University of Alberta

The effects of culverts on upstream fish passage in Alberta foothill streams.

by

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A thesis submitted to the Faculty of Graduate Studies and Research
in partial fulfillment of the requirements for the degree of

Master of Science

in

Wildlife Ecology and Management

Renewable Resources

Fall 2011

Edmonton, Alberta

Abstract

In the Rocky Mountain foothills of Alberta, Canada, activities of the forestry and energy sectors have resulted in the installation of tens of thousands of stream-crossing structures. In fifteen Athabasca River basins I found that culverts impeded upstream movements of non-sportfish species relative to reference bridge sites. Conversely, abundances of Rainbow Trout significantly increased upstream of culverts. I suggest that culverts that exclude Burbot, a voracious predator, or high temperatures above culverts allow for increased productivity of Rainbow Trout. Water quality and substrate composition did not noticeably change upstream and downstream of bridges, while culverts had significantly higher water temperatures and silt/sand upstream. In evaluating the effectiveness and temporal biases of common sampling techniques, I found that backpack electrofishing and angling had the highest Arctic grayling detection probabilities. Angling detected larger juvenile and adult fish (>110 mm), while young-of-the-year were more easily detected using backpack electrofishing in later summer.

Acknowledgements

I thank Shannon Stambaugh, Stephen Spencer, Don ‘the Fish’ Hildebrandt, Owen Watkins, Troy Furukawa and Sierra Sullivan for all their hard work in field sampling. In addition, the 2009 field crew proved that three, sometimes four people and a dog could all coexist happily 24/7 in a small trailer.

The ‘bumps’ along the road to completion of this study were made a lot easier given the continued encouragement, advice and support of my amazing graduate committee, Dr. Lee Foote, Dr. Michael Sullivan, Dr. Cameron Stevens and Dr. Peter Aku. A special thank you to Stephen Spencer and David Park for additional guidance in project design and continuously challenging and motivating conversations. A thank you to my graduate lab mates who were always fantastic resources for all things thesis related and especially knowledgeable when it came to statistics.

This study would not have been possible without the generous funding support and collaboration with the Alberta Conservation Association and Alberta Sustainable Resource Development. Further funding was provided by NSERC and the University of Alberta.

Last, but never least, I am indebted to my family, boyfriend and friends for their support in times when completing my graduate degree seemed like a frustratingly insurmountable task.

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Chapter 1 - General Introduction

Culverts are ubiquitous anthropogenic features in lotic systems worldwide. In Alberta, large road networks are necessary to support rapidly expanding forestry and energy sectors; consequently, culverts continue to be installed at road-stream crossing intersections throughout the province. Current provincial and federal management practices and legislation recognize the importance of mitigating the negative impacts of watercourse crossings on streams and aquatic organisms, yet it is only in recent years that the impacts of artificial barriers and the lack or remediation efforts have come under closer scrutiny (e.g. Warren and Pardew 1998, Harper and Quigley 2000, Park 2006). While bridges are generally considered ecologically benign structures (Warren and Pardew 1998, Poplar-Jeffers 2005, Park et al. 2008), culverts can cause habitat loss and degradation by altering the streambed and stream hydraulics, increasing sedimentation and erosion, and may restrict organism movements (Warren and Pardew 1998, Harper and Quigley 2000, Park et al. 2008, Burford et al. 2009).

Unlike terrestrial species, stream dispersers are restricted to aquatic systems and cannot move between distant stream patches without first passing through the entire series of stream patches (Fagan 2002). As a result, aquatic ecosystems may be less resilient to reduced levels of stream connectivity (Fagan 2002, Cote et al. 2009). Watercourse crossings must allow the movement of processes and organisms throughout the watershed. Fragmentation of stream habitats by culvert crossings may cause declines in fish populations by reducing fish dispersal behaviour among the stream habitats necessary to maximize their fitness and ensure long-term persistence (Faush and Young 1995, Jungwirth et al. 2000). Although movement scales and patterns vary among different fish species, in general, at a small scale, dispersal allows them to seek thermal refugia (e.g. Kaeding 1996), avoid predators (e.g. Harvey 1991) and access foraging areas (e.g. Clapp et al. 1990). On larger scales, movements can allow access to spawning and overwintering habitats (e.g. Isaak et al. 2007) and maintain genetic variability within a population (e.g. Morita and Yamamoto 2002). Indeed, under the federal *Fisheries Act* 'habitat' is defined as "spawning grounds and nursery, rearing, food supply and migration areas on which fish depend directly or indirectly to carry out their life processes". Legally, watercourse crossings must maintain connectivity to all portions of fish habitat. While the full ecological effects caused by anthropogenic barriers are not entirely understood, the number of potential barriers to fish migration raises important questions regarding the cumulative impacts of road crossings on fish population persistence.

Several culvert parameters can restrict fish movements from downstream to upstream of culverts. Physical factors include: hang height (i.e. the physical drop or distance from the culvert outlet to the stream below) (Mueller et al. 2008, Burford et al. 2009, Norman et al. 2009), outlet plunge pool depth (Mueller et al. 2008), high water velocities caused by stream channel constriction or high culvert slopes (Belford and Gould 1989, Warren and Pardew 1998, Macdonald and Davies 2007, Burford et al. 2009), and length (Dubé and Gravel 1980).

A physical drop (hang height) at a culvert outlet will obstruct upstream fish passage if it exceeds their swimming and jumping abilities (Mueller et al. 2008, Burford et al. 2009, Norman et al. 2009). A sufficiently deep pool below the culvert outlet may facilitate passage of certain fish species by increasing their jumping abilities (Lauritzen 2002, Mueller et al. 2008). Improperly installed and undersized culverts constrict stream flow, increase stream velocities and cause sedimentation and scouring at the culvert outlet (Tchir et al. 2004, Park 2006). Often the diameter of the installed culvert is small relative to the upstream stream channel width (Tchir et al. 2004, Park 2006), consequently, water is forced through culverts at higher water velocities. In turn, this can lead to scouring at the culvert outlet (Park 2006). As improperly installed and/or maintained culverts age, scouring at the outlet only intensifies the physical drop (hang height) (Park 2006). Excessive water velocities are often cited as a major factor inhibiting fish movements through culverts (Warren and Pardew 1998). Water velocity through culverts may affect fish swimming distance and frequency (Toepfer et al. 1999), and at higher water velocities the likelihood of fish passage through a crossing is reduced because of increased energetic stress (Adams et al. 2000). Long streamlined, hydraulically smooth culverts may also prevent fish from passing upstream if they can not sustain the necessary swimming time to navigate the entire culvert distance (Belford and Could 1989, Macdonald and Davies 2007).

Fish species distributions can also be dictated by their tolerance to the surrounding water quality, stream substrates and substrate embeddedness (Lyons 1992, CCME 2001). Sedimentation in streams has been known to decrease the quality of spawning habitat, reduce spawning activity and contribute to greater egg and fry mortality by filling the interstitial spaces between substrate that are necessary to provide oxygen (Ryan 1991, Argent and Flebbe 1999). Higher sedimentation at watercourse crossings may degrade habitat important to litho-obligate species that are 'clean' gravel and cobble spawners (Lyons 1992, Haskins and Mayhood 1997).

The permeability of culvert barriers will vary by fish species (Warren and Pardew 1998, Poplar-Jeffers 2005). While some culverts may allow movement, others may act to partially or entirely restrict fish movements (Warren and Pardew 1998, Park et al. 2008, Burford et al. 2009). The fragmentation of stream habitat may result in immediate effects for some fish species, while for others, the adverse effects may emerge gradually as the habitat is altered and population dynamics shift (Wofford et al. 2005, Park 2006). Fragmentation can result in smaller fish populations that are more vulnerable to local extirpations from reduced genetic diversity and local stochastic events (Morita and Yamamoto 2002, Park 2006). Culverts can cause the loss of genetic diversity, loss of resident fish species that seasonally migrate upstream, and changes in fish community assemblages exacerbated by disturbance events (e.g. drought, fire, winter kill) which may leave stream fragments uncolonized (Robison et al. 1999, Park 2006).

Despite the federal *Fisheries Act* definition of 'fish' as all fish species at all life stages, industrial representatives, scientists and managers alike continue to prioritize the conservation and persistence of sportfish species. For instance, the scientific literature often focuses on salmonid fish passage studies (e.g. Burford et al. 2009). Given the complexity of stream communities, the exclusion of non-sportfish species is a dangerous mentality since it does not stress the ecosystem context necessary to ensure the long-term persistence of 'healthy' stream systems (Sylte 2002, Park 2006). With growing emphasis on the importance of maintaining biodiversity, there is a need to adopt a scientific approach that acknowledges the roles of all species in contributing to aquatic ecosystem 'health' and integrity (Sylte 2002).

In this thesis, I identify how culverts are impeding upstream passage of stream fish. Due to ineffective sampling of Arctic Grayling, I also develop a monitoring protocol to effectively sample declining Arctic Grayling populations. In chapter 2, I identify culvert parameters affecting instream movements of several common Alberta foothill sportfish and non-sportfish species in Athabasca River tributaries. I also present the findings from an extensive watercourse crossing inventory of fifteen Athabasca River basins. In chapter 3, I examine sampling limitations identified in Chapter 2, to ascertain temporal and gear biases of common sampling techniques used to detect and determine abundance estimates of low-density Arctic Grayling populations in wadeable tributary streams. I also create a management decision framework to aid in effective monitoring of Arctic Grayling populations. In summary, I highlight key findings and present recommendations for future research.

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Chapter 2 - Effects of Culverts on Stream Fish Assemblages in the Alberta Foothills[1]

Introduction

Watercourse crossing structures are ubiquitous anthropogenic features across human modified landscapes (e.g Tchir et al. 2004, Park et al. 2008). In the Rocky Mountain foothills of west-central Alberta, Canada, activities of the forestry and energy sectors have resulted in the construction of large road networks and the installation of tens of thousands of stream-crossing structures (Tchir et al. 2004, Park et al. 2008). Culverts are the dominant stream crossing structures in the region and were previously used solely as a means to redirect water quickly and efficiently, with little regard for the impacts on the surrounding stream habitat and biota (Tchir et al. 2004). Although attitudes have shifted and current practices and legislation recognize the importance of mitigating the impacts of watercourse crossings, low-cost culverts continue to be widely installed despite their known negative effects, particularly on stream fish movements (e.g. Warren and Pardew 1998, Harper and Quigley 2000, Norman et al. 2009). Bridges, which are more expensive to install and less common in the Rocky Mountain foothills, are generally considered ecologically benign structures because they often retain the natural streambed, bank integrity and sinuosity (Warren and Pardew 1998, Pluym et al. 2008).

Culverts can cause immediate and long-term effects on fish populations by altering habitat characteristics such as stream hydrology, water quality, substrate composition, and by fragmenting fish habitat and impeding movements necessary for growth, survival, reproduction, gene flow and colonization (Warren and Pardew 1998, Harper and Quigley 2000, Morita and Yamamoto 2002, Park et al. 2008, Burford et al. 2009). Physical factors of culverts that influence fish movements, particularly passage from downstream to upstream locations include: hang height (i.e. the physical drop or distance from the culvert outlet to the stream below) (Mueller et al. 2008, Burford et al.

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[1] A version of this manuscript has been submitted to the *North American Journal of Fisheries Management*. Corresponding styles and format apply to citations, figures and tables.

2009, Norman et al. 2009), outlet plunge pool depth (Mueller et al. 2008), high water velocities caused by stream channel constriction or high culvert slopes (Belford and Gould 1989, Warren and Pardew 1998, Macdonald and Davies 2007, Burford et al. 2009), and length (Dubé and Gravel 1980). A physical drop (hang height) at a culvert outlet will obstruct upstream fish passage if it exceeds fish swimming and jumping abilities (Mueller et al. 2008, Burford et al. 2009, Norman et al. 2009). A sufficiently deep pool below the culvert outlet may facilitate passage of certain fish species by increasing their jumping abilities (Lauritzen 2002, Mueller et al. 2008). High water velocities and steep culvert slopes typically obstruct passage where they surpass the physiological and behavioural capabilities of fish species (Belford and Gould 1989, Haro et al. 2004). Similarly, long streamlined, hydraulically smooth culverts may also prevent fish from passing upstream if they can not sustain the necessary swimming time to navigate the entire culvert distance (Belford and Could 1989, Macdonald and Davies 2007).

Previous research has focused on fish passage for recreationally valuable trout species (e.g. Burford et al. 2009). In this study, I provide a broader examination of how watercourse crossings influence several stream fish species. The main objective of my study was to determine if culvert features influence upstream abundance of several Alberta stream fish species in the Athabasca River basin: Burbot (*Lota lota*) (Linnaeus), White Sucker (*Catostomus commersoni*) (Lacepède), Longnose Sucker (*Catostomus catostomus*) (Forster), Lake Chub (*Couesius plumbeus*) (Agassiz), Pearl Dace (*Margariscus margarita*) (Cope), Spoonhead Sculpin (*Cottus ricei*) (Nelson), Arctic Grayling (*Thymallus arcticus*) (Pallas), and Rainbow Trout (*Oncorhynchus mykiss*) (Walbaum). I predicted that culverts would more severely affect upstream abundances of weaker swimming fish species (i.e. Burbot, Spoonhead Sculpin and minnow species) by acting as partial or complete barriers to movement. I predicted however, that the upstream passage of stronger swimming species (i.e. suckers, Rainbow Trout and Arctic Grayling) would only be partially obstructed. Upstream-downstream trends in fish abundance may not be determined entirely by the ability of a fish species to traverse a crossing structure, but are likely a function of many interacting and potentially confounding factors. For instance, site-scale physical habitat characteristics such as water quality, water depth, velocity and substrate can also regulate where fish occur (Edwards 1983, Raleigh et al. 1984, Lyons et al. 1996). Thus, a secondary objective of this study was to identify poorly maintained and improperly installed culverts in the Rocky Mountain foothills and determine whether there were differences in stream habitat characteristics between upstream and downstream locations. Given the large number of culverts that may be acting as potential barriers in Alberta foothill watersheds, it is important for fisheries managers to understand how stream fish species respond to characteristics of culverts and how watercourse crossings generally alter adjacent stream habitat in order to make informed regulatory and planning decisions.

Study Site

My study was in tributaries of the Athabasca River in the Foothills ecoregion, near the towns of Whitecourt, Edson and Hinton in west-central Alberta (Figure 2.1) (NRC 2006). Located in the Lower Foothills and Upper Foothills subregions (NRC 2006), study sites extended as far southwest as Pinto Creek near Hinton (53˚36'N, 118˚05'W), and as far northeast as the Freeman River drainage near Swan Hills (54˚30'N, 115˚23'W) (Figure 2.1). The Foothills natural region receives high annual precipitation and is characterized by cold winters and short warm summer (Strong and Leggat 1992, NRC 2006). The maximum July temperature from 2007 to 2009 in my study area ranged from 29.7 to 30.6°C (Environment Canada 2011). The majority of study sites sampled for fish occurred in the Freeman River (n=36) and Sakwatamau River (n=37) drainages near the city of Whitecourt (Table 2.1). Typically, average discharge of the Athabasca River near study sites ranged from 150 m^3/s to 600 m^3/s from May to August and peaked during the spring freshet (600m^3/s) (Alberta Environment 2010). Industrial land-use activities in this area include timber harvest, oil and gas exploration and extraction. Recreational activities include hunting, fishing, camping and all terrain vehicle use. Study sub-basins varied in topographical features and industrial activity; however, watercourse crossing structures were widespread throughout all sub-basins.

