted widths) and decreased with increasing elevation and stream size (Figure 18). In general, regression models explained relatively little of the overall variance in total density and density of the most abundant species (Overall range in r^2= 0.11 to 47%) and non-linear models did not typically explain appreciably more variance than linear models (Table 14).

Total fish density and density of the predominant fish species were typically weakly or unrelated to watershed disturbances including harvest blocks, road networks, pipe lines and seismic lines.

The notable exceptions include the positive relations between: i) total density and stream crossing density, ii) density of dace stream crossing density.

With the two exceptions, our analyses also showed that watershed disturbance, stream crossings and their underlying attributes (e.g., percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines) typically did not explain significant amounts (P>0.10) of variance in fish density (Table 14). The notable exceptions include the positive relations between: i) total density and stream crossing density, ii) density of dace and stream crossing density.

Total fish biomass and biomass of the predominant species in the Simonette watershed were primarily related to variables reflective of steam size, elevation and the relative abundance of small gravels within the river substratum.

Relations between fish biomass and watershed attributes

Total fish biomass and biomass of the predominant species in the Simonette watershed were primarily related to variables reflective of steam size (e.g., watershed area, mean water depth, mean stream bankfull width), elevation and the relative abundance of small gravels within the river substratum (Figure 19, Table 14). Fish biomass typically decreased with indicators of stream size, elevation and the abundance of small gravel in the river bed (Figure 19). In general regression models explained relatively little to only moderate amounts of the overall variance in total density and density of the most abundant species (Overall range in R^2= 0.05 to 67%) and with the exception of the non-linear relationship between biomass of dace and log bankfull width, non-linear models did not typically explain appreciably more variance than linear models (Table 14).

With some exceptions, fish biomasses were typically unrelated to watershed disturbance variables.

However, our analyses did reveal statistically significant relationships between: 1) total fish biomass and percent watershed disturbance, ii) biomass of sculpin and seismic density and iii) biomass of shiners and stream crossing density suggesting that current disturbance levels may increase biomass of some fish species.

With three exceptions, our analyses showed that fish biomasses were typically not related (P > 0.05) to watershed disturbance variables expressed as the total watershed disturbance, total stream crossings or their underlying attributes (e.g., i] percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, ii] stream crossings by roads, seismic lines, power lines and pipe lines). However, our analyses did reveal statistically significant relationships between: 1) total fish biomass and percent watershed disturbance, ii) biomass of sculpin and seismic density and iii) biomass of shiners and stream crossing density (Figure 19C, N, R). For each case, however, fish biomasses were positively related with percent watershed disturbance (total fish biomass), seismic density (sculpin), and density of stream crossings (shiner) indicating that industrial disturbances may increase biomass of selected species, at least within the observed ranges of watershed disturbances in the Simonette watershed.

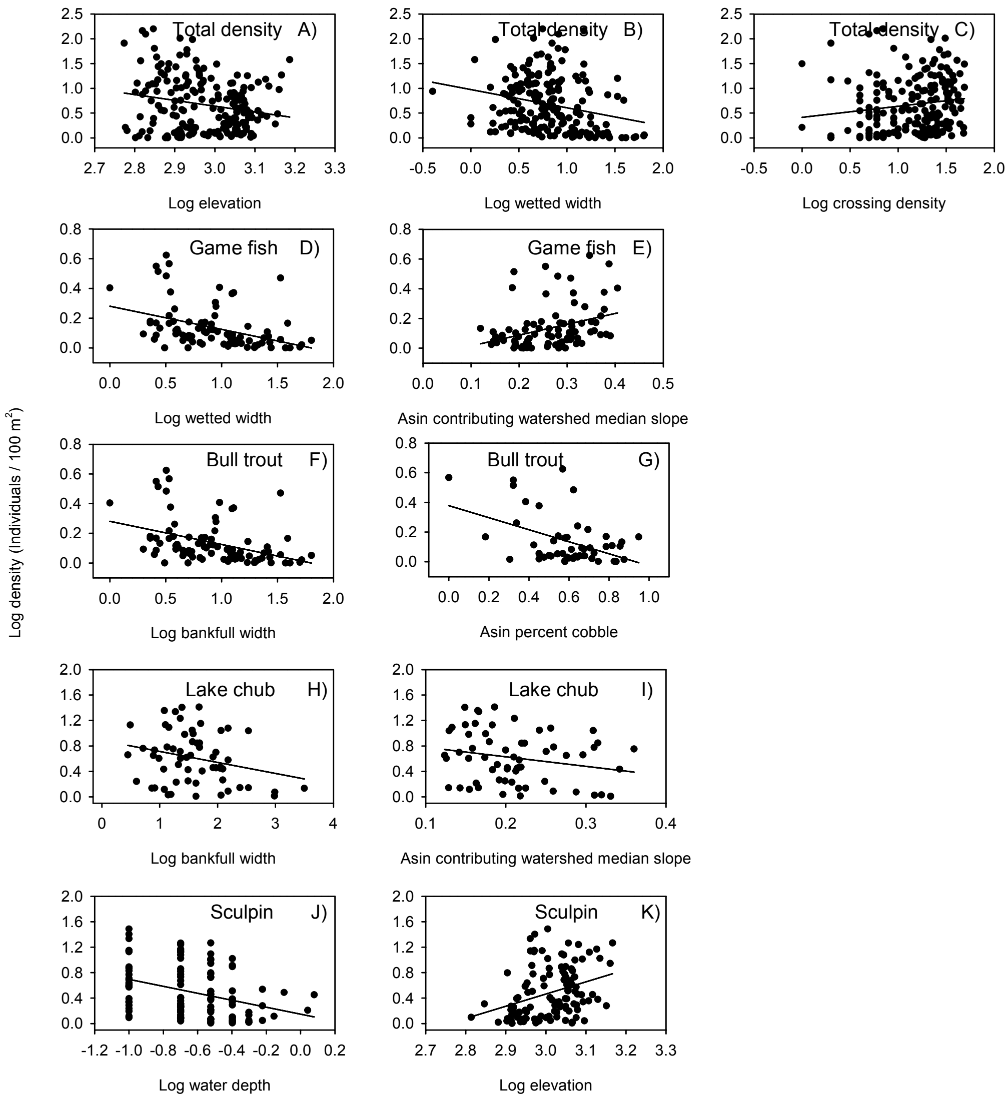

Figure 18. Linear regression models describing relations between total and species-specific fish density and watershed attributes in the Simonette River Basin, Alberta. Data transformation abbreviations: asin = arc-sin squareroot transformed, log = $\log_{10}$ (x+1). Regression equations are provided in Table 14. Wetted width = width of the stream water surface.

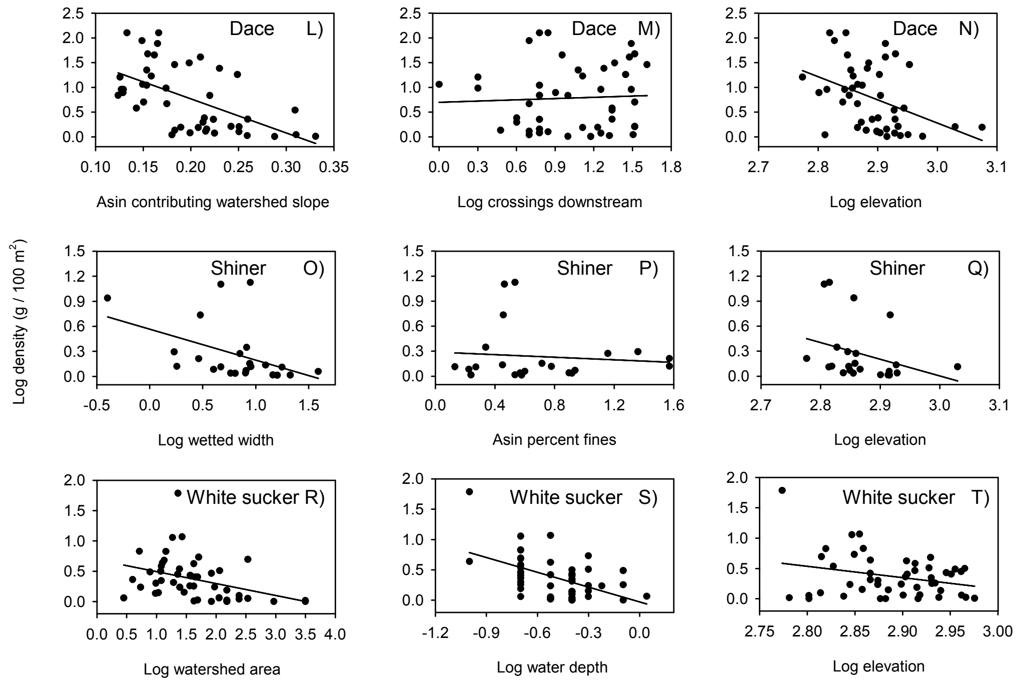

Figure 18. - continued

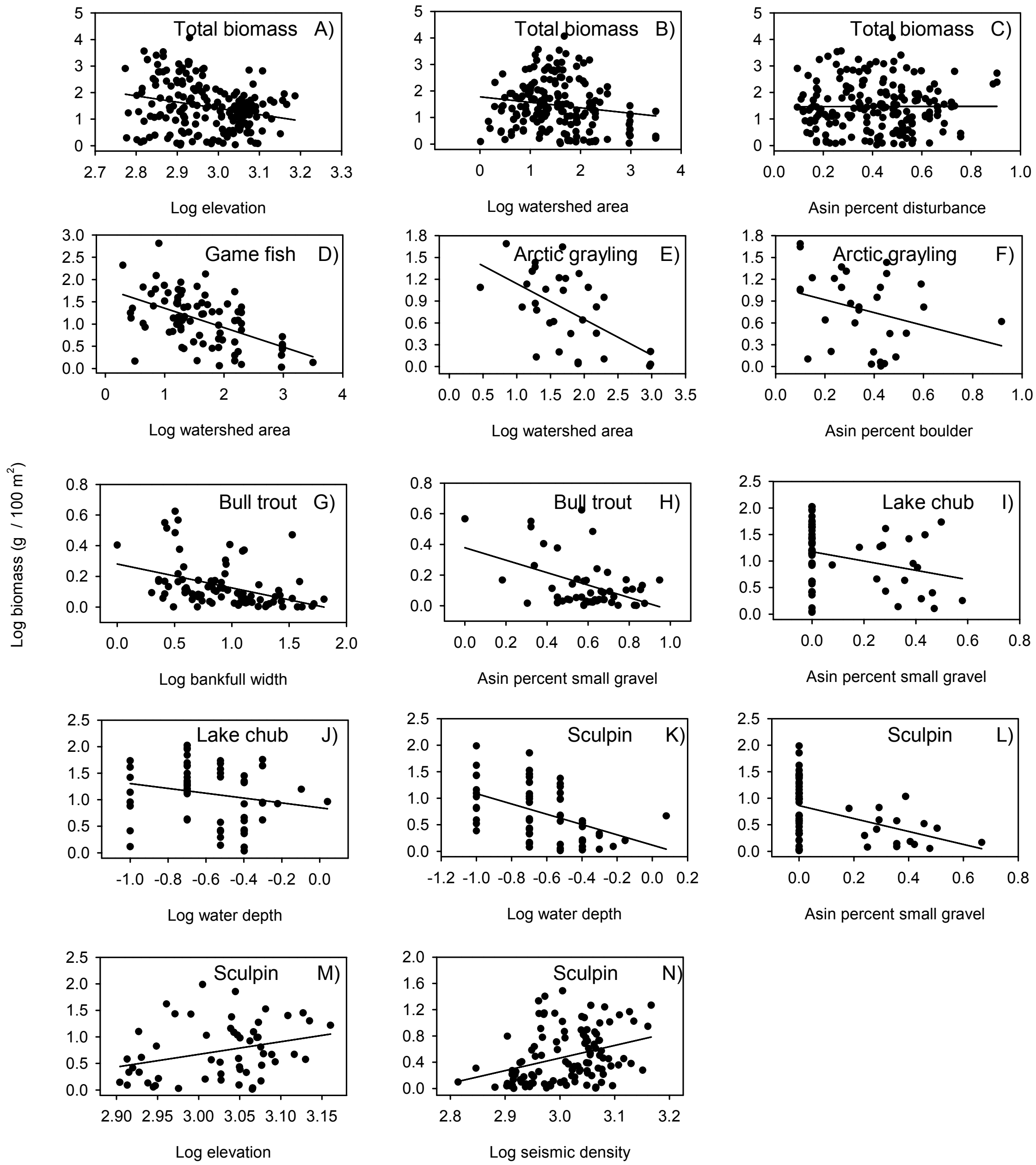

Figure 19. Regression models describing relations between total and species-specific fish biomass and watershed attributes in the Simonette River Basin, Alberta. Data transformation abbreviations: asin = arc-sin squareroot transformed, lg = $\log_{10}$ (x+1). Equations include linear and non-linear regressions (i.e., power functions and polynomial equations). Regression equations are provided in Table 14. Wetted width = width of the stream water surface.

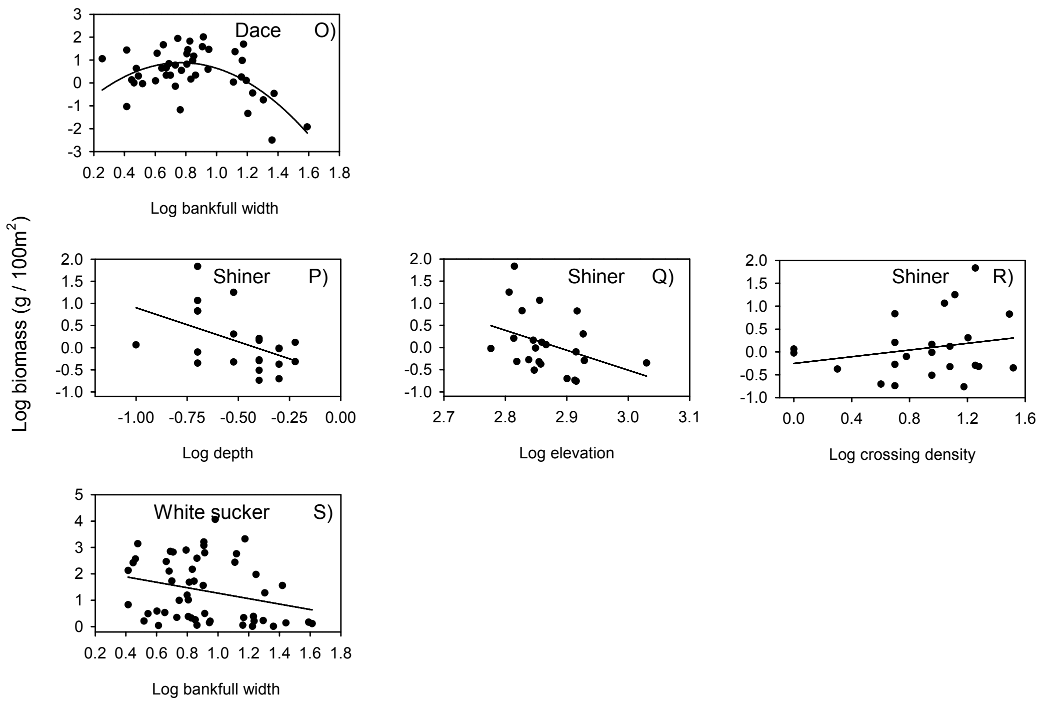

Figure 19. - continued

Table 14. Summary of linear and non-linear regression models between total and species-specific fish density and biomass and watershed attributes in the Simonette River Basin. Transformations: log x = $\log_{10}$, log x+1 = $\log_{10}(x+1)$, asin = arc-sin squareroot. Equations include linear and non-linear regressions (i.e., power functions and polynomial equations). CROSSING DENSITY = number of crossings within 40 km downstream of the site. BRIDGE = number of bridges within 40 km downstream of site. † = statistically non-significant model. Asin contributing watershed slope = asin median contributing watershed slope.

	Model	F	d.f	r^2	P	Figure
Density						
Total density	Y = 6.00 – 1.83 lg ELEVATION – 0.35 lg WETTED WIDTH + 0.36 lg CROSSING DENSITY	12.52	3, 185	0.15	<0.0001	17A, B, C
Game fish	Y = 0.13 - 0.15 lg WETTED WIDTH + asin CONTRIBUTING WATERSHED SLOPE	13.59	2, 80	0.21	<0.0001	17 D,E
Arctic grayling	†					
Bull trout	Y = 0.41 – 0.20 lg BANKFULL WIDTH – 0.28 asin PERCENT COBBLE	15.7	2, 47	0.37	<0.0001	17F, G
Lake chub	Y = 1.04 – 0.17 lg WATERSHED AREA - 3.15 CONTRIBUTING WATERSHED AREA SLOPE	3.62	2,58	0.11	0.03	17H, I
Sculpin	Y = -7.55 – 0.86 MEAN WATER DEPTH + 2.49 lg ELEVATION	16.70	2,52	0.39	<0.0001	17J, K
Dace	Y =10.0 – 6.11 asin CONTRIBUTING WATERSHED SLOPE + 0.53 lg NUMBER OF CROSSINGS DOWNSTREAM – 2.96 lg ELEVATION	9.23	3, 41	0.36	<0.0001	17L, M, N
Shiner	Y = 8.90 – 0.58 WETTED WIDTH – 0.44 asin PERCENT FINES – 2.74 lg ELEVATION	4.24	3,19	0.46	0.019	17O, P, Q
White sucker	Y = 7.35 – 0.22 lg WATERSHED AREA – 0.55 lg WATER DEPTH – 2.38 lg ELEVATION	10.81	3,43	0.39	<0.0001	17R, S, T
Biomass						
Total biomass	Y = lg ELEVATION – lg WATERSHED AREA + PERCENT WATERSHED DISTURBANCE	10.20	3, 187	0.13	<0.001	18A, B, C
Game fish	Y = 1.79 – 0.43 lg WATERSHED AREA	16.05	1, 82	0.26	<0.001	18D

Arctic grayling	Y = 2.01 – 0.51 lg WATERSHED AREA – 0.99 asin PERCENT BOULDER	11.37	12, 28	0.41	0.0002	18C, D
Bull trout	Y = 1.87 – 0.68 lg BANKFULL WIDTH – 1.52 asin PERCENT SMALL GRAVEL	14.42	2, 48	0.34	<0.0001	18G, H
Lake chub	Y = 0.87 – asin PERCENT SMALL GRAVEL - lg WATER DEPTH	3.94	1, 56	0.09	0.025	18I, J
Sculpin	Y = -5.81 – 1.05 lg WATER DEPTH – 1.09 asin PERCENT SMALL GRAVEL + 1.88 ELEVATION + 0.27 lg SEISMIC DENSITY	14.48	4, 50	0.50	<0.0001	18K, L, M, N
Dace	Y = -1.80 + 6.98 Lg BANKFULL WIDTH - 4.54 lg BANKFULL WIDTH ^{2}	11.60	2, 42	0.33	0.0001	18O
Shiner	Y = 23.70 – 2.15 lg DEPTH – 8.88 lg ELEVATION + 0.90 log CROSSING DENSITY	15.24	3, 19	0.67	<0.0001	18 P, Q, R
White sucker	Y = 2.30 – 1.03 lg BANKFULL WIDTH	3.86	1, 50	0.05	0.055	18S

Focal Question 3:

Is fish community structure affected by watershed disturbances?

Hierarchical cluster analyses identified three relatively discrete fish assemblages in the Kakwa River Basin.

Fish assemblages differed primarily based on the abundance of bull trout, sculpin and mountain whitefish.

4.8 Focal Question 3: Is Fish Community Structure Affected by Watershed Disturbances?

4.8.1 Kakwa watershed

<u>Identifying fish assemblages in reference (least-impacted) sites</u>

Hierarchical cluster analyses of 62 reference sites (i.e., disturbance attributes: total percent disturbance < 10%, range in percent disturbance = 0.08 to 9.9%, standard deviation = 2.1%) using percent composition data (based on fish densities), identified three discrete fish assemblages in the Kakwa River Basin (Figure 20). Assemblage 1 consisted primarily of bull trout (ca. 95% of all individuals collected) with Arctic grayling (ca 3%), mountain whitefish and rainbow trout accounting for < 5% of all the entire assemblage (Figure 21). Sculpin were absent from assemblage 1. In contrast, sculpin comprised about 81% of all individuals in assemblage 2, with bull trout (13%), mountain whitefish (2.3%), rainbow trout (1.6%) and Arctic grayling (1%) accounting for about 19% of all other fish collected. While assemblages 1 and 2 were largely dominated by a single fish species, assemblage 3 was numerically dominated by mountain whitefish (ca 51%) and to a lesser extent rainbow trout (18.3%), bull trout (13.3%), Arctic grayling (11.1%) and sculpin (7.6%) (Figure 21).

