# Dam removal as a tool for restoring fish connectivity – a literature review and field study

By

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**Dedication**

To my boyfriend Andrew Bond, for supporting me in all my life's crazy adventures, whether that be living at a biology station near Kingston, Ontario, or driving across Canada with me to help with fish research. You have been my constant support system and have watched me grow from an undergrad to a graduate student, that is crazy about fish. I can't wait to see where life takes us and I am so happy you have been by my side for every step of this journey. I also dedicate this thesis to my parents for always encouraging me to pursue a career in a field that I love, you have both had a huge influence on me and have shaped the determined and passionate woman I am today.

**Abstract**

Little is known about whether dam removal achieves fish restoration objectives. In Chapter 2, I document the characteristics of dams that have been removed along with the methods and trends in fish response to dam removal. In addition, this chapter provides guidance for those embarking on dam removal projects to improve the evidence base (e.g., reliability, replicability, relevance) so that a systematic review that advances the science will be possible in the future. In Chapter 3, I document the effectiveness of a nature-like fishway in supporting up- and down- stream movement of a threatened salmonid, bull trout (*Salvelinus confluentus*) in Forty Mile Creek, Banff National Park. This chapter explores the biotic and abiotic factors influencing the probability of fish to approach and pass through the fishway as well as passage duration. This information will expand our understanding on system connectivity as a whole by combining both dam removal and fishway research together.

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**Co-Authorship**

**Chapter 2**. Approaches for investigating the effects of dam removal for achieving fish restoration objectives: building the evidence base by improving the science. Brittany Sullivan, Steven Cooke.

While this study is my own, the research was undertaken as part of a collaborative effort with Dr. Cooke. The project was conceived by Sullivan and Cooke. Data were extrapolated by Sullivan. All figure construction and analyses were conducted by Sullivan. Data were interpreted by Sullivan and Cooke. All writing was conducted by Sullivan. Cooke also provided comments and feedback on the manuscript. This manuscript has been prepared for submission to the Journal of Ecological Engineering.

**Chapter 3.** Bull trout (*Salvelinus confluentus*) passage behaviour at a nature-like fishway following a partial dam removal in a national protected area. Brittany Sullivan, Chris Carli, Taylor Ward, Robert Lennox, Mark Taylor, Steven Cooke.

While this study is my own, the research was undertaken as part of a collaborative effort and each co-author played a role in its completion. The project was conceived by Sullivan, Taylor and Cooke. Data were collected by Sullivan, Carli and Ward. The figures were constructed by Sullivan with the assistance of Phil Harrison and Lee Gutowsky. The map of the study area was constructed by Carli. Data interpretation and analysis were conducted by Sullivan with direction from Taylor, Lennox and Cooke. All writing was conducted by Sullivan. All co-authors provided comments and feedback on the manuscript. This manuscript has been prepared for submission to River Research and Applications.

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## Chapter 1: General Introduction

The dam building era was a prominent time beginning in the late 1800's, when riverine ecosystems became altered for water supply, hydropower and flood control (Billington et al., 2005). The construction of dams have negatively affected ecosystems in a number of ways (reviewed in Dugan et al., 2010). For fish in particular, migration delays, habitat loss, and changes in environmental factors (i.e., temperature, flow alterations, water quality) have put a number of populations at risk (e.g., Liermann et al., 2012) especially migratory fishes (e.g., Marschall et al., 2011) that are unable to access historical spawning habitat. Fishways have been designed to provide connections between previously fragmented systems by supporting up- and down-stream movement with varying degrees of success (Roscoe & Hinch, 2010). Our understanding of fishways and how they influence reconnected ecosystems is constantly improving, where both the positive and the unintended consequences of reconnecting these systems have been observed (McLaughlin et al., 2013).

The dam building era has now transitioned towards a period of dam decommissioning. Many aging dams are being considered for removal due to the economic costs of continued maintenance and the associated risks of potential dam failure (Poff & Hart, 2001). Complete dam removal (e.g., Hirethota et al., 2005, Flejstad et al., 2012) or partial breaching (e.g., Maloney et al., 2008, Helms et al., 2011) have both been documented. Complete dam removal leaves the system entirely free of the anthropogenic structure, while partial breaching reconnects the system in the presence of the structure. Partial breaching can occur on purpose where a section of the dam is removed (e.g., Maloney et al., 2008) or because of dam failure (e.g., Helms et al., 2011). Nevertheless, the newly flowing passageways often mimic a nature-like fishway with natural substrate along the stream or river bottom. In this way, partial breaching and nature-like fishways are one and the same, but the connection between the two has not been defined in the literature to date (e.g., Maloney et al., 2008; Raabe & Hightower 2014a,b).

The aim of this thesis is to advance the science behind dam removal around the globe, and the knowledge base for restoring riverine longitudinal connectivity in previously fragmented ecosystems. Chapter 2 focuses on characterizing the current state of dam removal science by exploring the methodologies used and overall trends in fish response for both partial breaching and complete dam removals. There has been a growing number of dam removal studies over the past few decades. By understanding the general trends and methodologies to date, an opportunity to improve the evidence base for future studies will be possible. In Chapter 3, the effectiveness of a nature-like fishway as a result of a partial dam removal is explored. This is done by considering the abiotic and biotic factors that may influence the probability to approach (as a means of assessing fishway permeability), probability of passage and passage duration through the fishway for a threatened resident species, bull trout, *Salvelinus confluentus,* in a small montane stream within Banff National Park, Alberta, Canada. The present introduction is intentionally brief as context and details about dam removal are discussed at length in the literature review provided in Chapter 2.

## Chapter 2: Approaches for investigating the effects of dam removal for achieving restoration objectives: building the evidence base by improving the science


### 2.1 Abstract

Dam removal has become an increasingly popular method for attempting to restore fish populations in fragmented river systems. However, little is known about whether dam removal promotes the achievement of fish restoration objectives. Here we review the characteristics of dams that have been removed, the metrics used and the overall trends in fish response to dam removal. We also share recommendations for the study of future dam removals to improve the evidence base in order to predict with better certainty the effects of dam removal in the future. Our synthesis revealed that most studies included dams that were small in size (≤15-m), focusing on a single dam removal. However, several studies considered multiple dams on a river or in a watershed implying systems-level thinking. It was common for studies to provide less than 1-year of pre-removal monitoring and/or infrequent use of the before-after-control-impact design. A variety of endpoints have been used to assess fish responses to dam removal (e.g., species richness, abundance, density), where upstream community-level responses were often linked to positive outcomes and downstream responses tended to be negative, at least over short timescales. The use of multiple endpoints, appropriate reference sites (when available) and longer pre- and post-removal monitoring is advised. In cases where there is limited baseline or reference conditions, we suggest researchers use data in creative ways (e.g., use of comparative historical records and/or stakeholder knowledge, pooling resources). As the evidence base expands it will be possible to conduct a systematic review and meta-analysis on the effectiveness of dam removal for achieving fish restoration objectives. However, the quality of evidence must be improved (e.g., reliability, replicability, relevance) before this can occur.

