

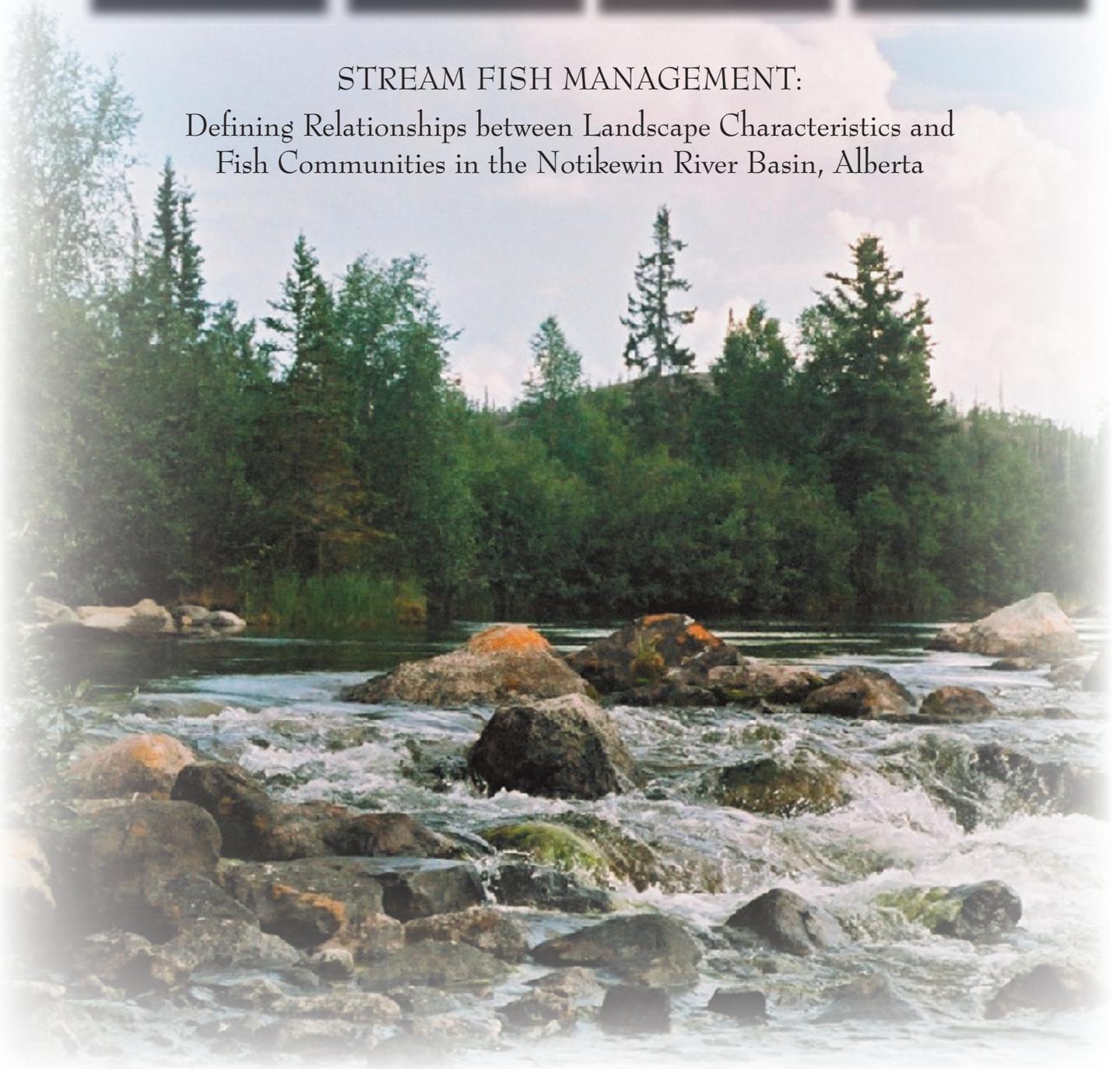
NORTHERN WATERSHED PROJECT

Project Report #2



STREAM FISH MANAGEMENT:

Defining Relationships between Landscape Characteristics and
Fish Communities in the Notikewin River Basin, Alberta



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



STREAM FISH MANAGEMENT:

DEFINING RELATIONSHIPS BETWEEN LANDSCAPE CHARACTERISTICS AND FISH COMMUNITIES IN THE NOTIKEWIN RIVER BASIN, ALBERTA

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EXECUTIVE SUMMARY

Study Focus

Spatial and temporal patterns in stream fish communities in the Notikewin River Basin, Alberta, Canada were evaluated using data from 266 stream reaches sampled between 1995 and 2001. Using these data, we addressed the following five focal questions:

- 1) How temporally variability is fish presence and density?
- 2) Is the presence of fish predictable based on stream habitat and watershed variables?
- 3) Do stream reach and watershed characteristics shape fish assemblage structure?
- 4) To what extent are relations between fish assemblages and watershed variables applicable to adjacent watersheds?
and
- 5) To what extent are fish assemblages shaped by forest cover attributes.

Focal question 1: How temporal variability is fish presence and fish density?

At the watershed scale, stream fish communities comprised 17 species from 8 familial groups. Arctic grayling, longnose sucker, lake chub, brook stickleback, trout perch and northern pike were the most frequently encountered species and occurred at between 5% and 32% of all 266 sampling sites. Comparisons of relative abundance showed that Arctic grayling, lake chub, brook stickleback, longnose sucker, finescale dace, trout perch and stickleback were numerically dominant and, when combined, accounted for 89% of all fish collected.

Overall density of fish in the Notikewin Basin was low (Mean = 0.92 individuals / 100 m²) and only densities of brook stickleback, Arctic grayling, finescale dace and lake chub typically exceeded of 0.1 individuals / 100 m². Within taxonomic groups, mean densities of cyprinid minnows (overall mean = 0.34 individuals / 100 m²) exceeded that of salmonids (0.22 individuals / 100 m²) and gasterosteids (0.27 individuals / 100 m²). In general, fish communities were numerically dominated by small bodied fishes (i.e., cyprinids, gasterosteids, percopsids, cottids) compared to larger bodied forms (e.g., percids, salmonids, esocids, and catostomids).

Analysis of fish community data from 27 sites (i.e., stream reaches) sampled on two occasions between 1995 and 2001 showed that the presence of fish was highly temporally variable and overall only 52% of all sites that contained fish in one year also contained fish one to five years later. Concordance in the presence of fish between sampling years increased with stream size from about 33% in first order streams (i.e., small streams) to 67% fifth order stream reaches (i.e., larger rivers). Similarly, concordance in the absence of fish between sampling events was also variable; on average only 39% of all sites that did not contain fish on one sampling occasion were also devoid of fish one to five years later.

Regression analyses indicated that the number of species replacements (i.e., number of species gained or lost between sampling years) was positively related to stream size, measured as stream bankfull width, and multiple regression showed that the relative number of species replacements was negatively related with bankfull width and positively related with time between repeated samples suggesting that the presence of fish was more temporally variable in small compared to larger stream reaches.

Fish density was also temporally variable and varied by about 10-fold between 1995 and 2001 (Highest annual mean±1SE [standard error] = 2.2±1.39 100 m², lowest annual mean = 0.24±0.13 100 m²). Density of Arctic grayling was also highly variable and differed 20-fold during the six year period (1.82±1.1 to

0.088±0.05 100 m²). Densities of brook stickleback and other numerically dominant fish species were also variable among years and varied between 5 and 25 fold during the six year study period.

Focal question 2: Is the presence of fish predictable based on stream habitat and watershed variables?

At the stream-reach-scale, the occurrence of fish was strongly affected by stream size. While only about 40% of first and second order stream reaches and small permanent stream reaches contained fish, the majority (65% to 90%) of third, fourth, fifth order stream reaches and large permanent stream reaches contained fish. Stream size also strongly affected the presence of individual species and species groups. Game fish (all species of game combined as one group), cyprinids, catostomids, and individual species of Arctic grayling occurred relatively infrequently (<35% occurrence) in small streams (1st to 3rd and small permanent streams) but were recorded between 20 and 40 times more frequently in larger streams. Brook stickleback, lake chub, longnose sucker, white sucker, northern pike, and trout perch occurred most frequently in larger streams.

At the landscape-scale, logistic regression analyses showed the presence of fish, game fish (all game fish species combined as one group), cyprinid minnows and Arctic grayling were highly predictable based on stream bankfull width, UTM northing and percent gravel. Reach elevation was the strongest predictor of the presence of suckers, longnose sucker, lake chub and brook stickleback. Single variable models using these predictors explained less variance in the presence of northern pike and resulted in lower classification success. Logistic regression models generally explained between 42% to 74% of the variance in the presence of fish, fish type and individual species and correctly classified sites where fish should be present or absent compared to presence based on electroshocking between 71% and 93% of the time.

Logistic regression models completed at the stream-scale (i.e., for 1st to 5th order reaches and small and large permanent streams) also performed well. In the majority of cases, variables that predicted the presence or absence of fish were measures of stream size or related to measures of stream size. At the stream reach level, stream size measured as bankfull width, water temperature, reach slope and to a lesser extent, elevation and gravel were significant predictors of the presence of fish, game fish, taxonomic groups and individual species. After excluding models which were not calculated due to low sample sizes, the majority of logistic regression models were statistically significant at $P < 0.05$ and typically explained between 51% and 87% of variance in the presence or absence of fish and on average correctly classified about 73% of sites as containing fish or as fish-absent.

The effects of stream classification method on the ability of logistic regression models to predict the presence and absence of fish was evaluated by comparing model fit and classification success between models created using Strahler stream orders and that using Alberta watercourse classes. With two exceptions, the stream classification method had only minor effects on the fit of logistic regression models and overall classification success. However, the ability of logistic regression models to correctly classify large permanent reaches as containing game fish and Arctic grayling were low (classification successes = 53% and 59%) and these models are problematic if the objective of modeling is to correctly predict their presence of these groups.

Focal question 3: Do stream reach and watershed characteristics shape fish assemblage structure?

Canonical correspondence ordinations indicated that instream habitat and watershed variables appear to shape fish assemblage structure and explained 24.6% of the variance in fish densities. When constrained by environmental variables, Axes 1 and 2 explained 80.8% of the variation between species abundances and environmental variables. Increased densities of white sucker, northern pike, sculpin and trout perch

were positively related with percent cobble, percent gravel, water depth and bankfull width. Density of Arctic grayling was positively related to elevation whereas increased density of longnose sucker was negatively related with elevation. Densities of dace, shiner, brook stickleback and chub were negatively related with cobble, gravel and water depth and bankfull width.

Multiple regression using forward selection showed that total fish density and density of the dominant groups were only weakly or moderately strongly related to environmental variables. Total density and density of individual species and species groups were typically significantly ($P < 0.05$) related to cobble, gravel, elevation, water depth and width and elevation. These regression models explained between 5% and 32% of variance in fish densities and often (5 of 9 regression models) accounted for 18 to 32% of variance in the abundance of individual species and species groups.

Density of Arctic grayling was positively related with elevation and negatively related with water depth and forest cover. Density of brook stickleback was negatively related with cobble, bankfull depth and percent gravel within the substratum whereas density of longnose sucker and trout perch were negatively related with elevation and positively related with bankfull width, respectively.

Focal question 4: To what extent are relations between fish assemblages and watershed variables applicable to adjacent watersheds?

Hierarchical cluster analyses, using fish density data, identified three relatively discrete fish assemblage types in the Notikewin River Sub-basin. Assemblage 1 consisted primarily of Arctic grayling, brook stickleback and dace whereas Assemblage 2 consisted of Arctic grayling, chub, white sucker, longnose sucker and northern pike. For these assemblages, mean total density and density of the three most numerically abundant groups ranged from 1.42 to 1.77 / 100 m² and 0.15 to 0.69 / 100 m², respectively. In contrast, Assemblage 3 was comprised of high densities of brook stickleback, dace, chub and to a lesser extent longnose sucker. Overall, mean total density and density of the three most abundant species was 12 to 15 and 18 fold higher than that in Assemblages 1 and 2.

We used a discriminant function model to differentiate among the three fish assemblage types using instream and watershed-scale variables. Based on fish density data, the forward selection discriminant analysis identified site elevation, percent gravel and cobble, and reach slope as significant (Wilks' Lambda, $P < 0.0001$) discriminators among the three fish assemblages. The linear discriminant function model had an overall classification success of 76.2% (i.e., 48 of the 63 sites were classified correctly) and correctly classified 73.3% (i.e., 33 of 45), 75.0% (i.e., 9 of 12), and 100% (6 of 6) of sites into assemblages 1, 2 and 3, respectively.

Stream reaches that supported Assemblage 1 were typically located at higher elevations and had higher amounts of gravel within the substratum compared with those belonging to Assemblages 2 and 3. In contrast, water depths and percent cobble at sites supporting Assemblages 1 and 2 were greater than stream reaches that supported Assemblage 3.

We tested the utility of the discriminant function model (based on density and percent composition) derived for fish communities in the Notikewin River Sub-basin by applying the model to fish communities described at 39 sites in the adjacent Hotchkiss and Meikle River Sub-basins. When applied to the Hotchkiss and Meikle River Sub-basins, the discriminant function models correctly classified 74.4% (i.e., 48 of the 63 sites were correctly classified) of all sites and correctly classified 4 of 5 sites into assemblage 1 (i.e., classification success = 80.0%), 17 of 23 sites into assemblage 2 (classification success = 73.9%) and all 8 of the 11 sites into assemblage 3 (i.e., classification success = 72.7%). These data suggest that the discriminant function model developed from the Notikewin River Sub-basin is a relatively good predictor of fish community types in the adjacent Hotchkiss and Meikle River Sub-basins.

These results suggest that relations between fish communities and watershed attributes derived in the Notikewin Sub-basin are also applicable to that in the Hotchkiss and Meikle River Sub-basins.

Focal question 5: To what extent are fish assemblages shaped by forest cover attributes?

Logistic regression analyses showed that forest cover attributes were typically poor predictors of the presence of fish, game fish, taxonomic groups and individual species compared with regression models derived using both non-forest physiographic attributes and forest cover attributes.

Overall, only 50% (16 of the 32 models) of empirical models using forest attributes were statistically significant ($P < 0.10$) predictors of the presence of fish. The remaining models were either statistically non-significant or were questionable because maximum likelihood estimates could not be calculated. In contrast, 97% (31 of 32 models) of empirical models using both physiographic and forest cover attributes were significant predictors ($P < 0.10$) of fish presence and in the majority of cases (23 of 32 models) models were statistically significant at an alpha of 0.05. Lastly, forest cover attributes were included within only 12.5% (4 of 32) of best-fit models.

Analyses based on Strahler stream orders showed that fish community structure was typically not statistically related (mean model $P = 0.12$) to forest cover attributes compared with the predominance of statistically significant models derived using non-forest physiographic ($P = 0.04$). Empirical models derived solely with forest cover attributes also had lower classification success but only marginally lower explanatory power. Analyses using Alberta watercourse stream classes also showed that forest cover attributes had minimal explanatory power in explaining fish community structure and were less powerful than empirical models derived using non-forest physiographic attributes.

The extent to which forest cover attributes explained variation in fish community structure was evaluated by partitioning the variance in fish communities into that which could be explained by: 1) forest cover attributes and 2) physiographic, non-forest cover attributes using variance partitioning techniques.

For this analysis, forest cover attributes included those measured at the watershed scale and those measured within the three riparian zones areas (i.e., areas of 3.2 ha, 28.1 ha, 78.1 ha) delineated by establishing radii of 100 m, 300 m and 500 m around sampling points.

Multiple canonical correspondence analyses showed that the environmental variables explained a relatively small proportion of the overall variance in fish density. When decomposed, the majority (45.6%) of explained variance in fish communities was attributed to non-forest cover attributes of elevation, mean water depth, reach slope, and the percentage of gravel and cobble within the substratum. Forest cover attributes including percent of watershed forested, and percent conifer, deciduous accounted for 29% of the variance that could be explained with the remaining 25.4% attributed to the interaction of forest cover and non-forest cover attributes.

Synopsis of Management Implications and Challenges

Taken together, our data indicate that logistic regression is a potentially powerful and relatively simple method to predict the presence of fish, game fish and some individual species in the Notikewin watershed. The presence of fish was moderately to highly predictable based on a small suite of recurring variables. The majority of these variables are easily measured either by querying existing GIS layers or by completing field surveys and are moderately robust to measurement errors. Logistic regression may prove to be a useful management technique to predict the presence of fish.

In contrast, empirical models predicting total fish density and density of the numerically dominant species and species groups from instream and watershed variables performed relatively poorly. If the lack of model fit reflects inherent variability, rather than poor selection of predictor variables, these watershed-scale models may be of limited value to fisheries biologists and resource managers.

In contrast, community-based models identified three relatively distinct fish assemblages in the Notikewin River Basin; these assemblage types were predictable based on reach elevation and percent gravel. Of equal importance, is that the fish assemblage model developed in the Notikewin Sub-basin was broadly applicable to that in the adjacent Notikewin and Meikle River Sub-basins, suggesting that the model may be broadly applicable at least at the local regional scale. The extent that these models can be applied to larger spatial scales, such as ecoregions remains to be determined, but if successful, could provide fisheries managers with an effective larger-scale predictive tool.

Based on our understanding of current stream fish management practices we identify eight areas where additional information and changes in management practices would benefit the conservation of stream fish communities. In an abbreviated form they include:

- 1) Assessments of the potential effects of industrial, municipal and agricultural activities on stream fish are seldom completed using spatially accurate and up-to-date information. This paucity of information challenges resource managers because they may not be fully aware of species-complexes present in stream and thus may not be able to identify practices that minimize impacts from industrial activities.
- 2) Fisheries assessments are typically completed at small spatial scales with little consideration of larger landscape patterns in fish communities. This approach precludes identifying broad-scale patterns in fish abundance or community types that should be included as part of evaluations of ecological risks. Management of stream fish communities would be enhanced if evaluations were based on assessments completed at multiple spatial scales.
- 3) The absence of an effective stream monitoring program compromises our ability to manage stream fish communities in Alberta. 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, iii) evaluate the effectiveness of restoration measures, and iv) to critically assess the effectiveness of current watershed management practices.
- 4) Current approaches to stream fish management do not take full advantage of quantitative tools or recognized quantitative relationships. The use of these approaches and tools can assist stream fish management by: i) providing techniques to understand large-scale patterns in fish communities, ii) gaining insights to potential cause-effect relationships that drive fish communities, iii) evaluating environmental impacts, iv) quantifying temporal variance in fish communities, and vi) monitoring the effectiveness of current management actions.
- 5) A better understanding of the causal mechanisms responsible for the degradation of stream fish communities would enhance stream fish management by: i) identifying the major causes of negative impacts on fish populations, ii) allow scenario modeling exercises to forecast long-term and large scale consequence of alternative management actions and iii) providing an ecologically sound approach to restoration actions.
- 6) An improved understanding of the cumulative effects of watershed disturbances on stream fish communities is required including the effects of land use conversions (i.e., conversion and loss of forested watersheds to agriculture).

1.0 GENERAL INTRODUCTION

1.1 Distribution and Abundance of Fish Communities in Forested Landscapes

The distribution and abundance of stream fish is strongly affected by watershed characteristics.

Stream fish communities vary as a function of geological history, climate, landform and watershed and riparian vegetation that affect 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 can be viewed as being 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).

Fish communities are often structured along gradients in environmental conditions.

In an attempt to better understand factors affecting the distribution and abundance of stream fish, fish ecologists have sought to develop empirical models predicting the occurrence, density and biomass of fish from watershed attributes (e.g., Tonn *et al.* 1983, Fausch *et al.* 1988, White 1996, Platts and McHenry 1988, Sharma and Hilborn 2001). In general, these models have often resulted in moderately powerful predictive relations between fish density and standing crop with watershed and instream variables, or have provided biologically important insights on the factors affecting the distribution and abundance of stream fish communities (e.g., Barton *et al.* 1985, Fausch *et al.* 1988, Platts and McHenry 1988, Marcus *et al.* 1990, Minns *et al.* 1996).

1.2 Geological History of the Boreal Region and Fish Assemblages

In geological terms, Boreal rivers and the fish assemblages that they support are very young and most likely developed within the last 10,000 years.

In geological terms, Boreal rivers and the fish assemblages that they support are very young and most likely developed within the last 10,000 years, as the Laurentide and Cordilleran ice sheets receded at the end of the Wisconsinian 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. Following glacial retreat, fish colonized the Boreal Plains and Cordillera as the continental glaciers melted to create a complex and dynamic drainage system about 5000-9000 years ago. 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 fauna 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 comprising only 58 species including 51 native and eight introduced species (Nelson and Paetz 1992). Biogeographic patterns in stream fish assemblages within the provinces nine major watersheds, including that within the Notikewin watershed located in the Peace River drainage, are generally poorly understood.

1.3 Study Rationale

Recent expansion of industrial activities has raised concerns about the ecological sustainability of stream fish communities in northern Alberta.

The expansion of Alberta's forest industry since the mid-1980's combined with the conversion of forest lands to agriculture and increased oil and gas activities has raised concerns about the ecological sustainability of stream fish communities in northern Alberta. These industrial activities potentially alter a myriad of interrelated stream and watershed attributes that alter 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).

The lack of empirical models predicting the distribution and abundance of stream fish within Alberta's boreal forest is a central challenge to the management of stream fish assemblages.

The lack of empirical models predicting the distribution and abundance of stream fish within Alberta's boreal forest is a central challenge to the management of stream fish assemblages. In the absence of time to complete field studies to describe fish communities, watershed management planning is often based on using historical information on fish assemblages and is often derived from watersheds other than those where activities are planned. This paucity of information typically results in conservative land use management prescriptions that are poorly tailored to extant fish communities and which preclude evaluations of environmental effects at multiple spatial scales (i.e., reach, stream, watershed scale, landscape scales).

The development of predictive tools will allow risk screening and assessments of proposed projects.

If ecological considerations are to be a part of sustainable forest management, we need to understand how fish communities are distributed at the landscape scale and how multiple, small-scale changes in stream fish assemblages influence large-scale patterns in fish populations. Empirical models defining how fish communities are distributed can also provide resource managers with an improved understanding of ecological risks by: 1) predicting the presence of fish, 2) identifying fish community types, 3) quantifying the extent to which empirical relations may be broadly applicable to other watersheds, 4) determining the extent to which fish abundance or presence varies through time and 5) describing whether fish communities are related to forest cover type. These relationships can allow resource managers to complete initial risk scoping exercises, based on known levels of predictive power.

1.4 Study Objectives

Study objective: To describe spatial, and to a lesser extent, temporal patterns in fish community structure in the Notikewin watershed.

The objectives of the present study were to describe spatial and to a lesser extent temporal patterns in fish community structure in the Notikewin watershed. Specifically, we addressed the following questions:

- 1) How temporally variable are fish communities?
- 2) Is the presence of fish, game fish and individual species predictable based on stream habitat and watershed variables?
- 3) Do stream reach and watershed characteristics shape fish assemblage structure?
- 4) To what extent are relations between fish assemblages and watershed variables applicable to adjacent watersheds?
- 5) To what extent are fish assemblages shaped by forest cover attributes?

We predicted that that the presence of fish, game fish and individual species would be highly predictable based on stream reach and watershed-scale attributes.

We predicted that: 1) the presence of fish, game fish and individual species at the reach-scale would be highly predictable based on stream size whereas the presence of fish in 1st to 5th order stream sites would be predictable based on watershed area, adjacency, stream reach slope and mean depth, 2) instream and watershed variables would explain a significant proportion of variance in the abundance of fish communities and assemblage structure, 3) relations between fish assemblages and watershed variables in the upper Notikewin Sub-basin would be similar to relations between fish communities and watershed variables in the Hotchkiss and Meikle Sub-basins, 4) the presence, abundance and composition of fish communities in the Notikewin Basin would be highly variable among years and 5) forest cover attributes would be moderately poor predictors of fish community structure compared with non-forest watershed attributes.

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

2.0 STUDY AREA

2.1 Description

Fieldwork was completed in the Notikewin River Basin located in northern Alberta.

Fieldwork was completed in the Notikewin Basin (Drainage area = 9,799 km²) located east Manning, Alberta (Figure 1). The Notikewin Basin is comprised of the Notikewin (7th order), Hotchkiss (5th order) and Meikle (6th order) River Sub-basins. All Sub-basins originate in the foothills of the Rocky Mountains of Alberta and flow east to the Peace River (Figure 2).

Monthly flow of the Notikewin River at Manning averages 14 m³ s⁻¹ (1961-1999) with peak flows typically occurring in May after snowmelt (Mean monthly and mean maximum monthly discharge = 149 m³ s⁻¹) and lowest flows in February (0.2 m³ s⁻¹) (Figure 3). Analyses of historical data (i.e., collected between 1961-1999) showed mean monthly flow is highly variable within months (Coefficients of variation based on among year monthly means between 1961 to 1999 = 104%).

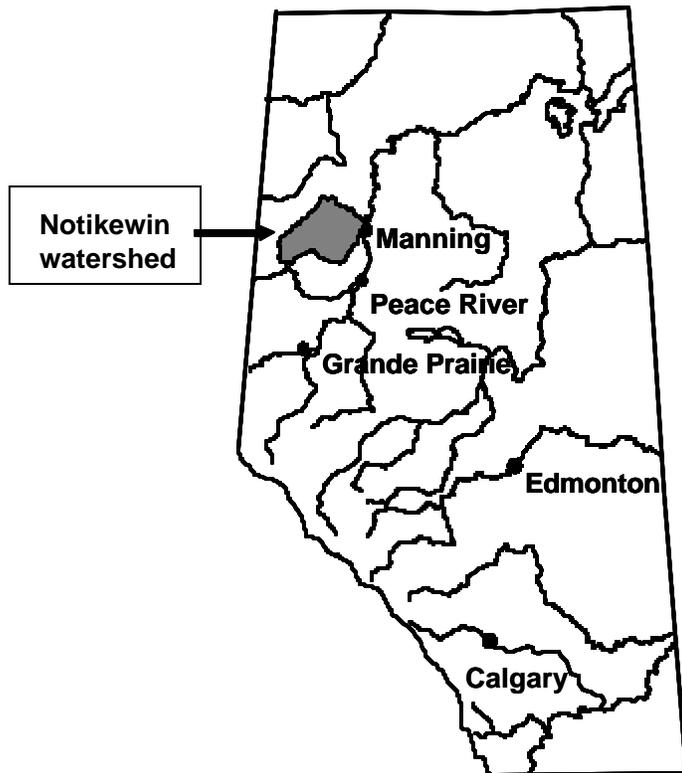


Figure 1. Location of the Notikewin River Basin in northwest Alberta, Canada.