Table 2.1 Number of sites angled, electrofished, and egg kick surveyed in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

Sub-basin	Number of culvert sites assessed	Number of bridges assessed	Number of culverts electrofished	Number of bridges electrofished
Freeman river	76	29	25	11
Sakwatamau river	96	21	21	16
Chickadee creek	6	0	4	0
Windfall creek	27	10	16	0
Canyon creek	1	0	1	0
Baseline creek	1	0	1	0
Wolf creek	2	4	2	1
Embarrass river	0	1	1	1
Pinto creek	3	1	1	0
Oldman creek	1	4	1	1
Sundance creek	2	3	1	2
Marsh head creek	1	3	1	0
Unnamed creek	1	0	1	0
Prest creek	1	0	1	0
Emerson creek	1	0	1	0
Total	219	76	78	32

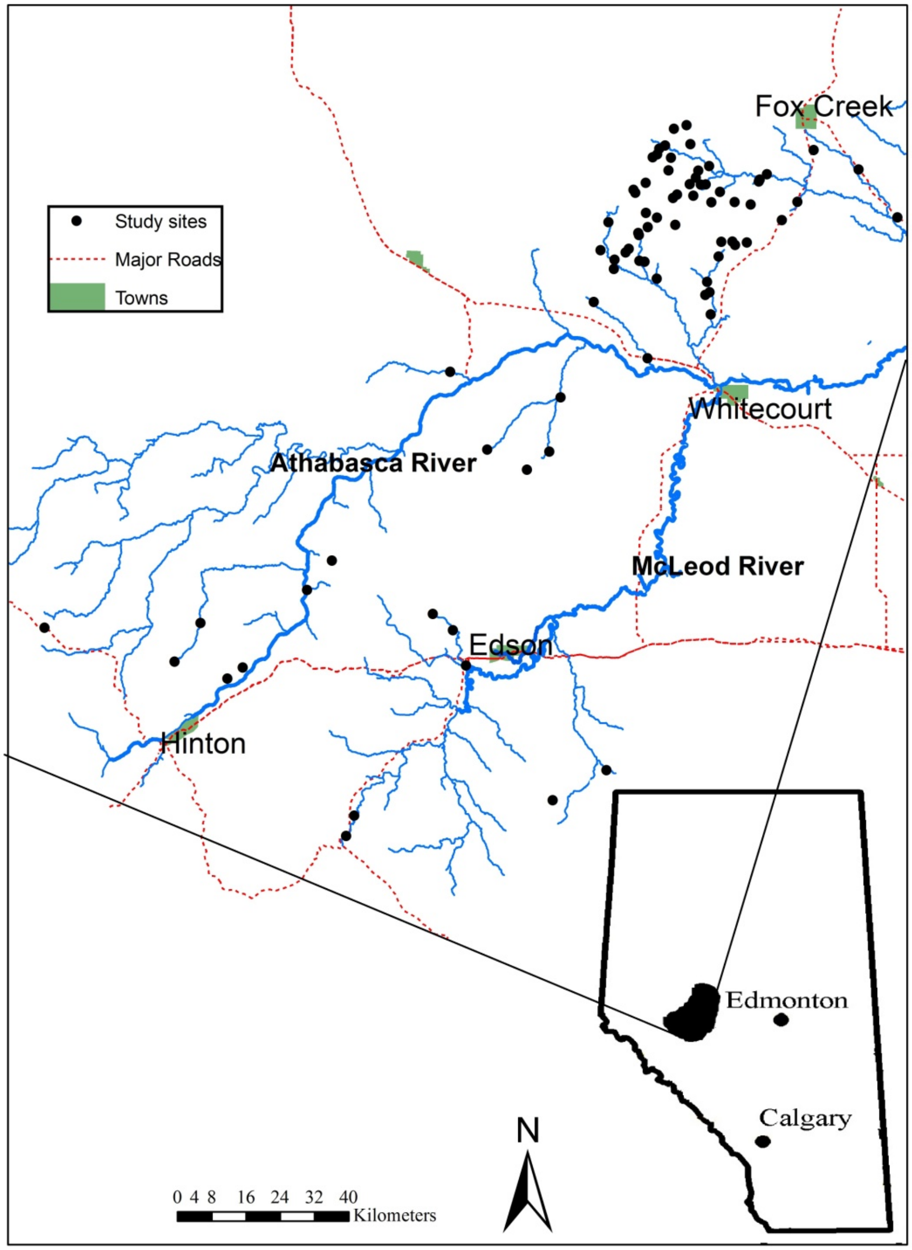

Figure 2.1 Map of Athabasca River basin, 2008 and 2009 study sites.

Methods

Field surveys and GIS analysis

I selected road-stream crossing study sites and assigned them a unique ID number using ArcView 9.3 and geographic information system (GIS) data for west-central Alberta. A total of 295 watercourse crossing sites in 15 different sub-basins were selected where I performed physical and habitat assessments in the summer and early fall (May-October) of 2007, 2008 and 2009 (Table 2.1). There were 76 crossing sites with bridges, and 219 with one culvert or multiple culverts. I assessed a total of 302 culverts. I did not perform assessments on culverts in drainage ditches and first-order (Strahler 1964) ephemeral streams with very low fish potential (i.e. sites that were nearly dry and without a visible channel) (Park 2006). I performed extensive assessments in the Freeman River, Chickadee Creek, Sakwatamau River and Windfall Creek. In the remaining sub-basins I assessed stream crossings that were sampled for fish, including all downstream watercourse crossing structures (Table 2.1).

I performed physical and habitat assessments of watercourse crossing structures and the surrounding stream habitat in accordance with the Foothills Research Institute stream crossing inspection manual (McCleary et al. 2007). I assessed fish passage parameters by determining culvert type (i.e. pipe, or open-bottomed arch), culvert diameter (m), hang height (m), outlet plunge pool depth (m), and outlet drop (m). Additionally, I determined the slope (%) and length (m) of the culvert using a TruPulse Laser Range Finder. For further information bridge and culvert assessment protocols refer to McCleary et al. (2007).

I measured average water velocity (m/s) at the culvert outlet and underneath bridge crossings using a digital velocity meter, and then measured bankfull channel width (m) and cross-sectional water depths at 50-meters upstream of crossings to avoid the influence of culverts on stream morphology (McCleary et al. 2007). I visually separated the composition of stream substrates into four categories: silt and sand (<2 mm), gravel (2 mm-64 mm), cobble (64 mm-256 mm), and boulder (> 256 mm) (Scrimgeour et al.

2008) every 50-meters downstream and upstream and at the culvert inlet and outlets or underneath the bridge for the entire 600-meters study reach. At the same stream locations I visually evaluated substrate embededdness (the degree to which larger particles are covered with finer particles) as a measure of sedimentation (Platts et al. 1983). A level of embeddedness was assigned into five categories, where five is 0 to <5% sand between substrate and one is > 75% sand between substrate (Platts et al. 1983). At 150-meters downstream and upstream locations, I took measurements of dissolved oxygen (mg/L), pH, and temperature (°C).

Fish sampling

To sample for stream fish I selected wadeable streams that were crossed by roads with culverts or bridges, and where the stream size was sufficient to support relatively high fish abundances (Norman et al. 2009). Of the assessed watercourse crossing sites, I sampled for fish at 110 sites (32 bridge and 78 culverts crossing sites). Of 51 culvert-crossing sites where I recorded fish, three sites were re-visited in a different study year. I excluded five sites (two bridges and three culverts) from analysis because it was later determined that the streams were too deep to effectively backpack electrofish (i.e. the catchability was obviously low).

Using a model 12 or LR24 Smith-Root backpack electrofishing unit, the experienced three-person crew used a single-pass of electrofishing (Kruse et al. 1998) to sample 300-m downstream, followed by 300-m upstream of stream crossing structures (ASRD 2008). Fish sampling typically occurred the same day as habitat and crossing assessments. Electrofishing proceeded from downstream to upstream to avoid silting the stream and to maximize catchability. I deferred electrofishing if stream flows or turbidity were elevated and may have affected fish capture. One person operated the electrofisher, while two people netted fish. I made an effort to sample all stream habitat types (i.e under cover, riffle, run, pools). No block nets were used (Bouska and Paukert 2009). Voltage, frequency, and duty cycle were adjusted to maximize capture efficiency without injuring any fish species (settings range: voltage = 400 – 500 V, frequency = 40 – 60 Hz, duty cycle = 20 - 25%). Stream reaches encompassed several pool-riffle-run sequences to reflect several habitat types. I identified all captured fish to species and fork length was measured to the nearest millimeter. I then returned fish to their respective sections of the stream. Fish identifications were based primarily on the taxonomic descriptions in Nelson and Paetz (1992).

At study sites sampled for fish, I also angled with dry flies. Combined with backpack electrofishing, this method allowed me to effectively sample for all Arctic Grayling life history stages (MacPherson et al. 2011 in prep.). Angled sites were sampled with a three-person crew, one experienced (> 5 years experience) angler and two moderately experienced anglers (2-5 years experience). We sample angled with fly rods and dry flies. Fly sizes ranged from 8 to 14. Arctic Grayling are opportunistic feeders that are highly susceptible to angling (Nelson and Paetz 1992; ASRD 2005) and typically do not show a preference for specific flies (Sullivan and Johnson 1994). We fished with stimulators, humpies and elk hair caddis fly types. We normally used 5X tippets (diameter 0.152 mm) with a leader length of 2.1-2.7 meters. Dry flies were presented both by dead drift and dragged through seams located in pools and deeper runs.

Data Analysis

In my analysis I established a reference condition defined as the proportion (or percent relative abundance) of a fish species found upstream of bridges (n=32) of the total catch at bridge-stream crossings. While a more suitable reference would have been natural stream reaches, study basins had high levels of industrial development and extensive stream crossing networks. Thus, it would have been difficult to find suitable reference sites free from the influences of downstream culverts. I assessed the influence of culvert hang height (m), outlet plunge pool depth (m), water velocity (m/s), length (m), and slope (%) on upstream fish passage. These were the independent variables for my analysis. I determined the influence of these parameters on the proportion of a fish species found upstream of culverts. Different levels of each culvert parameter were evaluated, and were typically determined by the natural breaks in my data. I used bootstrapping analysis (Efron 1993) to assess the impacts of culvert parameters on the upstream proportion (values between 0 and 1) of a stream fish species. For both bridges and culverts the re-sampling approach was repeated 10,000 times, which allowed me to determine a bootstrap mean and 95% confidence intervals (CIs) for upstream fish species distributions. If the mean upstream proportion was 0 (i.e. when no Burbot were found upstream of hanging culverts) a binomial distribution was used to determine CIs. To determine if there were significant effects, I compared culvert mean and 95% CIs to those at reference bridge sites. I only included species that were found at $\geq$ 6 culvert sites in each parameter category (e.g. no hang height, hang heights 0.0m-0.38m and hang heights 0.42m-0.84m) in my bootstrap analysis. Distributions were then compared to my control; the upstream proportion of a fish species observed at bridge crossings. Using proportion of individuals of a fish species allowed me to remove, in part, the influence of stream size from my analysis, since one would expect to find higher fish abundances in larger streams. Prior to model assessment and analysis, Pearson's correlations between all culvert characteristics were used to identify redundancies. If two variables were highly correlated ($r > |0.7|$) only one was retained (Dauwalter and Fisher 2007).

For my analysis I grouped the two sucker species (Longnose and White Suckers) together and two minnows species (Pearl Dace and Lake Chub) together because they may have been misidentified in the field and shared similar life history traits that are important to habitat fragmentation (i.e. how far they will move and swimming ability). I analyzed Burbot, Spoonhead Sculpin and Rainbow Trout data individually. I excluded Brook Stickleback (*Culaea inconstans*), Trout-Perch (*Percopsis omiscomaycus*), Longnose Dace (*Rhinichthys cataractae*), Brook Trout (*Salvelinus fontinalis*), and Mountain Whitefish (*Prosopium williamsoni*) from my analysis because they were uncommon species and captured at <10 sites. Further, I excluded sites from analysis that had very low fish densities (< 4 fish) (Burford et al. 2009). Although I often captured low densities of Arctic Grayling, I retained them in my analysis given that culverts are cited as a major factor exacerbating extreme provincial population declines (ASRD 2005). At culvert crossing sites with multiple culverts, often there was a vertically offset pipe or culvert that would be completely impassable to fish (i.e. outlet completely crushed, exceedingly large hang heights, jammed with debris etc.). In these cases, I only included the most passable culvert in my bootstrapping analysis (Park et al. 2008).

Using a *t*-test I compared upstream bankfull channel width (m) with culvert diameters and bankfull widths under bridges (Zar 2010). At study sites with multiple culverts, I combined diameters of all passable culverts (i.e. culverts that were vertically offset were excluded). I used a paired *t*-test to compare common water quality variables (dissolved oxygen (mg/L) and temperature (˚C)) and substrate (% gravel/cobble and % silt and sand) to determine if stream habitat differed upstream and downstream of culverts and bridges (Zar 2010). I accepted statistical significance at an alpha level of 0.05.

Results

Watercourse crossings and stream fish characteristics

I captured fish at 83 sites (32 bridge and 51 culvert crossing sites). At culvert and bridge study sites, I captured 666 Lake Chub (22.8% of total catch), 527 Rainbow Trout (18.0%), 304 Pearl Dace (10.4%), 301 White Suckers (10.6%), 203 Burbot (6.9%), 148 Spoonhead Sculpin (5.1%), 105 Longnose Suckers (4.9%), and 72 Arctic Grayling (2.5%). Stream fish species that were excluded from my analysis because they were found at < 10 culvert and bridge crossing sites included 187 Trout-Perch (6.4%), 163 Brook Trout (5.6%), 142 Longnose Dace (4.9%), 86 Brook Stickleback (2.9%) and 10 Mountain Whitefish (0.3%). I found that outlet drop (m) and hang height (m) were highly correlated (Pearson, $r=0.94$), therefore only culvert hang height was retained for analysis. All other culvert parameters were not highly correlated (Pearson, $r<0.49$).