## 2.2 Introduction

Humans have been a dominant presence on the landscape for centuries (Vitousek et al. 1997). Indeed, forests have been cleared, roads, buildings, and other infrastructure have been constructed, and watercourses have been altered. One of the most obvious ways in which humans can alter natural watercourses is to construct dams for flood control, hydropower or water abstraction (Poff & Zimmerman, 2010; Elosegi & Sabater, 2013). Dams now feature prominently on small and large watersheds around the globe. There are more than 87,000 dams above 7-m in height in the United States alone (USACE, 2013), with over 58,000 large dams (i.e., >15-m) worldwide (ICOLD, 2015). These structures influence the ecology and hydrology of river systems (Bednarek, 2001; Renofalt et al., 2010) by regulating the transport of sediment, water flow, nutrients and biota (Ligon et al., 1995; Ellis & Jones, 2013). This has transformed watercourses from lotic to lentic systems in upstream impoundments, while depleting downstream waters of nutrients, sediment, and natural water flow (Gregory et al., 2002; Graf, 2006).

Among the negative impacts that dams have on aquatic ecosystems, the most prevalent is the fragmentation associated with impassable barriers (Nilsson et al., 2005). Dams (especially large ones) serve as a direct barrier to fish passage and thus impede biological connectivity (Fullerton et al., 2010). Although some dams are equipped with fish passage facilities, upstream and downstream passage remains imperfect (Bunt et al., 2012). In some cases, fragmentation has led to declines in both resident and migratory species, where foraging, reproduction and colonization movements have been inhibited (Bednarek, 2001; Dudgeon et al., 2006). Moreover, given the numerous ecosystem services provided by freshwater fish populations (Holmlund & Hammer 1999; Lynch et al., 2016) such as nutrient cycling, the impacts of fragmentation have often extended beyond fish to affect riverine ecosystems as well as the riparian zone (Helfield &

Naiman 2001). Nonetheless, it is worth noting that dams have also created some unintended positive consequences where invasive species expansion and the spread of certain pathogens have been controlled (Rahel & Olden, 2008; Hurst et al., 2012; McLaughlin et al., 2013). Recognizing that improved connectivity may support the dynamics that once thrived in such locations, dam removal has often become a "desirable" method for river restoration in the last few decades (Hart & Poff 2002; Bernhardt & Palmer, 2011). However, little is known about whether dam removals can meet fish restoration objectives. There is much debate regarding the consequences of dam removal along with significant need for a rich evidence base to support policy and management (Doyle et al., 2003).

Here we identify (1) the characteristics of dams that have been removed, (2) provide an overview of how fish response to dam removal(s) has been quantified, (3) consider the general trends in fish response in the form of a narrative review and (4) provide guidance on how to improve the evidence base for future dam removals concerned with meeting fish restoration objectives.

**2.3 Methods**

In an attempt to deliver the most transparent methods possible, we have provided a very thorough explanation of how our literature search was conducted so that it can be replicated in the future when additional studies become available. Although we do not attempt to conduct a systematic review, we do adopt some of those principles in an effort to improve the reliability and replicability of the literature review (Haddaway et al., 2015).

### *2.3.1 Key words for primary literature search*

To conduct a primary literature search, a number of key words were selected in specified strings. We used * to define each search word. To avoid overlooking any possible studies we provided a wide range of search strings that in some instances led to repetition. However, this ensured that the greatest number of studies could be accounted for on the topic for fish restoration with dam removal. The search strings were as follows: (1) dam* + removal*, (2) weir* + removal* (3) partial* + dam * + removal* (4) breached* + dam* (5) dam*+ removal* + biological* (6) dam* + removal + fish* (7) dam* + removal* + fish* + passage*, (8) dam* + removal* + fish* + community*, (9) dam* + removal* + restor*, (10) dam* + removal* + ecological*, (6) weir* + removal* + fish*, (11) weir* + removal* + fish* + passage*, (12) weir* + removal* + fish* + community*, (13) weir* + removal* + restor*, (14) weir* + removal* + ecological*, (15) breached* + dam* + fish*, (16) breached* + dam* + fish* + passage*, (17) breached* + dam* + fish* + community*, (18) breached* + dam* + restor*, (19) breached* + dam* + ecological*, (20) river* + restor* + fish*, (21) stream* + restor*+ fish* (22) barrier* + removal* +fish* (23) barrier* + removal* +fish* +passage* (24) barrier* + removal* + fish* + community* (25) barrier* + removal* + ecological* (26) weir* + removal* + biological* (27) breached* + dam* + biological* (28) partial* + dam* + removal* + biological*.

### *2.3.2 Electronic Database selection*

Multiple databases were considered for this review. The databases used were as follows; (1) Google Scholar (2) Scopus (3) Web of Science (4) USGS Dam Removal Science Database (5) DFO Waves Database. Google Scholar, Scopus and Web of Science were used to locate articles in other jurisdictions, where keywords used in search strings were expanded to include English speaking countries such as Australia*, New Zealand* and a number of European countries. Our search was inclusive of all articles that were published up until May 10, 2016 which was when the search was conducted.

### *2.3.3 Primary and Grey Literature Selection for Review*

Articles that were identified based on search strings in the selected databases were imported into a reference management program, Mendeley. Once imported, multiple screening processes were conducted to ensure that the selected articles were appropriate based on the scope of the review. This was done by first screening the articles based on titles to ensure they focused on dam removal and removing articles deemed inappropriate. Second, the abstracts of all the articles were viewed to ensure they explicitly mentioned the evaluation of fish response to dam removal in some form. Finally, there was a review of the articles themselves, which focused specifically on the methods and results section to ensure that fish response to dam removal had been addressed and quantified. If an article was published in grey literature as well as a peer-reviewed publication, the peer-reviewed publication was included in our synthesis.

The search provided 143 studies with empirical data concerned with varying responses to dam removal (e.g., geomorphology, water quality). There were 37 studies that were relevant to fish restoration and dam removal that were included in this review. If the same dam was used for multiple studies, we only included the height of the dam once in our synthesis on dam size.