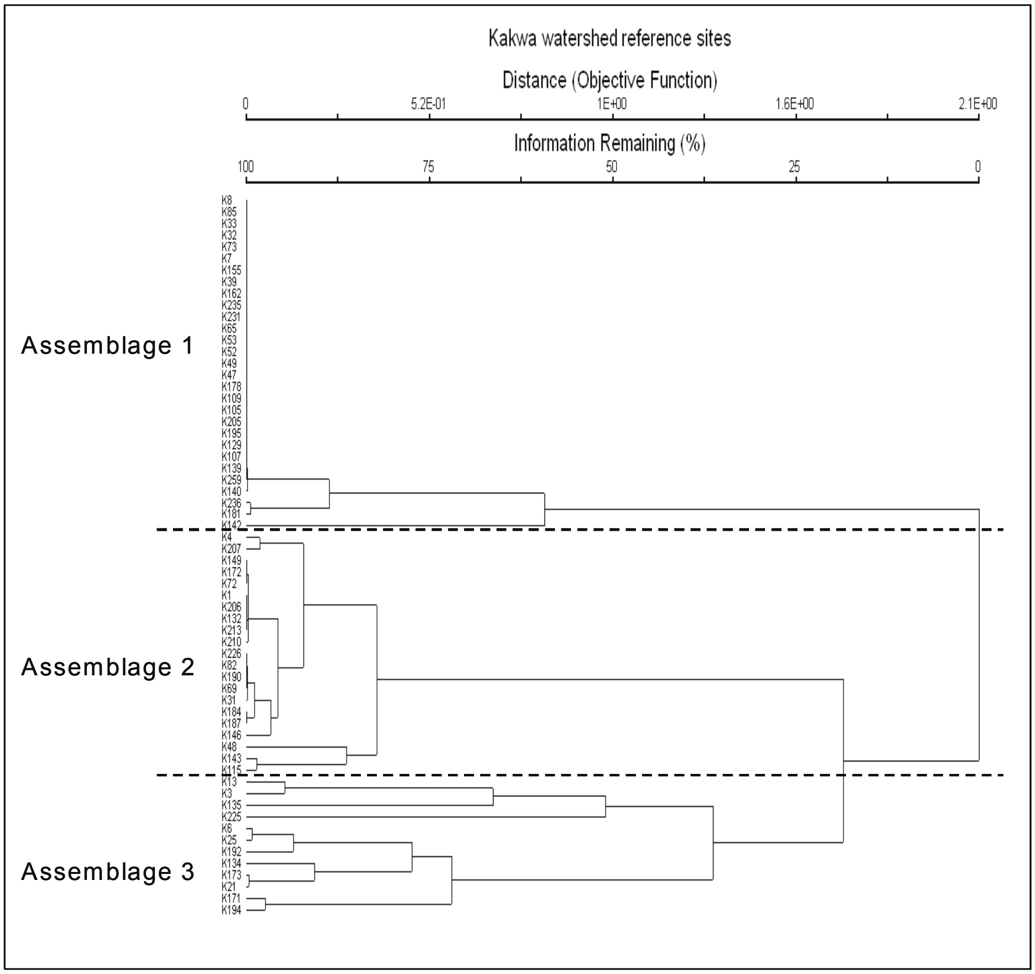

Figure 20. Agglomerative hierarchical cluster analyses of fish communities from 62 reference sites in the Kakwa River Basin. Analyses identified three fish assemblages.

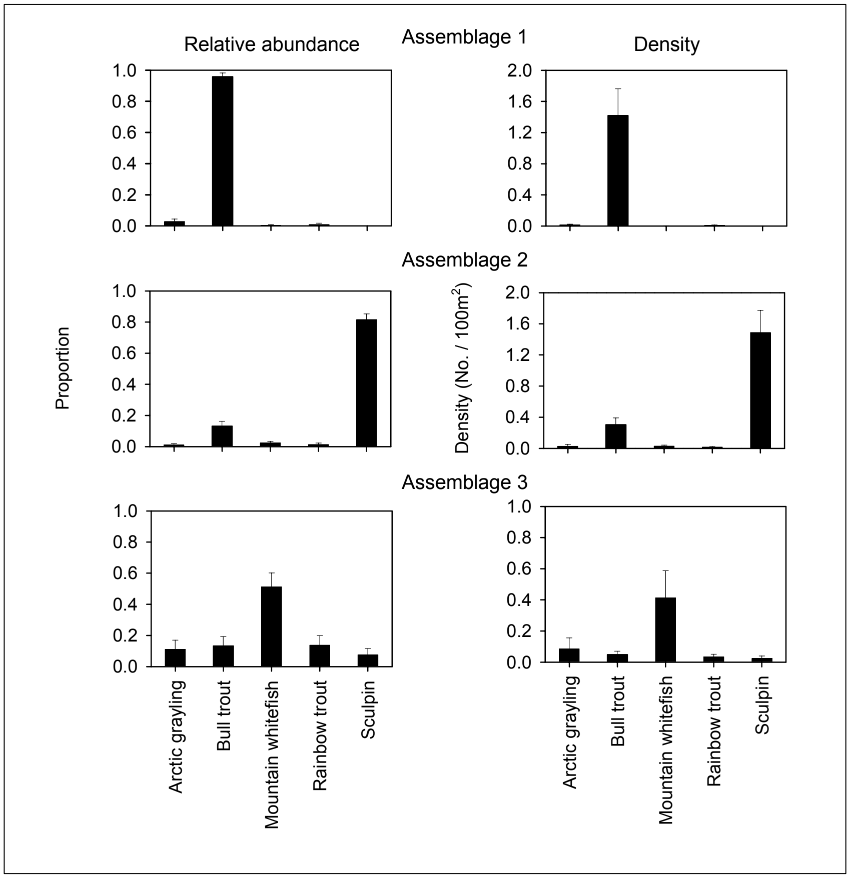

Figure 21. Mean (±1SE) percent composition and density of the five numerically dominant fish species and species groups comprising the three fish assemblages in the Kakwa River Basin.

Assemblage 1 was dominated by bull trout whereas sculpin and mountain whitefish were dominant in fish assemblage 2 and 3, respectively.

Analysis of variance tests indicated that Arctic grayling were proportionately more abundant in assemblage 3 than that in assemblage 1 (Bonferroni adjusted, orthogonal contrasts, Figure 22 A) whereas bull trout were proportionately more numerous in assemblage 1 than in assemblages 2 and 3 (Figure 22 B, H). Mountain whitefish and rainbow trout comprised a significantly higher proportion of the fish assemblage 3 compared to assemblages 1 and 2 (Bonferroni adjusted, orthogonal contrasts Figure 22, C, D).

Total fish density in assemblages 1 and 2 was greater then that in assemblage 3.

In addition to differences in relative abundance, total fish density and density of individual species and species groups also differed among assemblages (Figure 22 F-K). Total fish density was highest in assemblages 1 and 2 compared with assemblage 3 and resulted from: i) high densities of bull trout in assemblage 1 compared with that in assemblage 2 and 3 and ii) high densities of sculpin in assemblage 2 compared with that in assemblage 3 (Figure 22 H, K). Species richness was relatively low but was significantly higher in assemblages 2 and 3 compared with that in assemblage 1 (Bonferroni adjusted orthogonal contrasts, Figure 22).

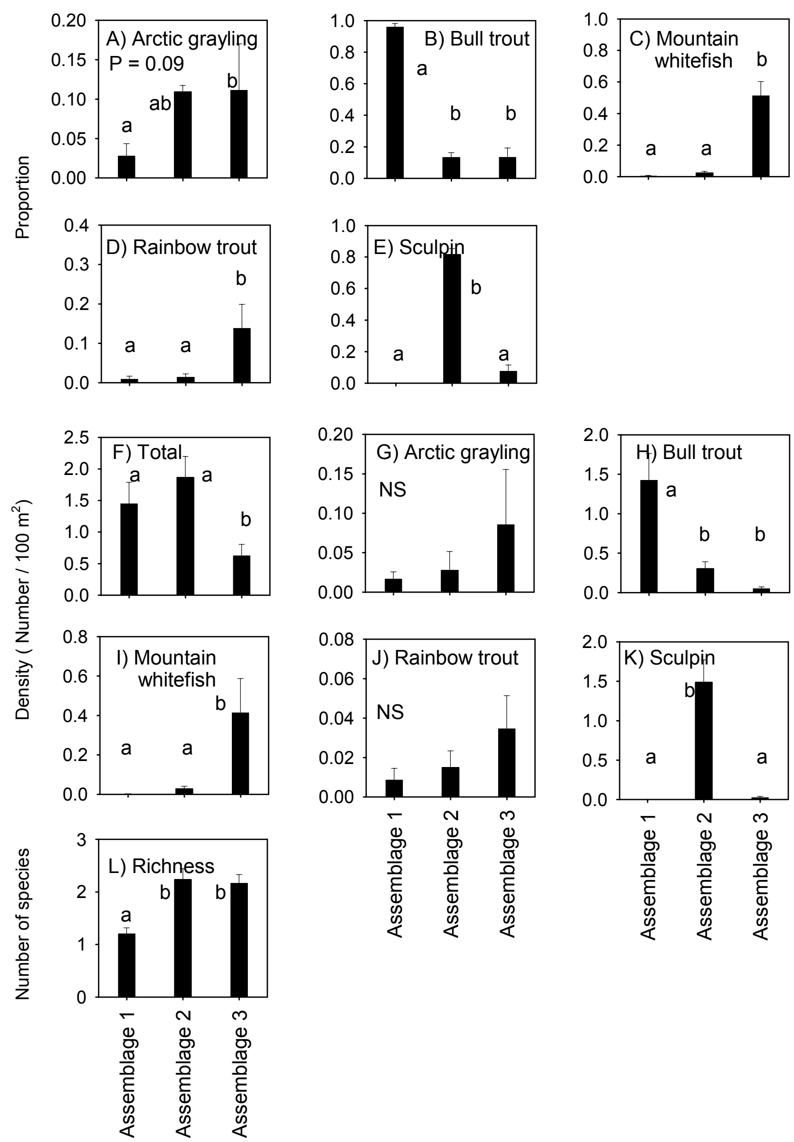

Figure 22. Mean (±1SE) total density, density of the four numerically dominant fish species and species richness comprising the three fish assemblages in the Kakwa River Basin. Histograms sharing the same letter are not significantly different (Bonferroni adjusted orthogonal contrasts). NS = not statistically significant. Analysis of variance tests were completed using log (density and richness) or arc-sin squareroot transformed (proportion) data.

Discriminating among fish assemblages in reference (least-impacted) sites

Watershed variables of stream wetted width, stream reach slope, site elevation and percent small gravels were moderately successful discriminators among the three fish assemblages.

Based on relative abundance data, the forward selection discriminant analysis identified log stream wetted width, asin stream reach slope, log site elevation and arcsin percent small gravel as significant discriminators among the three fish assemblages identified from reference (i.e., least-impacted) sites. The quadratic discriminant function model had an overall classification success of 71.0% (i.e., 44 of the 62 sites were classified correctly) and correctly classified 18 of the 29 sites into assemblage 1 (i.e., classification success = 62.1%), 18 of 21 sites into assemblage 2 (classification success = 85.7%) and eight of the 12 sites into assemblage 3 (i.e., classification success = 66.7%).

Stream reaches supporting assemblage 1 were relatively narrow, with high reach slope and stream beds with low amounts of small gravel.

Reaches supporting fish assemblage 2 were typically located at generally lower elevations, with broad stream channels that contained increased proportions of small gravels compared to reaches supporting assemblage 1.

Stream reaches that supported assemblage 3 were typically located at low elevations, were relatively broad with moderate reach slope with low amounts of small gravel within the river bed.

Stream reaches supporting fish assemblage 1 were typically located at high elevations (1313 m.a.s.l.), were relatively narrow (i.e., low stream wetted width [3.7 m]), with high reach slope (0.035) and stream beds with low amounts of small gravel (0.07 %) (Table 15). In contrast, reaches supporting fish assemblage 2 were typically located at generally lower elevations (1301 m.a.s.l.), with broader stream channels (i.e., mean stream wetted widths = 4.5 m) and contained increased proportions of small gravels (0.12%) compared to reaches supporting assemblage 1. Stream reaches supporting fish assemblage 3 were characteristically located at low elevations (1230 m), were relatively broad (i.e., stream wetted width = 5.6 m), with moderate reach slope (0.03) and stream beds with low amounts of small gravel (0.03 %) (Table 15).

Table 15. Description of fish assemblages and related environmental variables from 62 reference sites in the Kakwa River Basin. N = number of sites within each fish assemblage. Data are means ± 1SE. Burbot, dace and white sucker were absent from all reference sites.

Descriptor	**Assemblage**		
	1 (N = 29)	2 (N = 21)	3 (N = 12)
Fish community structure			
Density			
Total density	1.45±0.34	1.87±0.33	0.62±0.18
Arctic grayling	0.02±0.01	0.02±0.02	0.09±0.07
Bull trout	1.42±0.34	0.30±0.09	0.05±0.02
Longnose sucker	0	0.002±0.002	0
Mountain whitefish	0.002±0.002	0.03±0.01	0.41±0.17
Rainbow trout	0.009±0.006	0.01±0.008	0.03±0.02
Sculpin	0	1.49±0.28	0.02±0.02
Trout-perch	0	0	0.01±0.01
Percent composition			
Arctic grayling	0.03±0.02	0.01±0.008	0.11±0.06
Bull trout	0.96±0.02	0.13±0.03	0.13±0.06
Mountain whitefish	0.004±0.004	0.02±0.009	0.51±0.09
Rainbow trout	0.009±0.008	0.02±0.009	0.14±0.06
Sculpin	0	0.82±0.04	0.08±0.04
Species dominance			
Numerically dominant species	Bull trout	Sculpin	Mountain whitefish
Abundance rankings of the five numerically species (high to low)	Bull trout > rainbow trout > Arctic grayling > Mountain whitefish	Sculpin > bull trout > Mountain whitefish >Arctic grayling > rainbow trout > longnose sucker	Mountain whitefish > Arctic grayling > bull trout > rainbow trout > sculpin > trout-perch
Species richness			
Total number of species in assemblage	4	6	6
Mean site species richness	1.2	2.2	2.2
Environmental variables			
Wetted width (m)	3.7±0.5	4.5±0.5	5.58±0.8
Slope (^{o})	0.035±0.003	0.024±0.002	0.03±0.01
Elevation (m.a.s.l.)	1313±25	1301±30	1230±45
Percent small gravel	0.07±0.01	0.12±0.02	0.03±0.02
Percent large gravel	0.26±0.03	0.28±0.03	0.24±0.03
Percent boulder	0.27±0.04	0.27±0.04	0.28±0.06
Order	3.6±0.3	3.3±0.1	3.6±0.2
Bankfull width (m)	4.7±1.2	8.14±1.2	15.0±4.1
Watershed area (km^2)	34.4±8.8	27.1±2.8	52.3±16.0

Evaluating the cumulative effects of watershed disturbances on fish community structure in the Kakwa watershed using a reference-condition approach

We quantified the cumulative effects of watershed disturbances on fish community structure using a reference condition approach.

We quantified the cumulative effects of watershed disturbances on fish community structure a reference condition approach. This approach evaluates the extent to which potentially impacted sites contain fish assemblages predicted by relationships between fish community structure and habitat variables derived from reference (i.e., least-impacted) sites. If the empirical model (discriminant function model) fish assemblage-habitat model derived from the reference sites also explains variation in the potentially impacted sites then it is assumed that impacts are not detectable.

We evaluated larger patterns in fish communities by completing cluster analyses of all 108 sites comprising reference sites from (i.e., least-impacted sites, N = 62) and potentially impacted sites (46 sites) where the percent watershed disturbance ranged from 10 to 43% (Figure 23). In addition to identifying the three fish assemblages from the least-impacted sites (assemblages 1, 2 & 3, Table 15), clustering also identified two small additional clusters of sites (Test assemblages 1 and 2) (Figure 23). Test assemblage 1 consisted of 7 sites and was numerically dominated by Arctic grayling (86.2% of all individuals encountered) with bull trout (9.6%) and to a lesser extent mountain whitefish (2.6%) and rainbow trout (1.5%). In contrast Test assemblage 2 comprising 4 sites) was numerically dominated by rainbow trout (97.2%) and bull trout (2.7%). These two cluster of sites consisted predominantly (7 of 8 sites, Test - Assemblage 1) or exclusively (4 of 4, Test - Assemblage 2) of potentially impacted sites (i.e., range of percent watershed disturbances = 12.47 to 36.1%). On average the mean level of watershed disturbances in Test assemblage 1 (Mean±1STD = 34.29±0.45%, N = 4) was about 50% higher than that in Test assemblage 2 (21.62±12.34%, N = 8). These data suggest that fish communities from some, but not all, of the potentially impacted sites differ from those present at the least-impacted, reference sites.

These data suggest that fish communities from some, but not all, of the potentially impacted sites differ from those present at the least-impacted, reference sites.

While increased levels of watershed disturbance could potentially produce fish assemblages dominated by rainbow trout (i.e., Test assemblage 2), it is more likely that this assemblage type results from the emigration of rainbow trout stocked into the adjacent Musreau Lake. Watersheds located adjacent to the Musreau Lake have also been subject to relatively high rates of forest harvesting largely because they are located at low elevation in the Kakwa watershed and are adjacent to a main haul road. However, why the reference sites failed to identify fish assemblage dominated by Arctic grayling is less well understood and may be a response to industrial activities, especially given our result from the logistic regression analyses showing a statistically significant and negative relation between bull trout occurrence and percent watershed disturbance.