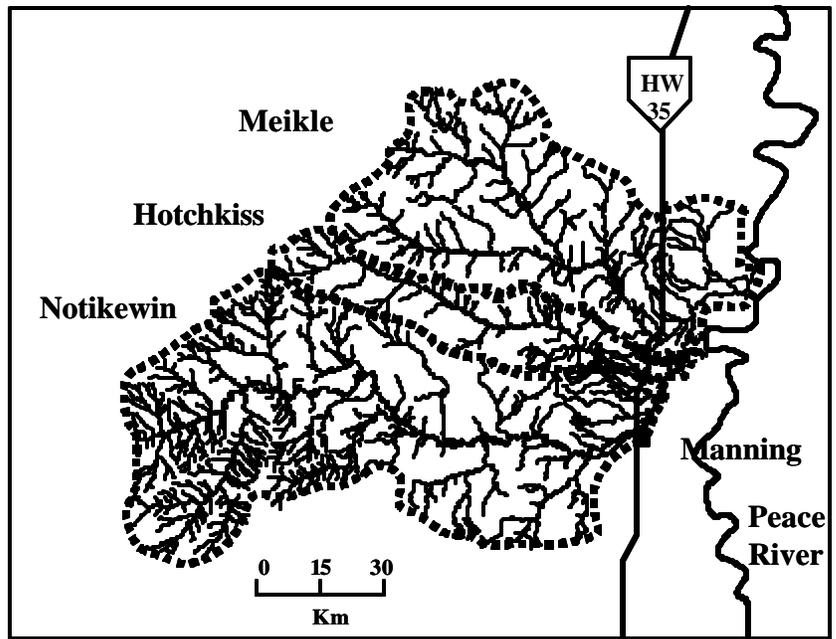


Figure 2. Location of the upper Notikewin, Hotchkiss and Meikle River Sub-basins of the Notikewin River Basin in northwest Alberta.

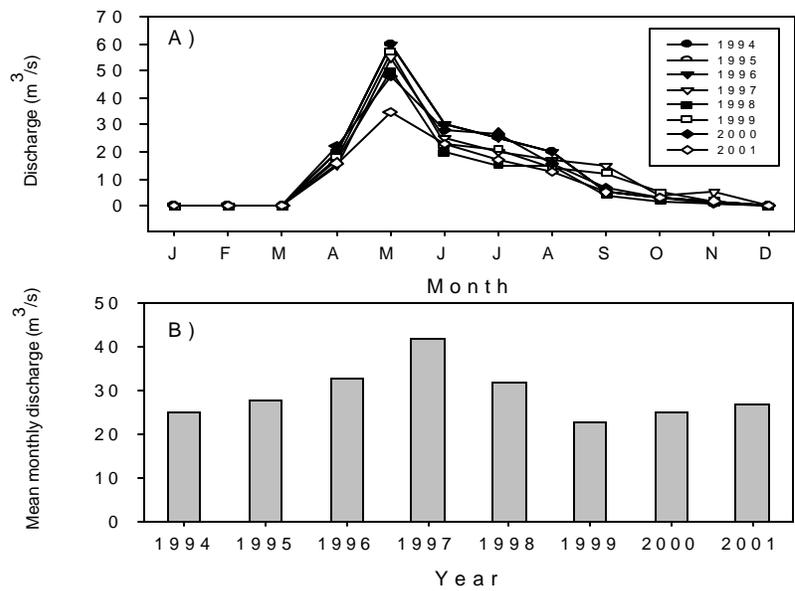


Figure 3. Mean monthly discharge of the Notikewin River at Manning Alberta, 1961-1999.

2.2 Ecoregions, Forest Cover and Soils

The Notikewin River Basin is located within the Upper and Lower Boreal Cordilleran and the Low Boreal Mixedwood ecoregions of Alberta.

The Notikewin watershed is encompassed within the Upper and Lower Boreal Cordilleran and Low Boreal Mixedwood ecoregions (Figure 4). The majority of the Notikewin is located within the Lower Boreal Cordilleran Ecoregion and all rivers flow through the Low Boreal Mixedwood Ecoregion into the Peace River (Figure 4).

Vegetation in the Upper Boreal Cordilleran ecoregion is dominated by closed canopy coniferous forest comprising lodgepole pine (*Pinus contorta*) that restricts the growth of deciduous species of trembling aspen (*Populus tremuloides*), and balsam poplar (*Populus balsamifera*) located on Gray luvisols and Brunisols (Strong and Legatt 1992). The Lower Boreal-cordilleran is an ecotone between boreal and cordilleran climatic conditions and reflects the transition from deciduous boreal and coniferous cordilleran vegetation. Lodgepole pine is the dominant species on well drained Gray luvisols with trembling aspen, balsam poplar, paper birch (*Betula papyrifera*), lodgepole pine, white spruce (*Picea glauca*) black spruce (*Picea mariana*), 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.

The Low Boreal Mixedwood is a transition between the Aspen Parkland and Mid Boreal Mixedwood ecoregion with Orthic Gray luvisols predominant (Strong and Legatt 1992). Forest communities are dominated by aspen on moderately well, to well-drained Gray luvisols and Eutric brunisols whereas an aspen, balsam poplar complex typically occur on imperfectly drained Gray luvisols and Gleysols. Black spruce occur predominantly on poorly drained sites.

Mean summer temperatures range from 11.5°C in the Upper Boreal Cordilleran to 13.8°C (Low boreal mixedwood) with mean winter temperatures ranging from ˆ6°C (Upper Boreal Cordilleran) to ˆ10.5°C (Low Boreal Mixedwood) (Strong and Legatt 1992; Table 1). Extreme winter temperatures can fall to ˆ40°C. The majority of annual precipitation falls during the summer period in all three ecoregions of the Notikewin River Basin (Table 1). The area is characterized by a mean summer and winter temperatures of 13.5°C and–13.2°C, respectively and annual precipitation of ~ 400 mm (Strong and Legatt 1992). Undulating morainal plains are overlain by Gray Luvisols with some Eutric Brunisol soils on more xeric sites (Rowe 1972, Strong and Legatt 1992).

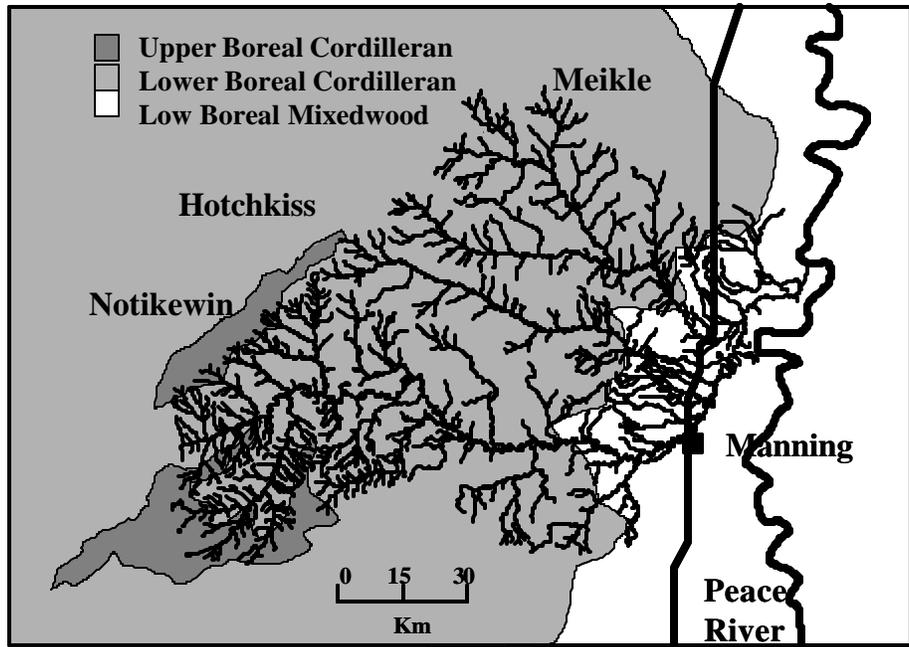


Figure 4. Location of the Upper Boreal Cordilleran, Lower Boreal Cordilleran and Low Boreal Mixedwood ecoregions in the Notikewin River Basin, Alberta.

Table 1. Summary of summer and winter air temperatures and annual precipitation in the Upper boreal cordilleran, Lower Boreal Cordilleran and Low Boreal Mixedwood Ecoregions in the Notikewin watershed.
Summer = May-August, Winter = November-February.

	Ecoregion		
	Upper Boreal Cordilleran	Lower Boreal Cordilleran	Low Boreal Mixedwood
Air Temperatures			
<u>Summer</u>			
Mean	11.5	12.8	13.8
Mean minimum	5.9	6.9	7.0
Mean maximum	16.7	18.3	20.4
<u>Winter</u>			
Mean winter	-6.0	-7.8	-10.5
Mean maximum	-12.5	-14.3	-15.8
Mean minimum	0.5	-2.1	-5.3
Precipitation			
Summer	340	295	235
Winter	60	60	61
Annual	538	464	380

2.3 Fish Communities

Seventeen species of fish from 8 Families have been recorded in the Notikewin River Basin.

Seventeen species of fish from 8 Families have been recorded in the Notikewin River Basin (Table 2). Cyprinid minnows are the dominant species group (8 species) followed by suckers (2 species) and salmonids (2 species). The Notikewin River Basin supports four game species of Arctic grayling, mountain whitefish, northern pike and walleye.

Table 2. Common and scientific names of fish recorded in the Notikewin River Basin. The presence of spoonhead sculpin in the Notikewin watershed is debated and may result from the misidentification of slimy sculpin. * = game species

Family	Common name
Cyprinidae	
Lake chub	<u>Couesius plumbeus</u> (Agassiz)
Flathead chub	<u>Platygobio gracilis</u> (Richardson)
Finescale dace	<u>Phoxinus neogaeus</u> Cope
Pearl dace	<u>Margariscus margarita</u> (Cope)
Longnose dace	<u>Rhinichthys cataractae</u> (Valenciennes)
Northern redbelly dace	<u>Phoxinus eos</u> (Cope)
Emerald shiner	<u>Notropis atherinoides</u> Rafinesque
Northern pikeminnow	<u>Ptychocheilus oregonensis</u> (Richardson)
Percopsidae	
Trout-perch	<u>Percopsis omiscomaycus</u> (Walbaum)
Gasterosteidae	
Brook stickleback	<u>Culaea inconstans</u> (Kirtland)
Percidae	
Walleye*	<u>Stizostedion vitreum vitreum</u> (Mitchill)
Salmonidae	
Arctic grayling*	<u>Thymallus arcticus</u> (Pallas)
Mountain whitefish*	<u>Prosopium williamsoni</u> (Girard)
Esocidae	
Northern pike*	<u>Esox lucius</u> Linnaeus
Catostomidae	
Longnose sucker	<u>Catostomus catostomus</u> (Lacepède)
White sucker	<u>Catostomus commersoni</u> (Forster)
Cotti dae	
Slimy sculpin	<u>Cottus cognatus</u> Richardson
Spoonhead sculpin	<u>Cottus ricei</u> (Nelson)

3.0 GENERAL MATERIALS AND METHODS

Empirical relations between fish communities and watershed characteristics in the Notikewin River Basin were based on data from 266 stream reaches.

Empirical relations between fish communities and watershed characteristics in the Notikewin River Basin were quantified using historical data on fish communities and instream habitats collected at 100 sites from 1995-1997 (Hvenegaard 1998) combined with inventories at 166 stream sites in 1999-2001 completed by the Northern Watershed Project. These data were combined with landscape and watershed-scale characteristics derived primarily from queries of Geographic Information Systems (GIS) databases describing watershed morphometry and forest cover.

3.1 Data Screening

Our primary interest was to establish relations between fish communities and watershed characteristics in the absence of industrial activities.

Our primary interest was to establish relations between fish communities and watershed characteristics in the absence of industrial activities. In the Notikewin River Basin, we minimized the potentially confounding effects of industrial activities on fish-watershed relations by excluding potentially impacted stream reaches. We accomplished this by quantifying the area that forest harvest blocks, rail lines, roads, well sites, open pit mines, and transmission rights-of-way reset forest cover to early successional stages or created non-vegetated areas within all study watersheds. Based on these analyses, 4 study sites were excluded from statistical analyses because industrial activities within their watersheds exceeded 20% on an areal basis. On average, industrial activities in the remaining sites accounted for only 4.1% of contributing watershed areas (Standard deviation = 4.0, Maximum = 18.7% Minimum = 0%). The exclusion of sites where industrial activities occupied >20% of their contributing watersheds is probably somewhat conservative.

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

A second data screening process was completed to exclude 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.

3.2 Watershed Characteristics

Predictors of fish communities were divided into the two classes of: i) instream and riparian attributes and ii) watershed attributes.

Predictors of fish communities were divided into the two classes of: i) watershed characteristics and ii) stream reach and riparian attributes (Table 3). Watershed-scale characteristics were derived to: 1) evaluate the extent that large-scale features shape fish communities and 2) allow partitioning of variance to understand the relative importance of large-scale versus small-scale features (i.e., stream reach) as drivers of fish community structure. The majority of watershed-scale variables were

quantified using databases created within the Provincial Governments' Geographic Information System (GIS) (i.e., Base Features).

Table 3. Environmental variables used to explain spatial and temporal variation in stream fish communities in the Notikewin River Basin. All forest cover attributes were calculated for contributing watersheds.

Variable	Abbreviation	Units
<u>Watershed</u>		
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	7 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
Watershed slope ^h	CW-SLOPE	Degrees
<u>Stream reach</u>		
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
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
Riparian slope – 100m ^m	RIP-SLP-100	Percent
Riparian conifer – 300m ^{k,n}	%RIPC-300	Percent
Riparian deciduous – 300 m ^{k,n}	%RIPD-300	Percent
Riparian conifer & deciduous - 300 m ^{k,n}	%RIPCD-300	Percent

Table 3 continued

Variable	Abbreviation	Units
Riparian deciduous & conifer - 300 m ^{k,n}	%RIPDC-300	Percent
Riparian productive forest - 300 m ^{k,n}	%RIP-PR-300	Percent
Riparian non-productive forest - 300 m ^{k,n}	%RIP-NPR-300	Percent
Riparian non-classified forest - 300 m ^{k,n}	%RIP-NONC-300	Percent
Riparian slope - 300m ^m	RIP-SLP-300	Percent
Riparian conifer - 300 m ^{k,o}	%RIPC-500	Percent
Riparian deciduous - 500 m ^{k,o}	%RIPD-500	Percent
Riparian conifer & deciduous - 500 m ^{k,o}	%RIPCD-500	Percent
Riparian deciduous & conifer ^m - 500 m ^{k,o}	%RIPDC-500	Percent
Riparian productive forest - 500 m ^{k,o}	%RIP-PR-500	Percent
Riparian non-productive forest - 500 m ^{k,o}	%RIP-NPR-500	Percent
Riparian non-classified forest - 500 m ^{k,o}	%RIP-NONC-500	Percent
Riparian slope - 500m ^m	RSLOPE-500	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)

^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.

ⁱ Slope (rise over run) calculated from points located 150 m upstream and 150 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.
- ^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 Nephelometric Turbidity Unit).
- ^v WTW Multiline P4 portable dissolved oxygen meter fitted with a OxiCal 325 probe.

The Base Features database was used to describe the physical location of study watersheds (i.e., longitude, latitude, elevation) 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 adjacent to sampling sites (i.e., 100, 200 and 300m areas).

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. We also defined forest cover in terms of: 1) potentially productive, 6) productive, non-forested and 7) unproductive, non-forested and 8) non-classified. 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 from field surveys and queries of GIS databases.

Physical characteristics of stream reaches (average stream reach length = 199 m) were determined during field surveys between 1995 and 2001 combined with queries of GIS databases in 2001. 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, and using GIS tools, 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).

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

We established three (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 estimates were 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 three locations along each of the five transects within each study reach.

Size composition of the substratum was determined by visually estimating the percent cover of silt and sand (<4.7 mm), gravel (4.8 - 76 mm), cobble (76.1 – 304.7 mm) and boulder (>304 mm) within three 1 m² areas, located along three (1999-2001) or five (1995-1997) transects 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 within 86 study reaches by counting and measuring all pieces of woody material that exceeded 20 cm in length within a randomly selected 40 m section of the study reach. For each piece, we quantified the length, 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 (m³/100 m²) by dividing the volume of woody debris by the area sampled (i.e., mean width x 40 m).

Streams were defined using Alberta Watercourse Course and Strahler stream classes.

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).

Forest practice codes in Alberta identify five classes of stream based largely on width, water flow period and channel development (Table 4). For the purpose of this study we refer to this classification as the Alberta watercourse classification, but acknowledge that other classifications exist. 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).

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 ^a order
Water source areas	Saturated soils and seepages.	All year.	N/A	1 st
Ephemeral	Often vegetated draws.	Only during or after rainfall.	Minimal, often vegetated or snow melt.	1 st
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.	1 st
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.	2 nd to 4 th
Large permanent	Major streams or rivers. Well defined floodplains. Valley width usually exceeds 400 m.	All year.	Non-vegetated channel. Channel width > 5m.	3 rd to 7 th

3.4 Fish Communities

Fish communities at 266 sites in the Notikewin watershed were described by electroshocking.

Fish communities at 266 sites in the Notikewin watershed were described by electroshocking stream reaches (average reach = 199 m, range in reach lengths = 20 to 492 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) between 1995 and 2001 (Table 5). The majority of sampling was completed in third (N = 80 sites) and fourth order (N = 75) sites that represent small permanent streams (Table 6, Figure 5).

Because our primary objective was to describe large-scale patterns in fish community structure in the Notikewin watershed, we used a single, rather than a multi-pass approach, to describe fish communities because it allowed us to sample substantially higher 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 – focussed studies (e.g., Kilgour and Barton 1999).

Data from electroshocking was used to describe: 1) fish presence-absence, 2) total fish density and 3) density of individual species.

Fish captured during electroshocking were identified using taxonomic keys in Nelson and Paetz (1992) or Scott and Crossman (1998), sometimes measured and released after the reach had been shocked. In several cases, fish were often shocked but not retrieved because the current swept them past capture nets or they became entangled in interstitial spaces or woody debris. Non-captured fish that were identified with confidence to species were included in total density and species density estimates but likely accounted for less than 2% of all fish identified. Data from electroshocking was used to describe: 1) fish presence-absence 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). Because fish densities in the Notikewin River Basin are relatively low, we converted density per m² to density per 100 m².

Table 5. Summary of stream fish and habitat surveys completed in the Notikewin River Basin, 1995 to 2001. Data collected between 1995 and 1997 were provided by the Cooperative Fisheries Inventory Program (Hvenegaard 1998) whereas surveys between 1999-2001 were completed as part of the Northern Watershed Project. Numbers of sites includes those sampled on two occasions between 1995 and 2001.

Year	Period	Number of sites
1995	7 June – 10 Sept	29
1996	7 May – 5 Sept	53
1997	30 May – 6 Oct	18
1999	20 July – 5 Aug	34
2000	13 June – 21 June	77
2001	18 June – 21 June	55
Total		266

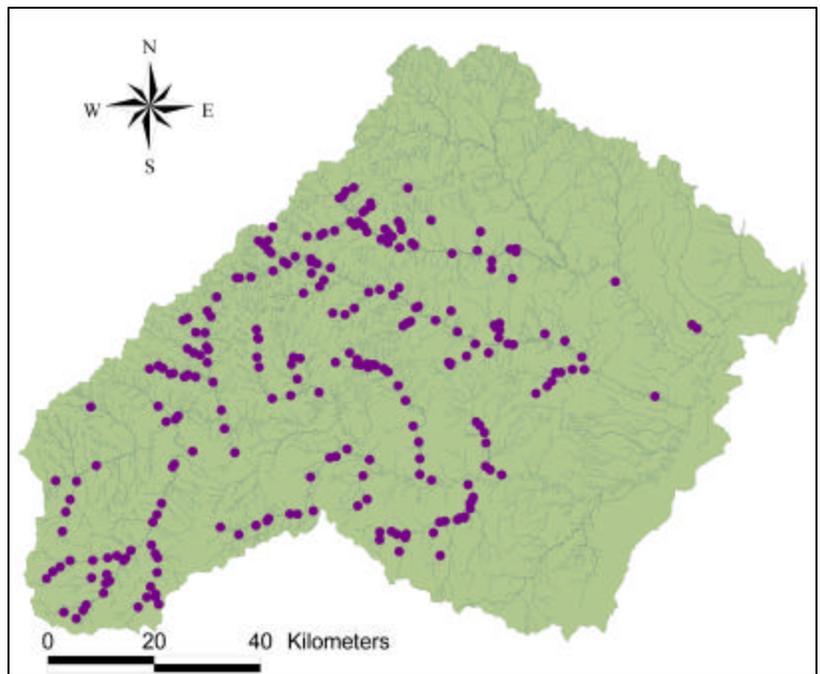


Figure 5. Location of sampling sites in the upper Notikewin, Hotchkiss and Meikle River Sub-basins of the Notikewin River Basin, Alberta. Sampling locales are shown as purple filled circles.

Table 6. Total number of stream reaches sampled in the Notikewin River Basin defined using (A) Strahler stream order and (B) Alberta watercourse classifications, 1995 to 2001. Data collected between 1995 and 1997 were provided by the Co-operative Fisheries Inventory Program (Hvenegaard 1998) with surveys between 1999-2001 were completed by the Northern Watershed Project. Stream classification using Alberta watercourse crossing criteria were based on mean bankfull width.

A)						
Strahler stream order						
Year	First	Second	Third	Fourth	Fifth	Total
1995	1	2	5	10	12	30
1996	4	8	13	14	14	53
1997	3	6	10	0	0	19
1999	0	6	9	13	5	33
2000	6	13	25	29	2	75
2001	12	11	18	9	6	56
Total	26	46	80	75	39	266

B)				
Alberta watercourse classification				
Year	Intermittent (Width <0.5 m)	Small permanent (Width 0.5-5m)	Large permanent (Width >5 m)	Total
1995	0	13	17	30
1996	0	26	27	53
1997	0	15	4	19
1999	0	23	10	33
2000	0	48	27	75
2001	1	39	16	56
Total	1	164	101	266

Focal Question 1: How temporally variable are fish communities in the Notikewin River Basin?

4.0 FOCAL QUESTION 1 – HOW TEMPORALLY VARIABLE ARE FISH COMMUNITIES IN THE NOTIKEWIN RIVER BASIN?

4.1 Design and Statistical analyses

Temporal variation in the presence of fish, total density and density of selected species, and absolute and relative number of species replacements were evaluated using data from 27 sites which were sampled on two occasions between 1995 and 2001. Because our primary focus was to describe patterns among independent stream reaches, we excluded sites from analyses when they were located: i) within the same first or second Sub-basin, and ii) within 3 km from an adjacent third, fourth, fifth order site. These conditions resulted in a total of 27 sites from 1st to 5th order streams (Table 7).

Table 7. Classification of 27 stream reaches used to quantify temporal variation in fish community structure in the Notikewin watershed, 1995-2000. Data were collected from each site on two occasions between 1995 and 2001. Classifications are based on (A) Strahler stream orders and (B) Alberta watercourse classifications (using mean reach bankfull width).

A)				
Strahler stream order				
First	Second	Third	Fourth	Fifth
1	5	10	9	2
B)				
Alberta watercourse classification				
Intermittent (Width <0.5 m)	Small permanent (Width 0.5 to 5 m)	Large permanent (Width >5 m)		
0	12	15		

Temporal variation in fish communities was evaluated using logistic and multiple regression and Wilcoxon rank tests.

The extent to which the presence of fish, game fish and selected individual species could be predicted from stream size (i.e., bankfull width) and years between the two repeated sampling events (i.e., one to six years between 1995 and 2001) 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 regressions were based on the statistical significance of the model, Nagelkerke R^2 , classification success, using a jackknife process to quantify cross-validation (CTABLE option in Proc Logistic) to reduce classification bias and the $-2LL$ statistic. The jackknife procedure calculates classification success after sequentially removing one observation and is based on N (number of sites) permutations.

Relationships between the: 1) number of species replacements and 2) relative number of species replacements with time between sample collection (years) and stream size (bankfull width) was evaluated using multiple linear regression (forward selection option) with an entry and removal criteria of 0.10. Prior to analyses we constructed a correlation matrix to exclude highly correlated variables from regression analyses. These analyses showed that time since sample collection was poorly correlated to stream bankfull width (Pearson correlation coefficient $r = 0.2$, $P > 0.05$).

Due to low sample sizes, comparisons of total fish density and density of selected species were restricted to pairwise comparisons between 1996 versus 2002, 1996 versus 2001 and 1999 versus 2002. Differences in density between year were completed using Wilcoxon paired rank tests which test the hypothesis that ranks of the differences between pairs of samples (i.e., between years) equals zero. Because these tests are based on rankings rather than absolute values, it allowed us to combine stream reaches of differing classes.

4.2 Results

4.2.1 Overall Patterns in Fish Communities

Arctic grayling, longnose sucker, lake chub, brook stickleback, trout perch and northern pike were the most frequently encountered species and occurred at between 5% and 32% of all sites.

Fish communities in the Notikewin River Basin comprise 17 species from 8 familial groups (Table 8). Arctic grayling, longnose sucker, lake chub, brook stickleback, trout perch and northern pike were the most frequently encountered species and occurred at between 5 and 32% of all sites. Comparisons of relative abundance showed that Arctic grayling, lake chub, brook stickleback, longnose sucker, finescale dace, trout perch and brook stickleback were numerically dominant and, when combined, accounted for 89% of all fish collected.

Table 8. Frequency of occurrence (%), percent composition (%) and mean (± 1 SE) density (number / 100 m²) of fish from the Notikewin River Basin, 1995-2001. Data are overall averages from all 266 first to fifth order stream reaches. Percent composition data were calculated from density estimates (i.e., numbers of fish / 100 m²).

Common name	Frequency of occurrence	Percent composition	Mean density composition
<u>Cyprinidae</u>			
Lake chub	16.92	11.9	0.11 \pm 0.03
Flathead chub	0.75	0.18	0.002 \pm 0.002
Finescale dace	2.26	17.03	0.16 \pm 0.11
Pearl dace	3.01	3.18	0.029 \pm 0.018
Longnose dace	2.63	1.18	0.01 \pm 0.01
Northern redbelly dace	0.38	1.92	0.018 \pm 0.018
Emerald shiner	1.88	1.51	0.01 \pm 0.01
Northern pikeminnow	0.38	0.30	0.003 \pm 0.003
<u>Percopsidae</u>			
Trout-perch	5.64	1.29	0.012 \pm 0.004
<u>Gasterosteidae</u>			
Brook stickleback	10.5	29.20	0.27 \pm 0.15
<u>Percidae</u>			
Walleye	0.75	0.02	0.0002 \pm 0.0001
<u>Salmonidae</u>			
Arctic grayling	31.95	24.45	0.22 \pm 0.04
Mountain whitefish	0.38	0.01	0.0001 \pm 0.0001
<u>Esocidae</u>			
Northern pike	6.39	0.56	0.005 \pm 0.002
<u>Catostomidae</u>			
Longnose sucker	13.91	6.02	0.06 \pm 0.02
White sucker	6.39	1.09	0.01 \pm 0.003
<u>Cottidae</u>			
Sculpin species	2.26	0.16	0.002 \pm 0.0005

Overall density of fish in the Notikewin Watershed was low (Mean = 0.92 individuals / 100 m²). Densities of brook stickleback, Arctic grayling, finescale dace, lake chub and brook stickleback exceeded densities of 0.1 individuals 100 m². Within taxonomic groups, mean densities of cyprinid minnows (overall mean = 0.34 individuals /100 m²) exceeded that of salmonids (0.22 individuals / 100 m²) and gasterosteids (0.27 individuals / 100 m²). In general, fish communities were numerically dominated by small bodied fishes (i.e., cyprinids, gasterosteids, percopsids, cottids) compared to larger bodied forms (e.g., percids, salmonids, esocids, and catostomids).

