Stream Crossings in the Western Boreal Forest: Assessing Impacts and Prioritizing Restoration for Native Freshwater Fishes

by

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Abstract

Growing anthropogenic development in response to rising demands for natural resources is a major concern for freshwater fish, particularly in resource rich regions such as Canada's boreal forest. Expanding networks of industrial resource roads has led to the installation of hundreds of thousands of stream-crossing structures that are becoming increasingly common anthropogenic features on North American riverscapes. These structures can reduce available fish habitat, deteriorate instream habitat, and disrupt ecological connectivity by acting as barriers to fish and aquatic organism movement. My objectives were (*i*) to determine the extent to which commonly installed stream crossings affect stream fish communities in a boreal forest watershed, and (*ii*) to assess the application of operational research tools that utilize an optimization framework for mitigating the effects of fragmentation on native freshwater fish and informing restoration planning in the boreal forest. I used mixed-effects modeling and multivariate analyses to determine the effects of stream crossings from 33 culverted, bridged, and reference streams in an industrializing region of the boreal forest in west-central Alberta. Instream habitat characteristics such as mean depth, percent fines, and turbidity showed significant between- as well as within-stream differences among stream crossings. I found that the majority of fish species exhibited significantly lower densities ($n \cdot m^{-2}$) in upstream habitats as compared to downstream habitats, including a significant reduction in Slimy Sculpin densities in streams with culverts compared to reference streams. Multivariate tests showed that fish assemblages differ as a function of stream type and location. The prioritization method utilized in this study suggests that large gains in potential connectivity could be realized with a moderate investment (~$200K to $500K). I found that the operational research tool can be used to develop cost-benefit curves from the study watersheds, which can be used to minimize overall restoration costs to achieve particular management objectives in watersheds of interest, as well as provide defendable evidence for budget planning to regulators and decision-makers. Additionally, varying model parameters that account for species-specific differences in habitat use (e.g. dispersal distance) affected prioritization solutions, and should be considered in future prioritization analyses. In addition to effecting fish passage and stream connectivity, my results suggest that culverts may also be altering fish habitat, further contributing to large-scale changes in stream fish communities in the boreal forest. Further, my research highlights the efficacy of a novel, easy to use optimization-based barrier prioritization toolset that has minimal data requirements, is applicable to both stream-resident and long-range migratory species, and significantly reduces the mathematical and technical expertise needed to perform relatively complex optimization analyses.

Preface

This thesis is an original work by Bryan M. Maitland. The research presented within received research ethics approval from the Animal Care and Use Committee under: “Stream Crossings,” animal care protocol no. AUP-00000757 on 7 July 2013. Fish collection was completed under approved fish research licenses (13-1009 & 13-1011). A version of Chapter 2 of this thesis has been published as: “Maitland, B.M., Poesch, M., Anderson, A.E., Pandit, S. In Press. Stream crossings drive changes in fish assemblages in an industrializing Boreal watershed. *Freshwater Biology*. DOI: 10.1111/fwb.12671.” B. Maitland was responsible for the field study design, data collection, analysis, and manuscript composition. S. Pandit assisted with data analysis and manuscript edits. M. Poesch and A. Anderson were the supervisory authors and were involved with concept formation and manuscript composition.

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Chapter I: Introduction

Expanding anthropogenic development across the globe is a major concern for freshwater fish. Rising demands for fossil fuels, forest products, natural gas, and minerals has driven the expansion of road networks across the planet (Laurance and Balmford 2013), particularly in resource rich regions such as Canada's boreal forest. These road networks have in turn lead to the installation of hundreds of thousands of stream-crossing structures which are becoming increasingly common anthropogenic features on North American riverscapes (Tchir et al. 2004, Park et al. 2008, MacPherson et al. 2012, Miller 2012, Januchowski-Hartley et al. 2013). These structures can reduce available fish habitat (Gibson et al. 2005), deteriorate instream habitat (Eaglin and Hubert 1993), and disrupt ecological connectivity by acting as barriers to fish and aquatic organism movement (Warren and Pardew 1998, Park et al. 2008, MacPherson et al. 2012).

Connectivity is a concept that defines linkages between and among different ecosystem elements in both space and time, and is integral to the healthy functioning of ecological systems (Pringle 2003, Fullerton et al. 2010). Hydrologically, connectivity is parsed into four dimensions: longitudinal (i.e. along the stream channel), lateral (i.e. stream channel to riparian zones), vertical (i.e. water surface to hyporheic zone), and temporal (Ward 1989). Longitudinal connectivity plays a vital role in the maintenance and resilience of many fish species (Fausch et al. 2002) as biotic and abiotic ecosystem components (i.e. organisms, sediment, woody debris, etc.) need to be able to move up and down a system. Accordingly, longitudinal connectivity is considered the most important dimension for freshwater fish as it provides for access to spatially and temporally separated essential habitat (e.g. spawning and foraging grounds, seasonal refugia, etc.; Lucas et al. 2001, Fausch et al. 2002).

Exploring the connectivity concept in riverine systems raises many challenges due to their high degree of spatial and temporal complexity (in contrast to terrestrial ecosystems; (Fullerton et al. 2010). However, recent work has highlighted the importance of considering riverscapes as dendritic ecological networks (DENs; Campbell Grant et al. 2007). In these systems, ecological processes (e.g. dispersal, population growth, and community interactions) are carried out within the branches of the network (i.e. habitat patches) with nodes serving as transfer points. Conversely, in terrestrial systems, nodes serve as habitat patches connected by multiple movement pathways. Thus, freshwater fish and their distribution within DENs tend to respond strongly to connectivity loss as dispersal and movement routes are constrained to a single path (i.e. the stream channel; Fagan 2002, Campbell Grant et al. 2007).

Freshwater fish distributions also tend to respond to habitat alterations (Gorman and Karr 1978, Karr 1981). Changes to physical habitat structure, hydrological processes, or water quality from stream crossing structures may thus have the potential to reduce population densites or alter community assemblages (Gordon et al. 2013, Ottburg and Blank 2015). Increased sediment delivery to streams, for example, is a major environmental impact of road development and stream crossing installation (Waters 1995, Ottburg and Blank 2015). Increases in suspended and deposited sediments have indeed been shown to have deleterious effects on feeding behavior, spawning success, and species composition and richness (Chapman et al. 2014). Alterations to fish habitat are also of concern as they may contribute to fish species homogenization by facilitating the dominance of tolerant species (e.g. Brook Stickleback *Culaea inconstans*) over sensitive species in degraded streams (e.g. Bull Trout *Salvelinus confluentus*; Walters et al. 2003, Rahel 2010).

Loss of connectivity can be considered a primary driver shaping the distribution of species in riverscapes (MacArthur and Wilson 1967, Levin 1974, Park 2006). When instream barriers compromise longitudinal connectivity, flow regimes can be altered which can lead to serious habitat modifications. These changes can in turn affect existing biotic communities in myriad ways. Impacts of isolation are most often observed in river systems crossed by dams and weirs (Barry 1990, Baum 1994, Morita and Yamamoto 2002, Mueller et al. 2011), though research done in recent years has highlighting the urgent need to consider the impact of connectivity loss on smaller streams crossed by roads where culverts and bridges are installed (Warren and Pardew 1998, MacPherson et al. 2012, Januchowski-Hartley et al. 2013, Neeson et al. 2015). While these impacts are especially pronounced for migratory species as downstream barriers completely isolate diadromous fish from their upstream spawning grounds, there is ample evidence to suggest impacts to potadromous and stream-resident fish are widespread (Belford and Gould 1989, Fagan 2002, Morita and Yamamoto 2002, Park 2006, Bouska and Paukert 2010, MacPherson et al. 2012, Perkin and Gido 2012, Diebel et al. 2014).

Northern environments are variable and stressful locations for fish and have received little attention regarding the effects of connectivity loss. Small-bodied Cyprinids, Catostomids, and Gasterosteids that range from small stream-resident water-column species to larger potadromous species dominate streams in Canada’s western boreal forest and foothills regions. Also present are populations of Arctic Grayling (*Thymallus arcticus*), Bull Trout, and Mountain Whitefish (*Prosopium williamsoni*). These species are adapted to living in harsh, northern environments where extreme conditions drive ecosystems dynamics. For example Park (2006) and Park et al. (2008) hypothesized that, similarly to desert environments (Fagan et al. 2002), environmental dynamics in the boreal forest may be a powerful factor regulating fish populations as long periods of ice cover combined with drought, fire, and other stochastic disturbances are very common. Environmental dynamics in northern environments thus contribute to the vulnerability of northern fishes to impacts from industrial resource development (Reist et al. 2006, Cott et al. 2015). Accordingly, assessing these impacts is imperative to inform management and restoration decisions (Brandt et al. 2013).

Additionally, systematic restoration efforts are needed to offset the growing deleterious impacts of expanding human resource development. Restoring longitudinal connectivity for fish at the watershed level is a key component in this process, as has been described as a priority in restoration planning (Roni et al. 2002, Roni et al. 2008). In the Canadian province of Alberta, traditional management approaches and decades of non-compliance with Federal and Provincial regulations have failed to prevent to development of problem stream crossings, and have in turn led to declines in fish population (Park et al. 2008, MacPherson et al. 2012, AESRD 2015) from effects of fragmentation (Walker 2005) and increased sediment delivery (ASRD 2009).

Restoration of longitudinal connectivity can be approached using a variety of methods including barrier repair or retrofitting, though barrier removal is by and large the most effective method to remedy fragmented habitats (Roni et al. 2002). Unfortunately, the ability to achieve restoration objectives is often stymied by high costs of remediation efforts and budget limitations (Bernhardt et al. 2005). Given the limited amounts of time and money available for remediation efforts, formal methods to prioritize restoration actions are needed to maintain fish populations (Fullerton et al. 2010, O’Hanley 2011). With the growing appreciation of the importance of connectivity in dendritic networks (Fagan 2002, Poole 2002, Fullerton et al. 2010), prioritization techniques have emerged as a major theme in riverscape restoration (Roni et al. 2002, Nilsson et al. 2007, Kemp and O'Hanley 2010). Until recently however, many tools for prioritizing connectivity restoration have been available only in regional and government reports making them difficult for decision-makers and practitioners to access (Kemp and O'Hanley 2010, Branco 2013). Additionally, most rely on simple scoring and ranking methods (Taylor and Love 2003, Karle 2005, Kocovsky et al. 2008, WDFW 2009, Lawson et al. 2010), and research has since highlighted the inefficiencies of these methods (O'Hanley and Tomberlin 2005), and explored the benefits of operational research approaches (e.g., heuristics and optimization) for prioritizing barrier removal in an systematic framework (Kuby et al. 2005, O'Hanley and Tomberlin 2005, Zheng et al. 2009, Kemp and O'Hanley 2010, O'Hanley 2011, O'Hanley et al. 2013, Segurado et al. 2013, Branco et al. 2014, Diebel et al. 2014, King and O'Hanley 2014, Neeson et al. 2015).

Understanding the degree to which these stream crossings are affecting fluvial fish communities and how management actions can mitigate potential impacts to stream connectivity is needed to guide best management practices, particularly with expansion of activities from extractive industries across Alberta's boreal landscape. My objectives were to determine the extent to which common stream crossings affect stream fish communities in a boreal forest watershed, and to assess the applicability of a recently developed stream crossing prioritization method for mitigating fragmentation effects and informing restoration planning. A second objective was to gather baseline data for a Before-After-Control-Impact study to validate the effectiveness of barrier removal for restoring longitudinal connectivity. Chapter 2 examines the effect of stream crossings on instream habitat and fish communities in an industrializing boreal watershed in west-central Alberta. Chapter 3 considers the use of the isolation concept for prioritizing barrier removals and remediation through a case study and provides recommendations for use in Alberta. Chapter 4 summarizes and discusses the implications for conservation and management, and presents recommended future research questions.

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Chapter II: Industrial Road Crossings Drive Changes in Community Structure and Instream Habitat for Freshwater Fishes in the Boreal Forest

Abstract.—Stream crossing structures are an increasingly prevalent anthropogenic feature on North American riverscapes, particularly in watersheds affected by industrial resource development in sensitive boreal environments. If improperly managed, stream crossings have the potential to alter fish habitat and impede fish movement. This study assessed instream habitat characteristics and fish communities from 33 culverted, bridged, and reference streams in an industrializing region of the boreal forest in west-central Alberta. Mixed-effects modeling and multivariate analysis were used to determine impacts of stream crossings at three scales: whole-stream scale, within-stream scale and the interaction of scales. Instream habitat characteristics such as mean depth, water velocity, percent fines, turbidity, water temperature, and dissolved oxygen showed significant between- as well as within-stream differences among stream crossings. The majority of fish species exhibited significantly lower densities ($n \cdot \mathrm{m}^{-2}$) in upstream habitats as compared to downstream habitats, including a significant reduction in Slimy Sculpin densities in culverted streams. Multivariate tests corroborated these results, showing that fish assemblages differ as a function of stream stream type and location. This study suggests industrial stream crossings influence abiotic habitat characteristics in freshwater ecosystems, restrict biotic connectivity, and affect fish community structure at the whole-stream and within-stream scales. In addition to effecting fish passage and stream connectivity, my results suggest that culverts may also be altering fish habitat, further contributing to large-scale changes in stream fish communities in the boreal forest. With expanded development expected in much of North America's boreal forest, mitigation measures that limit impacts from stream crossings are needed to ensure proper ecosystem function in freshwater systems.

2.1 Introduction

Freshwater ecosystems are globally imperiled by threats from anthropogenic development (Dudgeon 2014). As humans continue to alter the landscape (Vitousek et al. 1997), their footprint is rapidly growing and causing alterations to freshwater environments and fish communities (Maitland 1995, Schindler 2001). Expansion of this footprint is predominately driven by rising demands for fossil fuels, natural gas, minerals, and forest products (Laurance and Balmford 2013) resulting in the development and proliferation of linear features (Cott et al. 2015). In North America's boreal forest region—an oligotrophic biome comprising 58.5% of Canada's landmass (Anielski & Wilson, 2009) and containing 25% of the world's remaining intact forests (Lee *et al.*, 2003)—the amount and extent of exploration and development activities has increased in recent years (White et al. 2011, Kreutzweiser et al. 2013) and is expected to affect fish and fish habitat in myriad ways. A direct result of the proliferation of linear features such as resource roads has been the installation of hundreds of thousands of stream crossing structures (Miller 2012, Januchowski-Hartley et al. 2013). These crossings are particularly prevalent on small streams in forested boreal watersheds experiencing industrial resource development (Prévost et al. 2002, Scrimgeour et al. 2003, Park et al. 2008) where culverts and single-span bridges are the dominant structures used (Park et al. 2008, MacPherson et al. 2012a). Considering the vulnerability of sensitive northern fishes to impacts from resource development (Reist et al. 2006), understanding how stream crossings impact aquatic ecosystems in the boreal region is imperative for making informed management and restoration decisions (Brandt et al. 2013).

Crossing structures such as culverts and bridges alter stream systems through changes to physical habitat structure, hydrology, and water quality (Ottburg and Blank 2015), potentially leading to species loss and altered communities (Gordon et al. 2013). Habitat quality is affected from sediment mobilization and deposition which can affect fish both directly (health and behavior) and indirectly (changes to habitat). For example, a meta-analysis on the effects of sediment on freshwater fish showed that increases in suspended and deposited sediments had negative effects on feeding behavior, spawning success, and species composition and richness (Chapman et al. 2014). Deposited sediments, in turn, can have significant impacts on fish occurrence and distribution as species differ in their substrate preferences and requirements. For example, benthic species such as Slimy Sculpin *Cottus cognatus* and Salmonids such as Bull Trout *Salvelinus confluentus* are highly influenced by substrate composition; if excess fine sediment enters a stream it can form a mat over streambeds of coarse rocky substrate and severely degrade habitat suitability (Gordon et al. 2013). Alterations to fish habitat are also of concern as they may contribute to fish species homogenization by facilitating the dominance of tolerant species (e.g. Brook Stickleback *Culaea inconstans*) over sensitive species (e.g. Bull Trout) in degraded streams (Walters et al. 2003, Rahel 2010). For instance, Bull Trout are expected to be extirpated from 24% to 43% of stream reaches that support them in a boreal forest watershed by 2025 due to their negative relation to forest harvest and percent fines (Ripley et al. 2005).

Stream crossings such as culverts can act as barriers to instream fish movement, thus disrupting ecological connectivity, and reduce the quantity of available suitable habitat. Improperly designed or installed culverts result in physical barriers (e.g. outlet drop, velocity, slope) to fish movement (Belford and Gould 1989, Norman et al. 2009, MacPherson et al. 2012a) and prevent fish from accessing different habitats necessary for life history processes or colonization (Warren and Pardew 1998, Nislow et al. 2011, MacPherson et al. 2012a). As road networks expand, stream crossing densities have grown rapidly in stream networks (Kemp and O'Hanley 2010). For example, in the Great Lakes Basin, Januchowski-Hartley et al. (2013) found ~268,000 potential stream crossing barriers, of which they estimated 36% to be impeding fish movement. Similar trends are found in western Canada. In British Columbia, there is an estimated 320,000 stream crossings along ~550,000 km of resource roads (Miller 2012). In Alberta's boreal forest, it is estimated that several thousand culverts are fragmenting tens of thousands of stream kilometers (Park et al. 2008). Stream habitat fragmentation has additionally been shown to alter fish assemblages (Perkin and Gido 2012), as well as reduce population resilience to environmental disturbance and reduce genetic mixing (Torterotot et al. 2014).

In addition to growing anthropogenic stressors, northern boreal environments host a range of significant abiotic stressors that will likely intensify human impacts (Park *et al.*, 2008; Schindler & Lee, 2010). Environmental dynamics in the boreal forest, such as long, annual periods of ice cover combined with other relatively common stochastic events, such as drought and fire, have been shown to act as a strong regulator of fish population dynamics (Park, 2006; Park *et al.*, 2008). Fish sensitivity to environmental dynamics is exacerbated in northern environments, and the relationship between habitat size and population persistence may be especially strong (Park et al. 2008). Thus, the influence of environmental dynamics in northern environments contributes to the vulnerability of northern fishes to habitat fragmentation and degradation impacts arising from industrial resource development (Reist *et al.*, 2006; Cott *et al.*, 2015). Given the rapid increases in expansion of human activities in the boreal forest over the past four decades (Schindler & Lee, 2010), assessing the impacts of fragmentation from industrial road crossings is imperative to informing adaptive management, restoration decisions, and future land-use planning (Brandt *et al.*, 2013).

Relative to stream systems in Appalachia, the North American Prairies, Eastern Europe, and the Pacific Northwest, boreal stream systems have received less attention in regards to the direct influences of stream crossings on fish populations and instream habitat. This is an important research gap as the North American boreal forest is facing increasing pressure from industry (Henry 2005, Schindler and Lee 2010), and there are signs that resource development may negatively impact freshwater fish through increased sediment loads (Ripley et al. 2005, Scrimgeour et al. 2008), or problem culverts (Browne 2007, Park et al. 2008). The aim of this study is to assess the influence of culverts and bridges on instream habitat characteristics and stream fish communities in a boreal forest watershed undergoing expanding resource development. I compared physicochemical habitat characteristics and fish populations among three stream types: (*i*) streams crossed by culverts, (*ii*) streams crossed by bridges, and (*iii*) reference streams without culverts or bridges; stream types were further stratified by location within the stream (i.e. upstream or downstream of crossing structures). Fish response was measured in terms of fish density and species richness as suggested by Nislow et al. (2011). I assessed four hypotheses within the context of this study. (*i*) Instream habitat characteristics differ significantly among stream types. This would suggest whole-stream scale impacts on fish habitat, perhaps from altered flow regimes, changes to bed morphology or sinuosity. (*ii*) Total fish density and species richness would be significantly lower in culverted and bridged streams as compared to reference streams. This would suggest whole-stream-scale impacts of crossings on fish populations, likely from stream-wide extirpations induced by habitat fragmentation (Hanski et al. 1995, Favaro et al. 2014). (*iii*) Habitats upstream of culverts would have significantly lower fish densities and richness than downstream habitats. This would provide evidence of impeded dispersal at the within-stream scale (Bouska and Paukert 2010, Nislow et al. 2011, Favaro et al. 2014). (*iv*) Fish assemblages differ among stream types, and in particular, culverted streams will be associated with species more tolerant to increased levels of sediment (e.g. *Chrosomus* spp., Brook Stickleback, or Lake Chub *Couesius plumbeus*).

2.2. Methods

The boreal forest has long, cold winters and short, warm summers (mean annual temperature 0.2° C – 1.1° C), and contains a principal forest type of closed-canopy mixedwoods (NRC 2006). This study focused on tributary streams in the Simonette watershed (5,390 km^2) located in the Upper Peace River Basin of west central Alberta, Canada (Fig. 2.1). The watershed is located predominately within the boreal forest Natural Region of Alberta where it flows northward and drains into the Smokey River, and eventually the Peace River (NRC 2006). The streams in the Simonette are characterized by low gradient, meandering reaches (Scrimgeour et al. 2008) which feed the Simonette mainstem. This region has experienced high levels of land-use disturbance from intensive forest harvesting and oil and gas exploration/extraction activities (e.g. roads, forest cutblocks, oil and gas well sites, pipelines; Scrimgeour, Hvenegaard & Tchir, 2008; White *et al.*, 2011). In 2003, industrial activity accounted for 18.7% of land disturbance in the Simonette; of that, forest harvest accounts for 84%, with roads, pipelines, and seismic lines making up the remainder (Scrimgeour et al. 2003). The aggregate area of industrial activity has increased in recent years (White et al. 2011), and is expected to further grow over the next century (Kreutzweiser et al. 2013). Indeed, overall mean road density in individual sub-watersheds in the Simonette has increased from 0.33 ± 0.02 km·km^{-2} (Scrimgeour et al. 2008) to 0.47 ± 0.29 km·km^{-2} between 2008 and 2015. This study focused on watercourses south of the confluence of the mainstem Simonette and the Latornell Rivers because the objective was to understand the impacts of resource road stream crossing structures, and land use in the northern portion of the watershed is dominated by private land and agriculture.