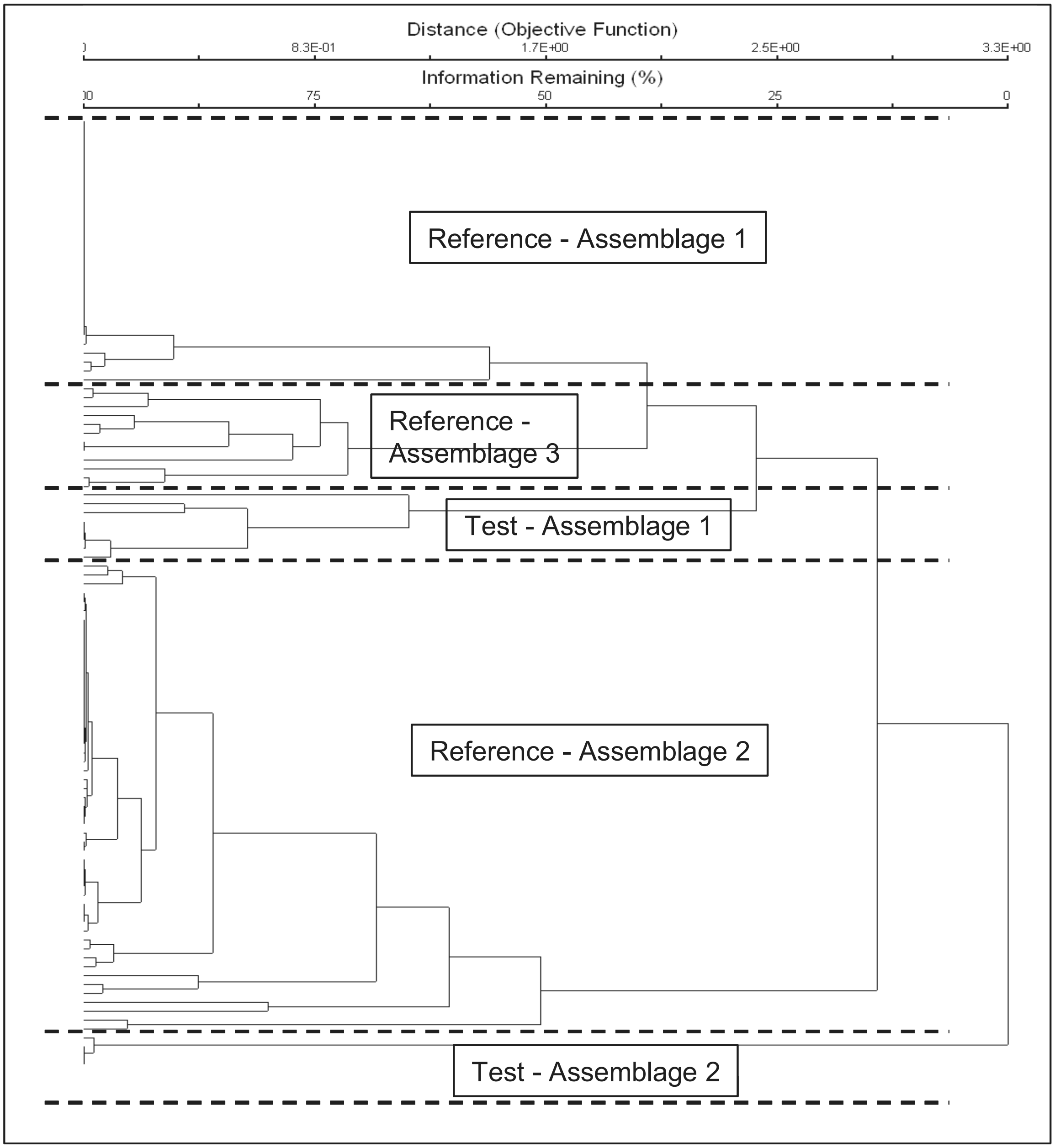

Figure 23. Agglomerative hierarchical cluster analyses of fish communities from 62 reference sites and 46 potentially impacted sites in the Kakwa River Basin. Analyses identified the three fish assemblage types dominated by reference (i.e., least impacted sites) sites (reference assemblages 1, 2 & 3) and two additional fish assemblages predominantly, or exclusively, comprised of sites from potentially impacted watersheds (test assemblage 1 & 2)
.

We used discriminant function analyses to quantify how well fish communities at all of the 46 potentially impacted sites (i.e., test sites where watershed disturbance ranged from 10 to 43%) were predictable based on the four watershed variables (i.e., stream wetted width, reach slope, elevation and percent small gravel) identified as predictors of fish communities from the 62 reference sites (i.e., least-impacted). If the discriminant function model derived from the reference sites successfully predicts fish community at the 46 tests sites, then we assume that the cumulative effects of watershed disturbance do not have detectable effects of fish community structure. Conversely, if the discriminate function model is a poor predictor of fish communities at the test sites, then we can assume that these sites differ from references sites and that such differences may, to some extent, reflect differences in watershed disturbance.

The outcome of the analysis of all 46 potentially impacted sites (See analysis 1) is in part predetermined because 12 of the 46 sites failed because they did not cluster within the three reference assemblages (i.e., assemblages 1, 2 & 3). However, the adjacency of four of the test sites, dominated by rainbow trout, to Musreau Lake suggests that these communities result from anthropogenic activities of fish stocking rather than industrial activities. Inclusion of these data in the discriminant function analyses would bias the results and for the purposes of this study were not included in additional analyses other than for comparative purposes to quantify the effect of including these data on classification success.

The discriminant function derived from the reference sites was a relatively poor predictor of fish assemblage membership of the potentially impacted test sites.

The discriminant function mode derived from the least-impacted sites had an overall classified success of only 50%.

These data suggest that the cumulative effects of human activities has resulted in impairment of stream fish communities in the Kakwa River Basin.

As anticipated, classification success of the discriminant function analysis improved slightly when the four sites dominated by rainbow trout were excluded from analyses (classification success of all test sites = 45.7%) compared to analyses completed using these data (classification success of reduced tests sites = 50.0%) (Figure 24A). Irrespective of this result, the discriminant function developed from the reference sites was a poor predictor of fish assemblage membership and correctly classified the 1 site from assemblage 1 (classification success = 100%), 18 of the 30 sites into assemblage 2 (classification success = 60%) and 2 of the 3 sites into assemblage 3 (i.e., classification success = 66.7%) (Figure 24A). When combined with the eight other sites that did not cluster into any of the three reference assemblages, the model had classification success of 50% (21 of the 42 sites). These data suggest that the discriminant function model developed from the reference sites is a poor predictor of fish assemblage structure in the potentially impacted sites and suggests that the cumulative effects of human activities has resulted in impairment of stream fish communities.

We completed a final analysis (Figure 24B) to determine whether the relative classification success of the discriminant function analysis is affected by the percent of watershed disturbance. For this analysis we ranked all test sites based on percent watershed and arbitrarily divided all sites into two groups with relatively equal number representing low (11 to 25%, mean = 17%, N = 16) and high (27 to 43%, mean = 32%, N = 18) percent watershed disturbance. This analysis results in a relative classification success because the analysis did not include the eight failed sites. Results from this analysis showed that classification success did not differ markedly when applied to sites defined as having low watershed disturbance compared to high watershed disturbance (Figure 24 B).

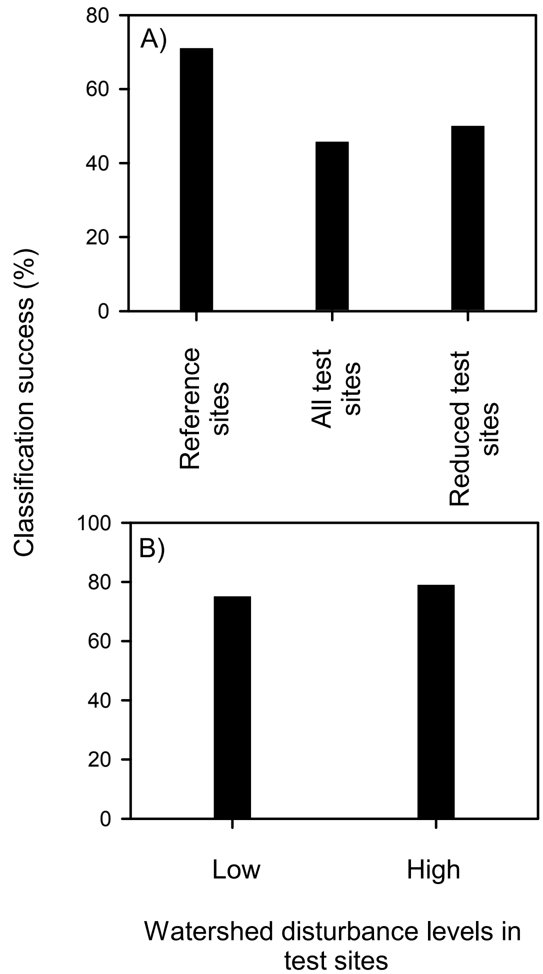

Figure 24. Comparisons of percent classification success of the discriminant function analysis from reference sites to test sites (A) and low and high disturbance levels within test sites (B) in the Kakwa River Basin. Classification success for the potentially impacted sites was calculated with and without the four sites dominated by rainbow trout (A) that likely emigrated as stocked rainbow-trout from Musreau Lake.

4.8.2 Simonette watershed

<u>Identifying fish assemblages in reference (least-impacted) sites</u>

Hierarchical cluster analyses of fish density identified five discrete fish assemblages in the Simonette River Basin.

Hierarchical cluster analyses of fish communities from the 106 reference (i.e., least-impacted) sites (i.e., disturbance attributes: total percent disturbance < 20%, range in percent disturbance = 0.1% to 19.48%, standard deviation = 5.85%) identified five relatively discrete fish assemblages and three smaller clusters of sites comprising fish communities from one to three sites (Figure 25). In total, fish communities from 98 of the 106 sites clustered into five relatively discrete fish assemblages.

Assemblage 1 consisted primarily of white sucker and to a lesser extent finescale dace, lake chub, trout-perch and redside shiner.

Assemblage 2 were dominated by lake chub and to a lesser extent white sucker, longnose sucker, sculpin and finescale dace.

Northern redbelly dace and to a lesser extent lake chub, pearl dace, Brook stickleback and longnose sucker dominated assemblage 3.

Assemblage 4 was dominated by mountain whitefish, pearl dace and Arctic grayling whereas sculpin were predominant in assemblage 5.

Assemblage 1 consisted primarily of white sucker (ca. 33% of all individuals collected) and to a lesser extent finescale dace (ca. 15%), lake chub (ca 13%), trout-perch and redside shiner (combined ca. 8%) whereas communities comprising assemblage 2 were dominated by lake chub (ca. 60%) and to a lesser extent white sucker (ca. 7%), longnose sucker (6%), sculpin (ca.6%) and finescale dace (Figure 26). Northern redbelly dace (ca. 69%) and to a lesser extent lake chub (ca. 9%), pearl dace (ca. 8%), brook stickleback (ca. 4%) and longnose sucker dominated assemblage 3. In contrast, fish communities comprising assemblages 4 and 5 were numerically dominated by mountain whitefish, pearl dace and Arctic grayling (assemblage 4) and sculpin (ca. 87%) (assemblage 5) (Figure 26).

Analysis of variance tests combined with Bonferroni-adjusted orthogonal contrast, showed significant differences in the relative abundance (i.e., proportion) and density of species and species groups among assemblages (Figure 27). For example, relative abundance, but not density, of Arctic grayling in assemblage 4 exceeded that in assemblage 2, 3 and 5 (Bonferroni adjusted orthogonal contrasts, Figure 27 A, J). Lake chub were also relatively more abundant in assemblage 2 compared with assemblages 1, 3, 4 & 5. Density of lake chub in assemblage 2 exceeded that in assemblage 4 and 5 whereas density of lake chub in assemblage 3 exceeded that in assemblage 5 (Figure 27 C, L). Relative and absolute abundance of mountain whitefish in assemblage 4 and sculpin in assemblage 5 exceeded that in all other assemblages (Figure 27 E, N and Figure 27 H, Q). Lastly, white sucker were proportionately more abundant in assemblage 1 compared to all other assemblage types whereas the density of white sucker in assemblage 1 only exceeded that in assemblages 4 and 5 (Figure 27 I, R).

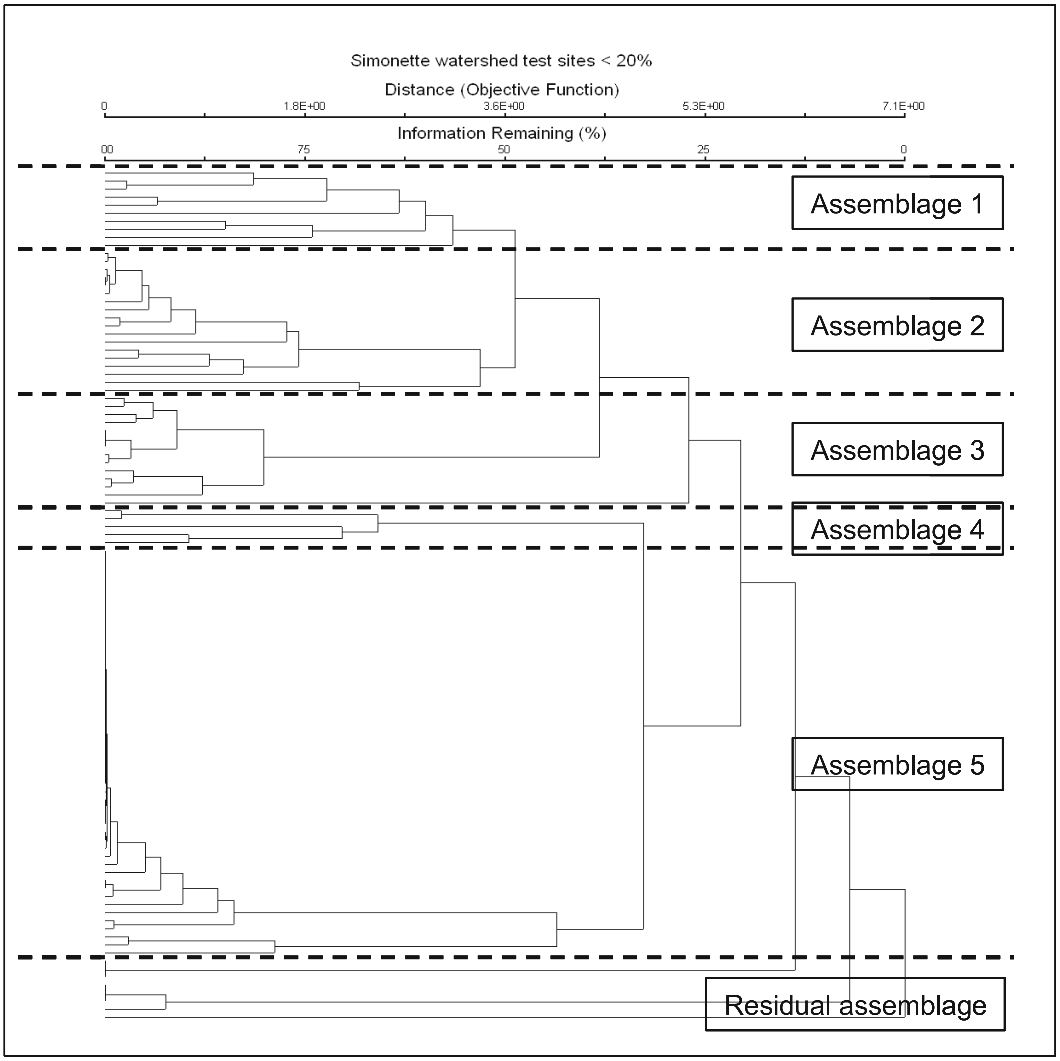

Figure 25. Agglomerative hierarchical cluster analyses of fish communities from 98 reference sites in the Simonette River Basin. Analyses identified five fish assemblages and a sixth small cluster of residuals sites.

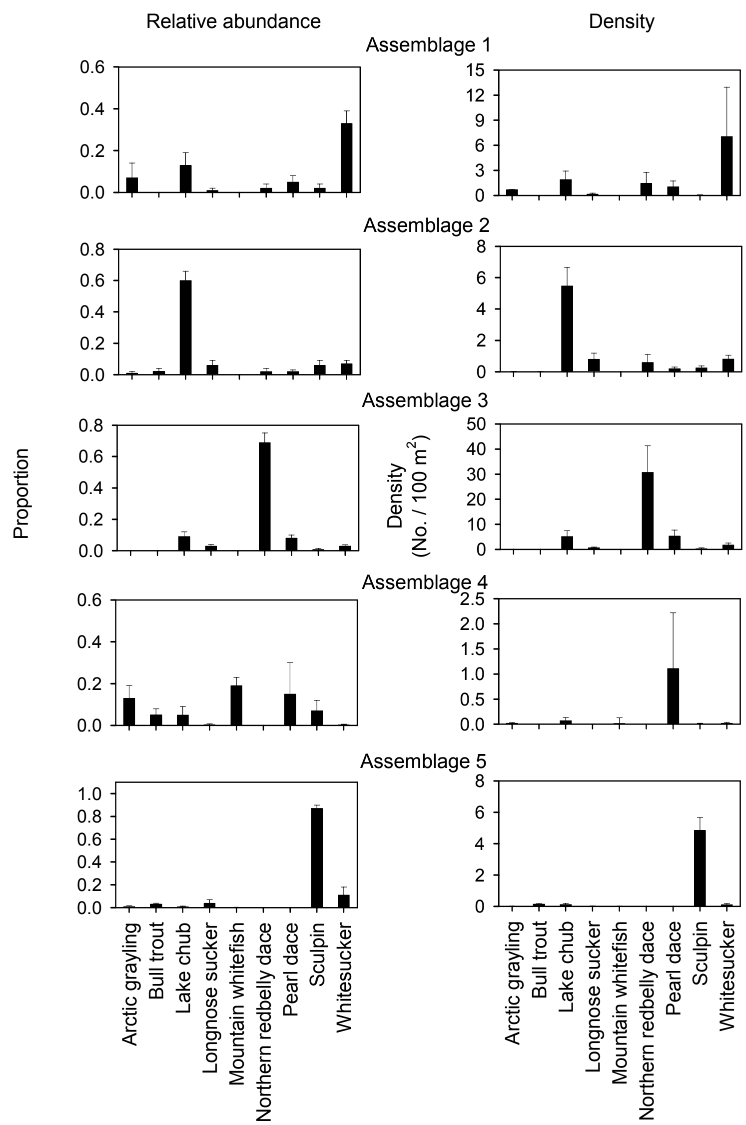

Figure 26. Mean (±1SE) percent composition and density of the nine numerically dominant fish species and species groups comprising the five fish assemblages in the Simonette River Basin.

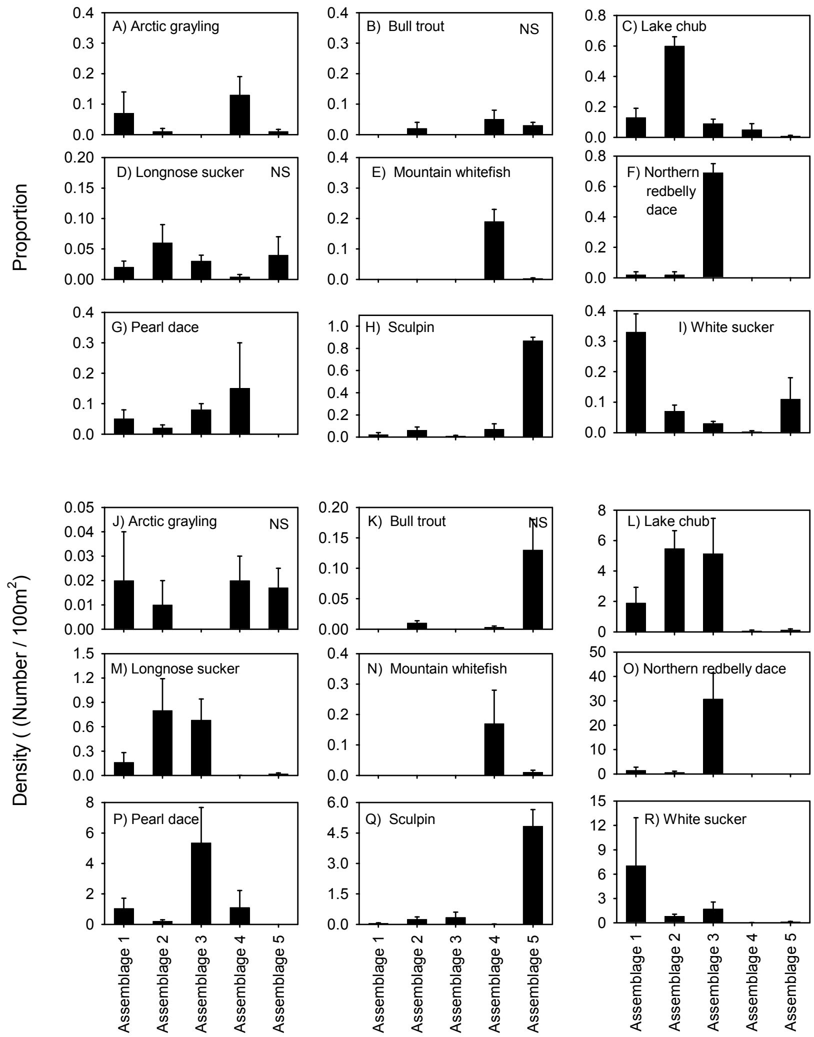

Figure 27. Mean (±1SE) total density, density of the nine numerically dominant fish species comprising the five fish assemblage types in the Simonette River Basin. Analysis of variance tests were completed using log (density and richness) or arc-sin squareroot transformed (proportion) data. Comparisons of means are described in text and based on Bonferroni adjusted orthogonal contrasts. NS = not statistically significant.

Watershed variables of stream wetted width, stream reach slope, site elevation and percent of fine materials in the substratum were moderately successful discriminators among the five fish assemblages.

Sites supporting assemblage 1 were moderately wide with low slope and were located at low elevations with riverbeds dominated by fine sediments.