### *2.3.4 How metrics were identified*

A number of metrics were identified in this synthesis for assessing fish response to dam removal. For our purposes, we identified species abundance as a relative measure of abundance based on sampling effort (i.e., catch-per-unit effort) which was commonly reported as number of individuals captured per electrofishing seconds. In contrast, fish density was identified as an absolute measure of abundance for the area being sampled and was reported as the number of individuals captured per unit area. Fish biomass was also identified as an absolute measure of abundance for the area being sampled and was reported as the mass of individuals captured per unit area (Bradford et al., 2016).

## 2.4 Analysis and Discussion

### *2.4.1 Characteristics of dams that have been removed*

*Publication information*

Of the 37 studies included in our review, 65% (N=24) were peer-reviewed publications, 19% (N=7) were technical reports and 16% (N=6) were theses.

*Geographic location of studies*

Although dam removal for fish restoration (that can be paired with dams that have safety concerns or obsolete in use) is becoming more common in North America, it is still a relatively new concept in other regions. We found that 92% (N=34) of the dam removals included in our analyses were from North America, while 5% (N=2) were from Europe, and an additional 3% (N=1) were from Asia. Primary and grey literature on fish response to dam removal at the international scale is still quite rare; for example, there has only been one peer-reviewed study published on this subject in Norway (Fjeldstad et al., 2012). Since we limited our search to English it is likely that some international grey literature sources were excluded from the study.

*Temporal range of studies*

The temporal range of the studies is highly focused within the last 20 years, as dam removal became an issue of concern in the late 1980's (Maloney et al., 2008; O'Connor et al., 2015; Bellmore et al., 2016). The first study on fish restoration with dam removal was published in 1994 (Hill et al.,1994) while the most recent study (included in this review) was published in 2016 (Magillian et al., 2016; see Figure 2.5.1). Although statistical analysis of the trend is unwarranted, there appears to be growing research activity on this topic. For example, the studies published between 2010 to 2015 alone have accounted for 60% (N=22) of the total number of publications included in this review.

*Relative size and number of dams considered for removal*

Out of the 37 studies used in our synthesis, 65% (N=24) studies completely (or partially) (3%, (N=1)) removed one dam, there were 19% (N=7) of studies that removed two dams, 3% (N=1) removed three dams, and 11% (N=4) removed four or more dams (including partially removed/relict dams). This was done to restore connectivity in highly fragmented river systems (e.g., Catalano et al., 2007, Raabe & Hightower, 2014a). Out of the 24 studies that identified the height of the structures, a total of 33 dams were removed, the majority were ≤15-m in height 97% (N=32), with only one dam (3%, (N=1)) >15-m in height.

*Purpose of dam removal*

Of the 30 studies that provided reasoning for dam removal, 47% (N=20) considered fish restoration as a main objective for dam decommissioning, 32% (N=14) of studies considered dams that were obsolete in use, 16% (N=7) considered structural and safety concerns, and 5% (N=2) were due to public demand (i.e., social pressures from grassroots organizations, NGOs). Over half of the studies (67%, (N=20)) provided one reason for barrier removal, while 33% (N=10) provided two or more reasons, which were primarily a combination of fish restoration and dams that were obsolete in use (40%, (N=4)) or fish restoration and safety concerns (30%, (N= 3)).

A review conducted by Bellmore et al., (2016) stated that over 1200 dams have been removed in the United States alone, with only 9% of them being accompanied by published scientific studies (which have spanned topics including sediment transport, water quality, and biota). The small number of scientific studies that have accompanied dam removal(s) have likely influenced our results as we only accounted for dam removals that were coupled with scientific studies. In addition, we focused on dam removals where "fish restoration" was considered and consequently excluded scientific studies that solely looked at hydrologic and geomorphic responses (to name a few).

It is also likely that scientific studies on dam removal have been paired with dams that were already supposed to be removed for other reasons, but failed to identify as such. This is probable given that out of all 58 dam removals included in this synthesis, the majority of dam removals provided only ≤ 1-year pre-removal monitoring (40%, (N=23)), or none at all (26%, (N=15)) (see Figure 2.5.2). A single year of baseline data has little power to statistically detect changes from the natural variability of a system (i.e., influences from hydrology, climate and/or stochastic events; Kibler et al., 2011). However, there are instances where certain end points are only worthy of study after dam removal. For example, consider a scenario in which a dam was blocking all upstream passage of a diadromous species, such as a Pacific salmonid. Abundance/presence of that species immediately prior to dam removal would be "zero/absent" such that it would only be necessary to monitor re-establishment after removal. In that case, appropriate baselines may be from periods prior to dam construction.

There were many dam removal projects that lacked appropriate reference sites, with only 22% (N=13) of dam removals using the Before-After-Control-Impact (BACI) design. If long-term pre-removal monitoring on a system is possible, the number of pre-removal monitoring years will largely depend on the objectives of the study. For example, at least 3-yrs of baseline data is recommended for fish restoration, especially when funding is limited (see Smokorowski et al., 2017). In contrast, an ecosystem approach to dam removal will require far more pre-removal monitoring years to capture the natural variability in the system (e.g., riparian zone, invertebrates, water chemistry), which has been shown in the Hubbard Brook experiments (see Likens & Busso, 2006; Holmes & Likens, 2016). In this synthesis, the number of post-removal monitoring years has largely been short-term and has ranged from 1-yr (24%, (N=14)), 2-yrs (29%, (N=17)) or 3-yrs (28%, (N=16)). Since dam removal is not a gentle process and can act as an initial disturbance on the system, it is important to recognize the need for more long-term studies on dam removal that address the objectives of the study (e.g., fish restoration or ecosystem approach) with the appropriate monitoring timescales.

We also recognize that in some instances it may be difficult to find appropriate reference locations (e.g., lack of appropriate habitat). In these circumstances, researchers should evaluate the strengths and weaknesses of alternative experimental designs (see Kibler et al., 2011) and select the most appropriate design based on the inferences they intend to make on the population. Alternatively, researchers can use data in creative ways to provide a more balanced study design (see Section 2.4.4) or combine dam removal with other areas of research (e.g., nature-like fishways) to advance the science in other ways.

***2.4.2 How fish response has been quantified for dam removals***

*Characteristics of Species Considered*

The majority of studies considered in our review focused on community level response to dam removal (54%, (N=20)), or community level response with a species of interest (11%, N=4)). The remaining studies (35%, (N=13)) failed to recognize the community and considered a single (or multiple) species of interest, this subset of studies largely focused on diadromous species (N=11), but riverine (N=2) species were also noted.