4.2.2 Presence-absence

After removing adjacent sites from analyses, temporal patterns in the presence and abundance of fish communities was based on analyses of data from 27 sites comprising 1st to 5th order reaches (Table 9).

Table 9. Mean ($\pm 1SE$) bankfull widths of 27 first to fifth order stream reaches (A) and small and large permanent stream reaches (B) in the Notikewin River, 1995-2001. Due to low sample sizes, sites from 1st & 2nd and 4th & 5th order sites were pooled.

Stream class	Mean	Range	N
Strahler			
1 st and 2 nd	5.05 \pm 1.87	2.1 – 14.25	6
3 rd	5.71 \pm 0.77	3.4 – 11.2	10
4 th and 5 th	9.92 \pm 1.63	2.8 – 20.95	11
Alberta watercourse			
Small permanent	3.64 \pm 0.25	2.1 – 4.7	12
Large permanent	10.18 \pm 1.17	5.1 – 20.95	15

The presence of Arctic grayling, lake chub and Cyprinid minnows was significantly related to either stream bankfull width or mean water depth.

Despite relatively low sample sizes, logistic regression showed that the presence of Arctic grayling ($P = 0.051$), lake chub and Cyprinid minnows was significantly related to either stream bankfull width or mean water depth (Table 10). The presence of all three species increased with stream bankfull width or mean water depth and overall regression models explained between 68 to 73% of the variance in the presence of fish. In contrast, models predicting the presence of fish, game fish and catostomid suckers were not statistically significant ($P > 0.05$) (Table 10).

Logistic regression was also used to compare the extent to which sites where fish were observed to be present or absent could be predicted using

stream bankfull width or mean water depth. These comparisons showed that between 74 and 93% of all sites that contained fish or where fish were absent were classified correctly using the jack-knife procedure (Table 10).

Table 10 Summary of logistic regression models predicting the presence of fish based on mean water depth (m) or mean bankfull width (M) from 27 sites sampled twice between 1995 and 2001 in the Notikewin watershed. Analyses were restricted to commonly encountered species or species groups. Game fish = Arctic grayling, walleye, mountain whitefish, northern pike; Catostomidae = white sucker and longnose sucker; Cyprinidae = lake chub, flathead chub, finescale dace, pearl dace, longnose dace, northern redbelly dace, emerald shiner and northern pikeminnow. NS = not statistically significant ($P > 0.05$). Classification success was calculated using a jack knife procedure. Significance levels for intercepts and main factors are shown within brackets. Highly non-significant values are not shown (i.e., $P > 0.10$). Class success = classification success.

Species group	Logistic equation	P	R ²	Class.	success
Fish	Y = -28.28 (NS) + 8.55 (NS) width	0.31	0.62	0.89	
Game fish	Y = -0.80 (NS) + 0.052 (NS) depth	0.15	0.65	0.78	
Arctic grayling	Y = -2.56 (NS) + 0.71 (0.05) width	0.051	0.68	0.93	
Lake chub	Y = - 6.84 (**) + 0.66 (*) depth	0.02	0.67	0.89	
Catostomidae	Y = - 1.35 (NS) + 0.06 depth (NS)	0.10	0.68	0.70	
Cyprinidae	Y = -4.01 (**) + 0.50 width (**)	0.01	0.73	0.74	

4.2.3 Concordance in the presence or absence of fish

Concordance (i.e., the presence of fish at the same site on both sampling events and the absence of fish at the same site on both sampling events) in the presence or absence of fish between sampling events varied with species group and stream size (Table 11). In general, the presence of fish was highly temporally variable and overall only about half (i.e., 52%) of all sites that contained fish in one year also contained fish one to five years later. Concordance in the presence of fish between sampling years increased with stream size from about 33% to 67% irrespective of whether streams were defined using Strahler or Alberta watercourse classifications. Concordance based on the presence of game fish, individual species or species groups differed with stream size and was about 5-fold higher in third and fourth and fifth order stream reaches compared with first and second order reaches and 3-fold higher in large permanent compared to small permanent streams (Table 11).

The presence of fish was highly temporally variable and overall only 52% of all sites that contained fish in one year also contained fish one to five years later.

Concordance in the presence of fish between sampling years increased with stream size.

Concordance in the absence of fish between sampling events was also variable; on average only one third (i.e., 39%) of all sites that did not contain fish on one sampling occasion were also devoid of fish one to five years later (Table 11). Alternatively, 61% of sites that contained fish on one sampling event did not contain fish one to five years later. The absence of fish between sampling years was also highly variable and did not decline markedly with increasing stream reach size (i.e., Strahler stream order) (Table 11).

4.2.4 Species replacement

Linear regression indicated that the relative number of species replacements (i.e., number of species gained or lost between sampling years) was negatively related to stream bankfull width and positively related to the number of years between sampling events.

Linear regression showed that the number of species replacements (i.e., number of species gained or lost between sampling years) was related to stream bankfull width. However this relationship largely arises because of the positive correlation between stream size and richness ($r = 0.74$). To adjust for the positive relationship between species richness and stream size, we divided the number of species replacements at a site by the mean number of species at that site. Using these data, multiple regression showed that the relative number of species replacements was negatively related with bankfull width ($P < 0.01$) and positively related with time between repeated samples ($P < 0.01$) (Relative number of species replacements = $1.14 - 0.06 \text{ width} + 0.17 \text{ time}$ ($F_{(2,24)} = 8.34$, $P < 0.005$, Adjusted $R^2 = 0.41$) (Figure 6).

4.2.5 Fish density

Our ability to better quantify temporal variation in fish density and density of predominant species using data collected between 1995 and 2001 was compromised because few sites were sampled consistently through time and sampling frequency (i.e., years between repeated sampling) varied among sites. This design precludes analyses using repeated measures or split plot approaches because of low and highly unbalanced numbers of replicate streams. Thus, overall variation in fish densities reflects spatial and temporal variation because few sites were sampled repeatedly through time.

As a result, we describe semi-quantitative patterns in fish density through time combined with quantitative pairwise comparisons among specific years using paired rank sign tests (i.e., 1996 versus 2002, 1996 versus 2001 and 1995 versus 1996). While the use of paired rank tests is beneficial because differences between years are based on ranks of differences rather than absolute densities, our statistical tests suffer from low sample sizes (i.e., repeated measurements of 4 to 7 sites between years) and presumably low statistical power.

Table 11. Comparison of mean percent concordance in the presence and absence of fish between two sampling periods in the Notikewin River, 1995-2002. To reduce inflation of concordance based on absence, comparisons were restricted to commonly encountered species or species groups. Fish class and species abbreviations are shown in Table 10. Means are based on the six fish groups. Sample means are provided as descriptive purposes only.

Strahler stream order				
	Present on both sampling occasions			
	First & second	Third	Fourth & fifth	
Fish	33.33	60.00	63.64	
Game fish	0	50.00	18.18	
Arctic grayling	0	40.00	27.27	
Lake chub	0	0	18.18	
Catostomidae	0	0	9.09	
Cyprinidae	0	0	9.09	
Means (± 1 SE)	5.55 \pm 5.55	26.50 \pm 10.84	24.24 \pm 8.35	
	Absent on both sampling occasions			
	First & second	Third	Fourth & fifth	
Fish	0	0	9.09	
Game fish	16.67	0	18.18	
Arctic grayling	16.67	0	18.18	
Lake chub	66.67	100.00	63.64	
Catostomidae	66.67	80.00	36.36	
Cyprinidae	66.67	100.00	36.36	
Means (± 1 SE)	38.89 \pm 12.67	46.67 \pm 16.33	30.30 \pm 5.92	
B) Alberta watercourse classification				
	Small	Large	Small	Large
	Permanent	permanent	permanent	Permanent
	Present at both Time 1 & Time 2	Present at both Time 1 & Time 2	Absent at both Time 1 & Time 2	Absent at both Time1 & Time2
Fish	33.33	66.67	8.33	0
Game fish	8.33	40.00	16.67	6.67
Arctic grayling	8.33	46.67	16.67	6.67
Lake chub	0	6.67	83.33	60.00
Catostomidae	0	6.67	83.33	46.67
Cyprinidae	0	6.67	83.33	46.67
Mean (± 1 SE)	8.33 \pm 3.72	28.89 \pm 6.68	48.61 \pm 11.01	27.78 \pm 6.75

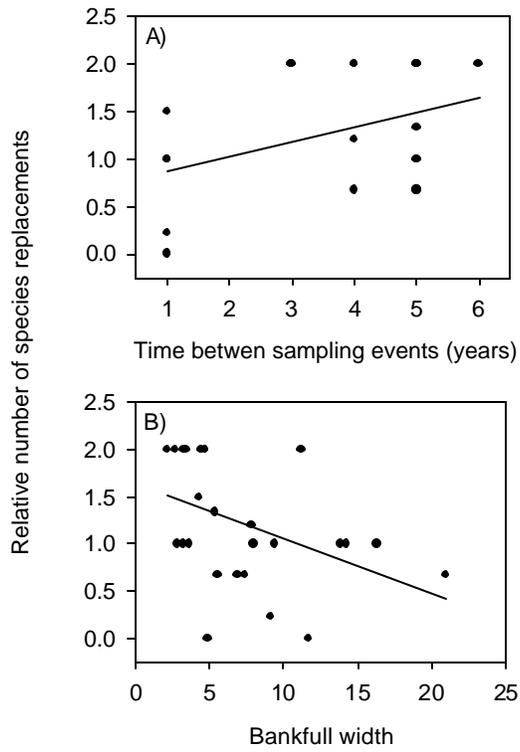


Figure 6. Linear regressions of relative number of species replacements versus time between sampling (A) and stream bankfull width (B) for 27 sites sampled on two occasions between 1995 – 2001 in the Notikewin watershed.

Mean total fish density in the Notikewin watershed was low and ranged from 0.24 ± 0.13 to 2.2 ± 1.39 100 m^2 . Densities of Arctic grayling and brook stickleback, the most numerically dominant species at the 27 sites sampled repeatedly during 1995 – 2001, were also low and typically ranged between 0.1 to 1.8 and to 0.01 to 0.3 100 m^2 , respectively. Densities of lake chub, finescale dace, longnose suckers, pearl dace, sculpin, trout perch and white sucker seldom exceeded 0.2 100 m^2 .

Densities of individual species was also temporally variable and varied by about 10-fold between 1995 and 2001.

Densities of individual species was also temporally variable and varied by about 10-fold between 1995 and 2001 (Highest annual mean $\pm 1\text{SE} = 2.2 \pm 1.39$ 100 m^2 , lowest annual mean = 0.24 ± 0.13 100 m^2) (Figure 7). Density of Arctic grayling was also highly variable and differed 20-fold during the six year period (1.82 ± 1.1 to 0.088 ± 0.05 100 m^2). Densities of stickleback and other numerically dominant were also variable among years and ranged between 5 and 25 fold during the six year study period (Figure 7).

Despite low sample sizes, total fish density and density of Arctic grayling ($P = 0.06$) varied significantly between 1996 and 2001 and 1999 and 2002 (Wilcoxon paired sign rank tests) (Figure 8). Total density and density of

Arctic grayling were significantly lower in 2001 and 2002 compared with that in 1996 and coincided with an overall decline in fish densities between 1999 and 2000 (Figure 7). Significant differences in density of the remaining species between years were not detectable presumably due to low sample sizes (N = 4 to 7) and reduced statistical power (Figure 8).

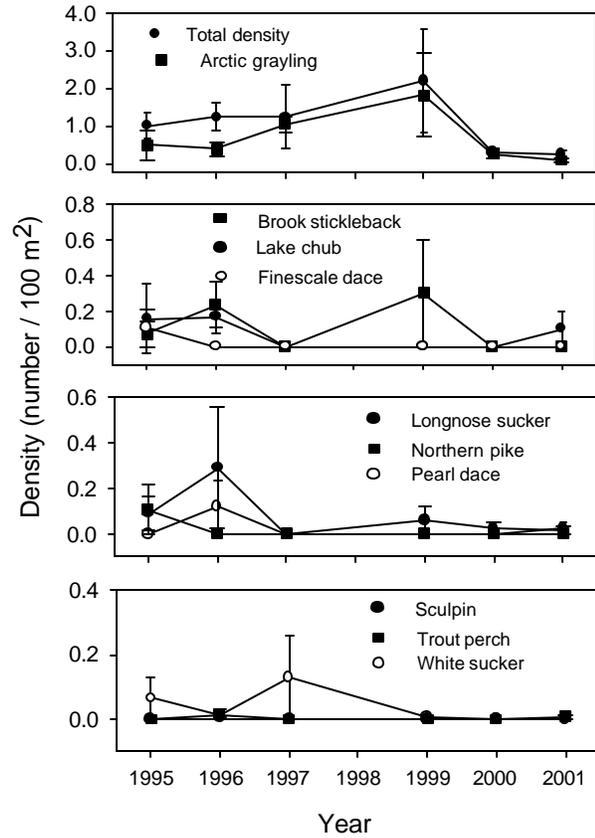


Figure 7. Temporal patterns in total fish density and density of selected species in the Notikewin watershed, 1995-2001.

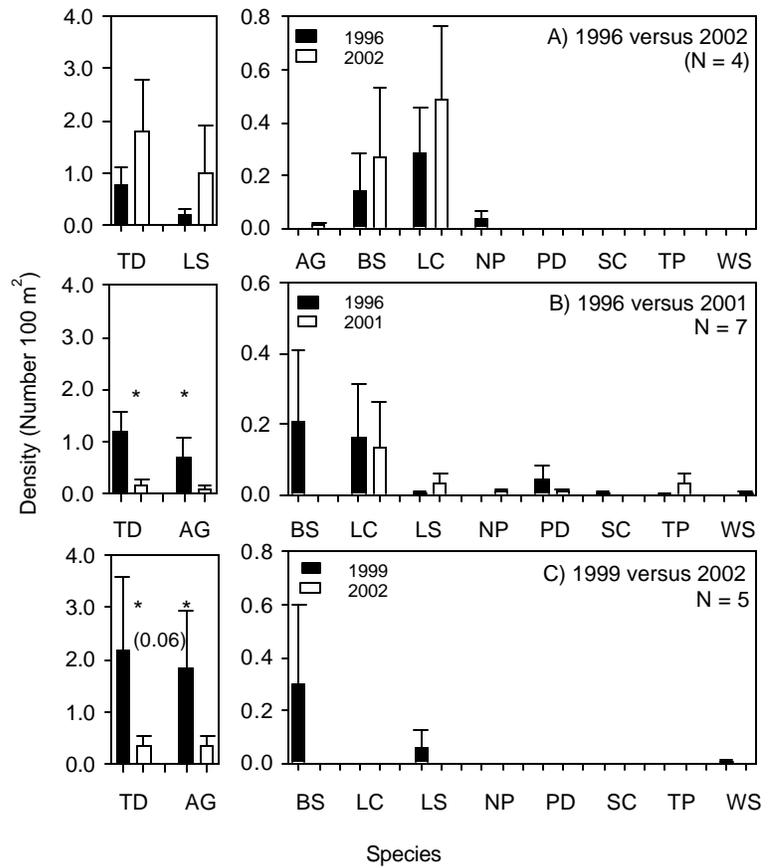


Figure 8. Pairwise comparisons of mean fish density and density of selected taxonomic groups between 1995 and 1996 (A), 1996 and 2001 (B) and 1999 and 2001 (C) in the Notikewin River. Comparisons were based on sites sampled on each of the two years occasions with Wilcoxon paired sin rank tests. * = significant. Species codes: TD = total density, AR = Arctic grayling, BS = brook stickleback, LC = lake chub, NP = northern pike, PD = pearl dace, SC = sculpin, TP = trout perch, WS white sucker.

Focal question 2 – Is the presence of fish, game fish and individual species predictable based on stream habitat and watershed variables?

Models predicting the presence of fish were also derived for: 1) 1st to 5th order reaches and 2) small and large permanent streams.

5.0 FOCAL QUESTION 2 - IS THE PRESENCE OF FISH, GAME FISH AND INDIVIDUAL SPECIES PREDICTABLE BASED ON STREAM HABITAT AND WATERSHED VARIABLES?

5.1 Design and statistical analyses

Logistic regression was used to predict the presence of fish, game fish, and individual species at the landscape and stream-scales (i.e., stream order). At the landscape level, the majority of variables describing stream size (i.e., 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 landscape models we chose water depth as the measure of stream size because it was unrelated ($P > 0.05$) to elevation, percent deciduous forest, percent undercut banks and canopy cover. We also derived single variable models predicting the presence of fish based on bankfull width, wetted width and discharge.

Models predicting the presence of fish were also derived separately for 1st to 5th order reaches (i.e., based on Strahler stream classes) and small permanent and large permanent streams (Alberta watercourse classes). Stream reach-scale models provide the opportunity to quantify the predictive power of a larger set of variables on fish presence because many of the variables related to stream size at the landscape scale are not significantly related to stream, channel and riparian attributes within stream 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.

5.2 Results

5.2.1 Data screening

Our data screening exercise indicated that 75 of the 266 sites were located within 3 km of an adjacent site sharing the same stream order, sampled ineffectively due to low conductivity, or were sampled more than once during the study period. When these sites were excluded and overall averages used to describe sites sampled on two or more occasions, spatial patterns in the presence of fish were evaluated using a total of 191 sites. When classified into Strahler stream orders and Alberta watercourse classifications, sites included 1st to 5th order reaches and small and large permanent streams (Table 11).

Fish communities at the reduced number of 191 sites (Table 12) in the Notikewin watershed were similar to that defined using the original 266 sites although assemblages comprised 16 rather than 17 species (Table 8). Arctic grayling, longnose sucker, white sucker, lake chub, brook stickleback, trout perch and northern pike were the most frequently encountered species and occurred at 7.8 to 36% of all sites (Table 12). Comparisons of relative abundance showed that Arctic grayling, lake chub, brook stickleback, longnose sucker, and finescale dace were numerically dominant and, when combined, accounted for almost 86% of all fish collected. Overall density of fish in the Notikewin Watershed was also low and small-bodied fishes numerically dominated assemblages.

5.2.2 Frequency of occurrence

The occurrence of fish was strongly affected by stream size). While about 40% of first and second order and small permanent reaches contained fish, the majority (0.65 to 0.90%) of third, fourth, fifth and large permanent stream reaches contained fish.

The occurrence of fish was strongly affected by stream size (Figure 9). While about 40% of first and second order and small permanent reaches contained fish, the majority (0.65 to 0.90%) of third, fourth, fifth and large permanent stream reaches contained fish (Figure 10). Stream size also strongly affected the presence of individual species and species groups. Game fish, cyprinids, catostomids, and Arctic grayling occurred infrequently (<35% occurrence) in small streams (1st to 3rd and small permanent streams) but occurred 20 to 40 times more frequently in larger streams (Figure 10). Brook stickleback, lake chub, longnose sucker, white sucker, northern pike, and trout perch occurred most frequently in larger streams (Figure 10).

5.2.3 Logistic regressions

At the landscape level, the presence of fish, game, cyprinid minnows and Arctic grayling were highly predictable based on watershed and instream habitat variables.

Landscape level models. At the landscape level, logistic regression indicated that the presence of fish, game, cyprinid minnows and Arctic grayling were highly predictable (jack knife classification success = 0.71 to 0.78) based on stream bankfull width or bankfull and UTM northing (Cyprinidae) or bankfull width and percent gravel (Arctic grayling) (Table 13). In contrast, reach elevation was the strongest predictor of the presence of suckers, longnose sucker, lake chub and brook stickleback. While reach slope was the strongest predictor of the presence of northern pike, bankfull width and elevation, which were correlated (Pearson correlation $P < 0.05$) with reach slope were also statistically significant single variable predictors (Table 13). However, single variable models using these predictors explained less variance in the presence of northern pike and resulted in lower classification success. These models generally explained between 0.42 to 0.74 of the variance in the presence of fish, fish type and individual species and correctly classified sites where fish should be present or absent compared to presence based on electroshocking between 71% and 93% of the time, respectively. In the majority of cases, variables that predicted the presence or absence of fish were measures of stream size or related to measures of stream size (Table 14). Characteristics of first to fifth order stream reaches and small and large permanent stream reaches are provided in Table 15.

In the majority of cases, variables that predicted the presence or absence of fish were measures of stream size or related to measures of stream size.

Table 12. Frequency of occurrence (%), percent composition (%) and mean (± 1 SE) density (number / 100 m²) of fish from 191 sites in the Notikewin watershed, 1995-2001. Data are overall averages from first to fifth order stream reaches. Percent composition data were calculated from density estimates (i.e., numbers of fish 100 m²).

Common name	Frequency of occurrence	Percent composition	Mean density composition
<u>Cyprinidae</u>			
Lake chub	20.42	15.88	0.26 \pm 0.11
Flathead chub	1.05	0.24	0.004 \pm 0.003
Finescale dace	3.14	13.31	0.22 \pm 0.15
Pearl dace	3.14	2.21	0.04 \pm 0.02
Longnose dace	4.19	1.60	0.03 \pm 0.01
Northern redbelly dace	0.52	1.52	0.02 \pm 0.02
Emerald shiner	2.62	1.58	0.03 \pm 0.01
Northern pikeminnow	0.52	0.15	0.002 \pm 0.002
<u>Percopsidae</u>			
Trout-perch	7.33	1.52	0.02 \pm 0.01
<u>Gasterosteidae</u>			
Brook stickleback	13.9	25.19	0.41 \pm 0.21
<u>Percidae</u>			
Walleye	1.05	0.12	0.002 \pm 0.001
<u>Salmonidae</u>			
Arctic grayling	36.1	17.31	0.28 \pm 0.05
Mountain whitefish	0	0	0
<u>Esocidae</u>			
Northern pike	7.85	1.94	0.03 \pm 0.01
<u>Catostomidae</u>			
Longnose sucker	18.85	13.80	0.23 \pm 0.11
White sucker	9.95	3.09	0.05 \pm 0.01
<u>Cottidae</u>			
Sculpin species	3.66	0.55	0.009 \pm 0.004

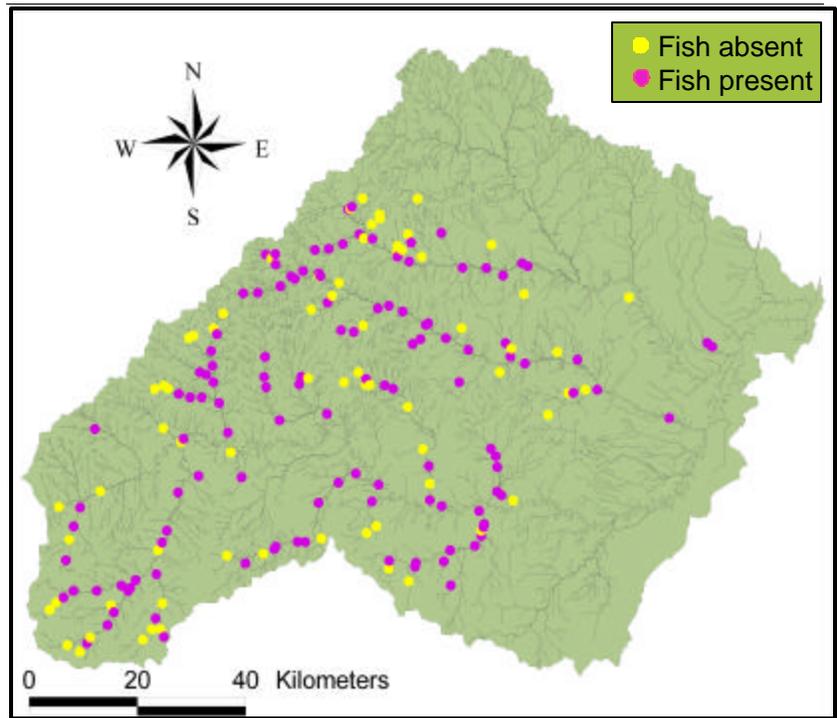


Figure 9. Comparison of sites in the upper Notikewin, Hotchkiss and Meikle Sub-basins. Showing sites supporting fish (purple filled circles) and those where fish were not detected (yellow filled circles).

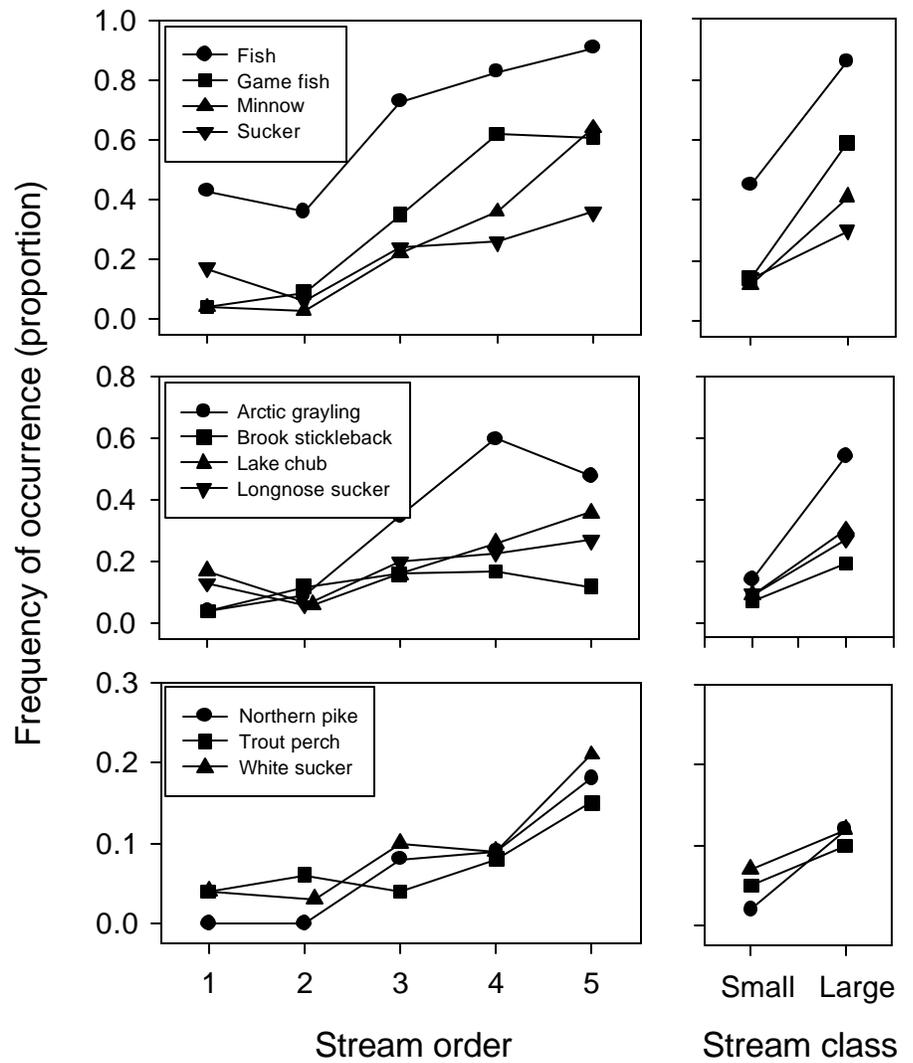


Figure 10. Frequency of occurrence of fish, game fish, fish species groups and individual species in first to fifth order streams and small and large permanent streams in the upper Notikewin, Hotchkiss and Meikle Sub-basins. Stream order = Strahler stream order, Stream class = Alberta watercourse classes: small = small permanent, large = large permanent streams

Table 13. Summary of landscape-scale logistic regression models predicting the presence of fish (logit) based on watershed and stream characteristics in the Notikewin watershed. Analyses were restricted to commonly encountered species or species groups. Game fish = Arctic grayling, walleye, northern pike; Catostomidae = white sucker and longnose sucker; Cyprinidae = 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 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 ($P > 0.05$), * = $P < 0.05$, ** = $P < 0.01$, *** $P < 0.001$. UTM-N= Universal Transverse Mercator Northing. The highly non-statistically model ($P > 0.20$) for white sucker is not shown.