Sites supporting assemblage 2 were relatively narrow with low slope and were located at moderate elevations with riverbeds dominated by fine sediments.

Sites supporting fish assemblage 3 were very narrow with high slope and were located at low elevations where riverbeds were dominated by fine sediments.

Stream reaches supporting assemblage 4 were wide with high slope and were located At high elevations with substrata dominated by large materials.

Sites supporting assemblage 5 were moderately small with high slope and were located at high elevations with moderate amounts of fine materials within the riverbed.

<u>Discriminating among fish assemblages from reference (least-impacted) sites</u>

Based on relative abundance data, the forward selection discriminant analysis identified log stream wetted width, log stream reach slope, log site elevation and the arcsin percent of fine materials in the substratum as significant discriminators among the five assemblages. The quadratic discriminant function model had an overall classification success of 71.1% (i.e., 69 of the 97 sites were classified correctly) and correctly classified 5 of the 10 sites into assemblage 1 (i.e., classification success = 50.0%), 15 of the 17 sites into assemblage 2 (classification success = 88.2%) and 10 of the 13 sites into assemblage 3 (i.e., classification success = 76.9%), 5 of the 6 sites into assemblage 4 (classification success = 83.3%) and 34 of the 51 sites into assemblage 5 (i.e., classification success = 66.7%).

Assemblage 1 was dominated primarily by white sucker and were located within moderately large (mean stream water width [wetted width] = 9.2 m), low elevation stream reaches (mean = 752 m.a.s.l.) characterized by low reach slope, substrata dominated by fine sediments and found in large watersheds (mean = 410 km^2) (Table 16). Fish communities comprising assemblage 2 were dominated by lake chub and were located in moderately small (mean wetted width = 5.4 m, mean watershed areas = 277 km^2), low gradient streams (mean slope = 0.007), at intermediate elevations of 825 m.a.s.l. Assemblage 3, dominated by northern redbelly dace were typically located in small, (mean wetted width = 3.0 m) low elevation (mean elevation = 740 m.a.s.l.) streams. In contrast, fish communities comprising assemblages 4 and 5 were numerically dominated by pearl dace (assemblage 4) and sculpin (assemblage 5) and were both typically located at high elevations (901 and 1044 m.a.s.l.) with moderate slope but differed in stream size and the percent of fine materials within the stream bottom. Streams comprising assemblage 4 were typically large (mean wetted width = 13.7 m) with a predominance of fine materials in the river bed compared with the small streams comprising assemblage 5 (mean wetted width = 5.9 m) where fine sediments accounted for only 9% of substratum materials (Table 16).

Table 16. Description of fish assemblages and related environmental variables from 98 reference sites in the Simonette River Basin. N = number of sites. Data are means ± 1SE.

Descriptor	**Assemblage number**				
	1 (N = 10)	2 (N = 18)	3 (N = 13)	4 (N=6)	5 (N=51)
Community structure					
Density					
Total density	13.7±8.1	9.90±2.15	46.1±15.5	1.45±1.18	5.25±0.85
Arctic grayling	0.02±0.02	0.01±0.01	0	0.02±0.01	0.017±0.008
Bull trout	0	0.01±0.004	0	0.003±0.002	0.13±0.05
Brook stickleback	0.10±0.08	0	0.77±042	0.	0
Burbot	0	0	0	0.014±0.014	0.02±0.006
Emerald shiner	0	0.26±0.02	0	0	0
Finescale dace	0.57±0.49	0.98±0.96	0.53±0.53	0	0
Flathead chub	0	0	0	0	<0.0001
Lake chub	1.9±1.03	5.47±1.18	5.14±2.32	0.07±0.06	0.12±0.08
Largescale sucker	0	0.001±0.001	0	0	0
Longnose dace	0	0.30±0.12	0.22±0.12	0.02±0.02	0
Longnose sucker	0.16±0.12	0.80±0.39	0.68±0.26	0.0005±0.0005	0.02±0.01
Mountain whitefish	0	0	0	0.17±0.11	0.01±0.007
Northern redbelly dace	1.44±1.29	0.60±0.49	30.7±10.6	0	0
Northern pike	0	0	0	0	0
Pearl dace	1.04±0.68	0.20±0.10	5.36±2.32	1.11±1.11	0
Redside shiner	1.27±1.22	0.15±0.01	0.13±0.10	0	0
Rainbow trout	0	0	0	0	0
Sculpin	0.05±0.03	0.25±0.12	0.34±0.26	0.01±0.01	4.84±0.82
Trout-perch	0.12±0.06	0.18±0.08	0.49±0.47	0	0
White sucker	7.04±5.90	0.81±0.25	1.72±0.84	0.02±0.02	0.11±0.07
Percent composition					
Arctic grayling	0.07±0.07	0.01±0.01	0	0.13±0.06	0.011±0.008
Bull trout	0	0.02±0.02	0	0.05±0.03	0.03±0.01
Brook stickleback	0.002±0.002	0	0.04±0.02	0	0
Burbot	0	0	0	0.02±0.02	0.015±0.01
Emerald shiner	0	0.03±0.03	0	0	0
Finescale dace	0.15±0.10	0.04±0.04	0.02±0.02	0	0
Flathead chub	0	0	0	0	0.007±0.007
Lake chub	0.13±0.06	0.60±0.06	0.09±0.03	0.05±0.04	0.009±0.005
Largescale sucker	0	0.0001±0.0001	0	0	0
Longnose dace	0	0.04±0.01	0.004±0.003	0.003±0.003	0
Longnose sucker	0.02±0.01	0.06±0.03	0.03±0.01	0.004±0.004	0.04±0.03
Mountain whitefish	0	0	0	0.19±0.04	0.003±0.002
Northern redbelly dace	0.02±0.02	0.02±0.02	0.69±0.06	0	0
Northern pike	0	0	0	0	0

Pearl dace	0.05±0.03	0.02±0.01	0.08±0.02	0.15±0.15	0
Redside shiner	0.08±0.04	0.01±0.01	0.003±0.002	0	0
Rainbow trout	0	0	0	0	0
Sculpin	0.02±0.02	0.06±0.03	0.008±0.007	0.07±0.05	0.87±0.03
Trout-perch	0.12±0.06	0.03±0.01	0.005±0.003	0	0
White sucker	0.33±0.06	0.07±0.02	0.03±0.007	0.003±0.003	0.010±0.007
Species dominance					
Numerically dominant species	White sucker	Lake chub	Northern redbelly dace	Mountain whitefish	Sculpin
Abundance rankings of the five numerically species (high to low)	White sucker > finescale dace > lake chub > trout-perch > redside shiner	Lake chub > white sucker > longnose sucker > sculpin > finescale dace	Northern redbelly dace > lake chub > pearl dace > brook stickleback > longnose sucker	Mountain whitefish > pearl dace > Arctic grayling > lake chub > bull trout	Sculpin > longnose sucker > longnose > bull trout > Arctic grayling
Species richness					
Total number of species in assemblage	11	15	12	11	9
Mean site species richness	3.9	4.4	4.5	3.7	1.8
Environmental variables					
Wetted width (m)	9.2±4.3	5.4±0.7	3.0±0.3	13.7±5.4	5.9±1.2
Slope (°)	0.008±0.003	0.007±0.001	0.015±0.003	0.015±0.009	0.017±0.002
Elevation (masl)	752±42	825±25	740±18	901±79	1044±21
Percent fines (%)	0.42±0.08	0.36±0.07	0.63±0.09	0.09±0.05	0.25±0.03
Percent small gravel (%)	0.03±0.02	0.06±0.02	0	0.07±0.04	0.06±0.01
Order	3.0±0.6	3.5±0.3	2.7±0.3	2.5±0.3	3.0±0.15
Bankfull width (m)	12.6±3.9	9.4±1.3	5.8±0.9	28.2±9.7	9.3±1.5
Watershed area (km^2)	410±306	277±176	17±4	398±184	109±36

We evaluated the cumulative effects of watershed disturbances on fish community structure the Simonette watershed using a reference -condition approach.

The cumulative effects of watershed disturbances on fish community structure were evaluated by determining how well habitat variables from the least-impaired sites predicted fish assemblage type at the potentially impacted sites.

The discriminant function model derived from the least-impacted sites had an overall classified success of only 57%.

These data suggest that the cumulative effects of human activities has impaired stream fish communities in the Simonette and Kakwa River basins.

Evaluating cumulative effects of watershed disturbances on fish community structure in the Simonette River Basin using a reference-condition approach.

We used discriminant function analyses to quantify how well fish communities at all of the 79 potentially impacted sites (i.e., test sites where watershed disturbance ranged from 10 to 43%) were predictable based on the four watershed variables (i.e., stream wetted width, reach slope, elevation and percent small gravel) identified as predictors of fish community types from the 106 reference (i.e., least-impacted sites, mean watershed disturbance = 9.5%) sites.

Overall, the discriminant function model had a classification success of only 57.1% (i.e., 44 of the 77 sites were correctly classified). Additional evaluations indicated that while the overall classification success was low, the ability of the discriminant function model to correctly classify sites was highly variable. In fact, the discriminant function model correctly classified both sites from assemblage 1 (i.e., classification success = 100%), 7 of the 17 sites into assemblage 2 (i.e., classification success = 41.2%), 5 of the 6 sites into assemblage 3 (i.e., classification success = 83.3%), 7 of the 14 sites into assemblage 4 (i.e., classification success = 50.0%), and 23 of the 38 sites into assemblage 5 (i.e., classification success = 60.5%). That the discriminant function model developed from the reference sites is a moderately poor predictor of fish assemblage structure in the test sites suggest that current levels of anthropogenic disturbance in the Simonette watershed result in detectable cumulative effects on fish community structure despite relatively low levels of watershed disturbance (Range = 21 to 61.8%, overall mean = 31.1%).

We completed two additional analyses to determine whether the inability of the discriminant function model to correctly classify fish assemblages based on habitat variables from reference (i.e., least-impacted) sites was related to differences in: 1) percent watershed disturbance or ii) stream crossing density. We predicted that if stream crossing density, rather than percent watershed disturbance, contributed to the low classification success of the discriminant function model, then the classification success of the discriminant function model using sites: i) with low stream crossing density should exceed that of sites with higher stream crossing density and ii) with low levels of watershed disturbance should not differ appreciably from that using sites with higher levels of watershed disturbance.

For the first analysis we ranked all test sites based on stream crossing density and arbitrarily divided these data into two groups with similar numbers of sites of low (range = 3 to 14, mean = 7.2, n = 38) and high (range 15 to 48, mean =27, n = 39) stream crossing density. While the absolute difference was not large, the classification success of the discriminant function model applied to the group of low stream crossing sites was higher (60.5%) than that applied to high stream crossing group (56.4%) (Figure 28A).

For the second analysis we ranked all test sites based on percent watershed disturbance and arbitrarily divided these data into two groups with similar numbers of sites with low (range = 21.3 to 29%, mean = 24.2%, n = 42) and high (range 29% to 61.8%, mean =39.3%, n = 35) percent watershed disturbances. Results from this analysis showed that the ability of the discriminant function model to correctly classify sites in the high disturbance group (88.6%) was substantially higher than that in the low disturbance group of sites (57.1%). This suggests that the low performance of the discriminant function in the Simonette watershed is poorly related to percent watershed disturbance.

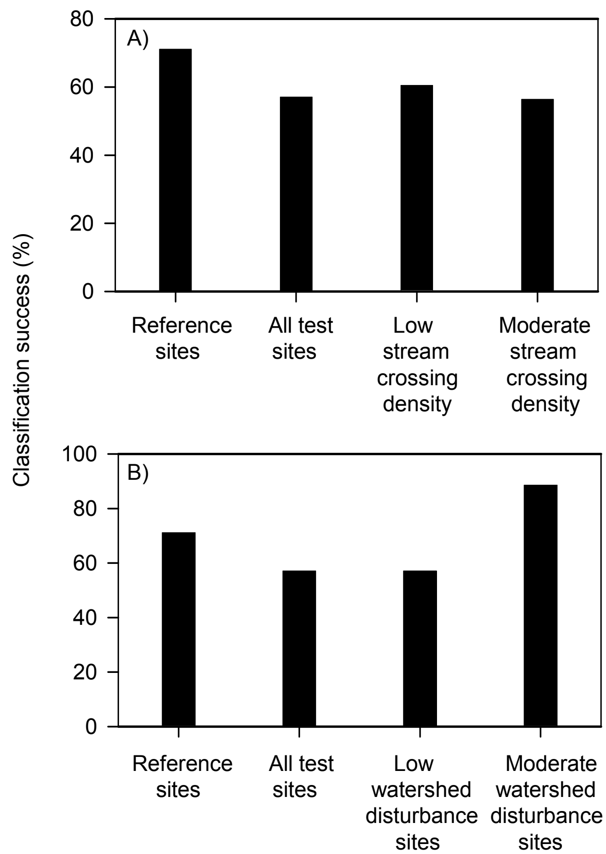

Figure 28. Comparison of percent classification success of the application of the discriminant function model from reference sites to all test sites and test sites with low and moderate densities of stream crossings (A) and all test sites and those with low and moderate watershed disturbance levels within the test sites (B) in the Simonette River Basin. Number of sites within each disturbance type: Reference sites N = 108, Test sites N = 42 (test site group 1) and N = 35 (test sites group 2).

5.0 DISCUSSION

5.1 Study Rationale and Focus

Developing management strategies that minimize cumulative effects of human-induced disturbances on ecological systems is arguably the single largest challenge to sustainable resource management.

Developing management strategies that minimize cumulative effects of human-induced disturbances on ecological systems is arguably the single largest challenge to sustainable resource management. In Alberta and elsewhere, rapid expansion of the forestry and oil and gas sectors, combined with conversion of forested lands to agriculture has raised concerns about the ecological sustainability of the boreal forest. For aquatic ecosystems, these industrial activities, including forest harvesting and linear disturbances, affect a myriad of watershed attributes that can alter the quantity and quality of habitat for stream fishes (e.g., Karr *et al.* 1985, Barton *et al.* 1985, Thedinga *et al.* 1989, Platts *et al.* 1989, Eaglin and Hubert 1993, Davies and Nelson 1994, Binns and Remmick 1994, Wichert and Rapport 1998, Kreutzweiser and Capell 2001, Kreutzweiser and Capell 2001, Sharma and Hilborn 2001).

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Cumulative effect assessments acknowledge that current environmental states result from the interactive effects of past and present activities and that such effects may not occur independently of each other.

Cumulative effect assessments acknowledge that current environmental states result from the interactive effects of past and present activities and that such effects do not occur independently of each other. Questions regarding what constitutes a cumulative impact typically focus on the sum of all individual impacts occurring in space and time (Contant and Wiggins 1991, Cocklin *et al.* 1992a). Studies completed under the umbrella of cumulative effect assessments can be broadly defined as those that: i) define conceptual or methodological issues, including potential indicators of cumulative impacts (e.g., Cocklin *et al.* 1992a, b, Parker and Cocklin 1993, Abbruzzese and Leibowitz 1997, Bevenger and King 1995, Dixon and Montz 1995, Spalling and Smit 1995), ii) identify, but do not quantify the cumulative effects of multiple stressors (Rice 1980, Salo and Cederholm 1980, Hemond and Benoit 1988, Preston and Bedford 1988, Fraley *et al.* 1989a, b, Wu *et al.* 2000), and iii) quantify the cumulative effects of multiple stressors or disturbances or predict rehabilitation from cumulative impacts, perhaps including partitioning the consequences of individual stressors (Johnson *et al.* 1988, Li *et al.* 1994, Chambers *et al.* 1997, Schnackenberg and MacDonald 1998, Scrimgeour and Chambers 2000, Wu *et al.* 2000).

Approaches to quantify cumulative effects have always been tailored to the problem being addressed and have resulted in a diversity of study designs.

Approaches to quantify cumulative effects have always been tailored to the problem being addressed and have resulted in a diversity of study designs. For example in aquatic ecosystems, Chambers *et al.* (1997) and Scrimgeour and Chambers (2000) quantified the cumulative effects of pulp mill and municipal effluents on algal biomass, nutrient and under-ice dissolved oxygen concentrations in the Athabasca River, Alberta. Neither study attempted to partition impacts from the multiple stressors. Wu *et al.* (2000) predicted the cumulative effects of improvements in riparian conditions on juvenile steelhead abundance in two Oregon streams and Li *et al.* (1994) quantified the cumulative effects of riparian disturbance by grazing on standing crops of rainbow trout in high desert streams.

In the present study, we evaluated the cumulative effects of industrial activities on stream fish communities in two moderately large watersheds located in northern Alberta.

In the present study, we evaluated the cumulative effects of industrial activities on stream fish communities in two moderately large watersheds in northern Alberta. Our objective was to describe the cumulative effects of watershed disturbances on fish communities by: 1) defining the areal extent that study watersheds were altered by patch (e.g., forest harvest blocks, well sites) and linear disturbances (e.g., roads, railway lines, power lines, pipelines, seismic lines), and 2) quantifying the adjacency, type (e.g., crossings by roads, pipelines, power lines, seismic) and density of stream crossings. Using these data we developed empirical models that: 1) quantified the variance in fish population and community characteristics that could be explained by watershed disturbance attributes. This approach allowed us to address the following three focal questions within a cumulative effects framework:

This approach allowed us to address the following three focal questions within a cumulative effects framework:

1) Is the presence of fish, game fish and individual species affected by watershed disturbances?

2) Are species density and biomass affected by watershed disturbances?

3) Is fish community structure affected by watershed disturbances?

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2) Are species density and biomass affected by watershed disturbances?

3) Is fish community structure affected by watershed disturbances?

We predicted that: 1) fish presence would be moderately to strongly predictable based on watershed attributes, 2) fish presence would not be strongly affected by watershed disturbances, 3) fish abundance and biomass would be affected by cumulative watershed disturbances, 4) cumulative effects of watershed disturbances would be more severe in the Simonette rather than in the Kakwa watershed, commensurate with differences in levels of industrial activity between these watersheds, and 5) cumulative effects of watershed disturbance would be fish species-specific.

Lastly, we discuss the management implications of our work and how our approach and results can assist with ecological assessments and planning of forest harvest blocks and road networks and their broader application to sustainable management of stream fish communities.

Lastly, we discuss the management implications of our work and how our approach and results can assist with ecological assessments and planning of forest harvest blocks and road networks within the broader context of sustainable resource management and the conservation of stream fish communities.

5.2 Cumulative Effects of Watershed Disturbance on Fish Presence

With two exceptions, our results from logistic regression analyses provided little evidence of cumulative effects of watershed disturbances on the presence of fish, game fish or individual species at the landscape, watershed or stream-order scales. In fact, of all the logistic regression models developed, only 1 of the 41 derived from the Simonette River Basin and only three of 20 statistically meaningful (i.e., those with relatively good sample sizes and where logistic models produced statistically valid models) models derived from the Kakwa River Basin identified watershed disturbance attributes as significant predictors of fish presence. While our analyses did not typically identify watershed disturbance attributes as statistically significant variables, logistic regression models usually produced highly statistically significant models that correctly classified fish presence or absence between 70 to 80% of the time. In the majority of cases, one to three, often-reoccurring variables, related to stream size (e.g., watershed areas, stream bankfull width, wetted width, mean depth, elevation), stream reach slope, reach elevation, and size composition of the substratum provided moderately powerful predictors of fish occurrence.

The two notable exceptions to the general lack of detectable effects of watershed disturbances on fish occurrence were the statistically significantly effects of watershed disturbance on the presence of bull trout in the Kakwa and Simonette River Basins.

