Disturbance Related Patterns in Fish Community Structure and Function in River Systems of the Lower Athabasca Oil Sands Region, Alberta

by

Shannon Ashley McFadyen
BSc., University of Guelph, 2008

A Thesis Submitted in Partial Fulfillment
of the Requirements for the Degree of

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in the Department of Geography

Supervisory Committee

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Abstract

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Anthropogenic development is altering watersheds and threatening freshwater ecosystems and the resources therein. Direct impacts of industry including conversion of land cover and increased water withdrawals from rivers, compounded with indirect influences such as climate change, collectively affect the health and sustainability of freshwater ecosystems. Many studies have indicated a suite of ecological impacts that large-scale anthropogenic land use and development impose on the structure and function of riverine systems. The overarching goal of this thesis was to examine the potential impacts associated with land use disturbance and Oil Sands (OS) mining operations on fish community composition patterns in three rivers located in the Athabasca Oil Sands Region (AOSR). Using historical data sets, this thesis attempted to evaluate disturbance-related patterns in fish community composition. Fish community-environmental relationships were investigated on a temporal scale, across which community composition could be constrained or altered by development. Structural and trait-based changes in fish community composition were analyzed to determine whether significant variation between levels of development (pre versus post) in the AOSR could be attributed to observed changes in fish community metrics. No significant difference in community composition patterns was observed between levels of development; however, there was a significant decline in fish species richness on a regional scale. The lack of significant results could be attributed to the limitations of the collected data, including temporal gaps, inconsistent sampling methods, and seasonal sampling inconsistencies. Furthermore, the scale of interpretation between individual tributaries and the regional datasets, demonstrates that studies of fish communities on a regional scale can elucidate different states of community change, implying that local controls can play a role in species presence/absence.

An assessment of the features and patterns of the hydrograph that could explain variation in fish communities was constrained due to dataset and subsequent methodological limitations. Currently, there is an inability to link changes (historical) to hydrologic regimes, land use or development within these systems, and how they have impacted fish communities therein due to inconsistencies in the methods and sampling during most of the pre-development and for a portion of post- development time span (until 2009). Long-term, standardized community monitoring will be critical to gain a greater understanding of how land management practices affect fish communities and what kind of ecosystem management can mitigate impacts to streams, rivers and the biota therein. Further recommendations were made from synthesizing these findings in conjunction with relevant literature and are intended to provide an improved understanding of the long-term cumulative changes within these systems and to help guide and improve future monitoring plans in the AOSR.

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Dedication

To my sisters Amanda and Katelyn for being there through it all. Thank you.

Chapter 1. Introduction

1.1 Current State of Knowledge

Anthropogenic development is altering watersheds and threatening freshwater ecosystems and the resources therein (Schindler, 2001). Activities such as agriculture, urbanization, industrial development and forestry, for example, are quickly changing the form and function of terrestrial ecosystems (Allan, 2004). The direct impacts of industry including conversion of land cover and increased water withdrawals from rivers, compounded with indirect influences such as climate change, collectively affect the health and sustainability of freshwater ecosystems (Schindler, 2001; Seitz et al., 2011; Meyer et al., 1999). Many studies have indicated a suite of ecological impacts that large-scale anthropogenic land use and development impose on the structure and function of riverine systems (Allan, 2004; Gergel et al., 2002; Wang et al., 2001). Together, these impacts are expected to change the natural hydrologic regime and; result in habitat loss and modification, which limits the ability of local fish and other freshwater fauna to avoid potential stressors (Allan, 2004; Jelks et al., 2008; Katz, 2014).

The Athabasca Oil Sands Region (AOSR; Alberta, Canada) has been developed for oil sands (OS) extraction since the 1970s. Rivers are influenced by the landscapes through which they flow (Hynes, 1970), and large-scale anthropogenic development like the OS have demonstrated many ecological consequences on surrounding structure and function of freshwater rivers, including changes in the flow regime (Allan, 2004; Figure 1.1). Impacts or changes to flow regimes have been cited to be one of the leading causes decline in freshwater fish species (Katz, 2014; Naiman and Turner, 2000; Richter, et al., 1997). It is difficult to untangle the effects of the collective direct and indirect anthropogenic impacts on these systems (Taylor et al., 2008). However, these disruptions are hypothesized to impose changes on the native fish communities in rivers within the AOSR that will manifest at a local (within each river) and regional (across the AOSR) scale. Fish communities have been identified as an important ecosystem component and shifts in the community composition are often used as indicators of changes in environmental conditions (Karr, 1981). Collecting and analyzing retrospective data will provide a fundamental, historical perspective to detect and quantify changes to river biota, including fish communities (Taylor et al., 2008). Furthermore, characterizing long-term trends in these systems can aid in identifying ecological limitations that should be taken into consideration for conservation and to set targets for management and restoration (Gido et al., 2010).

Figure 1.1 The Shell Albian Sands tailing pond are part of the large-scale anthropogenic OS development adjacent to the Muskeg River. (Source: globalforestwatch.ca)

1.2 Impacts of Landscape Changes on Rivers

The hydrology, geomorphology, water chemistry and biota in a lotic environment are all determined by regional factors, including geology, climate, local land cover and local land use (Wang et al., 2001). The direct physical impacts of land use on rivers have been well documented in literature. The principal mechanisms through which landscape alterations can impact rivers include hydrologic alterations, such as increase runoff volume to streams (Argent and Carline, 2004; Poff and Allan, 1995; Wang et al., 2001). Such increases can result in changes to channel morphology (Wang et al., 1997; Wang et al., 2001), hydrologic and thermal regimes (Argent and Carline, 2004), nutrient loading (Argent and Carline, 2004; Scott et al., 1986), flow patterns, water quality, an increase in precipitation runoff rate, and the volume, frequency and magnitude of floods (Wang et al., 2001). These changes to the hydrologic regime of a river can lead to a decrease in channel stability and an increase in bank erosion, turbidity, stream bed scouring and deposition of sediments on and within a streambed (Allan, 2004; McCart and Mayhood, 1980; Wang et al., 1997).

1.3 Impacts of Changing River Hydrology on Fish Communities

Environmental variability, as a factor in structuring biological communities, is a prominent topic of interest in ecology. In stream ecology, flow extremes and variability is important temporally, within and between lotic environments (Poff and Ward, 1989). The presence, diversity and abundance of fish have historically been influenced by a suite of geological, physical and biological variables (Argent and Carline, 2004). According to Hynes (1970), flow can play a central role in structuring stream ecology through its ability to change stream characteristics and physical attributes effectively shaping habitat for fish and other river biota. Habitat alterations such as landscape disturbance can potentially alter fish communities through changes in niche form and suitability (ter Braak and Verdonschot, 1995).

From a physiological perspective, the primary ecological consequences of land disturbance in relation to fish communities are altered flow regimes. For aquatic biota, this includes changes in habitat-related structural features such as: discharge regime(s), current velocities, morphological structure of a riverbed and banks, erosion, substratum stability, habitat and siltation (Bunn and Arthington, 2002; Poff and Ward, 1989; ter Braak and Verdonschot, 1995). Variation in river flow regimes and extreme conditions (e.g., floods and low to zero flow events) are, according to Stanford and Ward (1983), primary sources of environmental variability and disturbance. The impact of the aforementioned alterations to streams and rivers has been widely recognized as a principal threat to the ecological integrity of river ecosystems through the (demonstrated) influence on local habitat, biota, and water quality and quantity in streams and rivers (Allan, 2004; Poff and Allan, 1995; Wang et al., 2001).

More specifically, transitions from a natural to disturbed landscape have been observed to decrease abundance and diversity of fish communities (Allan, 2004; Lenat and Crawford, 1994). Fish communities are sensitive to anthropogenic influences for many reasons. A community of fish can occupy a variety of niches, feed across trophic levels, demonstrate different migratory behaviors and spawning strategies, and utilize aquatic respiration by obtaining oxygen from water (Argent and Carline, 2004). Fish communities are thus structured by their interactions with the surrounding biotic and abiotic environment, and changes to a landscape can have direct and indirect impacts on a community (Allan, 2004; Argent and Carline, 2004; Olden et al., 2010; Wang et al., 2001).

Over time, fish have evolved traits alongside the natural flow regime of the rivers they inhabit to survive, exploit and persist in (Lytle and Poff, 2004). For many species the natural variability in habitat types, regulated by the average long-term dynamics of the flow regime, is essential for survival (Lytle and Poff, 2004; Poff et al., 1997; Sparks, 1995). For example, the lifecycles of many fish species are timed to avoid or exploit different environmental cues, including changes in base flow that initiate events such as spawning, reproduction or migration (Bunn and Arthington, 2002; Nesler et al., 1988; Poff and Allan, 1995; Poff et al., 1997; Sparks, 1995). Changes to these largely flow-determined habitat patterns could affect the presence and distribution of species within a system, if the changes in conditions are outside those adapted to by native biota (Poff et al., 1997). As a result, changes in fish communities are a strong bio-indicator of changes to river ecosystem health, alterations in water quality and integrity of the river ecosystems.

1.4 Background: The Athabasca Oil Sands Region

The AOSR is located in the Boreal Plains of Alberta, north of the city of Fort McMurray. Due to rich oil-deposits and global-demands for oil supply, large-scale development and land use began in the AOSR in the 1970s. Development in the region is proposed to continue over the next few decades, continuing to alter the landscape through land clearing, mining operations, and infrastructure development (Humphries et al., 2008).

Despite the vast literature available on anthropogenic disturbances in riverine systems and watersheds (see Allan, 2004), few investigations, if any, have been done on the effects of development in the AOSR on changes in the ecology of fish communities. Similar studies have found that areas with high levels of disturbance at the watershed scale have led to spatial and temporal changes in the geomorphic processes. These disturbances have demonstrated effects on river ecosystems and the fish communities therein, through habitat alteration (Allan, 2004). While fish in the AOSR have been studied for changes to morphology, health, ecotoxicology and populations, there remains a gap in the literature looking at shifts or changes to the community structure in these rivers. According to King (2014), lacking the information on causal linkages between the impacts of landscapes and community responses in river ecosystems, such as those in the AOSR, will be a continued source of major problems for management, conservation and eventually restoration.

1.4.1 Athabasca Oil Sands – McMurray formation

The McMurray formation underlies 140,200 square kilometers (km^2) of land, and is the source of OS in the AOSR in Northern Alberta (Government of Alberta, 2007; Carrigy, 1959). The McMurray formation is exposed at the surface and, as a result of erosion, it occurs fairly shallow along the Athabasca River (mostly < 75 m). The shallow depth of the deposit makes the mining of these sands possible (Yasuda, 2006).

Oil sands (OS) development began in the region in the 1960s and bitumen has been extracted and produced from the McMurray formation since 1967 through open-pit, subsurface, and *in-situ* techniques, including Steam-Assisted Gravity Drainage (SAGD) (Government of Alberta, 2007). In 2012, crude oil production from the OS was 88.4

thousand m^3/day and 32.3 million m^3/year (Government of Alberta, 2014). The Alberta Government forecasts that production could reach 560 thousand m^3/d by 2020 and is projected to reach 790 thousand m^3/d by 2030 (Government of Alberta, 2007).

Studies have been conducted in the AOSR to monitor and assess potential environmental impacts from OS development on the terrestrial and aquatic ecosystems by the Alberta Oil Sands Environmental Research Program (AOSERP), Alberta Environment (AENV), Environment Canada (EC), and most the Regional Aquatics Monitoring Program (RAMP), to name a few. In 2011, the Canada-Alberta Joint Oil Sands Monitoring Program (JOSMP) was assembled to provide *"rigorous, comprehensive, integrated and transparent environmental monitoring to the region"* (Joint Oil Sands Monitoring Program [JOSMP], 2014). According to JOSMP, mineable and *in situ* oil sand developments could affect fish through habitat loss, or landscape fragmentation through land clearing, and changes in water regimes resulting from hydrological disturbances (JOSMP, 2012). With possible co-occurring processes including changes in water quantity by climate change, other regional industry such as forestry (Prowse et al., 2006), and projected future development, the impacts of the disturbance to biological communities in river ecosystems in response to further OS development needs to be quantified.

1.4.2 Local land use

Historically, land use in the lower Athabasca River region was predominantly dedicated to agriculture and pulp and paper production (Government of Alberta, 2012). In the past 40 years; however, OS development has expanded to cover more than 767 km^2 with a total of 4,800 km^2 of surface mineable area (Government of Alberta, 2014).

Underlain by minable OS, all three rivers have been, and continue to be, developed for resource extraction (Government of Alberta, 2011).

The development of the AOSR has led to an increase in population, urban development and associated infrastructure in and around the city of Fort McMurray, Alberta. Secondary impacts from development have been cited as increased recreational use, angling and fishing pressures on rivers in the AOSR from roads that provide increased access to rivers and lakes (Gunn and Sein, 2000; Post et al., 2002; Schwleb et al., 2014).

1.5 Study Sites

Long-term changes in fish community composition are evaluated for the Muskeg, Steepbank and Ells River watersheds of the AOSR. All three rivers are tributaries of the Athabasca River (Figure 1.2). Tributaries that were selected for analysis were included due to the availability of data, both fish and environmental (flow) to draw meaningful conclusions from, as well as their proximity to existing and approved development plans in the AOSR.

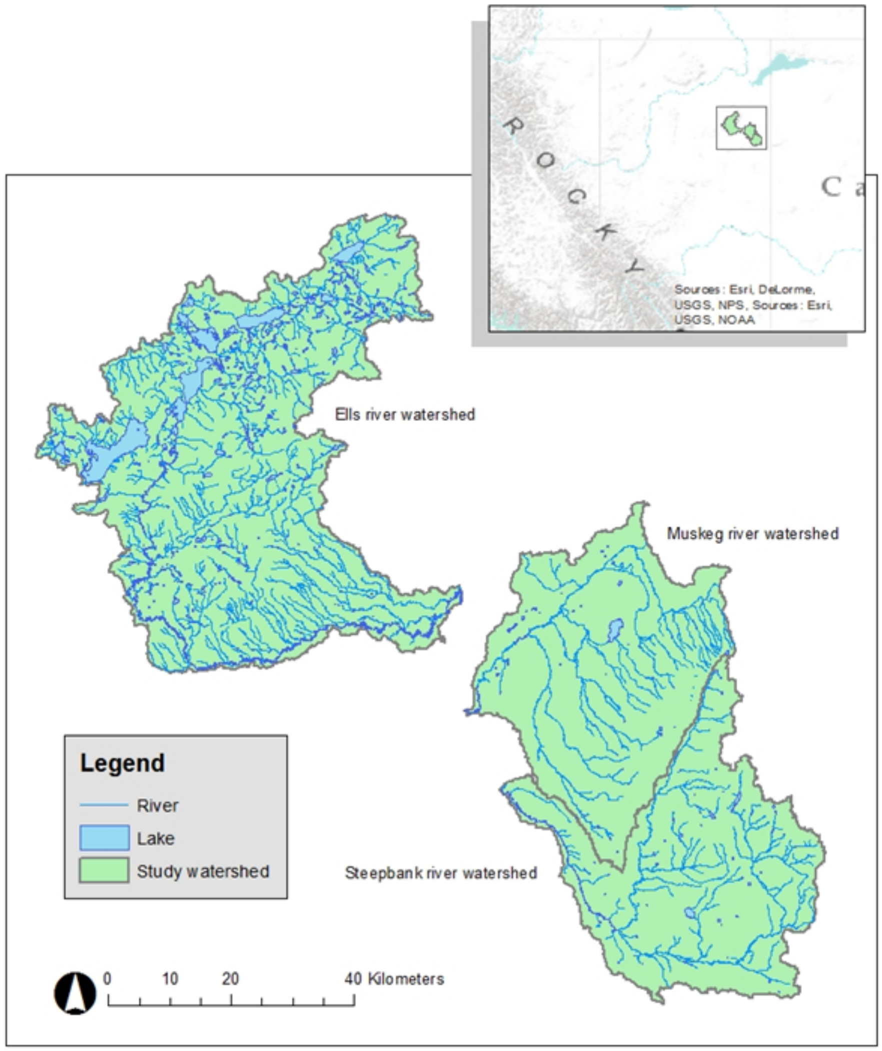

Figure 1.2 Long-term changes in fish community composition were evaluated in the Muskeg, Ells and Steepbank watersheds of the AOSR. Source: Regional Aquatics Monitoring Program / L. Reshitnyk.

1.5.1 Muskeg River Watershed

The Muskeg River Watershed is located within a Boreal Forest and covers an area of about 143,304 ha. Located approximately 55 km north of Fort McMurray, the Muskeg is a tributary of the Athabasca River in the Regional Municipality of Wood Buffalo.

Within the watershed, 14.71% of land was disturbed in 2012 (Regional Aquatics Monitoring Program [RAMP], 2012). Future activities are projected to disturb 50-60% the watershed area (Government of Alberta, 2011). Currently, there are five operational OS projects and one limestone quarry in the watershed (with another two projects approved for development) (Government of Alberta, 2014).

The Muskeg River is a forth-ordered, medium sized stream with hydrology typical of Boreal forest systems (Government of Alberta, 2011). Across the greatest number of days within a given year for which data were available (March 1st to November 6th), pre-development mean annual discharge, from 1974 to 1995, is approximately 5.53 m^3/s, and post-development, from 1996-2012, is 4.48 m^3/s (Hydrological Survey of Canada, 2014). Seasonal events affect the streamflow including snowmelt, summer thaw of peat lands, and winter ice cover, with peak flow usually during the freshet (late April/May). For the remainder of the year, shallow groundwater is the main source of streamflow to the Muskeg (Government of Alberta, 2011).

1.5.2 Steepbank River Watershed

The Steepbank River watershed covers an area of approximately 136,395 ha (RAMP, 2012) and is a tributary of the Athabasca River, residing on the eastside of and flowing westerly towards the main stem. The headwaters originate in the Muskeg Mountains, and the watershed drains an area of 1,424 km^2 (RAMP, 2012), majority of which is treed muskeg. Substrate in the lower reaches of the Steepbank is made up of gravel, boulders and tar sands, in the riffles and with silt, sand and OS in the pools. Banks of the river are vegetated with grass, willows, and a forest of poplar and spruce (Machniak and Bond, 1979). The lower reaches where fish samples were taken; are characterized by steep and eroded banks (Conly et al., 2007). According to RAMP (2012), approximately 3.70% of the land cover in the watershed has been disturbed in 2012, with one operational project located in the watershed (Government of Alberta, 2014).

The Steepbank River is a fifth order stream and flow has been recorded since 1974. Across the greatest number of days within a given year for which data were available (February 27th to October 16th), pre-development mean annual discharge, from 1972 to 1993, is approximately 7.25 m^3/s, and post-development, from 1994-2012, is 6.69 m^3/s (Hydrological Survey of Canada, 2014). The upper reaches form a pattern of meandering until the river enters the McMurray formation where the pattern becomes weaker as a reflection of the need to dissipate energy. The basin stores relatively large amounts of groundwater during wet periods. A rapid increase in flow rates during the freshet occurs from the slopes of the Muskeg Mountain. This usually results in the maximum annual flow rates (RAMP, 2013). There has been recorded another high flow event in the summer or early autumn due to an increase in precipitation (RAMP, 2013). Low flows occur in the winter months, as are characteristic for the region, when precipitation is commonly stored in snow. Additional low flows are recorded during the dry season (summer months) (RAMP, 2013). Furthermore, the Athabasca can reach discharge rates in excess of 1130 m^3/s, which according to Barton and Wallace (1979) can cause an increase in water levels in the Steepbank, altering the flow regime in the lower reaches.

1.5.3 Ells River Watershed

The Ells River originates in the southeast slopes of the Birch Mountains at an elevation of approximately 730 m, and descends to an elevation of approximately 250 m (RAMP, 2013). A large portion of the river meanders through a region dominated by boggy landscape, and is the only river of the three that flows into the Athabasca from the west (Headley et al., 2001). This is the lesser-developed watershed of the three used in this study. The watershed area is 270,944 ha with approximately 1.07% land disturbed in 2012 (RAMP, 2012). There are two proposed and one active mine in the watershed (Government of Alberta, 2014).

Superficial deposits of glacial till dominate the area drained by the Ells. The reach closest to the mouth of the Ells moves into the McMurray formation described above and is characterized by steep cutting banks along various portions of the river with OS deposits forming part of the bed material (Headley et al., 2001; Griffiths, 1973). This is where a majority of the fish samples have been collected.

The Ells River is a fourth order stream and flow has been recorded since 1974. Limited flow monitoring over the period from 1976 through 2012 suggests that the mean annual flow dropped from approximately 9.0 m^3/s to 8.50 m^3/s. The headwaters of the Ells River is located within the Birch Mountains, are inundated with many small lakes (approximately 44 in the watershed). A substantial amount of the watershed hydrology is influenced by the storage capacity of these lakes, including moderated surface runoff and high base flows through the winter (Headley et al., 2005; Sekerak and Walder, 1980).

1.6 Fish Communities in the AOSR

Some fish species in the AOSR are long-lived; they utilize a broad range of river reaches for migration at different times of the year, feed from a variety of trophic levels, and have varying requirements for spawning (Argent and Carline, 2004; Karr et al., 1981; Metcalfe et al., 2013; Munkittrick et al., 2009; Schwleb et al., 2014). As a result, fish communities are often identified as important ecosystem components and shifts in the community composition often an indicator of changes in physical and chemical properties of river systems. A comprehensive assessment of fish communities is a good indication of ecological condition compared to single species assessments (Metcalfe et al., 2013).

The AOSR supports a rich community of fish species (Table 1.1). Many species in these communities are important for social and economic reasons, including Arctic grayling, which are valuable recreational species, heavily sought after by anglers (Armstrong, 1982; Armstrong, 1986). The Muskeg River is home to 26-recorded fish species, the Steepbank River to 24 fish species and the Ells River to 18 (Table 1.1).

At present, a long-term, standardized monitoring database for fish communities in these tributaries does not exist. Early research was carried out in the late 1970s by the AOSERP to create a baseline for potential future development. For each of the Muskeg, Steepbank and Ells River(s), large gaps exist in the database from the early 1980s until 1995/96. In 1977 and then again from 1995 onwards, the Alberta Sustainable Resources and Development (ASRD) has collected fish data maintained by the Fisheries and Wildlife Management Information System (FWMIS), a centralized database that contains occurrence records for fish and wildlife species in the region. Since 1997, the RAMP and

ASRD have been the dominant fish and freshwater monitoring program for the areas directly affected by the Oil Sands and associated developments. More recently, the JOSMP monitors the region. Since 2011, JOSMP has begun to apply the Joint Canada|Alberta Implementation Plan for Oil Sands Monitoring, committed to a comprehensive monitoring approach with a larger spatial focus aimed at improving the understanding of the long-term cumulative effects of development. Specific to the water quantity and quality portion of the JOSMP implementation plan, the sources, transport, flux and fate of materials and contaminants entering the watersheds from the OS has been identified as a key concern for fish in the region. Current proposals for monitoring efforts include establishing the status of fish population health in the Athabasca main stem and selected tributaries, collecting baseline data for measuring future change and potentially impacted areas against, measuring incidence of physical abnormalities and changes in contaminant concentrations downstream from the OS for effects on the health or functioning of the aquatic ecosystem (JOSMP, 2012).

1.7 Knowledge Gaps, Data Availability and Database Challenges

In all three rivers, the availability and the consistency of the long-term data aggregation created constraints and raised questions related to whether the analysis would be sensitive to the changes in the fish community composition. These differences have arisen from motivations for sample collection, dissimilar techniques and methods used, different sampling seasons and consistency/accuracy in records across all years sampled. For example, it was observed in the collective database that electrofishing and fish fences were the predominant methods used to capture and measure populations since 1995; however, a variety of other methods including fish fences, seine netting, gill netting, trawling, angling, trap nets, minnow traps, dip nets, kick sampling and hoop netting were used to gather fish for samples both pre- and post-development.

Table 1.1 List of fish species, including abbreviation (code) and river found in, (X) indicating its presence.

Code	Common Name	Scientific Name	MKG	STBK	ELLS
ARGR	Arctic Grayling	*Thymallus arcticus*	X	X	
BLTR	Bull Trout	*Salvelinus confluentus*	X	X	
BRST	Brook Stickleback	*Culaea inconstans*	X	X	X
BURB	Burbot	*Lota lota*	X	X	X
CISCO	Lake Cisco	*Coregonus artedii*	X	X	
DLVR	Dolly Varden	*Salvelinus malma*	X		
EMSH	Emerald Shiner	*Notropis atherinoides*	X		
FLCH	Flathead Chub	*Platygobio gracilis*		X	X
FNDC	Finescale Dace	*Chrosomus neogaeus*	X	X	X
FTMN	Fathead Minnow	*Pimephales promelas*	X	X	
GOLD	Goldeye	*Hidon alosoides*	X	X	X
LKCH	Lake Chub	*Couesius plumbeus*	X	X	X
LKWH	Lake Whitefish	*Coregonus clupeaformis*	X	X	X
LNDC	Longnose Dace	*Rhinichthys cataractae*	X	X	X
LNSC	Longnose Sucker	*Catostomus catostomus*	X	X	X
MNWH	Mountain Whitefish	*Prosopium williamsoni*	X	X	X
NRDC	Northern Redbelly Dace	*Phoxinus eos*	X	X	
NRPK	Northern Pike	*Esox Lucius*	X	X	X
NSST	Ninespine Stickleback	*Pungitius pungitius*	X		
PRDC	Pearl dace	*Margariscus margarita*	X	X	X
SLSC	Slimy Sculpin	*Cottus cognatus*	X	X	X
SPSC	Spoonhead Sculpin	*Cottus ricei*	X	X	
SPSH	Spottail Shiner	*Notropis hudsonius*	X	X	X
TRPR	Trout Perch	*Percopsis omiscomaycus*	X	X	X
WALL	Walleye	*Sander vitreus*	X	X	X
WHSC	White Sucker	*Catostomus commersonii*	X	X	X
YLPR	Yellow Perch	*Perca flavescens*	X	X	X
		Total Sp. Rich	**26**	**24**	**18**

A second discrepancy to note is that communities may differ in species composition across seasons. While sampling took place in the spring, summer and fall, the effort to capture measurements of species richness in each season was not done consistently across seasons/years. Moreover, no fish sampling was carried out in the winter months in any of the systems due to logistical constraints related to access and under ice sampling. Finally, with the nature of the data available for each tributary, which as mentioned above, is often missing consecutive years. The scale of the interpretation on a tributary level compared to a regional perspective can therefore be difficult. With such a small temporal scale, and with the nature of our dataset aggregation, some of the applied statistical analyses, while appropriate, may not be sensitive to the changes in these communities.

There are additional challenges with using fish communities as an indicator of change in these systems. River systems are highly dynamic and fish communities therein can change in composition and distribution both seasonally and annually (Habit et al., 2006; Ostrand and Wilde, 2002; Pegg and McClelland, 2004;). For example, finding signals of change within migratory fish populations can be difficult depending on the consistency and effectiveness of monitoring efforts. An accurate account of the presence of migratory fish species is difficult to capture with one or two sampling expeditions throughout the year, and with varying methods used for capture over time. Without a standardized monitoring program, individuals recorded could potentially be representative of a temporary immigration or migration of individuals to or from other areas, rather than a change in migratory guilds.

Despite the limitations of the database, the historical fish data collection was used to examine the specific ecological hypothesis laid out in this chapter (examples of literature with similar limitations see Horwitz, 1978; Poff and Allan, 1995; Taylor et al., 2008;). Due to the nature of the data available, the database and questions asked were structured to provide large-scale ecological patterns best described using the coarse-grain data that was available (Margalef, 1986; Poff and Allan, 1995). Furthermore, the high variability of natural and anthropogenic stressors in these environments, including development, climate change, OS, etc., in combination with synergistic and cumulative interactions of these stressors, can complicate attributions and evaluation of the effects of development-related stressors on aquatic communities. This will involve an evaluation of the causal relationship between development and community response.

1.8 Assessment Endpoint and Hypothesis

Collective disturbances from modifications of the terrestrial and aquatic landscape due to development, both locally and downstream of changes, can result in degradation of stream habitat, impacting the biotic community therein (Argent and Carline, 2004; Scott et al., 1986; Wang et al., 2001). The characterization of long-term trends in community and environmental metrics were used to investigate the effects of landscape development (pre versus post) for the Muskeg, Steepbank and Ells River watershed has affected community composition and taxonomic richness. Below are the assessment endpoints used to determine changes in community composition (Table 1.2) and environmental mechanisms (Table 1.3) that appear important in the ecological maintenance of fish community structure indirectly through habitat suitability are analyzed in this study, and a brief rationale for the inclusion of each.