Species group	Logistic equation	N	P	R ²	Classification success
Fish	$Y = -2.25 (***) + 0.338 (***)$ bankfull width	189	<0.001	0.74	0.75
Game fish	$Y = -1.93 (***) + 0.195 (***)$ bankfull width	189	<0.001	0.71	0.71
Cyprinidae	$Y = 128.1 (*) + 0.18 (***)$ bankfull width + 0.00002 (*) UTM-N	189	<0.0001	0.69	0.78
Catostomidae	$Y = 5.51 (***) - 0.0009 (***)$ elevation	191	<0.0001	0.66	0.76
Arctic grayling	$Y = -3.05 (***) + 0.15 (***)$ bankfull width + 3.74 (*) gravel	189	<0.0001	0.73	0.73
Lake chub	$Y = 5.81 (***) - 0.01 (***)$ elevation	191	<0.001	0.64	0.76
Brook stickleback	$Y = 1.42 (NS) - 0.004 (***)$ elevation	191	<0.05	0.55	0.86
Northern pike	$Y = -1.42 (***) - 168.2 (*)$ reach slope	189	<0.05	0.42	0.92
Longnose sucker	$Y = 4.91 (***) - 0.009 (***)$ elevation	191	<0.0001	0.62	0.81

Table 14. Comparisons of mean (± 1 STDEV) environmental variables predicting the presence of fish in the Notikewin watershed. Predictors were based on logistic regression models.

Species group	Absent Mean	Min	Max	N	Present Mean	Min	Max	N
Fish								
Bankfull width (m)	3.86 (2.93)	0.40	13.8	80	10.01 (6.29)	1.9	31.40	109
Game fish								
Bankfull width (m)	5.22 (4.50)	0.40	23.4	116	10.88 (6.38)	2.1	31.40	73
Cyprinidae								
Bankfull width	5.72 (4.83)	0.4	31.4	136	11.74 (6.44)	2.6	25.8	53
UTM-N	6327244.0 (23373.3)				6318616(19099.1)			
	6281941.5	6363116	138	6283465		6352697.5	53	
Catostomidae								
Elevation	797.1 (113.5)				695.3 (111.0)			
	448.5	1060.3	149	457.6		941.1	42	
Arctic grayling								
Bankfull width	5.67 (5.18)	0.40	31.4	121	10.51 (6.10)	2.10	25.8	68
Gravel	0.26 (0.23)	0.00	0.96	121	0.41 (0.18)	0.00	0.91	68
Lake chub								
Elevation	798.2(117.6)	448.5	1060.3	151	685.8 (84.6)	457.6	898.9	40
Brook stickleback								
Elevation	782.8 (111.8)	448.5	1060.3	165	723.1 (157.7)	457.6	1036.9	26
Northern pike								
Slope	0.0097 (0.0084)	0.0	0.049	176	0.0040 (0.0052)	0.0	0.0185	15
Longnose sucker								
Elevation	794.4 (115.4)	448.5	1060.3	155	690.0 (104.6)	457.6	921.7	36

Table 15. Mean (± 1 STD) and ranges of environmental variables and fish assemblage richness from 191 sites in the Notikewin watershed, Alberta. Stream order and classes: Strahler stream orders: 1st to 5th, Alberta watercourse: small and large permanent. All sites = all 191 sites, m.a.s.l. = meters above sea level, dissolved oxygen = mg/L, woody debris volume = m³/100 m², adjacency to 5th = distance (km) to nearest 5th order reach. Data for water pH and dissolved oxygen at 1st to 5th order streams and woody debris volumes at 5th order sites are not presented due to low sample sizes. Woody debris volumes and instantaneous discharge were measured at about one half of all study sites.

Variables	Stream order and class				
	1 st N = 23	2 nd N = 33	3 rd N = 49	4 th N = 53	5 th N = 33
<u>Watershed</u>					
Watershed area (km ²)	115.9±172.7 (0.2–1217.2)	123.1±248.9 (0.5–1305.4)	146.6±224.65 (1.44–1217.3)	103.6±164.0 (2.24–808.94)	169.6±190.2 (2.35–635.1)
Percent forest	0.46±0.29 (0.02 – 0.80)	0.44±0.28 (0.01 –0.91)	0.49±0.25 (0.00 – 0.89)	0.58±0.27 (0.02 – 0.96)	0.62±0.20 (0.26 – 0.99)
Percent conifer	0.34±0.21 (0–0.85)	0.33±0.21 (0.03–0.71)	0.36±0.19 (0–0.72)	0.33±0.20 (0.02–0.85)	0.36±0.18 (0.0–0.66)
Percent deciduous	0.10±0.14 (0–0.77)	0.07±0.14 (0–0.40)	0.05±0.09 (0–0.32)	0.17±0.15 (0–0.64)	0.15±0.18 (0–0.77)
Slope (%)	2.34±1.0 (0.68–4.53)	3.01±1.33 (0.81–7.01)	2.65±0.14 (0.68–4.47)	2.56±1.10 (0.76–5.08)	0.02±0.01 (0.98–3.44)
<u>Stream and riparian</u>					
Elevation (masl)	830±153 (449–1060)	837±102 (583–1030)	814±115 (458–1037)	721.8±88.5 (464.1–904.1)	669.5±87.7 (556.8–931.9)
Discharge (m ³ s)	0.13±0.16 (0.01–0.57)	0.20±0.27 (0.05–1.08)	0.54±0.42 (0.01–1.74)	1.19±0.77 (0.17–3.13)	3.52±3.34 (0–8.21)
Bankfull width (m)	3.2±3.4 (0.4–12.7)	3.8±3.7 (0.9–18.3)	5.1±2.7 (1.2–12.1)	8.7±4.6 (1.3–23.2)	15.2±6.6 (3.3–31.4)
Wetted width (m)	1.85 ±1.49 (0.40–7.80)	2.22±1.48 (0.70–8.30)	3.74±1.75 (0.90–8.40)	6.69±3.46 (1.3–19.5)	11.14±5.49 (2.20–25.0)
Depth (m)	0.39±0.20 (0.10–0.90)	0.40±0.18 (0.10–0.90)	0.38±0.14 (0.10–0.80)	0.45±0.16 (0.2–1.0)	0.51±0.19 (0.2–1.0)
Percent fines	0.68±0.32 (0.07–1.00)	0.59±0.34 (0–1.19)	0.48±0.32 (0–1.0)	0.34±0.25 (0–1.0)	0.38±0.29 (0–1.0)
Percent gravel	0.22±0.23 (0–0.63)	0.28±0.25 (0–0.86)	0.33±0.22 (0–0.82)	0.37±0.20 (0–0.96)	0.33±0.20 (0–0.86)
Percent cobble	0.09±0.14 (0–0.47)	0.09±0.10 (0–0.31)	0.49±0.14 (0–0.49)	0.19±0.15 (0–0.60)	0.19±0.16 (0–0.53)
Temperature (°C)	10.9±4.5 (3.0–23.0)	11.1±4.3 (4.0–24.0)	12.3±3.8 (5.2–20.0)	12.0±3.6 (6.5–19.0)	13.3±3.1 (7.5–18.0)
Canopy cover	0.35±0.33 (0–0.94)	0.31±0.29 (0.01– 0.89)	0.18±0.20 (0–0.80)	0.16±0.16 (0–0.68)	0.27±0.30 (0.03–0.93)
Stream reach slope	0.94±0.81 (0–3.66)	1.02±0.91 (0–3.53)	0.72±0.52 (0–1.95)	1.03±0.94 (0–4.89)	0.95±0.94 (0–3.60)
Adjacency to 5 th	13.7±13.5 (0–58.6)	9.9±9.9 (0–42.1)	14.7±19.0 (0–74.9)	3.92±5.89 (0–21.9)	0±0 (0 - 0)
Woody debris	0.46±0.7 (0–2.62)	1.38±3.13 (0–15.9)	0.95±1.36 (0–4.74)	0.13±0.13 (0–0.34)	-
<u>Fish community</u>					
Number of species	0.1±0.46 (0–2)	0.3±0.57 (0–2)	1.0±1.0 (0–4)	1.6±1.7 (0–6)	3.1±1.9 (0–8.0)

Variables	Stream class		
	All N = 191	Small permanent N = 85	Large permanent N = 105
<u>Watershed</u>			
Watershed area (km ²)	115.3±172.7 (0.2–1217.2)	86.3±128.8 (0.4–644.4)	139.0±200.5 (0.25–1217.3)
Percent forest	0.52±0.27 (0 – 0.99c)	0.418±0.26 (0 - 0.872)	0.60±0.24 (0.05- 0.99)
Percent conifer	0.34±0.21 (0–0.85)	0.34±0.22 (0–0.72)	0.34±0.19 (0–0.85)
Percent deciduous	0.10±0.14 (0–0.77)	0.02±0.04 (0–0.50)	0.16±0.16 (0–0.77)
Slope (%)	2.53±1.1 (0.68–7.01)	2.76±1.16 (0.68–7.01)	2.38±0.99 (0.76–5.08)
<u>Stream and riparian</u>			
Elevation (masl)	775±120 (449–1060)	871.6±83.0 (723.5–1060.3)	701.6±77.4 (457.6–875.7)
Discharge (m ³ s)	0.86±1.37 (0–8.21)	0.33±0.35 (0.01–1.38)	1.48±1.81 (0–0.82)
Bankfull width (m)	7.4±6.0 (0.4–31.4)	2.8±1.4 (0.4–9.4)	11.2±5.6 (5.0–31.4)
Wetted width (m)	5.4±4.4 (0.4–25)	2.3±1.1 (0.4–7.0)	7.9±4.5 (1.10–25.0)
Depth (m)	0.42±0.17 (0.1–1.0)	0.40±0.17 (0.1–0.9)	0.45±0.17 (0.20–1.0)
Percent fines	0.47±0.32 (0–1.0)	0.58±0.32 (0–1.0)	0.37±1.0 (0-1.0)
Percent gravel	0.31±0.22 (0-0.96)	0.27±0.22 (0–0.79)	0.36±0.21 (0–0.96)
Percent cobble	0.15±0.15 (0–0.6)	0.11±0.13 (0–0.49)	0.18±0.15 (0–0.60)
Water temperature (°C)	12.1±3.8 (3.0–24.0)	11.5±4.2 (3.0–24.0)	12.5±3.5 (4.0–20.0)
Canopy cover	0.24±0.25 (0–0.94)	0.28±0.29 (0–0.94)	0.17±0.17 (0–0.74)
Dissolved oxygen (mg/L)	8.1±1.2 (4.1–10.4)	7.7±1.4 (4.1-10.4)	8.6±0.8 (7.2–10.2)
Water pH	7.2±0.7 (5.4–8.9)	7.1±0.7 (5.4–8.7)	7.3±0.6 (5.7–8.9)
Channel slope (%)	0.92±0.83 (0–4.89)	0.93±0.01 (0–3.66)	0.91±0.90 (0–4.89)
Distance to 5 th order (km)	11.5±13.9 (0–74.9)	14.9±16.7 (0–74.9)	8.7±10.62 (0-46.1)
Woody debris (m ³ /m ²)	0.90±1.97 (0–15.89)	0.93±2.01 (0-15.89)	0.19±0.14 (0–0.34)
<u>Fish community</u>			
Number of species	1.3±1.6 (0–8)	0.4±0.8 (0.1–4)	2.0±1.8 (0 – 8)

Stream bankfull width, water temperature, slope and to a lesser extent, elevation and gravel were significant predictors of the presence of fish, game fish, taxonomic groups and individual species in first to fifth order reaches.

Reach-scale models. At the stream reach level, bankfull width, water temperature, slope and to a lesser extent, elevation and gravel were significant predictors of the presence of fish, game fish, taxonomic groups and individual species in first to fifth order reaches (Table 16). After excluding models which were not calculated due to low sample sizes, 13 of the 20 logistic regression models were statistically significant ($P < 0.05$) and all were statistically significant at $P < 0.08$. These models explained between 0.51 and 0.75% of variance in the presence or absence of fish ($\text{Mean} \pm 1 \text{ standard deviation} = 0.683 \pm 0.07$) and on average correctly classified sites as containing fish or where fish were absent in the majority of cases ($\text{Mean} \pm 1 \text{ standard deviation} = 0.756 \pm 0.09$, Range = 0.59 to 0.91) (Table 16). Relationships between the probability of occurrence of fish, game fish and individual species and species groups are shown in Figure 11.

When analyses were based on classifying reaches using Alberta watercourse classes, reach elevation, bankfull width and to a lesser extent, slope and gravel were statistically significant predictors of the presence of fish, game fish, taxonomic groups and individual species in small and large permanent stream reaches. In total, 10 of the 12 models (i.e., 83.3%) were statistically significant ($P < 0.05$) and 11 of the 12 models were statistically significant at $P < 0.06$. These models explained between 0.53 and 0.87% of variance in the presence or absence of fish ($\text{Mean} \pm 1 \text{STDEV} = 0.68 \pm 0.08$) and on average correctly classified sites as containing fish or where fish were absent in the majority of cases ($\text{Mean} \pm 1 \text{STDEV} = 0.714 \pm 0.10$, Range = 0.53 to 0.87).

We evaluated the effects of stream classification method on the ability of logistic regression models to predict the presence and absence of fish.

We evaluated the effects of stream classification method on the ability of logistic regression models to predict the presence and absence of fish by comparing model fit and classification success between models created using Strahler stream orders and that using Alberta watercourse classes (Table 16, 17). With two exceptions, the stream classification method had only minor effects on the fit of logistic regression models and overall classification success. However, the ability of logistic regressions to correctly classify large permanent reaches as containing game fish and Arctic grayling were low (classification successes = 0.53 and 0.59) and potentially problematic if the ability to predict the presence of game fish and Arctic grayling is an important criterion in developing predictive models (Table 16, 17).

With two exceptions, the stream classification method had only minor effects on overall classification success.

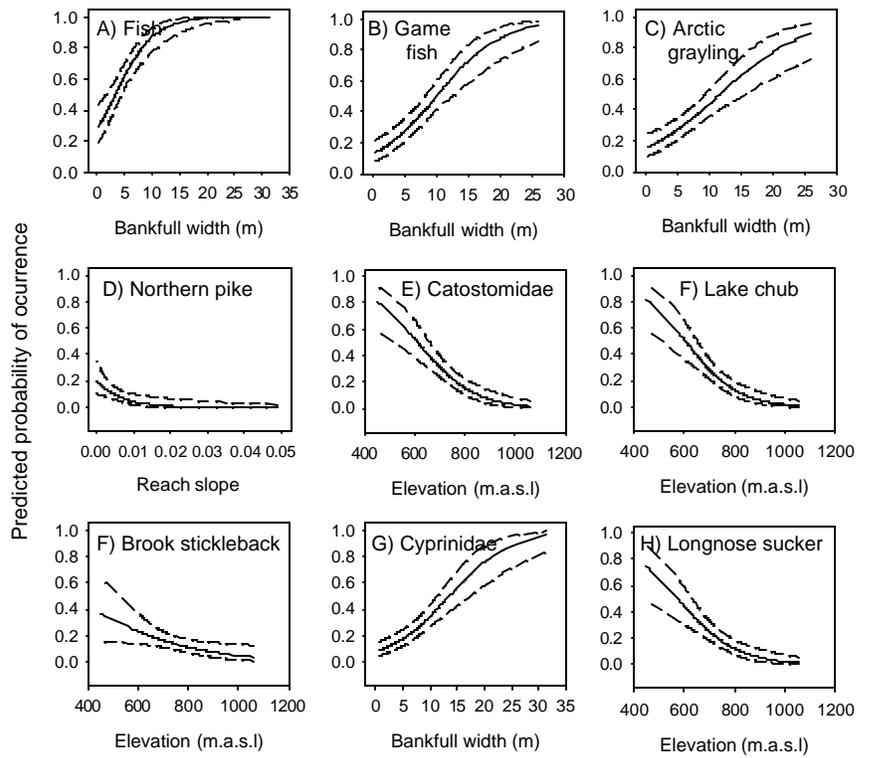


Figure 11. Predicted probability of occurrence of fish, game fish and individual species and species groups versus the first logistic regression predictor in streams in the Notikewin River Basin.

Table 16. Summary of logistic regression models predicting the presence of fish (logit) in first to fifth order reaches (A) and small and large permanent reach sites (B) in the Notikewin watershed. Analyses were restricted to commonly encountered species or species groups. Game fish = Arctic grayling, walleye, northern pike; 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. Highly non-statistically significant models ($P > 0.25$) are not shown. 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.

Species group	Logistic equation	N	P	R ²	Classification success
A) Strahler stream order					
First order reaches					
Fish	Y = -5.57 (0.07) - 0.007 (0.06) elevation	23	0.056	0.75	0.65
Sucker	Y = 6.93 (0.078) - 0.011 (0.037) elevation	23	0.060	0.60	0.83
Second order reaches					
Fish	Y = -2.82 (0.05) + 0.20 (0.06) temperature	32	0.071	0.73	0.59
Brook stickleback	Y = -3.00 (0.001) + 0.21 (0.07) bankfull width	32	0.042	0.60	0.78
Third order reaches					
Fish	Y = -0.71 (0.367) + 0.39 (0.032) bankfull width	49	0.033	0.69	0.76
Game fish	Y = -3.63 (0.012) + 0.28 (0.045) bankfull width + 3.96 (0.034) conifer-300	49	0.032	0.73	0.63
Minnow	Y = -5.46 (0.039) + 0.15 (0.018) temperature	44	0.011	0.68	0.77
Sucker	Y = 0.95 (0.22) - 361.6 (0.011) reach slope	49	0.042	0.68	0.75
Arctic grayling	Y = -1.94 (0.001) + 3.73 (0.034) gravel	49	0.019	0.72	0.67
Fourth order reaches					
Game fish	Y = -1.96 (0.046) + 0.31 (0.001) bankfull width	52	0.013	0.74	0.68
Minnow	Y = -2.92 (0.001) + 0.27 (0.045) bankfull width	52	0.003	0.73	0.75
Sucker	Y = 5.49 (0.059) - 0.009 (0.034) elevation	53	0.067	0.69	0.74
Arctic grayling	Y = 0.50 (0.70) + 0.26 (0.021) bankfull width - 0.18 (0.047) temperature	50	0.026	0.74	0.74
Lake chub	Y = -1.54 (0.42) - 147.5 (0.007) reach slope + 1.76 (0.089) conifer-300	53	0.019	0.68	0.71

Species group	Logistic equation	N	P	R2	Classification success
Fifth order reaches					
Game fish	$Y = -2.02 (0.067) + 0.17 (0.020)$ bankfull width	33	0.022	0.74	0.73
Minnow	$Y = -3.98 (0.031) + 0.35 (0.013)$ temperature	33	0.060	0.73	0.67
Sucker	$Y = 1.01 (0.23) - 657.4 (0.041)$ reach slope	33	0.042	0.51	0.82
Arctic grayling	$Y = -0.30 (0.90) + 0.15 (0.12)$ bankfull width - $0.36 (0.162)$ temperature + $8.13 (0.02)$ gravel	33	0.041	0.65	0.82
Lake chub	$Y = 10.33 (0.063) - 1028.7 (0.024)$ reach slope - $0.48 (0.12)$ temperature	33	0.052	0.73	0.85
Longnose sucker	$Y = 12.69 (0.022) - 598.9 (0.042)$ reach slope - $0.87 (0.19)$ temperature	33	0.057	0.69	0.91
B) Alberta watercourse classes					
Small permanent (channel widths = 0.5 to 5 m)					
Fish	$Y = -1.32 (0.026) + 0.56 (0.007)$ bankfull width - $1.68 (0.14)$ proportion gravel	84	0.022	0.75	0.68
Game fish	$Y = -4.41 (0.001) + 0.81 (0.007)$ bankfull width	84	0.051	0.57	0.87
Minnow	$Y = -2.30 (0.011) + 0.41 (0.069)$ bankfull width - $3.61 (0.043)$ conifer	84	0.022	0.75	0.68
Sucker	$Y = -1.32 (0.009) - 58.37 (0.282)$ reach slope	85	0.28	0.56	0.68
Large permanent (channel widths >0.5 m)					
Fish	$Y = -1.10 (0.904) + 0.198 (0.031)$ elevation	104	0.031	0.56	0.86
Game fish	$Y = -0.63 (0.186) + 0.091 (0.029)$ bankfull width	104	0.028	0.76	0.53
Minnow	$Y = -2.03 (0.001) + 0.150 (0.005)$ bankfull width	104	0.0005	0.74	0.68
Sucker	$Y = 18.01 (0.001) - 0.024 (0.0001)$ elevation - 4.86 forest cover	105	0.0001	0.71	0.77
Arctic grayling	$Y = -0.86 (0.001) + 0.205 (0.001)$ bankfull width - $3.42 (0.413)$ elevation	105	0.0029	0.75	0.59
Brook stickleback	$Y = 4.345 (0.062) - 0.008 (0.014)$ elevation	105	0.014	0.62	0.83
Lake chub	$Y = 9.88 (0.0009) - 0.013 (0.0006)$ elevation - $2.79 (0.015)$ forest	105	0.0017	0.71	0.68
Longnose sucker	$Y = 0.25 (0.001) - 197.9 (0.001)$ reach slope	105	0.004	0.69	0.70

Table 17. Comparisons of mean (\pm 1 STDEV) environmental variables predicting the presence of fish in the Notikewin watershed. Predictors were based on logistic regression models of first to fifth order streams (i.e., Strahler stream classification) and small and large permanent streams (i.e., Alberta watercourse classification).

Species group	Absent Mean	Min	Max	N	Present Mean	Min	Max	N
First order reaches								
Fish								
Elevation	888.1 (125.5)	627.5	1060.3	13	754.9 (159.3)	448.5	921.7	10
Sucker								
Elevation	867.4 (117.7)	627.5	1060.3	19	653.4 (197.7)	448.5	921.7	4
Second order reaches								
Fish								
Temperature	3.50 (3.26)	1.0	13.8	21	4.27 (4.59)	0.9	18.3	11
Brook stickleback								
Bankfull width	3.26 (2.84)	0.90	13.8	29	7.58 (7.22)	2.8	18.3	4
Third order reaches								
Fish								
Bankfull width	3.62 (2.07)	1.2	8.1	13	5.63 (2.75)	1.9	12.1	36
Game fish								
Bankfull width	4.50 (2.64)	1.2	12.1	32	6.22 (2.55)	2.8	11.0	17
Conifer-300m	0.381 (0.275)	0	1.0	32	0.267 (0.327)	0	0.966	17
Cyprinidae								
Temperature	11.41 (3.60)	5.2	20.0	33	15.14 (2.91)	10.6	19.0	11
Catostomidae								
Reach slope	0.0084 (0.005)	0	0.0195	37	0.0034 (0.0022)	0	0.0065	12
Arctic grayling								
Gravel	0.267 (0.216)	0.00	0.789	32	0.435 (0.203)	0.01	0.820	17

Species group	Absent Mean	Min	Max	N	Present Mean	Min	Max	N
Fourth order reaches								
Game fish								
Bankfull width	6.305 (2.875)	1.30	12.40	20	10.220 (4.948)	5.0	23.20	32
Cyprinidae								
Bankfull width	7.04 (3.74)	1.30	23.20	33	11.63 (4.74)	5.40	20.95	19
Catostomidae								
Elevation	738.7 (75.9)	578.3	875.7	39	675.0 (106.2)	464.1	904.1	14
Arctic grayling								
Bankfull width	6.60 (3.12)	1.30	12.60	20	10.14 (5.01)	5.00	23.20	30
Temperature	13.09 (3.72)	6.60	19.00	20	11.33 (3.35)	6.50	18.00	30
Lake chub								
Reach slope	0.012 (.010)	0.00	0.049	39	0.0053 (0.0048)	0.00	0.0159	14
Conifer 300 m	0.344 (0.320)	0.00	1.00	39	0.490 (0.344)	0.00	1.00	14
Fifth order reaches								
Game fish								
Bankfull width	11.72 (6.73)	3.30	23.40	13	17.45 (5.52)	9.10	31.40	20
Cyprinidae								
Temperature	11.48 (3.12)	7.50	18.00	12	14.34 (2.70)	10.0	18.0	21
Catostomidae								
Reach slope	0.013 (0.001)	0.0009	0.036	21	0.0028 (0.0028)	0.0	0.007	12
Arctic grayling								
Bankfull width	12.97 (7.67)	3.30	31.40	17	17.56 (4.22)	12.1	25.8	16
Temperature	14.31 (3.07)	7.50	18.00	17	12.23 (2.94)	9.00	18.00	16
Gravel	0.215 (0.157)	0.00	0.533	17	0.444 (0.168)	0.13	0.860	16
Lake chub								
Reach slope	0.014 (0.009)	0.001	0.036	21	0.002 (0.005)	0.00	0.006	12
Temperature	13.54 (3.27)	7.50	18.0	21	12.88 (2.99)	9.0	18.0	12

Species group	Absent Mean	Min	Max	N	Present Mean	Min	Max	N
Longnose sucker								
Reach slope	0.0119 (0.0098)	0.00	0.0360	24	0.0030 (0.0030)	0.00	0.0071	9
Temperature	14.14 (2.78)	9.0	18.0	24	11.06 (3.08)	7.5	18.0	9
Small permanent reaches								
Fish								
Bankfull width	2.50 (1.19)	0.80	4.80	46	3.24 (1.51)	0.80	9.40	38
Gravel	0.288 (0.239)	0.0	0.789	46	0.261 (0.209)	0.00	0.600	38
Game fish								
Bankfull width	2.61 (1.19)	0.80	4.80	72	4.10 (1.84)	2.10	9.40	12
Cyprinidae								
Bankfull width	2.72 (1.44)	0.80	9.40	74	3.58 (0.55)	2.60	4.40	10
Conifer	0.351 (0.220)	0.00	0.723	74	0.201 (0.199)	0.0	0.534	10
Catostomidae								
Reach slope	0.0096 (0.008)	0.00	0.0366	73	0.0071 (0.0060)	0.00	0.0185	12
Large permanent reaches								
Fish								
Elevation	722.3 (78.2)	594.0	875.7	14	695.5 (80.6)	457.6	866.0	90
Game fish								
Bankfull width	9.71 (4.49)	5.20	23.40	43	12.21 (6.10)	5.00	31.40	61
Cyprinidae								
Bankfull width	9.45 (4.93)	5.00	31.40	61	13.64 (5.64)	5.40	25.80	43
Catostomidae								
Elevation	722.9 (74.4)	556.8	875.7	73	645.5 (67.4)	457.6	793.8	32
Forest	0.633 (0.269)	0.00	0.992	73	0.518 (0.168)	0.146	0.797	32

Species group	Absent Mean	Min	Max	N	Present Mean	Min	Max	N
Arctic grayling								
Bankfull width	10.38 (5.37)	5.20	31.40	48	11.87 (5.76)	5.00	25.80	56
Elevation	676.5 (83.37)	457.6	875.7	48	718.6 (73.17)	464.1	866.0	57
Brook stickleback								
Elevation	709.1 (70.3)	578.3	875.7	85	657.8 (106.4)	457.6	842.4	20
Lake chub								
Elevation	717.5 (80.9)	464.1	875.7	73	658.0 (63.1)	457.6	805.6	32
Forest	0.626 (0.266)	0.00	0.992	73	0.534 (0.187)	0.094	0.855	32
longnose sucker								
Reach slope	0.0012 (0.0095)	0.00	0.0489	77	0.0037 (0.0036)	0.00	0.0159	28

Focal question 3 - Do stream reach and watershed characteristics shape fish assemblage structure?