Small streams in the boreal forest of Alberta support a relatively depauperate ichthyofauna as a result of its northern location and harsh climate (Nelson and Paetz 1992). Provincial records indicate that twenty-two species of fish representing nine families have been recorded in the Simonette River watershed (Alberta Fish & Wildlife Management Information System, accessed April 2015). Common fishes included those from the family Cyprinidae (Northern Redbelly Dace *Chrosomus eos*; Finescale Dace *Chrosomus neogaeus*; Lake Chub; Pearl Dace *Margariscus margarita*; Longnose Dace *Rhinichthys cataractae*; Redside Shiner *Richardsonius balteatus;* Northern Pikeminnow *Ptychocheilus oregonensis*), Cottidae (Slimy Sculpin; Spoonhead Sculpin *Cottus ricei*), Catostomidae (White Sucker *Catostomus commersoni*; Longnose Sucker *Catostomus catostomus*), and the salmonids (Arctic Grayling *Thymallus arcticus;* Bull Trout*;* Mountain Whitefish *Prosopium williamsoni*). Less common species include Brook Stickleback, Trout-perch *Percopsis omiscomaycus*, Burbot *Lota lota*, Walleye *Sander vitreus*, and Northern Pike *Esox lucius* (Nelson and Paetz 1992, Joynt and Sullivan 2003, Scrimgeour et al. 2003, Scrimgeour et al. 2008).

2.2.1 Study Design.—I used a balanced spatial comparison with replication among and within streams (McLaughlin et al. 2006, Mueller et al. 2011, Favaro et al. 2014) to investigate patterns in physicochemical habitat characteristics and fish communities. I sampled 33 watercourses (see supplementary material, Table A1) of similar size (order 2-4; Strahler 1957) during summer low flow, 2013. Sample stream reaches were selected using a stream crossing inventory completed in the watershed in 2001 (Johns et al. 2004), field scouting, and local knowledge of crossings and fish occurrence from government employees. Streams crossed by culverts (n = 11), crossed by bridges (n = 11) and reference streams (n=11) were evaluated (hereafter "stream type"). Stream types were stratified into upstream and downstream reaches (i.e. above and below crossing structures, and above and below a hypothetical crossing structure on reference streams); yielding 66 sample reaches (Fig. 2.1). Each sample reach was 300 m in length (AESRD 2013) and contained seven transects spaced 50 m apart where physicochemical habitat measurements were taken (Fig. 2.1). Pertinent physical features of crossings were measured (e.g., culvert hang height, outlet water velocity, slope, length). Potential fish passability ratings were determined (i.e. complete barrier to all fish, partial barrier to some fish, no barrier) for each culvert based on physical measurements of the structure (AESRD 2014). All culverts were closed-bottom corrugated metal pipes and all but two were devoid of substrate within. All bridges were single-span with either concrete or wood abutments. While reference streams did not have a stream crossing within or downstream of sampling, there are four instances in which a reference location is downstream of a culvert, one instance in which a culvert is located below another culvert, and one in which a bridge is located below a culvert. While the bridge located downstream of the culvert does not confound fish passage inference, the culvert located below another culvert may. This factor was impossible to avoid given to the non-random distribution of stream crossings across the riverscape, along with access and logistical constraints.

2.2.2 Physicochemical habitat characteristics.—I used a modified habitat assessment procedure to evaluate common physicochemical habitat parameters known to influence fish presence (Gorman and Karr 1978). Each stream was sampled consistently with respect to rain events to reduce their influence on water chemistry observations (i.e. sampled at least 24 h after rainfall events). At each transect, I measured wetted width (m), bankfull width (m), and mean water depth (averaged between three point measurements at 25%, 50%, and 75% of cross-sectional width). Temperature (°C), dissolved oxygen (DO; mg·L^{-1}), pH, velocity (m·s^{-1}), specific conductance (μs·cm^{-1}), and turbidity (NTU) were measured. Temperature, DO, pH, and specific conductance were measured using a handheld multi-probe meter (YSI, Yellow Springs, Ohio), and velocity with a handheld acoustic Doppler velocimeter (SonTek / Xylem Inc., San Diego, California). The velocity at each transect was recorded as the average of three point measurements (at 25%, 50%, and 75% of cross-sectional width). Substrate composition was visually classified by separating substrate components within 1·m^{-2} quadrats into four categories based on size (<2 mm [fines], 2-64 mm [gravel], 64-256 mm [cobble], >256 mm [boulder]) and the proportion of each within each transect was estimated (MacPherson et al. 2012a). The proportion of instream habitat types (i.e. pool, riffle, and run habitat) was qualitatively estimated for sections between transects (AESRD 2013). Substrate and habitat type measurements were all done by a single observer to reduce observer bias. Substrate components were subsequently grouped into a “Fines” category and a “Coarse Rocky” category (gravel, cobble, and boulder). Physicochemical variables were averaged for each stream reach from the seven transects. I measured physical characteristics of culverts related to fish passage including, diameter, length, culvert slope, and hang height (Table A2) as outlined in AESRD (2014).

2.2.3 Fish Data Collection.—During low-flow of summer and early fall 2013, I captured fish using single-pass backpack electrofishing in an upstream direction (Reid et al. 2009; Smith Root LR-24 Backpack Electrofisher with one dip-netter). Consistent with other studies (McLaughlin et al. 2006, Nislow et al. 2011) and given site field assessments, I established a buffer area (25 m; Fig. 2.1) that separated the crossing structures from sample reaches to reduce the influence of local habitat alterations from the road and crossing structures (e.g. plunge pools).

Direct current voltage and frequency settings were set in accordance with variations in stream conductivity. Mean voltage was 296 V (range: 250-330 V) at 30 Hz and 4 or 6 millisecond pulse width. Average time fished was 16.1 min (± 0.98 SD). Care was taken to ensure adequate sampling of all habitat types (i.e. riffles, runs, pools, undercut banks, etc.). All fish collected were identified to species, enumerated, measured for fork length (mm), and released alive. Vouchers specimens of small-bodied species were retained for laboratory confirmation of identification. Electrofishing effort was recorded in seconds per 300 m reach (mean = 963.9 ± 58.6 $\mathrm{s \cdot reach^{-1}}$).

2.2.4 Data Analyses.—I used a combination of mixed-effects modeling and multivariate analyses to examine differences in patterns of variation observed in the physicochemical habitat characteristics and fish metrics among and within streams. I grouped together salmonids (Arctic Grayling, Bull Trout, Mountain Whitefish), *Chrosomus* spp. (Northern Redbelly Dace, Finescale Dace, Northern Redbelly Dace × Finescale Dace), and *Catostomus* spp. (White Sucker, Longnose Sucker) because of small sample sizes (salmonids), broadly similar life histories (salmonids, *Chrosomus* spp., *Catostomus* spp.), field identification errors (*Chrosomus* spp.), and they gave comparable quantitative results (data not shown). For each stream-reach, I computed variable means for each habitat parameter. Total fish density ($n \cdot \mathrm{m}^{-2}$) and species richness were computed for each stream reach. Relative densities ($n \cdot \mathrm{m}^{-2}$) were then calculated for each species for each stream reach.

Given the nested structure of the data (i.e. stream location is nested within stream type), I used the analytical framework developed by Favaro et al. (2014). Mixed-effect models were used in three steps. In each model, I examined two main effects (i.e. stream type, location) and their interaction (i.e. stream type × location). Stream type was treated as a fixed factor with three levels (culvert, bridge, reference) and stream location with two levels (upstream, downstream). To account for stream-specific random variation in environmental variables, a random intercept term for each stream was included in all models (Zuur et al. 2009). In interpreting model outputs, I follow Favaro et al. (2014): a main effect for stream type would indicate whole-stream scale impacts (from stream-wide extirpations); a main effect for stream location would indicate general differences in upstream versus downstream locations (from gradients in fish distributions along the stream corridor); an interaction effect would suggest impacts of culverts and bridges at the within-stream scale. All models were built in the open-source software R (R Core Team 2014) with the *lme* function of the nlme package (Pinheiro et al. 2015) and the *glmmadmb* function of the glmmADMB package (Skaug et al. 2015). Statistical significance was declared at $\alpha = 0.05$. Residual plots of all models were visually inspected to ensure variance homoscedasticity.

2.2.5 Mixed Effects Modeling.—Linear mixed effects models (LMEs) were first used to test for differences in the physicochemical habitat characteristics among stream reaches. Multiple habitat variables were log transformed (depth, velocity, fines, pool habitat, turbidity) to ensure normal residuals. Each variable was modelled as a function of stream type, stream position, the interaction of stream type and location, and a random intercept for stream-specific effects. Following this analysis, I used generalized linear mixed models (GLMMs) to test whether significant physical habitat variables affected individual fish responses. If significant, these variables would need to be accounted for in subsequent models testing the effect of stream crossings on fish response metrics. I tested if mean depth, wetted width, velocity, percent fines, or percent coarse substrate (physical variables commonly associated with fish distributions; Favaro *et al.*, 2014) affected fish densities. For GLMMs, I specified a negative binomial error structure for species counts and a Poisson error structure for species richness data (Zuur et al. 2009). In addition, I included an offset for reach area in all models (log m^2) except for the species richness model (O'Hara and Kotze 2010) to present count data as densities and account for different areas sampled (Favaro et al. 2014).

The effects of stream crossings on overall fish density, richness, and species-specific densities were then examined using GLMMs. Fish responses were modelled as a function of stream type, stream location, the interaction of stream type and location, a random intercept for stream-specific effects, and with a reach area offset. Wetted width and mean depth were included as fixed covariates as they were found to influence species-specific density responses (Table A6).

2.2.6 Multivariate Analysis.—Multivariate analyses were used to explore differences at the assemblage level and assess the importance of physicochemical habitat characteristics on the observed fish communities among sites (Mueller et al. 2011, Favaro et al. 2014). Bray-Curtis dissimilarly indices (Clarke and Warwick 2001) were calculated for all stream pairs based on fish density data and ordinated using nonmetric multidimensional scaling (NMDS). NMDS is unconstrained by environmental variables and thus reflects only dissimilarities between species composition data. Environmental gradients were identified by fitting, as regressed vectors, the physicochemical habitat variables to the ordination in a second step. Vectors are interpreted as the direction of environmental change (i.e. of a gradient); the length of which indicates the strength of the correlation between the NMDS configuration and environmental variables. Significance tests for these correlations were done using permutation tests with 10,000 randomly permuted correlations. I used two-way permutational multivariate analysis of variance (PERMANOVA; Anderson 2001) on rank dissimilarities from the Bray-Curtis dissimilarity matrix to test for significant differences in species assemblages among stream types, within streams, and at the interaction between stream type and location. The PERMANOVA model included a strata term to account for stream-specific differences. Multivariate analyses were done in R (R Core Team, 2014).The NMDS analysis was done using the *nmds* function in the ecodist package (Goslee and Urban 2007). Permutation tests were done with the *envfit* function and PERMANOVA analysis with the *adonis* function in the vegan package (Oksanen et al. 2013) with 9,999 permutations.

2.3 Results

I captured 2,987 individuals representing 16 species and 7 families among 66 sample stream reaches (Table A3). Slimy Sculpin was the most abundant and common species (35 percent of catch) followed by Lake Chub (19 %), Northern Redbelly Dace (12 %), Finescale Dace (11 %), and Brook Stickleback (8 %). Following in decreasing order of abundance were Reside Shiner (4 %), Longnose Dace (3 %), White Sucker (3 %), Longnose Sucker (2 %), Trout-perch (1 %), Pearl Dace (0.8 %), Mountain Whitefish (0.7 %), Burbot (0.5 %), Arctic Grayling (0.4 %), Northern redbelly dace × finescale dace (0.1 %), and Bull Trout (0.03 %). Overall mean density of fish was 7.48 individuals$\cdot$100 m^{-2}, similar to previous estimates from this watershed (7.32 individuals$\cdot$100 m^{-2}; Scrimgeour, Hvenegaard & Tchir, 2008). Of the 11 culverts examined, eight were classed as complete barriers to fish movement (all species and life stages), two as partial barriers (passage inadequate for benthic species and fry of all species), and one as fully passable (Table S3).

2.3.1 Physicochemical habitat characteristics.—Consistent with our predictions, I observed strong differences in physicochemical habitat characteristics both among and within streams (Table A4; Fig. 2.2). Wetted width was similar across stream type and location. Mean water depth was on average 0.5 m lower in bridged versus culverted streams (LME: β = -0.5, standard error [SE] = 0.2, degrees of freedom [df] = 30, t = -2.8, *P* = 0.01), and 0.3 m higher upstream versus downstream of culverts (LME: β = -0.3, SE = 0.1, df = 30, t = 2.8, *P* = 0.009). Water velocity was on average 0.7 $m \cdot s^{-1}$ slower in culverted versus bridged streams (LME: β = 0.7, SE = 0.3, df = 30, t = 2.2, *P* = 0.038). Culverted streams had an average of 14% more fines than reference streams (LME: β = -13.5, SE = 5.8, df = 30, t = -2.3, *P* = 0.026). The percent of fines was also 21% higher upstream compared to downstream in culverts streams (LME: β = 20.6, SE = 4.4, df = 30, t = 4.7, *P* < 0.001), and 28% lower upstream compared to downstream in bridged streams (LME: β = -27.8, SE = 6.2, df = 30, t = -4.5, *P* < 0.001). Coarse rocky substrates percentages followed a similar trend. The percent of coarse rocky substrate was 4% higher at reference streams compared to culverted streams (LME: β = 4.3, SE = 2.0, df = 30, t = 2.2, *P* = 0.037), 7% lower upstream compared to downstream in culverted streams (LME: β = -7.0, SE = 1.5, df = 30, t = -4.6, *P* < 0.001), and 9% higher upstream versus downstream in bridged streams (LME: β = 9.2, SE = 2.1, df = 30, t = 4.3, *P* < 0.001). Pool, riffle, and run habitat percentages varied within stream types. There was 21% more pool habitat (LME: β = 20.9, SE = 3.4, df = 30, t = 6.2, *P* < 0.001), 11% less riffle habitat (LME: β = 10.5, SE = 2.6, df = 30, t = -4.0, *P* < 0.000), and 9% less run habitat (LME: β = 8.8, SE = 2.2, df = 30, t = -4.0, *P* = <0.001) upstream of culverted streams compared to downstream reaches. There was also 20% less pool habitat (LME: β = -20.2, SE = 4.8, df = 30, t = -3.4, *P* < 0.001), 8% more riffle habitat (LME: β = 7.6, SE = 3.7, df = 30, t = 2.1, *P* = 0.047), and 11 % more run habitat (LME: β = 10.8, SE = 3.2, df = 30, t = 3.3, *P* = 0.002) upstream versus downstream of bridged streams.

Chemical water characteristics also varied by the interaction of stream type and location (Table A5; Fig. 2.2). Mean water temperature was 2.8° C colder in bridged stream (LME: β = -

2.8, SE = 1.1, df = 30, t = -2.5, $P = 0.017$) and 6.3° C colder in reference streams (LME: β = -6.3, SE = 1.1, df = 30, t = -5.7, $P < 0.001$) as compared with culverted streams. Mean water temperature was on average 1° C warmer upstream versus downstream of culverted streams (LME: β = 1.0, SE = 0.35, df = 30, t = 3.0, $P = 0.006$). Dissolved oxygen was 0.8 $mg{\cdot}L^{-1}$ lower upstream versus downstream of culverted streams (LME: β = -0.8, SE = 0.13, df = 30, t = -6.3, $P < 0.001$). pH was 0.3 units higher in bridged (LME: β = 0.3, SE = 0.1, df = 30, t = 2.4, $P = 0.022$) and reference streams (LME: β = 0.3, SE = 0.1, df = 30, t = 2.8, $P = 0.009$) as compared with culverted streams, and 0.1 units lower upstream versus downstream on culverted streams (LME: β = -0.1, SE = 0.03, df = 30, t = -3.6, $P = 0.001$). Conductivity did not vary across stream type or location. Finally, turbidity was on average 2.7 NTUs lower in bridged (LME: β = -2.7, SE = 0.99, df = 30, t = -2.7, $P = 0.012$) and 3.0 NTUs lower in reference streams (LME: β = -3.0, SE = 0.2, df = 30, t = -3.0, $P = 0.005$) versus culverted streams, and 2.1 NTUs lower upstream versus downstream of culverted streams (LME: β = -2.1, SE = 0.48, df = 30, t = -4.3, $P < 0.001$).

Both wetted width and mean water depth influence fish response metrics (Table A6). Total density and Brook Stickleback densities were negatively associated with larger wetted widths (GLMM: β = -0.28, SE = 0.14, $z = -1.99$, $P = 0.046$; GLMM: β = -0.43, SE = 0.09, $z = -4.74$, $P < 0.001$; respectively), while richness and Slimy Sculpin, Lake Chub, and Longnose Dace densities were positively associated with increased wetted width (GLMM: β = 0.55, SE = 0.23, $z = 2.34$, $P = 0.019$; GLMM: β = 0.43, SE = 0.21, $z = -1.99$, $P = 0.046$; GLMM: β = 0.0.85, SE = 0.38, $z = -2.24$, $P = 0.025$; GLMM: β = 2.45, SE = 0.77, $z = -3.19$, $P = 0.001$; respectively). Species richness and Slimy Sculpin and *Chrosomus* sp. densities were negatively associated with increasing mean water depth (GLMM: β = -11.24, SE = 1.86, $z = -6.06$, $P = <0.000$; GLMM: β = -9.66, SE = 0.1.73, $z = -5.58$, $P < 0.001$; GLMM: β = -0.43, SE = 0.09, $z = -4.47$, $P < 0.000$).

2.3.2 Stream fish communities.—Species richness was affected by stream type and the interaction of stream type and location, and total fish density ($n \cdot m^{-2}$) was affected by the interaction of stream type and location after controlling for variation in wetted stream width and depth (Table 2.1; Fig. 2.3). There were on average 0.5 fewer species on bridged versus culverted streams, providing evidence of whole-stream scale effects (GLMM: $\beta = -0.50$, SE = 0.24, $z = 2.04$, $P = 0.041$). On culverted streams, total density was on average 4.6% lower (GLMM: $\beta = -1.73$, SE = 0.17, $z = -10\text{-}43$, $P < 0.001$) and there were on average 1.9 less species upstream versus downstream (GLMM: $\beta = -1.18$, SE = 0.27, $z = -4.43$, $P < 0.001$), providing evidence of within-stream scale impacts. In addition to impacts on total density and richness, the effects of stream type were species-specific (Table 2.1, Fig. 2.4). Slimy sculpin were positively associated with bridged and reference streams. There were on average 6 times more sculpin in reference streams (GLMM: $\beta = 1.30$, SE = 0.57, $z = 2.29$, $P = 0.022$) and 6 times more sculpin in bridged streams (GLMM: $\beta = 1.65$, SE = 0.57, $z = 2.88$, $P = 0.004$) as compared with culverted streams. *Chrosomus* spp. and *Catostomus* spp. were positively associated with culverted streams as compared with bridged streams. On average there was 1.3 times more *Chrosomus* spp. (GLMM: $\beta = -3.26$, SE = 1.34, $z = -2.44$, $P = 0.015$) and 10 time more *Chrosomus* spp. (GLMM: $\beta = -2.91$, SE = 1.21, $z = -2.42$, $P = 0.016$) in culverted streams.

Slimy Sculpin (GLMM: $\beta = -1.49$, SE = 0.38, $z = -3.92$, $P < 0.001$), Lake Chub (GLMM: $\beta = -2.20$, SE = 0.45, $z = -4.89$, $P < 0.001$), *Chrosomus* spp. (GLMM: $\beta = -1.55$, SE = 0.27, $z = -5.78$, $P < 0.001$), Brook Stickleback (GLMM: $\beta = -3.66$, SE = 1.43, $z = -2.55$, $P = 0.011$), Redside Shiner (GLMM: $\beta = -2.35$, SE = 0.63, $z = -3.76$, $P < 0.001$), and Longnose Dace (GLMM: $\beta = -1.81$, SE = 0.61, $z = -2.98$, $P = 0.003$) densities were all affected by the stream type × location interaction (Table 2.1, Fig. 2.4). This provides evidence of effects at the within-stream scale. For each of these species, densities varied as a function of location, but only on culverted streams; average densities were markedly lower upstream as compared with downstream.

PERMANOVA analysis showed that fish assemblages varied with stream type, but it depended on stream location as the interaction term was significant ($F_{2,58} = 1.11$, $P = 0.011$; Table 2.2). Ordination differentiated culverted stream reaches from bridged and reference streams, and to a lesser extent bridged streams from references (Fig. 2.5a). Whereas strong dissimilarities in species composition between upstream and downstream reaches on culverted streams are apparent, they were not for bridged or reference streams (Fig. 2.5a). The stress of the NMDS ordination of species densities was evaluated at two dimensions. Results suggested a two-dimensional solution suitably represented fish assemblages and revealed broad patterns of dissimilarity in species composition between stream reaches (2-D stress = 0.24; Fig. 2.5). Correlation of NMDS ordination scores with physicochemical habitat parameters and individual fish species indicate the presence of an environmental gradient across study stream reaches with associated differences in where species plot out (Fig. 2.5b). A primary gradient in physicochemical habitat characteristics is discernable from the left to right. The percentage of fines, pool habitat, temperature, turbidity, and water depth are associated with each other whereas increasing percentages of cobble, gravel and boulder substrate, riffle and run habitat, DO, and water velocity align opposite. Brook Stickleback, *Chrosomus* spp., *Catostomus* spp., and Lake Chub are associated with each other and align opposite salmonids, Burbot, and Slimy Sculpin.