Table 1.2 Descriptors of fish community composition used in this study to investigate change in community structure and richness between pre- and post-development and the rationale for their inclusion.

Community Descriptor	**Rational**
Species richness (diversity)	Investigations into the development of landscapes and subsequent impacts on stream fish have shown changes to communities reflected in taxonomic summaries, including a decrease in species richness and diversity (see Infante and Allan, 2010; Karr et al., 1986). Through evaluating species richness we test the hypothesis that species richness decreases with an increase in land use through impacts to local habitat availability and suitability.
Functional groups (trophic, reproduction and migratory)	Changes to species richness from landscape development may not manifest as a clear loss of diversity. Employing a functional approach, according to Keddy (1994), is beneficial when investigating community-environment patterns, where functional traits may more readily reflect ecosystem constraints applied to systems over time (Infante and Allan, 2010; Poff, 1997; Poff and Allan, 1995). This approach further allows community composition to be compared where the taxonomic composition may not be consistent across years sampled, as it is in our dataset. Detailed description of individual groups in chapter 2.
Taxonomic composition	Large-scale ecological patterns are best described using coarse-grain data (Margalef, 1968). Data on the presence or absence of species (taxa) in a community across levels of development provides a larger picture or coarser grain of environmental tolerance for ecological communities (Poff and Allan, 1995).

Table 1.3 Environmental variables used in this study to investigate potential environmental patterns in relation to fish species richness.

Environmental Variable	**Rational**
Hydrologic Patterns	Variation in flow regimes can affect fish communities directly by influencing important life history processes (Welcomme et al., 1985) and indirectly through providing unique habitat features for many riverine organisms. For example, critical life history processes including spawning, recruitment and migrations all influenced by flow (Richter et al., 1995). Additionally, flow can shape a river's physical habitat, influence sediment movement, substrate composition (e.g., pebbles or sand), and water chemistry (Poff and Allen, 1995; Poff and Ward, 1997; Richter et al., 1995). Each of these variables can determine which organisms (fish species) will inhabit a certain environment (Stark, 1993).
Community Descriptor	According to Karr (1981), fish communities are ideal indicator organisms, as changes to the composition can be a reflection of ecosystem health and the biotic integrity of rivers (Karr, 1981). See Table 1.2 for species richness rational under community descriptors - used in Chapter 3.

Relationships between spatial and temporal fish community patterns and various disturbances, both natural and anthropogenic, have been discussed extensively in the literature (Argent and Carline, 2004; Infante and Allan, 2010; Pegg and McClelland, 2004; Pegg and Pierce, 2002; Poff, 1997; Poff and Allen, 1995; Scott et al., 1986; Wang et al., 2001). Increasingly, ecologists have moved towards measuring the changes in the functional traits of species, within communities, rather than taxonomic variation to investigate changes or shifts in composition in response to environmental changes (Olden et al., 2010). Employing a functional approach is a strong basis for comparison of community ecology studies of stream fish where community composition is likely to change due to biogeographic, abiotic or biotic constraints applied to systems over time (Keddy, 1994; Poff and Allan, 1995; Karr et al., 1986; Quinn and Adams 1996). This trait-based approach allows for a look at communities with a broad-scale approach to observe any patterns or changes in assemblage structure. From a review of the literature it would be expected that those environments and flow regimes, which have remained constant, or experienced fewer impacts pre- versus post development, would contain a greater proportion of specialist species, which are limited by stable habitat and resource availability. Comparatively, it would be expected that environments or flow regimes in states of fluctuation or change away from the natural state, would be typified by more generalist species that have strategies for exploiting resources from a variety of sources, and more flexible habitat preferences (Poff and Allan, 1995). Southwood (1977) proposed that the strategies employed by a species to survive and successfully reproduce are related to a template created by local habitat characteristics. This important theoretical construct widely used in habitat ecology is useful for assessing assemblages or stream communities (Block and Brennan, 1993; Poff and Allan, 1995).

1.9 Thesis Objective

This thesis addresses the historical perspective, important to understanding the ecology of these rapidly developing systems. This will aid in directing sampling efforts to detect and quantify temporal community or functional changes to biota in these systems (Taylor et al., 2008). The overarching purpose of this research is to examine the potential impacts associated with land use disturbance and OS mining operations on fish community composition patterns in the AOSR. This research uses historical data sets from a variety of monitoring and research programs (as outlined above) from the AOSR to reveal any disturbance-related patterns in fish community composition. Specific questions to be addressed are:

1. *Do patterns in fish community composition vary significantly between levels of development, when change is measured by differences in taxonomic composition or functional groups (trophic, reproduction and migratory)?* (Chapter 2)
2. *Are there features of the hydrograph that can explain the greatest amount of variation in patterns in fish community composition between levels of development, using the Muskeg River as a test watershed?* (Chapter 3)

Using the endpoints/criteria outlined in Tables 2 and 3, patterns will be characterized by examining changes in species richness, taxonomic composition (species presence/absence) and percent composition of functional groups (trophic, reproduction, and migratory guilds), during specified time periods (pre- versus post-development) (Chapter 2), and using hydrological data collected for the Muskeg River, during the same time periods, changes to community will be analyzed through changes or patterns in ecologically relevant flow variables (Chapter 3). Chapter 4 provides a synthesis and discussion of the results, outstanding gaps in knowledge, and recommendations for further fish community monitoring efforts in the AOSR.

A final goal of this thesis is to contribute to a greater understanding of long-term cumulative changes in these systems in order to help guide future monitoring programs in the AOSR.

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Chapter 2. Disturbance Related Patterns In Fish Community Composition In Selected Rivers Of The Lower Athabasca River Systems

2.1 Introduction

Through successive functional traits, including behavioural patterns, life history traits, and morphological characteristics, populations adapt and persist within a range of local environmental conditions (Quinn and Adams, 1996). These functional traits, often inheritable, are representative of characteristics linked to a species fitness or performance, defining a species ecological role, and how they interact with the biotic and abiotic environment (Diaz and Cadbio, 2001; Olden et al., 2010; Quinn and Adams, 1996). Changes to environments, such as land development (mining, agriculture, forestry, etc.), and the resulting impacts on species and communities have been studied, providing in-sight into the relationship between selective pressures (biotic and abiotic) and natural populations (Quinn and Adams, 1996).

Processes that control the structure of freshwater fish communities have been related to changes in the abiotic environment, species interactions, and biogeographical events (Rogriguez and Lewis, 1997). Following Shelford's Law of Tolerance, which postulates that each species thrives at a particular range of environmental values (i.e., its optima), species are distributed along an environmental gradient, where tolerant species are found in more disturbed environments (Shelford, 1931). Each individual within a community is present due to its species' range of tolerance along that gradient (ter Braak and Verdonschot, 1995). Species suitability to an environment can be shifted with land use depending on their tolerance for change. For example, the life cycles of many fish species are timed to avoid or exploit different environmental cues, including changes in base flow that initiate events such as spawning, reproduction or migration (Bunn and Arthington, 2002; Nesler et al., 1988; Poff et al., 1997; Poff and Allan, 1995; Sparks, 1995). Changes to flow, and thus to these largely flow-determined habitat patterns could affect the presence and distribution of species within a system, if the changes in conditions are outside those adapted to by native biota (Poff et al., 1997). This could inevitably affect the species richness of a river or region over time.

Investigations into the development of landscapes and subsequent impacts on stream fish have shown changes to communities are reflected in taxonomic summaries, including a decrease in species richness and relative abundance as described for example by Karr et al. (1986) and Infante and Allan (2010). Increasingly, ecologists have moved towards measuring the changes in the traits of species, within communities, to investigate changes or patterns in community composition in response to environmental changes (Infante and Allan, 2010; Olden et al., 2010). According to Frimpong and Angermeier (2010), the term trait is defined as a measurable morphological, physiological, behavioral, ecological, or life-history expression of an individual organism’s adaptations to its environment that may also be regarded as a property of the taxon or population to which the individual belongs and guild is defined as a group of co-occurring species that exploit a common resource in a similar way. Employing a trait-based approach to observe changes in relative abundances (proportions) of individuals traits is a strong basis for the comparison of community ecology studies of stream fish where community composition is likely to change due to biogeographic, abiotic or biotic constraints applied to systems over time (Karr et al., 1986; Keddy, 1994; Poff and Allan, 1995; Quinn and Adams, 1996). A functional analysis provides a broad-scale approach to observe any patterns or changes in community assemblage structure, where taxonomic composition may not be consistent across years.

2.2 Objectives

This study examines the potential relationships associated with land-use disturbance and oil sands (OS) mining operations on fish community composition patterns by using existing and historical data from the Athabasca Oil Sands Region (AOSR; Alberta, Canada) across a gradient of development (pre versus post). There is no previous research examining changes to fish community composition in the AOSR, in relation to development for the rivers of interest. These will serve as indicators of change within the selected rivers and from a regional perspective. In this chapter the response of fish communities was compared between levels of development (pre versus post), in the Muskeg, Steepbank, and Ells River watershed(s), as well as from a collective regional perspective (pooled data from all three rivers).

Landscape alterations have been demonstrated to impact stream fish through changes in taxonomic composition, including a reduction in species richness. Similar to Infante and Allan (2010), Poff et al., (1997), Poff and Allen (1995), and this research also looked at the response of fish communities by taxonomy (species richness and species present/absent) and traits (food habits (trophic), modes of reproduction, and migration as a life history trait; Table 2.1). Collective disturbances, including AOSR development, within these rivers and watersheds, are predicted to lead to a decrease in species richness, and an increase in generalist species, between levels of development (Table 2.2). These species are characterized by those with strategies for exploiting resources from a variety of sources and who have more flexible habitat preferences (Poff and Allan, 1995). Alternatively, constant environments, or environments with less development pre versus post, would have comparatively less change to community composition. Furthermore, these communities would consist of a greater proportion of species with specialists' strategies, limited by the stable state of their environment, versus generalists, which can withstand a greater degree of disturbance in their environment.

Table 2.1 A summary of the structural and trait-based approaches selected to measure the changes in fish community composition between levels of development in the AOSR.

Approach	Description/ Measurement	
Structural Approach	Species Richness (pre- versus post-development)	
	Relative abundance of taxa (pre- versus post-development)	**Measured Guilds**
Trait-Based Approach	Change in relative abundance of food habits (trophic), pre- versus post-development	Invertivores Carnivore Invertivores-carnivore Omnivore
	Change in relative abundance of modes of reproduction (pre- versus post-development)	Lithophil (NOL) Lithopelagophil (NOLP) Pelagophil (NOP) Phytolithophil (NOPL) Phytophil (NOPH) Guarder Speleophil (GNS) Guarder Anadnophil (GNA)
	Change in relative abundance of migration as a life history trait (spawning or seasonal)	Migratory Source Resident

The specific objectives of this chapter are to:

(1) compile a comprehensive dataset using existing and historic data from the AOSR on fish community data;

2) examine community patterns from a structural approach, including changes in average relative abundance of taxa and species richness between pre- and post-development;

3) examine community patterns from a trait-based approach through changes in average relative abundance of: food habits (trophic), modes of reproduction and migration as a life history trait, during pre- and post-development; and,

4) following Poff and Allan (1995), examine the hypothesis that habitats with more variability (post-development), will support more resource generalists species/functional groups compared to a more stable (pre-development) ecosystem that would be characterized by more specialist species/ functional groups.

Selected structural and trait-based approaches will be used to observe any changes in fish community composition pre- and post-development, and to test if community response to development was a function of the trait possessed by species that made up the communities (Frompong and Angermeier, 2010). The overarching goal is to contribute to the larger understanding of long-term cumulative changes to fish communities in these systems to help guide future monitoring programs in the AOSR.

An understanding of these environmental gradients and how fish communities react to changes within them can help to explain, and possibly aid in predicting further changes in fish community composition. Characterizing long-term and historical trends in fish community patterns, along with consistent monitoring efforts will aid in mitigating impacts and strengthening monitoring efforts in these systems through characterization and identification of important environmental drivers (Gido, et al., 2010; Poff and Allan, 1995). The importance of a historical perspective lies in the ability to detect signals of change that are more chronic or gradual in response to environmental change (Gido et al., 2010). Furthermore, a historical perspective can assist resource managers and ecologists to formulate when, where and for how long to sample a particular system to properly detect and quantify functional changes to the communities therein (Taylor et al., 2008).

There are many uncertainties in research associated with such long-term changes including access to true baseline data that could allow for a distinguished difference between natural and altered community states. Some signals of change within the biotic community can be slow to respond to change from persistent environmental modifications. As a result finding linkages in community structure with environmental changes can be difficult given the temporal availability of data (Gido et al., 2010), the inconsistency of methods and variable sampling protocols.

2.3 Study Sites

Long-term changes in fish community composition were evaluated the Muskeg River, Steepbank River and Ells River watersheds of the lower AOSR (Figure 2.1). All three rivers are tributaries of the Athabasca River; the Muskeg and Steepbank are located to the east, and the Ells to the west.

2.3.1 Muskeg River

The Muskeg River is covers an area of approximately 143,304 ha, 55 km north of Fort McMurray (Figure 2.1). There are currently five operational projects, one limestone quarry, and two projects slated for development within the Muskeg River watershed (Government of Alberta, 2014). Currently operational disturbances affect approximately

14.71% of the land within the watershed (RAMP, 2012), with an expected additional 50-60% landscape disturbance for future activities (Government of Alberta, 2011). The Muskeg River provides important habitat for a total of 26-recorded fish species (see Chapter 1, Table 1.1).

2.3.2 Steepbank River

The Steepbank River watershed covers an area of approximately 136,395 ha (RAMP, 2013) (Figure 2.1). Approximately 3.70% of the land in the watershed has been disturbed, with one operational open-pit mine (Suncore) within the watershed (RAMP, 2012). The Steepbank River provides important habitat for a total of 24-recoded fish species (see Chapter 1, Table 1.1).

2.3.3 Ells River

The Ells River watershed covers an area of approximately 270,944 ha, and is the least developed of the three watersheds, where 1.07% of the land is disturbed by development (RAMP, 2012) (Figure 2.1). One active mine, and two additional mine sites have been proposed within the watershed boundaries (Government of Alberta, 2014). The Ells River is home to 18-recorded fish species (see Chapter 1, Table 1.1).

Site selection was based at the lower reaches of each river where the greatest amount of fish community data was collected for both pre- and post-development. Further descriptions of the fish communities from all three rivers can be found in Table 1.1 and in Chapter 1.

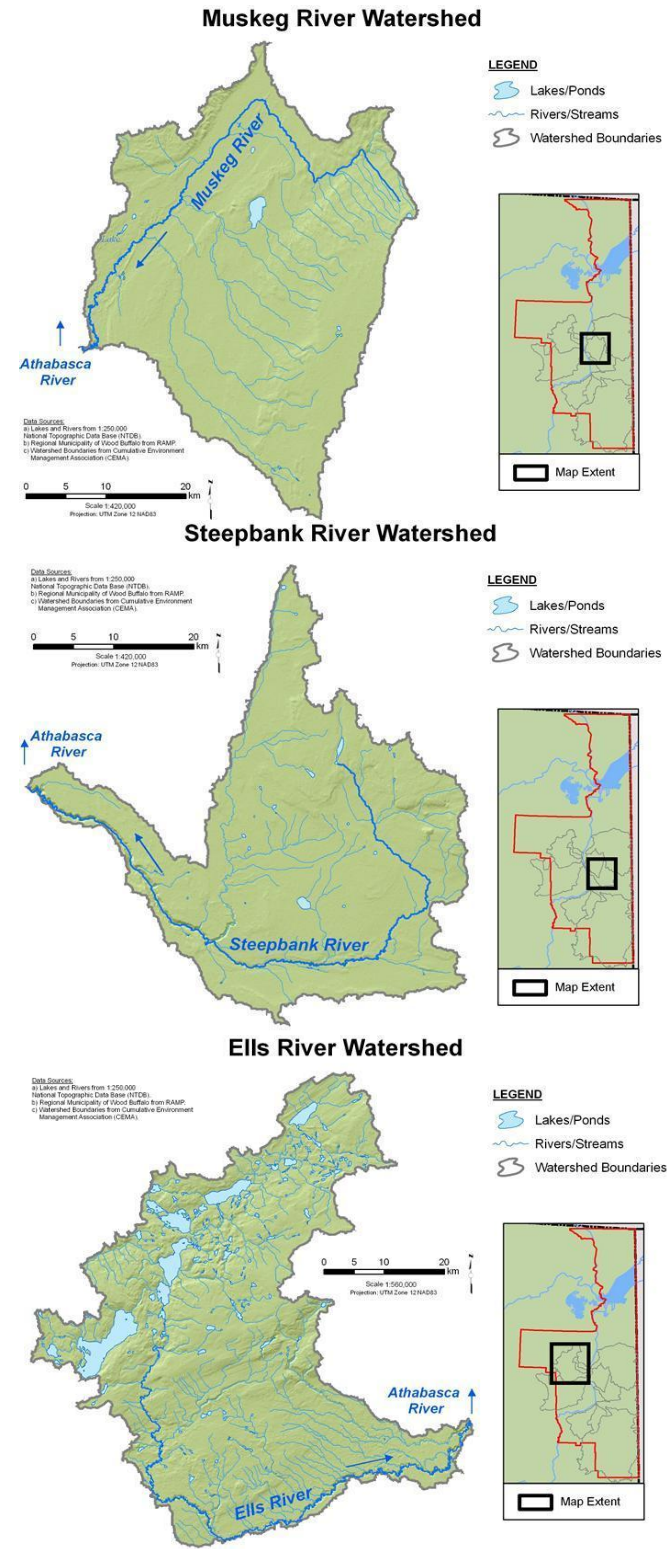

Figure 2.1 Study Sites: The Muskeg River Watershed, The Steepbank River Watershed, and The Ells River Watershed, situated in the province of Alberta, Canada (Source: RAMP, 2013).

2.4 Method

2.4.1 Fish Data

Historical datasets on fish community composition were compiled for the Muskeg, Steepbank and Ells River(s) to perform a retrospective analysis and evaluate temporal trends associated with changes in species richness and fish community structure and function (via. changes in defined taxonomic composition, trophic, reproductive, and migratory guilds). Fish community databases were compiled using current and historical datasets from a variety of monitoring and research programs in the AOSR, implemented in face of development including: Alberta Environment and Sustainable Resource Development (AESRD), the Regional Aquatics Monitoring Program (RAMP), the Alberta Oil Sands Environmental Research Program (AOSERP), and the Fish and Wildlife Management Information System (FWIMS). Fish collections were separated into sampling 'events' by season, method, year, and site sampled. The data collected provided many challenges, including a variety of motives for collecting it, dissimilar methods, techniques, and sampling periods. Given the lack of standardized estimates of abundance and survey methods, comparisons were restricted to measures based on presence/absence data for community composition.

Datasets were separated into levels of development to coincide with those used by Alexander and Chambers (*in press*) for consistency. Alexander and Chambers (*in press)* early and expanded time periods were combined into post-development time periods due to inconsistent and insufficient number of sampling events, within each time period. These periods are associated with no development (pre) and the initiation of development, land clearing, construction, infrastructure use, or potential for disruption to stream habitats (post) (Table 2.2). To assess whether OS mining affected broad scale community composition patterns, data for all three rivers were pooled into a regional database.

Table 2.2 Defined pre and post levels of development for each watershed and regional perspective, as well as respective time periods.

Level of Development	Explanation	Ells	Muskeg	Steepbank	Regional
Pre	No land use.	1978-2001	1979-1995	1976-1993	1976-1995
Post	Land clearing, construction, infrastructure use.	2002-2012	1996-2012	1994-2014	1996-2012

*Steepbank pre development data only extends to 1978, resulting in the 'Regional' classification of pre extending to 1995, not 1993

Presence/ absence data for each fish species was designated into trophic strategies, reproductive strategies and migratory behaviour guilds defined *a priori* (Table 2.2). Totals for each functional group within each guild were tallied and percent compositions calculated for each sampling event.

To explore the changes in guilds within functional groups, between pre- and post-development time frames, species traits were characterized based on trophic strategies, reproductive strategies and migratory behavior. Fish species were classified into trophic strategy categories based on data collected from Scott and Crossman (1998) and FishBase (2014). The Ontario Freshwater Fishes Life History Database (2014) was used to supplement any missing information. Reproductive guilds are based on classifications in Balon (1975) seen in Appendix A. Fish species were designated to one of three categories for migratory behaviours, based on guilds classified in Bond and Machniak (1979) and Machniak and Bond (1979).

2.4.2 Statistical Analysis

Multivariate statistical tests and ordinations were used to observe fish community structure patterns pre- and post-development.

Presence/ absence data for each fish species were collected into defined *a priori* trophic, reproductive and migratory guilds (Table 2.1). The total number of species within each functional group was tallied and percent compositions calculated for each sampling event. Percent composition of guilds was arcsine transformed with the following formula described in Legendre and Legendre (2012) and Roy et al. (2005):

$$\rho` = arcsine \sqrt{\rho}$$

The arcsine transformation was selected to stabilize the variance and normalize proportional data obtained from count data (in our case presence/ absence data for each fish species for each event), where ρ represents the proportion or percent composition data collected for reproduction and trophic guilds.

The three rivers were analyzed separately and collectively for structural and functional changes to the fish communities, with arcsine transformed data. To test the null hypothesis that the percent composition of each group remained consistent pre- versus post-development, the same interaction terms and parameters as the community comparison were used.

2.4.2.1 Species Richness Analysis

To test whether species richness decreases with an increase in land use through impacts to local habitat availability and suitability, changes in community composition were explored by calculating a measure of species richness, or the total number of species present, for both pre- and post-development periods. The mean number of species was calculated from species richness and a one-tailed t-test was used to determine whether a significant difference in the mean number of species present existed between levels of development. Species richness analysis was completed for the Muskeg, Steepbank and Ells River(s) and from a regional perspective (all three rivers).

2.4.2.2 Species Community Structure

a. Non-parametric multi-dimensional scaling ordination

To visually identify any temporal differences that may exist in fish community patterns pre- versus post-development, a nonparametric multi-dimensional scaling ordination (NMDS) was performed on a Bray-Curtis (BC) measure of dissimilarity matrix for trophic, reproductive and migratory guilds, and on binary data for community composition used to through clustering and ordination. The NMDS is an unconstrained ordination technique, which configures a map of the relative association or statistical distance between samples, and thus does not have a numeric axes typical of many other analyses (Clarke and Warwick, 1994). A BC distance measure of dissimilarity was applied to all data sets prior to the analysis. Due to a small dataset, no rare species were excluded.

The output of the NMDS model is a stress value, which is a measure of the distortion of the distance rankings of the similarity matrix among data points. These stress values range from 0 to 1, and they provide an indication of how accurate the community representation is. Stress values >0.3 indicate that the positioning of the points is arbitrarily placed on the graph (Clarke and Warwick, 2001). In the present study, all measured stress values were <0.17, which, according to Clarke and Warwick (1994) indicates an accurate community representation and a good ordination.

b. Analysis of similarity

To test for significant differences in community composition for pre- versus post-development the analysis of similarity (ANOSIM) permutation test on the BC dissimilarity matrix of the guilds was performed in PRIMER with 999 permutations. The BC measure of dissimilarity was used for all biotic data to construct resemblance matrices. Primarily used to compare the variation in species abundances, ANOSIM is distribution-free and does not rely on the assumption of multivariate normality of the data (Warwick and Clarke, 1991). The ANOSIM is a multivariate method, modified version of the Mantel test, and is based on rank correlation between two distance matrices. An ecologically meaningful distance measure (or difference) between samples was determined using the BC similarity coefficient based on relative abundance and composition of taxa between levels of development. Level of development was treated as fixed effects, examining and testing for patterns in community structure between levels of development was the *a priori* objective of the study. Sampling events, as described previously, were treated as random effects as they were used as replicates for the fixed effect 'level of development'.

c. Similarity percentage

A similarity percentage (SIMPER) analysis was conducted to determine which species or guilds were most responsible for any changes in community composition between pre- and post-development. The SIMPER further identified the contribution of each species to differences among functional groups using the procedure outlined in

Clarke and Warwick (2001). The outputs from SIMPER list all species or guilds by their contribution to the BC dissimilarity among samples within a group and to the dissimilarity among samples from different groups. The SIMPER was used to complement subsequent ANOSIM and NMDS analyses.

Each analysis was performed for all three rivers and for a regional perspective (pool of data from all three rivers), for changes in taxonomic composition and percent composition of trophic, reproduction and migratory guilds. These were done with the purpose of testing the null hypothesis that the percent composition of each group remained consistent pre- versus post-development, using the same interaction terms and parameters as the community comparison. For each SIMPER test, a two-sample Kolmogorov-Smirnov test was performed to see if there was significant difference between the average relative abundance between levels of development. Significance was observed if $p<0.05$.

2.4.2.3. Trait-based Analysis

Trait-based Classifications

The health and proportional representation of fish taxa or traits within a community can be used as an indicator of change over time and responses to a broad range of environmental changes (Karr, 1981; McCart and Mayhood, 1980; Munkittrick et al., 2009). For the trait-based analysis, species traits based on food habits (trophic), modes of reproduction, and migration as a life history trait was characterized. Each is described below. Species were divided up into their respective classifications as indicated by the literature, supplemented with information provided by Fishbase: http://www.fishbase.org and Scott and Crossman (1998) (Table 2.3).

a. Food Habits (Trophic Guilds)

Similar to methods used by Catins and Johns (2012), the four guilds representing food habits of the communities are based on the principle food source for each species. These habits are carnivores, omnivores, invertivores, and invertivores-carnivores. According to Karr et al. (1986), land use impacts within a watershed, such as changes to water and habitat quality can cause a shift in fish trophic dynamics or food resources. This shift results in species with more specialized feeding strategies- including carnivores. Some invertivores decrease in abundance as a result of their decreased foraging efficiency (esp. for river adapted species), as their prey base decreases (Renczak, 1995; Taylor et al., 2008). Omnivores display a greater plasticity in their feeding strategies and are more likely to adapt to anthropogenic and ecosystem disturbances including alteration of river food webs and food availability (Hughes et al., 1998; Karr et al., 1986; Karr et al., 1981). The resulting dietary shift has been well documented in the literature and reflects a community with an increase in omnivorous (generalist) species and a decrease in specialist feeding strategies (Brown et al., 2011; Hughes et al., 1998; Karr et al., 1986; Primack, 1988; Taylor et al., 2008).

The inclusion of the carnivore metric evaluates the ability of a system to produce enough fish and large invertebrates to support a population of relatively large predators (Hughes et al. 1998; Table 2.1). Carnivores are an indication of healthy, diverse communities and are predicted to decrease with anthropogenic disturbances as they employ a more specialized foraging strategy compared to the omnivore (Cantin and Johns, 2012).

Invertivores are those species that feed primarily on crustaceans, oligochaetes, mollusks, and insects from the benthos, water column and the surface (Poff and Allan, 1995; Ibanez et al., 2009). A change in abundance or proportional representation of insectivorous fish species may reflect a disturbance that has reduced the production of (benthic) insects (Emery et al., 2003; Cantin and Johns, 2012).

b. Modes of Reproduction

Many different variables can influence the presence or absence of different reproductive guilds within a community. For example, temperature has been demonstrated to influence the availability and quality of spawning substrates (Irz et al., 2007). Other research points to variables such as flow velocity, water depth, peak flow events, and local geomorphology as being important influences on the reproductive strategies of a fish community (Allan, 2004; Bunn and Arthington, 2002; Lamouroux, Poff and Angermeier, 2002; Murchie et al., 2008).

Allan (2004) further summarizes that alterations to flow regimes affect fish down-stream by physically altering the habitat availability and quantity. When water levels change beyond the river's hydrologic regime for a system, physical habitat features utilized by some fish species for spawning have changed and are no longer available (Allan, 2004). Species with reproductive requirements dependent on a natural flow regime would also be negatively impacted by the shift towards more lentic habitat conditions (Taylor et al., 2008; Taylor and Miller, 1990).

The percent composition of litho-obligates (lithophils) in the community represents fish species that need rocky substrates for spawning and are expected to decline if sedimentation increases, reducing availability of cobble, gravel and substrate quality (Bramblett et al., 2005; Karr, 1981; McCormick et al., 2001), and are vulnerable to water fluctuations (Grabowski and Isley, 2007). In this study, the metric is composed of lithophoils, guarder lithophils, lithopelagophils, phytolithophil groups from the Balon classification (Balon, 1975; Table 2.1, 2.2; Appendix A).