I grouped culvert hang heights into three categories given natural breaks in my data: no hang height, 0.04-0.38m (‘modest hang heights’) and 0.42-0.84m (‘high hang heights’). Upstream proportions (hereafter referred to as ‘relative proportion’ or ‘relative abundance’) of Burbot were most strongly influenced by culverts with a physical hang height (Table 2.2, Figure 2.2). The mean proportion of Burbot was 0.36 at non-hanging culverts, which was comparable to bridge crossings (mean=0.40 95% CI = 0.24-0.55), and much higher (almost a 0.36 unit increase) than that calculated for culverts with modest hang heights (Table 2.2). Spoonhead Sculpin, Longnose and White Suckers, and Lake Chub and Pearl Dace were also found upstream of non-hanging culverts but proportions were respectively 0.23, 0.13 and 0.28 units less than that observed upstream of bridges (Table 2.2). Conversely, proportions of Rainbow Trout were higher at upstream culverts (hanging and non-hanging) than those recorded upstream of bridges (a 0.06-0.20 unit increase) (Table 2.2). The difference between abundances observed at the highest hanging culverts (0.42-0.84m) and reference sites was statistically significant based on the mean lying outside of the 95% CI for bridges (Table 2.2, Figure 2.2). Arctic Grayling were only detected at a sufficient number of sites at non-hanging culverts. Abundances were significantly (0.08 units) higher than those observed at bridges (Table 2.2, Figure 2.2).

Culvert outlet plunge pool depths were divided into three categories for analysis: 0.10-0.58 m, 0.60-0.97 m and 1.00-3.50 m. All proportions of Burbot were lower than those observed at the bridge (reference) sites (Table 2.2, Figure 2.3). There was a statistical difference between abundances of Burbot observed at culverts with shallow to moderate plunge pool depths and bridge crossings given that the Burbot means fell outside of the 95% CI observed at bridges (Table 2.2, Figure 2.3). However, the abundances of Burbot found upstream of culverts increased by 0.19 units with deeper outlet plunge pools. Rainbow Trout had the highest abundances of all species when outlet plunge pools were the deepest (Table 2.2, Figure 2.3). There was a statistically significant difference between Rainbow Trout abundances observed at reference bridges and culverts with the deepest outlet plunge pools (Table 2.2). Additionally, sites with deep plunge pools had more Spoonhead Sculpin upstream of culverts relative to bridge crossings (a 0.20 unit increase) (Table 2.2, Figure 2.3). This was also statistically significant. Abundances of Arctic Grayling, sucker and minnow species were not influenced by outlet pool depth and were similar to that recorded at bridge sites (Table 2.2, Figure 2.3).

I grouped my analysis of culvert outlet velocities into two levels: velocities similar to control bridge sites (0.00-0.40 m/s) and those exceeding velocities under bridges (0.59-1.90 m/s). At culverts with lower outlet water velocities I found that there were significantly lower abundances of Burbot and minnows than those observed at my reference sites (Table 2.2, Figure 2.4). There were no significant difference between Arctic Grayling abundances at culverts with low outlet velocities and bridges. When culvert velocities exceeded 0.50 m/s there were less Burbot (a 0.32 unit decrease) and fewer minnows (a 0.34 unit decrease) compared to culverts with lower (<0.40 m/s) outlet velocities (Table 2.2, Figure 2.4). Spoonhead Sculpin and suckers were not found in a sufficient number at low velocity culvert sites to be included in this analysis. At higher culvert velocities, Spoonhead Sculpin proportions decreased by 0.11 units and suckers decreased by 0.29 units compared to bridge-crossings (Table 2.2, Figure 2.4). The difference between bridge-crossings and culverts with high water velocities was statistically significant for sucker species (Table 2.2). Contrarily, proportions of Rainbow Trout are comparable to bridges at lower culvert water velocities, while high water velocities lead to more (0.14 units) Rainbow Trout upstream of culverts (Table 2.2, Figure 2.4). The difference between Rainbow Trout abundances at bridges versus high culvert velocities was statistically significant (Table 2).

I followed natural breaks in my data and grouped culvert lengths into three categories: 8-15 m, 15-30 m and 30-111 m. Although there were significantly lower proportions of Burbot than at bridge crossings, there were no large differences between culverts 8-15 m versus 15-30 m long (Table 2.2, Figure 2.5). There were higher abundances of Rainbow Trout as the length of the culvert increased. There were significantly more Rainbow Trout at the medium length and longest culverts than there were at short culverts (0.17-0.19 unit increase) and bridge crossings (0.12-0.14 unit increase) (Table 2.2, Figure 2.5). At the longest culverts, there was no significant difference in Arctic Grayling abundances relative to reference sites. Both minnows and suckers were significantly less abundant at culverts 15-30 m long than at bridges and the longest culverts. For instance, suckers increased by 0.34 units and minnows increased by 0.39 units upstream of bridges than culverts 15-30 m long. They were, however, most abundant at the longest culverts (Table 2.2, Figure 2.5).

I grouped culvert slopes into three categories: 0.0-1.8%, 2.02-4.88% and 5.73-13.02%. Relative to reference bridge sites, Burbot, suckers and minnows all had lower abundances at culverts with low slopes (Table 2.2, Figure 2.6). This difference was statistically significant for minnows and Burbot. Abundances of Burbot decreased above higher culvert slopes (Table 2.2, Figure 2.6). Rainbow trout proportions at the three culvert slopes categories were similar or higher than that recorded for bridges (Table 2.2, Figure 2.6). However, only abundances observed upstream of culverts with gentle slopes significantly exceeded those observed at reference bridge sites. Culverts with the lowest slopes did not significantly affect Arctic Grayling abundances.

Overall, hang height, high outlet water velocities and slope were the culvert parameters that had the largest negative effects on upstream proportions of non-sportfish species. Of the non-sportfish species in my study, upstream proportions Burbot decreased the most (a 0.30 to 0.40 unit decrease) (Table 2.2). Generally, abundances of Spoonhead Sculpin, sucker and minnow species also declined but not as severely as Burbot (Table 2.2).

Table 2.2 Upstream proportion and 95% confidence intervals of Burbot (BURB), Spoonhead Sculpin (SPSC), suckers (Longnose and White suckers), minnows (Lake Chub and Pearl Dace), and Rainbow Trout (RNTR) from bootstrap analysis of data at bridges and culverts in the Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Numbers in brackets indicate the number of sample sites.

	BURB proportion US	95% CI	SPSC proportion US	95% CI	Suckers proportion US	95% CI	Minnows proportion US	95% CI	RNTR proportion US	95% CI	ARGR proportion US	95% CI
Bridges	0.4 (19)	0.24-0.55	0.43 (22)	0.32-0.54	0.4 (14)	0.28-0.53	0.51 (19)	0.38-0.63	0.33 (23)	0.23-0.44	0.32 (10)	0.15-0.38
Culverts												
Hang height (m)												
None	0.36 (15)	0.54-0.76	0.20 (6)	0.05-0.35	0.33 (9)	0.19-0.47	0.23 (13)	0.13-0.34	0.39 (20)	0.30-0.48	0.4 (10)	0.19-0.61
0.04-0.38	0.00 (7)	0.00-0.28	-	-	-	-	-	-	0.42 (14)	0.29-0.55	-	-
0.42-0.84	-	-	-	-	-	-	-	-	0.53 (6)	0.37-0.70	-	-
Depth of plunge pool (m)												
0.10-0.58	0.13 (10)	0.01-0.31	-	-	-	-	-	-	0.4 (17)	0.28-0.52	-	-
0.60-0.97	0.22 (10)	0.07-0.41	-	-	0.36 (6)	0.13-0.63	0.38 (8)	0.20-0.58	0.35 (13)	0.24-0.48	0.32 (6)	0.03-0.65
1.00-3.50	0.32 (6)	0.00-0.63	0.63 (6)	0.38-0.83	0.23 (6)	0.08-0.40	-	-	0.54 (10)	0.45-0.62	-	-
Water velocity (m/s)												
0.00-0.40	0.26 (10)	0.10-0.44	-	-	-	-	0.42 (7)	0.21-0.63	0.35 (13)	0.24-0.44	0.35 (7)	0.11-0.58
0.59-1.90	0.08(16)	0.01-0.16	0.32 (7)	0.11-0.57	0.11 (9)	0.01-0.22	0.17 (10)	0.06-0.30	0.47 (28)	0.38-0.56	-	-

Length (m)												
8.0-15.0	0.06 (9)	0.01-0.12	-	-	-	-	-	-	0.28 (13)	0.16-0.41	-	-
15.8-29.9	0.10 (14)	0.04-.19	-	-	0.06 (8)	0.00-0.15	0.12 (7)	0.01-0.30	0.45 (20)	0.37-0.53	-	-
30.7-111.4	-	-	-	-	0.49 (6)	0.29-0.68	0.56 (6)	0.44-0.71	0.47 (6)	0.20-0.75	0.37 (6)	0.08-0.71
Slope (%)												
0.00-1.80	0.22 (9)	0.00-0.44	-	-	0.29 (7)	0.11-0.47	0.29 (7)	0.14-0.45	0.48 (13)	0.35-0.61	0.28 (6)	0.03-0.61
2.02-4.88	0.10 (8)	0.03-0.18	-	-	-	-	-	-	0.31 (12)	0.21-0.40	-	-
5.73-13.02	-	-	-	-	-	-	-	-	0.44 (8)	0.28-0.61	-	-

“-“ = when a fish species was not found at > 6 sites at the culvert parameter it was excluded from analysis.

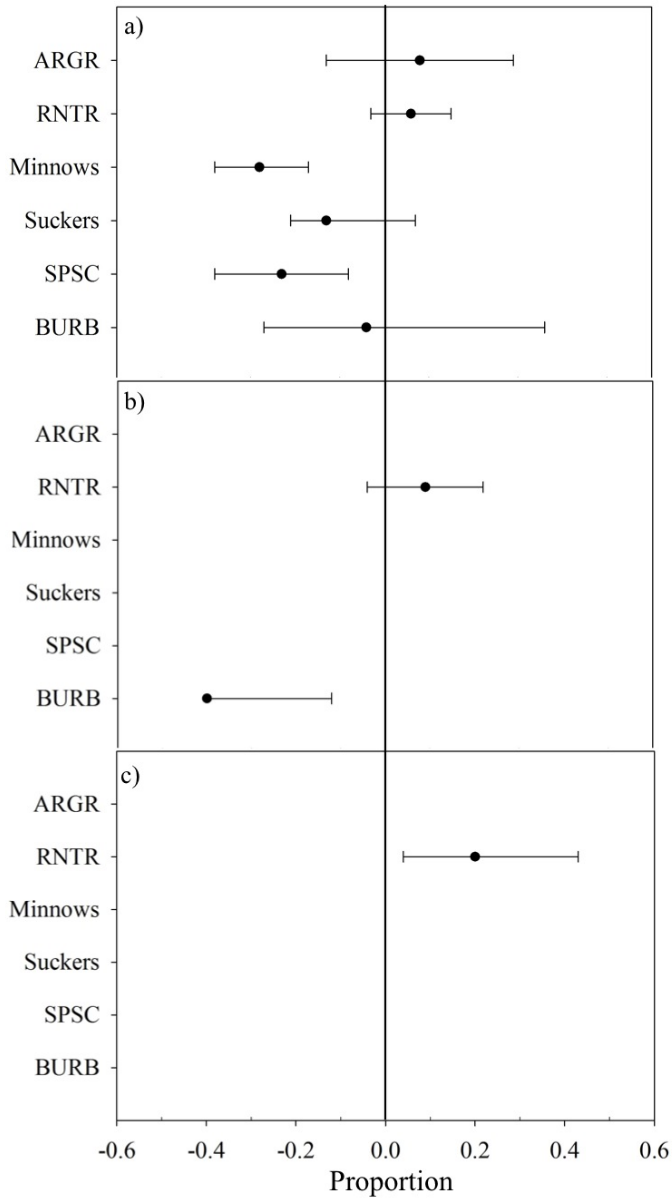

Figure 2.2 Bootstrap analysis for a) no hang height, b) hang heights 0.04-0.38m and c) hang heights 0.42-0.84m at culverts sampled for fish in Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Upstream abundance values for each species have been standardized to allow for bridge (reference) means to equate to 0. The vertical line represents the mean upstream proportions of stream fish species at bridge-crossings. If x values are negative, higher abundances were found downstream relative to bridges, if x values are positive, higher abundances were found upstream. ARGR=Arctic Grayling, RNTR=Rainbow Trout, SPSC=Spoonhead Sculpin, BURB=Burbot.

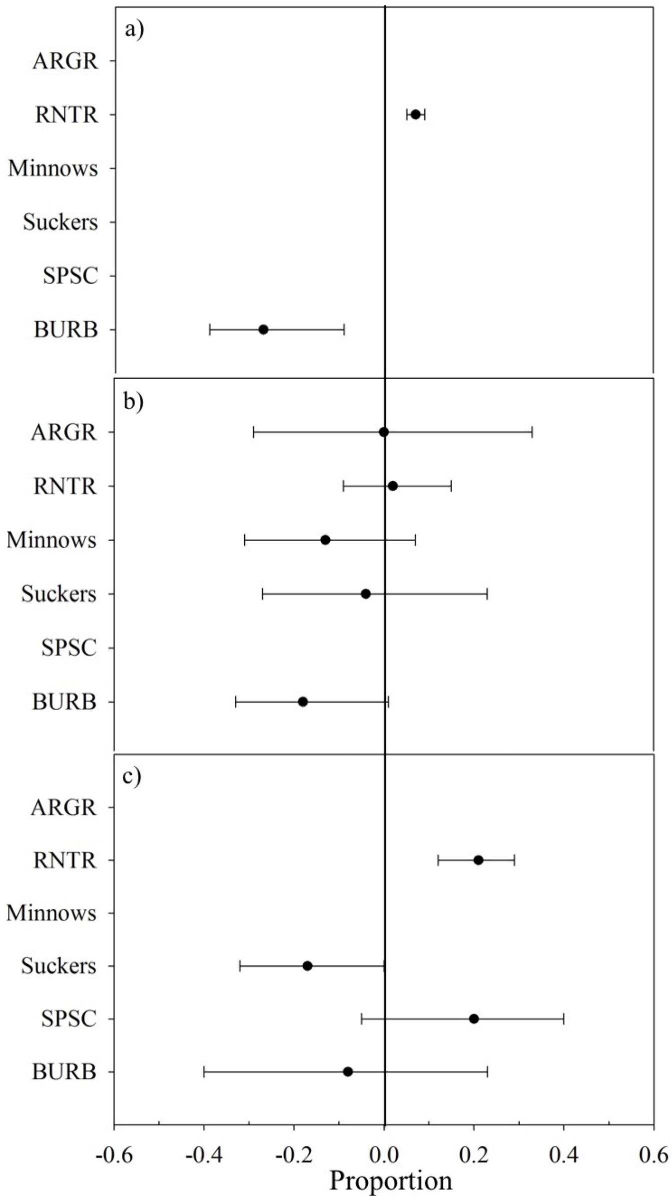

Figure 2.3 Bootstrap analysis for outlet plunge pool depth of a) 0.10-0.58m, b) 0.60-0.97 and c) 1.00-3.50m at culverts sampled for fish in Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Upstream abundance values for each species have been standardized to allow for bridge (reference) means to equate to 0. The vertical line represents the mean upstream proportions of stream fish species at bridge-crossings. If x values are negative, higher abundances were found downstream relative to bridges, if x values are positive, higher abundances were found upstream. ARGR=Arctic Grayling, RNTR=Rainbow Trout, SPSC=Spoonhead Sculpin, BURB=Burbot.