Relationships between fish assemblages and stream and watershed variables were determined with canonical correspondence analysis.

Community ordinations showed that instream and watershed variables were significant predictors of fish densities.

6.0 FOCAL QUESTION 3 - DO STREAM REACH AND WATERSHED CHARACTERISTICS SHAPE FISH ASSEMBLAGE STRUCTURE?

6.1 Design and Statistical analyses

Relationships between fish assemblages and stream and watershed variables were determined with canonical correspondence analysis (CCA) after a detrended canonical correspondence analysis revealed a gradient length of about 3 standard deviations (ter Braak and Šmilauer 1998).

Canonical correspondence analyses were performed on fish abundance data from 108 sites that contained fish. In total, six instream and watershed variables (i.e., mean percent gravel, mean water depth, mean bankfull width, forest cover, reach slope, elevation) were chosen via forward selection from an initial set of 16 variables. The remaining variables were excluded from analyses because they were moderately correlated ($r > 0.50$) with other variables. At each step, added variables were tested for statistical significance with Monte Carlo permutations ($N=999$ unrestricted, $P < 0.05$).

Fish species and taxonomic group locations and arrows of the environmental habitat variables in an ordination biplot reflect dominant patterns in community structure composition that can be best explained by the habitat variables. The length of the arrow determines the influence of an environmental variable on fish data (the longer the arrow, the stronger the influence). Perpendicular lines drawn from the arrow to fish species or taxonomic group determines the relative position of the taxonomic group along that environmental axis.

Lastly, we evaluated relationships between total and species abundances with a suite of watershed, stream reach and forest cover attributes using multiple regression (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 $\log_{10}(x + 1)$ to ensure that data were normally distributed.

6.2 Results

6.2.1 Canonical correspondence analyses

The canonical correspondence ordination indicated that the first two axes explained 24.6% of the total variation in fish density (Axis 1 = 13.21%, Axis 2 = 11.36%) (Figure 12). When constrained by environmental variables, Axes 1 and 2 explained 80.8% of the variation between species abundances and environmental variables (Table 18).

Increased densities of white sucker, northern pike, sculpin and trout perch were positively related with cobble, water depth, bankfull width and gravel. Density of Arctic grayling was positively related to elevation whereas density of longnose sucker was negatively related with elevation (Figure 13, Table 18). Densities of dace, shiner, brook stickleback and chub were negatively related with cobble, gravel and water depth and bankfull width (Figure 13).

6.2.2 Multiple regression

Multiple regressions using forward selection showed that total fish density and density of the dominant groups were only moderately or weakly related with environmental variables (Table 19). Total density and density of individual species and species groups was typically significantly related to cobble, gravel, elevation, water depth and width and elevation. Regression models explained between 5 and 32% of variance in fish densities and often (5 of 9 regression models) accounted for 18 to 32% of variance in individual species and species groups (Table 19). For instance, density of Arctic grayling was positively related with elevation and negatively related with water depth and forest cover. Density of brook stickleback was negatively related with cobble, bankfull depth and gravel whereas density of longnose sucker and trout perch were negatively related with elevation and positively related with bankfull width, respectively (Table 19).

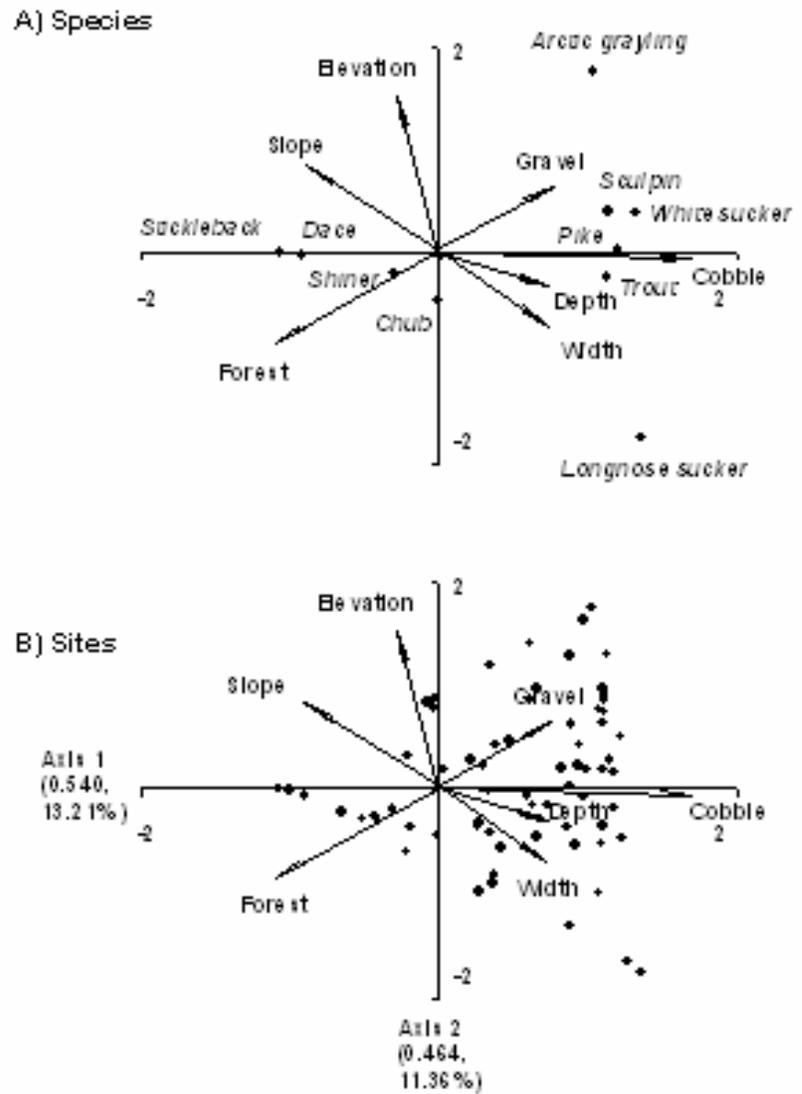


Figure 12. Canonical correspondence analysis (CCA) bi-plots of stream fish species and environmental relations (A) and sampling sites and environmental variables (B) from 108 sites in the Notikewin watershed, 1995-2001. Eigenvalues and variance explained by the first two axes are shown adjacent to each axis. Environmental variables (scores multiplied by 2.5) are shown as arrows. Species abbreviation: dace = finescale dace, pearl dace, longnose dace, northern redbelly dace, trout = trout perch, stickleback = brook stickleback, shiner = emerald shiner, pike = northern pike, chub = lake chub, flathead chub.

Table 18. Summary of canonical correspondence analyses of fish communities in the Notikewin watershed.

Variance in sites explained by Axes 1 and 2	Variance in fish -environment relationships explained by Axes 1 and 2	Environmental variable	Canonical coefficients		Correlation coefficients	
			Axis 1	Axis 2	Axis 1	Axis 2
24.57	80.8	Cobble	0.50	0.06	0.79	-0.001
		Water depth	0.18	0.03	0.34	-0.15
		Bankfull width	-0.07	-0.01	0.35	-0.35
		Gravel	0.11	0.14	0.35	0.32
		Elevation	-0.11	0.52	-0.12	0.75
		Slope	-0.07	0.22	-0.41	0.40
		Forest	-0.20	-0.09	-0.51	-0.41

Table 19. Summary of multiple regression models of relations between fish density and instream and watershed variables in the Notikewin watershed. Regression model r^2 values are shown in parentheses. asin = arc-sin squareroot transformed

Model		F	d.f.	r^2	P
Log ₁₀ total density	= 21.89 – 9.75 log ₁₀ depth (0.051) – 8.45 asin cobble (0.05)	5.91	2,105	0.10	0.004
Log ₁₀ Arctic grayling	= -1.91 + 0.20 log ₁₀ elevation (0.22) – 0.20 log ₁₀ depth (0.05) – 0.12 asin forest (0.04)	16.12	3,104	0.32	<0.0001
Log ₁₀ brook stickleback	= 1.06 – 0.38 asin cobble (0.14) – 0.44 log ₁₀ depth (0.10) – 0.19 asin gravel (0.04)	13.14	3,104	0.27	<0.0001
Log ₁₀ emerald shiner	= 0.04 – 0.06 asin cobble (0.05)	5.79	1,106	0.05	0.02
Log ₁₀ chub	= 2.55 – 0.69 asin elevation (0.09) – 0.24 log ₁₀ depth (0.05) – 0.16 asin gravel (0.04)	7.72	3,104	0.18	<0.0001
Log ₁₀ dace	= 0.16 – 0.24 asin cobble (0.07)	7.57	1,106	0.07	0.007
Log ₁₀ longnose sucker	= 3.22 – 1.10 log ₁₀ elevation (0.21)	28.39	1,106	0.21	<0.0001
Log ₁₀ northern pike	= 0.04 – asin adjacency (0.07)	8.43	1,106	0.07	0.0045
Log ₁₀ sculpin	= 0.02 + 0.03 log ₁₀ bankfull width (0.09)	11.69	1,106	0.10	0.0009
Log ₁₀ trout perch	= 0.06 + 0.08 log ₁₀ bankfull width (0.22)	29.50	1,106	0.21	<0.0001
Log ₁₀ white sucker	NS				

Focal Question 4: To what extent are relations between fish assemblages and watershed variables applicable to adjacent watersheds?

A Reference-condition approach was used to test the generality of empirical models between fish assemblages and watershed characteristics.

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

7.0 FOCAL QUESTION 4 – TO WHAT EXTENT ARE RELATIONS BETWEEN FISH ASSEMBLAGES AND WATERSHED VARIABLES APPLICABLE TO ADJACENT WATERSHEDS?

7.1 Design and Statistical analyses

7.1.1 Empirical models

We tested the generality of empirical relationships between fish assemblages and stream and watershed variables by comparing models area developed in the upper Notikewin Sub-basin (area = 4707.7 km²) with that developed in the adjacent Hotchkiss (area = 1305.4 km²) and Meikle (area = 2,589.7 km²) Sub-basins using a reference-condition approach (Reynoldson *et al.* 1997, 2001). In a traditional reference condition approach, the structure of fish assemblages at a group of test (i.e., potentially impacted) sites are compared to that expected from a set of minimally impacted sites (Reynoldson *et al.* 2001). If fish communities at the test sites are minimally impacted they should be predictable based on environmental variables that predicted fish communities at the east-impacted sites. While these analyses typically focus on minimally impacted sites versus potentially stressed sites, the approach can be used to test for similarity between two sets of data that are expected to be similar (See Reece *et al.* 2001).

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), especially with macroinvertebrate assemblages, but has also recently been applied to fishes (e.g., Joy and Death 2000). A conceptually similar approach was developed by Tonn *et al.* (1983) and Tonn *et al.* (In Press) to classify and predict fish assemblages in lakes.

7.1.2 The Reference Condition approach

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 Notikewin watershed 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 upper Notikewin Sub-basin. 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. Thus, we combined several species within familial groups to form the following taxonomic groupings: 1) chub (lake chub, flathead chub), and 2) dace (pearl dace, finescale dace, longnose dace, northern redbelly dace). To avoid potential problems with the misidentification of slimy sculpin as spoonhead sculpin these species were combined into a single taxonomic group (i.e., sculpin). Northern pike minnow occurred at only three sites and was excluded from analyses.

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 Notikewin Sub-basin through clustering, we used multiple discriminant function analysis (DFA) to create a model to predict assemblage type based on instream and watershed characteristics. While the CCA 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 could be robust to watershed disturbances and easily measured (e.g., mean bankfull width, forest cover, reach slope, elevation).

The spatial utility of the discriminant function model developed for the upper Notikewin Sub-basin was tested by applying it to data from the adjacent Hotchkiss and Meikle Sub-basins.

A multiple discriminant function analysis was used to identify variables that best separated 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).

Model assessment: comparisons to the Hotchkiss and Meikle Sub-basins. We tested the spatial utility of the discriminant function model by applying the model developed in the upper Notikewin Sub-basin to stream sites from the adjacent Hotchkiss and Meikle Sub-basins. This approach determines whether fish assemblages in the Hotchkiss and Meikle Sub-basins conformed with or differed from those expected based on environment-fish relations derived from the Notikewin Sub-basin. Departures in predictions derived from the Notikewin Sub-basins would suggest low model generality.

7.2 Results

7.2.1 Identifying fish assemblages in the Notikewin Sub-basin

Hierarchical cluster analyses of fish density identified three relatively discrete fish assemblages in the Notikewin Sub-basin.

Hierarchical cluster analyses of fish density identified three relatively discrete fish assemblages in the Notikewin Sub-basin and a fourth cluster of residual sites (Figure 13). Based on fish density data, assemblage 1 consisted primarily of Arctic grayling, brook stickleback and dace whereas assemblage 2 consisted of Arctic grayling, chub, white sucker, longnose sucker and northern pike (Table 19). For these assemblages, mean total density and density of the three most numerically abundant groups ranged from 1.42 to 1.77 / 100 m² and 0.15 to 0.69 / 100 m², respectively. In contrast, assemblage 3 was comprised of high densities of brook stickleback, dace, chub and to a lesser extent longnose sucker. Overall, mean total density and density of the three most abundant species was 12 to 15 and 18 fold higher than that in assemblages 1 and 2 (Table 19).

7.2.2 Discriminating among fish assemblages in the Notikewin Sub-basin

Instream and watershed variables of percent gravel and cobble, site elevation and reach slope were significant predictors of the three fish assemblages.

Based on fish density data, the forward selection discriminant analysis identified percent gravel and cobble, site elevation and reach slope as significant (Wilks' Lambda, $P < 0.0001$) discriminators among the three fish assemblages. The linear discriminant function model had an overall classification success of 76% (i.e., 48 of the 63 sites were classified correctly) and correctly classified 33 of 45 sites into assemblage 1 (i.e., classification success = 73%) 9 of 12 sites into assemblage 2 (classification success = 75.0%) and all six sites into assemblage 3 (i.e., classification success = 100%).

Stream reaches supporting fish assemblage 1 were typically located at higher elevations and had higher amounts of gravel within the substratum compared with those belonging to assemblages 2 and 3.

Stream reaches supporting fish assemblage 1 were typically located at higher elevations and had higher amounts of gravel within the substratum compared with those belonging to assemblages 2 and 3. In contrast, water depths and percent cobble at sites supporting assemblages 1 and 2 were greater than those for assemblage 3 (Table 19).

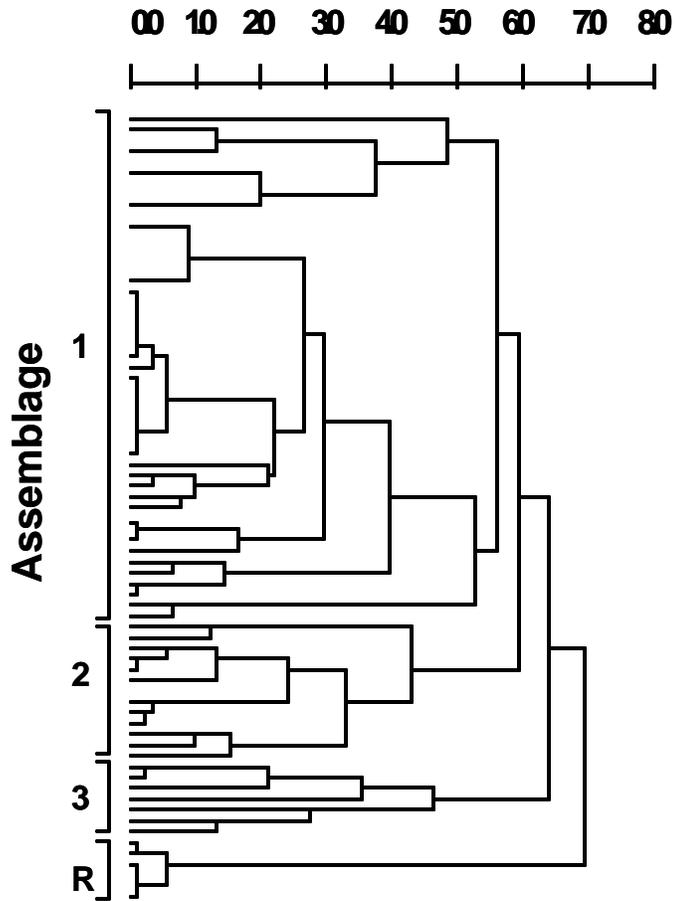


Figure 13. Agglomerative hierarchical cluster analyses of fish communities from the Notikewin Sub-basin of the Notikewin watershed. Analyses were completed using the Bray-Curtis Fish association measure and the unweighted pair-group method using arithmetic averages (UPGMA) on fish density data.

Table 20 Description of fish assemblages and related environmental variables from the three community types identified from the Notikewin Sub-basin.

	Assemblage 1 (N = 45 sites)			Assemblage 2 (N = 12 sites)			Assemblage 3 (N = 6 sites)		
	Mean±1STD	Min	Max	Mean±1STD	Min	Max	Mean±1STD	Min	Max
A) Density									
<u>Fish communities</u>									
Arctic grayling	0.69± 1.00	0	6.24	0.11±0.23	0	0.78	0.01±0.01	0	0
Brook stickleback	0.30±0.73	0	3.75	0.05±0.14	0	0.48	9.0±14.9	0	38.5
Emerald shiner	0±0	0	0	0±0	0	0	0±0	0	0
Dace	0.15±0.68	0	4.51	0.16±0.29	0	0.85	8.1±9.9	0.83	27.7
Chub	0.09±0.24	0	1.43	0.58±0.46	0	1.53	4.23±7.36	0	18.9
Longnose sucker	0.11±0.24	0	0.89	0.22±0.29	0	0.82	0.56±0.98	0	2.50
Northern pike	0.01±0.05	0	0.37	0.22±0.25	0	0.66	0±0	0	0
Sculpin	0.01±0.09	0	0.60	0±0	0	0	0±0	0	0
Trout perch	0.05±0.15	0	0.74	0.05±0.13	0	0.42	0±0	0	0
White sucker	0.03±0.13	0	0.69	0.38±0.32	0	0.97	0±0	0	0
Abundance rankings of the four numerically dominant species	Arctic grayling, brook stickleback dace, longnose sucker			Chub, white sucker, longnose sucker, northern pike			Brook stickleback, dace, chub, longnose sucker		
Total density	1.42±1.62	0.14	8.26	1.77±1.1	0.25	3.92	21.9±24.1	2.5	66.2
Richness	1.91±1.14	1.0	6.0	3.58±1.67	1.0	6.0	3.0±0.90	2.0	4.0

	Assemblage 1			Assemblage 2			Assemblage 3		
	Mean±1STD	Min	Max	Mean (1STD)	Min	Max	Mean (1STD)	Min	Max
B) Environmental variables									
Order	3.8±0.87	2	5	4.1±0.67	3	5	3.3±1.4	1	5
Elevation	818.8±96.3	598	1013	693±110	593	941	658.6±42.4	603	708
Forest	0.48±0.24	0.04	0.85	0.55±0.10	0.42	0.73	0.71±0.10	0.52	0.79
Bankfull width (m)	10.0±5.46	2.6	24.1	11.4±4.3	3.7	21.0	7.6±3.5	3.2	11.6
Reach slope	0.006±0.004	0	0.02	0.002±0.002	0	0.006	0.005±0.002	0.003	0.10
Adjacency	12.1±15.6	0	58.6	6.03±6.05	0	18.7	24.2±18.2	0	44.1
Depth	44.2±17.0	20.0	100	45.0±17.5	30.0	85.0	35.0±5.5	30.0	40.0
Gravel	0.41±0.20	0	0.86	0.28±0.17	0	0.52	0.21±0.15	0	0.45
Cobble	0.18±0.14	0	0.60	0.22±0.13	0	0.49	0.02±0.02	0	0.06

Testing the generality of the fish assemblage models: application to the Hotchkiss and Meikle Sub-basins.

The fish assemblage model derived in the Notikewin Sub-basin was a good predictor of fish assemblages in the adjacent Hotchkiss and Meikle Sub-basins.

7.2.3 Testing the utility of the discriminant function model: application to the Hotchkiss and Meikle Sub-basins

We tested the utility of the discriminant function model (based on density) derived for fish communities in the Notikewin watershed by applying the model to fish communities described at 39 sites in the adjacent Hotchkiss and Meikle Sub-basins. We predicted that the empirical model predicting fish community types in the Notikewin Sub-basin would be a good predictor of fish community types in the adjacent Hotchkiss and Meikle Sub-basin because these adjacent Sub-basins are share are also located in the upper Cordilleran ecoregion and have similar physiographical characteristics. Prior to these analyses, all 39 sites were classified into fish assemblage types using clustering methods described previously.

When applied to the Hotchkiss and Meikle Sub-basins, the discriminant function models correctly classified 74.4% (i.e., 48 of the 63 sites were correctly classified) of all sites. Comparisons of classification success for each of the fish assemblages showed that the discriminant function model correctly classified 4 of 5 sites into assemblage 1 (i.e., classification success = 80.0%), 17 of 23 sites into assemblage 2 (classification success = 73.9) and all 8 of the 11 sites into assemblage 3 (i.e., classification success = 72.7%). These data suggest that the discriminant function models developed from the Notikewin Sub-basin are relatively good predictors of fish community types in the adjacent Hotchkiss and Meikle Sub-basins.

7.2.4 Description of fish assemblages in the larger Notikewin watershed

Analysis of fish density data from all three Sub-basins of the Notikewin watershed (N = 108 stream reaches) also identified three fish assemblage types (Figure 14). As observed for the Notikewin Sub-basin, Arctic grayling, brook stickleback and dace were numerically dominant in assemblage 1 whereas Arctic grayling, chub and to a lesser extent white sucker, longnose sucker and northern pike were numerically dominant in assemblage 2 (Figure 14). For each assemblage, mean density of the most numerically abundant groups ranged between 0.25 to 0.6 / 100 m². In contrast, assemblage 3 was numerically dominated by brook stickleback, dace, chub and to a lesser extent longnose sucker. Densities of the three numerically dominant species groups in Assemblage 3 (Overall range = 2.11 to 3.2 / 100 m²) were between 3.5 to 12.8-fold higher in than that in assemblages 1 and 2 (Figure 14).

Total density and density of the numerically dominant seven species and species groups also differed significantly (ANOVA, P < 0.05) among assemblages (Figure 15). Total fish density and density of brook stickleback, longnose sucker, dace and chub in assemblage 1 was significantly higher than that in assemblages 1 and 2 (Bonferroni –adjusted orthogonal contrasts, P < 0.017). In contrast, densities of northern pike in assemblages 2 and 3 exceed that in Assemblage 1 (Figure 15).

Similarly, the relative abundance of fish species and species groups differed (ANOVA, $P < 0.05$) among assemblages (Figure 16). For instance, Arctic grayling were proportionately the dominant species in assemblages 1 and 2 (Bonferroni-adjusted, orthogonal contrast, $P < 0.017$) and accounted for significantly higher proportion of the community than that in assemblage 3. White sucker were proportionately more abundant in assemblage 2 than that in assemblages 1 and 3, whereas dace were proportionately more abundant in assemblage 3 than in assemblages 1 and 2. Lastly, chub were relatively more abundant in assemblages 2 and 3 than that in assemblage 1 (Bonferroni-adjusted orthogonal contrasts) (Figure 16).

Differences in fish assemblages were associated with significant differences in reach elevation but not water depth, and the relative composition of gravel and cobble within the substratum (Figure 18).

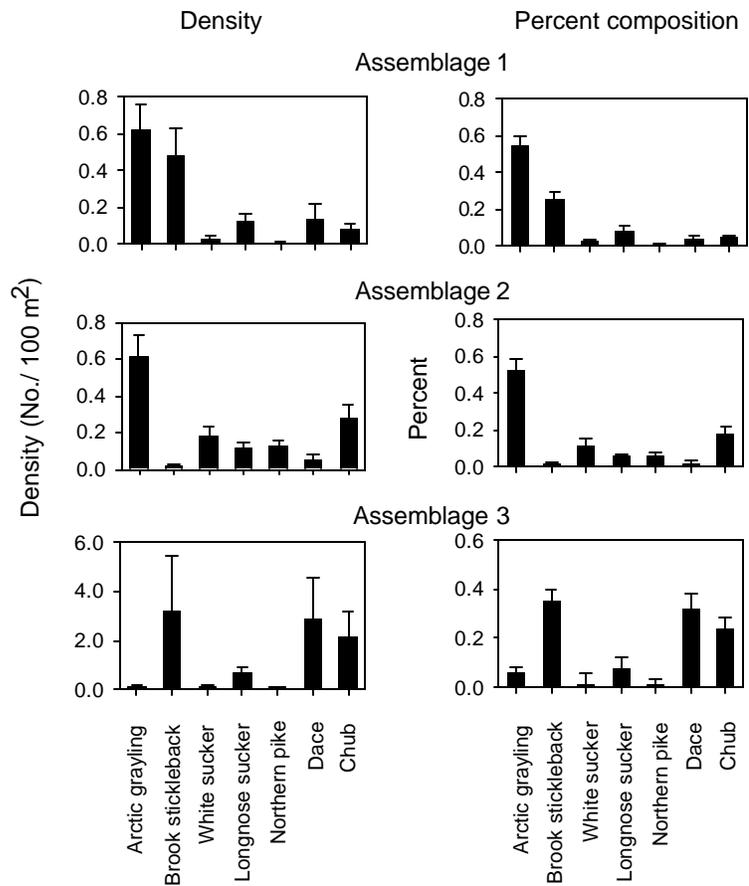


Figure 14. Mean ($\pm 1SE$) density of the seven numerically dominant fish species and species groups comprising the three fish assemblages in the Notikewin watershed. Analyses were based on communities from the Notikewin, Hotchkiss and Meikle Sub-basins.