Brood Hiders are those that practice spawning below substrate and lack in parental care including bull trout and dolly varden. Under the Balon (1975) classification, both species are considered Brood Hider: Lithophils (Table 2.2). Both species employ a reproductive strategy to hide their offspring in the substrate. This requires a loose gravel substrate into which females dig redds and deposits their eggs (Balon, 1975; Breder and Rosen, 1966). Due to them requiring a cobble substrate to survive, they have been classified as lithiophils.

Bramblett et al. (2005) outlines a more tolerant reproductive guilds defined by taxon who use parental care at nest sites or guarders, not including those that are litho-obligates (Bramblett et al., 2005). This category establishes and defends a territory surrounding or in a nest or cavity (Hitt and Chambers, 2014). These taxa have demonstrated increases in population or the ability to withstand disturbances such as elevated siltation and decreases in dissolved oxygen as they do not require gravel for spawning and are present to aerate their developing eggs (Bramblett et al., 2005). This group includes ariadnophils and speleophils (Table 2.1, 2.2).

Open substratum broadcast spawners are the least specialized of the guilds (Simon, 1998). According to the Balon (1975) classification only one group falls under this category, pelagophils (Table 2.1, 2.2).

c. Migratory Behaviour

Migratory fish are often larger-bodied and valuable components of river ecosystems as they are important for subsistence, recreation and commercial fisheries. Furthermore, as migratory species utilize different parts of a river system annually and seasonally, they are good bio indicators of change in habitat condition, water quality and quantity, throughout a stream network (Karr, 1981, Poff et al., 1997; Welcomme et al., 2006). These species operate in the mouths of the tributaries, as they are important habitat for nursing and resting areas while migrating (Bond and Machniak, 1979). Certain aspects of the flow regime are integral to ecological processes and the timing of rising flows can impact cue reception for migratory and spawning fish species (Bunn and Arthington, 2002). The timing of migrations are assumed adaptations to long term flow regimes, temperature, and other environmental factors and can cue the fish to changes in seasons and abundance of predators and prey (Quinn and Adams, 1996).

Fish species were designated to one of three life history traits related to migration, using previous studies on migration conducted by Bond and Machniak, (1979) and Machniak and Bond, (1979) as guidelines. These papers outline the extent to which the Steepbank and Muskeg watersheds form part of the home range of the various fish assemblages in both rivers. These range guilds were then applied to fish species on the Ells River. Three categories were defined. The first contains a number of species typical of the Athabasca River, which have been designated source species (Table 2.3). These species are rarely found in the middle and upper reaches of the tributaries and are captured close to the mouth of the river. The second group is those that have permanent populations established within the rivers. They have small home ranges and have been designated resident populations (Table 2.3). These include; pearl dace, brook stickleback, slimy sculpin, longnose dace, finescale dace, spoonhead sculpin and lake chub. The third category is those fish species that migrate and utilize the tributaries as a portion of their home range and have been designated migratory (Table 2.3). These include white sucker, longnose sucker, Arctic grayling, mountain whitefish, northern pike, trout-perch, bull trout and walleye.

To analyze trait-based changes (food habits, modes of reproduction and migratory behavior) between pre- and post- development, the same sequence of analysis was used as described in species community structure (methods 2.4.2.2 a-c). This was done for each of the Muskeg, Steepbank, Ells River(s) and from a regional perspective.

Table 2.3 A complete list of all species known to occur within the Muskeg, Steepbank and Ells River watershed(s) and their known trait-based characteristics.

Family	Common Name	Scientific Name	Food habit(s)[1]	Mode of Reproduction[2]	Migratory Trait[3]
Cyprinidae	Brassy minnow	*Hybognathus hankinsoni*	IN	NOPH	SO
	Emerald shiner	*Notropis atherinoides*	IN	NOP	SO
	Fathead minnow	*Pimephales promelas*	OM	GNS	SO
	Finescale dace	*Chrosomus neogaeus*	OM	NOPH	RE
	Flathead chub	*Platygobio gracilis*	IC	NOP	SO
	Lake chub	*Couesius plumbeus*	OM	NOL	RE
	Longnose dace	*Rhinichthys cataractae*	IN	NOL	RE
	Northern redbelly dace	*Phoxinus eos*	OM	NOPH	SO
	Pearl dace	*Margariscus margarita*	OM	NOL	RE
	Spottail shiner	*Notropis hudsonius*	IN	NOLP	SO
Catostomidae	Lognnose sucker	*Catostomus catostomus*	IN	NOL	MI
	White sucker	*Catostomus commersonii*	OM	NOL	MI
Cottidae	Slimy sculpin	*Cottus cognatus*	IN	GNS	RE
	Spoonhead sculpin	*Cottus ricei*	IN	GNS	RE
Esocidae	Northern pike	*Esox lucius*	CA	NOPH	MI
Gasterosteidae	Brook stickleback	*Culaea inconstans*	OM	GNA	RE
	Ninespine stickleback	*Pungitius pungitius*	IN	GNA	SO
Percidae	Yellow perch	*Perca flavescens*	IC	NOPL	SO
	Walleye	*Sander vitreus*	IC	NOLP	MI
Periscopidae	Trout-perch	*Percopsis omiscomaycus*	IC	NOL	MI
Gadidae	Burbot	*Lota lota*	IC	NOL	SO
Salmonidae	Lake whitefish	*Coregonus clupeaformis*	IC	NOLP	SO
	Arctic grayling	*Thymallus arcticus*	IC	NOL	MI
	Bull trout	*Salvelinus confluentus*	IC	NOL	MI
	Mountain whitefish	*Prosopium williamsoni*	IC	NOL	MI
	Lake cisco	*Coregonus artedii*	CA	NOP	SO
	Dolly varden	*Salvelinus malma*	IC	NOL	SO
Hiodontidae	Goldeye	*Hiodon alosoides*	IC	NOP	SO

[1]Food habit (trophic) Classifications: IN = Invertivore, IC = Invertivore Carnivore, CA = Carnivore, OM = Omnivore
[2]Mode of Reproduction Classifications: Phytophil = NOPH, Pelagophil = NOP, Guarder: Speleophil = GNS, Lithophil = NOL, Litho-pelagophil = NOLP, Guarder: Ariadnophil = GNA, Phytoplithophol = NOPL
[3]Migratory Behaviour Classifications: SO = Source, MI = Migratory, RE = Resident

2.5 Results

2.5.1 Species Richness

Total species richness remained constant for the Muskeg River with 20 species present pre- and post-development. Mean species richness was 4.56 species pre and 3.70 species post-development (Figure 2.2A, Table 2.4). One tailed t-test for total number of species n_{pre}=27 and n_{post} =56 showed no significant change ($p>0.05$).

In the Steepbank River, 24 species were present pre- and 19-post development. Mean species richness was 6.62 pre and 4.57 post (Figure 2.2B, Table 2.4). The total number of species present was significantly different between pre- versus post-development ($t= 2.3175$, $p= 0.011$).

A total of 15 species were present in the Ells River pre and 14-post development (Figure 2.2C, Table 2.4). Mean species richness was 4.8 pre and 4.38 post. Total species richness was not significantly different between levels of development ($P>0.05$).

From a regional perspective, using all rivers combined, a total of 25 species were present pre-development and 21-post development (Figure 2.2D, Table 2.4). Mean species richness pre- and post-development was 6.09 and 3.82 respectively. Total species richness showed a significant difference between levels of development ($t=4.852$, $p<0.0001$).

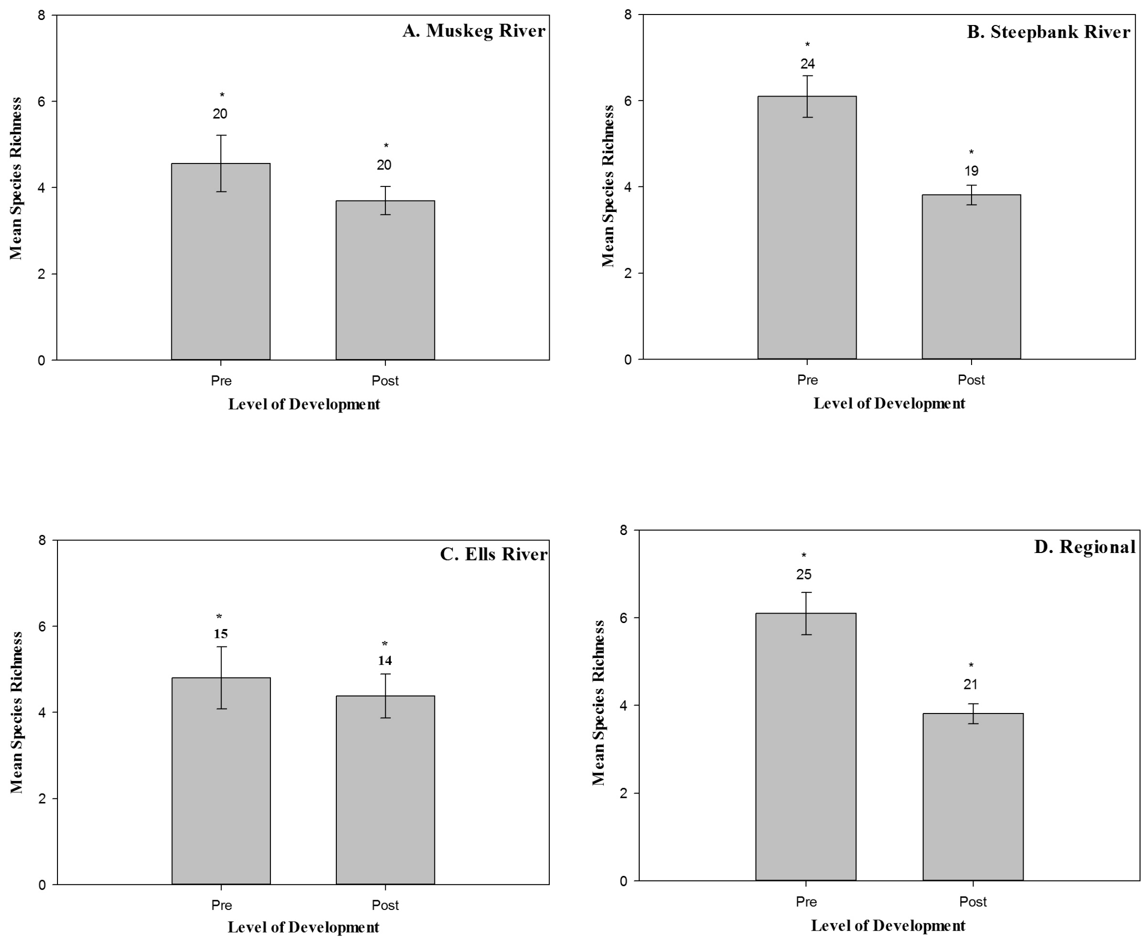

Figure 2.2 A. Muskeg River. Mean species richness (+/- SE) between levels of development and collective species richness (*). One-tailed t-test for total number of species pre (n=27) and post (n=56), t=1.306, p=0.098. Result is *not* significant ($p < 0.05$). B. Steepbank River. Mean species richness (+/- SE) between levels of development and collective species richness (*). One-tailed t-test for total number of species pre (counted for each sampling 'unit') and post, t= 2.318, p= 0.011. The result is significant ($p < 0.05$). C. Ells River. Mean species richness (+/- SE) between levels of development and collective species richness (*). One-tailed t-test for total number of species pre (n=15) and post (n=16), t= 0.488, p= 0.315. The result is *not* significant at $p > 0.05$. D. Regional perspective. Mean species richness (+/- SE) between levels of development and collective species richness (*) for all rivers. One-tailed t-test for total number of species pre (n=71) and post (n=130), t = 4.852, $p < 0.0001$. The result is significant at $p < 0.05$.

Table 2.4 Species richness (sp.rich) (total), sp. rich (mean), standard error (SE), and range of richness for pre- and post-development in the Muskeg, Steepbank and Ells River(s).

Level of development	Sp.rich (collective)	Sp.rich (mean)	SE (+/-)	Sp.rich range
Muskeg River				
Pre	20	4.56	0.65	1-15
Post	20	3.70	0.33	1-12
Steepbank River				
Pre	24	6.62	0.86	1-17
Post	19	4.57	0.47	1-12
Ells River				
Pre	15	4.8	0.72	1-10
Post	14	4.38	0.51	1-7
Regional				
Pre	25	6.10	0.481	1-17
Post	21	3.82	0.229	1-12

2.5.2 Changes in Species Community Structure

NMDS

In analysing the variation between levels of development, the dispersal in the NMDS ordination space suggests that community taxonomic composition is similar between pre- and post-development for the Muskeg, Steepbank and Ells River(s), and regionally (Figure 2.3). The two-dimensional NMDS ordination suitably represented the data (2d stress <0.3).

ANOSIM

There were no significant differences in taxonomic composition for any of the rivers between levels of development (ANOSIM global $R<0.25$; Table 2.5).

SIMPER

In the Muskeg River, eight species contributed over 70% of the variation in taxonomic community composition between pre- and post-development (Figure 2.4). Arctic grayling had a decline in average relative abundance from 0.74 (pre) to 0.16 (post) development, contributing to 13.2% of the change in taxonomic community composition (Figure 2.4). Additionally white sucker, northern pike, and pearl dace also declined between pre/post time periods. Lake chub increased in relative abundance from 0.11(pre) to 0.36 (post) contributing to 7.1% of the change in community composition. Slimy sculpin and walleye also increased in relative abundance.

In the Steepbank River, shifts in relative abundances of Arctic grayling, slimy sculpin, longnose dace, longnose sucker, lake chub, white sucker, pearl dace, trout-perch and spoonhead sculpin contributed 70% of the variation in the taxonomic community composition between pre- and post-development (Figure 2.5). Similar to the Muskeg: Arctic grayling, pearl dace and white suckers decreased in average relative abundance. The Arctic grayling accounted for the greatest change in relative abundance, decreasing from 0.62 to 0.36, and contributing 9.6% of the change in community composition. Longnose dace, longnose suckers, and lake chub also decreased. Slimy sculpin increased as they did in the Muskeg, while spoonhead sculpin increased in relative abundance by 0.07 to 0.34; however, they only contributed to 5.7% of the variation in taxonomic community composition. The average relative abundance of trout-perch remained the same between pre- and post-development. Northern pike, was the third greatest contributor to variation in the Muskeg, and did not contribute to the change observed in the Steepbank.

Unlike the Muskeg and Steepbank, no Arctic grayling were found in the Ells River. Similar to the Steepbank, however, lake chub and longnose suckers both decreased in average relative abundance, and similar to the Muskeg the northern pike decreased (Figure 2.6). Walleye, which increased in the Muskeg was only present in the pre level of development, and was not found in samples from post-development. Goldeye, only found in the Ells, were not found in samples collected post development. Northern pike results were similar to the Muskeg, decreasing in average relative abundance between pre- and post-development; however, they contributed much less to the overall variation in the community composition in the Ells compared to the Muskeg (5.26% versus 34.81% respectively). Longnose dace had the greatest increase in average relative abundance from 0.47 pre to 0.81-post development, and contributed 10.59% to the overall variation in community composition. Trout-perch increased in average relative abundance, where it stayed consistent in the Steepbank. White sucker and pearl dace also increased in their average relative abundance post development.

From a regional perspective, declines in average relative abundance were observed in Arctic grayling, white sucker, longnose dace, longnose sucker, pearl dace and northern pike and an increase in slimy sculpin followed a similar patter to the Muskeg and Steepbank communities (Figure 2.7). Average relative abundance of Lake Chub and trout-perch remained consistent between levels of development.

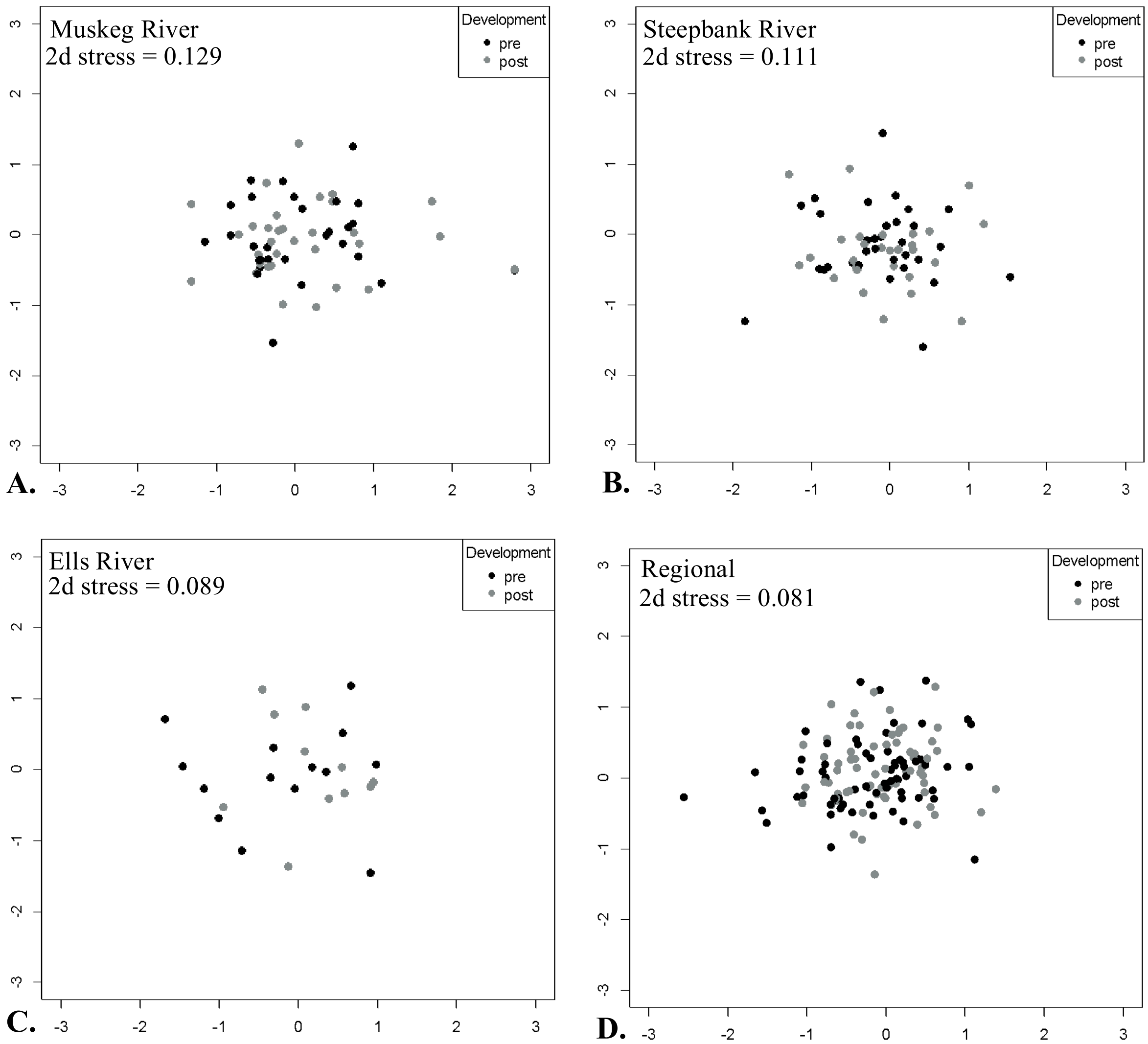

Figure 2.3 Two-dimensional plot of the fish community taxonomic composition in each river pre versus post development, showing the ordination resulting from the NMDS based on a similarity Bray-Curtis coefficient. **A**. Muskeg River Pre n= 27, post n=56. **B.** Steepbank River Pre n= 29, post n=58. **C.** Ells River Pre n= 15, post n=16. **D.** Regional (all three rivers), Pre n= 71, post n= 130.

Table 2.5 Result from the multivariate, two-way ANOSIM for Species Community Structure (presence/absence; percent composition with arcsine transformation), between levels of development for each of the rivers and for a collective regional perspective. R values below 0.25 are considered to be indistinguishable based on their species or guild composition (Clarke and Gorley, 2001).

River(s)	Global R*	P**
Species Community Structure		
Muskeg River	0.013	0.331
Steepbank River	0.016	0.32
Ells River	0.146	0.005
Regional	-0.014	0.734

*The global *R*-value is the comparative measure of the degree of separation of the groups used (Clarke and Warwick, 1994). High R-value (close to 1) indicates dissimilarity between groups.
**p = the significance level of the global R-value

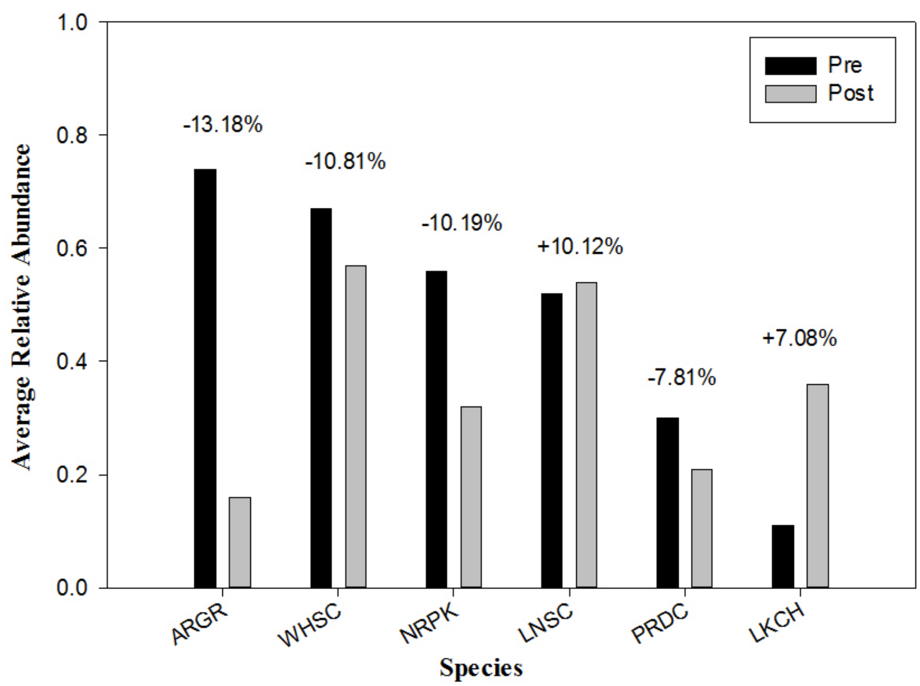

Figure 2.4 Average relative abundance of species found in the Muskeg River that represent approximately 70% of the variation in community composition between pre- and post-development. SIMPER identified the percentage of each species contribution to the observed

pattern of dissimilarity (Bray-Curtis). * Species codes: ARGR = Arctic Grayling, WHSC = White Sucker, NRPK = Northern Pike, LNSC = Longnose Sucker, PRDC = Pearl Dace, LKCH = Lake Chub.

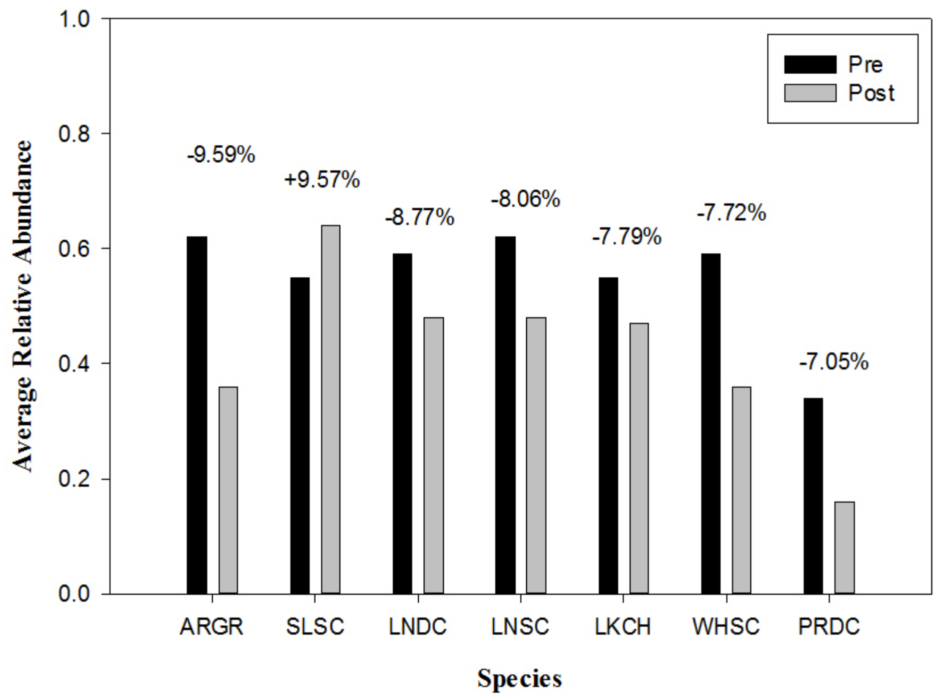

Figure 2.5 Average relative abundance of species found in the Steepbank River that represent approximately 70% of the variation in community composition between pre- and post-development. SIMPER identified the percentage of each species contribution to the observed pattern of dissimilarity (Bray-Curtis). *Species codes: ARGR = Arctic Grayling, SLSC = Slimy Sculpin, LNDC = Longnose Dace, LNSC = Longnose Sucker, LKCH = Lake Chub, WHSC = White Sucker, PRDC = Pearl Dace

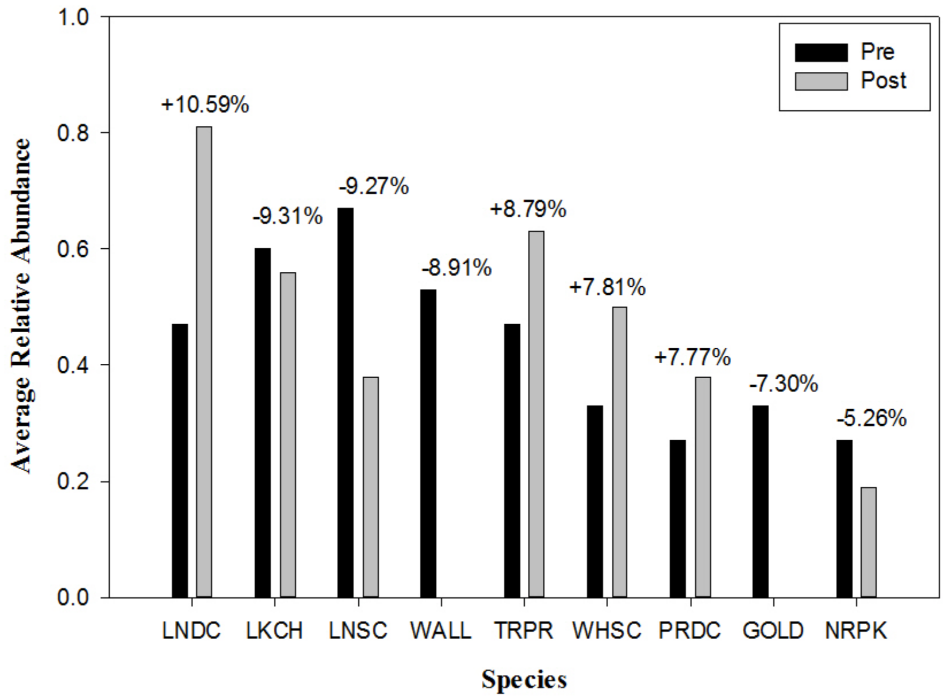

Figure 2.6 Average relative abundance of species found in the Ells River that represents approximately 70% of the variation in community composition between pre- and post-development. SIMPER identified the percentage of each species contribution to the observed pattern of dissimilarity (Bray-Curtis). *Species codes: LNDC = Longnose Dace, LNSC = Longnose Sucker, LKCH = Lake Chub, WALL = Walleye, TRPR = Trout Perch, WHSC = White Sucker, PRDC = Pearl Dace, GOLD = Goldeye, NRPK = Northern Pike.