2.4 Discussion

*2.4.1 Crossing Effects on Physicochemical Habitat Characteristics.—*Culverted streams in our study were associated with significantly higher percent fines, water temperature, water depth and turbidity and lower dissolved oxygen and water velocity. These observations are generally consistent with previous studies (Wellman et al. 2000, Park et al. 2008, MacPherson et al. 2012a). In particular, our findings support Wellman et al. (2000) and MacPherson et al. (2012a) in that sediment accumulation and water depth were greater in streams with culverts than in bridged or reference streams. This is consistent with Favaro et al. (2014) who found larger sediment sizes to be associated with reference streams as compared with culverted streams. The study also showed significant differences in upstream as compared with downstream habitats in culverted streams. Given our study sampling design (300 m stream reaches buffered from stream crossings by 25 m) and the low gradient nature of streams in the study region, this suggests culverts may be acting as constrictions causing upstream backwater effects. Stream channel constriction is often observed where culverts are present (Belford and Gould 1989, MacDonald and Davies 2007, MacPherson et al. 2012a) and can cause hydrological modifications which in turn alter geomorphological properties of streams (Gordon et al. 2013). In our study streams culverts may be constricting the downstream movement of water and abiotic materials, thus causing higher water depths and subsequent increases in pool habitat with concomitant decreases in riffle and run habitat. Higher upstream temperatures on culverted streams may then be the result of modified morphology (i.e. increased pool percentage and water depth) as similarly observed by MacPherson et al. (2012a).

While all lotic systems contain natural levels of sediment, road-stream crossing sites are often significant point-source locations for erosion and sedimentation (Ottburg and Blank 2015). Culverted and bridged streams had elevated turbidity levels, and bridged streams had more fines in downstream versus upstream reaches. This suggests significant sediment input from erosional processes occurring at the crossing road surface and stream bank. Indeed, I observed and documented evidence of erosion at all but one culverted stream, and all but two bridged streams. These observations are consistent with numerous studies examining the impact of resource roads and stream crossings on sediment loading (Spillios 1999, Lachance et al. 2008, Thomaz et al. 2013, Wang et al. 2013). While stream banks adjacent to crossings will, overtime, become re-vegetated and stabilize, heavy rain and flooding can destabilize banks and facilitate erosional processes (Chapman et al. 2014). As increases in suspended and deposited sediments can have serious negative impacts on stream fishes (e.g. feeding behavior, spawning success, species richness; Chapman et al., 2014) determining the tolerances of different species of fish to sediment loads associated with logging and oil and gas development is a high priority research need (Boyce and Poesch 2014), and should be examined broadly across the boreal region. Further, our results provide evidence that bridges may not be acting as ecologically benign structures as previously postulated (Warren and Pardew 1998, Pluym et al. 2008). By considering a reference stream condition, this study is able to show that bridges, along with culverts, are acting as significant point-source locations for sediment delivery into boreal streams. While data concerning stream conditions prior to stream crossing installation is unavailable, these findings together with numerous other studies (e.g. Spillios, 1999; Lachance *et al.*, 2008; MacPherson *et al.*, 2012) support the contention that stream crossings – and culverts in particular – can alter fish habitat at the whole- and within-stream scale through changes to habitat structure, hydrology and water quality, factors which can influence where fish occur (Gorman and Karr 1978).

*2.4.2 Crossing Effects on Stream Fish Communities.—*After accounting for physical differences in habitat that affect fish responses, I found evidence of fragmentation effects as species richness and Slimy Sculpin, *Chrosomus*, and *Catostomus* densities varied in response to stream type. These results are consistent with previous research that has demonstrated reductions in local abundances and species richness in relation to stream crossing structures (Nislow et al. 2011, Perkin and Gido 2012). Perkin and Gido (2012), for instance, found reduced species richness in fish communities isolated by stream crossings compared with those which maintained connectivity with the surrounding stream network. Our results are also consistent with research that has found reductions in species-specific densities as an effect of stream crossings (MacPherson et al. 2012a, Favaro et al. 2014). Slimy Sculpin densities were on average 6 times higher in our reference streams than in culverted streams. Similarly, Favaro et al. (2014) found densities of Coastrange Sculpin (*Cottus aleuticus*) and Prickly Sculpin (*Cottus asper*) to be on average 90 times higher in reference streams than in culverted streams, and MacPherson et al. (2012a) found that whereas 69% of their reference sites had Spoonhead Sculpin (*Cottus ricei*), only 12% of their culverted sites similarly did. This information demonstrates that both species richness and the densities of species differ among stream types. Furthermore, our results support the contention that sculpin are an effective indicator taxon for alterations to stream connectivity (Favaro et al. 2014) given their benthic habit and weak swimming abilities (Nelson and Paetz 1992, LeMoine et al. 2014). Conversely, Northern Redside Dace and Finescale Dace densities were positively associated with culverted streams in the study area. Species in the genus *Chrosomus* prefer slow, warm water streams and are commonly found over fine substrates (Nelson and Paetz 1992). As culverted streams in our study were characterized by deeper, warmer water with high levels of fine sediment, fragmentation effects on *Chrosomus* species may be buffered by their ability to tolerate habitat of reduced quality.

Local abundance and species richness should typically be depressed in the presence of a barrier to immigration (i.e. movement), and thus local reductions in these metrics may be appropriate proxies for fragmentation (Nislow et al. 2011). On average, total fish density and species richness, and densities of Slimy Sculpin, Lake Chub, *Chrosomus* spp., Brook Stickleback, Redside Shiner, Catostomids, and Longnose Dace were significantly reduced in upstream relative to downstream habitats in streams crossed by culverts. These findings support observations showing reduced upstream fish abundances and species richness as compared with downstream habitats on streams crossed by culverts (Warren and Pardew 1998, Wheeler et al. 2005, Nislow et al. 2011), particularly for small-bodied, non-game stream fish (MacPherson et al. 2012a, Perkin and Gido 2012, Favaro et al. 2014). Causes for the impediment of fish movement are related to physical factors including steep culvert slope and excess water velocity (Belford and Gould 1989, Burford et al. 2009, MacPherson et al. 2012a) or perching of the culvert above the water surface (Mueller et al. 2008, Burford et al. 2009, Norman et al. 2009, MacPherson et al. 2012a). These factors are exacerbated for weak-swimming stream fish commonly found throughout lowland boreal and foothills streams (e.g. Cottids, Cyprinids, and Catostomids). Of the 11 culverts examined in this study, eight were categorized as complete barriers, two as partial barriers, and one as completely passable (AESRD 2014; Table S3). Culverts in our study constricted stream channel width and increased instantaneous stream velocities at the crossing outlet (mean = $0.39\ \mathrm{m \cdot s^{-1}} \pm 0.4$ SD). Excessive water velocities, in addition to physically inhibiting fish movement (Warren and Pardew 1998, MacDonald and Davies 2007), can also lead to the development of large outwash scour pools and hanging culverts over time (Park et al. 2008). Outlet drop heights averaged 0.22 m (± 0.26 SD) for culverts in our study. MacPherson et al. (2012a) found that hang heights as small as 0.16 m (mean, ± 0.24 SD) may be complete barriers to upstream movement of Burbot, and partial barriers to Spoonhead Sculpin, Catostomids, and Cyprinids. In our culverted streams, Burbot were never observed, Catostomids were observed in eight streams and only once found in upstream habitats, and Slimy Sculpin found in seven streams and only once in upstream habitats. Accordingly, our results suggest that culverts are impeding upstream movement of non-game fish in boreal watersheds, possibly to the point of upstream extirpation. Multi-year studies that follow fish communities in habitats above culverts are warranted to further address the hypothesis that upstream extirpations are the result of stochastic environmental events (e.g. drought, flood, ice scour) combined with reduced or completely eliminated upstream immigration from downstream source populations (Eisenhour and Floyd 2013).

While I provide an expanded view of stream crossing effects on freshwater fish communities in boreal streams, there are a few caveats. This study was of a large-scale, comparative design in which stream type was not randomly assigned to streams due to the non-random placement of culverts and bridges on the landscape. Thus, the effect of stream crossing configuration within the stream network could not be evaluated (Chelgren and Dunham 2014). This is a common problem in stream crossing studies which investigate ecological patterns rather than their underlying mechanisms (Levin 1992, Favaro et al. 2014). Herein I attempted to control for landscape and habitat level differences in site selection and analyses, but unmeasured variables may have confounding effects on our observations of stream type effect. The fact that this study was carried out during summer low flow may have also influenced upstream fish densities as culvert passability for particular fish species has been shown to change relative to stream discharge (Bouska and Paukert 2010, Mahlum et al. 2014). Future studies should accordingly incorporate temporal scales into analyses to account for changes in culvert passability throughout the year. Finally, incorporating capture probability of fish into analyses of stream crossing effects may help further elucidate impact by accounting for species-specific differences in capture efficiencies (MacPherson et al. 2012b, Chelgren and Dunham 2014).

2.4.3 Cumulative Effects on Boreal Watersheds.—There is presently a limited amount of published literature on the effects of natural resource development on aquatic ecosystems in Canada's boreal forest (Kreutzweiser et al. 2013), though trends suggest principal impacts result from increased fine sediment loads (Anderson 1996, Ripley et al. 2005, Browne 2007, Scrimgeour et al. 2008) or malfunctioning culverts (Browne 2007, Park et al. 2008, MacPherson et al. 2012a, Torterotot et al. 2014). Collectively, our results support the hypothesis that the effects of culverts at the whole-stream scale are the result of fragmentation effects from stream-wide extirpations and within-stream effects likely the result of inhibited movement within streams. It is possible however, that multiple mechanisms (i.e. habitat degradation *and* fragmentation) are working synergistically and exacerbating the effects of stream crossings on boreal streams. Modifications to streams from the presence of crossing structures may be reducing habitat quality, thus leading to lower fish densities and altered communities. For example, shifts in habitat (e.g. lotic to lentic nature, temperature regime, increased fines) often favor generalist species over more specialized, sensitive ones, and can lead to species extirpations and biotic homogenization (Rahel 2000, Roberts 2001, Poff et al. 2007).

The cumulative effects of natural resource development and additional stressors, such as climate change and forest pests, also remain largely unknown (Kreutzweiser et al. 2013). For example, salvage logging in response to mountain pine beetle infestation and spread is expected to increase watershed disturbance in forested watersheds of western Canada (Redding et al. 2008), thereby exacerbating sediment loading (Chamberlin et al. 1991). It was subsequently found in Alberta that infestation rates were too low to cause significant impacts to aquatic systems, however this conclusion could not be substantiated by habitat or fish abundance data because it does not exist for vast portions of the province (Weiss 2011). Given the vulnerable nature of northern fishes to resource development (Reist et al. 2006), this poses serious concerns for long-term persistence and biodiversity maintenance of freshwater fish populations in boreal forest watersheds. With expanded development expected in much of North America's boreal forest, mitigation measures which limit impacts from stream crossings are needed to ensure proper ecosystem function in freshwater systems. This is particularly important given the general lack of consistent monitoring across remote northern environments (Weiss 2011, Brandt et al. 2013). To this end, this study provides a baseline to which remediation actions within our study watershed can be compared to validate the efficacy of culvert remediation to restore connectivity for fish populations.

In conclusion, this study shows that culverted streams in Canada's western boreal forest have higher levels of fine sediments, increased stream temperatures and water depth, and less coarse rocky substrate as compared with bridged and reference streams. Culverted streams were associated with lower sculpin densities and higher Chrosomus spp. densities, evidence of whole-stream scale fragmentation effects. These effects were also pronounced within-streams, where the majority of fish species exhibited significantly lower densities in upstream as compared to downstream habitats. Broadly, these results have negative implications for populations of stream-resident and potadromous species in the region, including Salmonids, Cyprinids, and Catostomids. Given our findings in the context of recent research, the widespread and growing distribution of culverts on the landscape (Prévost et al. 2002, Park et al. 2008, Miller 2012, Januchowski-Hartley et al. 2013), and increasing rate of natural resource exploitation in the boreal region (Kreutzweiser et al. 2013), I conjecture that alterations to fluvial stream systems associated with stream crossings may be driving changes in stream fish communities, potentially at a very large scale. This is likely facilitated by the cumulative effects of habitat connectivity loss, alterations to instream habitat, and other stressors. Regional studies such as this can fill gaps in our understanding of how anthropogenic features interact with freshwater environment and guide adaptive ecosystem management by identifying fragmentation hotpots where remediation and conservation dollars should be focused to ensure the greatest ecological return on remediation dollar invested.

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TABLE 2.1.—Parameter estimates from generalized linear mixed models (GLMMs) investigating fish response metrics across stream type (culvert vs. bridge vs. reference), stream location (upstream vs. downstream), and the interaction of stream type × location. Boldface indicates significance at $\alpha = 0.05$.

Response	Variable	Coefficient	SE	*z* value	P
Total density ($n \cdot \mathrm{m}^{-2}$)	Intercept	-2.31	0.33	-6.96	0.000
	Bridge	-0.05	0.24	-0.21	0.830
	Reference	0.38	0.24	1.60	0.110
	Upstream	-1.73	0.17	-10.43	**< 2e-16**
	Wet Width	-0.13	0.11	-1.19	0.240
	Depth	-0.62	1.07	-0.58	0.560
	Bridge × Upstream	1.55	0.19	8.05	**0.000**
	Reference × Upstream	1.67	0.19	8.66	**< 2e-16**
Richness	Intercept	0.52	0.30	1.75	0.079
	Bridge	-0.50	0.24	-2.04	**0.041**
	Reference	0.11	0.21	0.52	0.603
	Upstream	-1.18	0.27	-4.43	**0.000**
	Wet Width	0.27	0.08	3.20	0.001
	Depth	2.25	0.93	2.41	0.016
	Bridge × Upstream	1.22	0.36	3.37	**0.001**
	Reference × Upstream	1.18	0.33	3.60	**0.000**
Slimy Sculpin ($n \cdot \mathrm{m}^{-2}$)	Intercept	-5.43	0.70	-7.76	0.000
	Bridge	1.65	0.57	2.88	**0.004**
	Reference	1.30	0.57	2.29	**0.022**
	Upstream	-1.49	0.38	-3.92	0.000
	Wet Width	0.29	0.18	1.60	0.109
	Depth	-4.20	1.90	-2.20	0.027
	Bridge × Upstream	1.34	0.37	3.60	**0.000**
	Reference × Upstream	1.50	0.39	3.87	**0.000**
Lake Chub ($n \cdot \mathrm{m}^{-2}$)	Intercept	-6.97	1.18	-5.92	0.000
	Bridge	-1.46	1.15	-1.26	0.210
	Reference	1.49	1.05	1.43	0.150
	Upstream	-2.20	0.45	-4.89	**0.000**
	Wet Width	0.31	0.31	0.99	0.320
	Depth	2.71	3.03	0.89	0.370
	Bridge × Upstream	2.14	0.45	4.79	**0.000**
	Reference × Upstream	2.04	0.44	4.62	**0.000**
Chrosomus Spp. ($n \cdot \mathrm{m}^{-2}$)	Intercept	-3.68	1.18	-3.11	0.002
	Bridge	-3.26	1.34	-2.44	**0.015**
	Reference	-0.62	1.19	-0.52	0.602

Response	Variable	Coefficient	SE	*z* value	P
	Upstream	-1.55	0.27	-5.78	**0.000**
	Wet Width	-0.04	0.32	-0.14	0.890
	Depth	-3.85	1.67	-2.30	0.021
	Bridge × Upstream	1.57	0.37	4.27	**0.000**
	Reference × Upstream	1.53	0.30	5.05	**0.000**
Brook Stickleback (*n*·m^{-}	Intercept	-7.29	4.51	-1.62	0.106
	Bridge	-4.62	2.87	-1.61	0.107
	Reference	-2.39	2.49	-0.96	0.337
	Upstream	-3.66	1.43	-2.55	**0.011**
	Wet Width	-1.31	1.13	-1.16	0.247
	Depth	10.72	10.71	1.00	0.317
	Bridge × Upstream	5.56	2.29	2.43	**0.015**
	Reference × Upstream	2.88	1.47	1.96	0.050
Redside Shiner (*n*·m^{-2})	Intercept	-12.49	1.97	-6.35	0.000
	Bridge	-2.96	1.86	-1.59	0.113
	Reference	-0.24	1.44	-0.16	0.870
	Upstream	-2.35	0.63	-3.76	**0.000**
	Wet Width	1.40	0.66	2.12	0.034
	Depth	5.61	4.84	1.16	0.246
	Bridge × Upstream	1.79	0.64	2.80	**0.005**
	Reference × Upstream	2.58	0.73	3.52	**0.000**
Catostomus Spp. (*n*·m^{-2})	Intercept	-7.31	1.14	-6.44	0.000
	Bridge	-2.91	1.21	-2.42	**0.016**
	Reference	0.48	0.83	0.58	0.559
	Upstream	-2.10	0.58	-3.62	**0.000**
	Wet Width	0.13	0.30	0.43	0.664
	Depth	4.19	3.91	1.07	0.284
	Bridge × Upstream	1.11	0.86	1.29	0.196
	Reference × Upstream	2.30	0.60	3.81	**0.000**
Longnose Dace (*n*·m^{-2})	Intercept	-15.92	3.98	-4.00	0.000
	Bridge	-5.03	3.17	-1.59	0.113
	Reference	-1.83	2.09	-0.88	0.381
	Upstream	-1.81	0.61	-2.98	**0.003**
	Wet Width	2.75	1.26	2.19	0.028
	Depth	1.14	6.17	0.18	0.853
	Bridge × Upstream	1.02	0.64	1.60	**0.109**
	Reference × Upstream	2.32	0.90	2.59	**0.010**
Salmonids (*n*·m^{-2})	Intercept	-12.69	2.35	-5.41	0.000
	Bridge	0.42	1.57	0.27	0.788
	Reference	1.83	1.43	1.28	0.200
	Upstream	-0.11	1.26	-0.09	0.931
	Wet Width	1.08	0.65	1.66	0.097
	Depth	-1.36	6.32	-0.21	0.830

Response	Variable	Coefficient	SE	*z* value	P
	Bridge × Upstream	1.13	1.43	0.79	0.428
	Reference × Upstream	0.31	1.31	0.23	0.816

NOTE: SE = standard error. The intercept of each model represents fish counts for downstream, culverted streams; variable coefficients then represent their relationship to the intercept. Individual stream-level effects were accounted for by including it as a random intercept term in each model. Differences in area sampled between stream reaches was accounted for by including an offset (log m^2, excluding richness) in each model. Because wetted stream width and water depth were associated with stream type (Table A4) and influenced fish response metrics (Table A6), they were included as fixed effects to account for their variation.

TABLE 2.2.—Results from two-way permutation multivariate analysis of variance (PERMANOVA) testing the effects of stream type, location, and the interaction of stream type and location on Bray-Curtis dissimilarity of species assemblages. Boldface indicates significance at $\alpha = 0.05$.

Variable	df	Sums. Sqs	*F* value	R^2	*P* (perm)
Stream Type	2	2.87	5.80	0.16	**0.008**
Location	1	0.27	1.11	0.02	**0.006**
Stream Type × Location	2	0.55	1.11	0.03	**0.011**
Residuals	58	14.34		0.80	

NOTE: PERMANOVA model included a strata variable for stream to account for random stream to stream variation.

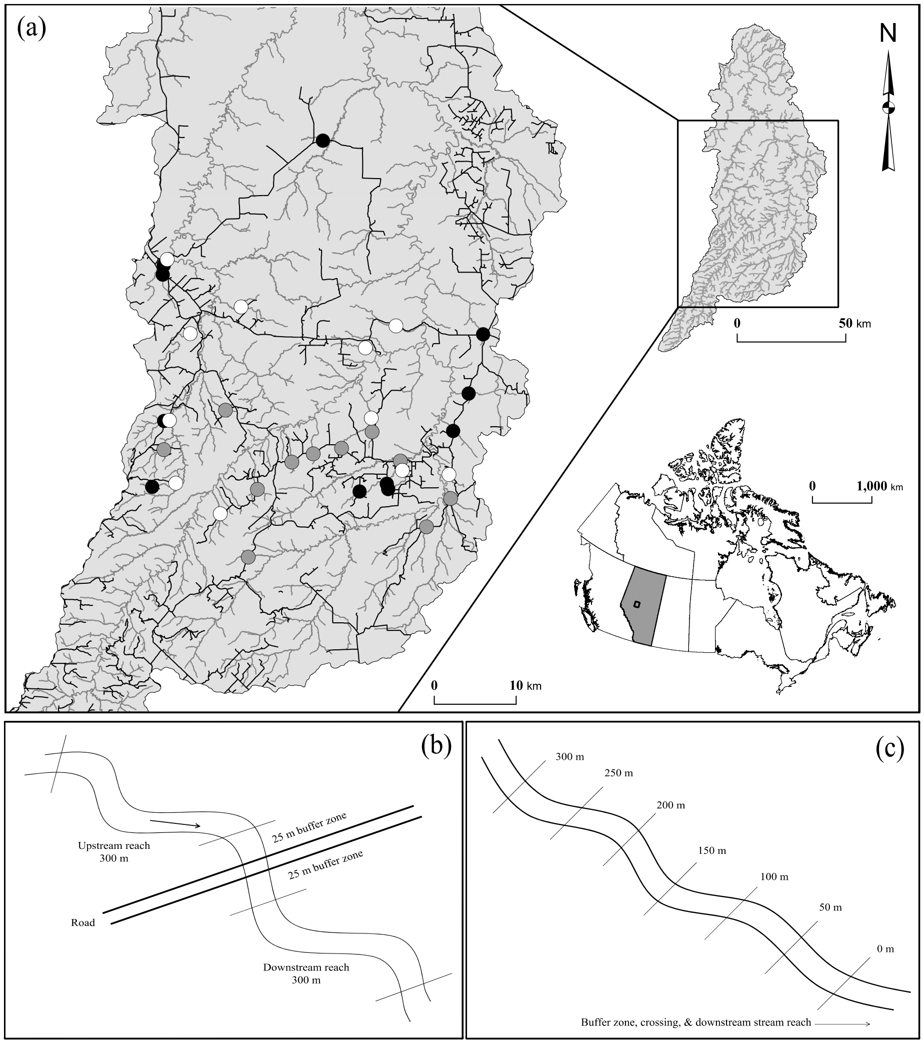

FIGURE 2.1.—Panel diagram depicting study watershed and sampling sites for summer 2013, and sample design for this study in the Simonette Watershed, Alberta, Canada. (a) Study map showing study stream locations. Circles represent stream types; black = culvert, grey = bridge, white = reference. Black lines represent resource roads. (b) Schematic of 300 m stream reaches above and below culverts, bridges, and hypothetical crossing structures (references). (Note 25 m buffer section between 300 m sample reaches and road crossing). (c) Within each 300 m stream reach, physicochemical habitat characteristics were quantified along seven transects spaced 50 m apart.