*Summary of metrics considered*

The metrics used to quantify fish response to dam removal (partial or full dam removal) were dependent on the species of interest (and/or community at large). The majority of studies included species composition (and/or a shift in species assemblage) (22%, (N=15)), while 16% (N=11) considered species abundance, 12% (N=8) considered species richness, 7% (N=5) considered fish biomass, 7% (N=5) considered fish density, while 6% (N=4) considered species diversity. A number of studies included metrics related to reproduction or recruitment (28%, (N=19)), with one study that focused on movement patterns in response to dam removal (outside of spawning season) (1%, (N=1)) (see Chen, 2012).

The metrics used to quantify fish response to dam removal have been applied to communities (or species of interest) at varying frequencies. We recognize that in most instances, populations are the fundamental unit that matter to managers. We suggest that future studies include as many metrics as possible on the same population. If applied, we will be able to identify which metric(s) are best at detecting population level effects when a larger evidence base becomes available in the future. This is also important as some metrics can be used to offset possible misinterpretation of results. For example, fish density can be paired with reproduction and recruitment metrics to assess population viability. This ensures that possible aggregation effects (described by fish density) are distinguished from population level increases (with reproduction and recruitment metrics; Bernhardt & Palmer, 2011). By recognizing these intricacies and the importance of using a wide range of metrics, a clear understanding of fish response to dam removal (that can be applied across studies) will be possible in the years to come.

### *2.4.3 General trends*

*Synthesis on upstream fish response*

There were 4 studies (with 4 dam removals) that looked at a single (N=3) or multiple (N=1) species upstream response, but failed to include community level responses. Most of these studies were concerned with diadromous species (Raabe & Hightower, 2014a) with only one study that looked at riverine species (e.g., smallmouth bass) of recreational significance (Kanehl & Lyons, 1997). Species abundance was shown to increase in upstream waters for all 3 dam removals it was quantified in (100%, (N=3)) and fish biomass was shown to increase upstream for the one dam removal it was considered (Kanehl & Lyons, 1997). In addition, there was one study that focused on movement patterns outside of spawning season for the Taiwan Salmon. This was done to assess connectivity up- and down- stream during draw-down and following dam removal. It was found that this species could indeed access upstream waters and would undergo large-scale movements motivated by translocation following dam removal (see Chen, 2012).

There were 23 studies that considered upstream community level response to dam removal, in which 34 dam removals were examined. We analyzed fish response to each dam removal separately, so that dam specific responses could be quantified. For the 8 dam removals that considered relative abundance, 75% (N=6) saw an increase in relative abundance, 13% (N=1) noted a decrease, which was likely due to further upstream fragmentation and flooding that occurred during monitoring (see Magilligan et al., 2016), while 13% (N=1) remained unchanged. For species richness, 9 dam removals were considered, which found that 67% (N=6) saw an increase in species richness, 11% (N=1) saw a decrease (see Magilligan et al., 2016), and 22% (N=2) remained unchanged (i.e., same study with multiple dam removals). For species diversity, 4 dam removals were considered and 75% (N=3) saw an increase in diversity, while

25% (N=1) remained unchanged. When considering upstream community level response for fish density, 4 dam removals were considered and found that 75% (N=3) saw an increase, 25% (N=1) saw a decrease. Out of the 4 dam removals that looked at fish biomass, it was found that 75% (N=3) saw an increase and 25% (N=1) saw a decrease. For changes in species composition (or assemblage shifts), 25 dam removals were considered. The majority saw a shift from lentic to lotic species (36%, (N=9)), or an overall increase in natives (tolerant and/or intolerant) (32%, (N=8)), 8% (N=2) saw in increase in invasive or non-native species that moved upstream from downstream waters, while 24% (N=6) remained unchanged.

There is a general consensus that upstream community level responses to dam removal have positive outcomes in which biodiversity of previously isolated reaches can be restored (Bushaw-Newton et al., 2002; Greathouse et al., 2006). Our synthesis found that fish communities generally improved following dam removal, with the exception of two instances where invasive/non-native species were found in upstream waters (see Gottgens et al., 2009; Copper, 2013). Mclaughlin et al., (2013) considered the unintended consequences of restoring connectivity to upstream reaches. The authors highlighted that restoring connectivity to previously isolated fish communities can cause unwanted predator-prey interactions, introgression between wild and hatchery fish, hybridization with introduced species, and potentially facilitate the spread of disease (Kiffney et al., 2009; Marks et al., 2010). Evidence of this has been largely limited to fishways. This is not surprising given the long history associated with fishway evaluation (Schwalme et al., 1985; Bunt et al., 2001; Roscoe et al., 2011). In contrast, the concept of dam removal is still relatively new and has had much less time to mature (Bellmore et al., 2016). This may explain why upstream community level response to dam removal have generally provided positive outcomes, with limited evidence to support the potential negative effects.

*Synthesis on downstream fish response*

There have been 3 studies (with 3 dam removals) that have looked at either a single (N=2) or multiple (N=1) species downstream response, but failed to include community level responses. All three studies considered relative abundance, one study noted an increase (N=1), one study noted an initial decline and then an increase (N=1), and the third study saw no clear trend in native species abundance (i.e., smallmouth bass), with a gradual decline in a non-native species abundance (i.e., common carp; N=1; Kanehl & Lyons, 1997).

There were 16 studies that considered downstream community level response to dam removal, in which 25 dam removals were examined. A subset of dam removals (N=7), only considered upstream community level response (monitoring impounded reaches and reference sites) or did not provide clear indication of downstream responses and so were excluded from this section (e.g., Chatham et al., 2007). We analyzed fish response to each dam removal separately, so that dam specific responses could be quantified. For the 8 dam removals that considered species abundance, 38% (N=3) saw a decrease, 25% (N=2) saw a decrease and then increase, 25% (N=2) remained unchanged, with one instance that followed species specific trends (13%, (N=1)). For species richness, a total of 9 dam removals were considered, 11% (N=1) saw an increase, 33% (N=3) saw an initial decline but then began to recover within one to two years (see Catalano et al., 2007), 44% (N=4) saw a steady decline, and one instance remained unchanged (11%, (N=1)). For fish density, 4 dam removals were considered, 75% (N=3) saw a decline, and there was one case that predator and prey densities were inversely correlated (which was consistent with pre-removal data) (25%, (N=1)). There were two instances that examined downstream response for fish biomass, where a decline was always seen (100%, (N=2)). For species composition (and/or assemblage shifts), 17 dam removals were considered, 24% (N=4) saw a decline in natives (including both tolerant and/or intolerant), an increase in non-natives or invasive species (12%, (N=2)), or did not see a clear transition within the specified monitoring period (65%, (N=11)). The increase in non-native or invasive species downstream was generally associated with impoundment species moving into downstream waters following dam removal.