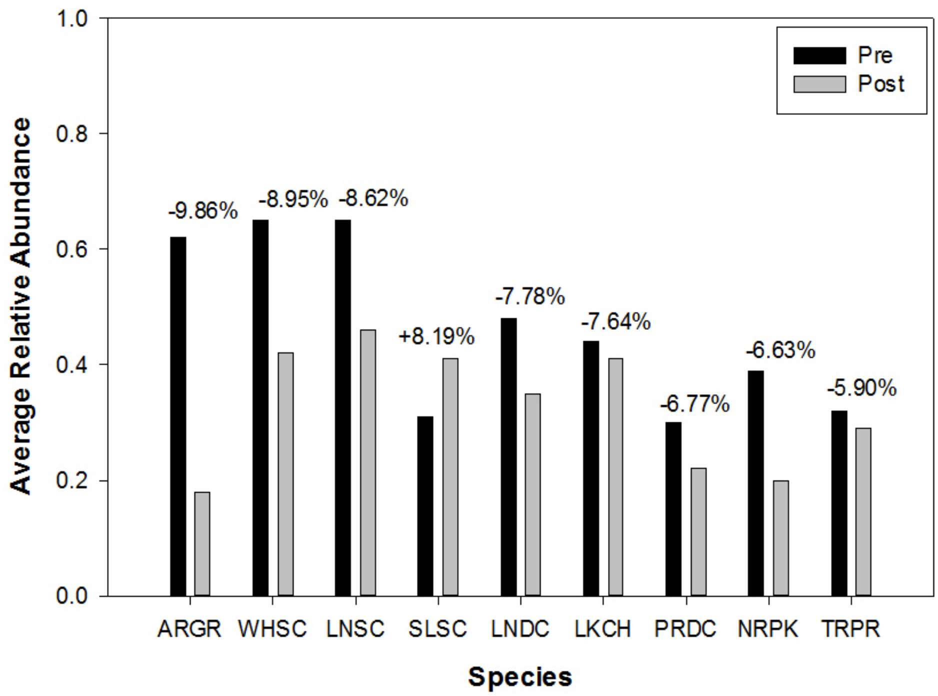

Figure 2.7 Average relative abundance of species found in all three rivers collectively that represents approximately 70% of the variation in regional community composition between pre- and post-development. SIMPER identified the percentage of each species contribution to the observed pattern of dissimilarity (Bray-Curtis). *Species codes: ARGR = Arctic Grayling, WHSC = White Sucker, LNSC = Longnose Sucker, SLSC = Slimy Sculpin, LNDC = Longnose Dace, LKCH = Lake Chub, PRDC = Pearl Dace, NRPK = Northern Pike, TRPR = Trout Perch.

2.5.3 Trait-Based Analysis

2.5.3.1 Trophic Guilds

NMDS

NMDS stress values for all ordinations were < 0.3 suggesting the two-dimensional solution suitably represented the data. The NMDS for percent composition of trophic guilds in the Muskeg, Steepbank and Ells River(s) reveal little observable differences in trophic guild composition, between levels of development (Figure 2.8)**.**

ANSOSIM

No clear difference was observed in percent composition of taxonomic guilds, for any river or regionally, between levels of development (ANOSIM global R <0.25; Table 2.6).

SIMPER

All four trophic functional groups were present in the each of the rivers pre- and post-development. The contribution to the variation in trophic guilds present between pre- and post-development in the Muskeg was greatest from the OM guild (28.9%) followed by IC (28.6%), IN (25.1%), and OM (17.4%) (Figure 2.9). There were no significant differences in average relative abundance between pre- and post-development. In contrast, in the Steepbank, the CA guild had the greatest contribution to variation (36.7%), followed by IN (24.6%), IC (22.1%), and OM (16.1%) (Figure 2.10). The Ells river demonstrated a different pattern in contribution to the variation in community composition between pre- and post-development leading with IC (33.3%), followed by OM (29.9%), IN (20.8%) and CA (16.1%) (Figure 2.11). Changes to average relative abundance of guilds between pre- and post-development for all three rivers were not statistically significant.

Regionally, carnivores demonstrated the greatest change in average relative abundance (27.3%), followed closely by IC (26.4%), OM (23.6%) and finally IN (22.7%) (Figure 2.12). All trophic guilds showed an increase in average relative abundance post development with the exception of IC.

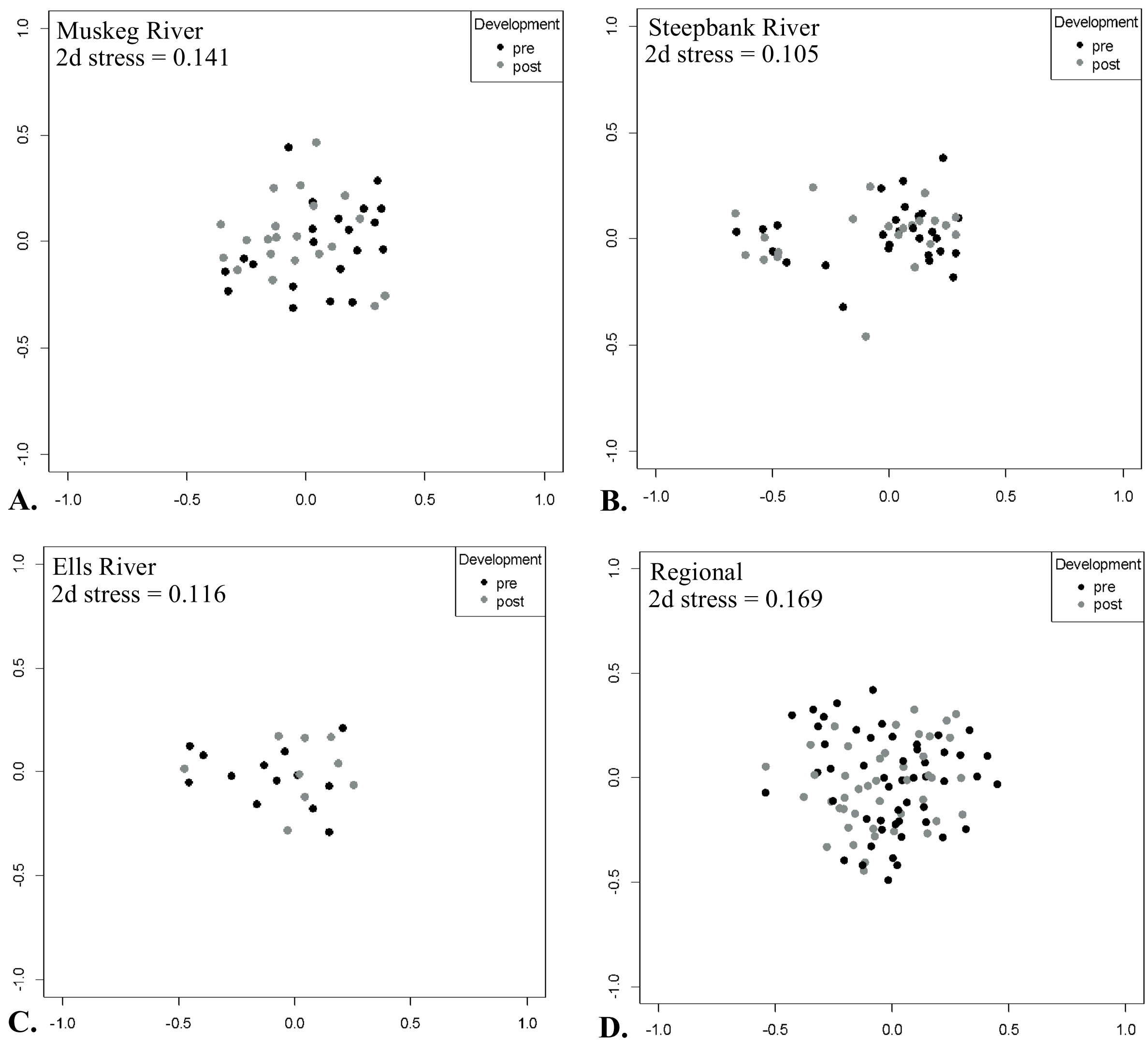

Figure 2.8 Two-dimensional plots for percent composition of trophic guilds (with arcsine transformation) for each river, pre versus post development, showing the ordination resulting from the NMDS based on a similarity Bray-Curtis coefficient. **A.** Muskeg River Pre n= 27, post n=56. **B**. Steepbank River Pre n= 29, post n=58. **C.** Ells River Pre n= 15, post n=16. **D.** Regional (all three rivers), Pre n= 71, post n= 130.

Table 2.6 Result from the multivariate, two-way ANOSIM for trophic guilds (percent composition with arcsine transformation), between levels of development for each of the rivers and for a collective regional perspective. R values below 0.25 are considered to be indistinguishable based on their species or guild composition (Clarke and Gorley, 2001).

River(s)	Global R*	P**
Trophic Guilds		
Muskeg River	-0.056	0.942
Steepbank River	0.119	0.008
Ells River	0.064	0.082
Regional	-0.002	0.512

*The global *R*-value is the comparative measure of the degree of separation of the groups used (Clarke and Warwick, 1994). High R-value (close to 1) indicates dissimilarity between groups.
**p = the significance level of the global R-value

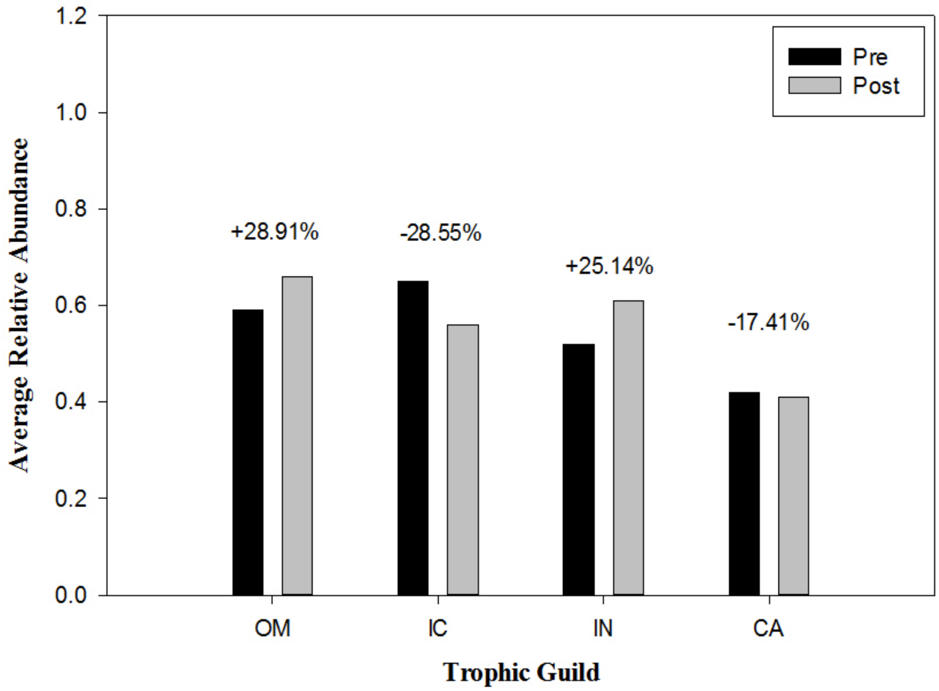

Figure 2.9 Average relative abundance of trophic guilds found in the Muskeg River that represent the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *OM = Omnivores, IC = Invertivore-carnivore, IN = Invertivore, CA = Carnivore.

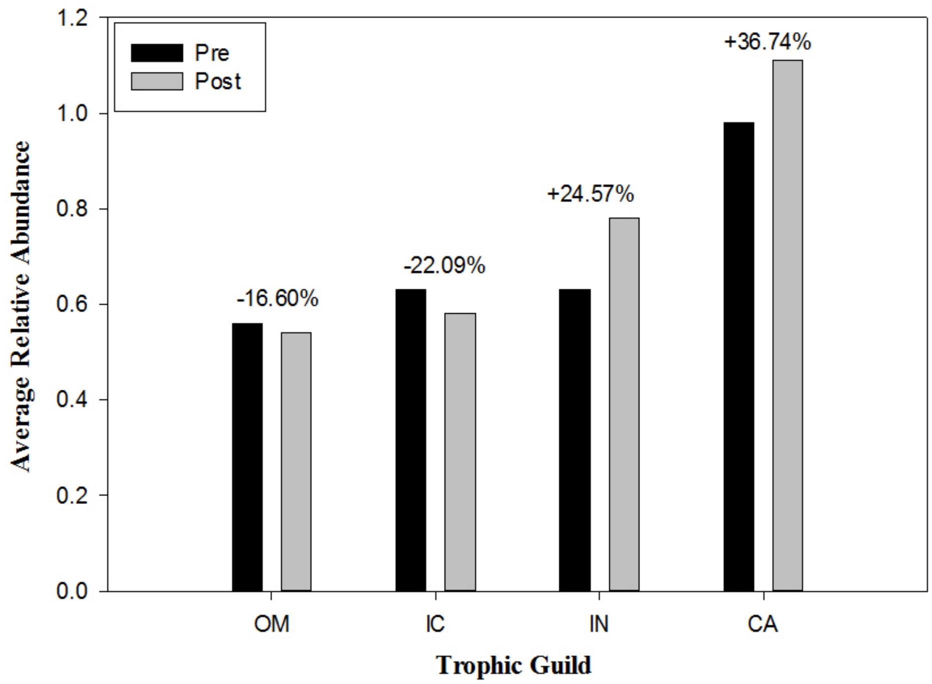

Figure 2.10 Average relative abundance of trophic guilds found in the Steepbank River that represent 50-60% of the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *OM = Omnivores, IC = Invertivore-carnivore, IN = Invertivore, CA = Carnivore.

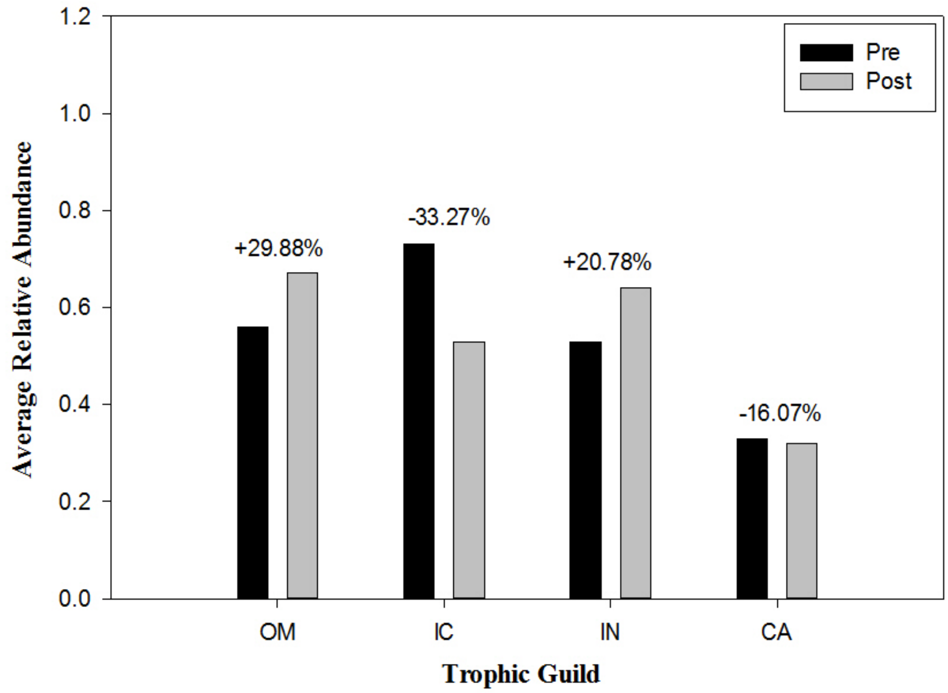

Figure 2.11 Average relative abundance of trophic guilds found in the Ells River that represents the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *OM = Omnivores, IC = Invertivore-carnivore, IN = Invertivore, CA = Carnivore.

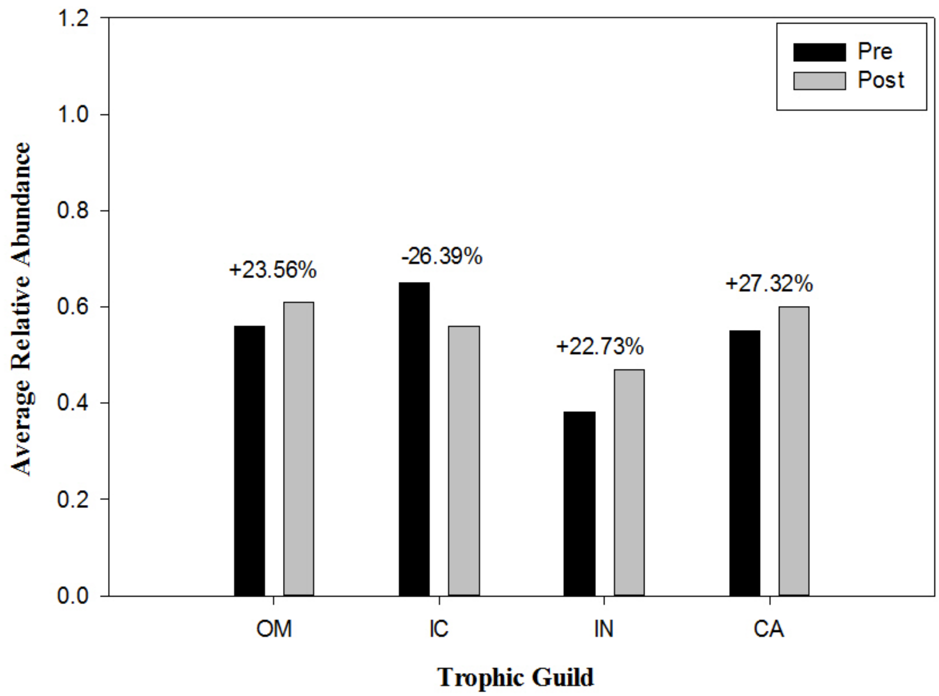

Figure 2.12 Average relative abundance of trophic guilds found in all three rivers collectively that represents the regional variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *OM = Omnivores, IC = Invertivore-carnivore, IN = Invertivore, CA = Carnivore.

2.5.3.2 Reproductive Guilds

NMDS

NMDS stress values for all ordinations were < 0.3 suggesting the two-dimensional solution suitably represented the data. The dispersal in the NMDS ordination space indicates there is no difference in the percent composition of reproductive guilds despite levels of development for the Muskeg, Steepbank and Ells River(s), and regionally (Figure 2.13).

ANOSIM

There was no significant differences in reproductive guild composition for any of the rivers, or regionally, between levels of development (ANOSIM global $R > 0.25$; Table 2.5).

SIMPER

In the Muskeg River, eight reproductive guilds were represented in the community composition: NOL, GNS, NOPH, NOLP, GNA, NOPS, NOP, and NOPL (Figure 2.14). NOL represented the greatest percent contribution to variation in reproductive guilds between pre- and post-development , the overall variability is consistently low for all guilds ranging from 9.1% to 15.0%. The two-sample Kolmogorov-Smirnov test revealed no significant differences between average relative abundance for any of the functional groups between levels of development ($p>0.05$).

In the Steepbank River, seven reproductive guilds were represented (Figure 2.15). Similar to the Muskeg, NOL, GNS, NOLP, GNA, NOPS, and NOPL were present; however, NOPH and NOP functional groups were not. GNL was found only in the Steepbank but not the Muskeg. While most of the guilds remained consistent in the average relative abundance between pre- and post-development, the range of percent contribution to the variation in community composition was larger than in the Muskeg, ranging from 9.9% to 20.2%. The two-sample Kolmogorov-Smirnov test revealed percent composition of GNS increased significantly in average relative abundance between pre- and post-development ($p<0.05$). No significant differences were found in average relative abundance in any other reproductive guilds ($p>0.05$).

There were five functional groups found in the Ells River (Figure 2.16). Similar to the Muskeg and Steepbank, NOL, GNS, NOPL and NOLP were present. Like the Muskeg, no members of the GNL or NOP guilds were present and NOPH were present. Additionally, the GNA and NOPS guilds were not observed. The average relative abundance between pre- and post-development was consistent, and the percent contribution of each guild was larger than in either of the other rivers, ranging from 9.5% to 27.0%. Percent composition of NOPL had significant decrease in average relative abundance between pre- and post-development ($p<0.05$). There were no significant differences between average relative abundance for any other functional groups pre- and post-development.

Regionally, with all reproductive guilds present, NOL showed the largest average relative abundance pre- and post-development and the greatest contribution to the observed pattern of dissimilarity. No significant changes in average relative abundances were observed (Figure 2.17).

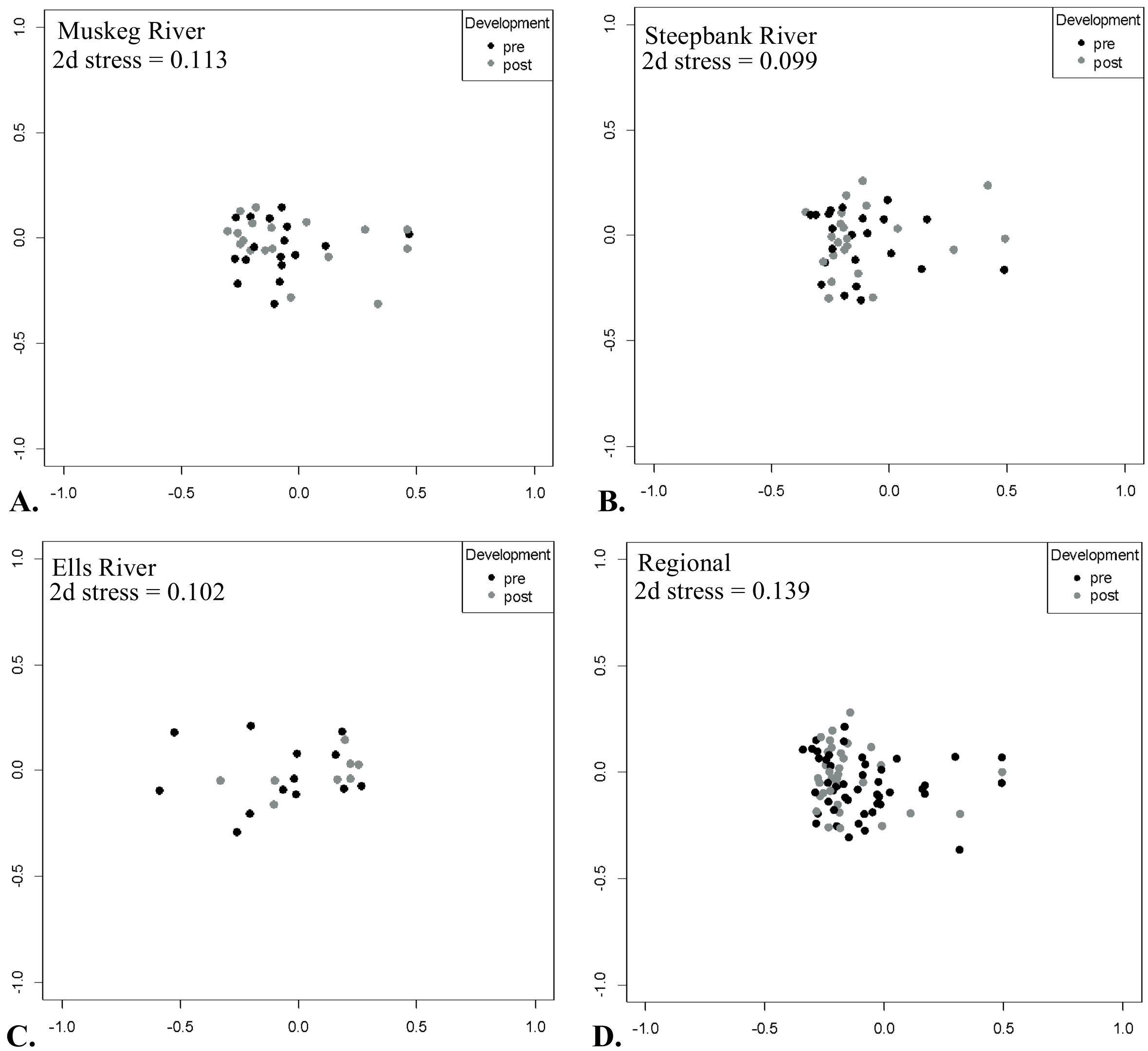

Figure 2.13 Two-dimensional plots for percent composition of reproductive guilds (with arcsine transformation) for each river, pre versus post development, showing the ordination resulting from the NMDS based on a similarity Bray-Curtis coefficient. **A.** Muskeg River Pre n = 27, post n = 56. **B**. Steepbank River Pre n = 29, post n = 58. **C.** Ells River Pre n = 15, post n = 16. **D.** Regional (all three rivers), Pre n = 71, post n = 130.

Table 2.7 Result from the multivariate, two-way ANOSIM for reproduction guilds (percent composition with arcsine transformation), between levels of development for each of the rivers and for a collective regional perspective. R values below 0.25 are considered to be indistinguishable based on their species or guild composition (Clarke and Gorley, 2001).

River(s)	Global R*	P**
Reproduction Guilds		
Muskeg River	-0.017	0.661
Steepbank River	0.008	0.285
Ells River	0.195	0.003
Regional	0.027	0.067

*The global *R*-value is the comparative measure of the degree of separation of the groups used (Clarke and Warwick, 1994). High R-value (close to 1) indicates dissimilarity between groups.
**p = the significance level of the global R-value

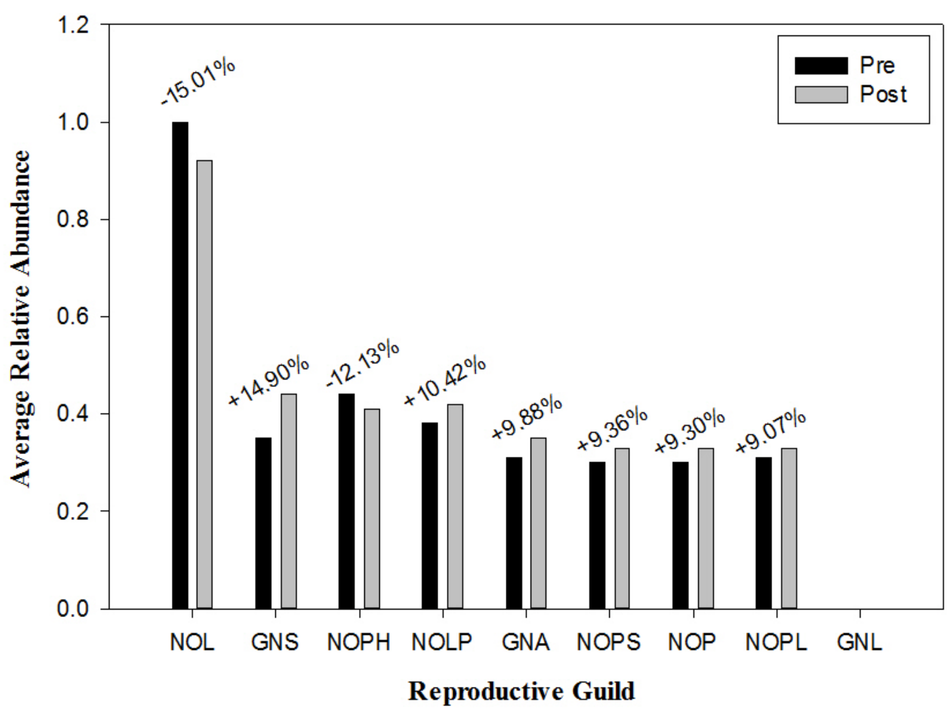

Figure 2.14 Average relative abundance of reproductive guilds found in the Muskeg River that represent the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *NOL = non-guarder open substratum lithophils, GNS = guarder speleophil, NOPH = non-guarder open substratum phytolithophil, NOLP = non-guarder open substratum litho-pelagophil,

GNA = guarder ariadnophil, NOPS = non-guarder open substratum litho-pelagophil, NOP = non-guarder open substratum pelagophil, NOPL = non-guarder open substratum phytolithophil, GNL = brood hider lithophil .