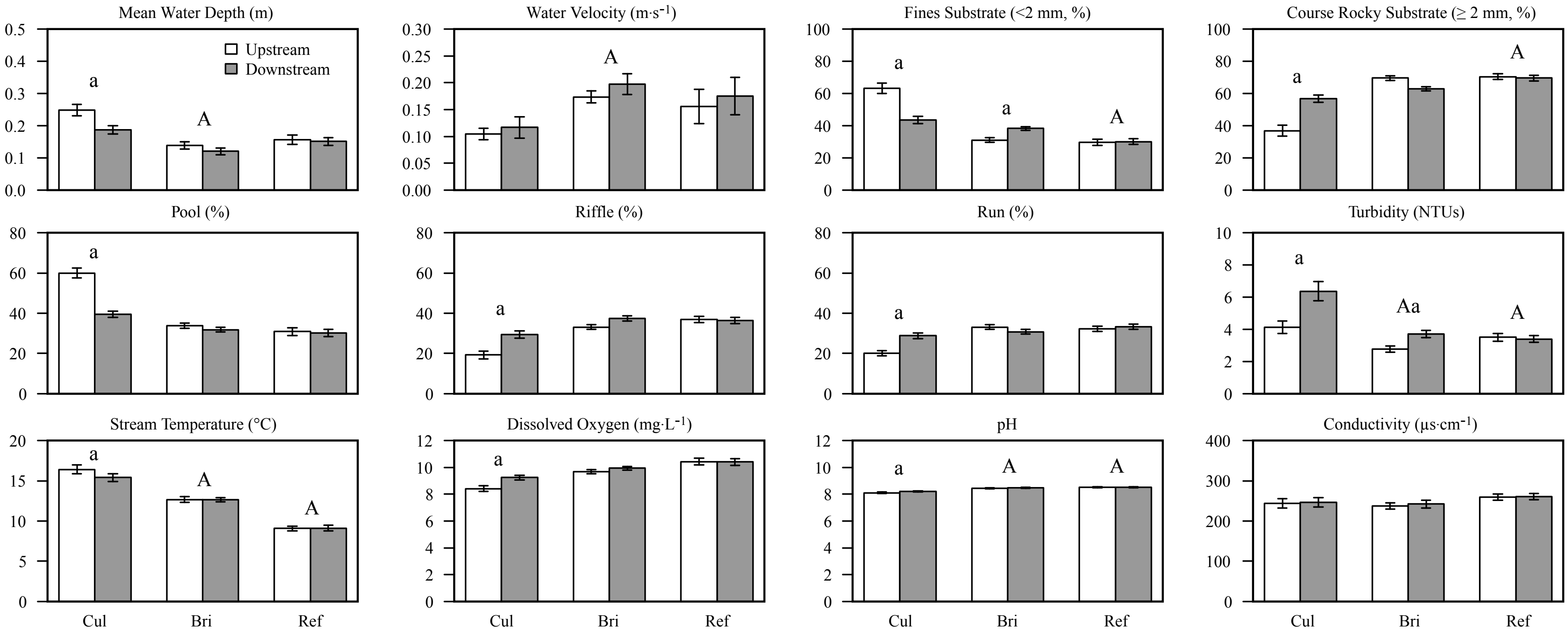

FIGURE 2.2.—Physicochemical habitat characteristics (mean ± SE) in each stream type and location. Sample sizes for stream types are: culvert, N = 11; bridge, N = 11; reference, N = 11. Abbreviations include: Cul = culverted streams, Bri = bridged streams, Ref = reference streams. Significant differences across stream types are identified by "A" above bars, while significance between upstream and downstream reaches within-streams are identified with lower case "a".

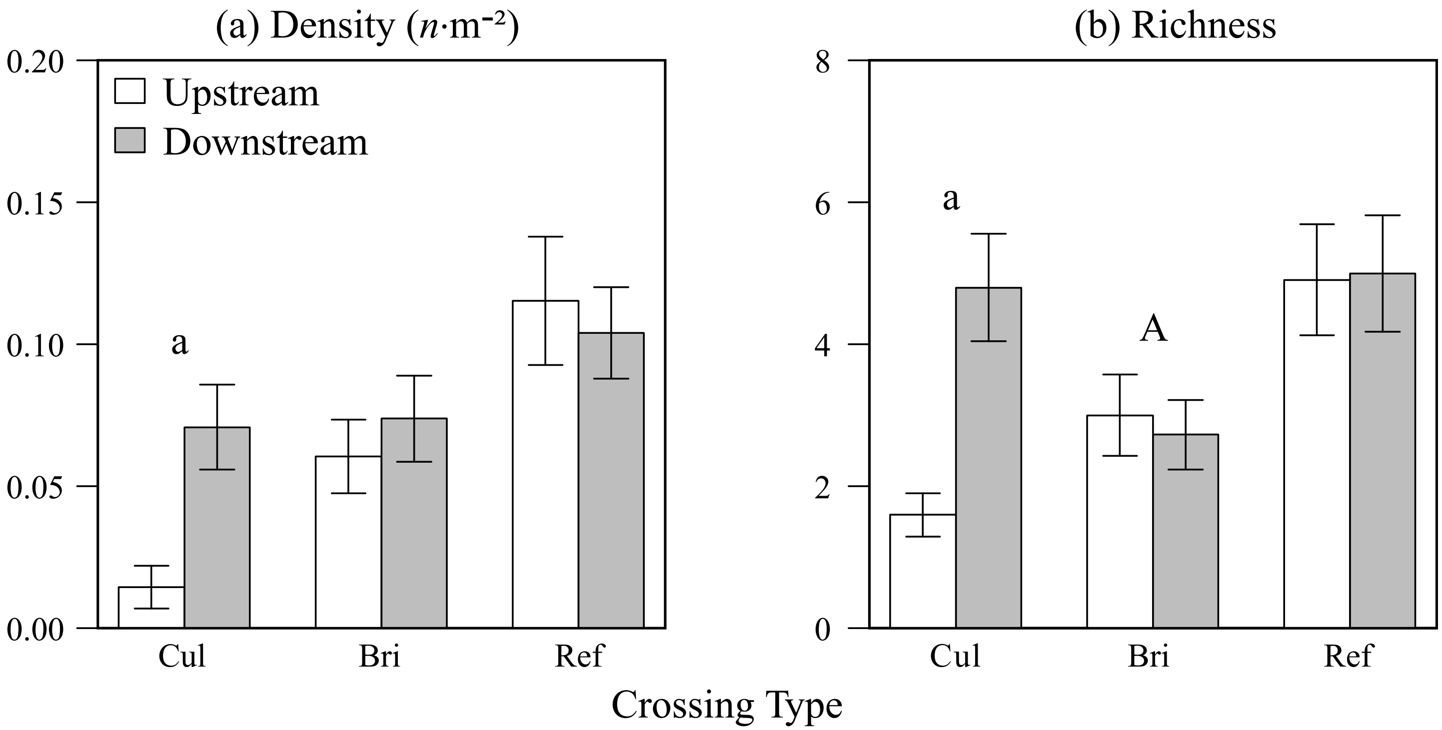

FIGURE 2.3.—Barplot of fish community metrics of (a) fish density (n • m^{-2}) and (b) species richness across stream types and upstream and downstream locations (mean ± SE). Sample sizes for stream types are: culvert, N = 11; bridge, N = 11; reference, N = 11. Abbreviations include: Cul = culverted streams, Bri = bridged streams, Ref = reference streams. Significant differences across stream types are identified by lower case by "A" above bars, while significance between upstream and downstream reaches within-streams are identified with lower case "a".

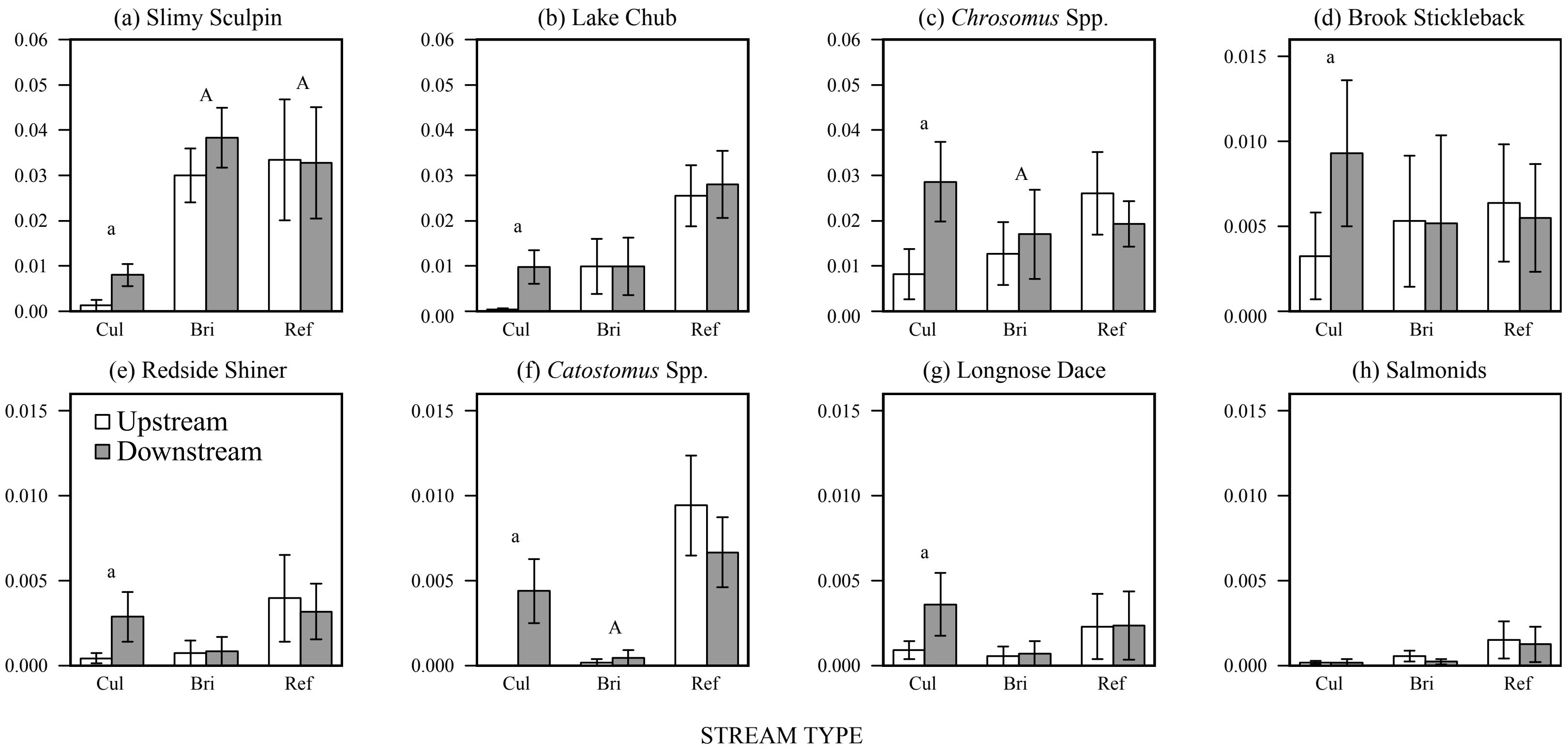

FIGURE 2.4.— Density (mean ± SE) of (a) Slimy Sculpin, (b) Lake Chub, (c) *Chrosomus* spp. (i.e. Northern Redbelly Dace, Finescale Dace, and their hybrid), (d) Brook Stickleback, (e) Redside Shiner, (f) *Catostomus* Spp., (g) Longnose Dace, (h) salmonids (i.e. Arctic Grayling, Bull Trout, Mountain Whitefish) by stream type and location. Sample sizes for stream types are: culvert, N = 11; bridge, N = 11; reference, N = 11. Abbreviations include: Cul = culverted streams, Bri = bridged streams, Ref = reference streams. Significant differences across stream types are identified by lower case by "A" above bars, while significance between upstream and downstream reaches within-streams are identified with lower case "a".

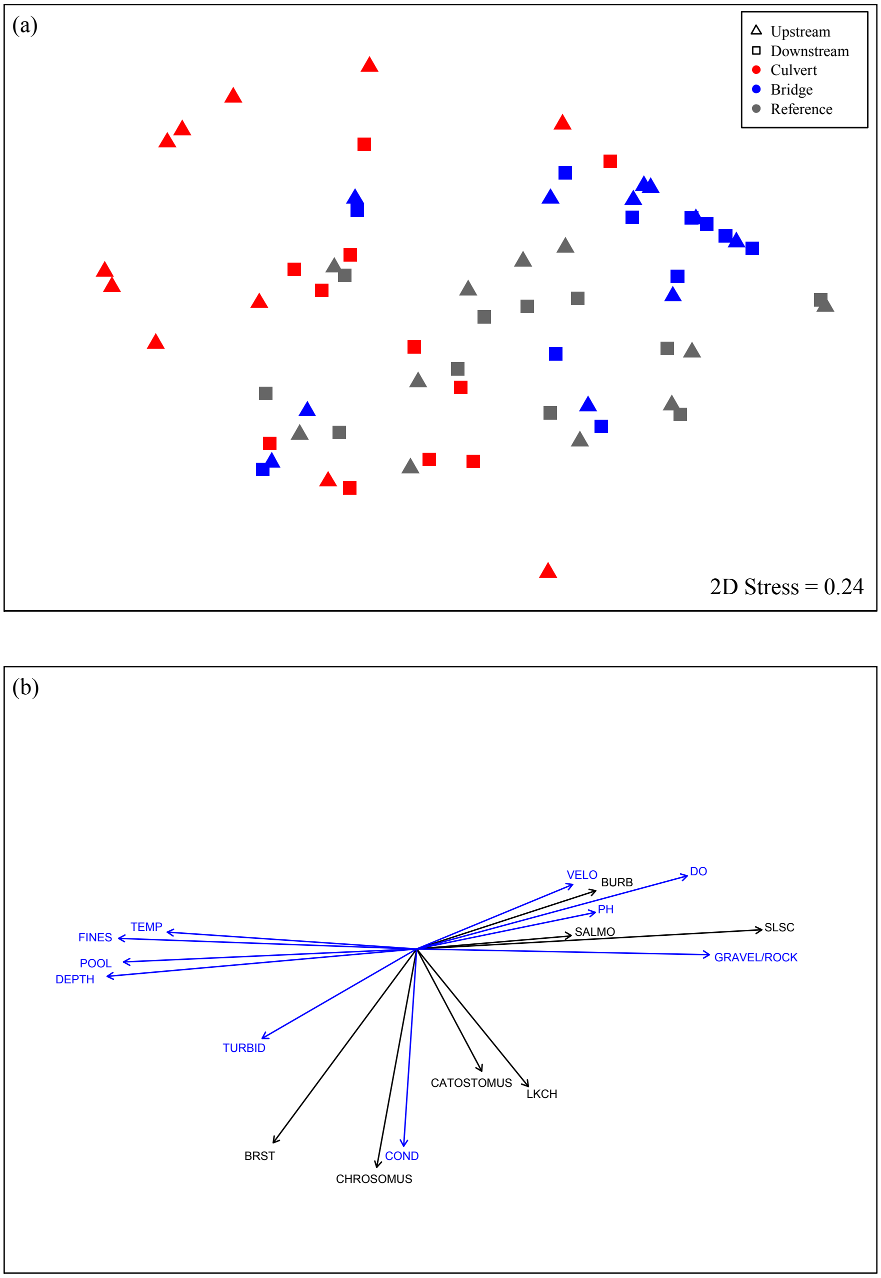

FIGURE 2.5.— Non-metric multidimensional scaling (NMDS) ordination based on species density in 66 stream reaches and relationships with physicochemical habitat variables, total fish density, and individual species. The stress level signifies the accuracy of the ordination for representing original dissimilarities in two dimensions. (a) Study streams are displayed with different colours. Upstream reaches are displayed with a triangle symbol and downstream

reaches with square symbols. (b) Physicochemical habitat variables and fish species ($P \leq 0.05$ based on 10,000 permutations) are displayed as vectors indicating the strength and direction of maximal correlations to the NMDS configuration. Vectors are distinguished by their colours; physicochemical habitat characteristics (blue), and fish (black). Note: Species abbreviations are as follows: SALMO = Arctic Grayling, Bull Trout, and Mountain Whitefish; CHROSOMUS = Northern Redbelly Dace, Finescale Dace, Northern Redbelly Dace X Finescale Dace; BRST = Brook Stickleback, LKCH = Lake Chub; CATOSTOMUS = White Sucker, Longnose Sucker; SLSC = Slimy Sculpin; BURB = Burbot.

Chapter III: Prioritization of stream crossings for restoration and management of freshwater biodiversity in a boreal watershed

Abstract.—The western boreal forest is a region with a wealth of natural resources, including vast expanses of harvestable timber and natural gas, as well as the majority of Canada's oil reserves. In west-central Alberta, extraction of these resources has increased dramatically in recent years and caused landscape-level changes, predominantly from forest harvest and the development of road networks. Culverted road-stream crossings in particular can decrease habitat quality and disrupt stream network connectivity. As the repair or removal of problem culverts is an expensive endeavor, systematic methods for prioritizing their mitigation are essential for the efficient allocation of restoration dollars. Here I assess the application of operational research tools that utilize an optimization framework to prioritize fish barriers by maximizing connectivity-weighted habitat availability for potadromous fish given budgetary constraints in two Albertan watersheds. Results indicate that a large proportion of potential connectivity gain can be realized with a moderately low investment (~$200K to $500K). This case study highlights the utility of this method for use in Alberta watersheds, particularly as it has minimal data requirements, is applicable to both stream-resident and long-range migratory species, and significantly reduces the mathematical and technical expertise needed to perform these relatively complex optimization analyses. Cost-benefit curves from these watersheds can be used to minimize overall restoration costs to achieve particular management objectives in watersheds of interest, as well as provide defendable evidence for budget planning to regulators and decision-makers.

3.1 Introduction

Globally, rising demands for natural resources including timber, oil and natural gas have driven the expansion of road networks across vast areas of land (Laurance and Balmford 2013), particularly in sensitive northern regions such as the North American boreal forest (Schindler 2001, Kreutzweiser et al. 2013). Expanding road development generally necessitates the installation of crossing structures (e.g. culverts, bridges) to convey good and services across streams or rivers where they occur. Mounting evidence over the past two decades has shown that these structures often disrupt longitudinal connectivity for a range of aquatic species by acting as barriers to movement and thus isolating populations (Dynesius and Nilsson 1994, Fullerton et al. 2010, Perkin and Gido 2012) and restricting their access to critical habitats necessary for reproduction, feedings, and overwintering (Gibson et al. 2005, Nislow et al. 2011, MacPherson et al. 2012).

Instream crossing structures pose a substantial threat to the sustainability of stream fish populations by means of habitat fragmentation and the disruption of longitudinal connectivity (ESRD 2012). Regulations and legislation have been developed globally to manage and enforce standards relating to the environmental risks of roads and stream crossing structures. However, decades of poor culvert installation practices and overlapping regulatory mandates have has led to negative impacts on fish resources (Warren and Pardew 1998, Park et al. 2008, MacPherson et al. 2012). As road networks continue to grow with the expansion of activities from extractive industries across landscapes (Park et al. 2008, White et al. 2011, Kreutzweiser et al. 2013, Laurance et al. 2015), examining how management actions can mitigate impacts of connectivity loss on freshwater fish is needed to guide best management practices. Further, given limited amounts of time and money available for restoration efforts, methods to prioritize management actions and measure their success are needed (Kemp and O'Hanley 2010).

With growing appreciation of the importance of connectivity in stream networks (Fagan 2002, Fullerton et al. 2010), prioritization techniques have emerged as a major theme for restoring habitat connectivity for native fishes (Table B1; Roni et al. 2002, Nilsson et al. 2007, Kemp and O'Hanley 2010). Broadly, these approaches attempt to maximize habitat or connectivity for species given limited resources that may be available to managers (Kemp and O'Hanley 2010, Oldford 2013, Diebel et al. 2014). Traditionally, scoring and ranking methods have been employed to prioritize river barrier restoration and simply require the assignment of a cost/benefit score to each barrier and ranking (prioritizing) the subsequent list accordingly (Karle 2005, Kocovsky et al. 2008, Nunn and Cowx 2012). However, this method tends to produce sub-optimal restoration solutions as they only consider the total amount of habitat restored (river km) instead of the connectivity between every pair of stream segments (and thus overall watershed connectivity); they also consider barriers independently thus ignoring the cumulative effects of barriers on longitudinal connectivity (Kemp and O'Hanley 2010). Accordingly, attention has shifted to more robust, systematic methods that are able to determine optimal or near-optimal restoration solutions.