O'Connor et al., (2015) stated that the physical properties of river channels are likely to stabilize within months to years rather than decades. Our synthesis found that downstream fish populations generally follow patterns of decline within at least 3-yrs of post-removal monitoring. It is likely that improvements in fish populations (e.g., function, structure) will take longer than the physical recovery of the system itself, given that the fish community will have to withstand the complex changes associated with sediment mobilization before channel stabilization occurs (Stanley & Doyle, 2002; Stanley et al., 2002). These changes however could be minimal depending on the amount of sediment released and how the system itself was re-channeled (see Chapter 3, Section 3.2 in this thesis). Researchers must recognize the importance of long-term monitoring programs as they are crucial to fully quantify fish response(s) to dam removal. We also recognize the importance of integrating different areas of research by taking an ecosystem approach to dam removal (e.g., geomorphology, water quality.) to get a better understanding of system-level responses and how they relate to fish.

*Synthesis on fish response in terms of reproduction and recruitment*

Of the 19 studies that examined reproduction and/or recruitment, 26 dam removals were considered, that focused on diadromous (N=12) or riverine (N=7) species. For riverine species, there were no dam removals that incorporated nesting success, and only one dam removal that saw an increase in upstream spawning activity (N=1). In contrast, there were 7 dam removals that considered changes in fish size structure upstream of the former dam site as evidence of recruitment into the system. One of these dam removal projects saw an increase in larval and egg densities (14%, (N=1)), 71% (N=5) saw an increase in juveniles and one instance saw an increase in multiple size classes (14%, (N=1)).

For diadromous species, there were 13 dam removals that considered spawning location, where 69% (N=9) found that spawning was primarily documented upstream of the former dam site, 15% (N=2) found that spawning occurred primarily downstream and 15% (N=2) were uniformly distributed up- and down- stream of the former dam site when compared to the distribution before dam removal occurred. There were 13 dam removals that considered nesting success, 69% (N=9) found nesting success to increase upstream, while 15% (N=2) saw an increase downstream and 15% (N=2) saw nesting success was comparable for both up- and down- stream waters. For changes in fish size structure, 10 dam removals were considered, 20% (N=2) noted an increase in the egg survival rate upstream, 20% (N=2) saw an increase of larvae upstream and downstream of the dam site, 60% (N=6) saw an increase in juveniles upstream (with one instance occurring downstream) of the former dam site. Out of the 5 dam removals that considered upstream migration timing, 80% (N=4) found that migrations started earlier (e.g., 1-mth) or were faster (20%, (N=1)). One study (encompassing 3 completely removed dams and one partially removed dam) found that male American shad *Alosa sapidissima* immigrated earlier than females but the study lacked reference data to directly associate these findings with dam removal (N=3). An additional study by Hogg et al., (2013), found that recolonization success to historical spawning grounds took 6-days in the first year of post-removal monitoring and only 3-days in the second year of monitoring. The authors suggested that this was likely a positive feedback in which larval recruitment upstream of the dam attracted adults earlier in the second year of post-removal monitoring with conspecific pheromone cues (N=1).

There has been a growing recognition of the importance of system connectivity to support the extensive movement patterns of riverine and diadomrous fishes, given that their population persistence often depends on access to upstream habitats (Bednarek, 2001; Gillenwater et al., 2006). Here, we show the value of restoring system connectivity for fishes that require more suitable spawning grounds or rearing habitats (especially for diadromous species). However, the concept of ecological traps should also be considered. Ecological traps occur when attractive environmental cues entice an organism to choose habitat where their fitness levels will likely be lower (Mclaughlin et al., 2013). When a dam removal has occurred, the complex geomorphic changes downstream of the former dam may encourage fish to move upstream, even when downstream spawning habitat might have been more suitable. In recognition of this, researchers should see the importance of conducting habitat assessments and identifying ways in which possible ecological traps can be offset if dam removal is likely to happen regardless of the environmental costs.

*Trends in Fish Response for Partially Removed and Relict Dams*

There were 4 studies in our synthesis that monitored one or more partially removed dams that were either taken down on purpose or because of dam failure/degradation of the structures. Raabe & Hightower (2014b) found that passage time was longer (i.e., delayed) for individuals migrating through the partially removed dam in comparison to 3 completely removed dams and overall passage success varied by species. Helms et al., (2011) found that species richness was generally lower upstream of partially removed dams in comparison to downstream waters, while Maloney et al., (2008) found that species richness and fish density tended to decline in downstream waters when pre-removal data were available. Maloney et al., (2008) also found that species composition became similar up- and down- stream of the partially removed dam following it's removal. Raabe & Hightower (2014a), looked at a partially removed dam in comparison to three completely removed dams and considered migration timing of American shad between sexes but failed to associate their migrations with the appropriate reference or control reach to differentiate if the timing of their spawning migrations was influenced by barrier removal or the natural biology of the species (Raabe & Hightower 2014a).

The small sample size of partially breached dams in our synthesis makes it difficult to identify how these structures influence fish populations. It appears that the presence of a structure in some form may limit connectivity (see Helms et al., 2011; Raabe & Hightower, 2014b), but to what extent is unclear. In some instances, partial dam removals and fishways can be seen as one and the same. This is because the habitat features at the former dam site may limit when species are able to pass through the newly connected reach. This can include but is not limited to, high flow, thermal barriers or the relict of the structure itself (Katapodis, 2012). Mclaughlin et al., (2013) have discussed the consequences for limited fishway use which can be applied to dam removal research, such as passage delay, selective passage and fallback behaviour. Fallback behaviour may be of concern if the species changes direction during and/or following their passage at the former dam site (Bjornn et al., 2000). This can largely be due to disorientation, or if they are physically unable to continue their migration based on the strain put on them from passing through a newly restored dam site. Selective passage can also affect restoration success if only certain species or life stages are able to freely move back and forth through the newly re-connected waterway (Bunt et al., 2012; Noonan et al. 2012). Researchers should identify the relative size of the breach that has been made, and when pre-removal data is not available, efforts should be focused towards understanding passage success as a measure of system connectivity. In addition, these same principles can be applied to full dam removal projects to understand any further limitations to fish movement. This is especially important in cases where dam removal has occurred in an otherwise highly fragmented system (e.g., Magilligan et al., 2016).