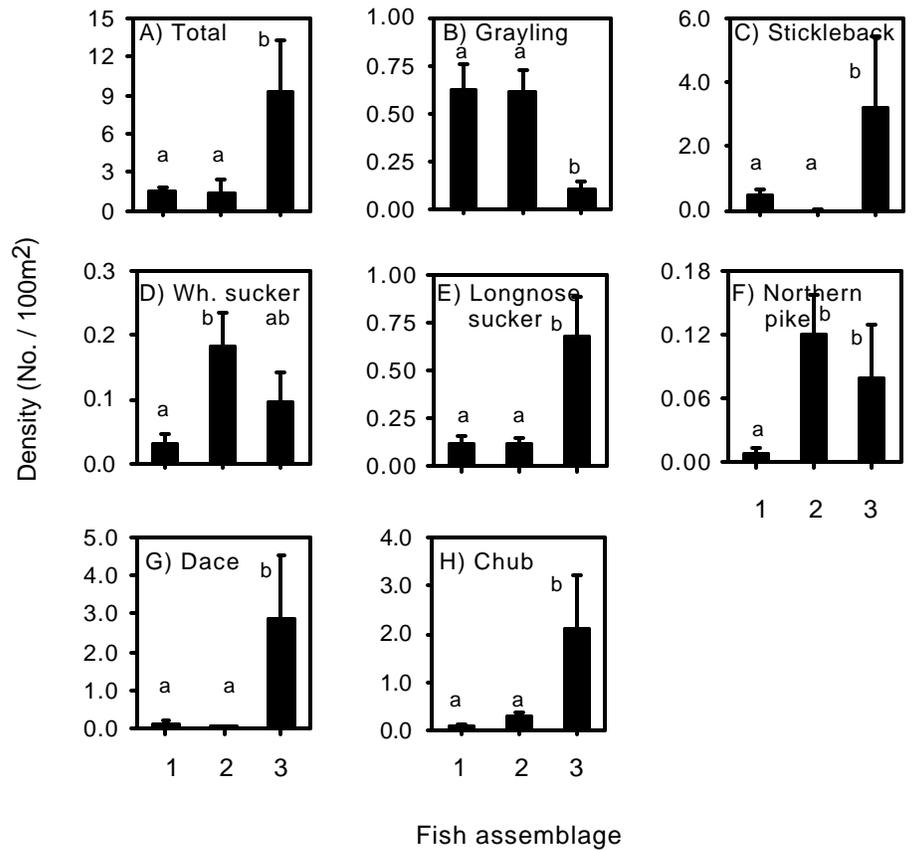


Figure 15. Comparisons of mean ($\pm 1SE$) total density and density of the seven numerically dominant fish species and species groups comprising the three fish assemblages in the Notikewin watershed. Analyses were based on communities from the Notikewin, Hotchkiss and Meikle Sub-basins. Histograms sharing the same letter are not statistically significantly different. Species and species group abbreviations: Grayling = Arctic grayling, stickleback = brook stickleback, Wh. sucker = white sucker.

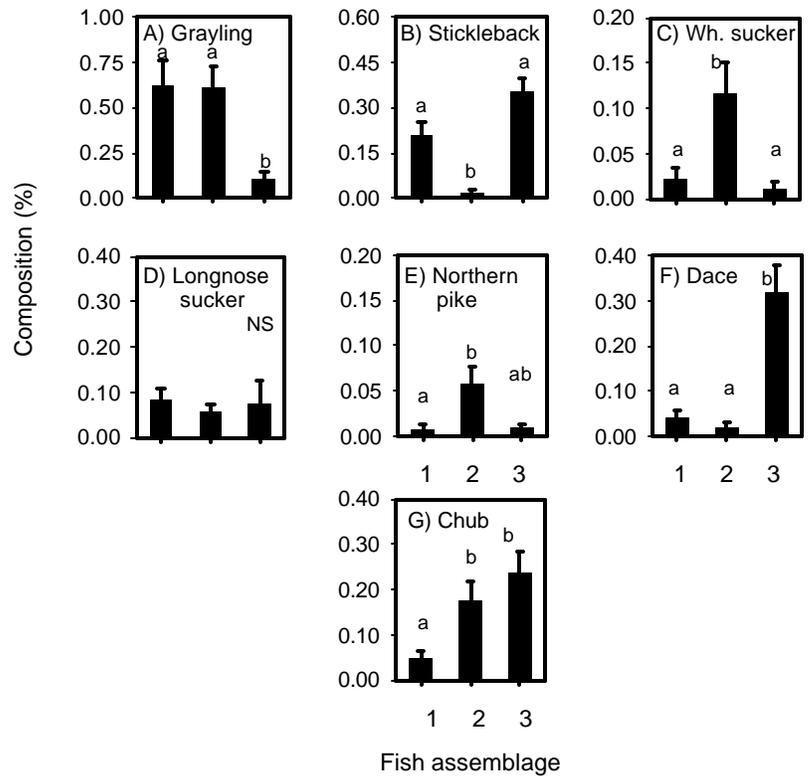


Figure 16. Comparisons of mean ($\pm 1SE$) percent composition of the seven numerically dominant fish species and species groups comprising the three fish assemblages in the Notikewin watershed. Analyses were based on communities from the Notikewin, Hotchkiss and Meikle Sub-basins. Histograms sharing the same letter are not statistically significantly different. Species and species group abbreviations : Grayling = Arctic grayling, stickleback = brook stickleback, Wh. Sucker = white sucker. NS = not statistically significant analysis of variance model.

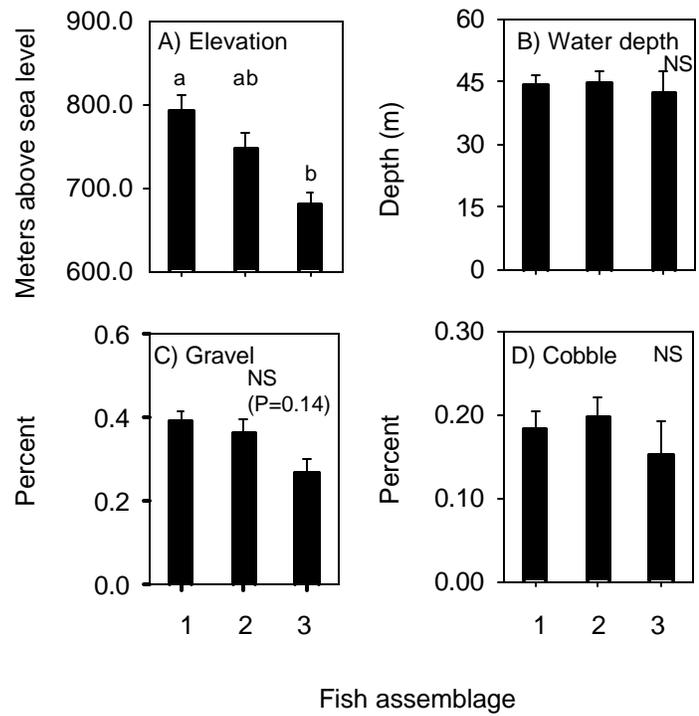


Figure 17. Comparisons of mean (± 1 SE) elevation, water depth and percent cobble and gravel at locations supporting the three fish assemblages in the Notikewin watershed. Analyses were based on communities from the Notikewin, Hotchkiss and Meikle Sub-basins. Histograms sharing the same letter are not statistically significantly different. NS = not statistically significant.

Focal Question 5: To what extent are fish assemblages shaped by forest cover attributes?

The importance of forest cover versus stream and watershed physiography characteristics as predictors of fish communities was evaluated using regression and ordination.

Forest cover attributes were typically poor predictors of the presence of fish, game fish, taxonomic groups and individual species.

8.0 FOCAL QUESTION 5 - TO WHAT EXTENT ARE FISH ASSEMBLAGES STRUCTURED BY FOREST COVER ATTRIBUTES?

8.1 Design and Statistical analyses

The importance of forest cover versus stream and watershed physiography characteristics as predictors of fish communities was evaluated using: i) logistic regression (i.e., fish presence-absence data) and ii) variance partitioning of canonical correspondence analyses (i.e., fish density data) of fish community data from 166 sites in the Notikewin watershed. Partitioning of variance was completed by performing sets of CCA as described by Borcard *et al.* (1992).

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. For logistic regression models, higher model removal criteria were often required to establish relationships between the presence of fish using forest cover attributes.

8.2 Results

8.2.1 Presence of fish

Logistic regression analyses showed that forest cover attributes were typically poor predictors of the presence of fish, game fish, taxonomic groups and individual species compared with regression models derived using both non-forest physiographic attributes and forest cover attributes (Table 21). Overall, only 50% (16 of the 32 models) of empirical models derived solely using forest attributes were statistically significant ($P < 0.10$) predictors of fish presence. The remaining models were either statistically non-significant or questionable because maximum likelihood estimates did not exist. In contrast, 97% (31 of 32 models) of empirical models using both physiographic and forest cover attributes were significant predictors ($P < 0.10$) of fish presence and in the majority of cases (23 of 32 models) models were statistically significant at an alpha of 0.05. In the majority of cases (>85%) empirical models based solely on physiographic attributes were statistically significant predictors of fish presence. Lastly, forest cover attributes were included within only 12.5% (4 of 32) of best-fit models (Table 21).

Comparisons between best fit models and those using forest cover attributes using Strahler stream orders indicated that on average forest cover models were not statistically significant (mean model $P = 0.12$) compared with models derived for stream defined using the Strahler stream order ($P = 0.039$), had lower classification success but only marginally lower explanatory power (i.e., R^2) (Table 21). Comparisons of model fit between sites defined using Alberta watercourse classes showed that

models derived using forest cover attributes were less powerful and had lower predictive power and classification success compared to those derived using non-forest physiographic attributes and forest cover attributes (Table 15).

Table 21. Comparison of best fit model logistic regression models versus model derived using forest cover attributes predicting the presence of fish (logit) in first to fifth order reaches (A) and small and large permanent reach sites (B) in the Notikewin watershed. Analyses were restricted to commonly encountered species or species groups. Game fish = Arctic grayling, walleye, northern pike; 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. Highly non-statistically significant models ($P > 0.25$) are not shown. Highly non-statistically P values are not shown (i.e., $P > 0.10$). Elevation (masl). Model abbreviations: temperature = water temperature, gravel = mean proportion gravel, conifer = proportion of conifer in watershed, deciduous = proportion of deciduous in watershed, gravel = proportion of gravel, conifer-100, 300 & 500 = percent cover by conifer species with 100, 300 and 500m of the study reach, deciduous-100, 300 & 500 = percent cover by deciduous species with 100, 300 and 500m of the study reach.

		N	P	R ²	Classification Success
A) Strahler stream order					
First order sites					
<i>Fish</i>					
Best model	Y = -5.57 (0.07) - 0.007 (0.06) elevation	23	0.056	0.75	0.65
Forest cover model	Y = -0.84 (0.19) + 1.62 (0.22) conifer-100	23	0.22 (NS)	0.60	0.52
<i>Sucker</i>					
Best model	Y = 6.93 (0.08) - 0.011 (0.037) elevation	23	0.040	0.60	0.83
Forest cover model	Y = -4.63 (0.05) + 5.82 (0.22) conifer-100	23	0.040	0.60	0.77
Second order sites					
<i>Fish</i>					
Best model	Y = -2.82 (0.05) + 0.20 (0.06) temperature	32	0.071	0.73	0.59
Forest cover model	Y = -1.86 (0.06) + 1.97 (0.22) deciduous-100	32	0.24 (NS)	0.73	0.56
<i>Brook stickleback</i>					
Best model	Y = -3.00 (0.001) + 0.21 (0.07) bankfull width	33	0.07	0.52	0.91
Forest cover model	Y = -2.73 (0.001) + 7.45 (0.04) deciduous	33	0.04	0.52	0.82

		N	P	R ²	Classification Success
Third order sites					
<i>Fish</i>					
Best model	Y = -0.71 (0.367) + 0.39 (0.032) bankfull width	49	0.033	0.69	0.76
Forest cover model	Y = 0.83 (0.051) + 0.86 (0.492) conifer-100	49	0.49 (NS)	0.69	0.73
<i>Game fish</i>					
Best model	Y = -3.63 (0.012) + 0.28 (0.045) bankfull width + 3.96 (0.034) conifer-300	49	0.032	0.73	0.63
Forest cover model	Y = -1.91 (0.051) + 3.39 (0.35) conifer	49	0.05	0.72	0.59
<i>Minnow</i>					
Best model	Y = -5.46 (0.039) + 0.15 (0.018) temperature	44	0.011	0.68	0.77
Forest cover model	Y = -1.88 (0.039) + 4.54 (0.001) deciduous-300	49	0.013	0.66	0.76
<i>Sucker</i>					
Best model	Y = 0.95 (0.22) - 361.6 (0.011) reach slope	44	0.042	0.68	0.75
Forest cover model	Y = 1.41 (0.001) + 4.59 (0.196) deciduous	49	0.40 (NS)	0.67	0.73
<i>Arctic grayling</i>					
Best model	Y = -1.94 (0.001) + 3.73 (0.034) gravel	49	0.019	0.72	0.67
Forest cover model	Y = -0.30 (0.36) - 4.45 (0.083) deciduous	49	0.083	0.72	0.61
Fourth order reaches					
<i>Game fish</i>					
Best model	Y = -1.96 (0.046) + 0.31 (0.001) bankfull width	52	0.013	0.74	0.68
Forest cover model	Y = 0.84 (0.047) + 3.97 deciduous -300	53	0.037	0.73	0.68
<i>Minnow</i>					
Best model	Y = -2.92 (0.001) + 0.27 (0.045) bankfull	52	0.003	0.73	0.75
Forest cover model	Y = -2.52 (0.001) + 4.55 (0.001) conifer-300	53	0.001	0.73	0.75
<i>Sucker</i>					
Best model	Y = 5.49 (0.059) - 0.009 (0.034) elevation	53	0.067	0.69	0.74
Forest cover model	Y = -1.46 (0.049) + 4.43 (0.021) deciduous-300	53	0.021	0.68	0.77

		N	P	R ²	Classification Success
<i>Arctic grayling</i>					
Best model	Y = 0.50 (0.70) + 0.26 (0.021) bankfull width – 0.18 (0.047) temperature	50	0.026	0.74	0.74
Forest cover model	NS				
<i>Lake chub</i>					
Best model	Y = -1.54 (0.42) – 147.5 (0.007) reach slope + 1.76 (0.089) conifer-300	53	0.019	0.68	0.75
Forest cover model	Y = -1.58 (0.001) + 1.34 (0.16) conifer-300	53	0.16 (NS)	0.69	0.74
Fifth order reaches					
<i>Game fish</i>					
Best model	Y = -2.02 (0.067) + 0.17 (0.020) bankfull width	33	0.022	0.74	0.73
Forest cover model	Y = -2.95 (0.01) – 4.43 (0.05) conifer – 3.03 (0.08) conifer-300	33	0.062	0.74	0.67
<i>Minnow</i>					
Best model	Y = -3.98 (0.031) + 0.35 (0.013) water temperature	33	0.060	0.73	0.67
Forest cover model	Y = -0.03 (0.95) + 2.18 (0.15) deciduous-300	33	0.15 (NS)	0.73	0.42
<i>Sucker</i>					
Best model	Y = 1.01 (0.23) – 657.4 (0.041) reach slope	33	0.042	0.51	0.82
Forest cover model	Y = 0.52 (0.32) – 9.39 (0.098) deciduous	33	0.10 (NS)	0.64	0.56
<i>Arctic grayling</i>					
Best model	Y = -0.30 (0.90) + 0.15 (0.12) bankfull width – 0.36 (0.162) temperature + 8.13 (0.02) gravel	33	0.041	0.65	0.82
Forest cover model	Y = 0.14 (0.81) – 4.84 (0.11) deciduous – 2.93 (0.12) deciduous-300	33	0.13 (NS)	0.75	0.52
<i>Lake chub</i>					
Best model	Y = 10.33 (0.063) – 1028.7 (0.024) reach slope – 0.48 (0.12) temperature	33	0.052	0.73	0.85
Forest cover model	Y = 0.17 (0.73) – 6.36 (0.058) deciduous	33	0.057	0.73	0.58

		N	P	R ²	Classification Success
<i>Longnose sucker</i>					
Best model	Y = 12.69 (0.02) – 598.9 (0.042) reach slope – 0.87 (0.19) temperature	33	0.057	0.69	0.91
Forest cover model	Y = -0.23 (0.64) – 7.59 (0.067) deciduous	33	0.073	0.69	0.73
B) Alberta watercourse classes					
Small permanent (channel widths = 0.5 to 5 m)					
<i>Fish</i>					
Best model	Y = -1.32 (0.026) + 0.56 (0.007) bankfull width – 1.68 (0.14) gravel	84	0.022	0.75	0.68
Forest cover model	Y = -0.26 (0.31) + 0.80 (0.477) deciduous	85	0.21 (NS)	0.75	0.51
<i>Game fish</i>					
Best model	Y = -4.41 (0.001) + 0.81 (0.007) bankfull width	84	0.051	0.57	0.87
Forest cover model	Y = -2.61 (0.003) + 3.616 (0.032) conifer – 2.43 (0.095) conifer-500	85	0.20	0.56	0.66
<i>Minnow</i>					
Best model	Y = -2.30 (0.011) + 0.41 (0.069) bankfull width – 3.61 (0.043) conifer	84	0.022	0.75	0.68
Forest cover model	Y = -1.09 (0.03) - 3.37 (0.054) conifer	85	0.054	0.52	0.68
<i>Sucker</i>					
Best model	Y = -1.32 (0.009) - 58.37 (0.282) reach slope	85	0.28	0.56	0.68
Forest cover model	Y = -1.99 (0.007) + 0.76 (0.42) conifer-100	85	0.42 (NS)	0.56	0.62
Large permanent (channel widths >0.5 m)					
<i>Fish</i>					
Best model	Y = -1.10 (0.90) + 0.198 (0.031) elevation	104	0.031	0.56	0.86
Forest cover model	Y = -2.19 (0.01) – 1.001 (0.265) conifer-500	105	0.265 (NS)	0.56	0.86

		N	P	R ²	Classification Success
<i>Game fish</i>					
Best model	Y = -0.63 (0.186) + 0.091 (0.029) bankfull width	104	0.028	0.76	0.53
Forest cover model	Y = -0.72 (0.02) - 0.954 (0.149) conifer-500	105	0.34 (NS)	0.74	0.56
<i>Minnow</i>					
Best model	Y = -2.03 (0.001) + 0.150 (0.005) bankfull width	104	0.001	0.74	0.68
Forest cover model	Y = -0.97 (0.022) + 1.75 (0.100) conifer	105	0.100	0.74	0.58
<i>Sucker</i>					
Best model	Y = 18.01 (0.001) - 0.024 (0.0001) elevation - 4.86 (0.001) forest	105	0.0001	0.71	0.77
Forest cover model	Y = 0.25 (0.64) - 1.863 (0.031) forest	105	0.031	0.71	0.61
<i>Arctic grayling</i>					
Best model	Y = -0.86 (0.001) + 0.205 (0.001) bankfull - 3.42 (0.413) elevation	105	0.003	0.75	0.59
Forest cover model	Y = 0.42 (0.88) - 0.68 (0.298) conifer	105	0.298 (NS)	0.75	0.54
<i>Brook stickleback</i>					
Best model	Y = 4.35 (0.062) - 0.008 (0.014) elevation	105	0.014	0.62	0.83
Forest cover model	Y = -1.23 (0.56) - 0.72 (0.349) conifer-100	105	0.35 (NS)	0.62	-
<i>Lake chub</i>					
Best model	Y = 9.88 (0.0009) - 0.013 (0.0006) elevation - 2.79 (0.015) forest	105	0.0017	0.71	0.68
Forest cover model	Y = 0.039 (0.94) - 1.485 (0.082) forest	105	0.082 (NS)	0.71	0.65
<i>Longnose sucker</i>					
Best model	Y = 0.25 (0.001) - 197.9 (0.001) reach slope	105	0.004	0.69	0.70
Forest cover model	Y = 0.00 (0.87) - 1.759 forest	105	0.047	0.69	0.69

Table 22. Mean (± 1 STD) statistical significance, Nagelkerke R^2 and classification success of best fit logistic regression models derived solely using forest cover attributes compared to best fit model using all potential predictor variables. N = number of comparisons, P = model P value. Comparisons are shown using Strahler (A) and Alberta watercourse stream classes (B). P = overall model P value, N= number of comparisons. Analyses were complete after excluding two highly non-significant forest cover models.

	N	P	Nagelkerke R^2	Classification success
(A) Strahler stream orders				
<i>Best model</i>	19	0.039 \pm 0.021	0.684 \pm 0.07	0.751 \pm 0.090
<i>Forest cover model</i>	19	0.120 \pm 0.135	0.685 \pm 0.06	0.674 \pm 0.111
Absolute difference	-	0.08	0.001	0.077
(B) Alberta watercourse stream classification				
<i>Best model</i>	11	0.041 \pm 0.081	0.680 \pm 0.08	0.714 \pm 0.108
<i>Forest cover model</i>	11	0.214 \pm 0.133	0.656 \pm 0.09	0.616 \pm 0.101
Absolute difference	-	0.17	0.024	0.098

8.2.2 Fish community structure

The extent to which forest cover explained variation in fish community structure was evaluated by partitioning the variance in fish communities into that which could be explained by: 1) forest cover attributes and 2) physiographic, non-forest cover attributes using variance partitioning techniques (Borcard *et al.* 1992, Paszkowski and Tonn 2000). For this analysis, forest cover attributes included those measured at the watershed scale and those measured within the three riparian zones areas (i.e., areas of 3.16 ha, 28.13 ha, 78.14 ha) delineated by establishing radius of 100 m, 300 m and 500 m radius around the sampling point.

Forest cover attributes were typically poor predictors of fish density.

Multiple canonical correspondence analyses showed that the environmental variables explained a relatively small proportion of the overall variance in fish density (Figure 18). When decomposed, the majority (45.6%) of explained variance in fish communities was attributed to non-forest cover attributes of elevation, mean water depth, reach slope, and the percentage of gravel and cobble within the substratum. Forest cover attributes including percent of watershed forested, and percent conifer, deciduous accounted for 29% of the variance that could be explained with the remaining 25.4% attributed to the interaction of forest cover and non-forest cover attributes (Figure 18).

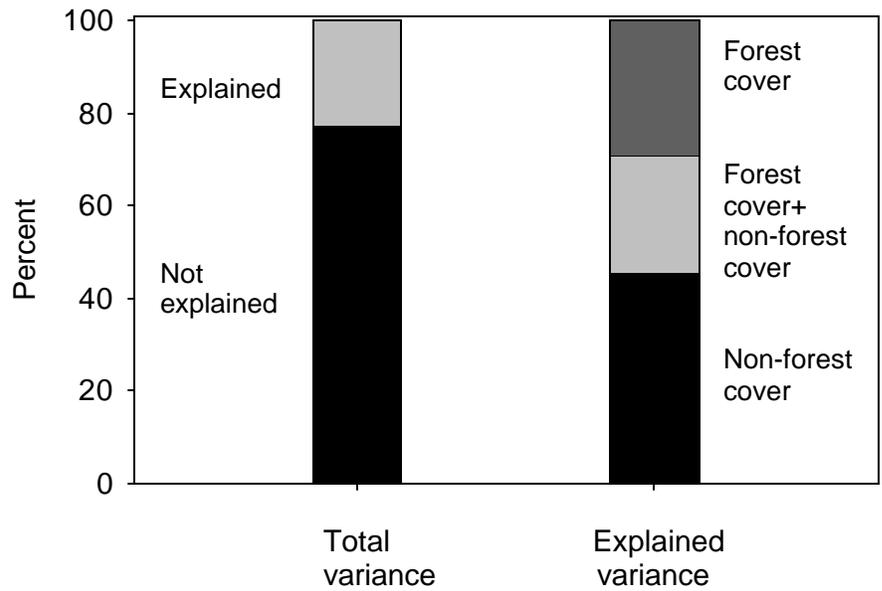


Figure 18. The percentage of explained and unexplained variance in the density of fish assemblages. From 108 stream reaches containing fish in the Notikewin watershed (A) and that which could be attributed to non-forest cover variables, non-forest cover combined with forest cover variables and -forest cover attributes (B). Variance partitioning was calculated from a series of canonical correspondence analyses (See text for details).

9.0 DISCUSSION

9.1 Temporal Variability in Fish Communities

Fish populations are known to highly temporally variable and this variability needs to be considered when describing: 1) relations between watershed attributes and biotic communities, 2) impacts of anthropogenic disturbances on biota and 3) recovery from disturbances.

Our data from the Notikewin watershed showed considerable temporal variation in both the presence of fish and significant differences in fish density between and among years.

Differences in concordance among species presence could reflect differences in life history patterns and general mobility that affect resistance and resilience to natural disturbances.

Fish populations are known to highly temporally variable (e.g., Moyle and Vondracek 1985, Platts and Nelson 1988, Grossman *et al.* 1990, Paul *et al.* 2000). Such variability needs to be considered when: 1) describing relations between watershed attributes and biotic communities, 2) evaluating impacts of anthropogenic disturbances on biota and 3) documenting recovery from disturbances (e.g., Bramblett and Fausch 1991, Scarnecchia and Bergerson 1987, Harding *et al.* 1998, Dobson *et al.* 2001). The explanatory power of empirical models predicting fish abundance (i.e., density and biomass) from watershed variables has also been shown to be variable (e.g., Fausch *et al.* 1988 and reference therein). For example, Scarnecchia and Bergerson (1987) reported that a habitat quality index model was moderately powerful ($R^2 = 0.44$) in explaining trout standing crop from 11 streams in northern Colorado using data collected in 1979 but was considerably less powerful ($R^2 = 0.16$) when applied to data collected in 1980.