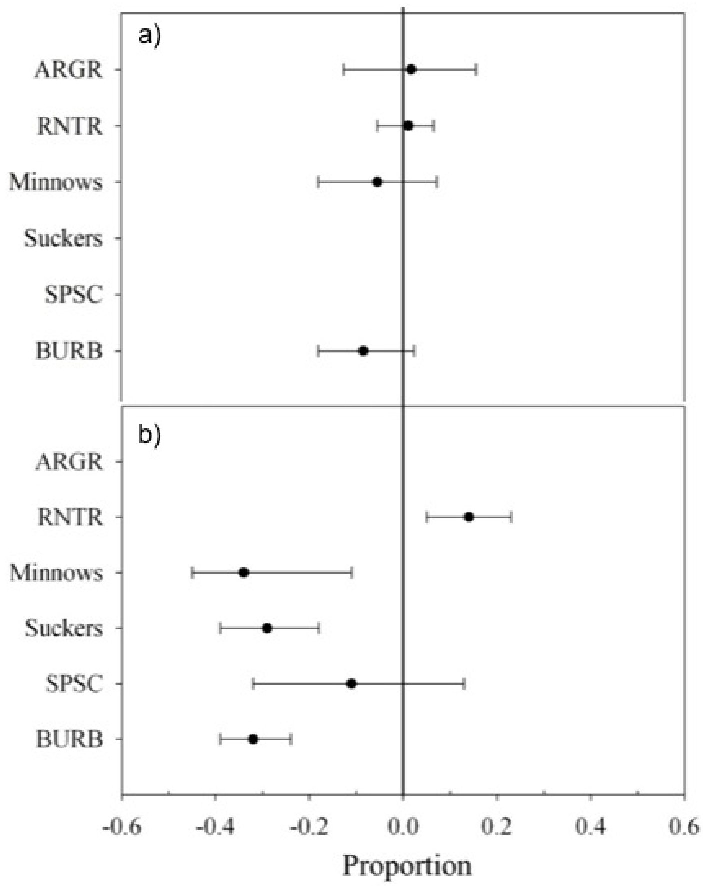

Figure 2.4 Bootstrap analysis for outlet water velocities of a) 0.00-0.40m/s and b) 0.59-1.90m/s at culverts sampled for fish in Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Upstream abundance values for each species have been standardized to allow for bridge (reference) means to equate to 0. The vertical line represents the mean upstream proportions of stream fish species at bridge-crossings. If x values are negative, higher abundances were found downstream relative to bridges, if x values are positive, higher abundances were found upstream. ARGR=Arctic Grayling, RNTR=Rainbow Trout, SPSC=Spoonhead Sculpin, BURB=Burbot.

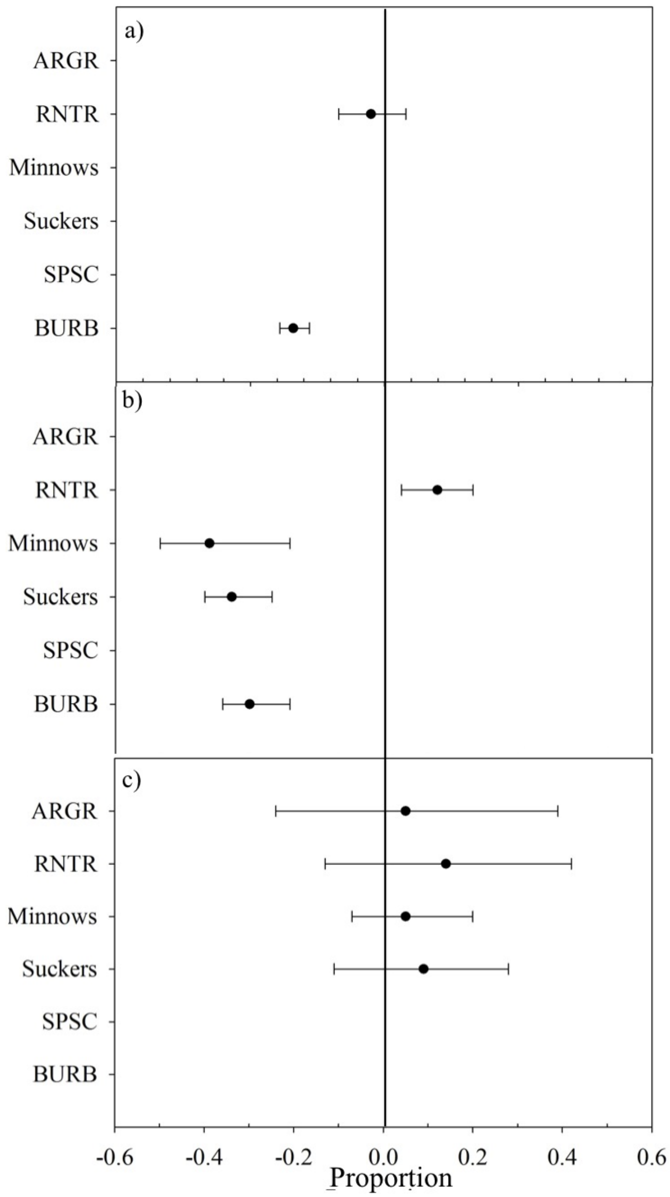

Figure 2.5 Bootstrap analysis for culverts lengths of a) 8.8-15.0m, b) 15.0-30.0m and c) 30.0-111.4m in Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Upstream abundance values for each species have been standardized to allow for bridge (reference) means to equate to 0. The vertical line represents the mean upstream proportions of stream fish species at bridge-crossings. If x values are negative, higher abundances were found downstream relative to bridges, if x values are positive, higher abundances were found upstream. ARGR=Arctic Grayling, RNTR=Rainbow Trout, SPSC=Spoonhead Sculpin, BURB=Burbot.

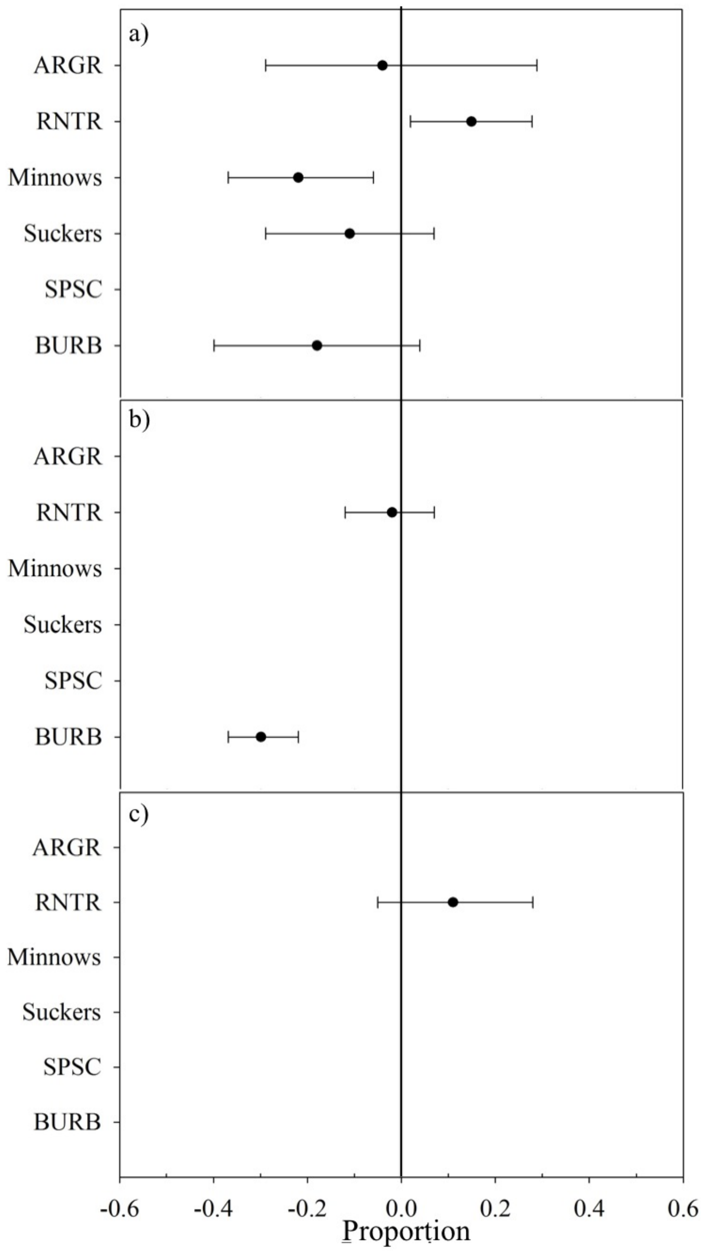

Figure 2.6 Bootstrap analysis for culvert slopes of a) 0.06-1.8%, b) 2.02-4.88% and c) 5.73-13.02% in Athabasca River tributary streams during the summers of 2007, 2008 and 2009. Upstream abundance values for each species have been standardized to allow for bridge (reference) means to equate to 0. The vertical line represents the mean upstream proportions of stream fish species at bridge-crossings. If x values are negative, higher abundances were found downstream relative to bridges, if x values are positive, higher abundances were found upstream. ARGR=Arctic Grayling, RNTR=Rainbow Trout, SPSC=Spoonhead Sculpin, BURB=Burbot.

Watercourse crossing inventory

On average, surveyed culverts had hang heights exceeding 0.15m, had slopes greater than 3.00%, were longer than 20.00m, outlet plunge pools deeper than 0.4 m and had average outlet velocities of 0.50m/s (Table 2.3). At culverts sampled for fish, on average, hang heights were 0.16m, slopes were greater than 2.50%, lengths were 26.53m, outlet pools were deeper than 0.8m and outlet water velocities averaged 0.70m/s (Table 2.3). Culvert characteristics differed between study sites. I found 53% of culverts were not hanging, 19% had hang heights from 0.01-0.20m, 16% from 0.21-0.50m and only (12%) had hang heights exceeding 0.51m (Figure 2.7). Most culverts (56%) had outlet plunge pools ranging from 0.01-0.50m deep, while 30% had outlet pools 0.51-1.00m (Figure 2.8). At surveyed crossing sites and those sampled for fish, average and maximum water velocities were higher through culverts than underneath bridges (Table 2.3). Most culverts had outlet velocities less than 0.4m/s (62%), while 38% outlet velocities exceeded 0.4m/s (Figure 2.9). Most culverts (87%) were less than 30m long (Figure 2.10) and most (77%) had slopes less than 5% (Figure 2.11).

Field data suggested that culverts constrict streams. The bankfull channel width 50 meters upstream was significantly different than diameters of assessed culverts (paired *t*-test, $t=4.53$, $df=233$, $p<0.001$) and culverts sampled for fish (paired *t*-test, $t=5.6$, $df=40$, $p<0.001$). There was no significant difference at assessed bridges (paired *t*-test, $t=0.88$, $df=78$ $p=0.38$) and bridges where fish were captured (paired *t*-test, $t=-1.5$, $df=27$, $p=0.152$).

Table 2.3 Minimum, maximum, mean and standard deviation (+/-SD) of measured bridge and culvert characteristics with and without fish present on Athabasca River tributary streams, 2007-2009.

	Velocity (m/s)	Length (m)	Slope (%)	Depth of plunge pool (m)	Hang height (m)
Assessed crossing sites					
Bridges					
Min	0	-	-	-	-
Max	1.53	-	-	-	-
Mean	0.31	-	-	-	-
Standard deviation (+/-)	0.37	-	-	-	-
Culverts					
Min	0	3.95	0.11	0	0
Max	2.16	162	15.04	3.5	3.9
Mean	0.5	20.01	3.55	0.47	0.18
Standard deviation (+/-)	0.57	19.21	2.64	0.4	0.36
Crossing sites with fish present					
Bridges					
Min	0.02	.	.	.	.
Max	0.58	.	.	.	.
Mean	0.18	.	.	.	.
Standard deviation (+/-)	0.19	.	.	.	.
Culverts					
Min	0.00	8.80	0.06	0.11	0.00
Max	1.90	111.43	12.98	3.50	0.84
Mean	0.70	26.52	2.60	0.89	0.16
Standard deviation (+/-)	0.59	20.29	3.00	0.61	0.24

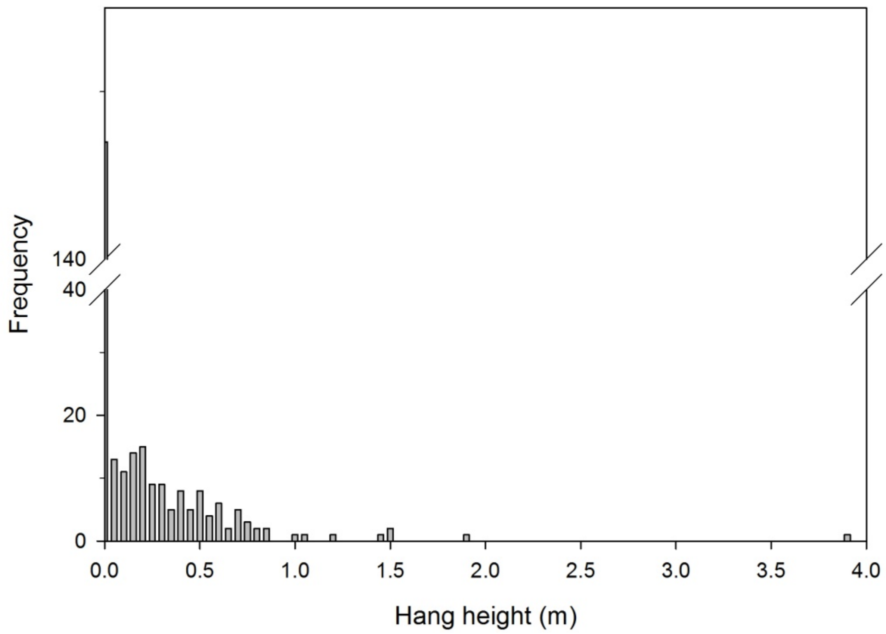

Figure 2.7 Hang heights (m) of assessed culverts in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