The two notable exceptions to the general lack of detectable effects of watershed disturbances on fish occurrence were the statistically significant effects of watershed disturbance on the presence of bull trout in both the Kakwa and in the Simonette River basins. In the Kakwa River Basin our landscape and third & fourth order watershed-scale level analyses showed that the presence of bull trout was negatively related to the percent watershed disturbed. Additional analyses showed that the negative association between bull trout presence and percent watershed disturbance arises primarily from forest harvesting which, on average, accounted for about 84% of human-induced disturbance within the Kakwa River Basin. In fact, our logistic regression model derived for third and fourth order stream reaches of: i) percent watershed harvested and ii) reach slope was highly statistically significant ($P<0.0001$) and correctly classified bull trout presence and absence in stream reaches 80% of the time. The notion that percent watershed harvested is the primary factor determining bull trout occurrence was further support by three additional analyses. First, the logistic regression model solely using percent watershed harvested was also highly significant and moderately powerful in predicting bull trout occurrence (classification success of analyses completed at the landscape level analysis = 71%). Secondly, additional analyses indicated that the presence of bull trout was unrelated to percent of the watershed disturbed by all other non-harvesting activities including percent disturbance by roads, pipe line and seismic lines and that reach slope alone was not a significant ($P = 0.44$) predictor of bull trout presence. Thirdly, the watershed disturbance attributes of total percent disturbed and percent watershed harvested were not strongly related to other predictor variables. However, we can not exclude the possibility that increased forest harvesting has coincided with increased bull trout mortality mediated by increased access and angler effort.

The negative relationship between bull trout presence and percent of the watershed harvested observed in the Kakwa watershed was strongly driven by data from the Prairie Creek sub-basin that has been subjected to moderate levels of forest harvesting.

The negative relationship between bull trout presence and percent of the watershed harvested observed in the Kakwa watershed was strongly driven by data from the Prairie Creek sub-basin that has been subjected to moderate levels of forest harvesting. Within this sub-basin, forest harvesting has set back about 23% of the watershed to early successional vegetative stages (overall range among individual watersheds = 6.8 to 44.8%) and in relative terms, represents one of the most disturbed region within the larger Kakwa River Basin. The mechanisms causing: i) the negative relation between bull trout occurrence and percent watershed harvested and ii) the absence of bull trout from many, but not all of the study sites in Prairie Creek watershed, are not known. However, this negative relationship may be associated with increased sediment loadings or changes in light and thermal regimes that affect either egg and juvenile

The negative effects of forested harvesting on the presence of bull trout in the Kakwa River Basin could be mediated through a diversity of mechanisms including those related to: i) decreased propensity of adults to migrate into Prairie Creek to spawn, ii) reduced availability of spawning habitat, iii) reduced egg and juvenile survival, iv) reduced propensity of adult fish reared in Prairie Creek to return to that sub-basin, or v) decreased densities of resident bull trout due to the loss of pool habitats.

survival or the propensity of adults from the main stem of the Kakwa River to spawn in tributaries of Prairie Creek. Previous studies of bull trout in the Kakwa River using radio transmitters have shown that relatively few adults spawn in Prairie Creek compared with adjacent and similar size tributaries such as Chicken and Lynx creeks (Hvenegaard unpubl. data). Lastly, anecdotal evidence, based on discussions with local residents who have a long history of angling within the Prairie Creek sub-basin, also suggests that bull trout may have used the Prairie Creek sub-basin to a greater extent prior to marked increases in forest harvesting. Interestingly, there is no evidence to suggest that the negative effects of forest harvesting on bull trout arose because forest companies did not abide by best management practices and provincial standards defined within Forest Harvesting Operating Ground Rules.

The negative effects of forested harvesting on the presence of bull trout in the Kakwa River Basin could be mediated through a diversity of mechanisms including those related to: i) decreased propensity of adults to migrate into Prairie Creek to spawn, ii) reduced ability of adults within Prairie Creek to locate suitable spawning habitat (i.e. reduced spawning habitat availability), iii) reduced egg and juvenile survival, iv) reduced propensity of adult fish reared in Prairie Creek to return to that sub-basin (i.e., reduced spawning site fidelity), or v) decreased densities of resident bull trout due to the loss of pool habitats perhaps related to the loss of large woody debris that can create and maintain deep pool habitat (Andrus *et al.* 1988, Fraley *et al.* 1989, Hicks *et al*. 1991a, b, Glova and Sagar 1994, Beechie and Sibley 1997, Watson and Hillman 1997, Hauer *et al.* 1999, McIntosh 2000). For example, Watson and Hillman (1997) reported that the presence of bull trout in the Pacific Northwest was positively related to large substrates, slow-water habitats with riparian zones dominated by trees and shrubs rather than grasses, forbs and barren areas.

Our suggestion that forest harvesting may reduce the reproductive success of bull trout by reducing the availability of spawning sites or through reduced egg survival is based on previous studies which have shown that bull trout spawn in low-gradient, low velocity areas accompanied by complex groundwater exchanges (Weaver and White 1985, Fraley and Shepard 1989, Baxter *et al.* 1999, Baxter and Hauer 2000). For instance, in the most detailed description of bull trout spawning sites, Baxter and Hauer (2000) found that bull trout redds were primarily located in bounded alluvial valley segments with extensive upwellings and complex hyporheic exchanges. Within these reach types, bull trout established redds within transitional bedforms with strong localized downwellings and high within-gravel flow rates. These habitats provide stable incubation conditions, and at the larger scale, reaches with high groundwater upwellings may also maintain migratory passages for adults during the moderately low flow fall conditions, and during winter months may reduce the formation of anchor ice (Watson and Hillman 1997). Weaver and White (1985) also identified substantial differences in survival of bull trout embryos depending on the size composition of substratum material. They reported 60% survival of incubating bull trout embryos in gravels that contained 30% fine materials compared to that zero survival of embryos incubated in gravels with 44% fine materials.

In the absence of additional harvesting, it is likely that the effects of forest harvesting on bull trout should moderate through time with forest regeneration and subsequent changes in water yields, sediment loadings, and light and thermal inputs.

In the absence of additional harvesting, it is likely that the effects of forest harvesting on bull trout should moderate through time with forest regeneration and subsequent changes in water yield, sediment loadings, and light and thermal inputs (Feller 1981, Hewlett and Fortson 1982, Barton *et al.* 1985, Platts *et al.* 1989, Hicks *et al.* 1991a, b, Eaglin and Hubert 1993). The recovery of bull trout populations in Prairie Creek however may take several years because: i) recent studies suggest that few adults currently spawn in Prairie Creek and ii) bull trout display a moderate level of spawning site fidelity in the Kakwa River Basin (Hvenegaard unpublished data). Recovery of bull trout in the Prairie Creek sub-basin may be hindered by additional forest harvesting, a possibility given that the sub-basin still contains moderate amounts of commercial grade forest. We suggest that additional research is required to: i) monitor the recovery of bull trout in the Prairie Creek sub-basin, and ii) determine whether additional forest harvesting in largely undisturbed areas of the Kakwa River Basin also result in reductions in the distribution of bull trout. The importance of bull trout populations within sub-basins of the Kakwa and Simonette watersheds to metapopulation stability is poorly understood (Dunham and Reiman 1999).

In the Simonette River Basin, our logistic regression analyses also identified a statistically significant and negative relationship between the presence of bull trout and the cumulative (i.e., total) density of stream crossings located downstream of the study reach.

In the Simonette River Basin, our logistic regression analyses also identified a statistically significant and negative relationship between the presence of bull trout and the cumulative (i.e., total) density of stream crossings located downstream of the study reach. On average, sites that contained bull trout had about 40% fewer stream crossing (15 crossings) located within 40 km downstream than those where bull trout were absent (24 crossings). Additional analyses also revealed statistically significant ($P < 0.05$) and negative relationships between bull trout presence and density of: i) seismic lines, ii) road crossings, and iii) pipelines located within 40 km downstream of study reaches. These relations occur because while seismic lines account for the majority of all stream crossings, density of seismic lines, roads and pipelines downstream were significantly correlated.

Stream crossings may affect the presence of bull trout through a diversity of routes that reduce habitat quality, quantity or accessibility.

Stream crossings may affect the presence of bull trout through a diversity of routes that reduce habitat quality, quantity or accessibility (Eaglin and Hubert 1993, Slawski and Ehlinger 1998). Stream crossings, including those resulting from roads, are known to be associated with increased sediment inputs and changes in thermal and light regimes due to the removal of riparian vegetation that reduces canopy cover adjacent to streams (See Reed 1977, Furniss *et al.* 1991). Bull trout are a cold-water species and are likely sensitive to the relatively small changes in thermal regimes. It is possible that small, incremental increases in solar inputs at multiple stream crossings, due to removal or reduction in canopy cover of riparian forests (Li *et al.* 1994) may result in meaningful increases in water temperatures and reductions in habitat quality for bull trout.

The negative association between bull trout presence and stream crossing density could also arise from stream fragmentation. However, our data from the stream crossing survey suggests the vast majority of culverts likely provide fish passage. In fact, our assessments of the potential for culverts to fragment stream reaches based on hang height measured in the summer and fall showed that very few of the culverts downstream of study reaches were classified as potentially imposing extreme (hang height: > 30 cm), high (hang height: 10 to 30 cm) or moderate (hang height: 2 to 10 cm) risk of fragmentation. Nevertheless, we are cautious in suggesting that culverts do not contribute to the negative relation between bull trout presence and stream crossing density because our assessments of fragmentation potential were not completed during the lower water period of late fall when bull trout migrate into spawning tributaries. Additionally, while culverts with high hang heights will impede fish migration, the presence of a continuous flow path for fish movement does not necessarily indicate that the structure is not an impediment. In fact, an improved understanding of the hydraulics of culverts, including the effects of water turbulence on fish passage, is required (e.g., Bates and Powers 1999).

Perhaps the two most disturbing aspects of the negative relations between bull trout presence and forest harvesting and density of stream crossing are that: i) negative effects of forest harvesting and stream crossings on bull trout were detectable at relatively low levels of watershed disturbance and ii) that the slope of the relation between bull trout presence and percent watershed harvested and density of stream crossings are relatively linear.

Perhaps the two most disturbing aspects of the negative relations between bull trout presence and forest harvesting and density of stream crossing are that: i) negative effects of forest harvesting and stream crossings on bull trout were detectable at relatively low levels of watershed disturbance and ii) that the slope of the relation between bull trout presence and percent watershed harvested and density of stream crossings are relatively linear (See Figure 12D). The latter result suggests that a disturbance threshold, even if it exists, likely reflects very low levels of watershed disturbance. Such a threshold effect could occur if the relation between bull trout presence and percent watershed harvested is characterized by a sigmoidal function or other functions where the presence of bull trout is initially unaffected by percent watershed disturbance (plateau stage) but subsequently declines with increasing levels of forest harvesting.

5.3 Cumulative Effects of Watershed Disturbance on Fish Density and Biomass

Our regression analyses generally provided relatively little evidence that cumulative levels of watershed disturbance have detectable effects on fish abundance in the Kakwa and Simonette watersheds.

Our regression analyses generally provided relatively little evidence that cumulative levels of watershed disturbance have detectable effects on fish abundance in the Kakwa and Simonette watersheds. Indeed, watershed disturbance attributes explained significant amounts of variance in fish density and biomass in only two of the 10 analyses completed in Kakwa and only four of the 17 analyses completed in the Simonette River basins. In contrast to our initial expectation, watershed disturbance attributes in some cases were positively related to fish density or biomass suggesting that fish abundance (of species like sculpin) may increase with increasing levels of watershed disturbance, at least within the ranges of watershed disturbance that currently exist. For example, our multiple regression analysis of total fish density, but not biomass, in the Kakwa watershed identified a statistically significant and positive association between total fish density and percent watershed disturbance. This relationship arises because of the positive relation between density of sculpin and percent watershed disturbance. Further, our data from the Simonette watershed showed that: i) total fish density was positively related to crossing density, ii) density of dace was positively related to density of stream crossings, iii) total fish biomass was positively related to percent watershed disturbance and iv) biomass of sculpin was positively related to the density of seismic lines.

While the mechanisms underlying the positive relations between fish density and abundance with watershed disturbance are not well understood, the positive relation between total density and biomass and density and biomass of some species could arise from increased nutrient loadings to streams and related trophic cascades that ultimately enhance some fish populations.

While the mechanisms underlying the positive relations between fish density and abundance with watershed disturbance are not well understood, the positive relation between total density and biomass and density and biomass of some species could arise from increased nutrient loadings to streams and related trophic cascades that ultimately enhance some fish populations (Hawkins *et al.* 1983, Li *et al.* 1994, Johnston *et al.* 1990, Deegan and Peterson 1992). For example, it is possible that low levels of forest harvesting may increase density or biomass of some fish species through disturbance-mediated nutrient pulses and increases in thermal and light regimes that stimulate primary and secondary production. In fact, several studies have shown that experimental fertilization can increase algal and benthic macroinvertebrate production (e.g., Johnston *et al*. 1990, Deegan and Peterson 1992). For instance, Johnston *et al.* (1990) reported increased size of steelhead trout (*Oncorhnuchus mykiss*) and coho (*Oncorhynchus kisutch*) following whole-river fertilization of a nutrient-deficient stream with inorganic phosphorus and nitrogen in the Keogh River, British Columbia. Deegan and Peterson (1992) found that experimental fertilization of phosphorus in the Kuparuk River, Alaska, increased the size of young of the year fish and adult growth and condition at least in some years following enrichment. Peterson (1993) reported higher trout density adjacent to electric transmission rights-of-way in forested headwater stream in New York compared to adjacent reference sites. Increases in density were associated with increased light levels, dense riparian vegetation and narrower and deeper stream channels. Watershed disturbances such as increased deposition of fine sediments in the substratum can decrease survival of eggs and embryos (e.g., Everest *et al.* 1987, Scrivener and Brownlee 1989, Waters 1995, Argent and Flebbe 1999), and reduce fish abundance. However, the net effect of watershed disturbances on fish population density and biomass is determined by the extent to which effects of disturbance that decrease fish abundance are countered by effects that increase fish abundance. As a result, net outcomes of the effects of watershed disturbances on total fish abundances are expected to be variable depending on: i) the suite of fish species present and their physiological and behavioural responses to various watershed disturbances and ii) levels of watershed disturbance.

For example, Li *et al*. (1994) reported that in northward flowing streams, biomass of rainbow trout was negatively correlated with solar radiation whereas positive relationships were observed with stream discharge and water depth. While augmented solar inputs may have increased the benthic invertebrate prey base, increased biomasses of rainbow trout may not have occurred at high elevations because enhanced solar inputs resulted in water temperatures that approached upper thermal limits.

Our data also support the prediction that the effects of watershed disturbance would be species-specific, similar to that reported previously. Garman and Moring (1993) identified species-specific responses in blacknose dace (*Rhinichthys atratulus*) and creek chub (*Semotilus atromaculatus*) following forest harvesting in the Piscataquis River Maine, U.S.A. Production of blacknose dace was significantly lower following logging whereas production of creek chub was significantly higher following logging. Differences were thought to be related to the extent that the two species were able to shift from a declining benthic prey resources to resources dominated by terrestrial arthropods (Garman and Moring 1993). Rutherford *et al.* (1992) also reported species-specific responses to forest harvesting and suggested that *r*-selected species (i.e., small size, short-lived) may respond quickly to forest harvesting whereas *k*-selected species (large species, long-lived) may exhibit a delayed response to forest harvesting. Jones *et al.* (1999) also described species-specific responses to the removal of riparian forest in southern Appalachian streams. Our regression analyses from the Kakwa River Basin identified positive relations between : i) total fish biomass and watershed disturbance, ii) biomass of sculpin and density of seismic lines, iii) density, but not biomass, of sculpin and percent watershed disturbed and iv) density of dace and density of stream crossings. Such relations could arise from increased nutrients loadings, thermal inputs that enhance primary production combined with trophic cascades.

Our regression models seldom identified watershed disturbance attributes as being statistically significant, powerful explanatory variables of variance in fish density and biomass. In the majority of cases watershed disturbance attributes explained only <5% of the overall variance in fish abundance.

Our regression models seldom identified watershed disturbance attributes as being statistically significant, powerful explanatory variables of variance in fish density and biomass. In the majority of cases watershed disturbance attributes explained only <5% of the overall variance in fish abundance. The general absence of detectable effects of watershed disturbance on fish density and biomass could arise because: i) current levels of watershed disturbance do not strongly affect fish abundance or ii) that such effects are present but that our analyses were not sufficiently powerful to detect such impacts.

While we are currently unable to discriminate between these two opposing hypotheses, we expect that they both contribute to the overall prevalence of watershed disturbance attributes as being poor predictors of fish abundance. For instance, a wealth of information has demonstrated that logging can negatively affect stream fish communities by altering a complex suite of processes related to hydrology (e.g., water balance, water quality [e.g., temperature, suspended sediment, dissolved oxygen, nutrients]), upland erosion, stream sediment dynamics (e.g., instream sediment processing, processing and retention of organic matter including nutrients), stream channel form and geomorphic processes (e.g., development and stability of channel bed) (Gibbons and Salo 1973, Coats and Miller 1981, Murphy *et al.* 1981, Berkman and Rabeni 1987, Meehan 1991). However, results of logging and other human-induced stressors on aquatic systems are profoundly affected by the intensity, magnitude and frequency of the disturbance as well as the geological and physical attributes of the harvested watershed. The majority of studies reporting strong, negative effects of forest harvesting on aquatic ecosystems were generally associated with the removal of large blocks of forest, often comprising the vast majority of small watersheds, typically through clear cutting, removal of all, or part of riparian stands, application of herbicides or burning of residual materials following harvesting and the removal of instream woody debris from relatively steep watersheds (e.g., See Campbell and Doeg 1989). For example, Scrivener and Brownlee 1989) reported ca.50% decline in survival to emergence of coho (*Oncorhynchus kisutch)* and chum salmon (*Oncorhynchus keta*) following harvesting of 41% of the Carnation Creek watershed between 1976–1981. In contrast, the effects of alternate logging practices, including small forest harvest blocks (< 40 ha), combined with non-harvested reserve areas, understory and soil protection measures, minimal physical disturbance of stream channels, and the retention of meaningful amounts of riparian forest on stream fish communities within the boreal forest are not well understood.

Lastly, while our results from regression analyses indicate that current levels of watershed disturbance in the Kakwa and Simonette watersheds are not strong predictors of fish density and biomass, we are cautious in suggesting that current levels of watershed disturbance are not affecting fish density and biomass.

Alternatively, the inability of our regression analyses to identify watershed disturbance could arise from low statistical power, in part due to high variability in fish density and biomass. High inter-annual variability in abundance estimates for fish have been reported previously (e.g., House 1995, Ham and Pearsons 2000) and may affect the ability of empirical models to predict density or biomass. For instance, Scarnecchia and Bergersen (1987) reported high annual variability affected the ability of a habitat quality model to predict trout biomass in 11 Colorado streams. High variability in fish abundance (e.g., Grossman *et al.* 1990) is frequently cited as being an important cause of the lack of detectable effects of industrial activities on aquatic biota (e.g., Rose 2000, Williams *et al.* 2002).

Our data revealed detectable effects of watershed disturbances on fish community structure in both the Kakwa and Simonette watersheds.

Lastly, while our results from regression analyses indicate that current levels of watershed disturbance in the Kakwa and Simonette watersheds are not strong predictors of fish density and biomass, we are cautious in suggesting that current levels of watershed disturbance are not affecting fish density and biomass for at least two reasons. First, our ability to detect such effects may have been low due to high variance. Second, it is possible that the full impacts of current levels of watershed disturbance may not be become fully apparent for several years associated with time lags (e.g., Findlay and Bourdages 1999) in detecting impacts which may be associated with slow responses from *k*-selected species (Rutherford *et al.* 1992). Finally, our analyses showing detectable relationships between fish density and biomass with watershed disturbance attributes may be an early warning indicator of future impacts.

Results from the discriminant function analyses suggest that cumulative effects of human activities has resulted in impairment of stream fish communities in the Kakwa and Simonette River Basins.

5.4 Cumulative Effects of Watershed Disturbance on Fish Community Structure

We used a reference-condition approach (Reynoldson *et al.* 1995, Reynoldson *et al.* 2001) to quantify the cumulative effects of watershed disturbances on fish communities in the Kakwa and Simonette River basins. In its broadest sense, reference-condition approaches use relations between biotic communities and environmental conditions from relatively undisturbed sites to predict the expected community types at potentially impacted locations (e.g., Reynoldson *et al.* 1995, 1997). They have been applied extensively to evaluate the effects of watershed disturbances on invertebrates (e.g., Bailey *et al.* 1998, Reece *et al.* 2001, Reynoldson 2001) and more recently on fish communities (e.g., Joy and Death 2002, Tonn *et al.* In Press).