### *2.4.4 Practical Considerations for Study Design*

The approaches for quantifying dam removal need improvement, and the trends in fish response may not be fully representative of the possible consequences of dam removal. We also recognize the realities of dam removal projects, especially those on smaller systems (e.g., streams), where dam removal may occur rapidly with little opportunity for monitoring. However, given that dam removal is often contentious and involves extensive stakeholder consultation, there is usually some knowledge of the potential for dam removal for years before it actually occurs. The exact timing may not be known and the project may proceed, but that should not preclude the collection of baseline data on fish. In reality, it seems to be rather common where researchers have to scramble to develop and implement a monitoring plan with little opportunity for pre-removal monitoring or evaluation of reference conditions in adjacent systems. In such instances, there is a need for creativity. One method is to identify where data already exists (i.e., long term monitoring programs) that could provide baseline information on the population (e.g., community structure, biomass, presence/absence) or appropriate reference conditions (e.g., data available from a separate watershed). Many natural resource management agencies conduct routine monitoring and in some instances these could align, such that they are relevant to dam removal monitoring. In doing so, researchers will be able to extract the necessary information to help shape a study design that asks critical questions pertaining to the information that is already available on the system.

We also recognize that funding opportunities may not always be available to satisfy site specific needs. Indeed, an obvious question is who pays for dam removal monitoring. The answer will of course be context specific but given the role of natural resource management agencies in stewardship of public resources, they are an obvious starting point. In instances where the dam owner is a utility or other entity with financial resources or where there are legal obligations to support monitoring, funding could be provided by those sources. However, there will still be many lost opportunities where the resources do not exist to conduct high-quality monitoring with the potential to deepen the evidence base. As such, we encourage funding bodies to pool resources together that provide long-term funding opportunities for developing appropriate study designs or monitoring programs in watersheds with aging dams that are likely to be decommissioned in the coming years. This would enable researchers to pull from this common pot of resources to better address questions related to dam removal or access information that is pre-emptively being collected to support the replicability, reliability and relevance that is needed for dam removal science. Waiting until a project is given “final approval” for dam removal is often far too late to mount an effective monitoring program using, for example, a BACI design. In cases where the dam has already been removed, and pre-removal data cannot be accessed (limited or unavailable), researchers may choose to advance the evidence base in other ways, this includes fishway research (see Chapter 3), where the concepts of dam removal and nature-like fishways can be combined to advance the science in other ways.

### *2.4.5 Conclusion*

The science supporting fish restoration using dam removals has been growing over the last two decades, and will likely continue to do so in the years to come (O'Connor et al., 2015). Although we are starting to learn more about the outcomes of dam removal studies, many gaps in knowledge remain. Moreover, the literature is still sufficiently sparse that a systematic-review with a meta-analysis is not possible. Of particular concern, is the fact that most studies conducted to date would be screened out during the critical appraisal phase of a systematic review because the studies lack the design to properly test whether dam removal is achieving fish restoration objectives. The intent of this review is to provide the basis for more critical decision-making in terms of allocating resources that focus on well-defined research and monitoring efforts over long-term timescales (see Table 2.6.1). More importantly, it should provide an understanding as to what areas should be examined further based on the knowledge gaps that currently exist. Since many dams are reaching the end of their lifespan, well-informed decisions related to dam decommissioning must be made (Doyle et al., 2003).

Our search would not have detected instances where there had been no formal reporting (e.g., technical reports, peer-reviewed papers) for dam removal or instances where dam removal occurred but there was no fish-related monitoring. As such, we offer a plea for those engaging in dam removals to share their findings with the broader community through case study reports in journals. Moreover, given the dramatic influence of dams on fish populations we submit that all dam removal studies should include as many fish-related endpoints as possible in their monitoring irrespective as to whether the dam removal had specific fish restoration goals (or using an ecosystem approach when possible). This will also provide the basis of assessing which metrics are best suited in identifying fish-related population level effects in the future.

With the direction provided here along with a commitment to strengthening the evidence base on dam removal, within the next decade it should be possible to conduct a systematic review where there is substantial, high-quality evidence to determine the extent to which such interventions are warranted. For large-scale dam decommissioning we are starting to see this, with many pre-removal monitoring studies being published (e.g., Woodward et al., 2008; Winans et al., 2017) and follow-up studies still to come. Failure to conduct studies that generate reliable and relevant data on the effectiveness of dam removal for achieving fish restoration objectives represents a lost opportunity and one that will require practitioners to continue to rely on narrative syntheses or to selectively pick/avoid individual empirical studies which is inconsistent with best practices for evidence-based conservation and environmental management (Sutherland et al. 2004).