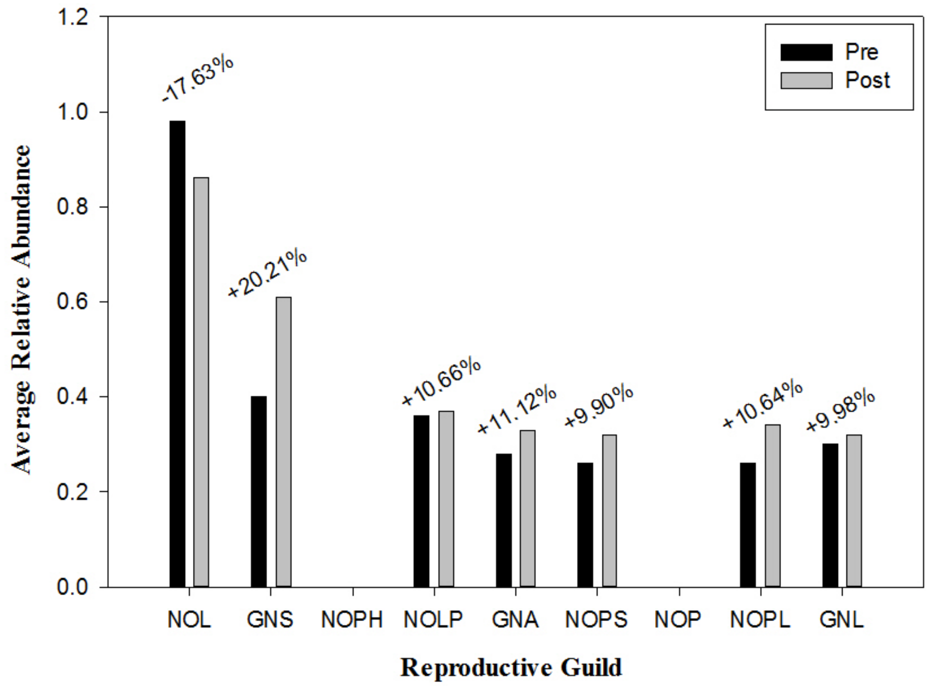

Figure 2.15 Average relative abundance of reproductive guilds found in the Steepbank River that represent the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *NOL = non-guarder open substratum lithophils, GNS = guarder speleophil, NOPH = non-guarder open substratum phytolithophil, NOLP = non-guarder open substratum litho-pelagophil, GNA = guarder ariadnophil, NOPS = non-guarder open substratum litho-pelagophil, NOP = non-guarder open substratum pelagophil, NOPL = non-guarder open substratum phytolithophil, GNL = brood hider lithophil.

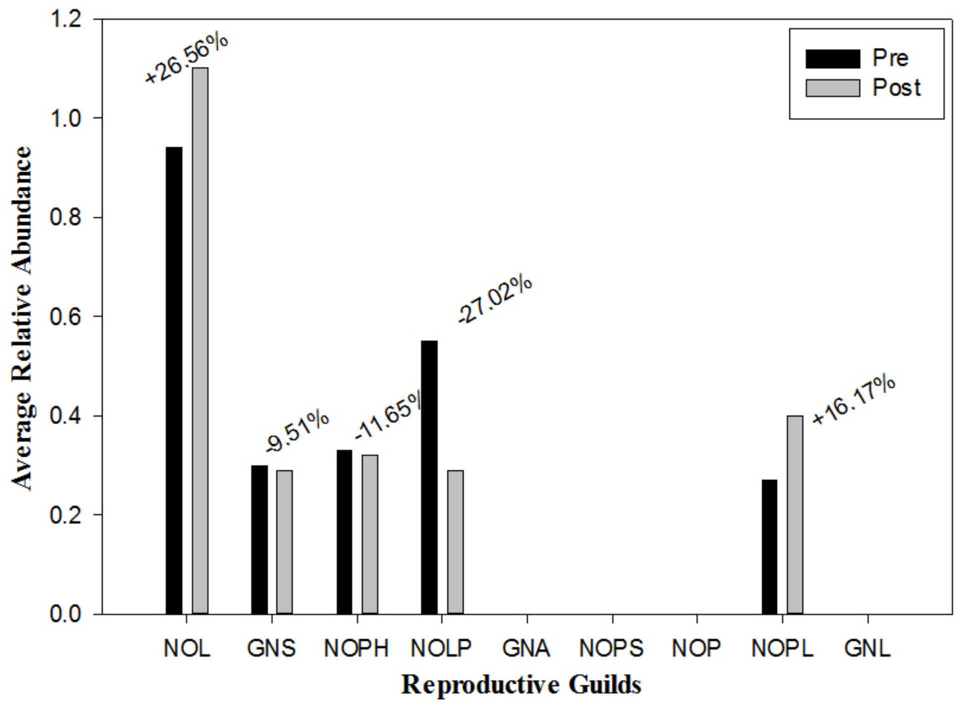

Figure 2.16 Average relative abundance of reproductive guilds found in the Ells River that represent the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *NOL = non-guarder open substratum lithophils, GNS = guarder speleophil, NOPH = non-guarder open substratum phytolithophil, NOLP = non-guarder open substratum litho-pelagophil, GNA = guarder ariadnophil, NOPS = non-guarder open substratum litho-pelagophil, NOP = non-guarder open substratum pelagophil, NOPL = non-guarder open substratum phytolithophil, GNL = brood hider lithophil.

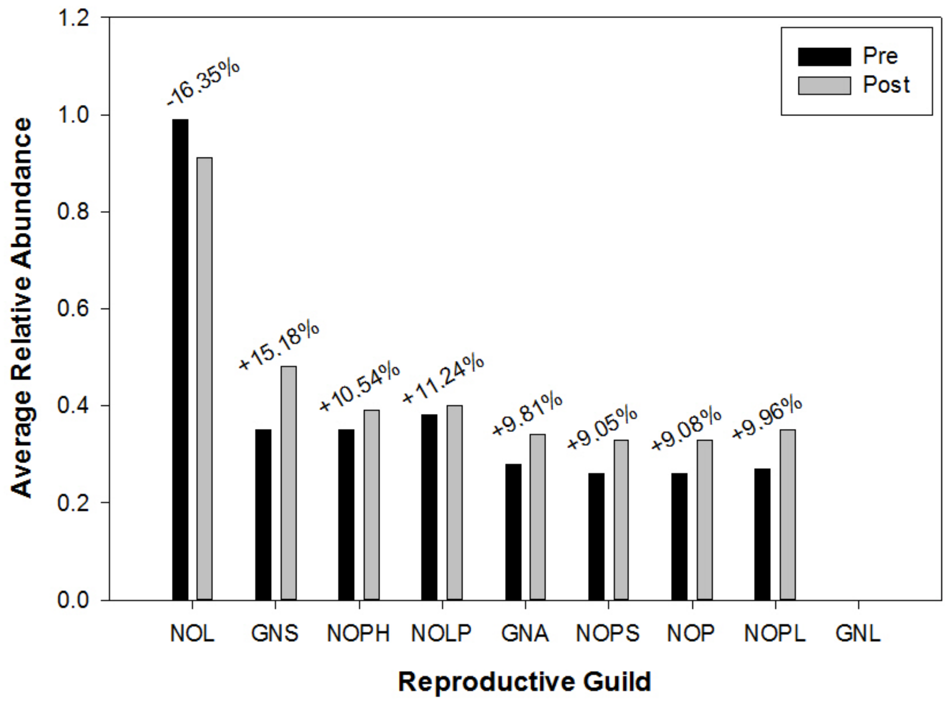

Figure 2.17 Average relative abundance of reproductive guilds found in all three rivers collectively that represent the regional variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis). *NOL = non-guarder open substratum lithophils, GNS = guarder speleophil, NOPH = non-guarder open substratum phytolithophil, NOLP = non-guarder open substratum litho-pelagophil, GNA = guarder ariadnophil, NOPS = non-guarder open substratum litho-pelagophil, NOP = non-guarder open substratum pelagophil, NOPL = non-guarder open substratum phytolithophil, GNL = brood hider lithophil.

2.5.3.3 Migratory Behaviour

NMDS

NMDS stress values for all ordinations were < 0.3 suggesting the two-dimensional solution suitably represented the data. The dispersal in the NMDS ordination space indicates the percent composition of migratory guilds in the Muskeg, Steepbank and Ells River(s) suggest that migratory guild composition is not affected by development (Figure 2.18).

ANOSIM

No significance was observed in community composition, for any of the rivers, or regionally, between levels of development (ANOSIM global R <0.25; Table 2.8).

SIMPER

In the Muskeg River, resident species contributed the greatest amount of variation between pre- and post-development (42.5%), followed by migratory (40.5%) and source (17.1%). Changes to the average relative abundance each range guild in the Muskeg River was not significant between pre- and post-development ($p>0.05$) (Figure 2.19). In the Steepbank River, variation followed the same pattern with resident species contributing 43.4% of the change followed by migratory (34.8%) and source populations (21.8%). The resident species had a significant change in average relative abundance between pre- and post-development ($p<0.05$), while migratory and source guilds did not (Figure 2.20). Resident species on the Ells contributed the greatest amount of variation between pre- and post-development (36.1%), followed by migratory (33.7%) and source (30.3%). Observations from the Ells showed that resident species had significant decrease in average relative abundance between pre- and post-development ($p<0.05$), while migratory and source groups showed no significant change (Figure 2.21). Regionally, no significant changes were observed in the average relative abundance between levels of development (p>0.05; Figure 2.22).

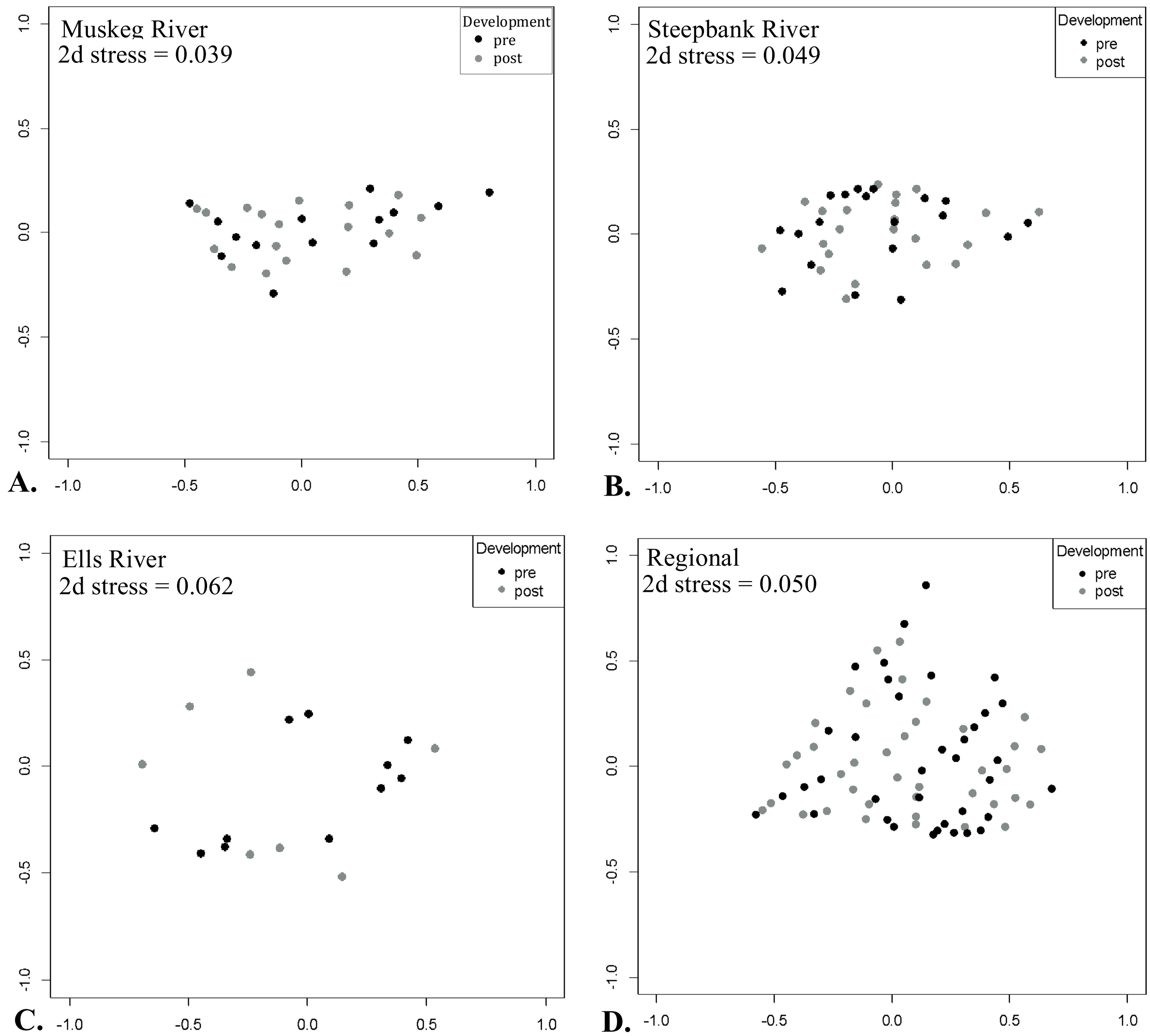

Figure 2.18 Two-dimensional plot for percent composition of migration guilds (with arcsine transformation) for each river, pre versus post development, showing the ordination resulting from the NMDS based on a similarity Bray-Curtis coefficient. **A.** Muskeg River Pre n = 27, post n = 56. **B**. Steepbank River Pre n = 29, post n = 58. **C.** Ells River Pre n = 15, post n = 16. **D.** Regional (all three rivers), Pre n = 71, post n = 130.

Table 2.8 Result from the multivariate, two-way ANOSIM for migration guilds (percent composition with arcsine transformation), between levels of development for each of the rivers and for a collective regional perspective. R values below 0.25 are considered to be indistinguishable based on their species or guild composition (Clarke and Gorley, 2001).

River(s)	Global R*	P**
Migration Guilds		
Muskeg River	-0.026	0.781
Steepbank River	0.076	0.014
Ells River	0.048	0.125
Regional	0.03	0.065

*The global *R*-value is the comparative measure of the degree of separation of the groups used (Clarke and Warwick, 1994). High R-value (close to 1) indicates dissimilarity between groups.
**p = the significance level of the global R-value

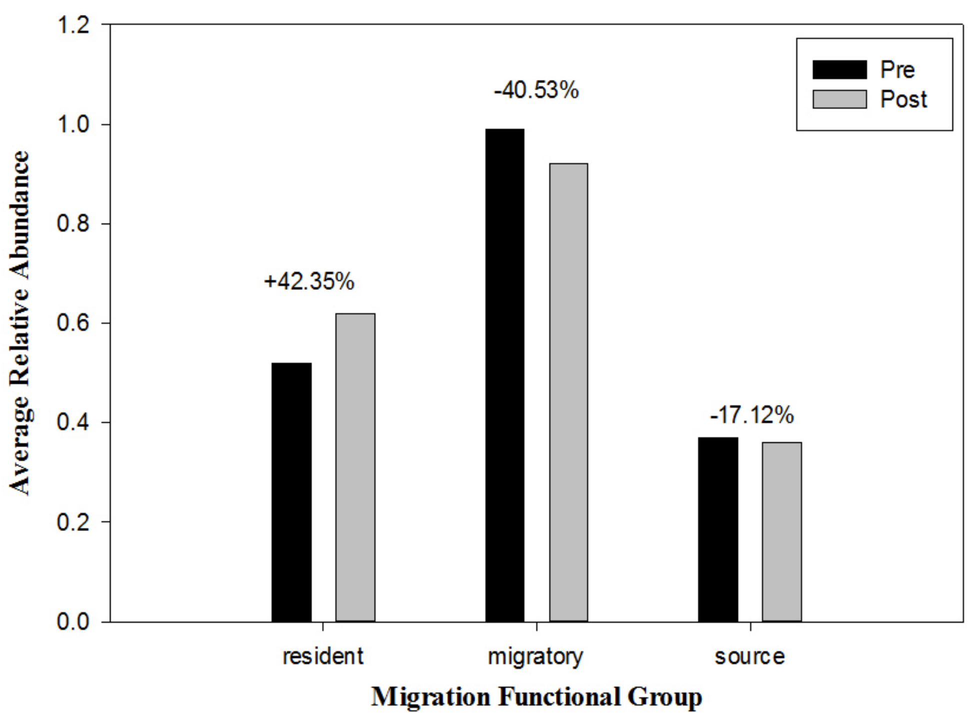

Figure 2.19 Average relative abundance of range guilds found in the Muskeg River that represents the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis).

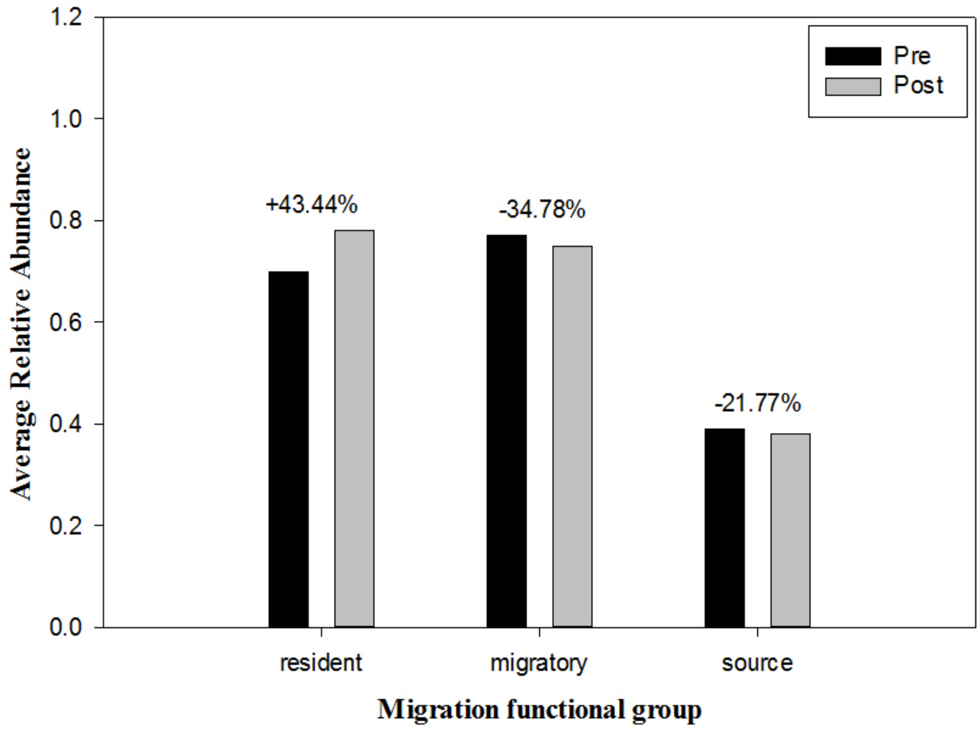

Figure 2.20 Average relative abundance of range guilds found in the Steepbank River that represents the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis).

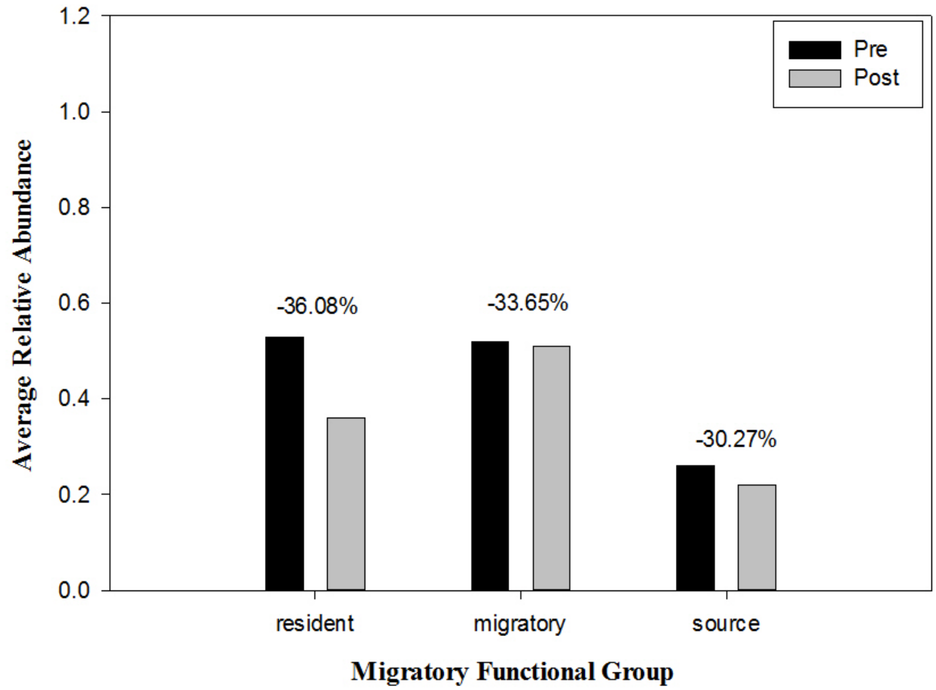

Figure 2.21 Average relative abundance of range guilds found in the Ells River that represents the variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis).

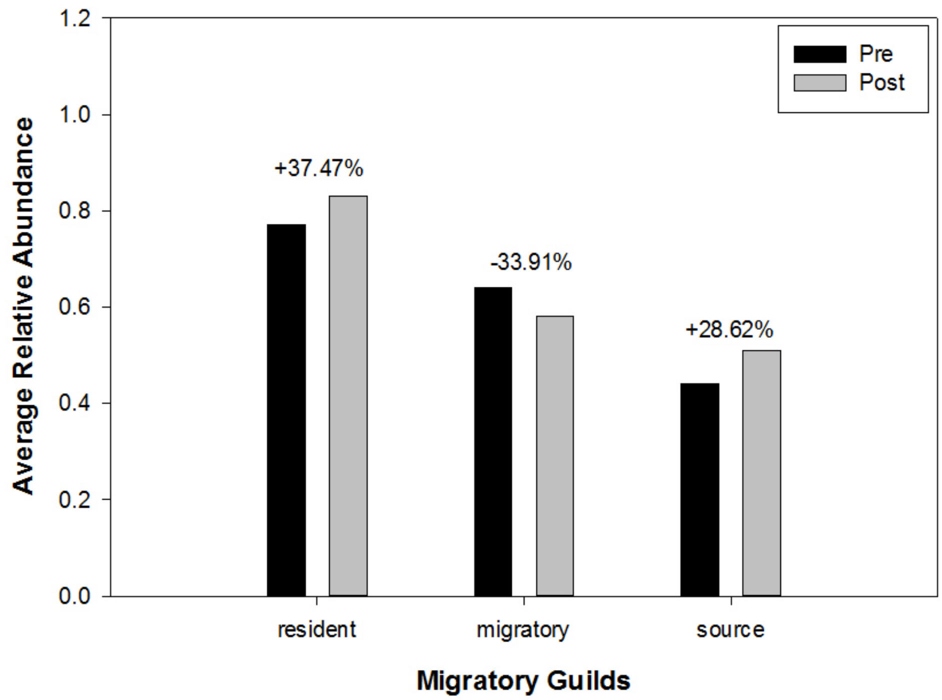

Figure 2.22 Average relative abundance of range guilds found in all three rivers collectively that represents the regional variation in community composition between pre- and post-development. SIMPER identified the percentage of each guild's contribution to the observed pattern of dissimilarity (Bray-Curtis).

PRIMER analytical software (v6.0, PRIMER Ltd, Plymouth, U.K.) with PERMANOVA+ was used for all multivariate routines, AMOSIM, and Similarity Percentage (SIMPER) analysis. Ordinations, hierarchical cluster analysis and descriptive statistics were run and plotted in R (http://www.r-project.org) with the aid of community ecology and graphic packages vegan, cluster and ggplot, as well as SigmaPlot version 12.0, from Systat Software, Inc., San Jose California USA, http://www.sigmaplot.com.

2.6 Discussion

This study was the first to investigate temporal community-environmental relationships and patterns in the AOSR, across which community composition could be affected by development. Overall, species community structure and trait-based patterns between levels of development within the AOSR remained unchanged; however, there was a significant decline in fish species richness when data were compiled into a regional database. Requiring further investigation, these results could be a plausible response to the hypothesis that species richness decreases as disturbance (land use or development) increases by means of impacts to local fish habitat availability and suitability.

2.6.1 Database Challenges

The river systems within the studied watersheds are dynamic, and fish communities therein can change in composition and distribution seasonally, and annually (Habit et al., 2006; Ostrand and Wilde, 2002; Pegg and McClelland, 2004). Inconsistencies in the compiled data created shortcomings in the database making it challenging to draw strong conclusions about the changes in the relative abundance of species. For example, sampling methods and efforts within each of these watersheds differed widely, between levels of development, seasons and tributaries. The Muskeg river sampling is predominantly fish fence (until 2009), and electrofishing. The Steepbank was measured with a wider variety of methods pre-development, including: fish fences, trawling, seine netting, gill netting and dip netting. Fish sampling efforts post- development was primarily done by electrofishing. The Ells on the other hand was sampled predominantly by gill and seine netting (pre), and electrofishing (post).

The difference in methods has led to different results than those of Schwalb et al. (2014), where evidence of reduced recruitment or increased mortality of migratory fish species was presented. Using fish counts collected from fish fences in the Muskeg River, Schwalb et al. (2014) observed a decline in the run of migratory species when taking into account dates of sampling. According to Schwalb et al. (2014), the total number of fish caught in the period from 1995 to 2006, was 78-87% lower than in the 1976/77. Furthermore, Schwalb et al. (2014) found a decline in three of the five migratory species outlined in Bond and Machniak (1979). Since 1995, Arctic grayling, longnose suckers and mountain whitefish have declined by 53-100%. Schwalb et al. (2014), characterize these results as a sign of reduced recruitment or increased mortality of migratory fish species.

Given the nature of the dataset (presence/absence), calculating a reduction in the number of fish and computing the level of significance from presence/absence data were not possible. Results were very conservative: four species decreased in average relative abundance, including Arctic grayling, white sucker, northern pike and pearl dace (Figure 2.4). Changes to the average relative abundance of each range guild, including migratory fish, were not significant between levels of development (Figure 2.19).

The discrepancy in results between this research and Schwlab et al. (2014) highlights three important things. First the importance of consistent methods, as differences in methods or study design can have a substantial effect on results (Allan and Johnson, 1997). Second, fish counts gathered from methods such as fish fences or electrofishing, compared to presence/absence data, are valuable as a measure of significance between levels of development can be generated if a change in community patterns is observed. Lastly, the seasonality of sampling, if not repeated at the same time annually, can cause discrepancy in the results.

Fish can be differentially sensitive to anthropogenic impacts (Karr et al., 1986), and while some are disperse for more favourable habitats, others migrate or move for reproduction (Daniel et al., 2014). If migratory species were going to be measured, sampling once annually on the same date would only capture those species that utilize these reaches at specific times of the year. As discussed in Schwalb et al. (2014), season may not be as important as the temperature of the water from which the fish could be cued to migrate or spawn, either entering or leaving the area being sampled. Water temperature and discharge are both shown to directly influence migratory behaviour of fish (Benjamin et al., 2014; Gowan and Fausch, 1996). For future monitoring efforts, temperature-defined sampling dates could be beneficial. For example, white suckers begin to migrate when the daily max water temperature is 10°C (Bond and Machniak, 1979), and mountain whitefish and bull trout both had migration patterns explained by water temperature (Benjamin et al. 2014; Homel and Budy, 2008). Taking this and other similar measurements into consideration, an optimal temperature window(s), or temperature-dependent times could be set for fish sampling within these rivers.

2.6.2 Scale of Interpretation

As discussed, rivers are strongly influenced by human activity that impacts land-river processes (Allan and Johnson, 2007). Many of these processes, including flow regimes, operate at spatial scales ranging from local (tributary) to regional, and can be affected differently by human activities and development. An understanding of scale-dependent processes will become important as development proceeds and management of these freshwater ecosystems grows increasingly more important.

Despite close geographic proximity and interconnectedness to the main stem Athabasca River, differences in fish species richness, community composition, and relative abundance were observed among all three tributaries. Moreover, differences were observed when the data were pooled into a regional dataset. As stated previously, the intensity of sampling effort and methods within versus between sub catchments could have influenced the findings in addition to the scale at which samples were analyzed (Allan and Johnson, 1997). While site specific (tributary) analysis is important for understanding local impacts of OS development on fish communities, a regional approach can help to characterize cumulative impacts to these systems and patterns of regional stress to fish communities (Daniel et al., 2014).

The difference seen in species richness here, between individual tributaries and the regional dataset, demonstrates that studies of fish communities on a regional and watershed scale can elucidate different sates of community change. According to Allan and Johnson (1997), there is an implicit assumption of local control, which exists, as demonstrated in this research through the different behaviours exhibited by all three rivers. For example Arctic grayling drove the majority of the change in average relative abundance of fish species within the Muskeg and Steepbank, but were not present in the Ells River. However, the analysis of the regional dataset in addition to the tributary dataset provides the opportunity to try and characterize further controlling mechanisms. According to Freund and Petty (2007, c.f. Daniel et al. 2014), mining can have wide spread influences across watersheds, including those catchments hydrologically connected but with no anthropogenic developments. The fish in these catchments could be impacted by mine development through restricted migratory routes and shifts in habitat quality and quantity, resulting in reduced species pools. Furthermore, a regional perspective has the possibility of capturing the presence/absence of fish species that may migrate from different locations within the region, and ensure there counted within the census (Daniel et al., 2014).