In the Kakwa River, the relatively low performance of the discriminant function model to predict fish community structure at the potentially impacted test sites, likely arises from: i) the negative effects of forest harvesting on bull trout presence and ii) that bull trout are the most frequently occurring fish species in the Kakwa watershed.

Our data revealed detectable effects of watershed disturbances on fish community structure in both the Kakwa and Simonette River basins. Based on differences in classification success, effects of watershed disturbances were most pronounced in the Kakwa compared to that in the Simonette River Basin. In the Kakwa River Basin analyses of fish communities from least impacted sites identified three distinct fish assemblages dominated primarily by bull trout (assemblage 1), sculpin (assemblage 2) and mountain whitefish and to a lesser extent rainbow trout, bull trout, Arctic grayling and sculpin (Assemblage 3). Additional analyses showed that stream wetted width, stream reach slope, site elevation and percent small gravel were moderately good predictors of the three fish assemblages. However, when applied to the potentially impacted sites (percent watershed disturbance > 10%, overall mean = 27%) the discriminant function model performed relatively poorly and correctly classified only 50% of them, even after the exclusion of the four sites where impacts likely arose from fish stocking practices in Musreau Lake rather than watershed disturbances.

We expect that the failure of the discriminant function model to correctly predict fish communities at the test sites in the Simonette River Basin may be primarily related to stream crossings by allowing increased angling effort, reductions in habitat quality, or increased habitat fragmentation.

The relatively low performance of the discriminant function model to predict fish community structure at the potentially impacted test sites, likely arises from: i) the negative effects of forest harvesting on bull trout presence and ii) that bull trout are the most frequently occurring and often most numerically abundant fish species in the Kakwa watershed. The inability to strongly predict fish community types in the potentially impacted sites is however, consistent with the statistically significant and negative relation between bull trout presence and percent forest harvesting. An additional analysis also showed that the application of the discriminant function model was not greatly improved when analyses of potentially impacted test sites were divided into those reflecting low (mean watershed disturbance = 17%) compared to moderate (mean watershed disturbance = 32%) levels of watershed disturbance. We expect that the failure of the discriminant function model to correctly predict fish communities at the test sites in the Kakwa River Basin arises largely from the effects of forest harvesting because it is the primary watershed disturbance activity.

In the Simonette River Basin, community analyses identified five discrete fish assemblages dominated by white sucker (assemblage 1), lake chub (assemblage 2), northern redbelly dace (assemblage 3), pearl dace, mountain whitefish and Arctic grayling (assemblage 4), and sculpin (assemblage 5). Similar to that observed in the Kakwa River Basin, stream wetted width, stream reach slope, site elevation and percent fine material within the substratum were also moderately good predictors of these fish assemblages (overall classification success = 71%). While relatively more successful than that in the Kakwa River Basin, when applied to the potentially impacted sites (percent watershed disturbance > 20%, overall mean = 27%), the discriminant function model also performed relatively poorly and correctly classified only 57% of sites. Additional analyses suggest that the inability of the discriminant function model to correctly classify fish communities was not strongly related to overall levels of watershed disturbance. In fact, our analyses showed that discriminant function model was moderately successful in correctly classifying fish communities (percent classification success = 88%) from sites whose watershed had been experienced moderate levels of watershed disturbance (overall mean = 39%, range = 29 to 62%) compared to less disturbed watersheds (overall mean = 24.2%, range = 21.3 to 292%. However, there was a small decline in the ability of the model to correctly classify fish communities from sites defined as having low compared to moderate numbers of stream crossings located within 40 km downstream of the study reaches.

We expect that the failure of the discriminant function model to correctly predict fish communities at the test sites in the Simonette River Basin may be primarily related to stream crossings that provide increased angling opportunities or through habitat alterations that reduce habitat quality, or increase habitat fragmentation. The mechanisms by which stream crossings, including those associated with crossing structures (e.g., bridges, culverts) affect fish communities inhabiting the boreal forest are poorly understood. In addition, while the effectiveness of stream crossing structures to provide fish passage has been evaluated elsewhere (e.g., Slawski and Ehlinger 1998, Warren and Pardew 1998), there is a paucity of information quantifying the effectiveness of crossing structures to provide fish passage in Alberta's boreal forest. However, if current trends continue, the rapid expansion in forest harvesting and continued exploration for oil and gas reserves in Alberta will dramatically increase stream crossing and road densities within the boreal forest. Due to the large number of predicted stream crossings, small improvements in their design have the potential to dramatically reduce impacts on fish communities and we suggest that additional research in this area is urgently required.

5.5 Management of Stream Fish and the Use of Empirical Tools

The use of empirical models to quantify the impacts of human activities on aquatic ecosystems has expanded rapidly over the last 15 years and biological assessments using multi-metric or multivariate approach are now common place.

In the present study we evaluated the cumulative effects of watershed disturbances on fish communities using three empirical modeling approaches of: 1) logistic regression, 2) linear and non-linear regression and, 3) the reference–condition approach. Using instream and watershed characteristics as explanatory variables, logistic regression, the reference condition approach, and to a lesser extent, linear and non-linear regression models often provided moderately powerful equations explaining variance in fish presence, community types, and fish density and biomass, respectively.

The use of empirical models to quantify the impacts of human activities on aquatic ecosystems has expanded rapidly over the last 15 years and biological assessments using multi-metric (Karr *et al.* 1985, Steedman 1988, Karr and Chu 1999) or multivariate approach (Barbour *et al.* 1996, Reynoldson *et al.* 1997, Bailey *et al.* 1988, Hawkins *et al.* 2000, Reynoldson *et al.* 2001, Austin 2002) are now common place. The use of logistic regression as a fisheries management tool also appears to be increasing in popularity. For example, Watson and Hillman (1997) and Paul and Post (2001) used logistic regression to predict the presence of bull trout in the stream reaches from the eastern slopes of the Rocky Mountains in Alberta, Canada and in the Pacific northwest of the United States. Oberdorff *et al.* (2001) used logistic regression to identify the probability of occurrence of 34 fish species in French rivers as part of the development of a national program to detect anthropogenic perturbations. Presence-absence data may also provide insights into temporal patterns in fish stock abundance (e.g., Mangel and Smith 1990). In contrast, other regression approaches, including those defining relations between fish communities and habitat and watershed variables, have been a prominent fisheries tool for several decades (e.g., Fausch *et al.* 1988, Platts and McHenry 1988, Marcus *et al.* 1990, Olden and Jackson 2002). The use of the reference-condition as part of an overall biomonitoring tool to assess impacts on stream fish is also increasing. For instance, using a reference-condition approach, Joy and Death (2002) reported that 14 environmental variables correctly classified 70% of fish communities into one of six assemblages identified from 200 reference (i.e., relatively pristine) sites.

We expect that the management of stream fish communities in Alberta would benefit from the increased use of a broad suite of empirical models, including those used in the present study.

We expect that the management of stream fish communities in Alberta would benefit from the increased use of a broad suite of empirical models, including those used in the present study. These statistical approaches can represent practical tools to evaluate the extent of ecological impairment, including quantifying cumulative effects and to forecast how impacts may be manifested through time (Boisclair 2001).

5.6 Cumulative Effects and the Search for Disturbance Thresholds

Research on the effects of forest harvesting on aquatic resources in Canada's Boreal forest has increased greatly over the last 10 years.

These and other studies have also prompted the search for and discussion of disturbance thresholds that describe non-linear relations between chemical and biological endpoints and the magnitude, intensity or frequency of watershed disturbances.

Research on the effects of forest harvesting on aquatic resources in Canada's boreal forest has increased greatly over the last 10 years. These studies include evaluations of: i) effects of forest harvesting on aquatic resources, ii) impacts of forest harvesting compared to those resulting from natural disturbances (e.g., fire), iii) the role of riparian stands to minimize the effects of forest harvesting in upland areas (Carignan *et al.* 2000, Garcia and Carignan 2000, McEachern *et al.* 2000, St-Onge and Magnan, Scrimgeour *et al.* 2000, Steedman and Kushneriuk 2000, Tonn *et al.* In Press). These and other studies have also prompted the search for and discussion of disturbance thresholds that describe non-linear relations between chemical and biological endpoints and the magnitude, intensity or frequency of watershed disturbances (e.g., Stevenson and Hauer 2002). Within a forest-harvesting context, a disturbance threshold can be viewed as the maximum amount (typically expressed as a percent) of forest harvesting within a watershed that does not result in a detectable effect on water quality or aquatic life. Disturbance thresholds can be calculated at a variety of spatial and temporal scales for a broad suite of structural (e.g., chemical, biological) or functional attributes (e.g., nutrient cycling).

For boreal stream ecosystems, relations between percent watershed disturbance and changes in fish presence and abundance do not exist. However, several studies have reported substantial changes in fish communities following clear cutting of the majority, or all of a watershed (See Campbell and Doeg 1989). Thresholds in increases in stream flow following patch harvesting combined with burning of residual material, have been reported at as low as 25% of a small (101 ha) watershed in western Oregon (Hicks *et al.* 1991a). Despite a relatively small disturbance, the increase in stream flow was relatively persistent (i.e., 16 years). In comparison, Hicks *et al.* (1991a) also reported that complete clear cutting of an adjacent and similar sized watershed (96 ha) also increased stream flow but that the effects were substantially less persistent (i.e., 8 years). While forest harvesting followed by burning exceeded the disturbance thresholds in both watersheds, differences in the persistence of such effects were thought to be due to differences in slope that affected the regeneration of riparian vegetation (Hicks *et al.* 1991a).

Studies of lakes from the boreal forest reported detectable effects of harvesting on lake water chemistry and mercury levels in fish under moderately low watershed disturbance levels (Garcia and Carignan 2000, Carignan *et al.* 2002). For example, Garcia and Carignan (2000) reported significantly higher levels of mercury in moderately large adult northern pike from lakes whose watersheds had been harvested (range of clear cut harvesting = 11 to 72%, overall mean = 43%) compared to undisturbed reference watersheds. In a related study, Carignan *et al.* (2002) also reported significantly higher concentrations of total phosphorus and total organic nitrogen from a larger selection of harvested and references lakes. However, Steedman and Kushneriuk (2000) found that clear-cutting of between 33 to 77% of individual lakes had minimal effects on late-summer thermoclines in three small northwestern lakes in Ontario.

From a management perspective, the use of disturbance thresholds is inherently appealing because they can provide a framework on which to schedule and if required, limit forest harvesting and other industrial activities to minimize ecological impacts.

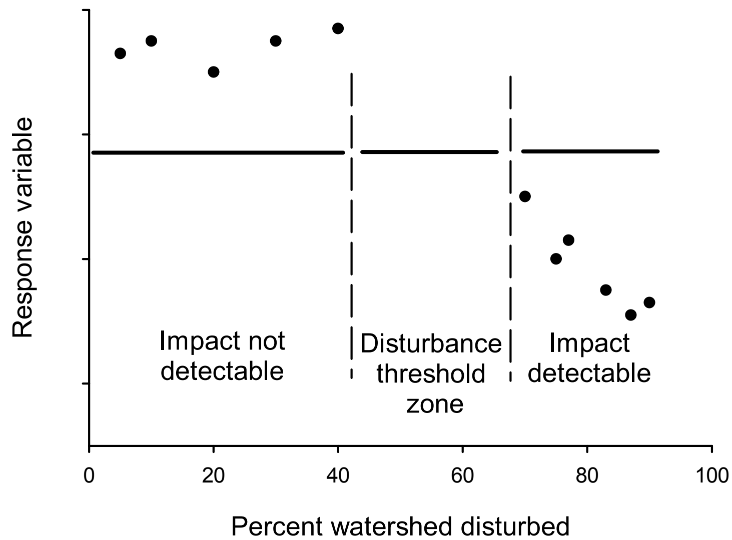

Figure 29. A hypothetical disturbance dose-response curve. The relationship between the response variable and percent watershed disturbed can conceptually be divided into three disturbance zones. The first zone reflects levels of watershed disturbance where impacts on the response variable are not detectable. The second zone reflects levels of watershed disturbance where the disturbance threshold is likely located and the third zone reflects levels of watershed disturbance above the disturbance threshold. In this hypothetical example, the disturbance threshold is located between disturbance levels of 40 and 70%.

From a management perspective, the use of disturbance thresholds is inherently appealing because they can provide a framework on which to schedule and if required, limit forest harvesting and other industrial activities to minimize ecological impacts. Results from our study, however, may not be overly useful in the search for disturbance thresholds. In fact, while our data identified negative relations between bull trout occurrence and: i) percent harvested in the Kakwa River Basin and ii) density of road crossings in the Simonette River Basin, relations between bull trout presence and disturbance attributes observed in the Kakwa and Simonette River basins were generally linear and displayed little evidence that low level of disturbance did not result in detectable effects on fish presence.

5.7 Stream Management and the Precautionary Approach

Land base managers face increasing pressure to provide quantitative evaluations of the impacts of current human activities on aquatic ecosystems.

Land base managers face increasing pressure to provide quantitative evaluations of the impacts of current human activities on aquatic ecosystems (Minns *et al.* 1996, White 1996, Angermeier 2000, Muotka and Laasonen 2002). Understanding the effects of human activities on ecological communities requires a detailed understanding of how populations and communities are distributed, including the environmental gradients of these patterns, and the extent to which anthropogenic impacts alter such patterns (e.g., Davies *et al.* 2000, Joy and Death 2000, Austin 2002, Joy and Death 2002, Oberdorff *et al.* 2002). In fact, the ability to distinguish between natural and human-induced variation in biological communities is a fundamental component of detecting ecological impairment (Davies *et al.* 2000, Hawkins *et al.* 2002, Oberdorff *et al.* 2002).

Resource managers also face enormous challenges when forecasting the ecological risks of proposed industrial activities and the extent to which current impacts may vary through time.

In the absence of an appropriate level of scientific information, resource managers often adopt a precautionary approach when reviewing proposed industrial activities.

Resource managers also face enormous challenges when forecasting the ecological risks of proposed industrial activities and the extent to which current impacts may vary through time. While the majority of proposed industrial activities require government approval including a public consultation process, the potential ecological impacts of many activities are poorly known. This paucity of scientifically rigorous information often results in a precautionary approach when reviewing proposed industrial activities. While adopting a precautionary approach may appear to be a valid approach, it is also problematic for at least two reasons. First, the precautionary approach typically results in a time-consuming project review process that greatly increases project costs. Second, the precautionary approach typically results in modification, but seldom rejection, of industrial activities. Thus, the level of watershed disturbance from single industrial activities typically increases, albeit at a low rate. Because the ecological effects of multiple industrial activities are not known, the precautionary approach does not guarantee that ecological resources are being conserved. In fact, our results indicating significant and negative relations between the presence of bull trout and percent forest harvesting (Kakwa River Basin) and stream crossing density (Simonette River Basin) challenges the notion that the precautionary approach is sufficient to conserve stream fish communities.

We suggest that the precautionary approach to project assessments should be viewed as an interim management measure during which time financial resources are invested to better understand the ecological impacts of industrial activities.

We suggest that the precautionary approach to project assessments should be viewed as an interim management measure during which time financial resources are invested to better understand the ecological impacts of industrial activities. Such information could be obtained from empirical models derived from: 1) long-term large-scale (e.g., watershed-scale) experiments and/or 2) large-scale biological monitoring programs that quantity dose–response relationships between industrial activities and ecological communities.

5.8 Science-Based Management: The Role of Improved Communication between Resource Managers and Researchers

In the present study we used the three empirical approaches of logistic regression, linear regression (simple and multiple regression) and the reference-condition approach to assess the cumulative effects of watershed disturbances on fish communities in two northern watersheds.

While the fundamentals of what these tests attempt to do is relatively simple, there are inherent difficulties in transferring study results into changes to resource management practices.

We suggest that the increased use of science in resource management (i.e., the science-based approach) represents a new paradigm and will require changes in how the research community interacts with the resource management community.

Several recent studies have called for an increased commitment to science-based management as a means to address large-scale environmental problems (e.g., Pringle 2000). In fact, the need for resource management to be based on the best scientific information available is the basis for the formation or continued support of several large-scale research initiatives in forested regions of Canada (e.g., Northern Rivers Basins Study, Sustainable Forest Management Network Centres of Excellence, Canadian Model Forest Association). While many stakeholders acknowledge that rigorous scientific information is required, the increased level of overlap between research and resource management requires an improved level of understanding between these and other stakeholders. The increased focus on applied research has resulted in a plethora of studies that quantify or forecast environmental impacts by linking changes in biological data with chemical or physical habitat data using relatively complicated empirical models (e.g., Ormerod and Edwards 1987, Wichert and Rapport 1998, Steedman 1988, Reynoldson *et al.* 1995, Reynoldson *et al.* 1997, Bailey *et al.* 1998, Harris and Silveira 1999, Davies *et al.* 2000, Scrimgeour and Chambers 2000, Joy and Death 2002, Oberdorff *et al.* 2002).

In the present study we used the three empirical approaches of logistic regression, linear regression (simple and multiple regression) and the reference-condition approach to assess the cumulative effects of watershed disturbances on fish communities in two northern watersheds. While the fundamentals of what these tests do is relatively simple, there are inherent difficulties in converting study results into positive improvements in resource management practices. We suggest that the increased use of science in resource management (i.e., the science-based approach) represents a new paradigm and that will require changes in how professionals within the research community interact with those in the resource management community. If the science-based approach to resource management is to be broadened, professionals will need to change how they allocate their time.

5.9 Summary of Key Findings

5.9.1 Fish communities in the Kakwa and Simonette River basins

- Analyses of fish communities from a total of 528 stream reaches revealed marked differences in fish communities between the Kakwa and Simonette River basins. The Kakwa watershed supports 9 species of fish comprising representatives from five familial groups whereas the Simonette River Basin supports 20 species of fish from nine Families.

- In the Kakwa River Basin, bull trout and sculpin were the most frequently occurring species followed by Arctic grayling, mountain whitefish and rainbow trout. Longnose sucker, longnose dace, burbot and white sucker occurred at relatively few sites.

- In the Simonette River Basin, sculpin, lake chub, bull trout, and white sucker were the most frequently occurring species followed by Arctic grayling, mountain whitefish, longnose sucker, northern redbelly dace, longnose dace, pearl dace, brook stickleback and redside shiner. Trout-perch, burbot, emerald shiner, finescale dace, flathead chub, northern pike and largescale sucker occurred at relatively few sites.

- Based on density estimates, fish communities in the Kakwa were numerically dominated by sculpin, rainbow trout, bull trout and Arctic grayling. When combined, these four species accounted for the vast majority of all fish encountered. In contrast, northern redbelly dace, sculpin, lake chub, white sucker, brook stickleback and pearl dace were the predominant species in the Simonette River Basin. Overall density of fish in Kakwa watershed was about four-fold lower than that in the Simonette watershed

5.9.2 Is the presence of fish affected by watershed disturbances?

- Logistic regression indicated that the presence of fish, game fish and individual species were moderately to highly predictable based on watershed area, stream width, elevation and to a lesser extent reach slope, size composition of the substratum and stream bank width.

- At the stream reach scale, the occurrence of fish in both the Kakwa and Simonette River basins was strongly affected by stream size (i.e., order) and to a lesser extent watershed type and in both watersheds increased with stream order.

- With only two exceptions, the presence of fish, game fish and individual species were unrelated to watershed disturbances arising from the cumulative effects of industrial activities.

- The two exceptions are the negative relations between bull trout presence and: i) the cumulative percent watershed disturbance in the Kakwa River Basin and ii) cumulative density of stream crossings in the Simonette watershed

5.9.3 Is fish density and biomass affected by watershed disturbances?

- Regression analyses generally showed that total fish density and density of the predominant species in the Kakwa watershed were primarily related to stream wetted width (i.e., the width of the water surface), elevation and reach slope. Fish density was generally highest in small streams or those at high elevation and decreased with increasing width or lower elevation. In general regression models explained relatively little of the overall variance in total density and density of the most abundant species and non-linear models did not typically explain appreciably more variance than linear models.

- With some exceptions, our analyses generally showed that total fish density and biomass and density and biomass of the numerically dominant species and species groups were unrelated or poorly related to watershed disturbances attributes including harvesting, stream crossing attributes and their underlying attributes (e.g., percent of the watershed disturbed by roads, harvest blocks, seismic lines, pipelines, and stream crossings by roads, seismic lines, power lines and pipe lines).