## 2.5 Figures

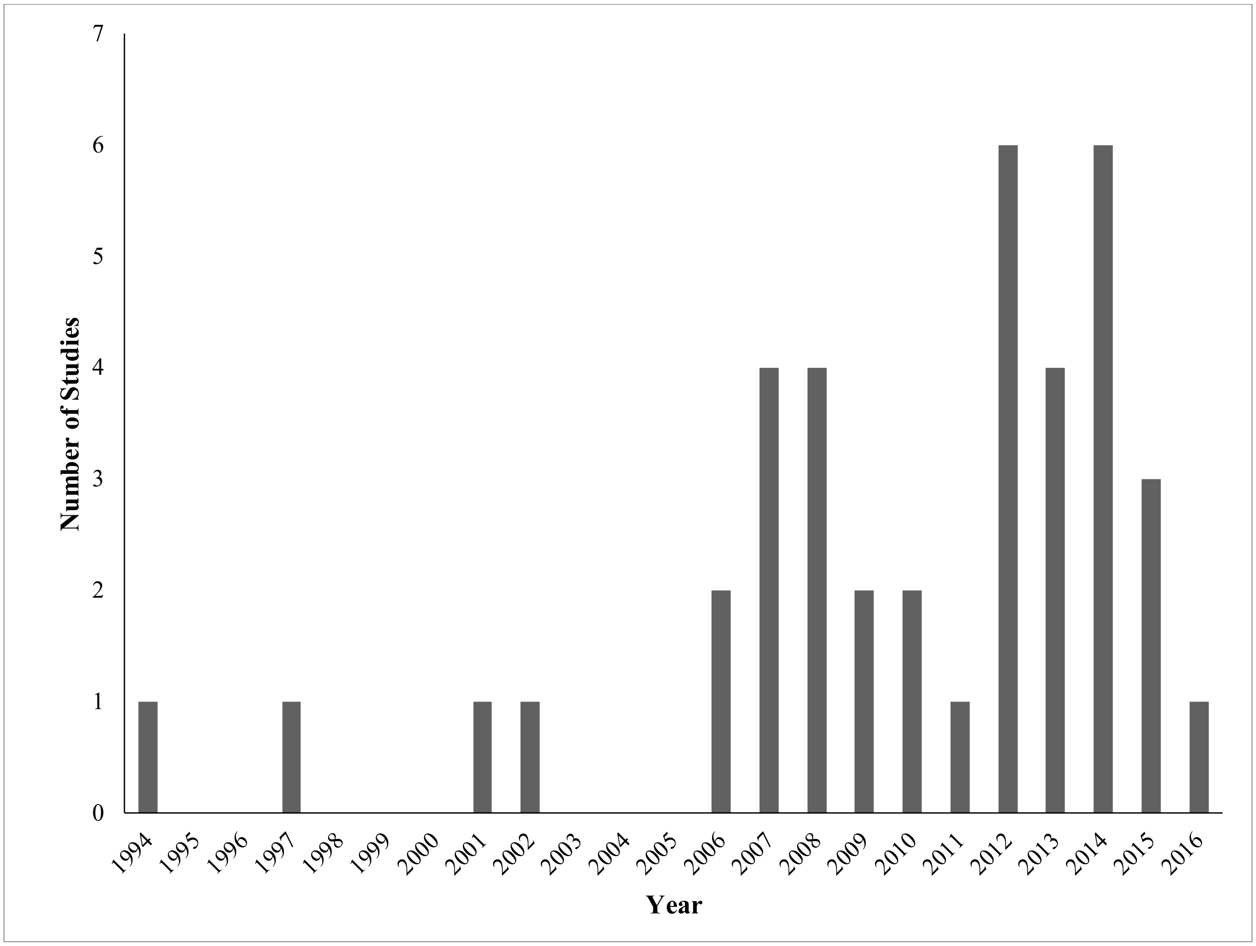


**Figure 2.5.1** Temporal patterns of dam removal studies that have focused on fish restoration objectives that were identified through our structured literature search. The search was conducted on May 10, 2016 such that the number for 2016 should be assumed to be incomplete.

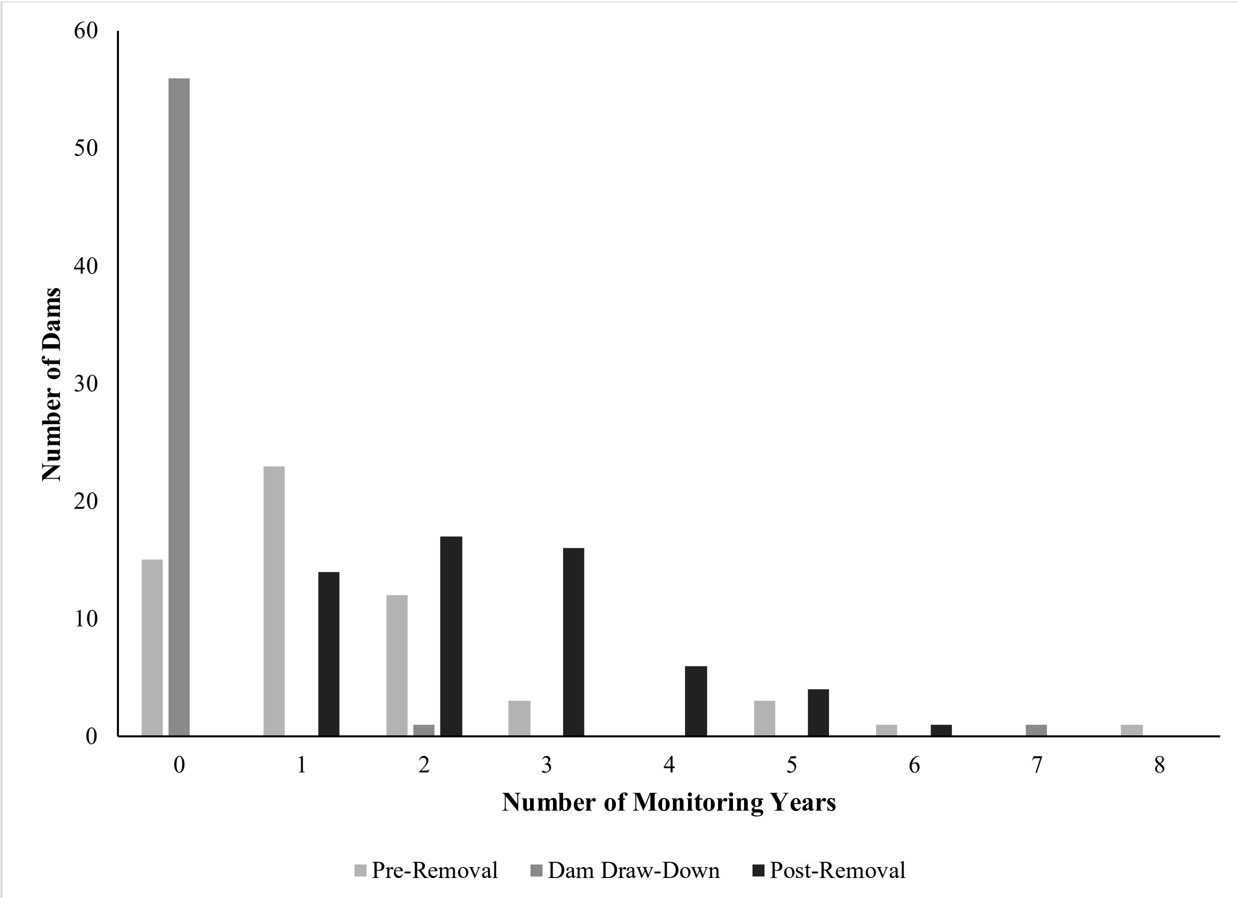


**Figure 2.5.2** The length of pre-, during, and post-removal monitoring for dam removal projects that have been concerned with meeting fish restoration objectives. The search was conducted on May 10, 2016 such that the number for 2016 should be assumed to be incomplete. All monitoring that was more than zero but less than a year was categorized under “1-yr” for simplicity.