2.6.3 Community Structure

The patterns observed here are inconsistent with the theoretical explanations outlined in the hypothesis (objective 4) that these stable or lesser-disturbed riverine systems contain more specialist species than do variable systems (Poff and Ward, 1989; Poff and Allan, 1995). Specifically, some of the patterns observed are consistent with trends in fisheries patterns within Alberta. For example, Arctic grayling showed a marked decline in average relative abundance post-development in both the Muskeg and Steepbank Rivers. Bond and Machniak (1979) predicted declines in recreation and sport fish on the Muskeg River, including Arctic grayling, in the late 1970s. This retrospective analysis shows a lack of proper implementation of effective management plans for fish in these regions, and furthers the importance of comprehensive, historical databases from which to make comparisons. Today, Arctic grayling is listed as a species of special concern though Alberta's Endangered Species Conservation Committee (ESC) (AESRD, 2014). Issues related to the population's decline are cited to include increased accessibility to remaining habitat-increasing angling pressure and blocked migration routes due to altered streamflow and improperly placed culverts (AESRD, 2014). Similar to the hypothesis tested here, explanations for the decline, including fishing pressures, could also lie in the species being a more specialized feeder (invertivores-carnivore) and a lithophil reproductive strategist: a more sensitive reproductive group.

The trophic guild structure appears to have remained static over time for all three rivers. As per Primack (1998) and Taylor et al. (2008), dietary shifts are common following anthropogenic alterations to river systems, and a signal of such was hypothesized to be stronger on the Muskeg, and possibly the Steepbank given the extent of development in these watersheds. Differences in the trophic representation were marked between each river. In the Muskeg, as hypothesized, omnivores had the largest average relative abundance post development; however, the difference was only marginally larger than the other guilds. Surprisingly, the Steepbank's carnivorous fish species possessed the greatest average relative abundance, with the omnivores possessing the smallest. These results contrast what was expected given the hypothesis. It is important to note that the contribution of each guild to the community is questionable. For example strictly carnivorous species were poorly represented (northern pike found in all three rivers and dolly varden found only in the Muskeg) in all three rivers, pre- and post-development.

2.6.4 Threshold Responses

The results here demonstrate a need for further data collection and exploration. Using Stedman (1988), Argent and Carline (2004), Waver and Gearnen (1994), and Wang et al. (1997), as development proceeds thresholds should be established to set limits to direct and indirect anthropogenic impacts from which fish species could potentially decline or be altered in response.

There is extensive literature covering changes in biological and fish assemblages in response to or associated with catchments being developed (*see* Allan, 2004). Steedman (1988) reported findings of marked fish community changes at 25-50% land use. More recently, Argent and Carline (2004), Weaver and Garmen (1994) and Wang et al. (1997) found similarities in their research where they reported that, as developed land use increased from 10 to 20%, fish communities changed (through a decline in Index of biological integrity (IBI) scores). In agriculture catchments, another study reported declines in fish IBI and habitat scores at 80% land use (Wang et al., 1997).

The observed range of relationships between an aquatic system characteristics and land use appears to be non-linear, and according to Gergel et al. (2002) and Daniel et al. (2014), a threshold response could be exhibited. These ranges, or limits, suggest that fish communities in the Muskeg, Steepbank and Ells river(s) may remain resilient or unaffected by the impacts of disturbance on their habitat.

Thresholds have been documented primarily for percent impervious surface in a catchment and are non-linearly related to changes in fish and invertebrate communities (Paul and Meyer, 2001; Gergel et al., 2002). More recently, Daniel et al. (2014) tested for thresholds in mined areas, at which fish community composition patterns changed substantially from best available fish habitat conditions to a state with increasing disturbance. The analysis found that threshold responses were occurring at very low mine densities (47.2 % of the significant thresholds found occurred at a density of only ≤ 0.01 mines/km^2). When compared to other land uses, including agriculture and urbanization, mining had more marked impacts on fish communities. Should development continue in these watersheds, these results make a strong case for increasing and standardizing the monitoring efforts of tributaries and regions.

2.7 Conclusion

According to Quist et al., (2005), in the absence of any anthropogenic impacts or changes to the river systems, large-scale and long-term changes in community composition patterns should not be observed. However, assigning attribution of environmental drivers that affect/cause the observed changes in community composition/abundance patterns may be difficult to prove due to confounding effects of different sampling methods and years between sampling events. Furthermore, it is increasingly difficult to untangle the collective anthropogenic and natural impacts on these systems, both direct (land disturbance) and indirect (climate change). The current lack of comprehensive data, and the conclusions presented here suggest further studies are needed and it reinforces the importance of comprehensive monitoring plans.

Currently, the database has produced a limited understanding of the fish communities in these rivers, ultimately, making it difficult to manage, protect and monitor changes resulting from anthropogenic land use and impacts (Infante and Allan, 2010). Furthermore, without a comprehensive baseline for status of these communities, it is challenging to find a benchmark for which to restore the system back to when development ceases.

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Chapter 3. Potential Impacts Of Changing Flow Regimes On Fish Community Structure And Function In The Muskeg River, Alberta

3.1 Introduction

Modification of streamflow is one of the most widespread human disturbances of stream environments (Ward and Stanford, 1983; Bain et al., 1988). Streamflow is considered to be the overriding force in rivers (Huphries et al., 2008) and is described as a major variable that regulates the ecological integrity of river systems (Poff et al., 1997).

Directly, streamflow variability can affect fish communities by influencing important life history processes through a variety of inter-related mechanisms (Bunn and Arthington, 2002; Humphries et al., 2008; Poff et al., 1997; Richter et al., 1996; Welcomme et al., 1995). A river's flow regime is described by five components: (1) magnitude, (2) frequency, (3) duration, (4) timing, and (5) rate of change (Poff et al., 1997; Richter et al., 1996; Figure 3.1). Each component of the flow regime individually and in combination provide natural biological cues and influence the critical life history processes of riverine biota, including fish. Flow regime, for example, can affect the occurrence and timing of spawning, recruitment, and (seasonal) migrations (e.g. Bunn and Arthington, 2002; Humphries et al., 2008; Poff et al., 1997; Richter et al., 1996; Welcomme et al., 1985). Aquatic organisms, which inhabit riverine ecosystems, have thus been strongly linked to a river's natural inter- and intra-annual flow regime (Poff and Ward, 1989; Richter et al., 1996).

Indirectly, flow plays a role in controlling aspects of stream ecology including habitat and biodiversity through the maintenance or changing of physical stream characteristics (Bain et al., 1988; Bunn and Arthington, 2002; Hynes, 1970; Poff and

Ward, 1989; Poff and Zimmerman, 2010; Resh et al., 1988). Major streamflow characteristics are related to different aspects of fish habitat, including: (1) seasonal pattern of flows where high flows may maintain ecosystem productivity and diversity and low flows provide changes in recruitment opportunities; (2) timing of extreme flows where variable magnitudes may favour the life cycles of some aquatic and riparian species but not others; (3) frequency and duration of floods and droughts, where different species have life histories that can accommodate changing flows or have different tolerances to flood or drought conditions; (4) seasonal and annual flow variability, which may create disturbances to benefit native species; and, (5) the change rate of flow conditions influencing species persistence and coexistence (Richter et al., 1996).

Other features of the flow regime that can determine the availability and quality of aquatic and riparian habitat riverine organisms are: current velocities, morphological structure of a riverbed and banks, erosion, substratum stability, sediment movement and water quality (Humphries et al., 2008; Poff and Ward, 1989; Poff et al., 1997; ter Braak and Verdonschot, 1995; Figure 3.1**).**

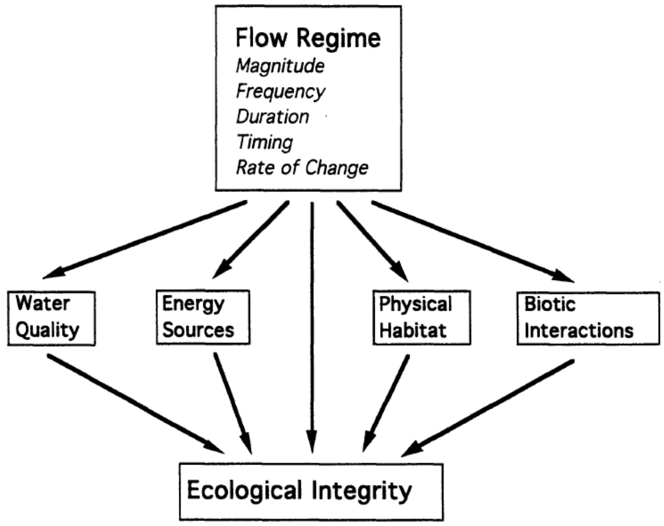


Figure 3.1. The five components of the flow regime (magnitude, frequency, duration, timing and rate of change), can directly and indirectly influence ecological integrity through effects on water quality, energy sources, physical habitat features and biotic interactions. *Source:* Poff et al. (1997).

Events such as floods and droughts are natural hydrologic disturbances in riverine ecosystems. Variations in river flow rates and extreme conditions are primary sources of environmental variability and disturbance, where disturbances are events in time that disrupt an ecosystem or community structure and can change resource availability, size and composition of substrate, or other components of the physical environment (e.g. woody debris, vegetation, etc.) (Pickett and White, 1985; Stanford and Ward, 1983). Other disturbances such as landscape alterations (anthropogenic development) and water withdrawal can also disturb a river's natural flow regime (Dunne and Leopold, 1978). According to Resh et al. (1988), these types of disturbances are events that fall outside the predictable range of habitat fluctuations a given organism or population is adapted to or can tolerate.

Furthermore, flow regimes have been identified as important in determining patterns of aquatic biodiversity. Bunn and Athrington (2002) provide a conceptual framework that describes the relationship between flow and biodiversity in four guiding principles (Figure 3.2). Principle 1 focuses on elements of the flow regime that can affect aquatic biodiversity through changes in the physical nature of habitat, including large events that affect channel form and shape. Alternatively, droughts and low-flow events are identified as limitations on habitat availability. Principle 2 describes the seasonality and predictability of flow events or patterns and concurrently evolved life history traits including spawning and recruitment, which take cues from such events. Principle 3 shows how lateral and longitudinal connectivity can be responsible for maintaining access to floodplains and dispersal triggers for riverine species. Without these connections, populations can experience reduced recruitment with potential for extinction. Lastly, principle 4, the alteration of flow regimes, can leave rivers susceptible to invasive species that may exhibit more ‘generalist’ characteristics, establishing, spreading and persisting in these rivers.

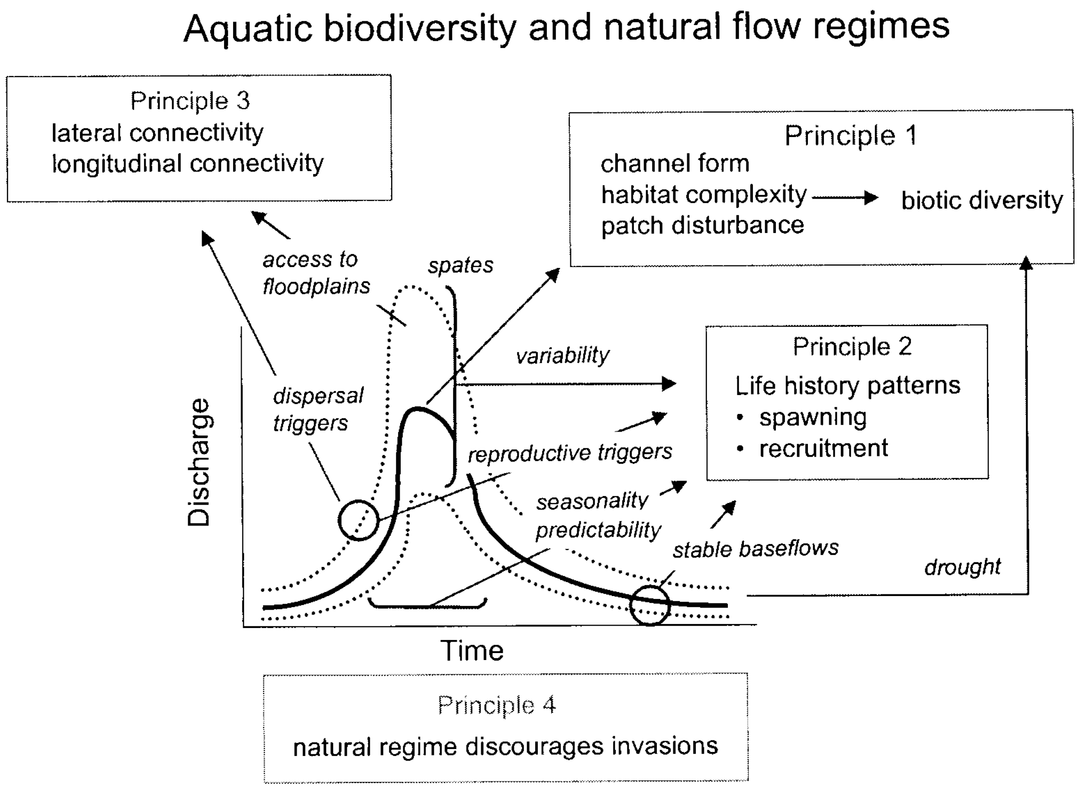


Figure 3.2. Aquatic biodiversity and natural flow regime. The natural flow regime of a river can influence biodiversity, both spatially and temporally, through several interrelated mechanisms outlined in four principles (Bunn and Arthington, 2002).

The organization, composition and successful persistence of biotic communities in river and stream systems, including fish, are therefore (in part) controlled by different variations of and disturbances to streamflow. These include dependability of normal flows, patterns of flooding (e.g., frequency and predictability), and low flows (Poff and Ward, 1989). Poff and Allen (1995), for instance, collected stream fish assemblage data, as well as long-term hydrologic variability data to test the hypothesis that the functional organization of fish communities is related to hydrological variability, as proposed by Poff and Ward (1989). Hydrologic regimes were defined as 'variable' or 'stable', and it was hypothesized that fish assemblages would associate with one or the other or arrayed across a gradient of habitat suitability. Associated with the hydrologically variable sites, communities were composed of species with more generalist feeding strategies, and a higher tolerance for silt, general substrata and slow-velocity streams and rivers. Rivers with more stable flow regimes were generally occupied by fish communities composed of species with more specialized trophic strategies, silt-intolerance and with a preference for fast to moderately flowing streams (Poff and Allen, 1995). Based on these hydrologic-assemblage relationships, hydrological factors are seen as environmental variables that can influence ecosystem integrity. These relationships demonstrate that disturbance or alterations to the flow regime have the potential to modify riverine fish community structure and composition.

3.2 Objectives

The quantity of water used by Oil Sands (OS) industry and development and the local and cumulative effects on hydrological regimes in the lower Athabasca region of Alberta is a growing concern (Grant et al., 2013; Peters et al., 2013; Sauchyn et al., 2015). Regionally, this is further compounded by land disturbance (Environment Canada and Alberta Environment, 2012) and climate change (Leong and Donner, 2015; Prowse et al., 2006). Focusing on the Muskeg River Watershed, one of the heavily developed river systems in the lower Athabasca region (RAMP, 2015), the overarching goal of this research is to assess whether historical and present changes in streamflow (discharge) regimes can be associated with alterations in observed fish community structure, in particular comparing pre- versus post-levels of OS developments. According to Karr (1981), fish communities are ideal indicators for the assessment of health and biotic integrity of rivers. As discussed in Chapter 1, a fish community can occupy a variety of niches, feed across trophic levels, demonstrate different migratory behaviours and spawning strategies, and contribute to overall river metabolism through respiration (i.e., oxygen uptake) from water (Argent and Carline, 2004). Fish communities are structured by their interactions with, and can reflect changes to, the surrounding biotic and abiotic environment (Allan, 2004; Argent and Carline, 2004; Olden et al., 2010; Wang et al., 2001). The specific objectives of this chapter are to:

(1) assess patterns of hydrologic change between pre- and post- development using indicators of hydrologic alteration (IHA) and categorization of annual hydrographs;

(2) examine patterns in fish species richness in relation to patterns in the hydrograph between pre- and post-development; and,

(3) propose additional criteria for in-stream flow needs (IFNs), and improving monitoring efforts for the Muskeg River in the face of ongoing development.

3.3 The Muskeg River Watershed Background Information

3.3.1 General Characteristics

The Muskeg River watershed is a tributary of the Athabasca Regional Municipality of Wood Buffalo (Government of Alberta, 2008), and covers an area of approximately 1460 km^2 (RAMP, 2012; Figure 2.1). The mouth of the river is located 55 km north of Fort McMurray where it connects to the main stem of the Athabasca river, 5 km east of Fort MacKay. In 2012, approximately 15% of the land within the watershed had been disturbed (RAMP, 2012), with an additional 50-60% slated for future activities (Government of Alberta, 2011).

Low topographic relief characterizes approximately 60% of the Muskeg River. Upper and lower reaches of the Muskeg River are dominated by peat soils that readily absorb snowmelt and precipitation. Combined with the river's low grade, the soil characteristics allow for high permeability and water storage, which may act as a barrier to water discharging back to the river. The Muskeg River watershed receives approximately 460 mm precipitation annually (Eum et al., *in press*). Strongly influenced by the watershed's topography, soil and climate, the Muskeg River watershed resembles other boreal forest systems, which influences the amount and variation in seasonal and annual water levels (Government of Alberta, 2008; McEachern and Noton, 2002).

The natural flow regime of the Muskeg River has seasonal variations in the streamflow that come primarily from the spring freshet, summer thaw of the peatlands, and winter ice cover. The natural flow regime is described in Alberta Environment (2008) as having a typical snowmelt pattern with streamflow peaking in late spring as the freshet melts into the river. Precipitation events typically cause small peaks in the fall (Alberta Environment, 2008). Freshet occurs in late April, early May when spring snowmelt contributes to approximately 50% of the Muskeg River's flow (McEachern and Noton, 2002). For the rest of the year, shallow ground water is the main source of streamflow to the Muskeg River watershed. Usually sitting less than 3 m below the surface, shallow groundwater can saturate the ground during the months when snow is melting. According to McEachern and Noton (2002), groundwater stored in the peat soils can contribute approximately 70% of the streamflow. It is this groundwater that the active OS mining projects have been licensed for use.

Using both RAMP and Water Survey of Canada data, annual mean daily discharge of the Muskeg River is $3.92m^3/s$ with a mean annual discharge is $3.75m^3/s$ (RAMP, 2013; Water Survey of Canada, 2015). During the 2013 open water season (May to October), minimum daily discharge was $1.15m^3/s$ and was 10% higher than mean historical minimum daily discharge of 1.05 m^3/s (RAMP, 2013). Maximum daily discharge in 2013 (open water season) was 80.6 m^3/s compared to the historical 25.3 m^3/s (RAMP, 2013). The highest mean monthly flow occurs in May and the lowest in February (RAMP, 2013; Water Survey of Canada, 2015).

In 2010, there were four active OS mining projects that were licensed under the *Water Act* to divert water within the Muskeg River watershed. These included Syncrude Aurora North and South Mines, Shell Muskeg River Mine and expansion, Shell Jackpine Mine and expansion, and the Imperial Oil Kearl Project (Government of Alberta, 2011). In addition, the Hammerstone Muskeg Valley Quarry is slated for development within the watershed (Government of Alberta, 2014). Below is a map of current OS lease boundaries and Water Act Licences (Figure 3.3).

While no active projects are diverting water directly from the Muskeg River, they are licensed to divert surface runoff and groundwater within the basin (Government of Alberta, 2011). Within the AOSR, companies operate under a zero discharge policy, where all water extracted for OS operations must be stored on site (Giesy et al., 2010). Based on a report from 2011, Syncrude Canada Limited, Aurora North mine was allocated 11,580,000 m^3 of surface runoff, 8,550,000 m^3 of shallow ground water, and 2,000,000 m^3 under a temporary diversion license in 2010. In 2010 a total 19,830,263 m^3

was withdrawn and only 10,149,000 m^3 returned to the Muskeg (Government of Alberta, 2011).

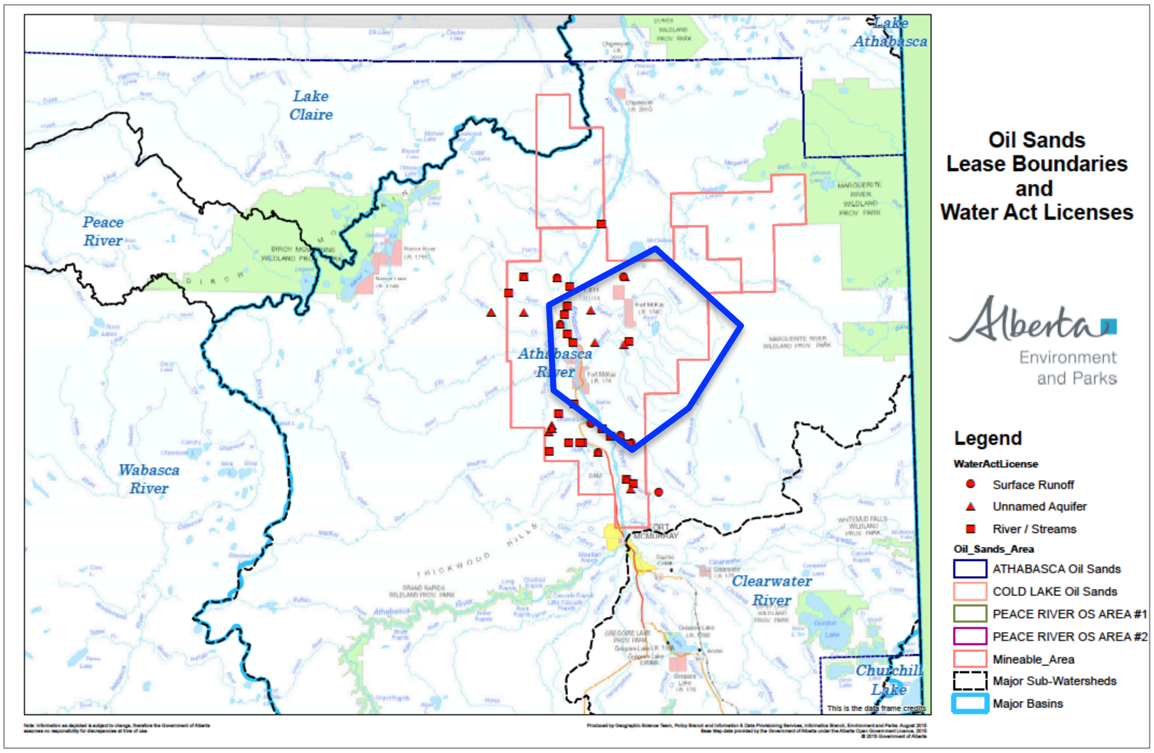


Figure 3.3. Oil Sands Lease Boundaries and Water Act Licenses. The Muskeg River watershed is outlined in blue. Source: Alberta Environmental Monitoring, Evaluation and Reporting Agency (AEMERA).

3.3.2 Muskeg River Watershed Interim Management Framework

Water is a key component of the OS industry, particularly in mining operations, where it is used to separate bitumen from sand and clay (CAPP [Canadian Association of Petroleum Producers], 2015a). While the use of freshwater has become more efficient, as the OS industry grows, so does the demand on freshwater resources (CAPP, 2015b). The Muskeg River watershed is underlain by mineable oil sands and is being developed for resource extraction.

In response to development concerns, the Muskeg River Interim Management Framework (IMF) was developed in 2008 by Alberta Environment (previous work done by Cumulative Environmental Management Association [CEMA]) to provide a structured approach to managing water quality and quantity within the Muskeg River watershed. The objective of the IMF is to maintain the natural seasonal variation of the system to reduce adverse effects on aquatic biota and maintain beneficial use of water as a resource. According to Alberta Environment (2008), the overarching goal of the IMF is to identify and maintain the biologically significant components of the flow regime in order to simulate the ecologically significant events of the Muskeg Rivers natural hydrograph. The ecologically significant flow conditions of the Muskeg River were identified as low flows, high flows, small flows and large floods. Inter- and intra-annually maintaining these conditions is key to sustaining the ecological integrity and functions of the river.

The IMF defines criteria for the evaluation of water quality and quantity and uses established limits, against which observed conditions and changes are compared. Seeking to preserve the natural flow hydrograph the water quantity objectives of the IMF utilizes the In-stream flow Needs (IFN) approach to set limits in order to maintain the natural flow required for aquatic organisms, aquatic or riparian habitat, water quality and the physical geomorphic process of the aquatic system.

The IMF monitors aquatic health, measured using population-level characteristics of aquatic organisms (density, diversity, recruitment and mortality). While aquatic health is difficult to measure compared to water quality and quantity, according to the IMF, it can act as a ‘litmus test’ for how well these parameters are being measured and monitored.

There are specific impacts estimated to affect the flow regime and ecological integrity of the Muskeg River. These include increased withdrawals, which are estimated to cause a more variable hydrologic regime, especially during initial phases of development, and lower variability as reclamation features, including pit lakes, are developed. Alberta Environment (2008) highlights how development is projected to have substantial negative impact on flows to tributaries of the Muskeg River.

3.3.3 Hydrologic Modeling of the Athabasca and Muskeg Rivers

In the Athabasca Region, Andrishak and Hicks (2011) explored the scenario of a $20m^3/s$ additional decrease in the average minimum winter flow in the main stem channel of the Athabasca River. While an arbitrary flow rate was selected for this scenario, the authors explain that it falls within the range considered similar to those used for impact assessments in the Lower Athabasca River (LAR) (Andrishak and Hicks, 2011). According to the model, winter, an already critical time for flow regimes in this region, could experience an approximate increase of 50% in complete flow interruptions, thereby likely to impact local and migratory fish populations in the region. Some fish species such as White and Longnose suckers use tributaries of the LAR, including the Muskeg River, for spawning and reproduction (Bond and Machniak, 1979). Similar to Bunn and Arthington's (2002) Principle 4, loss of habitat and connectivity in the area may act as barriers to completion of important life cycle events and recruitment for local fish communities.

Focusing on the Muskeg River, a modeling approach was used by Eum et al. (*in press*) to predict comprehensive hydrological responses to OS land-use and climate change within these systems. Using a suite of hydrological scenarios, climate projections

(2050s and 2080s; base period of 1980-2010), OS development projections, and land-cover types, Eum et al. (*in press*) were able to evaluate the impacts on the Muskeg River hydrological regime following land-use changes. Results showed hydrological responses to climate and land-cover changes contrasted, concluding that land-cover change played a significant role in alterations of hydrologic variables, with the exception of spring flows in the Muskeg River that were significantly influenced by climate change.

Eum et al. (*in press*) indicates that changes to the natural flow regime will occur in the Muskeg River basin, and it is projected to affect ecosystem integrity. Each of these examples highlights the importance of monitoring fish communities in tandem with changes or shifts in patterns of the natural flow regime. Ultimately, using historical data to gain an understanding of how each these variable interact and impact fish communities, as important indicators of change, will help to inform decisions regarding long-term water resource use, fisheries management and conservation.

3.4 Methods

3.4.1 Hydrologic Database

Long-term, daily measurements of hydrologic data (flow m^3/sec) for the Muskeg River was compiled from Water Survey Canada (WSC) approximately 10 km upstream, near Fort McKay (station identification number 07DA008, 57°11'28", -111°34'12"), during open water (approximately April to October) season. For data recorded since 2009, RAMP data were used to complete the flow records for the winter season (October to March).

3.4.2 Fish Community Database

The Muskeg River fish community data used in this chapter is the same as that collected for analysis in Chapter 2 (Appendix C). Historical datasets for Muskeg River fish community composition were compiled using current and historical datasets from a variety of monitoring and research programs in the AOSR, implemented in face of development including: Alberta Environment and Sustainable Resource Development (AESRD), the Regional Aquatics Monitoring Program (RAMP), the Alberta Oil Sands Environmental Research Program (AOSERP), and the Fish and Wildlife Management Information System (FWIMS). Fish collections were separated into sampling events by season, method, year, and site sampled. The data collected provided many challenges, including a variety of motives for collecting it, dissimilar methods, techniques and sampling periods. Given the lack of standardized estimates of abundance and survey methods, comparisons were restricted to measures based on presence/absence data for community composition.