Our data from the Notikewin River Basin showed considerable temporal variation in both the presence of fish and fish density between and among years. Comparisons of fish communities described at two time periods, showed that the concordance in the presence or absence of fish at each time period was strongly affected by stream size and differed among species and species groups. For instance, only one third of all first and second order stream reaches contained fish on both sampling years. While concordance in the presence of fish increased with stream order, only 60 and 63% of all third, fourth and fifth order reaches also contained fish on both sampling years. Conversely, less than one third of stream sites that contained fish in one year were also devoid of fish in the second year.

The presence of individual species was also variable and differed among species and species groups. For instance, repeated sampling showed that only 40% of third, fourth and fifth order reaches contained Arctic grayling on both sampling times but only 18 and 9% of sites contained lake chub, suckers and cyprinids on both sampling events. In contrast to species presence data, species absence data was relatively more consistent between years, at least for cyprinids and catostomids. In the majority of cases (67 to 83%) the absence of suckers and minnows from small streams reaches (i.e., 1st to 3rd order and small permanent stream reaches) in one year coincided with absence in the second sampling period.

Why is the presence of fish at the stream reach scale so variable? Ultimately, such variance results from the interactive effects of high annual variability in survival, adult reproductive success and recruitment with habitat quality and accessibility. Differences in concordance among species presence could reflect differences in life history patterns and general mobility that affect resistance and resilience to natural disturbances (e.g., floods, low water, river ice break-up, icing). For instance, Arctic grayling complete moderately long spawning migrations

(Northcote 1995) under high spring flows that may allow them to colonize previously inaccessible stream reaches, such as those which may have been fragmented by barriers (e.g., beaver dams) or those where habitat quality had increased sufficiently to allow occupation.

Irrespective of the mechanism, linear regression showed that relative species turnover was negatively related with stream size, measured as stream bankfull width, suggesting that the fish communities in small stream reaches are more variable than those in larger reaches.

Irrespective of the mechanism, linear regression showed that relative species turnover was negatively related with stream size measured as bankfull width suggesting that the smaller stream reaches are more variable than larger reaches. In addition, the relative number of species turnovers was positively related to the inter-sample period suggesting that changes in survival, recruitment or habitat quality may have a temporal component at least over relatively short time periods (10^1 years).

The lack of consistency in both the presence and to a lesser extent the absence of fish, game fish and individual species from stream reaches over relatively short time periods poses an interesting dilemma for fisheries management. Canada's Fisheries Act and related policies for fish habitat management enforced by the Department of Fisheries and Oceans (Fisheries and Oceans 1986, 1991) and through delegation to the Provincial Governments are based on the principle of no net loss. Alberta's Fish Conservation Strategy for 2002-2005 is also advocates no net loss as a working guideline for fish protection in the province (Alberta Environment 1998). Within this framework, the Department of Fisheries and Oceans strives to balance unavoidable habitat losses with habitat replacement. Inherent within this principle is that fish bearing streams, and reaches, can be identified so that reaches that provide fish habitat can be conserved. Alternatively, if site-specific habitat loss is unavoidable, such losses are either mitigated through habitat enhancements or through compensation so that site-specific habitat losses are balanced with habitat gains elsewhere. The high variability in fish populations and in predicting fish habitat creates uncertainty over what biologically constitutes fish habitat. Stream reaches that are devoid of fish in certain years can support fish populations at other times.

Despite a level of uncertainty, the use of empirical models to predict the presence of fish is a more rigorous approach than current practices.

Lastly, a paucity of data precluded our ability to predict site attributes that may drive high temporal variability in fish occurrence. However, we expect that larger data sets will allow the development of empirical models that predict: 1) stream reaches that display high non-concordance in the presence of fish, and 2) the location of these reaches within a larger landscape context. In the absence of these models, fisheries managers will need to acknowledge that at the stream reach scale, the presence of fish is variable even for moderately large third to fifth order stream reaches. Despite a level of uncertainty, the use of empirical models to predict the presence of fish is a more rigorous approach than current practices.

9.2 Predicting the Presence of Fish

Analysis of frequency of occurrence data showed that the presence of fish differed markedly among stream order and between stream classes.

Analysis of frequency of occurrence data showed that the presence of fish differed markedly among stream order and between stream classes. In contrast to the majority of first (57%) and second (64%) order and small permanent stream reaches (55%) which did not contain fish, 73 to 91% of

3rd to 5th order reaches and large permanent stream reaches were occupied by fish. The occurrence of all species increased with stream order and stream class, albeit only moderately for some species (e.g., brook stickleback, longnose sucker).

We used logistic regression models to predict the presence of fish, game fish, and individual species and species groups in the Notikewin River Basin.

Our analyses identified a small suite of recurring independent variables of elevation, water temperature, bankfull width, reach slope, and to lesser extent, percent gravel within the substratum as predictors of fish occurrence.

At the landscape level, the presence of fish, game fish, Cyprinids and Arctic grayling were significantly and positively related to stream size measured as stream bankfull width.

As an indicator of stream size, the increased propensity of larger stream reaches to support fish could arise from several interrelated factors including water permanence and species-specific habitat requirements including water depth and velocity, and dissolved oxygen requirements

We used logistic regression models to predict the presence of fish, game fish, and individual species and species groups in the Notikewin River Basin. Our primary focus was to develop simple predictive models using a few highly non-related (Pearson correlation coefficients $r < 0.5$) environmental predictors (Lanka *et al.* 1987). Using these criteria, logistic regressions identified moderate to strong predictive equations based on only a small suite of recurring independent variables of elevation, water temperature, bankfull width, reach slope, and to lesser extent, percent gravel within the substratum as predictors of fish occurrence. At the landscape level and for streams of different sizes (i.e., Strahler and Alberta watercourse classes), these predictors typically resulted in statistically significant models that explained between 65 to 80% of variance in the presence of fish with overall model classification success of between 70 and 85%. The performance of these models could have been increased by adding more predictors including those that were moderately related (i.e., Pearson correlation coefficients = 0.5 to 0.7) to the dominant explanatory variables (e.g., elevation and bankfull width). However, our results from the Notikewin Basin showed that in the majority of cases, one or two watershed and instream habitat variables explained the majority of variance in fish presence, and the addition of more variables, including those which were moderately related to the key explanatory variables did not greatly improve explanatory power (relative improvement in classification success = 0.05 to 0.20). Further, the development of logistic models that include variables moderately to highly correlated with key predictive variables raises serious concerns about the over inflation of regression model fit. While appearing to be more powerful, it is highly questionable whether such models are beneficial within a scientific or management context.

At the landscape level, the presence of fish, game fish, Cyprinids and Arctic grayling were significantly and positively related to stream size measured as stream bankfull width. As an indicator of stream size, the increased propensity of larger stream reaches to support fish could arise from several interrelated factors including water permanence and species-specific habitat requirements including water depth and velocity, and dissolved oxygen requirements. While relatively little is known about stream habitat conditions in small stream in northern Alberta, smaller streams are more vulnerable to complete or partial freezing depending on the intensity and length of the winter period and presence of snow cover that affects ice growth. Thus, while larger stream reaches can provide summer and winter habitats depending on species requirements, smaller stream that freeze during the winter experience bouts of recruitment through summer feeding (e.g., Buzby and Deegan 2000) and spawning migrations. In the absence of groundwater inputs and deeper permanent pool habitats, spawning in small stream reaches can result in minimal recruitment due to seasonal declines in discharge and accumulations of

frazil and anchor ice that profoundly affect the quantity and quality of fish habitat by altering habitat volumes (see Cunjak 1995 and references therein), and presumably exchanges of dissolved oxygen rich groundwater and species interactions (Chisolm *et al.* 1987).

Our results indicate that the occurrence of catostomids, lake chub and brook stickleback were negatively related with reach elevation indicating that fish were more likely to be present at lower rather than higher elevations in the watershed.

Alternatively, it is also possible that the reduced occurrence of fish in smaller stream reaches results because they are located at greater distances (i.e., reduced adjacency) from overwintering habitats and that fidelity in summer feeding habitats can reduce recolonization during the summer open-water period (e.g., Buzby and Deegan 2000). Lastly, larger stream reaches may be more likely to support fish because they provide a greater diversity of habitats types or are associated with thermal regimes that are sufficient for fish development and growth.

Similarly, our results indicate that the occurrence of catostomids, lake chub and brook sticklebacks were negatively related with reach elevation indicating that fish were more likely to be present at lower rather than higher elevations in the watershed. That the addition of stream bankfull width and reach slope did not appreciably improve model fit suggests that the effects of elevation may be largely related with thermal regimes or other habitat attributes unrelated to stream size.

While our empirical models predicting the presence of fish in first and second order stream reaches performed moderately well, the development of such models for low order stream reaches can be problematic.

While our empirical models predicting the presence of fish in first and second order stream reaches performed moderately well, the development of such models for low order stream reaches can be problematic due to: 1) the misclassification of first and second order stream reaches and 2) incorrect interpretation of classification success of logistic regression models. The classification of stream order is strongly influenced by the method and spatial resolution used to identify the stream channel. For the Notikewin watershed, the stream hydrography layer was created from 1:20,000-scale aerial photography. Errors in stream ordering between first and second order streams can arise from inconsistencies in identifying when the stream channel is sufficiently well formed to warrant classification as a first order reach. In fact, discriminating between ephemeral draws and first order stream is inherently difficult. In some situations, second order reaches can arise erroneously from the misclassification of ephemeral draws as first order streams. The presence of even a moderately well defined channel also does not imply that the channel provides continuous lotic habitat and even semi-permanently flowing streams can maintain defined channels depending on adjacent vegetation and soil conditions. Our experiences in many watersheds in Alberta's Boreal forest have shown that perhaps only 30 to 40% of all first order stream sites identified from aerial photography actually contain water during the late spring-early summer period when flows would be expected to be present. We suggest that practitioners should be cautious when developing predictive models for first stream order because of errors in stream classification. Lastly, the ecological importance of first and second order streams to the stability and persistence of stream fish communities at the landscape level is poorly understood.

The interpretation of logistic regression models can be misleading when such models are highly successful in correctly classifying one state (e.g., predicting fish presence) but poor in predicting the other (e.g., predicting fish absence).

Selection of environmental variables used to develop logistic empirical models is profoundly affected by the how the models are to be used.

Consideration of watershed boundaries, including those of first and second-order streams, will likely play an increasingly important role in the planning of harvest blocks and other industrial activities in the future.

Second, the overall classification success of logistic regression is often used as a measure of predictive capacity and is calculated from concordance between observed outcomes (e.g., observed presence and absence of fish) and those expected based on the predictive variables (e.g., instream and watershed variables). However, the interpretation of logistic regression models can be misleading when such models are highly successful in correctly classifying one state (e.g., predicting fish presence) but poor in predicting the other (e.g., predicting fish absence). Because we were interested in predicting both fish presence and absence, we avoided presenting logistic regression models which were powerful at correctly predicting fish absence but of limited utility in correctly predicting fish presence. The importance of misinterpreting overall classification success depends on the study question, but practitioners need to be aware of case-specific and overall model classification success.

The selection of environmental variables used to develop empirical models is profoundly affected by the how the models are to be used. Our empirical model predicting the presence of fish could have included several measures of stream size including stream bankfull width (i.e., the flow that maintains the stream channel), water depth, channel wetted width (i.e., stream width determined by the presence of water) and watershed area (i.e., total area drained by a tributary or tributaries to a given location) could have been used to account for spatial and temporal variance in fish presence, abundance and assemblage type. These measures of stream size are often highly correlated and depending on the study approach, there are potential problems in including multiple measures in the same model. In the present study, we chose stream bankfull width as a measure of stream size because it is easily measured (e.g., tape measure, digital range finder), robust to seasonal and short term changes in flow and could be easily integrated into cut block layout assessment or estimated from aerial photography.

In contrast, while estimates of wetted width were used in density estimates and can be measured easily, it was not used to describe stream size because it is highly temporally (e.g., daily to weekly spate-driven variance and high seasonal and annual variance) and spatially variables (e.g., among constrained and unconstrained reaches). Lastly, we also avoided the use of watershed area as a measure of stream size because it is quantified through either digital or manual exercises and is more time consuming than measures of stream size that can be estimated in the field as part of other activities. Further, harvest block layout in Alberta is strongly based on stand level management and watershed delineations are seldom considered at least for small watersheds. Thus, quantifying watershed area using GIS exercises is not a pervasive component of current practices and GIS queries to derive this measure of stream size is relatively expensive. Nevertheless, we expect that consideration of watershed boundaries, including those of first and second-order streams, will play an increasingly important role in the planning of harvest blocks and other industrial activities and in the future may prove to be a better metric of stream size than stream bankfull width.

We suspect that logistic regression has inherent potential as a fisheries management tool to detect moderate changes in population size.

The utility of logistic regression as a fisheries management tool is related to: 1) the ability to identify sites where fish are absent but where predictive models suggest that they should occur (i.e., site-based assessments) and 2) detect changes in population density (i.e., population trend detection (e.g., Strayer 1999)). The use of logistic regression to detect changes in abundance of fish populations is poorly understood and many questions related to statistical power remain unanswered (Mangel and Smith 1990). However, low statistical power arises when: i) low numbers of study sites are sampled, ii) encounter rates with species are low (e.g., sparse populations or low sampling effort at a site), iii) populations are highly spatially variable, and iv) if declines in populations are widespread (e.g., Strayer 1999). We suspect that logistic regression has inherent potential as a fisheries management tool to detect at least moderate changes in population declines because: i) the majority of watershed-scale studies are designed to include large numbers of sites, ii) effort (e.g., reach length sampled, number of passes through the reach) can be easily modified to increase the probability of detection (i.e., encounters) and can often be based on existing information on species abundance and spatial variance, iii) spatial variance can be well predicted based on watershed or landscape models and iv) large-scale population declines are relatively rare because of watershed connectivity or are predictable based on anthropogenic activities. Irrespective of how logistic regression is to be used, it will require that fisheries biologists become more acquainted with this techniques.

9.3 Relations Between Fish Communities and Watershed and Landscape Characteristics

The development of empirical models predicting the presence and abundance of fish communities is a central focus of fish ecology and fisheries management.

The development of empirical models predicting the presence and abundance of fish communities is a central focus of fish ecology and fisheries management (e.g., Fausch *et al.* 1988). In many cases, empirical models can yield relatively powerful relationships that explain 60 to 85% of the variation in fish abundance (i.e., density or biomass) (e.g., Fausch *et al.* 1988). Our models predicting total fish density and density of individual fish from instream habitat and watershed-scale attributes performed relatively poorly compared with model from other ecoregions.

Empirical models that predict the presence and abundance of fish from habitat variables assume that: 1) fish populations are limited by physical habitat conditions and other factors of predation, interspecific competition, fishing mortality play a minor role in influencing distributional patterns or are themselves related to habitat conditions (Clarkson and Wilson 1995), 2) disturbances such as extreme hydrological events (e.g., high and low flows, river ice break-up) that affect recruitment (e.g., egg survival, survival of juveniles [Clark 1992]) and adult survival are strongly linked with changes in habitat conditions (i.e., environmental descriptors of habitat) (Clarkson and Wilson 1995, Matthews 1986, Grossman *et al.* 1990) and 3) fish populations closely track changes in habitat condition. Despite these assumptions, an abundance of studies have identified relatively strong empirical models

predicting the abundance of salmonids from instream habitat attributes (See Fausch *et al.* 1988, Marcus *et al.* 1990, Sharma and Hilborn 2001).

Multiple regression indicated that watershed and instream habitat features were only moderate to poor predictors of fish density in the Notikewin River Basin.

Multiple regressions indicated that watershed and instream habitat features were only moderate to poor predictors of fish density in the Notikewin River Basin. Low to moderate predictive relations between fish standing crop or density and environmental variables has also been well documented (Fausch *et al.* 1988) in part because of high spatial and temporal variance in habitat quality, availability of spawning habitats (e.g., Baxter and Hauer 2000) and that catastrophic, often hydrological events, can potentially affect recruitment and year class strength without dramatically altering habitat quality. Variable fish growth and time to maturity (e.g., Carl *et al.* 1992), the availability of small-scale cold water and warm water refugia (e.g., Bilby 1984, Cunjak and Power 1986, Cunjak 1995) also affects variance in relations between habitat attributes and fish abundance. The occurrence of beaver (*Castor canadensis*) is also thought to affect the distribution of stream fish by altering dispersal corridors (Hägglund and Sjöberg 1999, Schlosser and Kallemeyn 2000) and under certain conditions can contribute to low dissolved oxygen concentrations (e.g., Devito and Dillon 1993). We also suspect that low predictive power of empirical relations between habitat and fish density arises from mortality induced by river ice break-up (Scrimgeour *et al.* 1994) and accumulations of frazil and anchor ice that may not manifest themselves in visible changes in stream habitat condition.

Lastly, the dearth of information on stream habitats and overwintering requirements of stream fish within the Boreal Cordillera and Plains is both noteworthy and surprising given the accumulating body of evidence that suggests that winter habitats can profoundly affect fish survival and that these habitats may be vulnerable to changes in land use practices (See Cunjak 1995 for a review, Quinn and Peterson 1996, Cunjak and Powers 1996). The importance of winter habitat to fish populations is also critical when evaluating potential effects of industrial activities on stream habitats in ecoregions where the ice-cover period can be extensive (5 to 6 months). We suspect that the availability of suitable overwintering habitats is a major variable affecting the predictability of fish abundance from instream and watershed variables (Cunjak and Power 1986, Chisholm *et al.* 1987, Cunjak 1995, Quinn and Peterson 1996).

While not overly powerful in explaining variance in fish density, our multiple regression and canonical correspondence models identified substratum composition (i.e., percent cobble and gravel), stream size (i.e., stream bankfull width), elevation, water depth, reach slope and to a lesser extent forest cover and reach adjacency as statistically significant predictors of fish density. In fact, these environmental variables have been reported as major drivers of fish community structure elsewhere (e.g., Rahel and Hubert 1991, Maret *et al.* 1997) and recognized as important attributes affecting habitat selection. Because many of these factors are not independent (e.g., elevation with water temperature and stream permanence, or substratum size composition with reach slope and

current velocity) attributing causal mechanisms is inherently speculative and cause-effect relations are best identified using experimentation.

The distribution and abundance of fish is known to be profoundly affected by stream size irrespective of whether it is measured as discharge or correlates of watershed area, wetted width or bankfull width.

Nevertheless, the distribution and abundance of fish is known to be profoundly affected by stream size irrespective of whether it is measured directly as discharge or correlates of watershed area, wetted width or bankfull width. Substratum size composition is also considered a major driver of stream fish community structure (Waters 1995 and references therein). For example, Maret *et al.* (1997) identified substratum size as a significant variable explaining occurrence of fish from 37 stream sites in western USA. The size composition of the substratum is also related to water velocity and combined can affect the abundance and type of invertebrate prey for fish and thus energetics related to feeding site selection (Rosenfeld and Boss 2001). Size composition of the substratum is also strongly affects habitat selection of individual species or life history stages.

Elevation has also been reported as a predictor of fish density (e.g., Rahel and Hubert 1991, Maret *et al.* 1997, Rahel and Nibblelink 1999). For example, Rahel and Hubert (1991) identified elevation to be dominant driver of fish assemblages in Wyoming, USA whereas Lanka *et al.* (1987) reported a significant relation between trout standing crop and reach elevation in forested streams in Wyoming. Maret *et al.* (1997) also showed differences in the structure of fish communities in lowland streams (i.e., < 1,600 m elevation) compared with those above 2000 m in the Upper Snake River Basins, western USA.

Elevation can affect fish populations either directly by affecting thermal regimes or indirectly because elevation is strongly related to a suite of fluvial processes that form and maintain stream channel morphology and thus fish habitat.

Elevation can affect fish populations either directly by affecting thermal regimes or indirectly because elevation is strongly related to a suite of fluvial processes that form and maintain stream channel geomorphology and thus fish habitat. Identifying how fish communities are related to elevation, like other habitat variables of stream size (e.g., bank full width, water temperature) is inherently difficult. For example, our results from the Notikewin River revealed significant positive relationships between elevation measured at individual stream sampling sites, stream bank full width and water temperature. Multiple regression analyses showed that the density of Arctic grayling was positively related to elevation while density of lake chub and longnose sucker were negatively related to site elevation. Elevation was also identified as the strongest variable separating the three fish assemblage types.

The extent to which stream reach slope affects fish populations is debated (Isaak and Hubert 2000). For example, Chisholm and Hubert (1986) reported a non-linear and inverse relationship between standing stock of brook trout (*Salvelinus fontinalis*) and reach gradient from 24 study reaches in Wyoming. They found that reach slope accounted for 25.5% of variance in standing stock. Sharma and Hilborn (2001) reported a negative relationship between the density of coho salmon smolts and stream gradient. Positive relations between trout density and reach-scale have also been reported (e.g., Hermansen and Krog 1984). In contrast, Isaak and Hubert (2001) suggest that the negative relationships reported

between trout abundance and reach slope arise because of confounding habitat and implicitly challenged the validity of previous studies. Our analyses indicated that overall fish density and density of individual species or species groups were unrelated to reach slope, although logistic regression showed that reach slope was a statistically significant and the most powerful predictor of the presence of suckers, perhaps reflecting feeding habits.

Woody debris profoundly influences the distribution and abundance of stream fishes by changing hydraulic complexity, creating meso-scale habitat and providing physical structure (e.g., Nickelson *et al.* 1992a, b, Culp *et al.* 1996, Beechie and Sibley 1997, Thevenet and Statzner 1999, Young *et al.* 1999). Woody debris can increase fish standing crop or density presumably by reducing encounters among conspecifics and predators, and providing spatial refugia during hydrologic events or a food source and habitat their invertebrate prey (e.g., Hauer and Benke 1991, Culp *et al.* 1996, Harvey *et al.* 1999). For example, during the winter coho salmon (*Oncorhynchus kisutch*) in Oregon streams are most abundant in habitats associated with woody debris (Nickelson *et al.* 1992a). Using an experimental approach, Giannico (2000) found that abundance of food and woody debris affected the distribution of juvenile coho salmon in pool habitats in two British Columbia streams. Under low food conditions, fish preferred pools that contained woody debris than pools where woody debris was absent.

Woody debris has also been shown to be important in creating deep pool habitats (Fausch and Northcote 1992, Hauer *et al.* 1999). For instance, Hauer *et al.* (1999) reported that the majority of pools in twenty stream reaches in northwest Montana were formed by bed scouring from either very large woody debris pieces located perpendicular to the stream or aggregates (i.e., multi-piece) of large woody debris.

Surprisingly, our analyses failed to identify the abundance of woody debris as a significant predictor of fish presence, abundance or discriminator among fish community types.

Surprisingly, our analyses failed to identify the abundance of woody debris as a significant predictor of fish presence, abundance or discriminator among fish community types. While speculative, it is possible that the linkage between woody debris and enhanced fish density is highly context dependent and affected by water clarity and overall fish density. We expect that the ability of woody debris to provide physical cover and shelter for fish may vary with water clarity. In fact, the vast majority of studies that have reported increased overall or life stage-specific abundance have been completed in clear to moderately clear water systems. Many of the stream reaches in the Notikewin watershed are moderately to highly coloured most likely resulting from inputs of dissolved organic carbon from wetlands (Prepas *et al.* 2001). Under these conditions, low water clarity may function to reduce encounters with conspecifics and predators typically attributed to increased habitat complexity in clear-water systems. The ability of woody debris to enhance fish density may also be low in systems such as the Notikewin where fish densities are low.

Alternatively, it is possible that woody debris was not identified as a major driving habitat variable because of low statistical power resulting from the absence of an appreciable gradient in woody debris abundance or 2) low variance in woody debris abundance. While we can not categorically deny the importance of these factors, we suspect that they played a minor role because our data suggest an appreciable range in woody debris even within and among stream orders (e.g., range among first and fifth order stream reaches = 0.13 to 1.38 cm³/m²).

For large-scale planning and assessment exercises, the use of watershed and landscape variables as predictors is advantageous because they can provide valuable information without requiring costly field assessments or where time constraints preclude field assessments.

In general fish community diversity at the reach level was low and individual reaches of 100 to 400 m in length typically contained two to four species. While few studies have quantified fish community diversity in Boreal Cordillera and Plains streams in Alberta and Saskatchewan Merkowsky (1998) also reported between one and four fish species in reaches of streams from the Boreal Plains ecoregion in Saskatchewan. However, at the watershed level, diversity was moderately high especially for Alberta with at least 17 species recorded from the Notikewin River Basin. While few studies have quantified fish community diversity in Boreal Cordillera and Plains streams in Alberta species richness in the Notikewin watershed is intermediate compared with that in relatively adjacent Kakwa (10 species) and the Simonette (20 species) basins of moderately similar size.

Empirical modeling of fish habitat requirements have historically focused on instream habitat variables of water depth, velocity and substratum size composition and a substantial body of evidence indicates that these variables are strong predictors of fish habitat preference (e.g., Marcus *et al.* 1990, Fausch and Northcote 1992). While our results also support these findings, our data suggest that watershed and landscape-level predictors performed equally well in explaining variance in fish density, albeit neither set of predictors performed overly well. For large-scale planning and assessment exercises, the use of watershed and landscape variables as predictors is advantageous because they can provide valuable information without requiring costly field assessments. We are not advocating the sole use of remotely acquired watershed and landscape level variables to predict fish community structure, however, the use of predictive models, in some situations, combined with field assessments could yield cost efficiencies. These cost efficiencies could be directed to fund conservation priorities which are currently not being addressed.

Land base managers are facing increasing pressures to provide quantitative evaluations of the potential effects of industrial activities and to better understand, and potentially mitigate, the effects of current activities on biological communities.

9.4 Testing Relations Between Fish Communities and Environmental Variables Using the Reference Condition Approach

Land base managers are facing increasing pressures to provide quantitative evaluations of the potential effects of industrial activities and to better understand, and potentially mitigate, the effects of current activities on biological communities. The reference condition approach represents one of several biological monitoring and assessment

The reference condition approach represents one of several assessment methods.

We used the reference-condition approach to test the spatial utility of fish habitat models developed in the Notikewin Sub-basin to that in the adjacent Hotchkiss and Meikle systems.

Our data showed that habitat models developed in the upper Notikewin Sub-basin performed well when applied to the adjacent Hotchkiss and Meikle Sub-basins.

The use of ecoregion-based models of fish communities has potential value for fish assessments and management purposes.

The use of regionally-based fish community models can also be used to evaluate the success of mitigation and compensation actions.

approaches (e.g., Wright *et al.* 1984, Karr *et al.* 1987, Rosenberg and Resh 1993, Karr and Chu 1999). In its broadest sense, it uses relations between biotic communities and environmental conditions from relatively pristine (i.e., relatively non-impacted sites) sites to predict the expected community types at potentially stressed locations (e.g., Reynoldson *et al.* 1997). It has been applied extensively to evaluate the effects of watershed disturbances on invertebrates (e.g., Reece *et al.* 2001, Reynoldson 2001) and more recently on fish communities (Joy and Death 2000, Tonn *et al.* In Press).