Two prioritization techniques have arisen with recent insights into watershed typology: iterative prioritization (stepwise scoring and ranking) and budget optimization (supplementary information, Table B1). These methods consider stream networks as Dendritic Ecological Networks (DENs) where ecological processes (e.g. dispersal, population growth, and community interactions) are carried out within the branches of the network with nodes serving as transfer points (Campbell Grant et al. 2007). Using a graph theoretical approach, DENs are constructed with habitat patches or river segments as nodes and river confluences as arcs (Padgham and Webb 2010, Erős et al. 2011, Segurado et al. 2013). Alternatively, DENs can be modeled with barriers as nodes and habitat patches as connections between nodes (Cote et al. 2009, Diebel et al. 2010, McKay et al. 2013, Diebel et al. 2014). This paradigm has led to the development of indices which can help to assess the degree of structural connectivity within a river/stream network for both diadromous (Saura and Pascual-Hortal 2007, Pini Prato et al. 2011, McKay et al. 2013) and potadromous/stream-resident fish species (Cote et al. 2009, Diebel et al. 2010, Diebel et al. 2014; Table B2). These indices can subsequently be incorporated into operational research techniques that can be used to find optimal restoration solutions for stream crossings.

The purpose of this study was to develop a procedure for prioritizing culvert removals that will maximize habitat availability for fish species at risk of extirpation in a boreal forest watershed. The Simonette watershed is used as a model system as it is a watershed of high conservation concern; both Arctic Grayling (*Thymallus arcticus*) and Bull Trout (*Salvelinus confluentus*) populations are at depleted levels—two species listed as “at risk” of extirpation within Alberta (Walker 2005, ASRD 2009). Specifically, the objective of this study is to use optimization models to select barriers for mitigation that maximize the amount of habitat for species at risk given trade-offs in budget allocation for barrier removal and species life history information. By examining these trade-offs, optimal gains in habitat can be obtained with minimal investment; an ideal structure for prioritizing species conservation and restoration (Murdoch et al. 2007).

3.2 Methods

3.2.1 Determining Habitat Availability for Fish Species at Risk—To measure habitat availability, the C metric was used (Diebel et al. 2010, Diebel et al. 2014). The C metric is a normalized measure of stream habitat connectivity. Importantly, the metric accounts for habitat quality, quantity, and accessibility of different habitat types that a fish can access when quantifying connectivity. The incorporation of these attributes make it superior to other stream-resident directed methods (e.g. DCIp; Cote et al. 2009) which quantifies connectivity based only on barrier location within a stream network. In particular, the C metric has the capacity to account for habitat variation and species-specific dispersal limitations for species of interest, and was recently incorporated into a budget-constrained optimization model formulated to maximize habitat connectivity for stream-resident fish in a DEN (O'Hanley et al. 2013). This model is thus able to account for different habitat types, habitat quality, and the dispersal range of a species of interest, as well as the spatial arrangement of barriers within a network. The formulation of C presented in O'Hanley et al. (2013) differs from its original formulation (Diebel et al. 2010) in that it accounts for both artificial and natural barriers on connectivity. Details of the C connectivity metric, as described in O'Hanley et al. (2013) and presented here for a single watershed with no natural barriers, are given below.

The availability (y_{sh}) of habitat type h accessible from stream segment s, taking into account the effects artificial barriers, is given by:

$$(1)\ \ y_{sh} = \sum_{t \in N} \theta_{th}\, W_{th} L_t D_{st} \varphi_{st}$$

The summation in equation (1) is over all segments in the watershed, indexed by *s* and *t*. θ_{th} is the proportion of habitat type h in stream segment s. W_{th} is the suitability (0-1) of habitat type h in stream segment s. L_t is the length (m) of stream segment s. D_{st} is the inverse-distance weighting term between segments *s* and *t*. φ_{st} represents the cumulative bidirectional passability of barriers between two segments *s* and *t*.

$$(2)\ \ \varphi_{rst} = \prod_{j \in B_{rst}} (p_j q_j)$$

The cumulative bidirectional passability shown in equation (2) is found by multiplying the individual upstream and downstream passability values for all artificial barriers between stream segments *s* and *t*, where B_{rst} is the set of artificial barriers between stream segments *s* and *t*, indexed by *j*. p_j is the upstream passability of barrier *j* and q_j is the downstream passability of barrier *j*. Thus, this term can be interpreted as, assuming passability is independent at each barrier, the probability that fish can navigate past all barriers while moving from stream segment *s* to segment *t* and back again.

The inverse distance weighting term D_{st} is defined as:

$$(3)\ \ D_{st} = \frac{1}{1 + \left(\frac{d_{st}}{d_0}\right)^2}$$

Where d_{st} is the distance along the stream network between the centroids of segments *s* and *t* and d_0 is the typical dispersal distance of target resident fish species.

From this, y_{sh} can be interpreted as follows: "the amount of habitat type h accessible from segment *s* is determined by the amount of quality-weighted habitat type h available in each stream segment *t* ($\theta_{th} W_{th} L_t$), adjusted by the inverse-distance (D_{st}) and level of cumulative bidirectional passability (φ_{st}) between segments *s* and *t*, summed over all segments *t* in the same watershed as *s* ((*s,t*) ∈ *N*)" (O'Hanley et al. 2013). Next, assuming no barriers are present, the baseline availability (y'_{sh}) of habitat type h accessible from stream segment s is given by:

$$(4)\ \ y'_{sh} = \sum_{t \in N} \theta_{th} W_{th} L_t D_{st}$$

which is then combined with equation (1) to form the C connectivity metric:

$$(5)\ \ Z_s = \frac{1}{m} \sum_{h=1}^{m} \frac{y_{sh}}{y'_{sh}}$$

Finally, with C defined, the total connectivity weighted habitat for a given watershed is given by:

$$(6)\ \ H = \textstyle\sum_{s \in N} L_s Z_s$$

3.2.2 Prioritizing Barrier Removals.—To determine the optimal portfolio of barriers to remove given a specified budget, the OptiPass tool was used (Cadmus Group, Inc). This method selects barriers to mitigate in a watershed in order to maximize the total connectivity-weighted habitat available to stream-resident/potadromous or diadromous fish. The internal optimization routine is derived from the Resident-Fish Passage Barrier Removal Problem (R-FPBRP; O'Hanley et al. 2013). In this framework, stream networks are represented by individual stream segments, each of which is bounded upstream by headwaters or a barrier and downstream by a barrier.

The R-FPBRP optimization procedure was formulated to maximize total connectivity-weighted habitat for stream-resident fish (O'Hanley et al. 2013). A key component of the optimization model is the allowance for partial passability values (0-1) which when multiplied determined the cumulative bidirectional passability. Because this introduced nonlinearity to the R-FPBRP, the model was reformulated as a Mixed Integer Linear Program, a more accurate method for prioritizing barrier removal than scoring and ranking or greedy heuristics (O'Hanley et al. 2013). Using the following decision variables:

$$x_{ij} \begin{cases} 1 & \text{if mitigation project } i \text{ is selected for barrier } j \\ 0 & \text{otherwise} \end{cases}$$

the optimization model formulated by O'Hanley et al. (2013) to maximize total connectivity-weighted habitat for stream resident fish (and presented here for a single watershed with no natural barriers) is given below:

$$(7)\quad \max H = \sum_{s\,\in\,N} L_s Z_s$$

s. t.

$$(8)\quad \varphi_{st} = \prod_{j \in B_{st}} \left(\left(p_j^0 + \sum_{i\,\in A_j} p_{ij} x_{ij} \right) \left(q_j^0 + \sum_{i\,\in A_j} q_{ij} x_{ij} \right) \right) \quad (s,t) \in N, s \neq t$$

$$(9)\quad \varphi_{st} = \varphi_{ts} \quad (s,t) \in N, s \neq t$$

$$(10)\ \varphi_{ss} = 1 \quad s \in N$$

$$(1a)\ \ y_{sh} = \sum_{t \in N} \theta_{th}\, W_{th} L_t D_{st} \varphi_{st} \quad s \in N, h\ \leq m$$

$$(5a)\ \ Z_s = \frac{1}{m} \sum_{h=1}^{m} \left(\frac{1}{y'_{sh}}\right) y_{sh} \quad s \in N$$

$$(11)\ \sum_{i\, \in\, A_j} x_{ij} \leq 1 \quad \forall j \in J$$

$$(12)\ \sum_{j \in J} \sum_{i\, \in A_j} c_{ij}\, x_{ij} \leq b$$

$$(13)\ \ x_{ij} \in \{0{,}1\} \quad \forall j \in J, i \in A_j$$

The objective function (7) maximizes the total amount of connectivity-weighted habitat *H* across a study watershed. Constraints (8) determine the amount of cumulative bidirectional passability between each unique pair of connected stream segments *s* and *t*; that is, the product of up- and downstream passabilities at barriers, accounting for the effects of increased passability via barrier mitigation. Constraint (9) states the cumulative passability from stream segment *t* to *s* is equal to the cumulative passability from *s* to *t*. Constraint (10) sets the cumulative passability of any segment *s* to itself equal 1. Constraints (1a) determine the amount of y_{sh} of available habitat *h* accessible from stream segment *s*. Constraints (5a) determine the level of connectivity Z_s for stream segment *s*. Constraint (11) sets the requirement that only one barrier mitigation project can be selected for each barrier *j*. Inequality (12) limits the cost of barrier mitigation to be less than or equal to the available budget. Finally, constraint (12) forces a binary restriction on the x_{ij} decision variable.

3.2.3 Case Study.—The Simonette Watershed (SW) is located is west-central Alberta, Canada in the Upper Peace River Basin (Fig. 3.1). Covering 5,390 km of Rocky Mountain foothill and boreal forests, the watershed is characterized by low gradient, meandering streams that feed the mainstem Simonette River (Scrimgeour et al. 2003, NRC 2006). Provincial records indicate that twenty-two species of fish representing nine families have been recorded in the SW (Alberta Fish & Wildlife Management Information System, accessed April 2015). This case study specifically considers the Latronell and Deep Valley Creek sub-watersheds of the Simonette River (LW and DVW, respectively). The majority of stream crossings that are impeding fish movement in the SW are located in these sub-watersheds (Fig. 3.1).

Geospatial data for river network features and streams crossings were acquired from the Foothills Stream Crossing Partnership (FSCP) and Alberta Government Informatics Branch. Streamlines were 1:24,000-scale polylines that were subsequently prepared to ensure network topology was fully connected; as such, their positions were often slightly modified. The LW covers 980 km^2 and contains approximately 532 km of total stream length and the DVW covers 964 km^2 and contains approximately 558 km of total stream length.

Crossing surveys were conducted in 2013 (see Chapter 2) using protocols developed by the Foothills Research Institute (fRI) and Alberta Government (AESRD 2014). This dataset was augmented with crossing data obtained from the FSCP. Two hundred and sixty stream crossings were identified in the SW; including 172 culverts, 78 bridges, and 10 fords. Crossings identified on inspections as bridges, fully passable culverts, and fords were removed from the dataset because bridges do not impede fish passage and fords were on ephemeral streams that dry completely in summer. The remaining crossings were visually inspected in ArcGIS to ensure locations were coincident with mapped streams; inconsistent survey points were snapped to stream polylines using the *snappoints* tool in Geospatial Modeling Environment (Beyer 2012). From this dataset crossings on first order streams were excluded as they have been shown to have a low potential of fish occurrence (Scrimgeour et al. 2003). The remaining dataset contained 14 identified barriers in the DVW and 9 barriers in the LW (Fig. 1). Attributes from surveys for each stream crossing include qualitative estimates of upstream fish passability. Qualitative estimates were converted into quantitative values (full barrier and partial barrier would be represented by 0 and 0.5, respectively; Diebel et al. 2014, King and O'Hanley 2014). As in previous studies (O’Hanley 2011, O’Hanley et al. 2013), downstream passability is assumed to be 1 in all cases (i.e. 100% passable).

Estimating costs associated with culvert restoration is site specific requiring detailed information acquired only through site assessments by qualified personnel, which were not available for this region. Literature values for culvert remediation projects vary from $10,000 CAD (Parker 1999) to $100,000 CAD (Fish Passage Technical Working Group 2012), however, foresters in the region reported higher costs that vary with stream size and road type. Based on input from foresters in the region, I used stream order and road type as a means to estimate culvert costs to be $50,000 for 2nd order, $100,000 for 3rd order, $150,000 for two 3rd order streams under major haul roads, and $250,000 for one 4th order stream with significant fill (~75 m). These values are not specific estimates and are only expected to be representative of the range of mitigation costs for this region.

Life history and habitat attributes of Arctic Grayling and Bull Trout were implicitly incorporated into the prioritization process in two ways. First, stream segment attributes included estimated length (m), Strahler stream order (1-7), and habitat quality (0-1). Habitat quality indices were calculated based on estimated road density in sub-watersheds of the greater SW. Road density has been shown to be strongly negatively related to the occurrence of Bull Trout (Ripley et al. 2005), and general declines in both Arctic Grayling and Bull Trout populations from fragmentation and habitat degradation has been attributed to increasing road development (ASRD 2005, 2009). In a nearby watershed, there is less than a 10% chance of Bull Trout occurrence when road density is as low as 0.8 $km \cdot km^{-2}$ (Ripley et al. 2005). Accordingly, road density was scaled so a value of 1.0 equates to a road density of 0.0 $km \cdot km^{-2}$, and a value of 0.0 equates to a road density of 0.8 $km \cdot km^{-2}$.

Next, dispersal distance (d_0) was estimated for Arctic Grayling and Bull Trout. Dispersal distance should be set to account for the distribution of seasonal movements a species of interest makes in a fully connected stream network (Diebel et al. 2014). Here fluvial populations of Arctic Grayling and Bull Trout were accounted for. Arctic Grayling may migrate from 10 to over 150 km to find spawning habitat (Nelson 1954), but generally average between 33 km and 50 km in other watersheds of the Upper Peace River Basin (Stanislawski 1997, Blackman 2002). Bull Trout similarly may travel up to 250 km to access spawning grounds (Burrows et al. 2001), but likely average significantly less in watersheds with habitat spaced closer together (ASRD 2009). In previous optimization analyses done in Wisconsin, a value of 20 km was used which approximated the spatial autocorrelation function of numerous stream species (Diebel et al. 2010, O’Hanley et al. 2013, Diebel et al. 2014). Given that differences in dispersal may greatly alter estimates of fish viability (Poos and Jackson 2012), a sensitivity analysis was done to explore how varying dispersal values representing reasonable approximations of seasonal movement of fluvial trout affected the prioritization process. These distances included 5, 10, 20, 50, 100, 150, and 250 km, representing a natural gradient from lowest to highest connectivity needed to maintain these species.

Barrier mitigation options were to repair passability at each culvert to 1.0 and assumed identical at all culverts (i.e. a binary decision variable; “repair” or “do not repair”). A sensitivity analysis was performed to investigate how optimal barrier removal solutions varied as budget levels increased; budget levels were raised in \$100K intervals from \$100K up to \$1.2 million. The OptiPass toolset is designed for use in ArcGIS 10.1, with the internal optimization routine implemented in Python 2.7.

3.3 Results

Varying the dispersal distance parameter to account for species-specific differences in spatial ranges has an effect on the objective function and thus the net gain in habitat (Fig. 3.2). Because species with longer dispersal ranges are more susceptible to connectivity impairment (Diebel et al. 2010), they benefit more from barrier removal than species with small ranges. Consistently, higher dispersal estimates resulted in lower objective function values at each budget level for each sub-watershed. For example, initial values for the objective function (budget = \$0) were always lower for analyses considering increasingly larger dispersal values. This trend remains constant until all barriers are restored at the maximum budget level. Varying the distance weighing parameter also resulted in variation in the number of barriers selected for removal as well as variation in the average cost per barrier at a particular budget level (Fig. 3.3b, 3.3c, 3.3e, 3.3f).

Generally, as budget levels increase for both sub-watersheds, marginal improvements in accessible habitat diminish (Fig. 3.2a, 3.2d). For the DVW, this proceeds in a roughly concave manner with distinctive inflections at \$100K and \$400K (Fig. 2a). The LW trend however increases at a more linear rate with a slight inflections at \$400K (Fig. 2d). Generally, this indicates that barriers adding the most to the objective function are selected for removal at lower budget levels, while those that contribute less are selected as additional funds become available. For the DVW, this trend is much more apparent where modest levels of investment yield large gains in accessible habitat. This indicates that a large proportion of the potential connectivity gain can be realized with a moderately low budget. For instance, considering a 20 km dispersal distance, an investment of \$200K (i.e. 20% of the total budget required to restore all barriers to full passability) would make accessible 60.7% (34.8 km) of currently inaccessible habitat. The more linear net habitat gain curve for the LW indicates many barriers in this watershed contribute similar gain to the objective function when selected for removal.

In terms of the number of barriers removed for each budget level, the increase was nearly linear with multiple inflections present for both sub-watersheds (Fig. 3.3b, 3.3e), though the location of inflections varied depending on the distance weighting parameter value. For the DVW at low budget levels, approximately one additional barrier is removed per \$100K increase in budget; however, the trend increased at higher budget levels where approximately two additional barriers are removed for each \$100K increase. Major inflections are apparent at \$500K and 600K. The LW follows a slightly steeper curve, with a prominent inflection at \$400K and \$500K. When increasing the budget from \$100K to \$200K in the DVW ($d_0 = 20$ km), only one additional barrier selected for mitigation increases the amount of available habitat from 499 km to 531 km, suggesting these barriers contribute the most to connectivity loss within the sub-watershed; this highlights the impacts are individual barriers.

The average cost to mitigate a barrier in the DVW was \$75,000 and \$88,889 in the LW (Fig. 3.3c, 3.3f). The mean cost for barrier portfolios at varying budget levels were always more expensive than the mean budget to mitigate all barriers in the DVW, and varied greatly for the LW. Additionally, varying d_0 resulted in different average costs for mitigation at different budget levels. The mean cost for selected barriers in the DVW remained constant at \$100K until a budget level of \$500K where it spikes to either a mean of \$125K ($d_0 = 20$ km), remains constant ($d_0 \geq 100$ km), or drops to \$83,333 ($d_0 = 5$, 10 km). At \$600K, the mean mitigation cost returns to \$100K ($d_0 \leq 50$ km) or drops to \$85,714 ($d_0 \geq 100$ km). At an investment of \$600K, the mean cost returns to \$100K, then gradually declines to the mean for all barriers at a budget of \$1.1M. The mean cost for selected barrier mitigation in the LW dropped from \$100K to \$66,667K from a budget of \$100K to \$400K, spiked at \$500K to a mean of \$125K, and then gradually declined to the mean of all barriers at \$75K. When d_0 was set to 50 km, the spike to a mean of \$125K occurred one budget level earlier. Interestingly, the trend of average barrier mitigation cost differed greatly when d_0 was set to 5 km. In this case, the average cost was generally lower than the average to mitigate all barriers in the sub-watershed. This indicates that cost, as opposed to connectivity improvement, is a primary driver in selecting barriers when low dispersal values are used. For example, in the LW, as the budget increased from \$400K to \$500K (when $d_0 \neq 5$ or 50 km), the number of barriers selected decreases by two while the average cost per barrier and net habitat gain increases (Fig. 3.3). At this point, the optimization routine selected fewer, but more expensive, barriers to remove that in turn provide more gain in net habitat than would less expensive solution sets.

At lower budget levels for both the LW and DVW, barriers selected for restoration tended to be centrally located along third order streams (Table 3.2, 3.3; see Fig. 3.4 for detailed spatial arrangement of barrier portfolios selected for removal across budget levels). As budget levels increased, barriers on smaller second order streams were selected more often. In general, this shows that at low budget levels, the optimization model attempts to restore the largest subnetwork to which additional barrier removal selections at higher levels are added to connect isolated headwater streams. Solutions for each watershed were also not perfectly nested (i.e. barriers selected at low budget levels were not always selected at higher budget levels) as seen in Figure 2b, Figure 3, and Table 2. Specifically, at low budget levels solutions were not perfectly nested, however as budget levels exceed \$500K, solutions become additive with all barriers at lower budgets selected at higher budgets. Further, while barrier portfolios were generally consistent across dispersal distance values at specific budget levels, there were observed differences (Table 3.2, 3.3). For example, at an initial investment level of \$100K in the LW, when $d_0 = 5$, two less expensive barriers located closer to headwaters are selected over a single more expensive and centrally located barrier selected when $d_0 \geq 10$.

3.4 Discussion

Over the past decade and in response to anthropogenic alterations to riverine systems, millions of dollars have been invested in efforts to restore ecological connectivity for aquatic organisms. Because restoration projects of this nature are often very costly, techniques are needed to prioritize restoration decisions systematically in order to achieve the greatest ecological return for each dollar invested. This case study highlights the importance of incorporating species-specific attributes into barrier optimization analyses, and further contributes to restoration planning in the region by elucidating the restoration potential for the Simonette Watershed given pre-determined levels of investment.

3.4.1 Importance of net gain vs budget curves.—Analyses examining the relationship between net habitat gain and monetary investments can be extremely useful when planning for systematic, watershed-based restoration planning (O’Hanley et al. 2013, Diebel et al. 2014, King and O'Hanley 2014). The analysis presented here should be beneficial to managers by elucidating how particular levels of investment or budgetary constraints will affect the amount of potential habitat that can be restored. Major benefits of this type of analysis include: *i*) the identification of critical investment levels required to effect the largest ecological gain in a particular watershed (O’Hanley 2011, O’Hanley et al. 2013) or larger geographic region such a state or province (King and O'Hanley 2014), and *ii*) the ability to provide defendable evidence for budget planning to regulators and other involved parties (O’Hanley 2011). Indeed, in this analysis I show that significant habitat gains for fish such as Arctic Grayling and Bull Trout can be achieved with a modest investment, particularly for the DVW. Further, I show how varying the dispersal distance parameter within the model can affect the resulting objective function and the amount of habitat regained at particular budget levels. This exemplifies the benefit of performing sensitivity analyses across a range of budgets and biological attributes for watersheds and species of conservation concern.