Datasets were separated into two levels of development to coincide with those used by Alexander and Chambers *(in press)* for consistency (Table 3.1). Data collected from 1979 to 1995 was designated as pre-development and 1996 to 2012 as post-development. These periods are associated with no development (pre) and the initiation of development, land clearing, construction, infrastructure use, or potential for disruption to stream habitats (post). Most fish sampling sites were located in the lower portion of the Muskeg River, <25 km from the Athabasca River (Appendix B). Those further upstream ($>25^{+}$ km) were not included given the spatial heterogeneity of the Muskeg River and relatively low sample size.

Table 3.1. Defined pre and post levels of development for the Muskeg River watershed and respective time periods.

Level of Development	Explanation	Muskeg
Pre	No land use.	1979-1995
Post	Land clearing, construction, infrastructure use.	1996-2012

3.4.3 Assessing Patterns of Hydrologic Change Between Pre- and Post-Development

Compiled daily discharge (m^3/s) values and IHA variables were used to measure changes in hydrologic patterns including average annual discharge for pre- and post-development (The Nature Conservancy, 2015; Stuart MacKinnon, *pers. comm.*). A spilt period IHA analysis of the observed WSC and RAMP Muskeg River data were performed on pre- and post-development from 1979-1995 and 1996-2010, respectively, to distinguish relevant parameters from daily hydrological data that correspond to streamflow characteristics (Appendix B; Nature Conservancy, 2009; Poff and Ward, 1989; Richter et al., 1996, 1997).

Paired t-tests were executed to test for patterns of hydrologic alteration between pre- and post- development. The paired t-test was used to compare average annual pre-development discharge rate (m^3/s) to post development discharge rate (m^3/s) and average monthly discharge rate between pre and post development. Paired t-tests using outputs from the IHA software were used to compare 1-day, 3-day, 7-day, 30-day, and 90-day minimum and 1-day, 3-day, 7-day, 30-day, and 90-day maximum discharge rates (m^3/s) and average annual base flow between pre- and post-development.

3.4.4 Examining hydrological changes in relation to patterns in fish species richness between levels of development

A second method was used to assess hydrologic changes between pre- and post-development and to examine the relationship between these patterns and fish species richness. Using daily flow measurements, a hydrograph was plotted for each of the 29 years of collected hydrological data (1979-2013). For each hydrograph, three domains were developed to visually characterize the dominant flow patterns: (1) dominant spring flow, (2) dominant fall flow and (3) evenly distributed flow. Seasonality was defined using RAMP classifications (winter = January, February, March and April; spring = May and June; summer = July and August; fall = September, October and November). Each of the hydrographs was characterized relative to the hydrographs from other years.

Imposed on these, and using species richness calculations from Chapter 2, for years where data were available, the total mean species richness was calculated for each year, across the corresponding season(s), during which the samples were taken.

Lastly, to explore whether broad changes in patterns in the hydrograph (as defined by the domains listed above) had a significant affect on fish species richness for corresponding years, a one-tailed t-test was performed. This was done for the years defined as post-development (1996-2013), where sampling methods were more consistent than years defined as pre-development and only for years where species richness data were available.

3.5 Results

3.5.1 Assessing Patterns of Hydrologic Change between Pre- and Post-Development

Discharge

Average pre- and post-development daily discharge was determined for the available recoded time periods for the Muskeg River. Average pre-development daily discharge (1979-1995) was graphed to illustrate the hydrological conditions prior to land disturbance. Post-development daily discharge (1996-2013) was graphed to illustrate any possible changes in the hydrological conditions away from pre-development. The hydrograph is presented in Figure 3.4.

Following a similar hydrologic pattern pre- and post-development, post-development mean discharge rate was slightly elevated in October compared to pre-development, possibly coinciding with an increase in fall precipitation (Figure 3.4).

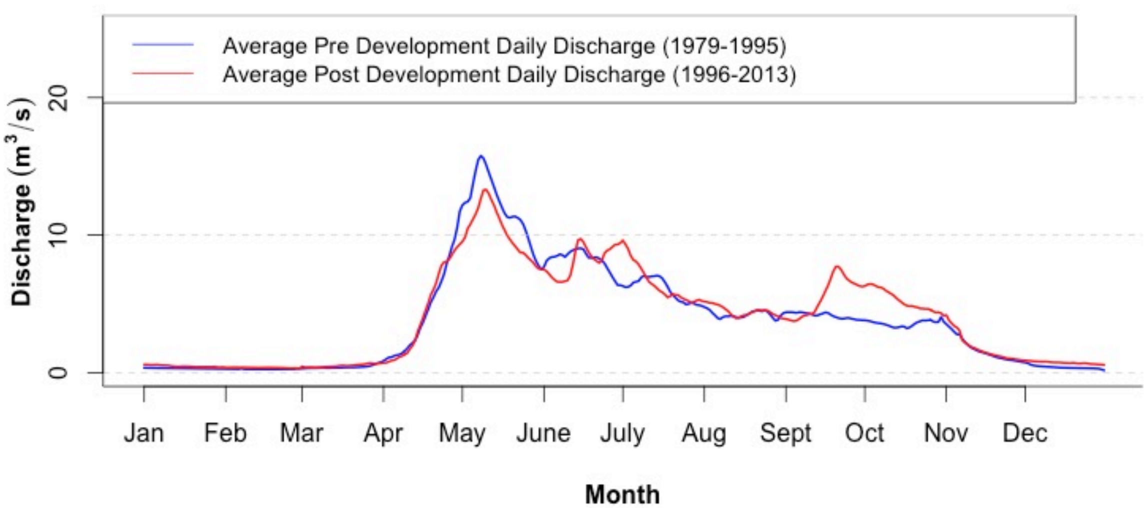


Figure 3.4 Muskeg River hydrograph displaying the average pre-development (1979-1995) daily discharge (m^3/sec) and average post-development (1996-2013) daily discharge (m^3/sec).

No significant changes in the patterns of hydrologic alteration were found between the average annual pre-development discharge rate (m^3/s) and post- development discharge rate (m^3/s) ($p>0.05$, Table 3.2). Similarly, no significant differences were observed in average monthly discharge rate between pre- and post- development ($p>0.05$, Table 3.2).

No significant differences were also found in average 1-day, 3-day, 7-day, 30-day or 90-day annual minimum discharge (m^3/s) or average 1-day, 3-day, 7-day, 30-day and 90-day annual maximum discharge (m^3/s). Finally, no significant differences were found in average annual base flow between pre- and post-development ($p>0.05$, Table 3.2).

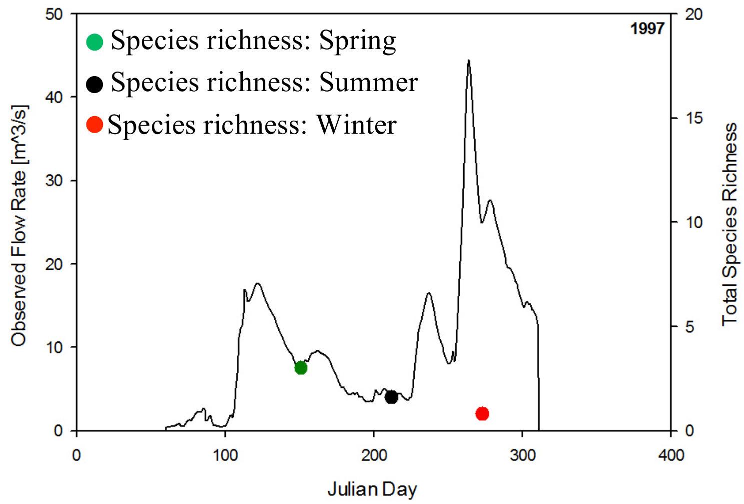
Table 3.2 Results from paired t-tests that were executed to test for patterns of hydrologic alteration between pre- and post- development for average annual discharge (m^3/s), average monthly discharge (m^3/s, January-December), average 1-day, 3-day, 7-day, 30-day or 90-day annual minimum, 1-day, 3-day, 7-day, 30-day and 90-day annual maximum discharge rates (m^3/sec) and average annual base flow.

Two-period analysis	**p**	**Summary Statistics** **n Pre**	**n Post**	**SD Pre**	**SD Post**	**Years**
Annual	0.65	17	19	1.30	1.98	1979-2014
January	0.50	17	19	0.66	1.38	1979-2014
February	0.33	17	19	0.25	0.54	1979-2014
March	0.14	17	19	0.12	0.35	1979-2014
April	0.37	17	19	2.45	3.62	1979-2014
May	0.30	17	19	9.94	7.26	1979-2014
June	0.34	17	19	5.47	9.11	1979-2014
July	0.83	17	19	4.85	4.53	1979-2014
August	0.88	17	19	5.59	4.54	1979-2014
September	0.19	17	19	4.84	5.69	1979-2014
October	0.17	17	19	2.59	4.86	1979-2014
November	0.53	17	18	1.49	3.13	1979-2013
December	0.68	17	18	1.17	2.23	1979-2013
1-day min	0.59	17	18	0.12	0.11	1979-2010

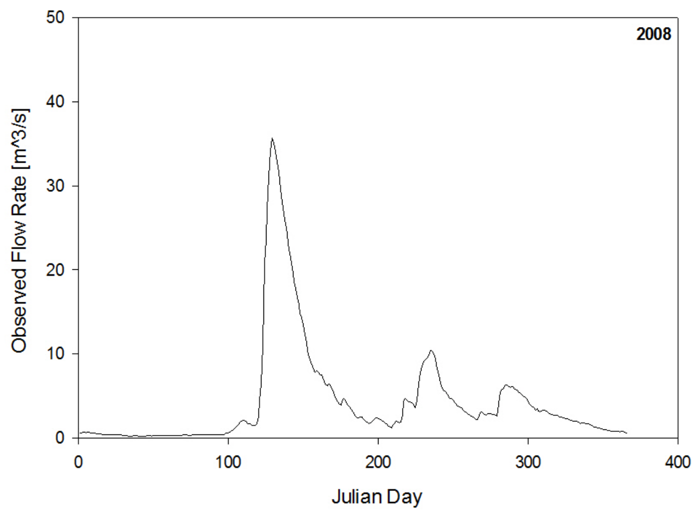
3-day min	0.32	17	15	0.12	0.12	1979-2010
7-day min	0.24	17	15	0.12	0.13	1979-2010
30-day min	0.10	17	15	0.13	0.20	1979-2010
90-day min	0.28	17	15	0.21	0.38	1979-2010
1-day max	0.26	17	15	16.70	18.37	1979-2010
3-day max	0.37	17	15	16.35	12.57	1979-2010
7-day max	0.38	17	15	14.15	11.70	1979-2010
30-day max	0.38	17	15	8.69	7.38	1979-2010
90-day max	0.44	17	15	4.46	4.31	1979-2010
Base flow	0.17	17	15	0.03	0.02	1979-2010

3.5.2 Examining patterns in fish community composition in relation to patterns in the hydrograph

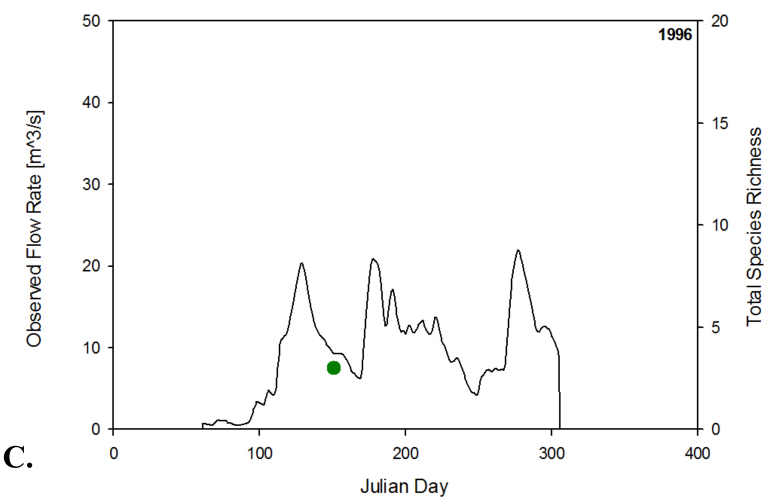
The methods used to both examine patterns in hydrology and to relate them to changes or patterns in fish species richness were inconclusive. Observed flow rates (m^3/sec) were plotted against Julian day, and species richness where possible for each year data were available (Figure 3.5; Appendix C). It could be suggested that this is due, in part, to deficiencies in the dataset, primarily the dominant data gaps in fish community data (from 1981 to 1995). Furthermore, this approach does not take into account any possible time lag between flow and potential impacts to species richness.



A.



B.



C.

Figure 3.5 Muskeg River average daily discharge plotted using observed flow rates (m^3/s). Each hydrograph represents one of the three domains created to characterize the dominant flow patterns. Where data were available, total species richness was plotted over the Julian days associated with the season sampling took place. A. The hydrograph from 1997 represents a domain with a dominant fall flow. Total species richness was recorded in spring, summer, and winter. B. The hydrograph from 2008 represents a domain with a dominant spring flow. No data were available to calculate total species richness. C. From 1996, the hydrograph representing a flow domain that is evenly distributed throughout the year. Total species richness was recorded in the spring.

Some further inconsistencies in the dataset were identified, including differences in methods used to gather fish samples and seasonal sampling inconsistencies from year to year.

To assess differences in observed frequency of seasons sampled and methods used pre- versus post-development, introducing a bias, a contingency table analysis was used. The methods used to gather fish samples pre-development were significantly different than the methods used post-development, χ^2 (d.f. = 7, n = 86, $p<0.05$; Figure 3.6). Pre-development methods consisted of hoop nets, fish fences, seine nets, kick nets, angling, hoop nets and less commonly, electrofishing. This changed post-development, where electrofishing was the dominant method used, followed by fish fence sampling.

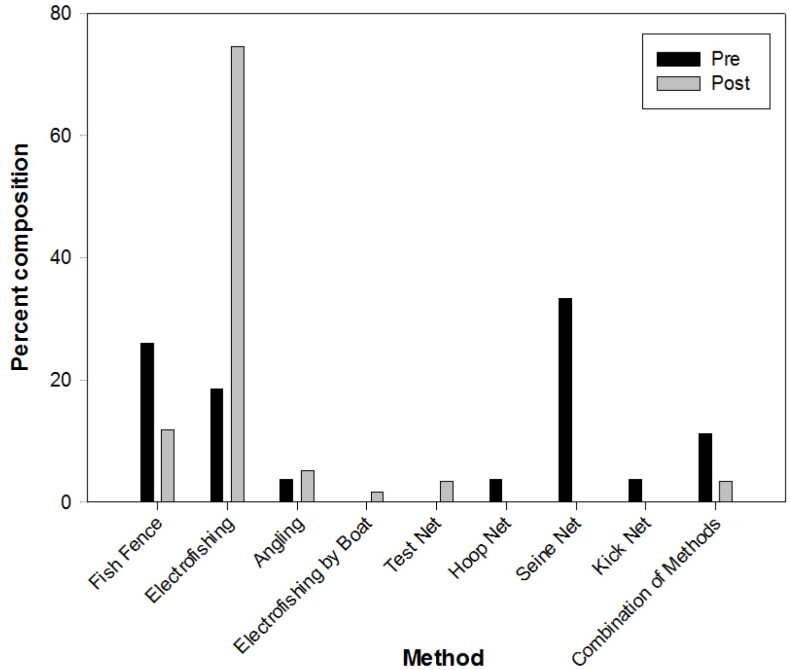


Figure 3.6 Percent composition of fish community samples taken from the Muskeg River using various methodologies: fish fences, electrofishing, angling, electrofishing by boat, test nets, hoop nets, seine nets, kick nets and a combination of methods, pre- and post-development.

Due to the data being collected from multiple sources, there is a significant difference in the seasons during which samples were collected, between pre- and post-development, χ^2 (d.f. = 4, n = 88, p <0.05; Figure 3.7).

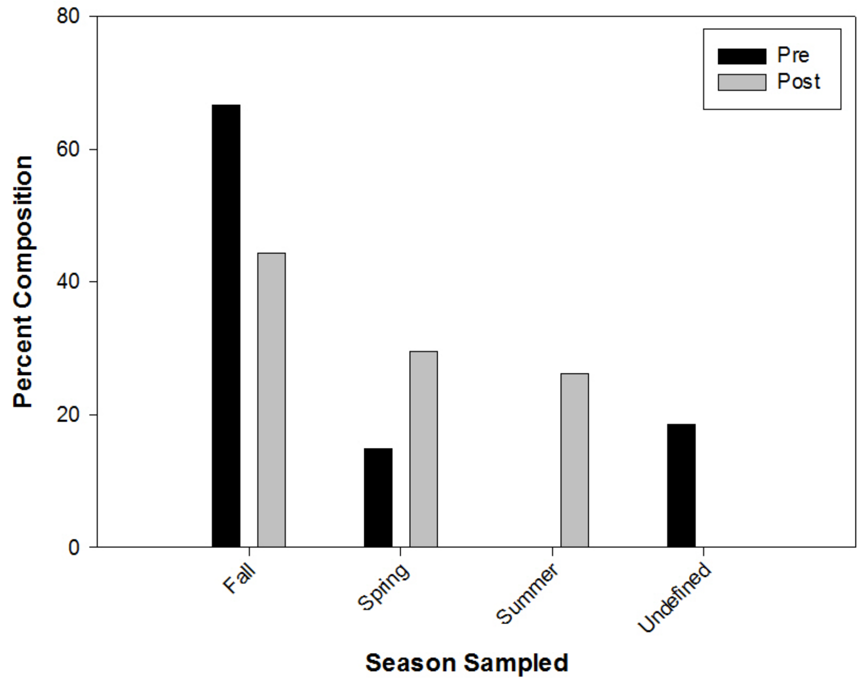


Figure 3.7 Percent composition of fish community samples taken from the Muskeg River during Fall, Spring, Summer and seasons that were undefined in the literature, pre- and post-development.

While samples were taken at various times of the year (seasonally), post-development had more standardized methodology, with electrofishing and fish fences primarily used to collect fish samples. A one-tailed t-test was used to determine whether mean species richness in domain two ($n_{years} = 9$) was significantly different than the mean species richness in domain three ($n_{years} = 7$) and no significant difference was found ($p = 0.81$; SD domain 2 = 3.64; SD domain 3 = 2.97). Domain one was not used in the analysis as there was only one year classified in that domain post-development. There was no fish species richness available for 2007, 2008 and 2010.

3.6 Discussion

3.6.1 Variation in Flow Regime

An assessment of patterns of hydrologic change, found no significant differences between pre- and post-development (Table 3.2). There were 12 years with periods of missing data greater than 50 days, the longest of which was 134 days. The data were not interpolated across the data gap(s) to prevent any anomalies in the statistics. Additionally, parameters calculated with fewer than twenty years of continuous data fall below the suggested 20 years of continual data ideal for capturing the natural variation in flow regimes (Richter et al., 1996).

The second method used to assess patterns of hydrologic change failed to identify any patterns or relationships between species richness and the three-hydrograph domains. This could due, in part to critical information gaps in the dataset, including a gap in fish community data from 1981 to 1995, differences in sampling methods and seasons sampled between pre- and post-development. Furthermore, the static nature of this method might not be able to capture or identify any time-lagged responses including accumulating stressors or gradual OS development over time on species richness, compared to more distinct events or abrupt changes in environment (e.g., dam construction).

According to Karr (1981), the nature of fish sampling including methods, motivations, location of sampling along a river and seasonal and annual migrations can cause an inaccurate representation of the fish community composition and changes. From the compiled Muskeg River fish community dataset, many challenges have been acknowledged and deficiencies identified, that could lead to erroneous conclusions. The following section elaborates on some of the challenges that prevented the completion of objective 2, which was to examine patterns in fish community composition in relation to patterns in the hydrograph between pre- and post-development.

3.6.2 Critical Information Gaps

Exploring the data and potential methods to complete the objectives resulted in the identification of critical information gaps, including: temporal gaps in the data, inconsistent sampling methods, and seasonal sampling from year to year. Further, there are many challenges due to the nature and composition of the fish community (for example the total number of species present during any sampling event), suitable control site/samples due to the gradual nature of development within the Muskeg River watershed, and contextualizing the challenges of monitoring in a region with multiple impacts that can be attributed to changes in fish community composition. Some studies exclude sites based on low species richness during a given sampling event. For example, in Poff and Allan (1995), any candidate sites with less than 20 species were excluded in order to reduce the chance of bias against streams with naturally low species richness. Compared to the dataset compiled for this study, the greatest number of species sampled during any event was 16 (Appendix C).

3.6.3 Temporal Gaps in the Data

In 1976, the Muskeg River watershed became the focus of OS development and since, fish community measurements have been taken through a variety of monitoring programs and research efforts including RAMP, AESRD, AOSERP and FWIMS. According to Poff and Zimmerman (2010), the ability to detect a response from a community (threshold or linear), along a gradient of flow alteration is challenging when data are missing. The largest gap in fish community data from the Muskeg River spans 14 years from 1981 to 1995. With critical ecological data missing during the initial stages of development within the watershed, the compiled data and sample sizes could have insufficient statistical power to detect a response from the community to varying levels of development (Poff and Zimmerman, 2010). Furthermore, a greater number of years of consistent and continuous sampling would be beneficial to measure any change in fish community composition, to ensure that the presence or absence of migratory species is accurately captured and to confirm the presence or absence of species is not a random event.

3.6.4 Inconsistent Sampling Methods

The methods used to gather fish samples pre-development were significantly different than the methods used post-development. Fish community sampling in the Muskeg River watershed is challenging for many reasons, including attempting to capture a wide array of fishes with varying physiology (size, life stage, etc.) and habitat preferences. The catchability of fish within a community, or the proportion of fish that are available to be captured by a defined unit of fishing effort, is – in part, related to the efficiency of the gear being used to sample the community (Portt et al., 2006; Ricker, 1975). Methods used to gather fish samples were inconsistent between years and levels of development in the Muskeg River watershed. Pre-development methods consisted of more netting (e.g. hoop, seine and kick nets), fish fences and less commonly, electrofishing. This changed post-development, where electrofishing was the dominant method used, followed by a few years of fish fence sampling (Figure 3.6).

It could be suggested that the consistency in post-development methods used to collect fish samples could make that data more comparable or similar, relative to the pre-development collections. However, when differences in mean species richness between two of the pre-defined hydrological domains for post-development years was assessed, no significance changes in patterns were observed. In the absence of baseline data, post-development data with continued standardization of methods for capturing fish would improve the possibility of linking hydrologic change to alterations in fish community composition and structure.

3.6.5 Seasonal Sampling Inconsistencies

Since data is being collected from multiple sources, there is a significant difference in the seasons during which samples were collected (Figure 3.7). Fish integrate their physiological functions with environmental cycles where many fish breed and migrate depending on seasonal cues (Northcote, 1984). Freshwater temperate zone fish spawn primarily in spring and early summer, while others such as the salmonids do so more commonly in autumn (September to December) (Sundararaj, 1981; Swanson, 1995). Inconsistent sampling during either or both of these seasons could further lead to sampling error or misrepresentation of the community (in both abundance and species richness) as some species can be present or absent seasonally.

Roy et al. (2005) discussed the importance of identifying patterns in altered flow regimes including high storm flows in the late spring/early summer spawning and during low flow events in autumn, when habitat availability can become restricted. Each of these hydrologic events has the potential to increase the mortality of eggs or young-of-year fishes (Bunn and Arthington, 2002; Poff et al., 1997; Roy et al., 2005), altering the fish community composition. While insufficient data were available to assess this, it is an important example of how seasonally, hydrologic patterns can change fish community composition.

3.6.6 Suitability of Control Sites and Replication of Samples

Another challenge was defining a suitable control, or reference condition against which to compare change within these systems. Years defined as pre-development was to act as the control or comparison for both flow and fish community metrics. This proved challenging due to the gradual nature of development within the region and the watershed, which is ongoing today. Development first began in the Muskeg River watershed in the late 1970s. In other studies, either a second or ‘control’ river was sampled with a similar effort, or the same river was sampled but from different sections. Repeated sampling from multiple locations in one river (ideally covering all the habitat types) would allow a greater coverage of fish species presence or absence, as well as repeatability of sampling, to increase statistical power to detect any change or response patterns. For example, Koel and Sparks (2002) had 26 fixed sites that were sampled per year, Armstrong et al. (2000) sampled from 27 sites, Pegg and Pierce (2002) sampled from 15 river segments, and finally Bain et al. (1988) sampled eight sites in one river annually to capture spatial-temporal variation in both fish assemblage structure and microhabitats. While there are (historical and current) samples taken in more than one location on the Muskeg River, the majority of samples that were taken are from close to the mouth of the river (approximately 57°7'57.60" N, 111°35'6.08" W). Due to the heterogeneous physical nature of the Muskeg River, it is important to understand whether samples collected are representative of those within the entire Muskeg River or a reflection of a more localized impact.

3.6.7 Alternative method: quantifying the relationship between flow patterns and species richness

Following Poff and Allen (1995), along with a more comprehensive data collection, a canonical discriminate analysis (CDA) could be used to determine if there are linear combinations of hydrologic variables that could discriminate fish community composition (presence / absence) between pre- and post-development, and to explore the suitability and application of multivariate methods for analysis due to the nominal classification of the two groups (pre- and post-development). CDA is a dimension-reduction method similar to a principle component analysis (PCA) and canonical correlation, which can summarize between-group variations in the same way PCA, may describe total variation.

Compared to Poff and Allen (1995) the most limiting factor in this study are the quality and scope of the fish community data collected to date. Where Poff and Allen utilized the combination of data taken from 34 sites along a river, this research was limited to samples taken from two sampling site (post-development) and various, but inconsistently monitored sites along the river (pre-development). Key requirements in sampling design that will need to be implemented to use such multivariate analytical methods include, an increase in temporal and spatial scale monitoring for an adequate description of fish community composition (i.e., an increase in sites monitored along the Muskeg River to capture the population of fish occupying different habitats and across seasons to capture the migratory fish population).

This research highlights important gaps in the data compiled, including differences in methods used to sample fish, seasonal sampling discrepancies and gaps in the data (both hydrologic and biotic). In other literature, (e.g. Koel and Sparks, 2002; Humphries et al., 2008) flow metrics (such as IHA) have successfully been used to capture a range of ecologically relevant natural and anthropogenic hydrologic variables important for sustaining fish communities.

Regarding the IMF implemented in 2008 by Alberta Environment, this analysis shows that because of a lack of consistent and long-term fish community monitoring program in the Muskeg River, the fish data available is of limited value for assessing the efficacy of current water quality and quantity management criteria in the system. If the IMF is going to be based on IFN (water quantity needed to sustain a healthy aquatic ecosystem and to meet the needs of human activities), further monitoring and evaluations will require a more comprehensive assessment of fish community composition.

Poff et al. (2007), suggests that current management practices often do not recognize the importance of the integrity of the natural flow regimes. Understanding the natural flow regime of a river and its relationship with fish communities will provide a more holistic, ecosystem-level understanding of the implications of anthropogenic development in the AOSR. This is increasingly important when taking into consideration the research by Andrishak and Hicks (2011), Alberta Environment (2008) and Eum et al. (*in press*). Given modelling projections, the impacts of OS development are projected to have regional cumulative effects flow regimes, with effects yet to be determined.

3.7 Conclusion

This research highlights the importance of standardized and continuous sampling to monitor important ecosystem changes in the Muskeg River and watershed. There are critical information gaps in the compiled datasets, preventing an analysis of structural patterns in fish community composition in relation to pre- and post-development patterns in flow regimes in the Muskeg River. These confines have limited the ability to draw conclusions about important ecological endpoints pertaining to changes in fish communities and flow patterns.

An appropriate monitoring program needs to be designed to address specific questions that allow for a quantitative analysis to examine the relationships between changes in the hydrological regime and fish community structure. These questions should incorporate potential impacts of other the cumulative affects including, climate change and other development pressures in the region. Alternative (or additional) approaches could be taken including monitoring habitat quality and quantity. While not the focus of this chapter, other studies looking to test a similar hypothesis (e.g., Armstrong et al., 2000; Bain et al., 1988; Bice et al., 2014) used habitats as a proxy or product of flow regimes to quantify how changes in habitat suitability affected fish community composition.

With ongoing development, and future changes to the Muskeg River watershed described earlier by Eum et al. (*in press*) and Andrishak and Hicks (2011), it is important for future research and monitoring efforts are coordinated to improve the ability to examine the relationship between the flow regime in the Muskeg River and fish communities therein. Despite challenges, fish communities, which collectively occupy a broad range of habitats, have a variety of habitat requirements, and depend on certain ecological processes, are more likely to display a wide range of sensitivities to changes in habitat modification, be it through changes to flow regimes or other anthropogenic impacts. Improved consistency and standardization of fish community monitoring efforts would help to measure changes in the integrity of local ecosystems within the Muskeg River and elsewhere in the AOSR.