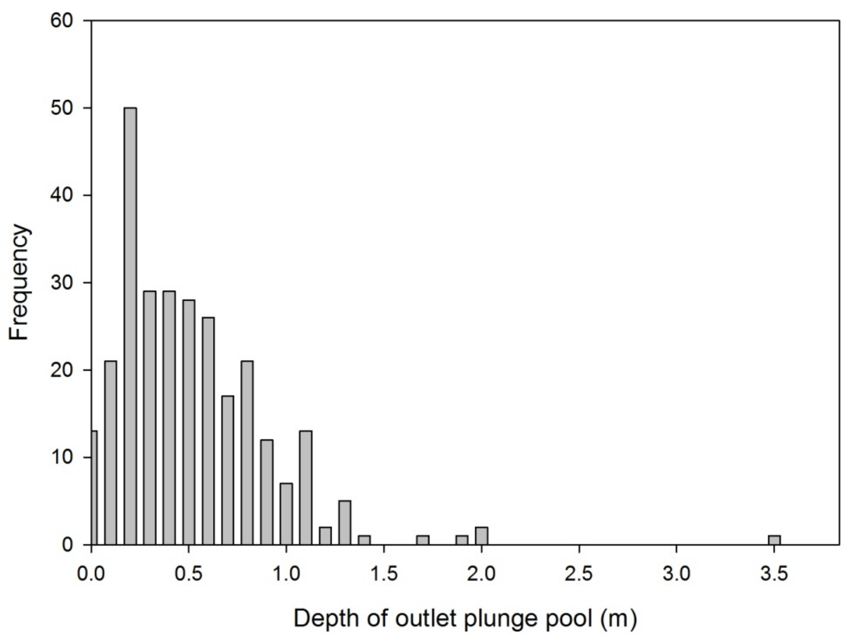

Figure 2.8 Depth of plunge pools (m) of assessed culverts in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

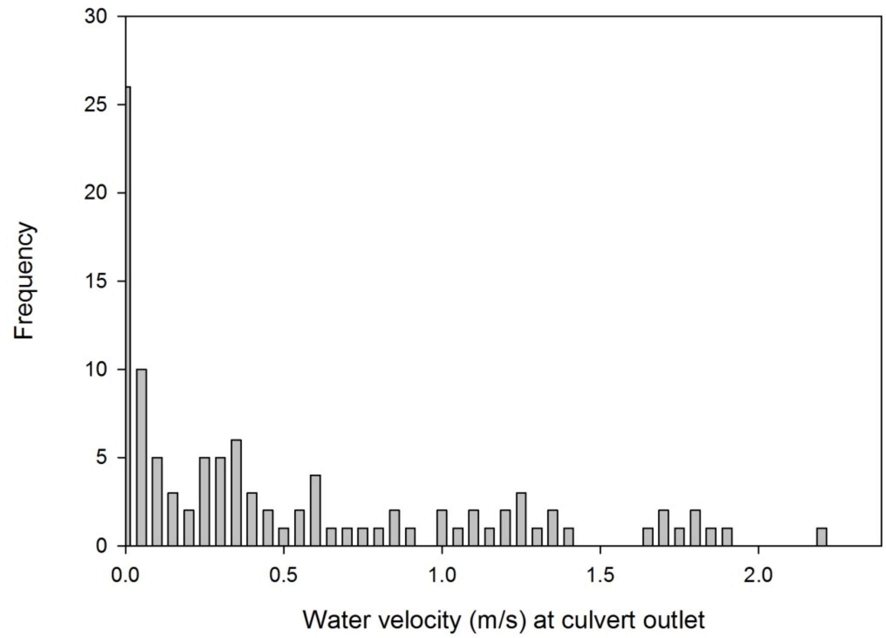

Figure 2.9 Outlet water velocities (m/s) of assessed culverts in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

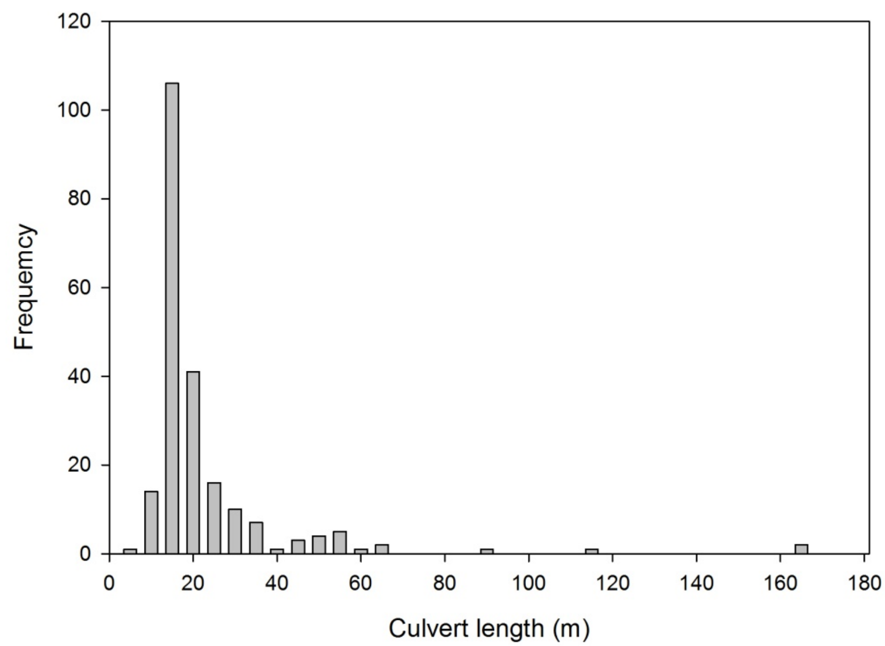

Figure 2.10 Length (m) of assessed culverts in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

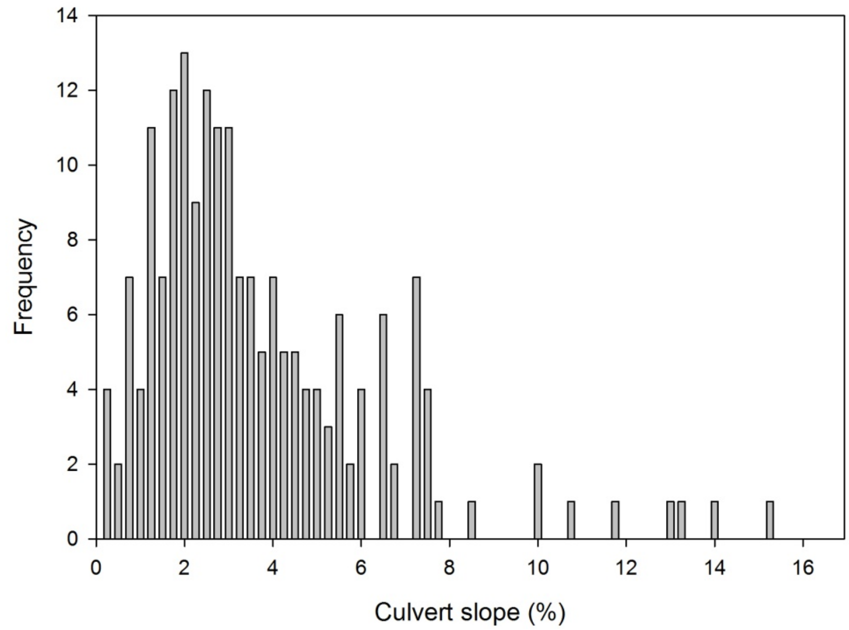

Figure 2.11 Slope (%) of assessed culverts in Athabasca River tributary streams during the summers of 2007, 2008 and 2009.

Watercourse crossings and stream habitat

I measured bankfull channel width (m), temperature (°C), dissolved oxygen (mg/L), percent sand, percent gravel and cobble at locations where I sampled for fish. The in-stream characteristics differed considerably between sites (Table 2.4). However, there were no significant difference in water temperature (°C) (paired *t*-test, t=-1.93, df=28 p=0.06) and dissolved oxygen (mg/L) (paired *t*-test, t=0.59, df=18 p=0.57) between upstream versus downstream of bridge crossings. There was no significant difference in percent gravel and cobble (paired *t*-test, t=0.79, df=29 p=0.44) and percent silt and sand (paired *t*-test, t=-0.82, df=29 p=0.42) between upstream and downstream locations at bridges. While there were no differences in dissolved oxygen (mg/L) upstream and downstream of culverts (paired *t*-test, t=-1.22, df=21 p=0.24), temperature (°C) was higher upstream of culverts (paired *t*-test, t=-2.17, df=30 p=0.02). Culverts also altered stream substrate. There were significantly higher percentages (42% more) of silt and sand at upstream locations (paired *t*-test, t=-3.96, df=28 p<0.001) and more (23%) gravel and cobble at downstream locations (paired *t*-test, t=2.20, df=28 p=0.02) of culverts.

Table 2.4 Minimum, maximum, mean and standard deviation (+/- SD) of instream characteristics surrounding bridges and culverts on Athabasca River tributary streams, 2007-2009.

	Bankfull channel width (m)	DO (mg/L)	Temperature (C)	% Sand	% Gravel and cobble
Bridges					
Min	2.24	9.15	5.00	0.00	0.00
Max	29.80	16.48	16.80	100.00	46.67
Mean	8.70	10.79	11.38	41.02	24.47
Standard deviation (+/-)	5.89	1.54	2.90	37.91	15.79
Culverts					
Min	0.80	5.92	4.00	3.50	0.00
Max	17.00	16.30	15.50	100.00	38.56
Mean	4.73	10.46	10.36	54.72	18.62
Standard deviation (+/-)	3.45	2.05	2.82	33.45	13.04

Discussion

The Canadian *Fisheries Act* states that watercourse crossing structures should allow passage of all fish species at all life-history stages. Despite this, there continues to be little to no remediation of impassable culverts. The majority of fish passage studies remained focused on recreationally valued salmonid species. My study provides a broader examination of stream fish passage by not only addressing how culverts may influence upstream movements of sport fish species, but burbot, minnow, sculpin and sucker species as well. I describe how several culvert parameters act as barriers and impede upstream fish passage. My evidence indicates that culverts alter stream habitat and negatively impact upstream fish densities of many stream fish species. The results of my watercourse crossing inventory suggest that the majority of culverts measured were acting as partial or full barriers to Alberta lotic foothill fish movements.

Species abundance upstream of culverts has commonly been used as indicator of a stream crossing barriers (Winston et al. 1991, McLaughlin et al. 2006). As predicted, I found that upstream abundances of weaker swimming fish species were most affected by culverts. Indeed, I found that suckers, a stronger swimming fish species, were affected by a few of the investigated culvert parameters. Physical drops at the outlet (hang heights), outlet water velocities and slope seem to be important culvert parameters shaping upstream non-sportfish distributions, such as suckers. In particular, hanging culverts appeared to act as complete barriers to upstream passage of Burbot and partially impeded Spoonhead Sculpin, sucker and minnow movements. When culverts were hanging I did not capture Burbot upstream. Although a culvert hanging a few centimeters seems insignificant, it can act as a complete physical barrier to instream and migratory movements for a weaker-swimming fish, benthic dwellers such as Burbot in particular. Similarly, Norman et al. (2009) did not observe any movement of benthic fishes through perched culverts in the Etowah River system in Georgia, while water column fishes (shiners and chub) could occasionally bypass perched culverts. If fish were present at all hanging culverts assessed in my inventory, based on my model of fragmentation 47% would act as complete barriers to Burbot upstream movements and would partially impair Spoonhead Sculpin, sucker and minnow passage.

Constriction of the natural streambed and steep slopes often cause high water velocities through culverts that can impede or prevent the passage of fish (Warren and Pardew 1998, MacDonald and Davies 2007). Water velocity through culverts may affect fish swimming distance and frequency (Toepfer et al. 1999), and at higher water velocities the likelihood of fish passage through a crossing is reduced because of increased energetic stress (Adams et al. 2000). Excessive water velocities are often cited as a major factor inhibiting fish movements through culverts (Warren and Pardew 1998). Where water velocities were similar to those recorded underneath bridge crossings (<0.40m/s), upstream densities of fish were comparable to those observed at bridges. Where culvert outlet water velocities surpassed 0.50 m/s however, I saw large reductions in upstream proportions of Burbot and minnow and sucker species. Of the studied culverts, 38% would potentially act as velocity barriers. Warren and Pardew (1998) found that the movement of stream fish through watercourse crossings was inversely related to water velocity, and that high water velocities through culvert crossings (> 0.4 m/s) acted as bidirectional barriers to sunfish and minnow species while bridges allowed unrestricted bidirectional instream movements (Warren and Pardew 1998). It should be noted that my measurements of culvert water velocities and hang heights were instantaneous measurements, and I could not specifically determine if or when fish passed through the culvert. For instance, all culverts were likely not hanging during higher water events for some period of my study. However, since culvert assessments were typically performed the same day as fish sampling, upstream and downstream fish distributions likely represent the culvert's influence at the time of survey.

Low culvert slopes (0.06-1.80%) led to higher downstream proportions of minnows and suckers. My findings support those of Bouska and Paukert (2009) who found that the proportion of cyprinids that moved upstream decreased with higher culvert slopes. When culvert slopes exceeded 2% Burbot were found primarily downstream of culverts. Not surprisingly, given the poor swimming stamina of Burbot (Jones et al. 1974) even slight culvert slopes appear to significantly impede their upstream movement. Of assessed culverts, 69% had slopes exceeding 2% and would be predicted to be partial barriers to upstream movements of Burbot. Different levels of each culvert parameter may influence upstream proportions of minnows (Lake Chub and Pearl Dace), suckers (Longnose Sucker and White Sucker) and Spoonhead Sculpin to degrees for which my limited sampling does not allow inference. The Burbot results, however, indicated a need for future studies of other fish species.

Contrary to the findings of other studies, I did not find evidence that culverts acted as barriers to upstream passage of Rainbow Trout. Surprisingly, I found that larger culvert hang heights resulted in higher upstream proportions of Rainbow Trout, even though other authors have found that upstream trout passage is impeded by perched culverts (e.g. Burford et al. 2009). Further, my findings indicate that Rainbow Trout are not influenced by culvert slope, as even the steepest culverts (>5.7%) led to higher upstream densities. On the contrary, Burford et al. (2009) found that Westslope Cutthroat

(*Oncorhynchus clarkii lewisi*) and Brook Trout (*Salvelinus fontinalis*) densities decreased upstream of culverts when slopes were higher than 4.5 %. Most notably however, upstream passage of Rainbow Trout and Spoonhead Sculpin appeared to be the highest when outlet pool depths were greater than 1m. Lauritzen (2002) suggested that an average ratio of the height of the barrier to the pool depth should be 1:1 to allow for optimal passage conditions. The largest culvert hang height in the fish study was 0.84m. For the deepest plunge pool category, my data exceeded Lauritzen's ratio in all cases and showed the highest upstream Rainbow Trout densities. Likewise, Mueller et al. (2008) suggested that the leaping ability and passage success of juvenile Coho Salmon (*Oncorhynchus kisutch)* through perched culverts improved when sufficiently deep and large pools were present at the culvert outlet. I speculate that culverts that exclude Burbot, a voracious predator, may offer both a competitive release for larger Rainbow Trout and a release from predation for smaller Rainbow Trout upstream of culverts. I cannot however, disregard the possibility that culverts may act as barriers to upstream Rainbow Trout movements. Rainbow Trout can likely maintain self-sustaining populations above complete barriers provided that there is sufficient upstream habitat despite the presence of culverts fragmenting their instream movements (Novinger and Rahel 2003, Wofford et al. 2005). In the absence of Burbot, Rainbow Trout populations above culverts may thrive. Alternatively, higher temperatures above culverts may allow increased productivity of Rainbow Trout upstream of culverts. For instance, in cold Alberta foothills streams, densities of Rainbow Trout appeared higher in streams with more sunlight and less shading (Craig Johnson pers. comm.). I extend caution to the interpretation of my results as culverts being beneficial to Rainbow Trout. More intensive studies are required before any definitive conclusions can be drawn.