3.4.2 Nestedness.—An important finding of this study was that barrier portfolios were not perfected nested. Nested solutions indicate that barriers selected for restoration at lower budget levels are also selected at higher budget levels. As shown by O'Hanley (2011), O'Hanley et al. (2013), and Weiter (2015), solutions to optimization-based prioritization models need not be nested from one budget level to the next. A lack of nestedness may be a potential concern as it indicates small changes in budget levels may result in large changes in net habitat gain and the portfolio of barriers selected for mitigation. For both sub-watersheds examined herein, lack of nestedness is more pronounced at low budget levels, while solutions become additive at higher budget levels (i.e. all perfected nested at budgets > $500K). Knowledge of the spatial configuration of solutions at different budget levels can increase confidence in decision-makers with regard to final restoration decisions as it can be assumed near-optimal gains will still be achieved even if particular barriers need be replaced with others (Weiter 2015).

3.4.3 Average Barrier Cost.—Selected barriers in this study tended to be more expensive to mitigate than the average barrier in either sub-watershed, a result consistent with King and O'Hanley (2014), but inconsistent with O'Hanley (2011), O'Hanley et al. (2013), and Weiter (2015) who found selected barriers to be less expensive than the average. This difference in average cost trends for selected barriers among datasets highlights the importance of incorporating accurate cost estimates for individual barriers; it further illustrates the mechanism by which a lack of perfect nestedness arises in optimal solutions at increasing budget levels. Erroneous cost estimates can lead to inefficient and sub-optimal solutions and reduces the capacity for restoration decisions to remain within pre-determined budgets (Weiter 2015). For the DVW, combining this information with reduced net habitat gains at higher budget levels indicate there is a large number of relatively expensive barrier mitigation portfolios that are significantly restricting habitat access to stream-resident fish. Also, accurate cost estimates allow the optimization routine to sometimes select fewer but more expensive barriers to remove at increasing budget levels which in turn provide more gain in net habitat than would less expensive solution sets. Further, the case may arise where, for economic or social reasons, a particular barrier cannot be removed (e.g., the barrier assigned a mitigation cost of $250K or possibly a culvert under a major highway). The optimization routine used in this study is able to consider this case by forcing that barrier to remain (i.e. never selected for mitigation), but still accounting for its effect on overall watershed connectivity. Accordingly, investing resources to estimate accurately barrier mitigation costs across watersheds is imperative for determining optimal barrier removals and likely well worth the initial investment.

3.4.4 Management Implications.—In the western boreal forest, both Arctic Grayling and Bull Trout are experiencing precipitous population declines and range contractions due to habitat fragmentation and degradation (Ripley et al. 2005, Walker 2005, ASRD 2009). Reducing the impacts of connectivity loss by reconnecting isolated habitat is an important first step in the restoration process (Roni et al. 2002), and towards recovering lost or severely depleted populations. Because populations of these species often exhibit substantial variation in their dispersal abilities (Stanislawski 1997, Burrows et al. 2001, Blackman 2002), and given that differences in dispersal may affect estimates of fish viability (Poos and Jackson 2012), accounting for variable dispersal is integral to determining meaningful restoration priorities. The results of this study suggest that incorporating appropriate dispersal estimates for species of concern in particular watersheds will add biological relevance the prioritization process, a finding consistent with research examining methods for maximizing return on investment in conservation (Murdoch et al. 2007). For instance, Murdoch et al. (2007) found that enormous savings as achievable when return-on-investment frameworks are utilized, particularly if the analyses incorporate both cost and biological measures.

The use of optimization modeling to prioritize barrier mitigation specifically assesses ecological connectivity within a stream network and omits the incorporation of other pertinent factors. For example, none of the current prioritization modeling frameworks available (Table B1) incorporate possibly confounding environmental or socio-economic factors in solutions such as recreational fishing impacts or flood control (O'Hanley et al. 2013), nor do they account for safety concerns of the crossing structure or sedimentation hazards. On resource roads in forested watersheds such as the SW, increased sediment transport into stream has been observed and attributed to provincial declines in sensitive fish species such as Bull Trout (Ripley et al. 2005), and is likely a leading factor contributing to altered fish assemblages (Scrimgeour et al. 2008, MacPherson et al. 2012). Incorporating more robust habitat quality metrics may aid in addressing these concerns, as would the formulation of multi-objective optimization models attempting to maximize stream connectivity and minimize sediment inputs (given data on sedimentation hazards for stream crossings exists).

In conclusion, this study contributes to the growing body of work describing the benefits of optimization procedures in prioritizing the mitigation of fish barriers. In particular, I show how the incorporation of different dispersal distance estimates affect optimization solutions and thus highlighting the importance of accounting for species-specific attributes when prioritizing barriers. Further, this study provides pertinent information for managers regarding restoration potential for a large watershed in Alberta experiencing increasing levels of industrial resource development, and should aid in prioritizing restoration action.

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TABLE 3.1.—List of culvert stream crossings in the Latronell Creek (LAT) and Deep Valley Creek (DV) sub-watersheds. UTM coordinates are in the North American Datum 1983, UTM zone 11N.

ID	Watershed	Passability	Cost ($)	Strahler	UTM_E	UTM_N
1394	LAT	0	100,000	3	291448.2	6056692
1580	LAT	0	250,000	4	290880.1	6037643
1600	LAT	0.5	100,000	3	289198.5	6029671
1604	LAT	0	50,000	2	293468.2	6070639
1715	LAT	0	100,000	3	298618.8	6061556
1755	LAT	0.5	50,000	2	291194.6	6055497
1759	LAT	0.5	50,000	2	302143.9	6065322
1765	LAT	0.5	50,000	2	291830.6	6039041
1894	LAT	0	50,000	2	296707.4	6059662
117	DV	0	100,000	3	314459	6028390
448	DV	0.5	50,000	2	320182.8	6030047
648	DV	0	100,000	3	317499.8	6016313
872	DV	0	50,000	2	317644.2	6016536
1291	DV	0	50,000	2	317352.3	6011524
1312	DV	0	50,000	2	319791.4	6010761
1500	DV	0	50,000	2	317996.6	6028541
1602	DV	0	150,000	3	328129.5	6039941
1632	DV	0.5	150,000	3	326063.8	6035454
1635	DV	0	100,000	3	317820.3	6029192
1692	DV	0	50,000	2	297694.4	6017577
1796	DV	0	50,000	2	301533.8	6022258
1797	DV	0	50,000	2	303176.3	6024420
1799	DV	0	50,000	2	304985.6	6024667

TABLE 3.2.—Results from optimization models for the Deep Valley Creek sub-watershed at different budget levels and with different dispersal distance values (d_0), including selected barrier portfolios.

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
5	$0	519.859	0	0	0	
	$100K	522.861	3.002	100,000	1	1635
	$200K	539.568	19.709	100,000	2	448, 1602
	$300K	543.17	23.311	100,000	3	448, 1602, 1635
	$400K	546.77	26.911	100,000	4	448, 1602, 1635, 117
	$500K	548.632	28.773	83,333	6	448, 1602, 1635, 872, 117, 1500
	$600K	550.967	31.108	100,000	6	448, 1602, 1635, 117, 1632, 1500
	$700K	552.744	32.885	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	554.3	34.441	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	555.009	35.15	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	35.46	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	35.461	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291
10	$0	508.78	0	0	0	
	$100K	512.903	4.123	100,000	1	1635
	$200K	536.362	27.582	100,000	2	448, 1602
	$300K	541.002	32.222	100,000	3	448, 1602, 1635
	$400K	545.156	36.376	100,000	4	448, 1602, 1635, 117
	$500K	547.401	38.621	83,333	6	448, 1602, 1635, 872, 117, 1500
	$600K	550.16	41.38	100,000	6	448, 1602, 1635, 117, 1632, 1500
	$700K	552.568	43.788	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	554.202	45.422	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	554.999	46.219	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	46.539	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	46.54	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
20	$0	496.413	0	0	0	
	$100K	499.494	3.081	100,000	1	1635
	$200K	531.252	34.839	100,000	2	448, 1602
	$300K	537.039	40.626	100,000	3	448, 1602, 1635
	$400K	539.755	43.342	100,000	4	448, 1602, 1635, 117
	$500K	544.456	48.043	125,000	4	1602, 1635, 117, 1632
	$600K	547.972	51.559	100,000	6	448, 1602, 1635, 872, 117, 1632
	$700K	549.726	53.313	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	551.966	55.553	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	552.948	56.535	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	553.599	57.186	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	553.7	57.287	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291
50	$0	486.827	0	0	0	
	$100K	494.307	7.48	100,000	1	1635
	$200K	526.17	39.343	100,000	2	448, 1602
	$300K	534.038	47.211	100,000	3	448, 1602, 1635
	$400K	540.289	53.462	100,000	4	448, 1602, 1635, 117
	$500K	543.847	57.02	100,000	5	448, 1602, 1635, 648, 117
	$600K	547.931	61.104	100,000	6	448, 1602, 1635, 872, 117, 1632
	$700K	551.54	64.713	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	553.794	66.967	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	554.879	68.052	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	68.492	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	68.493	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291
100	$0	482.019	0	0	0	

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
	$100K	490.505	8.486	100,000	1	1635
	$200K	522.223	40.204	100,000	2	448, 1602
	$300K	531.091	49.072	100,000	3	448, 1602, 1635
	$400K	538.258	56.239	100,000	4	448, 1602, 1635, 117
	$500K	542.549	60.53	100,000	5	448, 1602, 1635, 648, 117
	$600K	546.571	64.552	85,714	7	448, 1602, 1635, 648, 872, 117, 1500
	$700K	550.834	68.815	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	553.452	71.433	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	554.787	72.768	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	73.3	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	73.301	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291
150	$0	480.662	0	0	0	
	$100K	489.405	8.743	100,000	1	1635
	$200K	521.026	40.364	100,000	2	448, 1602
	$300K	530.15	49.488	100,000	3	448, 1602, 1635
	$400K	537.571	56.909	100,000	4	448, 1602, 1635, 117
	$500K	542.113	61.451	100,000	5	448, 1602, 1635, 648, 117
	$600K	546.319	65.657	85,714	7	448, 1602, 1635, 648, 872, 117, 1500
	$700K	550.568	69.906	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	553.309	72.647	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	554.753	74.091	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	74.657	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	74.658	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291
250	$0	479.858	0	0	0	
	$100K	488.745	8.887	100,000	1	1635
	$200K	520.302	40.444	100,000	2	448, 1602

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
	$300K	529.569	49.711	100,000	3	448, 1602, 1635
	$400K	537.138	57.28	100,000	4	448, 1602, 1635, 117
	$500K	541.836	61.978	100,000	5	448, 1602, 1635, 648, 117
	$600K	546.153	66.295	85,714	7	448, 1602, 1635, 648, 872, 117, 1500
	$700K	550.392	70.534	100,000	7	448, 1602, 1635, 648, 872, 117, 1632
	$800K	553.211	73.353	88,889	9	448, 1602, 1635, 1796, 648, 872, 117, 1632, 1500
	$900K	554.73	74.872	81,818	11	448, 1602, 1635, 1796, 1797, 648, 1692, 872, 117, 1632, 1500
	$1M	555.319	75.461	76,923	13	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500
	$1.1M	555.32	75.462	75,000	14	448, 1312, 1602, 1635, 1796, 1797, 1799, 648, 1692, 872, 117, 1632, 1500, 1291

TABLE 3.3.—Results from optimization models for the Latronell River sub-watershed at different budget levels and with different dispersal distance values (d_0), including selected barrier portfolios.

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
5	0	506.722	0	0	0	
	1	510.513	3.791	50,000	2	1604, 1755
	2	512.453	5.731	66,666	3	1604, 1394, 1755
	3	514.926	8.204	75,000	4	1604, 1394, 1715, 1755
	4	515.921	9.199	66,666	6	1604, 1894, 1394, 1715, 1755, 1759
	5	517.403	10.681	100,000	5	1604, 1765, 1580, 1394, 1755
	6	519.951	13.229	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	520.966	14.244	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	14.756	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
10	0	498.693	0	0	0	
	1	502.9	4.207	100,000	1	1394
	2	507.447	8.754	100,000	2	1394, 1715
	3	512.096	13.403	75,000	4	1604, 1394, 1715, 1755
	4	513.308	14.615	66,667	6	1604, 1894, 1394, 1715, 1755, 1759
	5	515.498	16.805	125,000	4	1580, 1394, 1715, 1755
	6	519.323	20.63	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	520.569	21.876	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	22.785	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
20	0	489.436	0	0	0	
	1	497.565	8.129	100,000	1	1394
	2	503.862	14.426	100,000	2	1394, 1715
	3	508.634	19.198	75,000	4	1604, 1394, 1715, 1755
	4	510.24	20.804	66,667	6	1604, 1765, 1894, 1394, 1715, 1755

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
	5	514.973	25.537	125,000	4	1580, 1394, 1715, 1755
	6	518.955	29.519	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	520.317	30.881	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	32.042	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
50	0	476.481	0	0	0	
	1	490.123	13.642	100,000	1	1394
	2	498.402	21.921	100,000	2	1394, 1715
	3	504.047	27.566	75,000	4	1604, 1394, 1715, 1755
	4	506.384	29.903	133,333	3	1580, 1394, 1755
	5	514.273	37.792	125,000	4	1580, 1394, 1715, 1755
	6	518.691	42.21	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	520.263	43.782	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	44.997	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
100	0	464.646	0	0	0	
	1	482.788	18.142	100,000	1	1394
	2	492.953	28.307	100,000	2	1394, 1715
	3	499.726	35.08	75,000	4	1604, 1394, 1715, 1755
	4	502.678	38.032	66,667	6	1604, 1765, 1394, 1715, 1755, 1759
	5	513.2	48.554	125,000	4	1580, 1394, 1715, 1755
	6	518.281	53.635	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	520.114	55.468	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	56.832	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
150	0	458.545	0	0	0	
	1	478.655	20.11	100,000	1	1394
	2	489.595	31.05	100,000	2	1394, 1715

d_0	Budget	Objective (km)	Net Gain (km)	Mean Cost ($)	No. Barriers	Portfolio
	3	496.859	38.314	75,000	4	1604, 1394, 1715, 1755
	4	501.227	42.682	133,333	3	1580, 1394, 1755
	5	512.566	54.021	125,000	4	1580, 1394, 1715, 1755
	6	518.015	59.47	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	519.955	61.41	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	62.933	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
250	0	453.448	0	0	0	
	1	474.954	21.506	100,000	1	1394
	2	486.407	32.959	100,000	2	1394, 1715
	3	494.011	40.563	75,000	4	1604, 1394, 1715, 1755
	4	500.161	46.713	133,333	3	1580, 1394, 1755
	5	511.98	58.532	125,000	4	1580, 1394, 1715, 1755
	6	517.755	64.307	100,000	6	1604, 1765, 1580, 1394, 1715, 1755
	7	519.764	66.316	87,500	8	1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759
	8	521.478	68.03	88,889	9	1600, 1604, 1765, 1894, 1580, 1394, 1715, 1755, 1759

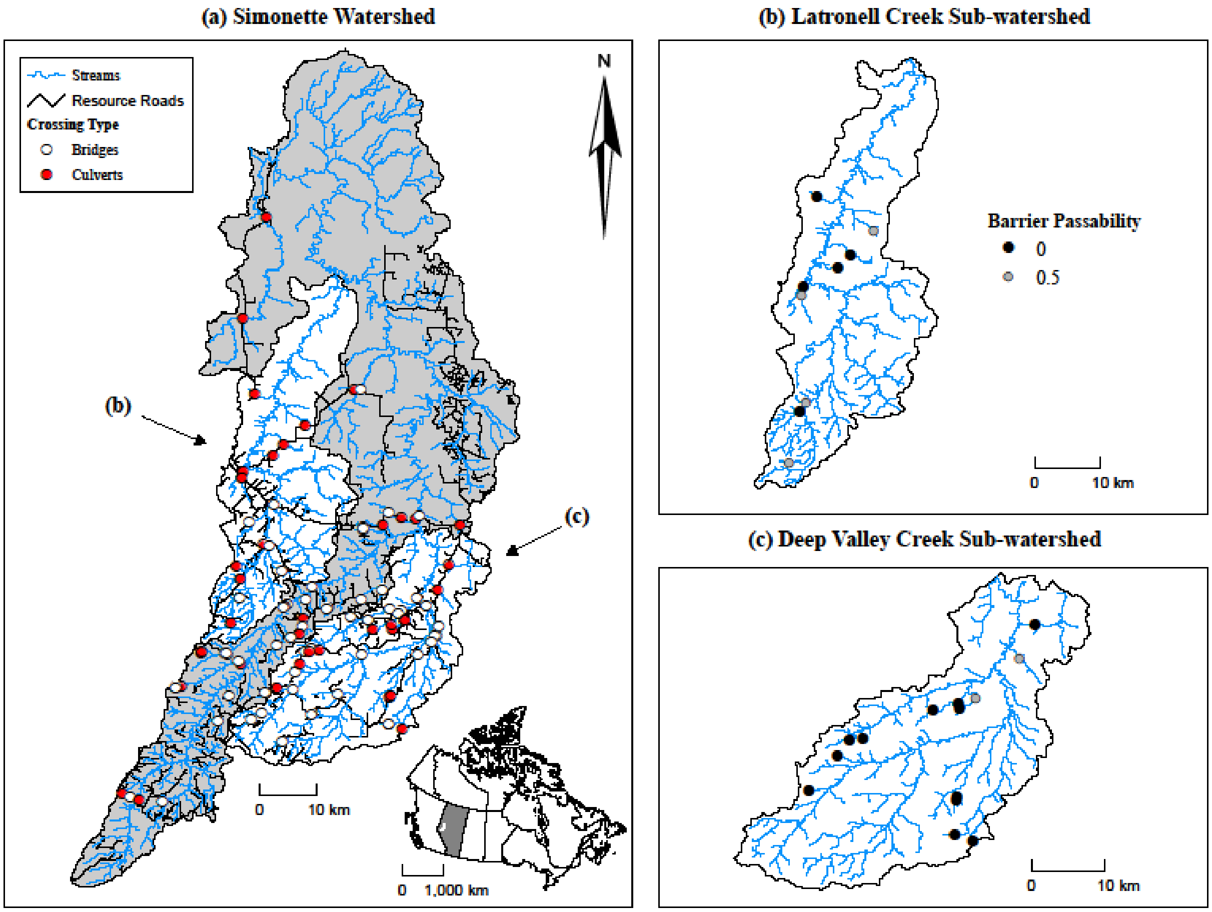

FIGURE 3.1.—Map of the Simonette Watershed in west-central Alberta: (a) Simonette Watershed showing stream (Strahler order ≥ 2) and resource road networks, and culvert (red circle) and bridge (white circle) stream crossing locations; (b,c) Latronell Creek and Deep Valley Creek sub-watersheds showing locations of culvert passage barriers with associated passability values.

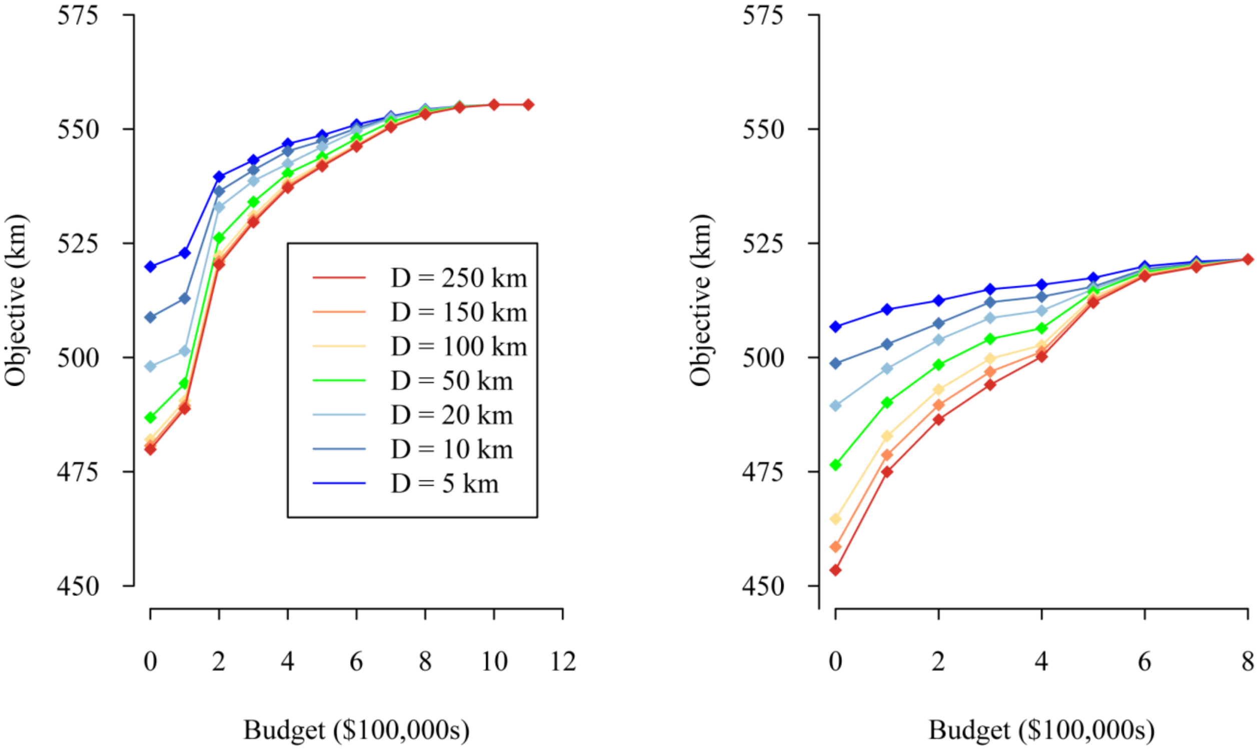

FIGURE 3.2.—Objective function across budget levels for the Deep Valley Creek and Latronell River sub-watersheds. D = typical dispersal distance of target resident fish species.