- In the Kakwa River Basin, the notable exceptions to these findings were the positive relationships between: i) total fish density and percent watershed disturbance and, ii) density of sculpin and percent watershed disturbance.

- In the Simonette watershed, the notable exceptions were a): the positive relationship between: i) total fish density and stream crossing density, ii) density of dace and density of stream crossings, iii) total biomass and percent watershed disturbance, iv) biomass of sculpin and seismic line density, and iv) biomass of shiner and crossing density.

- While we report some statistically significant relations between fish density and biomass and watershed disturbance attributes, the majority of these relations did not explain substantial amounts of variance in fish density or biomass.

5.9.4 To what extent is fish community structure affected by watershed disturbances?

- We quantified the cumulative effects of watershed disturbances on fish community structure in the Kakwa and Simonette River basins using a reference condition approach. This approach evaluates the extent to which potentially impacted sites contain fish assemblages predicted by relationships between fish community structure and habitat variables derived from reference (i.e., least-impacted) sites. If the empirical model (discriminant function model) derived from the reference (i.e., least-impacted) sites also explains similar amounts of variation in the fish assemblage membership, then it is assumed that impacts are not detectable.

Kakwa River Basin

- In the Kakwa River Basin, cluster analyses of 62 reference sites (i.e., least disturbed sites) using percent composition data identified three discrete fish assemblages. Assemblage 1 consisted primarily of bull trout, assemblage 2 was dominated by sculpin whereas mountain whitefish, and to a lesser extent rainbow trout, bull trout, Arctic grayling, and sculpin dominated assemblage 3.

- Discriminant function analyses were used to determine linkages between the three fish assemblage types and habitat variables. Results of these analyses showed that stream wetted width, stream reach slope, site elevation and percent small gravel were moderately powerful discriminators among the three assemblage types and had an overall classification success of 71.0%.

- Stream reaches supporting fish assemblage 1 were typically located at high elevations, were relatively narrow, with high reach slope and stream beds with low amounts of small gravel. Reaches supporting fish assemblage 2 were typically located at lower elevations, with broader stream channels, and higher amounts of small gravels. Stream reaches supporting fish assemblage 3 were typically located at low elevations, were relatively broad, with moderate reach slope and stream beds that contained low amounts of small gravel.

- We evaluated larger patterns in fish communities initially by completing cluster analyses of least-impacted sites and potentially impacted sites. In addition to identifying the three fish assemblages from the reference sites, clustering also identified two more assemblage types. The first new assemblage consisted primarily of sites dominated by Arctic grayling, and to lesser extent bull trout mountain whitefish and rainbow trout. The second new assemblage type was numerically dominated by rainbow trout which likely originated as fish stocked into the adjacent Musreau Lake.

- We used discriminant function analyses to quantify how well fish communities at the potentially impacted sites (i.e., test sites where watershed disturbance ranged from 10 to 43%) were predictable based on the four watershed variables of stream wetted width, reach slope, elevation and percent small gravel within the substratum.

- Our results showed that the four habitat variables of stream wetted width, reach slope, elevation and percent small gravel the, we poor predictors of fish assemblage type of the potentially impacted test sites and overall had a classification success of only 50%.

- These results indicate that the discriminant function model developed from the least-impacted reference sites was a poor predictor of fish assemblage structure in the potentially impacted sites and that fish communities in the Kakwa River Basin are affected by the cumulative impacts of current levels of industrial activities.

<u>Simonette River Basin</u>

- In the Simonette River Basin, cluster analyses of 106 reference (i.e., least impacted) sites identified five relatively discrete fish assemblages. Assemblage 1 consisted primarily of white sucker and to a lesser extent finescale dace, lake chub, trout-perch and redside shiner whereas assemblage 2 was dominated by lake chub and to a lesser extent white sucker, longnose sucker, sculpin and finescale dace. In contrast, northern redbelly dace, lake chub, pearl dace dominated assemblage 3. Fish communities comprising assemblages 4 and 5 were numerically dominated by mountain whitefish, pearl dace and Arctic grayling (assemblage 4) and sculpin (assemblage 5).

- As for the Kakwa River Basin, discriminant function analyses were used to determine linkages between the five fish assemblages and habitat variables. Results of these analyses showed that stream wetted width, stream reach slope, site elevation and percent of fine materials in the substratum were moderately powerful discriminators among the five fish assemblages and had an overall classification success of 71.1%.

- Stream reaches supporting fish assemblage 1 were found in moderately large, low elevation reaches with moderately large watershed areas and characterized by low reach slope, with substrata dominated by fine sediments. Reaches supporting assemblage 2 were moderately small, low gradient systems located at intermediate elevations whereas assemblage 3, were typically located in small, low elevation reaches. Fish communities comprising assemblages 4 and 5 were located at high elevations with moderate reach slope but differed in stream size and the percent of fine materials within the stream bottom. Streams comprising assemblage 4 were typically large with a predominance of fine materials in the river bed compared with the small streams comprising assemblage 5 where fine sediments were relatively rare.

- We used discriminant function analyses to quantify how well fish communities at the potentially impacted sites (i.e., test sites where watershed disturbance ranged from 21 to 61.8%) were predictable based on the four watershed variables of stream wetted width, reach slope, elevation and percent fine materials within the substratum.

- Our results showed that the four habitat variables of wetted width, reach slope, elevation and percent fine materials within the substratum were moderately poor predictors of fish assemblage type and overall had a classification success of only 57.1%.

- These results indicate that the discriminant function model developed from the least-impacted reference sites was a poor predictor of fish assemblage structure in the potentially impacted sites and that fish communities in the Simonette River Basin are affected by the cumulative impacts of current levels of industrial activities.

5.10 Management Implications and recommendations

5.10.1 Towards an improved understanding of the cumulative effects of human-induced activities of stream fish communities

Rationale

The expansion of Alberta's forest industry since the mid-1980's combined with conversion of forest lands to agriculture and increased oil and gas activities has raised concerns about the ability of ecological sustainability of stream fish communities in northern Alberta. These industrial activities have the potential to affect stream fish communities by influencing the quantity and quality of habitat for stream fishes. Our study showed that current levels of industrial activity have detectable cumulative effects on stream fish communities. Such effects were linked with forest harvesting and linear features (e.g., roads and seismic lines) that intercept streams.

Recommendation

If management of the boreal forest is to be based on ecological in addition to economic, social, political and cultural considerations, a more detailed understanding of the cumulative effects of multiple industrial activities on fish communities is required. Our pursuit of a knowledge base sufficient to understand the cumulative impacts of industrial activities on stream fish communities is in its infancy and this lack of knowledge challenges our ability to manage resources in a sustainable fashion. As a result we recommend an increased commitment to understand the cumulative effects of industrial activities on stream fish in forested regions of Alberta.

5.10.2 Development and implementation of stream fish monitoring program

Rationale

In Alberta, fish communities and aquatic environments are protected under both provincial and federal legislation. For instance, consequences of watershed activities on aquatic environments and the biological diversity that they support are considered within Provincial Acts and Regulations (e.g., The *Water Act*, Timber harvest and planning operational Ground Rules [Anonymous 1994]). These provisions include operating ground rules for forest harvesting practices, and provincial codes of practice for: 1) watercourse crossings, 2) pipeline and telecommunication line crossings and 3) temporary diversion of water for hydrostatic testing. Operating ground rules define a set of forest harvesting practices including the protection of 30 and 60 m buffers adjacent to small and large permanent streams and practices that reduce inputs of organic matter into stream and avoidance of stream channels.

Fish communities and aquatic habitats are also protected under habitat protection provisions (Section 35 [1]) of the *Federal Fisheries Act* which prohibits works or undertakings that result in the harmful alteration, disruption or destruction of fish habitat, while Section 35 [2], allows for authorization by the Minister, or under regulation, of harmful alteration, disruption or destruction of fish habitat. While these provisions are intended to protect stream fish populations, a scientifically rigorous program to monitor the consequences of industrial activities on stream fish communities does not exist.

<u>Recommendation</u>

The absence of effective monitoring programs compromises our ability to manage stream fish communities. Rigorous monitoring programs are required to: i) understand current trends in fish populations, ii) evaluate the ecological effects of anthropogenic and natural disturbances on stream fish communities and iii) evaluate the effectiveness of restoration measures, iv) to critically assess the effectiveness of current watershed management practices. As such, we recommend the allocation of resources to develop and implement a scientifically rigorous stream monitoring program in northern Alberta. This monitoring program should also include evaluations of focal watersheds, such as the Prairie Creek sub-basin, where ecological impacts have been detected and where information on recovery of fish communities is required.

5.10.3 The role of empirical modeling in stream fish management

<u>Rationale</u>

The use of empirical models to quantify the impacts of human activities on aquatic ecosystems has expanded rapidly over the last 15 years and biological assessments using multi-metric or multivariate approach are now common place. These statistical methods represents important tools that can assist with the management of stream fish communities.

<u>Recommendation</u>

We suggest that the increased use of empirical modeling would assist with the management of stream fish communities by: i) providing techniques to understand large-scale patterns in fish communities, ii) gaining insights to potential cause-effect relationships, iii) evaluating environmental impacts, iv) quantifying temporal variance in fish communities, v) identify fish community types and the environmental gradients that may drive them, and vi) monitoring the effectiveness of current management actions. As a result, we recommend the increased use of empirical and other modeling tools to assist with the management of stream fish communities.

6.0 LITERATURE CITED

Abbruzzese, B., Leibowitz, S.G. 1997. Environmental auditing a synoptic approach for assessing cumulative impacts to wetlands. Environmental Management **21**: 457-475.

Angermeier, P.L. 2000. The natural imperative for biological conservation. Conservation Biology **14**: 373-381.

Anonymous. 1994. Alberta timber harvest planning and operating ground rules. Alberta Environmental Protection, Land and Forest Service, Edmonton, Alberta, Canada.

Andrus, C, W., Long, B.A., Froehlich, H.A. 1988. Woody debris and its contribution to pool formation in a coastal stream 50 years after logging. Canadian Journal of Fisheries and Aquatic Sciences **45**: 2080-2086.

Austin, M.P. 2002. Spatial prediction of species distribution: an interface between ecological theory and statistical modelling. Ecological Modelling **157**: 101-118.

Argent, D., Flebbe, P.A. 1999. Fine sediment effects on brook tout eggs in laboratory streams. Fisheries Research **39**: 253-262.

Bailey, R. C., M. G. Kennedy, M. Z. Dervish, Taylor, R.M. 1998. Biological assessment of freshwater ecosystems using a reference condition approach: comparing predicted and actual benthic invertebrate communities in Yukon streams. Freshwater Biology **39**: 765-774.

Barbour, M., Gerritsen, T.J., Griffith, G.E., Frydenborg, R., McCarron, E., White, J. S., Bastian M. L. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. Journal of the North American Benthological Society **15**: 185-211.

Barton, D.R., Taylor, W.D., Biette, R.M. 1985. Dimensions of riparian buffer strips required to maintain trout habitat in southern Ontario streams. North American Journal of Fisheries Management **5**: 364-378.

Bates, K., Powers, P. 1999. Upstream passage of juvenile coho salmon through roughened culverts. Page 192-202. In: M. Jungworth, S. Schmutz and S. Weiss (Editors) Fish migrations and fish bypasses. Blackwell Publishing. Blackwell Science, Oxfordshire, UK.

Baxter, C.V., Frissell, C.A., Hauer, F.R. 1999. Geomorphology, logging roads, and the distribution of bull trout spawning in a forested river basin: Implications for management and conservation. Transactions of the American Fisheries Society **128**: 854-867.

Baxter, C.V., Hauer, F.R. 2000. Geomorphology, hyporheic exchange, and selection of spawning habitat by bull trout (*Salvelinus confluentus*). Canadian Journal of Fisheries and Aquatic Sciences **57**: 1470-1481.

Beaudoin, C.P., Prepas, E.E., Tonn, W.M., Wassenaar, L.I., Kotak, B.G. 2001. A stable carbon and nitrogen isotope study of lake food webs in Canada's Boreal Plain. Freshwater Biology **46**: 465-477.

Beechie, T.J., Sibley, T.H. 1997. Relationships between channel characteristics, woody debris, and fish habitat in northwestern Washington streams. Transactions of the American Fisheries Society **126**: 217-229.

Berkman, H.E., Rabeni, C.F., Boyle, T.P. 1986. Biomonitors of stream quality in agricultural areas: Fish versus invertebrates. Environmental Management **10**: 413-419.

Berkman, H.E., Rabeni, C.F. 1987. Effect of siltation on stream communities. Environmental Biology of Fishes **18**: 285-294.

Bevenger, G.S., King, R.M. 1995. A pebble count procedure for assessing watershed cumulative effects. United States Department of Agriculture, Forest Service Research Paper RM-RP-319. Fort Collins, Colorado, United States Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Binns, A.A., Remmick, R. 1994. Response of Bonneville cutthroat trout and their habitat to drainage-wide habitat management at Huff Creek, Wyoming. North American Journal of Fisheries Management **14**: 669-680.

Bisson, P.A., Quinn, T.P., Reeves, G.H., Gregory, S.V. 1992. Best management practices, cumulative effects, and long-term trends in fish abundance in the Pacific Northwest river systems. Pages 189-231 *In:* Watershed Management: Balancing sustainability and environmental change. R.J. Naiman (Editor). Springer-Verlag New York, Inc. USA.

Boisclair, D. 2001. Fish habitat modeling: from conceptual framework to functional tools. Canadian Journal of Fisheries and Aquatic Sciences **58**: 1-9.

Byorth, P.A., Magee, J.P. 1998. Competitive interactions between Arctic grayling and brook trout in the Big Hole River drainage, Montana. Transactions of the American Fisheries Society **127**: 921-931.

Campbell, I.C., Doeg, T.J. 1989. Impact f timber harvesting and production on stream: a review. Australian Journal of Marine and Freshwater Research **40**: 519-539.

Carignan, R., Steedman, R.J. 2000 Impacts of major watershed perturbations on aquatic ecosystems. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2): 1-4.

Carignan, R., P. D'Arcy, and S. Lamontagne. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2):105-117.

Carignan, R., D’Arcy, P., Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in boreal shield lakes. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2): 105-117.

Carlander, K.D. 1969. Handbook of freshwater fishery biology. Life history data on freshwater fishes of the United States and Canada, exclusive of the Perciformes. Iowa State. Press, Ames, Iowa, USA.

Chambers, P.A., Scrimgeour, G.J., Pietroniro, A. 1997. Winter oxygen conditions in ice-covered rivers: the impact of pulp mill and municipal effluents. Canadian Journal of Fisheries and Aquatic Sciences **54**: 2796-2806.

Coats, R.N., Miller, T.O. 1981. Cumulative silvicultural impacts on watersheds: a hydrologic and regulatory dilemma. Environmental Management **5**: 147-160.

Cocklin, C., Parker, S., Hay, J. 1992a. Notes on cumulative environmental changes I: concepts and issues. Journal of Environmental Management **35**: 41-49.

Cocklin, C., Parker, S. ., Hay, J. 1992b. Notes on cumulative environmental change II: a contribution to methodology. Journal of Environmental Management **35**: 51-67.

Contant, C.K., Wiggins, L.L. 1991. Defining and analyzing cumulative environmental impacts. Environmental Impact Assessment Review **11**: 297-309.

Crossman, E.J., McAllister, D.E. 1986. Zoogeography of freshwater fishes of the Hudson Bay drainage, Ungava Bay and the Arctic Archipelago. In: *The Zoogeography of North American Freshwater Fishes*. C.H. Hocutt and E.O. Wiley (Editors), John Wiley and Sons, New York, USA.

Davies, N.M., Norris, R.H., Thoms, M. 2000. Prediction and assessment of local stream habitat features using large-scale catchment characteristics. Freshwater Biology **45**: 343-369.

Davies, P.E., Nelson M. 1994. Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research **45**: 1289-1305.

Deegan, L.A., Peterson, B.J. 1992. Whole-river fertilization stimulates fish production in an Arctic tundra river. Canadian Journal of Fisheries and Aquatic Sciences **49**: 1890-1901.

Dixon, J., Montz, B.E. 1995. From concept to practice: implementing cumulative impact assessment in New Zealand. Environmental Management **19**: 445-456.

Dunham, J.B., Rieman, B.E. 1999. Metapopulation structure of bull trout: Influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications **9**: 642-655.

Eaglin, G.S., Hubert, W.A. 1993. Effects of logging and roads on substrate and trout in streams of the Medicine Bow National Forest, Wyoming. North American Journal of Fisheries Management 13: 844-846.

Everest, R.H., Beschta, R.L., Scrivener, J.C., Koski, K.V., Sedell, J.R., Cederholm, C.J. 1987. Fine sediment and salmonid production: A paradox. In: Streamside management: Forestry and Fishery Interactions, E.O. Salo & T.W. Cundy (Editors). Contribution No. 57 Institute of Forest Resources, University of Washington, Seattle, Washington Pp 98-142.

Fausch, K.D., Hawkes, C.L., Parsons, M.G. 1988. Models that predict standing crop of stream fish from habitat variables. United States Department of Agriculture. Forest Service. Pacific Northwest Research Station. General Technical report PNW-213. Portland, Oregon, USA.

Feller, M.C. 1981. Effects of clearcutting and slashburning on stream temperature in southern British Columbia. Water Resources Bulletin **17**: 863-867.

Findlay, C.S., Bourdages, J. 1999. Response time of wetland biodiversity to road construction on adjacent lands. Conservation Biology **14**: 86-94

Fisheries and Oceans. 1986. The Department of Fisheries and Oceans policy for the management of fish habitat. Department of Fisheries and Oceans Report 3209. Communications Directorate, Department of Fisheries and Oceans, Ottawa, Ontario, Canada.

Fisheries and Oceans. 1991. Canada's fish habitat law. Department of Fisheries and Oceans Report 4438. Communications Directorate, Department of Fisheries and Oceans, Ottawa, Ontario, Canada.

Fitzgerald, D.G., Kott, E., Lanno, R.P., Dixon, D.G. 1998. A quarter century of change in the fish communities of three small streams modified by anthropogenic activities. Journal of Aquatic Ecosystem Stress and Recovery **6**: 111-127.

Fraley, J.J., Shepard, B.B. 1989. Life history, ecology and population status of migratory bull trout (*Salvelinus confluentus*) in the Flathead Lake river system, Montana. Northwest Science **63**: 133-143.

Fraley, J., Wever, T., Vashro, J. 1989. Cumulative effects of human activities on bull trout (*Salvelinus confluentus*) in the upper Flathead drainage, Montana. Headwaters Hydrology **June**: 111-120.

Frenzel, S.A., Swanson, R.B. 1996. Relations of fish community composition to environmental variables in streams of Central Nebraska, USA. Environmental Management **20**: 689-705.

Furniss, M.J., Roelofs, T.D., Yee, C.S. 1991. Road construction and maintenance. Page 297-323. In: W.R. Meehan (Ed). Influence of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication **19**. Bethesda, Maryland, USA.

Garcia, E., Carignan, R. 2000. Mercury concentrations in northern pike (*Esox lucius*) from boreal lakes with logged, burned, or undisturbed catchments. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2): 129-135.

Garman, G.C., Moring, J.R. Diet and annual production of two boreal river fishes following clearcut logging. Environmental Biology of Fishes **36**: 301-311.

Gibbons, D.R., Salo, E.O. 1973. An annotated bibliography of the effects of logging on fish of the western United States and Canada. United States Department of Agriculture, Forest Service General Technical Report PNW-10. 145 pp.

Glova, G.J., Sagar, P.M. 1994. Comparison of fish and macroinvertebrate standing stock in relation to riparian willows (*Salix* spp.) in three New Zealand streams. New Zealand Journal of Marine and Freshwater Research **28**: 255-266.

Golder Associates 2000. Weyerhaeuser Canada Ltd., Grande Prairie Environment Effects Monitoring – Cycle 2. Report submitted by Weyerhaeuser Canada Ltd. to Environment Canada.

Grossman, G.D., Dowd, J.F., Crawford, M. 1990. Assemblage stability in stream fishes: A review. Environmental Management **14**: 661-671.

Ham, K.D., Pearsons, T.N. 2000. Can reduced salmonid population abundance be detected in time to time to limit management impacts? Canadian Journal of Fisheries and Aquatic Sciences **57**: 17-24.