The lack of success I encountered sampling Arctic Grayling using traditional sampling methods (i.e. backpack electrofishing), combined with their low populations densities in my study area did not allow me to draw any conclusions on the impacts of culverts on the upstream passage of Arctic Grayling. Fragmentation is currently suspected of being a significant factor exacerbating severe population declines in this species of concern. Perhaps, Arctic Grayling habitats in this region have declined severely and are now only present at bridges and the largest, non-hanging culverts with the lowest outlet water velocities.

Surprisingly, I found no evidence to suggest that the various culvert lengths altered the upstream passage of Alberta foothill stream fish species. The upstream passage of Burbot, sucker and minnows appeared to be negatively influenced by culverts with shorter lengths (8-30 m), while minnow, sucker and Rainbow Trout all had higher densities above the longest culverts (30-111m). Most fish passage studies have found an inverse relationship between upstream fish passage and culvert length. For instance, Bouska and Paukert (2010) found that the proportion of cyprinids that moved upstream of culverts decreased with larger slopes and lengths. My results may be explained by the initial culvert installation. Longer culverts are often found on larger roads and require more experienced engineers and personnel for their installation. Alternatively, shorter culverts are often found on smaller and less travelled roads. These are likely rapidly installed by relatively inexperienced crews. Therefore, the shorter and improperly installed culverts are more likely to act as fish passage barriers.

Most notably, the results of my study highlight how culverts could be a major driver shaping Burbot (and as a consequence Rainbow Trout) populations. Culverts are currently listed as a factor contributing to the worldwide declines of Burbot (Stapanian et al. 2010). Given the poor swimming stamina of Burbot (Jones et al. 1974) and the numerous parameters that may act as barriers to passage, culverts likely limit movement of Burbot from one waterbody to another. Although Burbot are generally widespread in Alberta and North America, they are locally sensitive to habitat disturbances (Stapanian et al. 2010). As a result of their susceptibility to anthropogenic disturbance and their role as a top predator in stream ecosystems (Arndt & Hutchinson 2000, Fisher et al. 2000, Nelson and Paetz 1992), I argue that there is a pressing need for more in depth North American studies of how Burbot and other weaker swimming non-sportfish species are influenced by watercourse crossing structures. The fragmentation of stream networks is reducing the movement of fish species from one stream segment to another with likely negative effects to both the structure and function of Alberta stream fish assemblages.

Even partial barriers may be significant at the population level to fish species that rely on movements through multiple stream fragments to complete life history processes. Culvert fragmentation research should focus on the weakest swimming fish species and early life history stages of the stream fish community to develop culvert remediation strategies that would allow the passage of all stream fish species regardless of their perceived recreational value.

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Chapter 3 - Evaluating sampling techniques for low-density Arctic Grayling (*Thymallus arcticus*) populations[2]

Introduction

Increased land use from expanding energy and forestry development, habitat fragmentation by improperly installed and maintained watercourse crossing structures, and increased angler access and angling pressure have resulted in severe declines of Arctic Grayling (*Thymallus arcticus*) populations across the province (Berry 1998; Blackburn and Johnson 2004; ASRD 2005). Although declines are relatively new to Alberta, similar trends have been previously observed over much of its North American range. Native Arctic Grayling populations have been extirpated in eastern North America (e.g., Michigan and Ontario), and have severely declined in their southern parts of their western range (e.g., Montana, Wyoming and British Columbia) (Northcote 1995). Since the 1950s in Alberta, Arctic Grayling have experienced high population reductions, with declines estimated at 90% for approximately 50% of subpopulations (ASRD 2005). Currently, Arctic Grayling is a fish species provincially listed as 'sensitive' (ASRD 2001).

The ability to monitor fish species abundance and assess the status of a species allows fisheries biologists to formulate management strategies and facilitate the protection and recovery of populations. The need for reliable data on distributions and abundances are especially important for species at risk, where a failure to obtain credible ecological information could result in inappropriately assigned conservation designations or management decisions (Mace 1994). Thus, reliable information on Arctic Grayling distributions and abundances are essential to their future persistence. However, provincial managers and fisheries biologists remain unsure on the extent of previous declines, current ranges and the overall status of Arctic Grayling. Monitoring fish populations requires both financial and human resources, as well as a cost-effective framework upon which programs can rely (Kennard and others 2006).

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[2] A version of this manuscript has been submitted to *Northwestern Naturalist.* Corresponding styles and format apply to citations, figures and tables.

To assess fish populations biologists rely on numerous methods. Electrofishing remains one of the most widespread fisheries sampling tools and is regarded as the most effective for sampling stream fish assemblages (Bohlin and others 1989; Reynolds 1996). Electrofishing only however, may not be suitable for all situations. Angling is a commonly used fisheries technique, and has been successfully applied when sampling Alberta Arctic Grayling and other salmonids (e.g. Fitzsimmons and Blackburn 2009; Paul and Post 2003). Another less frequently used technique to identify Arctic Grayling spawning habitat are egg kick surveys (R.L.&L. 1982a, 1995a, 1995b). Difficulties arise in conducting fisheries assessments using these methods because capture efficiencies and the vulnerability of fish life history stages may vary with sampling gear, size, behaviour, and time of sampling (Bayley and Austen 2002; Peterson and others 2004; Kennard and others 2006). Of particular importance is the influence of temporal variability on the success of Arctic Grayling sampling techniques. Arctic Grayling are a highly mobile species moving long distances seasonally between reaches to spawn in the spring (adults), to locate feeding grounds and find appropriate overwintering habitat (young-of-the-year, juveniles and adults) (Ward, 1951; Lucko 1992; Nelson and Paetz 1992; Stanislawski 1997). Many sites may contain a range of fish from young-of-the-year to the largest adults.

Three-pass removal and mark-recapture are commonly employed fisheries methods for estimating population abundance (Pine and others 2003), however, in Alberta the likelihood of detecting and accurately estimating Arctic Grayling abundances using these methods is largely unknown. Many authors caution against the use of the removal model because it can overestimate sampling efficiency and underestimate abundance estimates due to declining capture efficiencies with successive removal passes (Zippin 1958; Peterson and Cederholm 1984; Peterson and others 2004; Rosenberger and Dunham 2005; Dauwalter and Fisher 2007). Although the mark-recapture method is generally viewed as less susceptible to sampler and environmentally induced biases that can alter abundance estimates (Zippin 1958; Peterson and Cederholm 1984), its success hinges on a sufficient number of the population being marked and recaptured to account for and correct statistical biases (Ricker 1975; Jensen 1992). Both of these methods perform best when sample sizes are sufficiently large. Adequate sample sizes, however, are often difficult to achieve on rare species (Thompson 2004).

I initiated this study given the paucity of peer-reviewed literature on Arctic Grayling sampling methods and the lack of a standardized stream sampling protocol for Arctic Grayling in Alberta. My study was applied to populations within the southern limit of the species' range (Athabasca River and tributaries in north-central Alberta) where there have been drastic reductions in abundance (ASRD 2005). Given recent declines of Arctic Grayling, it is critical to establish the most efficient technique(s) to detect their occurrence and determine their abundance for this imperiled species. My primary objective was to evaluate common sampling techniques (backpack electrofishing and angling) to detect Arctic Grayling and determine population estimates in wadeable streams. As part of my evaluation, I also report results from an innovative, scoping-level survey design for detecting spawning habitat through the use of egg-kick surveys. Egg kick surveys for Arctic Grayling have the potential to be a monitoring tool (e.g. red counts; Gallagher and others 2007). I recommended the best methods as those, for example, that had the highest detection probabilities or results that met the assumptions of the analyses. I also described biases of gear type and season (early versus later summer) by comparing the structure and size catches of Arctic Grayling. I then summarize the findings of my results in an easy-to-use sampling decision flow diagram.

Methods

Study site

My study was performed on wadeable tributaries of the Athabasca River in the Foothills ecoregion, near the towns of Whitecourt, Edson and Hinton in west-central Alberta (Figure 3.1) (NRC 2006). Basins with high historical Arctic Grayling abundance were chosen for my study by referring to the province-wide fish and wildlife management information system (FWMIS) database, reviewing consultant and non-profit agency reports (e.g. Golder Associates, Alberta Conservation Association), and talking with provincial fisheries biologists and private consultants with experience sampling Arctic Grayling in Alberta. Located in the Lower Foothills and Upper Foothills subregions (NRC 2006), study sites extended as far south as the Embarras River (53˚17'N, 117˚00'W), and as far north as (54˚65'N, 115˚90'W) in the Freeman River drainage. The Alberta foothills have high annual precipitation, cold winters and warm summers (NRC 2006). The majority of study sites occurred in the Freeman River (n=27) and Sakwatamau River (n=36) drainages near the town of Whitecourt. These basins are cited to have had high historical Arctic Grayling populations and are believed to continue to support some of the highest Arctic Grayling densities in the area. Average discharge of the Athabasca River near study sites peaked at 600 m^3/s during the spring freshet, and ranged from 150 m^3/s to 600 m^3/s from May to August (Alberta Environment 2010). Study sites had an average bankfull channel width of 6.52m (range 0.71m – 29.8m). Land use in this area is dominated by the forestry, oil and gas exploration and extraction. Recreational activities include fishing, hunting, camping, and ATV use.

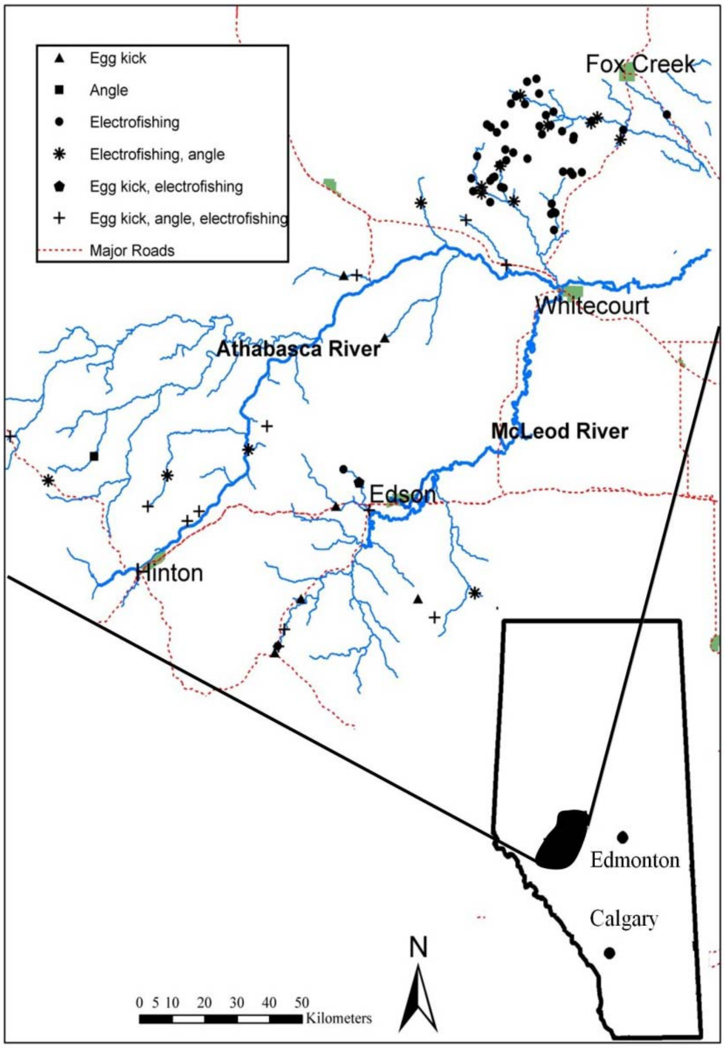

Figure 3.1 Map of Athabasca River basin, 2008 and 2009 study sites and sampling methods used.

Field sampling

Sampling for my study occurred from May - August (2008 and 2009), and consisted of a single pass of angling (July - Sept 2008, June - Aug 2009) and a single pass of electrofishing (May - Aug 2008, 2009) to detect Arctic Grayling. All fieldwork affecting fish was conducted under peer reviewed animal care protocols following the guidelines of the Canadian Council on Animal Care (#585806 2007-2009). I performed a mark-recapture and three-pass removal study during late summer (2008 and 2009) on a sub-sample of sites supporting Arctic Grayling to estimate population abundance.

In 2009 from May 12–27, I performed egg kick surveys at 20 sites in 13 sub-basins in an attempt to locate Arctic Grayling spawning areas. Of these sites, all 20 were also electrofished and angled for Arctic Grayling. Potential spawning sites were identified in second to fourth order streams and were described as riffle-run transitions with water depths of 0.15 - 0.5 m, water velocities of 0.35 - 0.55 m/s, and with gravel/cobble substrates (Stewart and others 1982; R.L&L. 1995a, 1995b; Berry 1998). For ease of access, surveyed sites were located upstream and downstream within 1 km of road/stream crossing intersections. Each site consisted of three riffle-run transitions within 500 m of one another.

Arctic Grayling spawning in Alberta boreal streams often coincides with the spring freshet where waters are often turbid from high rainfall. Further, Arctic Grayling have been observed spawning during the first two weeks of May, and sometimes into early June (Nelson and Paetz 1992; Berry 1998; Joynt and Sullivan 2003). Females lay between 5000 and 6000 eggs per 0.5 kg of body weight (Benhke 2002). Eggs are amber coloured, measure 2.4-2.7mm in diameter and easily distinguished from all other co-occurring fish eggs (Berry 1998; Benhke 2002). Arctic Grayling eggs hatch in 11 to 22 days with water temperatures between 7˚C and 11˚C (Berry 1998; Joynt and Sullivan 2003). The timing of my surveys followed the spring freshet and turbid conditions such that I could observe males occupying spawning territory, prior to hatch (May 12-27).

I sampled for eggs in suitable spawning habitat using a D-net placed downstream of 1-m^2 sampling plots (R.L&L 1995b), and then I disturbed the substrate for 1 minute. This was repeated at 3 - 9 locations within each riffle-run transition area. If no eggs were located after sampling three riffle-run transitions (18 sample plots in total), field crews continued to survey the next upstream site. If eggs were located at a site, sampling stopped immediately, and the tributary was deemed to support spawning Arctic Grayling.