In the present study, we used the reference-condition approach to test the spatial utility of fish habitat models developed in the Notikewin Sub-basin to that in the adjacent Hotchkiss and Meikle systems. Our data showed that models developed from the upper Notikewin Sub-basin performed reasonably well when applied to the adjacent Hotchkiss and Meikle Sub-basins. The utility of these models, likely arises because fish communities in the Sub-basins are structured by the same local and regional factors, events and processes (i.e., Tonn 1990). In fact, the vast majority of sites within the Notikewin, Meikle, and Hotchkiss Sub-basins were located within the within the Low Boreal Mixedwood ecoregion and located within the same river basin.

In fact, concordance between fish community types and ecoregions has been reported previously although absolute concordance in fish community structure among ecoregions is rarely shown at least from sites located in adjacent ecoregions (Hawkes *et al.* 1986, Larsen *et al.* 1986, Whittier *et al.* 1988). For example, Whittier *et al.* (1988) reported moderately clear differences in fish communities among ecoregions in Oregon streams, USA. While contrary to their conclusion, Maret *et al.* (1997) showed high overlap of fish community structure in the Middle Rockies, Northern Basin and Range, and Snake River Basin/High desert ecoregions in Idaho and Wyoming but considerably less overlap with fish assemblages in the Northern Rockies ecoregion.

The use of ecoregion-based models of fish communities has potential value for fish assessments and management purposes especially if conservation concerns extend to include community types in addition to individual species. The utility of our fish community models would also be greatly improved if fish communities described in the Notikewin watershed are applicable to other watersheds within the Low- Boreal Mixedwood ecoregions. Additional research is required to determine whether fish community types described in the Notikewin are representative of those in other watersheds within the Low Boreal Mixedwood.

Our analyses were completed after excluding potentially impacted river reaches that could have reduced the strength of relations between fish communities and environmental characteristics. While developed for a different purpose, relations between fish communities and habitat characteristics from minimally impacted sites, can be used to: 1) assess impacts of current and planned industrial activities, 2) document recovery

from existing activities, 3) evaluate longer-term changes in fish communities, such as that which could arise from climate change, and 4) complete environmental screening exercises to determine whether fish communities at a particular site, perhaps that adjacent to planned activities, resembles that expected from regionally-based habitat models. The use of regionally-based fish community models can also be used to evaluate the success of mitigation and compensation actions. While we are not suggesting that the reference-condition approach is the best of several assessment techniques (e.g., Barbour *et al.* 1996, Bailey *et al.* 1998), most practitioners recognize predictive empirical models as being an important management tool (e.g., Tonn *et al.* 1983, Hughes *et al.* 1987, Boisclair 2001) and their use in fisheries management in Alberta needs to be increased.

9.5 To What Extent are Fish Communities Shaped by Forest Cover Attributes?

Riparian forests are known to perform important biological functions by retarding nutrient and sediment inputs, and reducing incoming solar radiation that maintain cool stream water temperatures and, indirectly, low algal biomass.

Riparian forests are known to perform important biological functions by retarding nutrient and sediment inputs, and reducing incoming solar radiation that maintain cool stream water temperatures and indirectly low algal biomass (e.g., Barton *et al.* 1985, Osborne and Kovacic 1993, Lowrance *et al.* 1997). Riparian zones provide cover for fish either directly while located within the riparian zone or indirectly by providing woody debris that provides instream cover, and are important sources of terrestrial invertebrates for invertivorous fish and a source of allochthonous carbon for detritivores that are a food resource for fish.

Several studies have reported that fish population or community density can be positively or negatively related to riparian vegetation cover although responses are typically species-specific.

Several studies have reported that fish population or community density can be positively or negatively related to riparian vegetation cover although responses are typically species-specific (Hawkins *et al.* 1983, Li *et al.* 1994, Barton *et al.* 1985, Glova and Sagar 1994, Tait *et al.* 1994) and questions related to effect thresholds remain largely unknown (Jones *et al.* 1999). Negative relationships between density and riparian vegetation are hypothesized to result from increased autotrophic production, due to increased light attenuation water temperature and nutrient loadings, and trophic cascades to primary consumers (Scrivener and Anderson 1984, Barton *et al.* 1985, Jones *et al.* 1999). Negative relations between fish density and riparian vegetation cover are often attributed to habitat degradation associated with reduced habitat heterogeneity (i.e., habitat diversity) and changes in light and thermal regimes (Li *et al.* 1994, Barton *et al.* 1985, Tait *et al.* 1994).

For example, Glova and Sagar (1994) reported significantly higher density and biomass of brown trout (*Salmo trutta*) in stream reaches bordered with willow (*Salix* spp) in two of three streams compared reaches where riparian vegetation was dominated grasses and short brush whereas the presence of willow did not affect the density of eels (*Anguilla* spp.), bullies (*Gobiomorus* spp.) and native galaxids (*Galaxius* spp.). Tait *et al.* (1994) reported significantly lower densities of steelhead trout (*Oncorhynchus mykiss*) and sculpin (*Cottus* spp.) with increased incident radiation and water temperature whereas many warm-water species

(reidside shiners [*Richardsonius balteatus*], bridelip suckers [*Catostomus columbianus*], northern squawfish [*Phytocheilus oregonensis*] and chiselmouth chub [*Acrocheilus alutaceus*]) were positively related to solar input associated with minimal riparian vegetation cover. Barton *et al.* (1985) reported significant relationships between maximum water temperature and the percent of stream banks forested.

Our results did not identify the percent of forest cover nor the percent of dominant overstory forest types within riparian areas as consistent and strong predictors of stream fish presence or community structure.

In contrast, our results did not identify the percent of forest cover nor the percent of dominant overstory forest types within riparian areas as consistent and strong predictors of stream fish presence or community structure. The lack of riparian forest cover to account for variance in fish community structure could arise because of the lack of a substantive gradient in forest cover among sites. However, our analyses were based on large gradients (0 to 100%) in percent forest cover and percent cover of individual forest types (0 to 85%) within the 100 to 300 m areas delineated adjacent to study reaches. Thus, it is unlikely that the inability of riparian forest cover to explain variance in fish presence, density and assemblage type results from a short gradient in this variable among study reaches.

We expect that the apparent inconsistency of results from our study with those which have typically demonstrated the importance of riparian forest cover on fish communities arises from differences in vegetative states of the non-forested riparian zone and the actions that lead to forest cover removal. The majority of studies that have documented significant effects of riparian forest cover on fish communities were associated with deforestation and conversion of land to agriculture. These activities are associated with other practices including road construction that can complicate interpretation of causal effects (e.g., Jones *et al.* 1999). Another difference is that non-forested riparian zones adjacent to study sites in the Notikewin watershed were still well vegetated (e.g., grassed) and were likely capable of reducing sediment and nutrient inputs (e.g., Lyons *et al.* 2000).

At the watershed-scale, forest cover can also influence the structure of fish communities by altering water yield, channel morphometry (e.g., abundance of pool versus riffle habitats), water physio-chemistry and through inputs of allochthonous matter (Hicks *et al.* 1991, Maret *et al.* 1997, Kilgour and Barton 1999). For example, Maret *et al.* (1997) reported forest as an important environmental factor determining the fish species distributions in stream in western United States and Kilgour and Barton (1999) showed that colder streams with greater forest cover were dominated by brook, rainbow trout and brown trout whereas warmer water reaches were dominated by cyprinids, ictalurids and darters rainbow darter and pumpkinseed.

Our analyses showed that in the relatively pristine Notikewin watershed, forest cover attributes were infrequent predictors of fish density and community structure.

Our analyses showed that in the relatively pristine Notikewin watershed, forest cover attributes were infrequent predictors of fish density and community structure. Potentially, this result could arise because of at least two reasons. First communities may be relatively resistant and resilient to changes in forest cover (Detenbeck *et al.* 1990) at least within

the existing forest cover matrix where watershed are predominantly forested and where non-forested areas are still vegetated. While it is well recognized that forests within Boreal ecoregions are strongly affected by natural disturbances on fire, flooding, and pest outbreaks that drive species composition and successional stages (e.g., Payette 1992) the effects of deforestation on aquatic ecosystems are equivocal and relatively little is known of the effects of harvesting on Boreal streams compared to lake ecosystems. For example, experimental and post-disturbance studies of Boreal forest lakes suggest that watershed and shoreline disturbance by logging and wildfire are associated with only minor impacts to phytoplankton, zooplankton, game fish, and small littoral fish and that such effects may be relatively short-lived (Rask *et al.* 1998; Carignan *et al.* 2000, Carignan and Steedman 2000, Scrimgeour *et al.* 2000, Steedman 2000, St-Onge and Magnan 2000, Tonn *et al.* In Press). In part, resistance and resilience to changes in forest cover could arise due to evolutionary history and life history and behavioural adaptations that allow community persistence at least within low to moderate changes in forest cover.

Second, effects of changes in forest cover on stream fishes in the Boreal may be moderated because the Boreal Plains ecoregion is characterized by low runoff coefficients, hydraulic energy and erosion potential compared with other forested ecoregions (Steedman *et al.* 2003).

9.6 Summary of Key Findings

9.6.1 Structure of fish communities in the Notikewin watershed

Fish communities in the Notikewin watershed comprise 17 species from 8 familial groups. Arctic grayling, longnose sucker, lake chub, brook stickleback, trout perch and northern pike were the most frequently encountered species and occurred at between 5 and 32% of all 266 sampling sites. Comparisons of relative abundance showed that Arctic grayling, lake chub, brook stickleback, longnose sucker, finescale dace and trout perch stickleback were numerically dominant and, when combined, accounted for 89% of all fish collected.

Overall density of fish in the Notikewin Watershed was low (Mean = 0.92 individuals 100 m²). Densities of brook stickleback, Arctic grayling, finescale dace and lake chub exceeded densities of 0.1 individuals 100 m². Within taxonomic groups, mean densities of cyprinid minnows (overall mean = 0.34 individuals 100 m²) exceeded that of salmonids (0.22 individuals 100 m²) and gasterosteids (0.27 individuals 100 m²). In general, fish communities were numerically dominated by small bodied fishes (i.e., cyprinids, gasterosteids, percopsids, cottids) compared to larger bodied forms (e.g., percids, salmonids, esocids, and catostomids).

9.6.2 Focal Question 1: How temporal variability is the fish presence and density?

Analyses of fish community data from sites sampled on two occasions between 1995 and 2001 showed that the presence of fish was highly variable and only 52% of all sites that contained fish in one year also contained fish one to five years later.

Concordance in the presence or absence of fish between sampling years varied with species group and stream size. Concordance in the presence of fish between sampling years increased with stream size from about 33% in first order streams to 67% fifth order streams.

Concordance in the absence of fish between sampling events was also variable; on average only 39% of all sites that did not contain fish on one sampling occasion were also devoid of fish one to five years later. Alternatively, 61% of sites that contained fish on one sampling period did not contain fish one to five years later. Concordance in the absence of fish between sampling years was also highly variable and did not decline consistently with increasing stream reach size (i.e., Strahler stream order).

Linear regression indicated that the number of species replacements (i.e., number of species gained or lost between sampling years) was positively related to stream bankfull width. Multiple regressions showed that the relative number of species replacements was negatively related with bankfull width and positively related with time between repeated samples.

Mean total fish density in the Notikewin watershed was low and ranged from 0.24 ± 0.13 to 2.2 ± 1.39 100 m^2 . Densities of Arctic grayling and brook stickleback, the most numerically dominant species at the 27 sites sampled repeatedly during 1995 – 2001, were also low and typically ranged between 0.1 to 1.8 and to 0.01 to 0.3 100 m^2 , respectively. Densities of lake chub, finescale dace, longnose suckers, pearl dace, sculpin, trout perch and white sucker seldom exceeded 0.2 100 m^2 .

Fish density was also temporally variable and varied by about 10-fold between 1995 and 2001 (Highest annual mean $\pm 1\text{SE} = 2.2 \pm 1.39$ 100 m^2 , lowest annual mean = 0.24 ± 0.13 100 m^2). Density of Arctic grayling was also highly variable and differed 20-fold during the six year period (1.82 ± 1.1 to 0.088 ± 0.05 100 m^2). Densities of stickleback other numerically dominant were also variable among years and ranged between 5 and 25 fold during the six year study period.

9.6.3 Focal Question 2: Is the presence of fish predictable based on stream habitat and watershed variables?

The occurrence of fish was strongly affected by stream size. While only about 40% of first and second order and small permanent reaches contained fish, the majority (0.65 to 0.90%) of third, fourth, fifth and large permanent stream reaches contained fish. Stream size also strongly

affected the presence of individual species and species groups. Game fish, cyprinids, catostomids, and Arctic grayling occurred infrequently (<35% occurrence) in small streams (1st to 3rd and small permanent streams) but occurred 20 to 40 times more frequently in larger streams. Brook stickleback, lake chub, longnose sucker, white sucker, northern pike, and trout perch occurred most frequently in larger streams.

At the landscape level, logistic regression analyses showed the presence of fish, game, cyprinid minnows and Arctic grayling were highly predictable based on stream bankfull width or bankfull and northing (Cyprinidae) or bankfull width and percent gravel.

Reach elevation was the strongest predictor of the presence of suckers, longnose sucker, lake chub and brook stickleback. While reach slope was the strongest predictor of the presence of northern pike, bankfull width and elevation, which were correlated with reach slope were also statistically significant single variable predictors.

Single variable models using these predictors explained less variance in the presence of northern pike and resulted in lower classification success. These models generally explained between 0.42 to 0.74 of the variance in the presence of fish, fish type and individual species and correctly classified sites where fish should be present or absent compared to presence based on electroshocking between 71% and 93% of the time.

In the majority of case, variables that predicted the presence or absence of fish were measures of stream size or related to measures of stream size.

At the stream reach level, bankfull width, water temperature, slope and to a lesser extent, elevation and gravel were significant predictors of the presence of fish, game fish, taxonomic groups and individual species in first to fifth order reaches.

After excluding models which were not calculated due to low sample sizes, 13 of the 20 logistic regression models were statistically significant ($P < 0.05$) and all were statistically significant at $P < 0.08$. These models explained between 0.51 and 0.75% of variance in the presence or absence of fish and on average correctly classified sites as containing fish or where fish were absent in the majority of cases (Mean \pm 1 standard deviation = 0.756 \pm 0.09, Range = 0.59 to 0.91).

Analyses based on classifying reaches using Alberta watercourse classes, also identified reach elevation, bankfull width and to a lesser extent, slope and gravel as statistically significant predictors of the presence of fish, game fish, taxonomic groups and individual species in small and large permanent stream reaches. In total, 10 of the 12 models (i.e., 83.3%) were statistically significant ($P < 0.05$) and 11 of the 12 models were statistically significant at $P < 0.06$. These models explained between 0.53 and 0.87% of variance in the presence or absence of fish (Mean \pm 1STDEV = 0.68 \pm 0.08) and on average correctly classified sites as containing fish or

where fish were absent in the majority of cases (Mean±1STDEV = 0.714±0.10, Range = 0.53 to 0.87).

The effects of stream classification method on the ability of logistic regression models to predict the presence and absence of fish was evaluated by comparing model fit and classification success between models created using Strahler stream orders and that using Alberta watercourse classes. With two exceptions, the stream classification method had only minor effects on the fit of logistic regression models and overall classification success. However, the ability of logistic regressions to correctly classify large permanent reaches as containing game fish and Arctic grayling were low (classification successes = 0.53% and 0.59%) and potentially problematic if the ability to predict the presence of game fish and Arctic grayling is an important criterion in developing predictive models.

9.6.4 Focal Question 3: Do stream reach and watershed characteristics shape fish assemblage structure?

The canonical correspondence ordination indicated that the first two axes explained 24.6% of the total variation in fish density (Axis 1 = 13.21%, Axis 2 = 11.36%). When constrained by environmental variables, Axes 1 and 2 explained 80.8% of the variation between species abundances and environmental variables.

Increased densities of white sucker, northern pike, sculpin and trout perch were positively related with cobble, water depth, bankfull width and gravel. Density of Arctic grayling was positively related to elevation whereas increased density of longnose sucker was negatively related with elevation. Densities of dace, shiner, brook stickleback and chub were negatively related with cobble, gravel and water depth and bankfull width.

Multiple regressions using forward selection showed that total fish density and density of the dominant groups were only moderately or weakly related with environmental variables. Total density and density of individual species and species groups was typically significantly related to cobble, gravel, elevation, water depth and width and elevation. These regression models explained between 5 and 32% of variance in fish densities and often (5 of 9 regression models) accounted for 18 to 32% of variance in individual species and species groups.

Density of Arctic grayling was positively related with elevation and negatively related with water depth and forest cover. Density of brook stickleback was negatively related with cobble, bankfull depth and gravel whereas density of longnose sucker and trout perch were negatively related with elevation and positively related with bankfull width, respectively.

9.6.5 Focal Question 4: To what extent are relations between fish assemblages and watershed variables applicable to adjacent watersheds?

Hierarchical cluster analyses of fish density identified three relatively discrete fish assemblages in the Notikewin Sub-basin and a fourth cluster of residual sites.

Based on fish density data, assemblage 1 consisted primarily of Arctic grayling, brook stickleback and dace whereas assemblage 2 consisted of Arctic grayling, chub, white sucker, longnose sucker and northern pike. For these assemblages, mean total density and density of the three most numerically abundant groups ranged from 1.42 to 1.77 / 100 m² and 0.15 to 0.69 / 100 m², respectively. In contrast, assemblage 3 was comprised of high densities of brook stickleback, dace, chub and to a lesser extent longnose sucker. Overall, mean total density and density of the three most abundant species was 12 to 15 and 18 fold higher than that in Assemblages 1 and 2.

We attempted to distinguish among the three fish assemblage types using multiple discriminate function using watershed and instream variables. Based on fish density data, the forward selection discriminant analysis identified percent gravel and cobble, site elevation and reach slope as significant (Wilks' Lambda, $P < 0.0001$) discriminators among the three fish assemblages. The linear discriminant function model had an overall classification success of 76% (i.e., 48 of the 63 sites were classified correctly) and correctly classified 73%, 75% and 100% of sites into assemblage 1, 2 and 3, respectively.

Stream reaches that supported Assemblage 1 were typically located at higher elevations and had higher amounts of gravel within the substratum compared with those belonging to Assemblages 2 and 3. In contrast, water depths and percent cobble at sites supporting assemblages 1 and 2 were greater than those for assemblage 3.

We tested the utility of the discriminant function model (based on density and percent composition) derived for fish communities in the Notikewin watershed by applying the model to fish communities described at 39 sites in the adjacent Hotchkiss and Meikle Sub-basins. Prior to these analyses, all 39 sites were classified into fish assemblage types using clustering methods described previously.

When applied to the Hotchkiss and Meikle Sub-basins, the discriminant function models correctly classified 74.4% (i.e., 48 of the 63 sites were correctly classified) of all sites and correctly classified 4 of 5 sites into assemblage 1 (i.e., classification success = 80.0%), 17 of 23 sites into assemblage 2 (classification success = 73.9) and all 8 of the 11 sites into assemblage 3 (i.e., classification success = 72.7%). These data suggest that the discriminant function models developed from the Notikewin Sub-

basin are relatively good predictors of fish community types in the adjacent Hotchkiss and Meikle Sub-basins.

9.6.6 Focal Question 5: To what extent are fish assemblages shaped by forest cover attributes?

Logistic regression analyses showed that forest cover attributes were typically poor predictors of the presence of fish, game fish, taxonomic groups and individuals species compared with regression models derived using both non-forest physiographic attributes and forest cover attributes. Overall, only 50% (16 of the 32 models) of empirical models using forest attributes were statistically significant ($P < 0.10$) predictors of the presence of fish. The remaining models were either statistically non significant or that the maximum likelihood estimates did not exist and the model fits were questionable. In contrast, 97% (31 of 32 models) of empirical models using both physiographic and forest cover attributes were significant predictors ($P < 0.10$) of fish presence and in the majority of cases (23 of 32 models) models were statistically significant at an alpha of 0.05. Lastly, forest cover attributes were included within only 12.5% (4 of 32) of best-fit models.

Comparisons between best fit models and those using forest cover attributes using Strahler stream orders indicated that on average forest cover models were not statistically significant (mean model $P = 0.12$) compared with models derived for stream defined using the Strahler stream order ($P = 0.039$) and had lower classification success but only marginally lower explanatory power (i.e., R^2).

Comparisons of model fit between sites defined using Alberta watercourse classes showed that models derived using forest cover attributes were less powerful and had lower predictive power and classification success compared to those derived using non-forest physiographic attributes and forest cover attributes.

The extent to which forest cover explained variation in fish community structure was evaluated by partitioning the variance in fish communities into that which could be explained by: 1) forest cover attributes and 2) physiographic, non-forest cover attributes using variance partitioning techniques.

For this analysis, forest cover attributes included those measured at the watershed scale and those measured within the three riparian zones areas (i.e., areas of 3.16 ha, 28.13 ha, 78.14 ha) delineated by establishing radius of 100 m, 300 m and 500 m radius around the sampling point.

Multiple canonical correspondence analyses showed that the environmental variables explained a relatively small proportion of the overall variance in fish density. When decomposed, the majority (45.6%) of explained variance in fish communities was attributed to non-forest cover attributes of elevation, mean water depth, reach slope, and the percentage of gravel and cobble within the substratum. Forest cover

attributes including percent of watershed forested, and percent conifer, deciduous accounted for 29% of the variance that could be explained with the remaining 25.4% attributed to the interaction of forest cover and non-forest cover attributes.

9.7 Management Implications and Challenges to Stream Fisheries Management

9.7.1 Implications

Empirical models relating fish communities with watershed variables can provide fisheries managers with an effective management tool.

Our data indicate that logistic regression is a potentially powerful and relatively simple method to predict the presence of fish, game fish and some individual species in the Notikewin watershed. Variance in the presence of fish was accounted for by a small suite of recurring variables of stream bankfull width, temperature, percent gravel and to a lesser extent, forest cover attributes of percent conifer and the percent of conifer within 300 m of the study reach.

The development and refinement of multivariate statistical approaches (e.g., Canonical Correspondence Analysis [ter Braak 1986, Palmer 1993], variance partialling [Borcard *et al.* 1992], metric and non-metric multidimensional scaling [Kenkel and Orloci 1986]) and software (e.g., Canoco [ter Braak and Smilauer 1998], PC-ORD [McCune and Mefford 1999]) over the last 15 to 20 years has provided ecologists with an opportunity to identify and gain insights to larger-scale patterns in fish community structure. We suggest that these approaches could provide fisheries biologists with an improved understanding of the factors associated with spatial variation in fish communities. This information should be communicated to landscape managers to provide them the best scientific information on which to manage northern landscapes.

9.7.2 Challenges to stream fisheries management

A large body of scientific evidence has shown that fish communities are vulnerable to watershed disturbances such as those arising from forest harvesting, exploration and extraction of oil and gas and construction of road and telecommunication networks, urbanization and conversion of forested watershed to agricultural production.

A large body of scientific evidence has shown that fish communities are vulnerable to watershed disturbances such as those arising from forest harvesting, exploration and extraction of oil and gas and construction of road and telecommunication networks, urbanization and conversion of forested watershed to agricultural production (Berkman *et al.* 1986, Bisson *et al.* 1992, Binns *et al.* 1994, Weaver and Garman 1994, Hartman *et al.* 1996, Martin-Smith 1998, Baxter *et al.* 1999, McIntosh *et al.* 2000, Trombulak and Frissell 2000, Wildhaber *et al.* 2000, Kreutzweiser and Capell 2001). However, such negative impacts are often ambiguous, resulting in context dependent effects that are difficult to predict in other watershed types (Dunham *et al.* 1999, Rose 2000). More importantly, the detection of watershed disturbance impacts on fish communities and the interpretation of non-detectable events in the same region are seldom viewed in terms of disturbance thresholds (e.g., Bradford and Irvine 2000). Under a disturbance threshold management model, fish communities may be protected if the sum of all impacts does not exceed a

threshold value. Beyond this level, however, fish communities will be adversely affected.

In Alberta, fish communities are protected through the Canadian *Fisheries Act* (Fisheries and Oceans 1986, 1991). Habitat protection provisions (Section 35 [1]) of the Fisheries Act 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. Consequences of watershed activities on aquatic environments and the biological diversity that they support are also considered within Provincial Acts and Regulations (e.g., The *Water Act*, Timber harvest and planning operational Ground Rules [Anonymous 1994]). Provincial regulations include operating ground rules for forest harvesting practices, and provincial codes of practice for: 1) watercourse crossings, 2) pipeline and telecommunication lines crossings and 3) temporary diversion of water for hydrostatic testing. While these provisions may protect stream fish populations, a number of issues test our ability to manage stream fish assemblages and challenge the view that current management practices are based on the best scientific information available. They include:

- 1) Assessments of the potential effects of industrial, municipal and agricultural activities on stream fish are seldom completed using spatially accurate and up-to-date information. This paucity of information challenges resource managers because they may not be fully aware of species-complex present and thus can not accurately evaluate potential consequences of industrial activities of those species.
- 2) Fisheries assessments are typically completed at small spatial scales with little consideration of the importance of larger landscape patterns and processes. Management of stream fish communities would be enhanced if evaluations were based on assessments completed at multiple spatial scales.
- 3) 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.
- 4) Current approaches to stream fish management do not take full advantage of quantitative tools or recognized quantitative relationships. The use of these approaches and tools can assist stream fish management 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 process that may drive them, and vi) monitor effectiveness of current management actions.

5) A better understanding of the causal mechanisms responsible for the degradation of stream fish communities would enhance stream fish management by: i) identifying the major causes of negative impacts on fish populations, ii) allow scenario modeling exercises to forecast long-term and large scale consequence of alternative management actions and iii) providing an ecologically sound approach to restoration actions.

6) An improved understanding of the cumulative effects of watershed disturbances on stream fish communities is required including the effects of land use conversions (i.e., conversion and loss of forested watersheds to agriculture).

Increased research and improved planning and communications would greatly assist with the management of Alberta's stream fisheries.

While these issues represent a sobering challenge to current stream fisheries management approaches, we are optimistic that increased research combined with improved planning and communications will provide much of the knowledge base required to manage stream fish communities. Sustaining stream fish communities will require additional financial investments to provide fisheries managers with pragmatic tools that they can use to assess and forecast changes in fish communities based on watershed disturbances. For example, the use of empirical models that: i) define natural patterns in fish community structure and ii) forecast changes in fish communities based on changes in watershed disturbances has been only slowly adopted especially by non-academic institutions. The use of these tools to assist with the management of stream fish communities in Alberta is in its infancy. Resource managers need to be better equipped, including the allocation of additional resources, to understand the complexities of stream fish dynamics. Ultimately management decisions need to be based on the best scientific information available if the sustainable management of stream fish and fisheries is to be achieved.

Management decisions need to be based on the best scientific information available.

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The Northern Watershed Project Stakeholder Committee

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