Harris, J.H., Silveira, R. 1999. Large-scale assessments of river health using an index of biotic integrity with low-diversity fish communities. Freshwater Biology **41**: 235-252.

Hartman, G.F., Scrivener, J.C., Miles, M.J. 1996. Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, and their implication for restoring fish habitat. Canadian Journal of Fisheries and Aquatic Sciences **53**: 237-251.

Hauer, F.R., Poole, G.C., Gangemi, J.T., Baxter, C.V. 1999. Large woody debris in bull trout (*Salvelinus confluentus*) spawning of logged and wilderness watersheds in northwest Montana. Canadian Journal of Fisheries and Aquatic Sciences **56**: 915-924.

Hawkins, C.P., Murphy, M.L., Anderson, N.H., Wilzbach, M.A. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the Northwestern United States. Canadian Journal of Fisheries and Aquatic Sciences **40**: 1173-1185.

Hawkins, C.P., Norris, R.H., Hogue, J.N., Feminella, J.W. 2000. Development and evaluation of predictive models for measuring the biological integrity of streams. Ecological Applications **10**: 1456-1477.

Hemond, H.F., Benoit, J. 1988. Cumulative impacts on water quality functions of wetlands. Environmental Management **12**: 639-653.

Hewlett, J.D., Fortson, J.C. 1992. Stream temperature under an inadequate buffer strip in the southeast Piedmont. Water Resources Bulletin **18**: 983-988.

Hicks, B.J., Beschta, R.L., Harr, R.D. 1991a. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. Water Resources Bulletin **27**: 217-226.

Hicks, B.J., Hall, J.D., Bisson, P.A., Sedell, J.R. 1991b. Page 483-518. In: W.R. Meehan (Editor). Influence of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19. Bethesda, Maryland, USA.

House, R.A. 1995. Temporal variation in abundance of an isolated population of cutthroat trout in western Oregon. North American Journal of Fisheries Management **15**: 33-41.

Hughes, R.M., Gammon, J.R. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. Transactions of the American Fisheries Society **116**: 196-209.
Hughes, R.M., Rexstad, E., Bond, C.E. 1987. The relationship of aquatic ecoregions, river basins and physiographic provinces to the ichthyogeographic regions of Oregon. Copeia **2**: 423-432.
Hvenegaard, P.J. 1998. Cooperative fisheries inventory program: final report. Technical report produced by the Alberta Conservation Association for the Department of Fisheries and Oceans, Winnipeg, Manitoba. Alberta Conservation Association, Northwest Boreal Region, Peace River, Alberta, Canada.
Jackson, D.A., Harvey, H.H. 1989. Biogeographical associations in fish associations: local vs. regional processes. Ecology **70**: 1472-1484.
Johnson, C.A., Detenbeck, N.E., Bonde, J.P., Niemi, G.J. 1988. Geographic information systems for cumulative impact assessment. Photogrammetric Engineering and Remote Sensing **54**: 1609-1615.
Johnston, N.T., Perrin, C.J., Slaney, P.A., Ward, B.R. 1990. Increased juvenile salmonid growth by whole-river fertilization. Canadian Journal of Fisheries and Aquatic Sciences **47**: 862-872.
Jones, E.B. III., Helfman, G.S., Harper, J.O., Bolstad, P.V. 1999. Effects of riparian forest removal on fish assemblages in southern Appalachian streams. Conservation Biology **13**: 1454-1465.
Joy, M. 1996. Handling Uncertainty in GIS and Environmental Models: an application in forest management. M.Sc. Thesis, Department of Geography, University of British Columbia, Vancouver, British Columbia, Canada.
Joy, M.K., Death, R.G. 2000. Development and application of a predictive model of riverine fish community assemblages in the Taranaki region of the North Island of New Zealand. New Zealand Journal of Marine and Freshwater Research **34**: 241-252.
Joy, M.K., Death, R.G. 2002. Predictive modelling of freshwater fish as a biomonitoring tool in New Zealand. Freshwater Biology **47**: 2261-2275.
Karr, J.R., Toth, L.A., Dudley, D.R. 1985. Fish communities of midwestern rivers: a history of degradation. BioScience **35**: 90-95.
Karr, J. R., Chu, E.W. 1999. Restoring Life In Running Waters: Better Biological Monitoring. Island Press, Washington, D.C., USA.
Kilgour, B.W., Barton, D.R. 1999. Associations between stream fish and benthos across environmental gradients in southern Ontario, Canada. Freshwater Biology **41**: 553-566.
Kreutzweiser, D.P., Capell, S.S. 2001. Fine sediment deposition in streams after selective forest harvesting without riparian buffers. Canadian Journal of Forest Research **31**: 2134-2142.
Kruse, C.G., Hubert, W.A., Rahel, F.J. 1998. Single-pass electroshocking predicts trout abundance in Mountain streams with sparse habitat. North American Journal of Fisheries Management **18**: 940-946.

Lanka, R.P., Hubert, W.A., Wesche, T.A. 1987. Relations of geomorphology to stream habitat and trout sanding crop in small Rocky Mountain streams. Transactions of the American Fisheries Society **116**: 21-28.

Larsen, D.P., Omernik, J.M., Hughes, R.M., Rohm, C.M., Whittier, T.R., Kinney, A.J., Gallant, A.L., Dudley, D.R. 1986. Correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. Environmental Management **10**: 815-828.

Legendre, P., Legendre, L. 1998. Numerical ecology. Second English edition. Elsevier Scientific Publishing, Amsterdam.

Li, H.W., Lamberti, G.A., Pearsons, T.N., Tait, C.K., Li, J.L., Buckhouse, J.C. 1994. Cumulative effects of riparian disturbance along high desert trout streams of the John Day basin, Oregon. Transactions of the American Fisheries Society **123**: 627-640.

Lindsey, C.C., McPhail, J.D. 1986. Zoogeography of fishes of the Yukon and Mackenzie basins. In: *The Zoogeography of North American Freshwater Fishes*. C.H. Hocutt and E. O. Wiley (Editors). John Wiley and Sons, New York, New York, USA.

Lyons, J., Trimble, S.W., Paine, L.K. 2000. Grass versus trees: managing riparian areas to benefit streams of North America. Journal of the American Water Resources Association **36**: 919-930.

MacDonald, L.H. 2000. Evaluating and managing cumulative effects: process and constraints. Environmental Management **26**: 299-315.

Mangel, M., Smith, P.E. 1990. Presence-absence sampling for fisheries management. Canadian Journal of Fisheries and Aquatic Sciences **47**: 1875-1887.

Marcus, M.D., Young, M.K., Noel, L.E., Mullan, B.A. 1990. Salmonid-habitat relationships in the western United States: A review and indexed bibliography. Unite States Department of Agriculture, Forest Service. Rocky Mountain Forest and range Experiment Station. General Technical Report RM-188, Laramie, Wyoming, USA.

Maret, T.R., Robinson, C.T., Minshall, G.W. 1997. Fish assemblages and environmental correlates in least-disturbed streams of the Upper Snake River Basin. Transactions of the American Fisheries Society **126**: 200-216.

Martin-Smith, K.M. 1998. Effects of disturbance caused by selective timber extraction on fish communities in Sabah, Malaysia. Environmental Biology of Fishes **53**: 155-167.

McEachern P., Prepas, E.E., Gibson, J.J., Dinsmore, W.P. 2000. Forest fire induced impacts on phosphorus, nitrogen, and chlorophyll *a* concentrations in boreal subarctic lakes of northern Alberta. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2): 73-81.

McCune, B., Mefford, M.J. 1999. PC-ORD. Multivariate analysis of ecological data. Version 4. MJM Software Design, Gleneden Beach, Oregon, USA.

McIntosh, B.A., Sedell, J.R., Thurow, R.F., Clarke, S.E., Chandler, G.L. 2000. Historical changes in pool habitats in the Columbia River Basin. Ecological Applications **10**: 1478-1496.

Meehan,W.R. 1991. (Editor) Influence s of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication No. 19.

Minns C., Kelso, J.R.M., Randall, R.G. 1996. Detecting the response of fish to habitat alterations in freshwater ecosystems. Canadian Journal of fisheries and Aquatic Sciences **53** (Supplement 1): 403-414.

Mitro, M.G., Zale, A.V. Predicting fish abundance using single-pass removal sampling. Canadian Journal of Fisheries and Aquatic Sciences **57**: 951-961.

Murphy, M.L., Hall, J.D. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences **38**: 137-145.

Muotka, T., Laasonen, P. 2002. Ecosystem recovery in restored headwater streams: the role of enhanced leaf breakdown. Journal of Applied Ecology **39**: 145-156.

Nelson, L.A., Paetz M.J. 1992. The fishes of Alberta. Second Edition. University of Alberta Press, Edmonton, Alberta, Canada.

Oberdorff, T., Pont, D., Huguny, B., Porcher, J-P. 2002. Development and validation of a fish-based index for the assessment of ‘river health’ in France. Freshwater Biology **47**: 1720-1734.

Olden, J.D., Jackson, D.J. 2002. A comparison of statistical approaches for modeling fish species distributions. Freshwater Biology **47**: 1976-1995.

Ormerod, S.J., Edwards, R.W. 1987. The ordination and classification of macroinvertebrate assemblages in the catchment of the Wye River in relation to environmental factors. Freshwater Biology **17**: 533-546.

Parker, S., Cocklin, C. 1993. The use of geographical information systems for cumulative environmental effects assessment. Computers, Environmental and Urban Systems **17**: 393-407.

Parsons, M., Norris, R.H. 1996. The effect of habitat-specific sampling on biological assessment of water quality using a predictive model. Freshwater Biology **36**: 419-434.

Paul, A.J., Post, J.R. 2001. Spatial distribution of native and nonnative salmonids in stream of the eastern slopes of the Canadian Rocky Mountains. Transactions of the American Fisheries Society **130**: 417-430.

PC-ORD 2000. PC-ORD for Windows. Multivariate analysis of ecological data. Version 4. MJM Software Design. Gleneden Beach, Oregon, USA.

Platts W.S. McHenry, M.L. 1988. Density and biomass of trout and char in western streams. United States Department of Agriculture, Forest Service. Intermountain Research Station. General Technical Report INT-241. Ogden, Utah, USA.

Platts, W.S., Torquemada, R.J., McHenry, M.L., Graham. C.K. 1989. Changes in salmon spawning and rearing habitat from increased delivery of fine sediment to the South Fork Salmon River, Idaho. Transactions of the American Fisheries Society **118**: 274-283.

Preston, E.M., Bedford, B.L. 1988. Evaluating cumulative effects of wetland functions: a conceptual overview and generic framework. Environmental Management **12**: 565-583.
Rahel, F.J., Nibblelink, P. 1999. Spatial patterns in relation among brown trout (*Salmo trutta*) distribution, summer air temperature, and stream size in Rocky Mountain streams. Canadian Journal of Fisheries and Aquatic Sciences **56**: 43-51.
Reed, J.R. 1977. Stream community response to road construction sediments. Bulletin 97. Published by the Virginia Water Resources Centre, Virginia Polytechnic Institute and State University Blacksburg, Virginia, USA.
Reece, P.F., Reynoldson, T.B., Richardson, J.S., Rosenberg, D.M. 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.
Reynoldson, T.B., Bailey, R.C., Day, K.E., Norris, R.H. 1995. Biological guidelines for freshwater sediment based on BEnthic Assessment of SedimenT (the BEAST) using a multivariate approach for predicting biological state. Australian Journal of Ecology **20**: 198-219.
Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E. 1997. The reference-condition: a comparison of multi-metric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society **16**: 833-852.
Reynoldson, T.B., Rosenberg, D.M., Resh, V.H. 2001. A comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment. Canadian Journal of Fisheries and Aquatic Sciences **58**: 1395-1410.
Rice, R.M. 1980. A perspective on the cumulative effects of logging on streamflow and sedimentation. Pages 32-46. In R.B. Standiford and S.I. Ramacher (Editors) Proceedings of the Cumulative effects of forest management on California watersheds: an assessment. The Department of Forestry and Resource Management and Cooperative Extension. University of California, Berkeley, California, USA.
Rose, K.A. 2000. Why are quantitative relationships between environmental quality and fish populations so elusive? Ecological Applications **10**: 367-385.
Rutherford, D.A., Echelle, A.A., Maughan, O.E. 1992. Drainage-Wide Effects of Timber Harvesting on the Structure of Stream Fish Assemblages in Southeastern Oklahoma. Transactions of the American Fisheries Society **121**: 716-728.
Salo, E.O., Cederholm, C.J. 1980. Cumulative effects of forest management on watersheds – some aquatic considerations. Page 67-78. In R.B. Standiford and S.I. Ramacher (Editors) Proceedings of the Cumulative effects of forest management on California watersheds: an assessment. The Department of Forestry and Resource Management and Cooperative Extension. University of California, Berkeley, California, USA.

SAS Institute Inc. 1987. SAS user's guide: statistics. SAS Institute Inc. Cary North Carolina. U.S.A.

Scarnecchia, D.L., Bergersen, E.P. 1987. Trout production and standing crop in Colorado's small stream, as related to environmental features. North American Journal of Fisheries Management **7**: 315-330.

Schindler, D.W. 1998a. Sustaining aquatic ecosystems in boreal regions. Conservation Ecology [Online] 2(2): 18. Available from the Internet. URL:http//www.consecol.org/vol2/iss2/art18. 27 pp.

Schindler, D.W. 1998b. A dim future for boreal waters and landscapes: cumulative effects of climatic warming, stratospheric ozone depletion, acid precipitation, and other human activities. Bioscience **48**: 157-164.

Schlotzhauer, D.C. 1993. Some issues in using the LOGISTIC procedure for binary logistic regression. Observations **3**: 42-49

Schnackenberg, E.S., MacDonald, L.H. 1998. Detecting cumulative effects on headwater streams in the Routt National Forest, Colorado. Journal of the American Water Resources Association **34**: 1163-1177.

Scott, W.B., Crossman, E.J. 1998. Freshwater fishes of Canada. Galt House Publications, Ontario, Canada.

Scrimgeour, G.J. Chambers, P.A. 2000. Cumulative effects of pulp mill and municipal effluents on epilithic biomass and nutrient limitation in a large northern river ecosystem. Canadian Journal of Fisheries and Aquatic Sciences **57**: 1342-1354.

Scrimgeour, G. J., Tonn, W.M., Paszkowski, C.A., Aku, P.M.K. 2000. Evaluating the effects of forest harvesting on littoral benthic communities within a natural disturbance-based management model. Forest Ecology and Management **126**: 77-86.

Scrivener, J.C., Anderson, B.C. 1984. Logging impacts and some mechanisms that determine the size of spring and summer populations of coho salmon fry (*Oncorhynchus kisutch*) in Carnation Creek. Canadian Journal of Fisheries and Aquatic Sciences **41**: 1097-1105.

Sharma, R., Hilborn, R. 2001. Empirical relationships between watershed characteristics and coho salmon (*Oncorhynchus kisutch*) smolt abundance in 14 western Washington streams. Canadian Journal of Fisheries and Aquatic Sciences **58**: 1453-1463.

Slawski, T.M., Ehlinger, T.J. 1998. Fish habitat in box culverts: management in the Dark&quest. North American Journal of Fisheries Management **18**: 676-685.

Spalling, H., Smit, B. 1995. A conceptual model of cumulative environmental change of agricultural land drainage. Agriculture, Ecosystems and Environment **53**: 99-108.

Steedman, R.J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences **45**: 492-501.

Steedman, R.J., Kushneriuk, R.S. 2000. Effects of experimental clearcut logging on thermal stratification, dissolved oxygen, and lake trout (*Salvelinus namycush*) habitat volume in three small boreal forest lakes. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 1): 82-91.

Steedman, R.J., Tonn, W.M., Paszkowski, C.A., Scrimgeour, G.J. 2003. Forestry and fish in the Boreal Region of Canada. Chapter 20 - In: T.G. Northcote and G.F. Hartman [Editors]. Fishes and Forestry: Worldwide watershed interactions & management. Blackwell Science, Oxford, UK. (In Press).

Stevenson, R.J., Hauer, F.R. 2002. Integrating hydrogeomorphic and index of biotic integrity approaches for environmental assessment of wetlands. Journal of the North American Benthological Society **21**: 502-513.

St-Onge, I., Magnan, P. 2000. Impact of logging and natural fires on fish communities of Laurentian Shield Lakes. Canadian Journal of Fisheries and Aquatic Sciences **57** (Supplement 2):165-174.

Strahler, A.N. 1957. Quantitative analysis of watershed geomorphology. Transactions of the Geophysical Union **38**: 913-920.

Strayer, D.L. 1999. Statistical power of presence-absence data to detect population declines. Conservation Biology **13**: 1034-1038.

Strong, W.L., Leggat, K.R. 1992. Ecoregions of Alberta. Land Information Services Division, Alberta Forestry, Lands and Wildlife, Edmonton, Alberta, Canada.

Thedinga, J.F., Murphy, M.L., Heifetz, J., Koski, K.V., Johnson, S.W. Effects of logging on size and age composition of juvenile coho salmon (*Oncorhynchus kisutch*) and density of presmolts in southeast Alaska streams. Canadian Journal of Fisheries and Aquatic Sciences **46**: 1383-1391.

Tonn, W.M. 1990. Climate change and fish communities: A conceptual framework. Transactions of the American Fisheries Society **119**: 337-352.

Tonn, W.T., Paszkowski, C. A. Scrimgeour, G.J., Aku, P.K.M., Lange, M., Prepas, E.E. 2003. Effects of Forest Fire and Harvesting on Fish Assemblages in Boreal Plains Lakes: A Reference Condition Approach. North American Journal of Fisheries Management (In Press).

Trombulak, S.C., Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology **14**: 18-30.

Warren, M.L. Jr., Pardew, M.G. 1998. Road crossings as barriers to small-stream fish movement. Transactions of the American Fisheries Society **127**: 637-644.

Waters, T.F. 1995. Sediment in Streams: Sources, Biological Effects, and Control. American Fisheries Society Monograph, Bethesda, Maryland, USA.

Watson, G., Hillman, T.W. 1997. Factors affecting the distribution and abundance of bull trout: An investigation at hierarchical scales. North American Journal of Fisheries Management **17**: 237-252.

Weaver, L.A., Garman, G.C. 1994. Urbanization of a watershed and historical changes in a stream fish assemblage. Transactions of the American Fisheries Society **123**: 162-172.

Weaver, T.M., White, R.G. 1985. Coal Creek monitoring study Number III. Montana Cooperative Fisheries Research Unit, Montana State University, Bozeman, Montana, USA.

White, R.J. 1996. Growth and development of North American stream habitat management for fish. Canadian Journal of Fisheries and Aquatic Sciences **53** (Supplement 1): 342-363.

Wichert, G.A., Rapport, D.J. 1998. Fish community structure as a measure of degradation and rehabilitation of riparian systems in an agricultural drainage basin. Environmental Management **22**: 425-443.

Wildhaber, M.L., Allert, A.L., Schmitt, C.J., Tabor, V.M., Mulhern, D., Powell, K.L., Sowa, S.P. 2000. Natural and anthropogenic influences on the distribution of the threatened Neosho Madtom in Midwestern Warmwater stream. Transactions of the American Fisheries Society **129**: 243-261.

Williams, L.R., Taylor, C.M., Warren, M.L. Jr., Clingenpeel, J.A. 2002. large-scale effect of timber harvesting on stream systems in the Ouachita Mountains, Arkansas, USA. Environmental Management **29**: 76-87.

Wright J., Moss, D., Armitage, P.D., Furse, M. T. 1984. A preliminary classification of running-water sites in Great Britain based on macroinvertebrate species and the prediction of community type using environmental data. Freshwater Biology **14**: 221-256.

Wu, J., Adams, R.M., Boggess, W.G. 2000. Cumulative effects and optimal targeting of conservation efforts: steelhead trout habitat enhancement in Oregon. The American Journal of Agricultural Economics **82** (May): 400-413.

Ziemer, R.R., Lewis, J., Rice, R.M., Lisle, T.E. 1991. Modeling the cumulative effects of forest management strategies. Journal of Environmental Quality **20**: 36-42.

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