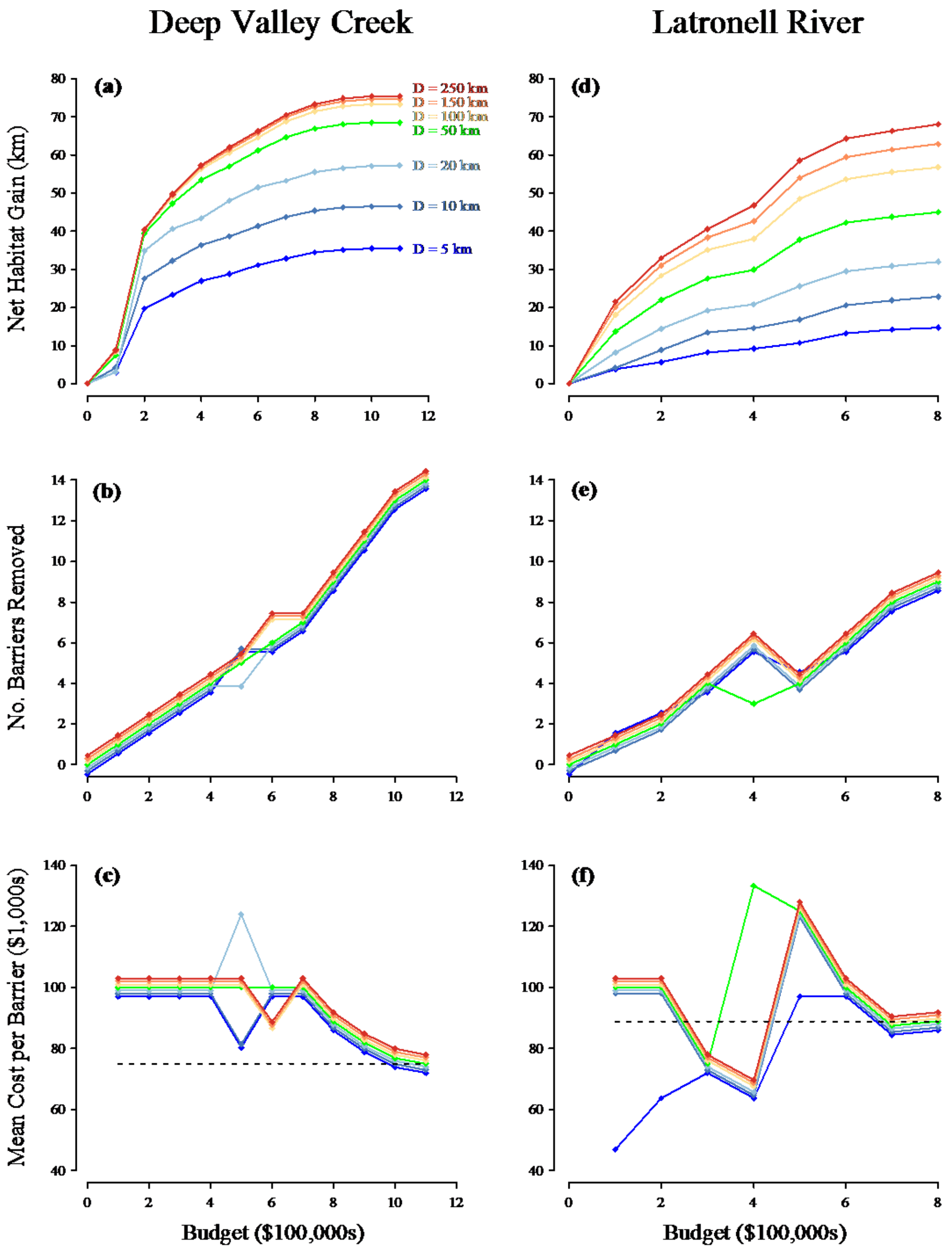

FIGURE 3.3.—For budgets of $100K to $1.1M for the Deep Valley Creek sub-watershed (a,b,c) and the Latronell River sub-watershed: (a,d) net habitat gain (km), (b,e) number of barriers removed, and (c,f) the mean cost of mitigation per barrier portfolio with the mean cost of mitigating all barriers in each sub-watershed given by horizontal lines.

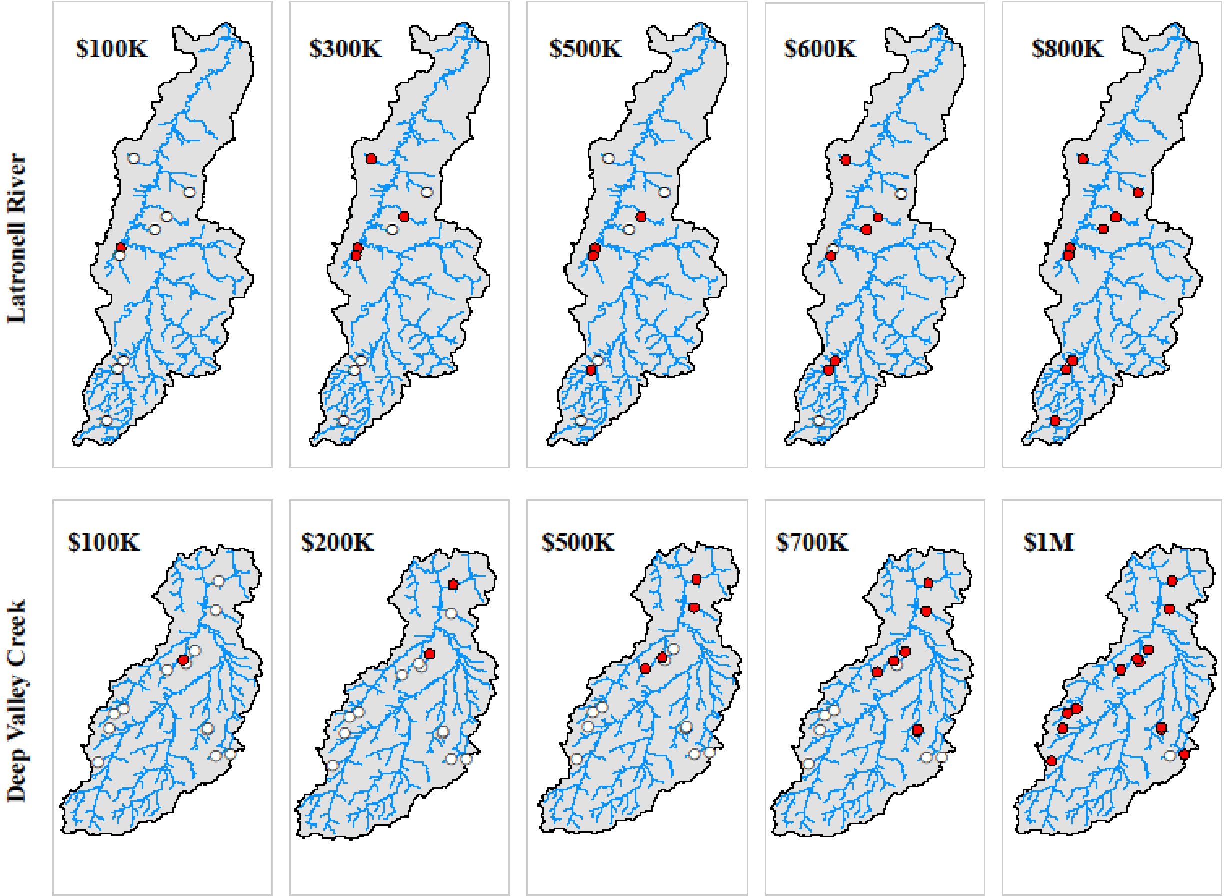

FIGURE 3.4.—Optimal barrier portfolios for the Latronell Creek (top row) and Deep Valley Creek (bottom panel) sub-watersheds at five budget levels (dispersal distance = 20 km). Solid red circles denote fish barriers selected for removal or repair, while all other barriers are denoted with solid white circles.

Chapter IV: Thesis Summary

Stream crossings on industrial resource roads are an increasingly dominant feature in watersheds of Alberta. Growing research in the province has provided evidence of the detrimental impacts these structures are having on fluvial fish populations. This thesis builds on previous research by investigating the degree to which these structures are affecting fluvial fish communities in the boreal forest and how management actions can mitigate potential impacts to stream connectivity. Herein, I show that stream crossing may be driving large-scale changes in stream fish communities in the boreal forest through alterations to fish habitat and habitat fragmentation. Further, my research highlights the efficacy of a novel, easy to use watershed-based barrier prioritization tool to maximize ecological restoration gains given a particular budget.

Chapter 2 presented a case study assessment of the impacts to fish communities and fish habitat from industrially installed stream crossings in the Simonette Watershed. While I can only speculate on the long-term effects of stream crossings, this research provides evidence that the synergistic effects of habitat alteration and fragmentation may be shifting stream fish assemblages towards those dominated by generalist, tolerant species. Accordingly, problem stream crossings, particularly those examined in this study, require restoration action to mitigate detrimental impacts. Specifically removal and replacement with open-bottomed culverts, or direct repair methods (e.g. baffle installation, riprap, bank stabilization) should be pursued. Following this, I recommend new stream crossing installations be either open-bottomed culverts or bridges, and that extreme care should be taken with regard to sediment mobilization during construction. Additionally, as highlighted by MacPherson (2011), mitigation solutions presented by Park (2006) should be adopted.

Chapter 3 built on chapter 2 by examining the restoration potential for barrier mitigation in the Simonette Watershed using novel optimization methods. I used a recently developed custom ArcGIS toolbox (OptiPass) that incorporates the C connectivity metric and optimization modeling routines to evaluate to barrier removal across a range of budget levels. I highlight the efficacy of this tool for use in Alberta watersheds, particularly as it has minimal data requirements, is applicable to both stream-resident and long-range migratory species, and significantly reduces the mathematical and technical expertise needed to perform these relatively complex optimization analyses. It should be noted that of the three barrier prioritization procedures discussed in Chapter 3 (i.e. scoring and ranking, iterative prioritization, and budget optimization; Table B1), each has conditions and limitations to its implementation in a particular jurisdiction including funding availability, decision-maker involvement and relative coordination, and the number a barriers that can feasibly be removed (see Diebel et al. 2014).

4.1 Future Studies

Generally, studies examining stream crossing improvements and the resulting effects on sedimentation reduction and improvement of stream connectivity are lacking. Restoration success is usually determined though qualitative site observations. For example, of 345 stream-restoration projects reviewed by Roni et al. (2008), only 5 examined culvert replacements, of which 3 failed to consider improvements in water quality or biotic health. The results of the study presented in Chapter 2 should be used as a benchmark to measure the efficacy and success of management actions after they are implemented. I specifically recommend a before-after-control-impact paired series (BACIPS) study (Osenberg et al. 2006) be conducted on culverts in the Simonette (or other watershed with higher provincially determined priority). Nislow et al. (2011) propose a reasonable management goal for barrier remediation efforts be restoring the relative longitudinal equivalence of species richness and abundance above and below potential barriers to fish movement. Considering the stark differences in species richness and density observed above and below culverts in Chapter 2, this management goal appears to be appropriate and feasible. Additional metrics accounting for changes in community composition could also be incorporated.

Regarding barrier prioritization, future studies on watershed assessments could incorporate more refined habitat quality indices or different metrics for habitat types such as spawning or overwintering habitat. A simple extension of the prioritization analysis presented herein could be to incorporate larger watersheds or even river basins into a single analysis. In this way managers could examine the restoration potential for connectivity improvement across broad ecological regions, as well as investigate their cost-effectiveness. Alternatively, where particular watersheds have been *a priori* identified for restoration action—as is the case in Alberta whereby fish stocks are assessed on a watershed-by-watershed basis through the Fish Sustainability Index—individual barrier prioritization assessments can be conducted on a case-by-case basis. Prioritizing watersheds for restoration action before any within-watershed prioritization assessments is done is good practice. This will allow managers to: *i*) identify problem watersheds and prioritize them for restoration action, *ii*) conduct directed, detailed surveys of problem watersheds to estimate barrier passability and replacement costs, and *iii*) perform optimization-based prioritization assessments to determine restoration potential.

The is a pressing need to undertake systematic inventories of stream crossings to determine the spatial extent of barriers at both provincial and national scales (Januchowski-Hartley et al. 2013). Knowing the spatial relationship of barriers within a watershed allows for the use of quantitative spatial prioritization methods that have been shown to be vastly superior to simple scoring and ranking techniques (O’Hanley and Tomberlin 2005, O’Hanley 2011, O’Hanley et al. 2013, Diebel et al. 2014). This may be particularly important in Alberta where studies have highlighted the negative impacts industrial resource development on freshwater fish (Ripley et al. 2005, Park et al. 2008, Scrimgeour et al. 2008, MacPherson et al. 2012). The spatial database of road crossing currently being developed by the FSCP and Alberta managers will make possible such analyses for watersheds across the province and is a major step toward informed and cost-effective decision-making. Moreover, while the prioritization presented above assesses connectivity for individual watersheds, optimization routines such as this have been shown to be highly scalable and efficient. For example, King and O'Hanley (2014) used a similar optimization model to assess barrier mitigation across the entire state of Maine. Indeed, their formulation was highly efficient and was able to account for many thousands of barriers.

A significant impediment to the use of optimization methods for watershed restoration planning, despite the numerous studies exemplifying their benefits (Kuby et al. 2005, O’Hanley and Tomberlin 2005, Zheng et al. 2009, O’Hanley 2011, O’Hanley et al. 2013, King and O'Hanley 2014), includes: *i*) the need for specialized expertise in programming or operations research to formulate models, and *ii*) their lack of transparency to decision-makers (O’Hanley and Tomberlin 2005, Beechie et al. 2008). The OptiPass tool is the first GIS-based spatial-decision support tool that allows end-uses and managers with limited mathematical expertise to perform detailed spatial analyses on riverscapes and watersheds similarly to other software available for terrestrially based, systematic conservation planning (e.g. ZONATION, MARXAN). I recommend the use of optimization modeling for prioritizing barrier restoration activities in Alberta, and in particular the use of the OptiPass tool for informing watershed restoration and decision-making. This tool has minimal data requirements, is applicable to both stream-resident and long-range migratory species, and significantly reduces the mathematical and technical expertise needed to perform these relatively complex optimization analyses.

As a caveat, while prioritization models may be mathematically correct, solutions are likely imprecise due to reliance on coarse cost and passability estimates (O'Hanley et al. 2013). As with all models, the efficacy of solutions produced is a function of the quality of input data, and thus excess care and effort should be employed when estimating costs, habitat quality, and other parameters. Accordingly, it should be bore in mind that barrier prioritization analyses are a single component of a larger prioritization process. Thus, even when it is assumed barrier input data is precise and complete, solutions must be reviewed by experts to confirm their feasibility (O'Hanley et al. 2013).

In conclusion, this thesis provides an expanded view of stream crossing effects on freshwater fish communities in boreal streams, along with recommendations for the use of barrier prioritization methods in Alberta for improving stream connectivity. The research presented herein should be used to guide management action and inform best practices for future roads development and stream crossing installations; for without prompt action, stream crossings will undoubtedly continue to impair fish habitat, isolate populations, and contribute to provincial declines in sensitive northern fishes.

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Appendices

Appendix A.—Supplementary information for Chapter 2

TABLE A1.—Selected physicochemical habitat characteristics in upstream (U) and downstream (D) reaches of study streams in the Simonette watershed in west-central Alberta, Canada, summer 2013.

Stream ID	Stream Name	Stream Type	Elevation (masl)	Strahler Order	Location	Stream Gradient (%)	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
101	Smuland Creek	Cul	910	3	U	0.010	3.8	0.7	6.9	1.7	0.24	0.07	0.16	0.04	19.7	0.2
					D	0.040	3.5	0.9	8.5	2.4	0.22	0.11	0.08	0.02	19.1	0.6
102	Wilke Creek	Cul	1072	3	U	0.010	4.1	0.6	5.2	0.8	0.36	0.19	0.09	0.01	18.9	0.3
					D	0.020	3.6	1.6	5.2	0.8	0.18	0.08	0.08	0.01	17.7	0.6
103	Shell Creek	Cul	1047	3	U	0.010	2.8	0.5	3.7	1.0	0.38	0.17	0.12	0.06	19.6	0.6
					D	0.030	2.5	0.5	4.0	0.9	0.19	0.13	0.17	0.09	17.9	0.6
104	Shell Creek	Cul	1022	2	U	0.006	1.6	0.3	1.9	0.5	0.24	0.09	0.10	0.02	18.6	0.2
					D	0.003	1.7	0.5	2.4	0.5	0.20	0.12	0.09	0.02	17.2	0.8
105	Shell Creek	Bri	1099	3	U	0.003	4.9	0.9	6.2	0.9	0.23	0.10	0.29	0.15	16.2	1.0
					D	0.016	4.7	1.0	5.9	0.8	0.19	0.05	0.28	0.08	16.0	0.1
106	Latronell River	Cul	1017	3	U	0.046	2.0	0.4	3.3	0.7	0.17	0.04	0.07	0.02	9.9	0.5
					D	0.016	3.0	0.7	4.1	1.1	0.11	0.04	0.04	0.03	9.5	0.1
107	Latronell River	Bri	1039	4	U	0.006	2.9	1.1	4.5	0.7	0.10	0.03	0.17	0.01	10.4	0.1
					D	0.020	3.1	0.9	4.1	0.5	0.15	0.07	0.20	0.01	10.4	0.0
108	Frying Pan	Cul	1033	4	U	0.003	2.8	0.7	4.4	1.2	0.29	0.14	0.15	0.03	12.5	0.2
					D	0.026	3.3	1.1	5.2	0.8	0.20	0.07	0.16	0.06	11.3	0.2
109	Marion Creek	Cul	1027	3	U	0.006	2.6	0.7	4.7	1.1	0.27	0.11	0.14	0.05	16.2	0.1
					D	0.010	2.4	0.7	5.4	2.5	0.23	0.08	0.14	0.07	14.9	0.0

Stream ID	Stream Name	Stream Type	Elevation (masl)	Strahler Order	Location	Stream Gradient (%)	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
110	Shell Creek	Cul	1031	3	U	0.030	2.6	0.9	5.7	2.3	0.24	0.33	0.22	0.08	12.7	0.2
					D	0.036	2.8	1.3	3.6	1.1	0.12	0.09	0.18	0.06	11.7	0.2
111	Smuland Creek	Cul	958	2	U	0.010	1.8	0.5	2.1	0.6	0.20	0.05	0.06	0.01	21.9	0.6
					D	0.010	2.0	0.5	1.9	0.6	0.28	0.15	0.28	0.38	18.0	0.3
112	Atte Creek	Cul	1126	2	U	0.030	1.4	0.6	2.3	0.6	0.23	0.13	0.01	0.01	15.1	0.4
					D	0.030	2.2	2.2	3.1	2.0	0.13	0.14	0.02	0.01	15.9	0.5
118	Latronell River	Cul	959	3	U	0.026	1.1	0.4	1.9	0.4	0.15	0.10	0.03	0.05	16.0	2.7
					D	0.020	0.8	0.2	2.1	0.3	0.20	0.09	0.04	0.06	16.7	0.2
126	Norton Creek	Bri	879	3	U	0.036	2.0	0.4	3.5	0.5	0.12	0.09	0.15	0.04	12.2	0.1
					D	0.036	2.4	0.6	3.1	0.4	0.07	0.02	0.20	0.04	12.1	0.0
127	Simonette River	Bri	933	3	U	0.026	1.4	0.5	4.1	1.7	0.15	0.08	0.20	0.03	13.8	0.2
					D	0.023	1.4	0.5	4.0	1.4	0.13	0.03	0.24	0.13	13.5	0.3
128	Simonette River	Bri	1009	3	U	0.026	1.9	0.3	3.2	0.6	0.10	0.07	0.18	0.04	11.1	0.1
					D	0.036	1.7	0.5	3.3	0.3	0.05	0.02	0.15	0.05	11.1	0.4
129	Latronell River	Bri	947	3	U	0.010	1.7	0.5	2.4	0.5	0.08	0.05	0.09	0.02	12.1	0.1
					D	0.010	1.5	0.3	2.1	0.4	0.20	0.24	0.09	0.05	12.3	0.2
130	Deep Valley	Bri	1033	2	U	0.013	2.0	0.2	4.0	0.4	0.26	0.19	0.26	0.05	12.4	0.5
					D	0.013	1.9	0.3	4.0	0.4	0.12	0.05	0.43	0.29	13.2	0.2
131	Simonette River	Bri	1088	2	U	0.023	1.5	0.3	3.7	0.6	0.09	0.03	0.18	0.01	10.5	1.7
					D	0.010	1.7	0.3	3.8	0.3	0.08	0.04	0.18	0.04	13.2	0.2
132	Unnamed	Bri	933	3	U	0.026	3.4	0.7	5.7	1.8	0.15	0.08	0.17	0.02	13.3	0.1
					D	0.023	3.8	1.2	6.0	1.9	0.13	0.08	0.19	0.01	13.8	0.3
133	Deep Valley	Bri	1040	3	U	0.026	3.1	0.6	5.1	1.5	0.12	0.11	0.14	0.03	11.2	0.1
					D	0.040	2.8	0.9	7.1	2.2	0.08	0.06	0.12	0.02	11.0	0.1
134	Simonette River	Bri	1162	2	U	0.023	1.1	0.1	1.2	0.1	0.14	0.06	0.09	0.02	16.4	3.5
					D	0.013	1.0	0.2	1.1	0.1	0.13	0.03	0.09	0.01	12.4	3.0
136	Unnamed	Ref	986	3	U	0.026	2.6	0.3	4.3	0.3	0.07	0.04	0.10	0.02	9.4	0.2

Stream ID	Stream Name	Stream Type	Elevation (masl)	Strahler Order	Location	Stream Gradient (%)	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
					D	0.020	2.0	0.8	4.1	0.6	0.06	0.02	0.11	0.04	9.6	0.1
137	Smuland Creek	Ref	940	3	U	0.010	3.2	0.9	8.2	1.9	0.10	0.04	0.30	0.42	11.4	0.0
					D	0.010	3.4	0.6	7.9	1.8	0.08	0.02	0.07	0.01	11.6	0.6
138	Frying Pan	Ref	1005	4	U	0.003	1.6	0.6	5.0	0.5	0.20	0.11	0.13	0.04	13.0	0.0
					D	0.003	2.0	0.5	5.3	0.8	0.19	0.07	0.14	0.04	13.2	0.1
139	Hodges Creek	Ref	1030	3	U	0.006	1.1	0.6	6.9	2.8	0.09	0.03	0.09	0.01	4.3	0.1
					D	0.006	1.2	0.3	6.1	1.9	0.11	0.05	0.09	0.02	4.3	0.1
140	Unnamed	Ref	939	3	U	0.003	2.5	0.7	9.0	1.5	0.12	0.03	0.68	0.03	9.5	0.1
					D	0.043	2.6	1.0	9.1	1.7	0.10	0.03	0.67	0.03	9.6	0.2
141	Unnamed	Ref	1098	3	U	0.020	1.8	0.4	7.3	1.0	0.12	0.04	0.15	0.04	7.2	0.1
					D	0.013	1.7	0.5	7.9	2.6	0.11	0.05	0.15	0.04	7.2	0.1
142	Unnamed	Ref	1116	3	U	0.003	1.5	0.7	4.0	1.4	0.13	0.05	0.07	0.02	9.0	0.1
					D	0.003	2.3	1.5	3.1	1.6	0.13	0.06	0.28	0.42	9.1	0.3
143	Unnamed	Ref	994	2	U	0.020	2.0	0.5	2.3	0.7	0.10	0.03	0.06	0.03	8.5	0.0
					D	0.010	2.0	0.5	2.3	0.6	0.11	0.02	0.25	0.38	8.5	0.0
144	Unnamed	Ref	1080	2	U	0.003	2.3	0.3	2.8	0.7	0.33	0.20	0.05	0.01	8.6	0.0
					D	0.003	2.7	0.5	2.9	0.5	0.38	0.10	0.06	0.01	8.6	0.0
145	Unnamed	Ref	1163	3	U	0.013	2.7	0.8	7.4	1.5	0.13	0.05	0.04	0.01	9.6	0.3
					D	0.026	2.0	0.7	7.4	1.6	0.12	0.02	0.05	0.01	9.5	0.1
146	Unnamed	Ref	960	2	U	0.006	1.9	0.3	2.2	1.0	0.32	0.23	0.06	0.01	9.2	0.1
					D	0.006	2.0	0.2	2.0	0.2	0.27	0.14	0.06	0.01	9.3	0.1

NOTE: Cul = culvert; Bri = bridge; Ref = reference.