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Chapter 4. Conclusion And Recommendations For Future Research

4.1 Introduction

The overarching goal of this thesis was to examine the potential impacts associated with land use disturbance and Oil Sands (OS) mining operations on fish community composition patterns in three rivers located in the Athabasca Oil Sands Region (AOSR). Using historical data sets from a variety of monitoring and research programs conducted in the region since 1978, this thesis attempted to evaluate disturbance-related patterns in fish community composition. The results from the studies conducted are intended to provide an improved understanding of the long-term cumulative changes within these systems and identify knowledge gaps in data collection to help guide and improve future monitoring plans for rivers in the AOSR.

Fish communities are sensitive to anthropogenic developments and are structured by their interactions with the surrounding biotic and abiotic environment including flow regimes and landscape development (Allan, 2004; Argent and Carline, 2004; Olden et al., 2010; Wang et al., 2001). More specifically, transitions from a natural to disturbed landscape have been observed to decrease abundance and diversity of fish communities (Allan, 2004; Lenat and Crawford, 1994). For these reasons, and those discussed throughout the thesis, alterations in fish communities are strong bio-indicators of environmental change.

The primary ecological consequences of land disturbance on the structure and distribution of fish communities are through altered flow regimes. For aquatic biota, this includes changes in habitat-related structural features such as: discharge regime(s), current velocities, morphological structure of a riverbed and banks, erosion, substratum stability, habitat and siltation (Bunn and Arthington, 2002; Poff and Ward, 1989; ter Braak and Vandonschot, 1995). The impact of these alterations to streams and rivers has been widely recognized as a principal threat to the ecological integrity of river ecosystems through the influence on local habitat, biota, and water quality and quantity in streams and rivers (Allan, 2004; Poff and Allan, 1995; Wang et al., 2001).

After reviewing the literature, two questions were developed (Chapter 2 and 3) to understand how fish communities respond to environmental change and variability across temporal scales in the AOSR, where development has been extensive and is projected to continue.

Question 1: ***Do patterns in fish community composition vary significantly between levels of development, when change is measured by differences in taxonomic composition or functional groups (trophic, reproduction and migratory)?***

To address this question, the research investigated fish community-environmental relationships on a temporal scale, across which community composition could be constrained or altered by development. Structural and trait-based changes in fish community composition were analyzed to determine whether significant variation between levels of development (pre and post) in the AOSR could be attributed to observed changes in fish community metrics. A comprehensive data set was compiled using existing and historical information from the AOSR on fish collected from the Muskeg, Steepbank and Ells River(s). All three rivers were analyzed separately and also from a collective ‘regional’ perspective. Change in fish community structure was first assessed using species richness and alterations in the average relative abundance of taxa. Subsequently, trait measurements included changes in average relative abundance in modes of reproduction, food habits (trophic) and migratory behaviour. No significant difference in community composition patterns was observed between levels of development within the AOSR within each river system; however, there was a significant decline in fish species richness when data were pooled into a broader regional database.

There are several possible reasons for why no significant change in fish community composition patters was observed between pre- and post-development periods. Contributing factors include the limitations of the collected data, including temporal gaps, inconsistent sampling methods, and seasonal sampling inconsistencies addressed further in Chapter 3. Furthermore, the scale of interpretation between individual tributaries and the regional datasets, demonstrated that studies of fish communities on a regional scale can elucidate different states of community change, implying that local controls (i.e., habitat availability, land development, etc.) can play a large role in species presence/absence and measurements of which, should be taken into consideration for future monitoring plans.

Question 2: ***Are there key features of the hydrograph that can explain the greatest amount of variation in patterns in fish community composition between levels of development?***

Initially this chapter was predicated on answering questions to assess the features and patterns of the hydrograph that could best explain variation in fish communities, commonly used as bio-indicators of river ecosystem health, across a gradient of development (pre versus post). Linking together the components of the natural flow regime (considered a dominant hydrological variable in driving many ecosystem and biotic processes within aquatic systems) and changes in fish community composition patterns was constrained due to dataset and subsequent methodological limitations. Chapter three presented a background on In-stream flow Needs (IFN) for the Muskeg River, two studies looking at key factors projected to alter hydrologic conditions and flow regimes in the AOSR, as well as methods and variables used in the literature to quantify the response of fish community composition. Drawing on the conclusion that the attempted methods could not adequately assess the linkage between change in hydrological regimes and resulting fish community structure, this research identified knowledge gaps, dataset limitations and future data collection and monitoring requirements so the initial hypothesis could be tested appropriately. Some of the factors identified to create a more robust dataset include standardized temporal sampling (including seasonal consistencies), standardized field collection methods, the identification of appropriate control sites, and the establishment of baseline data with suitable replication. Furthermore, future monitoring plans need to ensure that the questions being asked allow for a quantitative analysis to examine the relationships between changes in the hydrological regime and fish community structure.

4.2 Recommendations and Future Research

With continued development in the AOSR, there is a need for more standardized and extensive fish community monitoring within these rivers. This research has highlighted where fish community monitoring in the AOSR can be enhanced and integrated into a broader environmental assessment program. Provided below are specific recommendations for future research and monitoring of fish communities in the AOSR:

1) The design of future fish monitoring programs should take into consideration the connections between percent land use and changes in fish community composition patterns. The types of interactions and processes that occur on the landscape within the boundary of the watershed are largely responsible for river system health (Seitz et al., 2011) and ecologists have long recognized that rivers and streams are influenced by the landscapes through which they flow (Hynes 1975, Vannote et al., 1980, c.f. Allan 2004). According to Allan (2004), the variety and variability of habitat are important in influencing the biological diversity of streams and are linked to the larger system and surrounding landscape. Approaches highlighted in Hu et al. (2015) and Guse et al. (2015) where research efforts were aimed at investigating the effects of land use change and climate change on stream-flows and biota, using hydrological models and geographic information systems (GIS) should be taken into consideration to strengthen future monitoring programs.

2) Systematic and standardized quantification of aquatic habitat characteristics of importance in determining fish community structure and distribution need to be integrated into monitoring program designs. For example, aquatic habitat characteristics were not measured as part of the Muskeg River IMF, and were scarcely monitored in other historical programs (e.g., RAMP only collected habitat data on the Muskeg since 2009). Habitat measurements including substrate composition, particularly in areas important for spawning, should be investigated. While not the focus of this paper, other studies testing similar hypotheses (e.g., Armstrong et al., 2000; Bain et al., 1988; Bice et al., 2014) focused on quantifying habitats as a proxy or product of flow regimes to assess how hydrologic change affected fish community composition. These studies used a

range of habitat metrics, from simple visual counts of riffles versus pools to more complex habitat composition observations for physical structures such as brush, roots, etc. (Armstrong et al., 2000). Bain et al. (1988), first quantified representative habitat types within the river system, and subsequently stratified their fish sampling within these defined habitat categories. This extensive sampling included habitat variables: water depth, surface current, velocity and mean velocity, as well as stream substrate variables such as coarseness and heterogeneity. The measurement of such environmental variables is important, especially during spawning seasons, since for many species, habitat plays a very critical role in the reproductive cycle, and ultimately recruitment success as discussed in Chapter 2.

Although not the focus of this research, habitat modification is a prominent topic in river ecology and can reflect changes in the flow regime. This study further emphasizes the need for long-term, comprehensive monitoring efforts, with consistent sampling methods. The need for long-term, comprehensive studies is reinforced by the premise that studying fish communities over relatively short periods of time could lead to erroneous or inconclusive results (Pegg and McClelland, 2004; Schwalb et al., 2014).

3) As with many problems in ecology, monitoring programs with proper controls and replicates are needed to test assumptions underpinning all of the theory and observations. Collecting a standardized baseline of information will be important as development continues. This should include increasing the number of sample sites along the Muskeg River for repeatability and to ensure appropriate spatial and temporal coverage of fish species occupying varying habitat types. Furthermore, future monitoring plans should, given the varying degrees of disturbances and the differing

characteristics of each watershed, take into consideration the diverse responses of community composition patterns between watersheds as well as pre-and post development. Differences in behavior of each of the three rivers of the Lower Athabasca River system, and how they behave from a collective (regional) perspective were highlighted by this research. Enhanced reach-specific (or tributary) fish monitoring (both spatially and temporally) using standardized sampling protocols and the inclusion of landscape and regional measurements, would provide an opportunity to elucidate further ecosystem controls (for example, biotic and abiotic interactions that change with increased landuse).

4) The lack of conclusive findings in this study demonstrates the need for further data collection and exploration. In absence of that, and as development proceeds, environmental thresholds could be set where alteration in fish abundance and community composition in response to both direct and indirect anthropogenic impacts in the AOSR will not be acceptable. Previous studies by Steedman (1988), reported findings of marked fish community changes at 25-50% land use and more recently, Argent and Carline (2004), Weaver and Garmen (1994) and Wang et al., (1997) found similarities in their research where they reported that, as developed land use increased from 10 to 20%. Consideration should be made of using such approaches to establish precautionary thresholds to protect fish community composition and structure in the AOSR. A suggested starting point for future management plans would be to utilize approaches and protocols developed by the Lower Athabasca Regional Plan (LARP) – Land-use framework, and the limitations or thresholds they have established for water quality,

along with supporting and current fish literature, and apply them to fish community thresholds specific to the river systems or region.

5) Standardization of methods for sampling fish communities is needed to detect change. Consistent methodology, including consideration of methods for targeting the varying physiological states and habitat preferences of relevant fish species would produce a more robust and realistic comparison both spatially and temporally. It would further reduce the potential for error in species richness and abundance estimates. Furthermore, standardization of sampling efforts would improve data quality assurance and control across catchments, within river segments or areas being compared to provide a stronger spatial or temporal analysis of alteration of fish communities.

6) Fish-monitoring programs should standardize sampling protocols across all seasons and should take into consideration species-specific migrations, as well as species that spawn seasonally. Sampling across seasons with similar effort would increase the probability of capturing species that can be present or absent seasonally, thus reducing sampling error through a more accurate account of species abundance and richness.

Finally, a question posed by Olden et al. (2010) provides important context related to assessing trait-based community change and improvements that could be considered for future monitoring plans: "*Given the potentially large number of candidate biological traits, but lack of trait data for many fish species in particular regions, which subset of traits are most appropriate for defining functional diversity and offer the most promise for predicting responses to environmental change?"* (Olden et al. 2010, p. 498). This research was only able to utilize a limited number of traits that were consistently measured across sampling events. In moving forward with new monitoring plans, an important question that should be addressed is: *"Are there other traits that are more sensitive to change be more effective for monitoring changes in these communities and can these be measured consistently?"*

As further development within these watersheds is inevitable, knowledge of the specific mechanisms of impairment is critical when attempting to manage and protect systems from anthropogenic pressures (Infante and Allan, 2010). Fish metrics, if selected for sensitivity to disturbance, can provide critical information regarding how habitat alteration results in changes in fish community conditions (Dale and Beyeler, 2001). According to Daniel et al. (2014), fish communities with species sensitive to mining or development can be considered ecological indicators. Specifically, indicators including invertivores, lithophils, tolerant/ intolerant species, fish diversity (H'), and evenness (J') and game species, have all shown high sensitivity to increased mine density, and can collectively provide information about water quality, habitat and food web alteration, community structure and impacts form sedimentation (Olden et al., 2010; Daniel et al., 2014). Future monitoring plans should take these metrics such as these into consideration when deciding on endpoints or field measurements.

Currently, there is an inability to link changes (historical) to hydrologic regimes to land use or development within these systems (Schwalb et al., 2014), and how they have impacted fish communities therein due to the inconsistencies in the methods and sampling during most of the pre-development time span and for a portion of post-development until RAMP started monitoring (2009). Long-term, standardized community monitoring will be critical to gain a greater understanding of how land management practices affect fish communities and what kind of ecosystem management can mitigate impacts to streams and rivers and the biota within them.

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Appendix A. Reproductive Guild Definitions

Table A.1 Definitions for the reproductive guilds attributed to Canadian freshwater fishes found in the Muskeg, Steepbank and Ells River, as listed in Table 2.2. For further, detailed descriptions of these guilds and others, see Balon (1975, 1981).

Guild name	Description
Nonguarders: Open substratum spawners: Pelagophils	Large quantities of non-adhesive, near-neutral or positively buoyant eggs are released and scattered in open water. No parental care of eggs.
Nonguarders: Open substratum spawners: Litho-pelagophils	Eggs are deposited on rocks and gravel, but the eggs, eleutheroembryos, or larvae become sufficiently buoyant to be carried away from the spawning substrate by water currents. No parental care of eggs.
Nonguarders: Open substratum spawners: Lithophils	Deposit eggs on a rock, rubble, or gravel bottom where their embryo and larvae develop. No parental care of eggs.
Nonguarders: Open substratum spawners; Phyto-lithophils	Deposit eggs in relatively clearwater habitats on submerged plants, if available, or on other submerged items such as rocks, logs, or gravel, where their embryos and larvae develop. No parental care of eggs.
Nonguarders: Open substratum spawners: Phytophils	Scatter or deposit eggs with an adhesive membrane that sticks to submerged, alive or dead, aquatic plants, or to recently flooded terrestrial vegetation. Sometimes on logs and branches. No parental care of eggs.
Nonguarders: Brood hiders: Lithophils	Eggs are hidden in specially constructed places. In most cases the hiding places (called redds in salmonids) are excavated in gravel by the female. No parental care of eggs.
Guarders: Nest spawners: Ariadnophils	The nest building male has the ability to spin a viscid thread from a kidney secretion, which binds the nest of different material together. The eggs are guarded and ventilated by the male, who also guards the young once they hatch.
Guarders: Nest spawners: Speleophils	These fishes guard a clutch of eggs in natural holes or cavities, in specially constructed burrows, or where deposited on a cleaned area of the undersurface of flat stones.

Appendix B. Compiled Fish Community Information

Table B.1 Muskeg River fish community information collected from literature including location description, species present, season samples were taken, methods used to collect samples and source of the data.

Development	Year	Location Description	Species	Season*	Method	Source
PRE DEVELOPMENT	1973	Hoop net operated 17.5 km upstream from the mouth of the river	ARGR LNSC NRPK WHSC ARGR	Winter / Spring	Hoop net	Fedoruk, A.N. 1973. Supplementary Ecological Baseline Measurements of Tar Sands Lease C-13, Athabasca Tar Sands, Alberta, Canada. Prepared by Lombard North Group.
	1976	Fish fence operated near the river mouth - Upstream	BURB LKWH LNSC MNWH NRPK WALL WHSC	Winter / Spring	Fish fence	Bond, W.A., and K. Machniak. 1977. Interim report on an Intensive Study of the Fish Fauna of the Muskeg River Watershed of Northeastern Alberta. Prepared by the Department of Fisheries for the Alberta Oil Sands Environmental Research Program. AOSERP Project AF 4.5.1.
		Fish fence operated near the river mouth - downstream	ARGR BURB LKWH LNSC MNWH NRPK WALL WHSC	Winter/ Spring/ Summer	Fish fence	Bond, W.A., and Machniak, K. 1979. An intensive study of the fish fauna of the Muskeg River watershed of Northeastern Alberta. AOSERP Report 76.
	1977	Fish fence operated near the river mouth - upstream	ARGR BLTR BURB CISCO LKWH LNSC MNWH NRPK WALL WHSC	Winter / Spring	Fish fence	Bond and Machniak,1977.

		Fish fence operated near the river mouth - downstream	ARGR LKWH LNSC MNWH NRPK WALL WHSC	Winter / Spring	Fish fence	Bond and Machniak,1977
	1979	Reach 1 - (Km 0 to Km 0.5)	ARGR LKCH LNDC PRDC SLSC SLSC TRPC YLPR	Fall	Seine Combination of methods	Sekerak, A.D., and G.L. Walder. 1980. Aquatic Biophysical Inventory of Major Tributaries in the AOSERP Study Area, Volume I: Summary Report. Prepared for the Oil Sands Environmental Research Program by LGL Limited, Environmental Research Associates. AOSERP Report 114. c.f. Griffiths, W.E. 1973. Preliminary Fisheries Survey of the Fort McMurray Tar Sands Area. Fish and Wildlife Division, Alberta Department of Lands and Forests. and Walder, G.L., P.L. Strankman, E.B. Watton, and K.A. Bruce. 1980. Aquatic Biophysical Inventory of Major Tributaries in the AOSERP Study Area: Volume II, Atlas. Prepared for the Oil Sands Environmental Research Program by LGL Limited, Environmental Research Associates. AOSERP Project WS 3.4.
		Reach 2 - (Km 0.5 to Km 9.0)	ARGR LNDC PRDC SLSC	Fall	Seine	
		Reach 2 - (Km 0.5 to Km 9.0)	WHSC	Fall	Electrofisher	
		Reach 2 - (Km 0.5 to Km 9.0)	ARGR	Fall	Electrofisher	Bond and Machniak 1979
		Reach 2 - (Km 0.5 to Km 90)	SLSC	Fall	Kick Sample	
		Reach 2 - (Km 0.5 to Km 9.0)	ARGR PRDC	Fall	Seine	Sekerak and Walder, 1980, cited from Griffiths, 1973 and Walder et al 1980

			SLSC WHSC			
		Reach 2 - (Km 0.5 to Km 9.0)	LNDC PRDC TRPC WHSC	Fall	Seine	Bond and Machniak, 1979
		Reach 2 - (Km 0.5 to Km 9.0)	ARGR BURB DLVR CISCO LKWH LNSC MNWH NRPK WALL WHSC	Fall	Counting Fence	
		Reach 2 - (Km 0.5 to Km 9.0)	ARGR BURB FTMN LKCH LKWH LNDC LNSC MNWH NSST NRPK PRDC SLSC TRPR WHSC YLPR	Fall	Combination of methods	
		Reach 3 - (Km 9.0 to Km 16.5)	PRDC	Fall	seine	Sekerak and Walder, 1980, cited from Griffiths, 1973 and Walder et al, 1980

		Reach 3 - (Km 9.0 to Km 16.5)	NRPK	Fall	angling	
		Reach 3 - (Km 9.0 to Km 16.5)	LNSC NRPK PRDC	Fall	Electrofishing	Bond and Machniak, 1979
		Reach 3 - (Km 9.0 to Km 16.5)	ARGR BRST LNDC NRPK WHSC	Fall	Seine	Sekerak and Walder, 1980, cited from Griffiths, 1973 and Walder et al, 1980
		Reach 3 - (Km 9.0 to Km 16.5)	ARGR WHSC	Fall	Seine	Bond and Machniak, 1979
		Reach 3 - (Km 9.0 to Km 16.5)	LNSC PRDC	Fall	Seine	Sekerak and Walder, 1980, cited from Griffiths, 1973 and Walder et al, 1980
		Reach 3 - (Km 9.0 to Km 16.5)	PRDC SLSC	Fall	Electrofishing	
		Reach 3 - (Km 9.0 to Km 16.5)	ARGR LKCH WHSC	Fall	Seine	Bond and Machniak, 1979
	1981	Km 5.5 above creek mouth (Jackpine Creek)	ARGR LNSC NRPK WHSC	Spring	Hoop trap/fence	O'Neil, J., L. Noton, and T. Clayton. 1982. Aquatic Investigations in the Hartley Creek Area, 1981 (Sand Alta Project).
		Km 14.2 above creek mouth 1981	ARGR LNSC NRPK WHSC	Spring	Hoop trap/fence	
	1995	16.5 Km	ARGR	Spring	Fish fence	

		upstream of the river mouth - Upstream	LNSC NRPK WALL WHSC			Golder (Golder Associates Ltd.). 1996a. Aquatic Baseline Report for the Athabasca, Steepbank and Muskeg Rivers in the Vicinity of the Steepbank and Aurora Mines. Prepared for Suncor Inc., Oil Sands Group.
		16.5 Km upstream of the river mouth - Downstream	ARGR LNSC NRPK WHSC	Spring	Fish fence	
		16.5 Km upstream of the river mouth - Upstream	ARGR	Fall	Fish fence	
		16.5 Km upstream of the river mouth - Downstream	ARGR LNSC NRPK WHSC	Fall	Fish fence	
POST DEVELOPMENT	1996	FWIMS 12 463430N 6332390E (UTM)	LNSC MNWH WALL WALL	Spring	Sample Angling	Fish and Wildlife Management Information System (FWIMS). Available from: http://esrd.alberta.ca/fish-wildlife/fwmis/access-fwmis-data.aspx
	1997	FWIMS 12 463417N 6332366E (UTM)	WALL	Spring	Sample Angling	
		FWIMS 12 465357, 6338708 (UTM)	LKCH WHSC	Fall	Electrofishing	
		FWIMS 12 466412 N 6339829 E (UTM)	LKCH WHSC	Fall	Electrofishing	
		FWIMS 12 466479 N 6339865 E(UTM)	LKCH WHSC	Summer	Electrofishing	
		FWIMS 12 468973 N	ARGR LKCH	Summer	Electrofishing	

		6344046 E (UTM)	WHSC			
		FWIMS 12 469230 N 6344227 E(UTM)	LNSC	Spring	Electrofishing	
		FWIMS 12 469751 N 6344780 E (UTM)	LKCH LNSC WHSC	Summer	Electrofishing	
	1998	Fish Fence Data RAMP	ARGR FTMN LNSC NRDC SPSC MNWH WHSC	Spring	Fish fence	RAMP
		FWIMS 12 465398 6322901 (UTM)	ARGR NRPK	Spring	Sample Angling	FWIMS
		FWIMS 12 469894 N 6345293 E (UTM)	LKCH	Spring	Electrofishing	
	1999	FWIMS 12 463881 N 6332272 E (UTM)	BURB LNSC PRDC SLSC	Fall	Electrofishing	
		FWIMS 12 463938 N 6332076 E (UTM)	BURB LNDC LNSC PRDC SLSC TRPR	Fall	Electrofishing	
		FWIMS 12 465442 N 6338295 E (UTM)	LNDC LNSC PRDC SLSC SLSC TRPR	Fall	Electrofishing	

		FWIMS 12 465490 N 6338800 E (UTM)	ARGR PRDC SLSC SLSC TRPR	Fall	Electrofishing	
	2000	FWIMS 12 463884 N 6332252 E (UTM)	LKCH LNDC PRDC SLSC	Fall	Electrofishing	
		FWIMS 12 465218 N 6334403 E (UTM)	LNSC MNWH NRPK WALL WHSC	Spring	Electrofishing	
		FWIMS 12 465218 N 6334403 E (UTM)	LNSC NRPK WALL WHSC	Spring	Electrofishing	
		FWIMS 12 465481 N 6338797 E (UTM)	LNSC NRPK TRPR WALL WHSC WHSC	Spring	Electrofishing	
		FWIMS 12 465501 N 6338940 E (UTM)	LKCH LNDC LNSC PRDC SLSC	Fall	Electrofishing	
		FWIMS 12 465515 N 6338806 E (UTM)	LKWH LNSC NRPK WALL WHSC	Spring	Electrofishing	
		FWIMS 12 467355 N 6342047 E (UTM)	LKCH LNSC WHSC	Spring	Electrofishing	

		FWIMS 12 467355 N 6342047 E (UTM)	LNSC WHSC	Spring	Electrofishing	
	2001	Fish Fence Data RAMP Spring 2001	BRST LNSC NRPK WHSC	Spring	Fish Fence	RAMP
	2001	FWIMS 12 465469 N 6338138 E (UTM)	LKCH LNSC NRPK TRPR WHSC	Summer	Electrofishing	FWIMS
	2002	FWIMS 12 465148 N 6337613 E (UTM)	WHSC	Summer	Test Net	FWIMS
	2002	FWIMS 12 465263 N 6332533 E (UTM)	LKCH LNSC MNWH TRPR WALL WHSC	Summer	Electrofishing	FWIMS
	2002	FWIMS 12 465148 N 6337613 E (UTM)	WHSC	Summer	Test Net	FWIMS
	2002	FWIMS 12 465280 N 6332656 E (UTM)	GOLD LKCH LNSC NRPK TRPR WHSC	Summer	Electrofishing	FWIMS
	2002	FWIMS 12 465433 N 6338955 E (UTM)	TRPR	Fall	Electrofishing	FWIMS
	2002	FWIMS 12 465440 N 6338953 E (UTM)	LKCH LNDC LNSC MNWH	Summer	Electrofishing	FWIMS

			SPSC WALL WHSC			
		FWIMS 12 466501 N 6340000 E (UTM)	LNSC WALL WHSC	Summer	Electrofishing	
		FWIMS 12 467615 N 6341926 E (UTM)	NRPK	Fall	Electrofishing (BOAT)	
	2003	Fish Fence Data RAMP	ARGR LNSC MNWH NRPK WALL WHSC	Spring	Fish Fence	RAMP
		FWIMS 12 464048 N 6332069 E (UTM)	ARGR LKWH LNSC MNWH NRPK WALL WHSC	Spring	Trap nets, Minnow Traps, Dip Nets	FWIMS
	2004	FWIMS 12 463511 N 6332462 E (UTM)	SPSC	Fall	Electrofishing	
		FWIMS 12 463693 N 6332507 E (UTM)	SPSC	Summer	Electrofishing	
		FWIMS 12 463693 N 6332507 E (UTM)	SLSC	Fall	Electrofishing	
		FWIMS 12 466624 N 6340240 E (UTM)	NRPK PRDC WHSC	Summer	Electrofishing	
		FWIMS 12 467688 N	LNSC NRPK	Summer	Trap nets, Minnow Traps,	

		6341943 E (UTM)	PRDC		Dip Nets	
		FWIMS 12 467688 N 6341943 E (UTM)	LNSC NRPK TRPR WHSC	Summer	Electrofishing	
	2005	FWIMS 12 464244 N 6332021 E (UTM)	EMSH LKCH MNWH NRPK PRDC SPSH TRPR WHSC	Spring	Electrofishing	
	2006	FWIMS 12 463711 N 6332511 E (UTM)	SLSC SPSC	Summer	Electrofishing	
		FWIMS 12 463831 N 6332409 E (UTM)	SLSC	Fall	Electrofishing	
		Fish Fence Data RAMP	ARGR LKWH LNSC MNWH NRPK WALL WHSC	Spring	Fish Fence	RAMP
	2009	Fish Fence Data RAMP	ARGR LKWH LNSC NRPK WALL WHSC	Spring	Fish Fence	
		MUR-F1**	BRST BURB LKCH LNSC SLSC	Fall	Electrofishing	

			SPSC			
		MUR-F2†	ARGR	Fall	Electrofishing	
		FWIMS 12 463448 N 6332406 E (UTM)	SLSC	Summer	Electrofishing	FWIMS
		FWIMS 12 463511 N 6332462 E (UTM)	BRST BURB LKCH LNSC SLSC SPSH WHSC	Fall	Electrofishing	
		FWIMS 12 466312 N 6339503 E (UTM)	ARGR	Fall	Electrofishing	
	2010	MUR-F1**	BRST BURB FNDC LKCH LNDC LNSC PRDC SLSC SPSC WALL WHSC YLPR	Fall	Electrofishing	RAMP
	2011	MUR-F1**	BRST LKCH LNDC LNSC PRDC SLSC WALL WHSC	Fall	Electrofishing	

		MUR-F2†	NRPK WHSC	Fall	Electrofishing	
		MUR-F3 ‡	BRST PRDC	Fall	Electrofishing	
		FWIMS 12 463511 N 6332462 E (UTM)	BRST LKCH LNDC LNSC PRDC SLSC WALL WHSC WHSC	Fall	Electrofishing	FWIMS
	2012	MUR-F1**	LNDC NRPK YLPR SLSC	Fall	Electrofishing	RAMP
	2012	MUR-F3 ‡	BRST	Fall	Electrofishing	RAMP
	2013	MUR-F3 ‡	WHSC	Fall	Electrofishing	RAMP
	2013	MUR-F2†	LKCH LNSC NRPK PRDC WHSC	Fall	Electrofishing	RAMP
	2013	MUR-F1**	BURB LKCH LNSC NRPK SLSC	Fall	Electrofishing	RAMP

			SPSC WHSC YLPR				

* Seasons as per RAMP: Winter = January, February, March, and April; Spring = May, June; Summer = July, August; Fall = September, October, and November
** MUR-F1 upstream 12 463718N 6332499E; downstream 12 463543N 6332450E
† MUR-F2 upstream 12 466553N 6340424E; downstream 12 466399N 6340037E
‡ MUR-F3 upstream 12 479786N 6357048E; downstream 12 479743N 6357048E
FWIMS
RAMP