To detect Arctic Grayling, during the 2008 and 2009 field seasons I electrofished a total of 84 sites in 18 sub-basins, of which 33 sites were also sample-angled using dry flies. An experienced three-person crew electrofished sites for 600 m (300 m upstream and downstream of the stream crossing) using a model 12 or LR24 Smith-Root backpack electrofishing unit. Electrofishing proceeded from downstream to upstream to avoid silting the stream and to maximize catchability. I made an effort to sample all stream habitat types (i.e under cover, riffle, run, pools). No block nets were used. One person operated the electrofisher, while two people netted fish. Voltage, frequency, and duty cycle were adjusted to maximize capture efficiency without injuring any fish species (settings range: voltage = 400 – 500 V, frequency = 40 – 60 Hz, duty cycle = 20 - 25%).

Angled sites were sampled with a three-person crew, one experienced (>5 years experience) angler and two moderately experienced anglers (2-5 years experience). I sample angled with fly rods and dry flies. Fly sizes ranged from 8 to 14. Arctic Grayling are opportunistic feeders that are highly susceptible to angling (Nelson and Paetz 1992; ASRD 2005) and typically do not show a preference for specific flies (Sullivan and Johnson 1994), but generally anglers used stimulators, humpies and elk hair caddis fly types. I normally used 5X tippets (diameter 0.152 mm) with a leader length of 2.1-2.7 meters. Dry flies were presented in both by dead drift and dragged through seams located in pools and deeper runs. Typically, when fish were captured, they were held live in buckets with ambient stream water, and then processed every 50 m. I measured fish fork length (mm) (fish length from snout to fork in the tail) and identified the species, after which fish were returned to a downstream location. Fish were not released back into the stream unless they appeared healthy and uninjured. In rare cases (2.5%) when fish were severely injured and unable to recover, I euthanized them.

At nine sites confirmed to support Arctic Grayling, I attempted to calculate population estimates using mark-recapture and three-pass removal methods. Each site was approximately 300 m in length. Prior to sampling, the upstream and downstream ends of each transect were blocked using beach seine nets with a 3.18 mm mesh. I took care to ensure that block nets were completely secured to the streambed. After block nets were in place, I performed an initial pass of angling with dry flies. Arctic Grayling caught during sample angling were marked with a clip to their adipose fin. Immediately after angling and marking, I completed a single electrofishing pass. Fish captured from electrofishing were marked with a small clip to their pelvic fin. After the first angling and electrofishing passes, marked fish were returned to the closed section of stream where they were captured. Following the initial marking passes and an overnight recovery period (Rosenberger and Dunham 2005), field crews conducted three-pass removal sampling with an electrofisher the following day. Before proceeding from one removal pass to another, the field crew waited for a minimum of one hour to allow fish to recover from electrofishing activity (Poos and others 2007). Every 50 m during electrofishing passes I recorded the number of marked captures, measured fork length (mm), and immediately released all fish (marked and unmarked) to the stream outside of the block nets. Unmarked fish captured during the three-pass removal effort were not marked. All block nets remained in position for two days until the depletion study was completed.

Analyses

Data from 34 sites where angling and electrofishing were performed were used to estimate occupancy (ψ) and detection probabilities (p) for Arctic Grayling using program PRESENCE (version 3.1) (Hines 2006). Egg kick surveys were performed at 20 of these sites. Probabilities of detection were estimated from encounter histories over all sites using a maximum likelihood function (MacKenzie and others 2002). Bankfull channel width (m) was used as a covariate. For my analysis I assumed that species presence and detection probabilities were constant across time and sites. I assumed sites were closed spatially and temporally to changes in occupancy. Backpack electrofishing and sample angling occurred in the same day. In addition, although the timing of egg kick surveys differed, the goal was to detect non-mobile Arctic Grayling eggs. I explored how p, the probability that a species will be detected at a site (given it is present) differed between backpack electrofishing, angling and egg kick surveys.

I then explored possible biases of gear type and season (early versus later summer) by comparing the structure and size catches of Arctic Grayling using an analysis of variance (ANOVA). For electrofishing and angling this involved assessing how Arctic Grayling size was influenced by temporal variability (summer season and year) and gear type by performing an (ANOVA) ($\alpha = 0.05$). Average fish fork length at a site (mm) was the dependent variable, season (early summer (May-June) and late summer (July-August)), year (2008, 2009) and gear type (electrofishing, angling) were independent variables. I investigated all potential interactions.

To evaluate sampling methods for monitoring the abundance of low-density Arctic Grayling populations, I described minimum thresholds for each technique. I deemed the mark-recapture method unsuccessful and susceptible to systematic statistical bias if < 4 fish were recaptured during successive passes at a site (Ricker 1975). A three-pass removal site was discarded as a sampling method if the efficiency of capture (p) is ≤ 0.20 because, at that level, abundance estimates become biased and unreliable (Lockwood and Schneider 2000). Lockwood and Schneider (2000) describe capture efficiency as:

$$p = T / kN - X \qquad \text{[Equation 1]}$$

Where,

T = the total number of fish caught from all passes,

k = the number of removal passes,

N = the absolute abundance of fish, and

X = the intermediate statistic.

Results

Arctic Grayling detection

Of the sites electrofished (n = 84) and angled (n = 33), Arctic Grayling were located at only 25 different sites. Using electrofishing and angling, I captured a total of 277 Arctic Grayling in the Freeman River, Sakwatamau River, Wolf Creek, Sundance Creek, Pinto Creek, Chickadee Creek, Two Creek, and Marsh Head Creek drainages (Figure 3.1). At sites where both angling and electrofishing were performed, I detected Arctic Grayling at 19 of 34 sites for a naïve occupancy estimate of 0.56. The naïve estimate is occupancy given perfect detectability. The estimated probability of occupancy from my study was 0.75 ($\pm$ 9.77). Since the estimated probability of occupancy is 25% larger, this suggests that Arctic Grayling were never detected at 1 in every four occupied sites. The probability of detecting Arctic Grayling using backpack electrofishing was 0.68 ($\pm$ 0.11). The probability of detecting Arctic Grayling where angling was successfully performed was 1.00 ($\pm$ 0.00). Egg kick surveys had the lowest probability of detecting Arctic Grayling eggs (p=0.13 ($\pm$ 0.12)). At the 20 sites surveyed using the egg kick survey method, I located a total of 20 Arctic Grayling eggs at one site only: Sundance Creek. Of the three Arctic Grayling detection techniques evaluated, backpack electrofishing and angling had the highest Arctic Grayling detection probabilities and were retained for further analysis.

Arctic Grayling size selectivity: influence of gear type and temporal variation

At sites confirmed to support Arctic Grayling (i.e. they were detected using at least one sampling method), on average, I captured 0.37 fish/100 s of backpack electrofishing (range: 0.00-4.63 fish/100s). Using sample angling I caught an average of 1.36 fish per hour of angling (range: 0.00-4.31 fish/hr). There were significant differences (α=0.05) in size of Arctic Grayling captured by gear type (ANOVA, $F_{(1,33)}$= 10.14, P=0.004) and between summer seasons (ANOVA, $F_{(1,33)}$=5.99, P=0.021) (Figure 3.2). The year of survey did not significantly influence the size of Arctic Grayling captured (ANOVA, $F_{(1,33)}$=0.04, P=0.852). Further, no interactions between year, season and gear type were significant (Table 3.1). Although angling caught larger fish (mean=173 mm, SD=39, range=100-285 mm), angling frequently failed to capture young-of-the-year Arctic Grayling (Figure 3.2). In contrast, electrofishing captured a broader size range (mean=92 mm, SD=42, range=23-282 mm), including young-of-the-year Arctic Grayling (Figure 3.2). For instance, angling captured 3.1 times more large (>110 mm) Arctic Grayling per kilometer of sampling than backpack electrofishing. With the appearance of young-of-the-year, the average fork length of Arctic Grayling decreased in later summer (July-August) (mean=119 mm, SD=56, range=23-285 mm) compared to early summer (mean=144 mm, SD=55, range=73-282 mm).

Table 3.1 Results of an ANOVA determining if the average fork length (mm) of Arctic Grayling captured at a site differed by gear type (backpack electrofishing and angling), season (early versus late summer) and year of survey (2008 and 2009). All possible interactions were explored.

Source	df	MS	F	*p*
Model	6	6024.85	4.71	0.002
Gear	1	12976.18	10.14	0.003
Season	1	7659.53	5.99	0.021
Year	1	45.26	0.04	0.852
Gear X season	1	639.18	0.50	0.486
Gear X year	1	66.60	0.05	0.821
Season X year	1	20.61	0.02	0.899

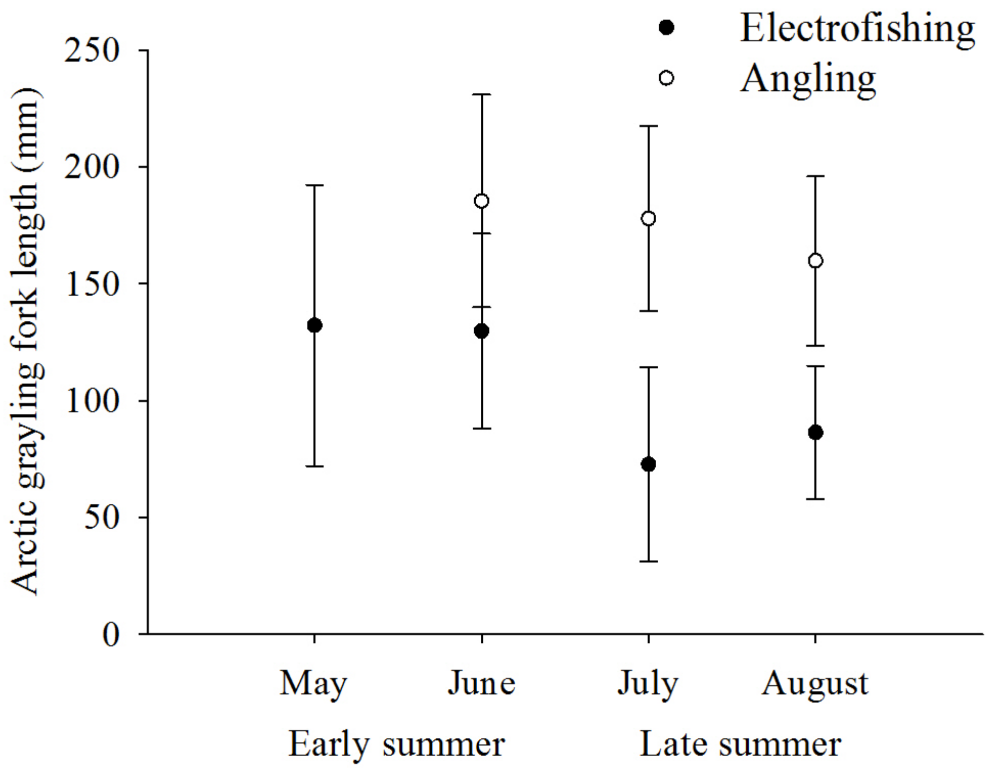

Figure 3.2 Average fork length (mm) of Arctic Grayling caught in 2008 and 2009 by angling and electrofishing in Athabasca River tributary streams in May (electrofishing: n=4), June (electrofishing: n=6; angle: n=3), July (electrofishing: n=4; angle: n=9) and August (electrofishing: n=6; angle: n=5). Data from both summers were combined for this graph.

Arctic Grayling abundance estimates

My results demonstrated that abundance estimates of low-density Arctic Grayling populations using the mark-recapture technique are likely unreliable as only 2 of the 9 sites (22%) recaptured a minimum of four fish at a site (Ricker 1975) (Table 3.2). I therefore rejected the mark-recapture technique as a potential method to estimate low-density Arctic Grayling abundances in my study streams. In addition, at only 2 of the 9 three-pass depletion sites (22%) Arctic Grayling had a capture efficiency (p) greater than my minimum threshold of p=0.20 (Table 3.2). As a result, I considered abundance estimates from this method potentially biased and unreliable.

Table 3.2 Total number of Arctic Grayling recaptured from mark-recapture, and the efficiency of capture (p) of Arctic Grayling captured in the three-pass depletion study in wadeable tributaries (n = 9) in the Athabasca River basin. Sites which met the minimum threshold for reliable and unbiased abundance estimates for mark-recapture (>4 recaptures (Ricker 1975)), and three-pass removal (p>0.20 (Lockwood and Schneider 2000)) are denoted with an 'X'. FM=Freeman River, JC=Judy Creek, SK=Sakwatamau River, CC=Chickadee Creek, TC=Two Creek study sites.

Site	Start date	End date	Total # recaptures	Threshold met?	Probability of capture (p)	Threshold met?
FM1	08/21/08	08/22/08	2		0.00	
JC1	08/22/08	08/23/08	8	X	0.41	X
SK1	08/24/08	08/25/08	0		0.04	
SK2	08/26/08	08/27/08	4		0.03	
FM3	09/05/08	09/06/08	1		0.01	
JC2	07/28/09	07/29/09	6	X	0.25	X
CC2	08/05/09	08/06/09	0		0.02	
CC1	07/24/09	07/25/09	0		0.00	
TC1	07/26/09	07/27/09	0		0.01	

Overall evaluation of sampling techniques

Given my findings, I created a decision flow diagram to guide the selection of sampling techniques to detect and determine the catch-per-unit-effort and abundance estimates of Arctic Grayling. Additionally, I address the detection of Arctic Grayling spawning habitat. To detect low-density Arctic Grayling stream populations, managers must decide if they need to confirm occupancy of young-of-the-year (≤110 mm) or juvenile and adult (>110 mm) fish since the gear type and timing of sampling will vary (Figure 3.3). Backpack electrofishing more reliably detected small young-of-the year fish while angling with dry flies detected juvenile and adult fish. Backpack electrofishing did detect some juvenile and adult Arctic Grayling, but only with moderate efficiency (Figure 3.3). Young-of-the-year Arctic Grayling were only detected in late summer (July and August) (Figure 3.3). Using egg kick surveys, I rarely detected spawning habitat in low-density Arctic Grayling populations despite a large amount of sampling effort (Figure 3.3). Unless the literature-derived minimum thresholds can be met and estimates will be unbiased and reliable mark-recapture and three-pass depletion methods should not be used for assessing abundance estimates of low-density Arctic Grayling (Figure 3.3). Egg kick surveys, mark-recapture and three-pass depletion sampling techniques appear inefficient and unreliable for these types of low-density populations (Figure 3.3).