TABLE A2.—Physical characteristics and fish passability rating of stream crossings examined in the Simonette watershed of west-central Alberta, Canada, summer 2013. **NOTE**: Cul = culvert; Bri = bridge; RMCP = round metal corrugated pipe; SSB = single-span Bailey. Passability rating as determined by the Alberta Watercourse Crossing Inspection Manual (AESRD, 2014).

Stream ID	Crossing Type	Strahler Order	Structure Type	Length (m)	Culvert Diameter (m)	Culvert Slope (%)	Outlet Drop (m)	Outlet Velocity ($m\cdot s^{-1}$)	Fish Pass Rating[a]
101	Cul	3	RCMP	112	2.50	4.9%	0.49	0.60	Barrier
102	Cul	3	RCMP	64.5	2.50	3.7%	0.07	0.12	Barrier
103	Cul	3	RCMP	36	1.82	2.1%	0.35	1.43	Barrier
104	Cul	2	RCMP	28.5	0.86	1.4%	0.75	0.66	Barrier
106	Cul	3	RCMP	20	1.19	0.7%	0.1	0.25	Barrier
108[a]	Cul	4	RCMP	117	3.00	0.9%	0.03	0.06	Barrier
109	Cul	3	RCMP	64.5	2.50	2.1%	0	0.32	Partial
110	Cul	3	RCMP	39.5	2.65	0.2%	0.07	0.32	Partial
111	Cul	2	RCMP	32.5	2.20	0.0%	0	0.02	None
112	Cul	2	RCMP	28	1.42	0.3%	0.5	0.36	Barrier
118	Cul	3	RCMP	81	1.65	8.75%	0.08	0.15	Barrier
105	Bri	3	SSB	8	-	-	-		None
107	Bri	4	SSB	10	-	-	-		None
126	Bri	3	SSB	11	-	-	-		None
127	Bri	3	SSB	10	-	-	-		None
128	Bri	3	SSB	10	-	-	-		None
129	Bri	3	SSB	12	-	-	-		None
130	Bri	2	SSB	10	-	-	-		None
131	Bri	2	SSB	10	-	-	-		None
132	Bri	3	SSB	10	-	-	-		None
133	Bri	3	SSB	10	-	-	-		None
134	Bri	2	SSB	10	-	-	-		None

TABLE A2.—Species collected in 66 stream reaches in the Simonette watershed of west-central Alberta, Canada, summer 2013. **NOTE**: POC = percent of catch.

Common name	Scientific name	Number of streams	N	POC
Slimy Sculpin	*Cottus cognatus* (Richardson)	27	1043	0.349
Lake Chub	*Couesius plumbeus* (Agassiz)	18	567	0.190
Northern Redbelly Dace	*Chrosomus eos* (Cope)	14	349	0.117
Finescale Dace	*Chrosomus neogaeus* (Cope)	15	331	0.111
Brook Stickleback	*Culaea inconstans* (Kirtland)	10	225	0.075
Redside Shiner	*Richardsonius balteatus* (Richardson)	8	130	0.044
Longnose Dace	*Rhinichthys cataractae* (Valenciennes)	8	88	0.029
White Sucker	*Catostomus commersoni* (Lacepède)	11	91	0.030
Longnose Sucker	*Catostomus catostomus* (Forster)	8	59	0.020
Trout-perch	*Percopsis omiscomaycus (Walbaum)*	3	29	0.010
Pearl Dace	*Margariscus margarita* (Cope)	4	25	0.008
Mountain Whitefish	*Prosopium williamsoni* (Girard)	6	21	0.007
Burbot	*Lota lota* (Linnaeus)	3	14	0.005
Arctic Grayling	*Thymallus arcticus* (Pallas)	5	12	0.004
Northern Redbelly Dace X Finescale Dace	*Chrosomus eos* X *Chrosomus neogaeus*	1	2	0.001
Bull Trout	*Salvelinus confluentus* (Suckley)	1	1	0.000

TABLE A4.—Parameter estimates for linear mixed effect models (LMEs) investigating variation in physical habitat characteristics across stream type (culvert vs. bridge vs. reference), stream location (upstream vs. downstream), and the interaction of stream type and location. Boldface indicates significance at $\alpha = 0.05$. **NOTE**: SE = standard error, df = degrees of freedom. The intercept of each model identifies the condition of downstream, culverted stream types. Model coefficients relate each variable to the intercept. Individual stream-level effects were accounted for by including it as a random intercept term in each model.

Response	Variable	Coefficient	SE	df	*t* value	P
Wetted Width	Intercept	2.5	0.3	30	9.4	0.000
	Bridge	-0.2	0.4	30	-0.4	0.681
	Reference	-0.4	0.4	30	-0.9	0.362
	Upstream	-0.1	0.1	30	-0.8	0.442
	Bridge X upstream	0.1	0.2	30	0.5	0.645
	Reference X Upstream	0.0	0.2	30	0.2	0.863
log(Depth)	Intercept	-1.7	0.1	30	-13.9	0.000
	Bridge	-0.5	0.2	30	-2.8	**0.010**
	Reference	-0.3	0.2	30	-1.8	0.079
	Upstream	0.3	0.1	30	2.8	**0.009**
	Bridge X upstream	-0.1	0.2	30	-1.0	0.344
	Reference X Upstream	-0.3	0.2	30	-1.6	0.112
log(Velocity)	Intercept	-2.4	0.2	30	-10.9	0.000
	Bridge	0.7	0.3	30	2.2	**0.038**
	Reference	0.5	0.3	30	1.5	0.137
	Upstream	-0.1	0.1	30	-0.8	0.442
	Bridge X upstream	0.0	0.2	30	0.2	0.868
	Reference X Upstream	-0.2	0.2	30	-1.0	0.317
Fines	Intercept	43.6	4.1	30	10.7	0.000
	Bridge	-5.3	5.8	30	-0.9	0.363
	Reference	-13.5	5.8	30	-2.3	**0.026**
	Upstream	20.6	4.4	30	4.7	**0.000**
	Bridge X upstream	-27.8	6.2	30	-4.5	**0.000**
	Reference X Upstream	-20.9	6.2	30	-3.4	**0.000**
Rocky	Intercept	18.9	1.4	30	13.7	0.000
	Bridge	2.0	2.0	30	1.0	0.306
	Reference	4.3	2.0	30	2.2	**0.037**
	Upstream	-7.0	1.5	30	-4.6	**0.000**
	Bridge X upstream	9.2	2.1	30	4.3	**0.000**

Response	Variable	Coefficient	SE	df	*t* value	P
	Reference X Upstream	7.3	2.1	30	3.4	**0.002**
Pool	Intercept	39.5	3.3	30	11.9	0.000
	Bridge	-7.9	4.7	30	-1.7	0.104
	Reference	-9.3	4.7	30	-2.0	0.056
	Upstream	20.9	3.4	30	6.2	**0.000**
	Bridge X upstream	-18.7	4.8	30	-3.9	**0.001**
	Reference X Upstream	-20.2	4.8	30	-4.2	**0.000**
Riff	Intercept	29.4	2.8	30	10.7	0.000
	Bridge	8.0	3.9	30	2.1	0.049
	Reference	7.0	3.9	30	1.8	0.080
	Upstream	-10.5	2.6	30	-4.0	**0.000**
	Bridge X upstream	6.2	3.7	30	1.7	**0.108**
	Reference X Upstream	10.9	3.7	30	2.9	**0.007**
Run	Intercept	28.8	2.0	30	14.0	0.000
	Bridge	2.2	2.9	30	0.8	0.450
	Reference	4.5	2.9	30	1.6	0.127
	Upstream	-8.8	2.2	30	-4.0	**0.000**
	Bridge X upstream	10.9	3.1	30	3.5	**0.002**
	Reference X Upstream	7.7	3.1	30	2.5	**0.019**

TABLE A5.—Parameter estimates for linear mixed effect models (LMEs) investigating variation in chemical habitat characteristics across stream type (culvert vs. bridge vs. reference), stream location (upstream vs. downstream), and the interaction between stream type and location. Boldface indicates significance at $\alpha = 0.05$. **NOTE**: SE = standard error, df = degrees of freedom. The intercept of each model identifies the condition of downstream, culverted stream types. Model coefficients relate each variable to the intercept. Individual stream-level effects were accounted for by including it as a random intercept term in each model. Differences in area sampled between stream reaches was accounted for by including an offset (log m^{-2}, excluding richness) in each model.

Response	Variable	Coefficient	SE	df	*t* value	P
Temp	Intercept	15.4	0.78	30	19.7	0.000
	Bridge	-2.8	1.11	30	-2.5	**0.017**
	Reference	-6.3	1.11	30	-5.7	**0.000**
	Upstream	1.0	0.35	30	3.0	**0.006**
	Bridge X upstream	-1.0	0.49	30	-2.0	0.053
	Reference X Upstream	-1.1	0.49	30	-2.2	**0.035**
DO	Intercept	9.2	0.41	30	22.7	0.000
	Bridge	0.7	0.58	30	1.2	0.238
	Reference	1.2	0.58	30	2.0	0.051
	Upstream	-0.8	0.13	30	-6.3	**0.000**
	Bridge X upstream	0.6	0.19	30	3.1	**0.004**
	Reference X Upstream	0.9	0.19	30	4.5	**0.000**
pH	Intercept	8.2	0.07	30	111.3	0.000
	Bridge	0.3	0.10	30	2.4	**0.022**
	Reference	0.3	0.10	30	2.8	**0.009**
	Upstream	-0.1	0.03	30	-3.6	**0.001**
	Bridge X upstream	0.1	0.04	30	1.6	0.120
	Reference X Upstream	0.1	0.04	30	2.5	**0.017**
Cond	Intercept	246.7	19.59	30	12.6	0.000
	Bridge	-4.7	27.70	30	-0.2	0.866
	Reference	14.2	27.70	30	0.5	0.612
	Upstream	-2.3	3.35	30	-0.7	0.502
	Bridge X upstream	-2.2	4.74	30	-0.5	0.647
	Reference X Upstream	0.5	4.74	30	0.1	0.923

Response	Variable	Coefficient	SE	df	*t* value	P
Turbid	Intercept	6.4	0.70	30	9.1	0.000
	Bridge	-2.7	0.99	30	-2.7	**0.012**
	Reference	-3.0	0.99	30	-3.0	**0.005**
	Upstream	-2.1	0.48	30	-4.3	**0.000**
	Bridge X upstream	1.1	0.68	30	1.7	0.106
	Reference X Upstream	2.2	0.68	30	3.2	**0.003**

TABLE A6.—Coefficient estimates from generalized linear mixed models (GLMMs) investigating variation in fish response metrics as a function of physical habitat characteristics. Boldface indicates significance at $\alpha = 0.05$. **NOTE:** SE = standard error, df = degrees of freedom. The intercept of each model identifies the condition of downstream, culverted stream types. Model coefficients relate each variable to the intercept. Individual stream-level effects were accounted for by including it as a random intercept term in each model. Differences in area sampled between stream reaches was accounted for by including an offset (log m^{-2}, excluding richness) in each model.

Response	Variable	Coefficient	SE	*z* value	P
Total density	Intercept	-1.63	4.41	-0.37	0.712
	Wet Width	-0.28	0.14	-1.99	**0.046**
	Depth	-0.44	1.90	-0.23	0.818
	Velocity	0.31	0.88	0.35	0.726
	Fines	-0.02	0.05	-0.39	0.699
	Course Sub.	0.01	0.13	0.09	0.931
Richness	Intercept	0.50	3.28	0.15	0.879
	Wet Width	0.55	0.23	2.34	**0.019**
	Depth	-11.24	1.86	-6.06	**0.000**
	Velocity	0.82	1.34	0.61	0.543
	Fines	-0.07	0.03	-1.95	0.051
	Course Sub.	-0.11	0.09	-1.18	0.238
Slimy Sculpin	Intercept	-0.40	3.04	-0.13	0.895
	Wet Width	0.43	0.21	1.99	**0.046**
	Depth	-9.99	1.76	-5.68	**0.000**
	Velocity	1.12	1.25	0.90	0.370
	Fines	-0.05	0.03	-1.70	0.090
	Course Sub.	-0.09	0.08	-1.06	0.289
Lake Chub	Intercept	-1.88	4.28	-0.44	0.661
	Wet Width	0.85	0.38	2.24	**0.025**
	Depth	-6.32	3.26	-1.94	0.052
	Velocity	-5.11	2.74	-1.86	0.062
	Fines	-0.08	0.04	-1.85	0.064
	Course Sub.	-0.07	0.11	-0.61	0.542
Chrosomus Spp.	Intercept	-8.61	11.18	-0.77	0.442
	Wet Width	0.33	0.54	0.60	0.549

Response	Variable	Coefficient	SE	*z* value	P
	Depth	-6.54	2.77	-2.36	**0.018**
	Velocity	-1.29	2.25	-0.57	0.567
	Fines	0.02	0.11	0.18	0.854
	Course Sub.	0.12	0.32	0.37	0.714
Brook Stickleback	Intercept	-1.99	3.02	-0.66	0.510
	Wet Width	-0.43	0.09	-4.74	**0.000**
	Depth	1.11	0.88	1.27	0.200
	Velocity	-0.16	0.57	-0.29	0.770
	Fines	-0.01	0.03	-0.34	0.730
	Course Sub.	-0.03	0.09	-0.36	0.720
Redside Shiner	Intercept	-9.65	5.64	-1.71	0.087
	Wet Width	2.09	0.91	2.29	0.022
	Depth	-1.99	6.40	-0.31	0.756
	Velocity	-7.30	5.76	-1.27	0.205
	Fines	-0.12	0.07	-1.68	0.094
	Course Sub.	-0.04	0.14	-0.28	0.783
Catostomus Spp.	Intercept	-8.59	4.95	-1.73	0.083
	Wet Width	0.76	0.39	1.92	0.055
	Depth	-1.25	3.81	-0.33	0.742
	Velocity	-2.82	2.72	-1.04	0.300
	Fines	-0.05	0.06	-0.98	0.326
	Course Sub.	0.09	0.14	0.63	0.526
Longnose Dace	Intercept	-15.13	5.01	-3.02	0.003
	Wet Width	2.45	0.77	3.19	**0.001**
	Depth	-5.33	4.61	-1.16	0.248
	Velocity	-7.68	6.50	-1.18	0.237
	Fines	-0.03	0.04	-0.89	0.374
	Course Sub.	0.03	0.11	0.31	0.756
Salmonids	Intercept	-7.91	10.35	-0.76	0.440
	Wet Width	0.53	0.53	1.00	0.320
	Depth	4.17	5.19	0.80	0.420
	Velocity	4.13	3.03	1.36	0.170
	Fines	-0.06	0.11	-0.57	0.570
	Course Sub.	-0.07	0.30	-0.22	0.820

Appendix B.—Supplementary information for Chapter 3

TABLE B1.—Literature review summary of prioritization methods (adapted and expanded from Oldford, 2013)

Prioritization Method	Reference	Region	Focus	Connectivity Measure	Number of barrier	Barrier Types
Scoring and Ranking	Karle (2005)	Alaska, USA	Diadromous	Downstream barriers present	n/a	n/a
	Hicks & Sullivan (2008)	Nova Scotia, CAN	Diadromous	"Presence of upstream barrier" score	268	culverts
	Mader & Maier (2008)	Austria	Migratory	NONE	230	weirs, falls, debris
	Kocovsky et al. (2009)	Susquehanna River Basin, Pennsylvania, USA	Diadromous	Distance from river mouth	20	dams
	Poplar-Jeffers (2009)	West Virginia, USA	Potadromous	NONE	120	culverts
	WDFW (2009)	Washington, USA	Diadromous	NONE	-	
	Lawson (2010)	NE Australia	Diadromous	NONE	5,536	road crossings
	Pini Prato et al. (2011)	Italy	Diadromous	"Continuity Index" (CI; Pini Prato, 2007)	16	small obstacles
	Nunn & Cowz (2012)	England	Diadromous	Likelihood of access (downstream barrier passage)	67	weirs
	Anderson (2012)	SE, USA	Stream-resident	Presence of upstream / downstream barrier	256	culverts
Iterative Scoring and ranking	Taylor & Love (2003)	California, USA	Diadromous	Presence of upstream barrier	n/a	culverts

Prioritization Method	Reference	Region	Focus	Connectivity Measure	Number of barrier	Barrier Types
(stepwise)						
	Cote et al. (2009)	Newfoundland, CAN	Diadromous, Potandromous	DCIp & DCId	15	culverts
	Diebel et al. (2010)	Wisconsin, USA	Stream-resident	C metric	121	culverts, dams
	Mount et al. (2011)	Pacific NW (B.C., CAN)	Diadromous, Potandromous	Number of barriers downstream	> 300,000	culverts
	Bourne et al. (2011)	Newfoundland, CAN	Diadromous, Potandromous	DCI_p & DCI_d	43	culverts
	Branco et al. (2014)	Iberia, Spain	Potadromous, Stream-resident	"Integral Index of Connectivity" (IIC; Pascual-Hortal & Saura, 2006)	29	dams
	Diebel et al. (2014)	Wisconsin, USA	Stream-resident	C metrics (C^{ind}, C^{avg})	190	road crossings
Combinatorial optimization	Kuby et al. (2005)	Pacific NW, USA	Diadromous	Presence of downstream barrier	150	dams
	O'Hanley & Tomberlin (2005)	Pacific NW, USA	Diadromous	Connectivity matrix	289	culverts
	Zheng et al. (2009)	Great Lakes	Migratory	Presence of downstream barrier	139	dams
	O'Hanley (2011)	Wisconsin, USA	Stream-resident	Connectivity matrix	125	culverts, dams
	O'Hanley et al. (2013)	Wisconsin, USA	Stream-resident	C metrics (C^{ind}, C^{avg})	130	culverts, dams
	McKay et al (2013)	Nevada, USA	Diadromous	HCIU index	30	culverts, dams

Prioritization Method	Reference	Region	Focus	Connectivity Measure	Number of barrier	Barrier Types
	Oldford (2013)	Newfoundland, CAN	Diadromous, Potandromous	DCI_p & DCI_d	579	culverts, dams
	King and O'Hanley (2014)	Maine, USA	Diadromous	Connectivity matrix	6,989	culverts, dams

TABLE B2.—Comparison of connectivity (habitat availability) metrics. Modified and expanded from Diebel et al. (2010).

Metric	Name	Application	Reference
Cind	Connectivity metric (individual segment)	Stream-resident	Diebel *et al.*, 2010, 2014
Cavg	Connectivity metric (watershed average)	Stream-resident	Diebel *et al.*, 2010, 2014
DCIp	Dendritic Connectivity Index	Stream-resident	Cote *et al.*, 2009
DCId	Dendritic Connectivity Index	Diadromous	Cote *et al.*, 2009
IIC	Integral Index of Connectivity		Pascual-Hortal & Saura, 2006
HCIU	Habitat connectivity index for upstream passage	Diadromous	McKay *et al.*, 2013
CI	Continuity Index	Diadromous	Pini Prato, 2007