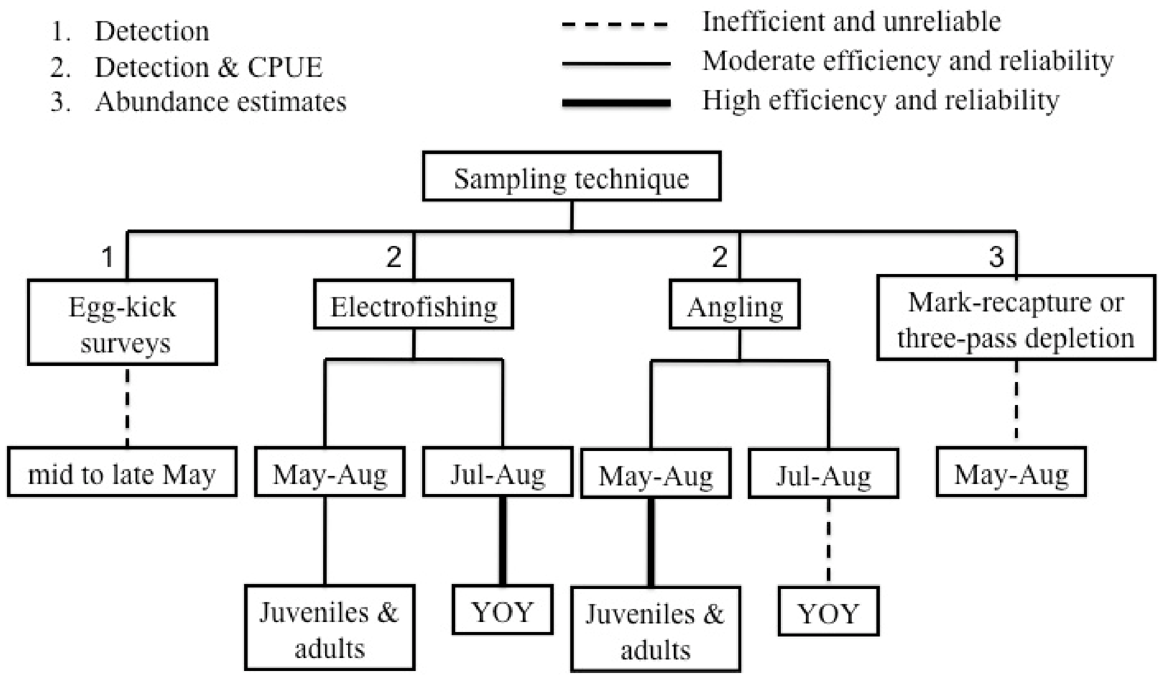

Figure 3.3 Decision flow diagram for detecting and determining catch-per-unit-effort (CPUE) and abundance estimates of Arctic Grayling young-of-the-year (YOY), juvenile and adults in Athabasca River tributary streams using five sampling techniques. The level of efficiency and reliability of each technique is indicated by the strength of the flow diagram line.

Discussion

Despite declining Arctic Grayling populations in Alberta, sampling and monitoring has been under development, and consequently, inconsistent throughout the province. A fundamental step in Alberta's Arctic Grayling management and conservation depends on the availability of accurate and consistent detection and population estimate monitoring protocols. The results presented in my study will provide resource managers with the information required to efficiently assess Arctic Grayling distributions in wadeable tributary streams of the Athabasca River.

In my comparison of three Arctic Grayling sampling techniques, I recommend angling and backpack electrofishing as they had the highest probability of detecting Arctic Grayling occupancy. Although electrofishing is often the preferred sampling method in wadeable streams (Bohlin and others 1989; Bonar and others 2009), my results indicate that angling using dry flies could be an equally effective or complimentary sampling method. In contrast, egg kick surveys appeared to be least successful of the three methods. This is despite the fact that they have been successfully used and detected high egg densities in the House River and Hartley Creek in northeastern Alberta (R.L.&L. 1982a, 1995a, 1995b). Compared to populations in northeastern Alberta, my inefficient egg kick survey findings may be a result of low densities of Arctic Grayling in sampled streams. Further, I may not have located appropriate spawning sites, precise spawning times may not have been well defined, or I should have sampled larger portions of study basins. Given these uncertainties and the potential damage to spawning grounds I recommend that egg kick surveys not be used to identify Arctic Grayling spawning habitat in low-density populations unless absolutely necessary. The conservation and management of a fish species requires knowledge of the distributions of existing populations (Peterson and Bayley 2004). For low-density southern range Alberta Arctic Grayling populations this is especially critical. From the findings of my study, I believe backpack electrofishing and angling are two of the most efficient sampling methods to detect and determine catch-per-unit effort of Arctic Grayling inhabiting streams.

Monitoring programs should direct sampling effort to gears and seasons when target species are most vulnerable (i.e., highest probability of detection) (Willis and Murphy 1996; Noble and others 2007). In my investigation of the biases of the sampling methods, I found that the effectiveness of electrofishing and angling differed temporally and by the size Arctic Grayling. Electrofishing was the most efficient method to capture small young-of-the-year Arctic Grayling, while juvenile and adult fish were more susceptible to angling with dry flies. Contrary to my findings, numerous electrofishing studies have found that larger fish are easier to capture (Bayley and Dowling 1993;

Peterson and others 2004; Buckmeier and Schlechte 2009). Theoretically, electrofishing efficiency should increase with fish length (Bohlin and others 1989). For instance, Peterson and others (2004) found that electrofishing capture efficiency was greatest for the largest size classes of Bull Trout (*Salvelinus confluentus*) and Westslope Cutthroat Trout (*Oncorhynchus clarki lewisi*). In practice however, differences in fish behaviour and the surrounding habitat are more important for capture efficiency (Bohlin and others 1989). The differences of my findings may be a result of Arctic Grayling behavioural avoidance of the electrical field. When disturbed in my study, Brook Trout (*Salvelinus fontinalis*) and Rainbow Trout (*Oncorhynchus mykiss*) were often observed concealing themselves under larger substrate or undercut banks making them easier to capture (pers. obs.). Conversely, upon an electrofishing disturbance, juvenile and adult Arctic Grayling were observed rapidly swimming away either upstream or downstream of sampling crews without stopping to conceal themselves (pers. obs.). Similarly, Ernst and Nielson (1981) found that European Arctic Grayling (*Thymallus thymallus*) avoided the electrical field, resulting in very few captures. Further, Fitzsimmons and Blackburn (2009) found that in the Little Smoky River in west-central, Alberta, Arctic Grayling capture efficiencies were extremely low using a boat electrofisher. Consequently, they relied solely on angling to determine Arctic Grayling abundances. In my study I suspect that angling was a more successful method for detecting juvenile and adult Arctic Grayling since field crews could remain relatively undetected when sampling. Conceivably, increasing the voltage, hertz and pulse width while backpack electrofishing may improve the capture efficiencies of larger Arctic Grayling, but it would be at the risk of seriously injuring or killing smaller Arctic Grayling and other stream fish species. Not surprisingly in my assessment of potential temporal biases, I found that there were no differences in the size of Arctic Grayling captured between 2008 and 2009. Given that sampling methods for Arctic Grayling have not been well defined, I wanted to ensure there were no differences between sampling years. Conversely, I found differences in Arctic Grayling size between early and late summer. I attribute this to the appearance of small young-of-the-year Arctic Grayling in numerous study streams in July. My results have established that researchers must not only be conscientious when choosing their Arctic Grayling sampling gear, but also of the timing of sampling. Often researchers are hindered by economic constraints, which results in the use of a single-gear sampling method without ensuring its effectiveness (Poos and others 2007). My study has demonstrated that sampling Arctic Grayling with a single gear would likely exclude entire life history stages. Fisheries managers should therefore choose the sampling method that will align with their study objectives.

I found that mark-recapture and three-pass removal methods failed to meet my literature-derived minimum requirements. As such, I do not recommend these techniques to determine population estimates of Arctic Grayling since estimates are likely biased and unreliable. Often, I recaptured very few, if any Arctic Grayling during the mark-recapture study. As a result, at many sites I was unable to estimate Arctic Grayling abundances or abundance estimates were likely subjected to statistical bias (Ricker 1975). Further, my abundance estimates derived from the three-pass removal method were likely inaccurate since capture efficiencies (p) were extremely low at most sample sites (Lockwood and Schneider 2000). Of the two methods, mark-recapture abundance estimates are considered more reliable because they are less susceptible to sampler and environmentally induced biases (Zippin 1958; Peterson and Cederholm 1984; Rodgers and others 1992). For instance, when sampling Rainbow Trout, Rosenberger and Dunham (2005) suggested the assumptions of the removal method were not met and decreasing sampling efficiencies over removal passes resulted in underestimated population sizes and overestimates of sampling efficiency. Using the removal method, Rainbow Trout sampling efficiency decreased with increasing stream size and amount of woody debris. Conversely, they concluded that Rainbow Trout abundances could be rigorously evaluated using the mark-recapture method (Rosenberger and Dunham 2005). In my study, unbiased Arctic Grayling abundance estimates were difficult to obtain. Increasing the site sample size and the number of removal passes may have helped increase the reliability of my three-pass removal abundance estimates. This however, would have been more costly and time consuming. Perhaps, Arctic Grayling juvenile and adult abundance estimates derived from the mark-recapture method would have been more reliable if only angling with dry flies was used to sample stream reaches similar to Fitzsimmons and Blackburn (2009). Unfortunately, since my abundance estimates using mark-recapture and three-pass removal techniques were unsuccessful, I was unable to determine Arctic Grayling capture efficiency (p) by gear type and fish size. This information would have allowed fisheries managers to correct catch data and provide a more reliable index of Arctic Grayling population sizes for Athabasca River tributary streams. Although my study failed to determine Arctic Grayling abundances in wadeable streams, these sampling techniques may warrant further investigation since reliable population estimates are critical to future Arctic Grayling management efforts.

Allocation of sampling effort towards the most efficient gears with the highest detection probabilities can minimize the variance in fish collections and lead to a more effective monitoring program (Peterson and Rabeni 1995). From this study I have developed a decision flow diagram to guide future sampling efforts to efficiently and reliably detect and assess Arctic Grayling abundances. Sampling for young-of-the-year should be avoided in the earlier summer months of May and June, as I did not detect them until July. I have shown that electrofishing enables the detection of small young-of-the-year Arctic Grayling, while angling with dry flies is the most efficient means to collect larger juvenile and adult Arctic Grayling. Angling and electrofishing can also be applied to determine a catch-per-unit-effort estimate of Arctic Grayling by size class. Further, mark-recapture and three-pass removal methods are not recommended unless population estimates meet the minimum literature derived criteria and can be considered reliable and unbiased. Undoubtedly, there will be variations in Arctic Grayling densities among my Athabasca River tributary study sites and throughout Alberta. Sampled streams in my study were generally on the southern limit of Alberta's Arctic Grayling range where population declines have been most pronounced. Sampled areas have experienced high angling pressure, forest and energy sector development and extensive culvert networks that have likely resulted in low Arctic Grayling densities. The difficulties I experienced in detecting and capturing Arctic Grayling are likely due in part to the low densities of my study populations. Therefore, the outlined sampling protocols should be used as a guide until similar region-specific studies have also been undertaken. Perhaps, in areas with higher densities of Arctic Grayling, sampling to determine population estimates using mark-recapture and three-pass removal techniques is feasible. My research emphasizes the need to determine the effectiveness of different sampling techniques to detect and determine abundances of Arctic Grayling.

Management implications

Due to differences in sampling effectiveness, the selection of gear types can influence the assessment of fish distributions and abundances (Holland-Bartels and Dewey 1997). My study highlights the value of assessing the suitability of several gear types and determining their biases when sampling low-density Arctic Grayling populations. Inaccurate knowledge of sampling gears can lead to unreliable species population information and poorly made management decisions. Reallocating sampling efforts in streams to both backpack electrofishing and angling can increase sample sizes and lead to a more efficient sampling protocol to detect long-term trends in Arctic Grayling abundances and management actions. The future success of Alberta's Arctic Grayling populations depends on consistent monitoring, management efforts and the formulation of sampling protocols. Standardized protocols would allow fisheries biologists to concentrate resources on improving fish populations rather than performing inefficient monitoring. It is my hope that this study will encourage fisheries managers to perform similar region-specific assessments and will provide an initial step to the formulation of rigorous Arctic Grayling standard sampling protocols.

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Summary

Despite the many thousands of culverts proliferating in Alberta's landscape, a surprisingly few studies have investigated their effects on stream fish assemblages. The results of my research highlight how culverts are partially or entirely impeding upstream passage of several Alberta foothills fish species and are altering adjacent stream habitat. Currently, the majority of foothill culvert crossings are in direct violation of the federal *Fisheries Act* that states that watercourse crossings should provide passage to all fish species at all life history stages. Further, findings of my research emphasize the need to consider gear biases and effectiveness when sampling Arctic Grayling, a species at the peril of road-building practices in the region.

Hanging and steep culverts with high water velocities are altering fish communities and fragmenting aquatic habitat. This research has demonstrated that culverts are obstructing upstream passage of Burbot and appear to partially impede upstream movements of minnows, suckers and Spoonhead Sculpin. Although the long-term ecological effects of this fragmentation are speculative, our study suggests that shifts to community structure (i.e. through the potential competitive release of Rainbow Trout upstream of culverts and possibly the absence of Arctic Grayling at barrier culverts) have already occurred. Impassable culverts require immediate remediation or, ideally, replacement with bridges, which retain the natural streambed and stream characteristics. Unfortunately, culverts are often installed with short sightedness and initial installation costs in mind. Additionally, very few are maintained to prevent future fish passage issues. Obviously, there is a pressing need for regulatory agencies to begin to monitor and prosecute offenders under the *Fisheries Act*. It is my recommendation that the solutions suggested for mitigating anthropogenic stream barriers in Park (2006) be adopted.

The difficulties encountered while sampling Arctic Grayling combined with the paucity of peer-reviewed literature on the topic prompted an investigation into the most effective Arctic Grayling sampling techniques. The findings of my research suggest that gear and temporal biases exist. I suggest that lotic Arctic Grayling population monitoring would benefit from the introduction of a provincial standard monitoring protocol. This would reduce the human resources and financial costs associated with ineffective monitoring and allow fisheries managers to make more consistent and informed management decisions for rapidly declining Arctic Grayling populations.

This study provides only a short duration assessment (3 years) that quantifies the negative impacts of culverts on Alberta foothill stream fish assemblages. More work is highly recommended. Future studies would benefit by directly monitoring tagged fish movements through watercourse crossings structures to confirm the findings of this research. This would allow for a more intensive direct study of fish movements through watercourse crossing structures. Additionally, species for which my initial sampling did not allow adequate inference (e.g. Arctic Grayling) should be revaluated in the future using our sampling protocol recommendations. Lastly, since streams are part of larger dendritic watersheds, fish passage studies needs to begin to investigate the landscape-level alterations that culverts networks have on the life history processes, distribution and abundances of fish species.

Without remediation and replacement, culverts will continue to be installed in Alberta lotic systems. Although the full ecological implications of habitat fragmentation are currently not understood, it would undoubtedly lead to further alterations of aquatic ecosystems and unforeseeable and irreversible long-term consequences.

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