## 2.6 Tables

**Table 2.6.1** Key considerations for future dam removal studies where fish restoration objectives are being considered

| **Considerations for Future Dam Removal Studies:** |
| --- |
| • What threats are present within a given system (i.e., invasive species, parasite introduction)?<br>• Will these threats hinder restoration success if system connectivity is restored?<br>• How fragmented is the system and will barrier removal provide access to more suitable fish habitat?<br>• What species (if any) are at risk within the system and are they likely to recover if system connectivity is restored? The natural history (i.e., spawning migrations etc.) of a species must be considered here.<br>• What is the state of downstream habitat prior to barrier removal, is this critical habitat for native fish? If this habitat is degraded following barrier removal, will it likely threaten native fish populations?<br>• What is the most appropriate experimental design based on restoration goals? Is there adequate time and funding to carry this out?<br>• Is it possible to identify an appropriate reference site? Is there access to pre-dam construction data to understand fish populations and community characteristics before the system was fragmented?<br>• How long should monitoring occur to understand if restoration goals have been met? (short term=passage success, long term recolonization, especially in downstream habitats).<br>• What additional measures can be taken to help meet restoration goals? |

## Chapter 3: Bull trout (*Salvelinus confluentus*) passage behaviour at a nature-like fishway following a partial dam removal in a national protected area

### 3.1 Abstract

Dams represent one of the major forms of river alteration. In recent years, many of these structures are reaching the end of their lifespan, where there has been need to consider either extensive refurbishments or dam removal. The partial removal of a small-scale water supply dam in Banff National Park (Alberta, Canada) created a nature-like fishway. This provided the opportunity to investigate probability to approach, probability of passage and passage duration of a threatened species, bull trout (*Salvelinus confluentus*) through a nature-like fishway. Using radio telemetry, we determined that the probability for a fish to approach the fishway was low (37%) and size dependent, but for those that approached, their probability to pass was high, with a passage efficiency of 78%. Passage success was related to water depth and time of day. Fish were likely to pass at high water levels (>0.40-m) in the late spring to summer months in this system. Although some passage events occurred during day-light, the probability to pass the fishway was significantly higher at night. Passage duration ranged from 5-min to 13-days, suggesting that this resident species could have used the fishway for a variety of purposes (e.g., foraging, cover) and not just transiting. Some individuals underwent large-scale movements 2-km upstream (N=11) or downstream (N=2) of the nature-like fishway following a successful passage event. This study provides new insights on how partial dam removals and nature-like fishways can be combined to expand the knowledge base on fishway permeability for newly restored ecosystems.

## 3.2 Introduction

Human-made structures (e.g. dams, water mills, water diversion facilities) have influenced stream connectivity for centuries. It has only been in the last few decades that their negative ecological and environmental effects have been recognized (e.g., Ligon et al., 1995; Rosenburg et al., 1997; Vörösmarty et al. 2010). Of particular concern are migrating fishes (including both potamodromous and diadromous species) that may be limited or have lost complete connection to upstream waters, often associated with critical spawning or rearing habitats (Peter, 1998; Lucas & Baras, 2000). Longitudinal connectivity in fluvial ecosystems is regarded as important for gene transfer, nutrient cycling, and population persistence (Wiens 2002; Pringle 2003). As such, contemporary perspectives on river restoration typically call for efforts to re-establish or enhance ecological connectivity in fragmented systems (Jansson et al. 2007; Cooke et al. 2012).

Negative effects arising from river fragmentation have been mitigated through strategies such as dam removal (ranging from partial removal to full removal) and construction of different types of fishways. Fishways have been used in various forms for decades (See Clay, 1961; Katopodis & Williams 2012) and range in appearance from highly engineered (e.g., Denil or vertical slot fishways) structures to designs that are meant to more closely mimic natural channels (i.e., nature-like fishways; Katapodis et al., 2001). Passage success rates are influenced by fish physiology (e.g., swimming capacities), species life stage and other biotic and/or abiotic factors (e.g., Mallen-Cooper & Stuart, 2007; Bunt et al., 2012). Complete dam removal has become more common in the last few decades, especially for aging structures that pose a liability (i.e., dam failure), or impractical costs for continued dam maintenance (Poff & Hartt, 2002). This process requires substantial efforts to not only remove the structure but also restore the system itself (e.g., Hartt et al., 2002; Stanley & Doyle, 2002). In some instances, complete dam removal is not possible (e.g., limited funding or higher environmental risk) and so partial dam removal is considered, whereby a section of the dam is removed to enhance connectivity. Partial dam removal can be seen as a nature-like fishway by providing up- and down- stream access to resident and/or migrating individuals, but the connection between the two concepts has not been made in the literature so far (e.g., Maloney et al., 2008; Raabe & Hightower 2014a,b). This connection is important to consider when there is limited to no pre-removal or reference data on the system such that understanding dam removal as the "intervention" is not possible (for which this Chapter is a case study for).

For the purpose of this study, we focus on Forty Mile Creek Dam in Banff National Park, Alberta, Canada. This is an example of a nature-like fishway that was created as the result of a partial dam removal. For this case study, we focus on fishway permeability rather than fish response to dam removal given that there was insufficient pre-removal data on the system to understand dam removal as the "intervention". The Forty Mile Creek Dam was built in several stages starting in the early 1900s as a source of the town's drinking water and for fire protection. However, in the mid-1980s the dam ceased to have a function after deep-water wells were drilled in the area. The Town of Banff expressed interest in removing the dam as it was rendered a liability to the town and had fundamental costs associated with continued dam maintenance (e.g., routine inspections, maintaining road access to the dam). The dam stood without purpose until 2014 when the access road was demolished during a 100-yr flood. Managers wanted to invest funds to demolishing the dam instead of re-building the access road. However, the budget was estimated to not be enough to demolish the full concrete structure. Furthermore, removing the entire dam would require fluming (i.e., redirecting) the entire creek while simultaneously demolishing 8-m high and 2-m thick concrete walls. By constructing a nature-like fishway beside the remaining structure it allowed an existing bypass channel within the dam to act as a flume managing the water while the breach was created. The breach was therefore created in a section beside the bypass channel without water passing through, which mitigated the need for complicated sediment control measures.

Bull trout, *Salvelinus confluentus*, are the species of interest for this study, which have been protected under the Alberta Wildlife Act (Alberta Sustainable Resource Development 2015) and more recently have been assessed as “threatened” by Canada’s Committee on the Status of Endangered Wildlife in Canada (COSEWIC, 2012). In Alberta, 57% of bull trout populations are declining, with the Bow River Watershed having experienced the greatest declines due to habitat fragmentation as well as an increase in the cumulative effects of industrial and recreational activities (COSEWIC, 2012; Alberta Sustainable Resource Development, 2015).

The objectives of this study were to (1) assess the probability to approach, the probability of passage and passage duration of bull trout through the nature-like fishway, (2) identify the biotic and abiotic factors influencing the probability to approach (as a means of assessing fishway permeability), probability of passage and passage duration, and (3) determine the distance travelled by a fish following a passage event (i.e., 2-km, 6-km up- or down- stream of the fishway). If the fishway is deemed permeable, it will likely benefit bull trout by reducing habitat fragmentation and thus restoring connectivity to the system.

## 3.3 Methods

### *3.3.1 Study Site*

This study was conducted in Forty Mile Creek in Banff National Park, Alberta at the site of a nature-like fishway (50° 07'N; 96° 01'W). Forty Mile Creek is groundwater fed and flows into the Bow River approximately 6-km downstream of the nature-like fishway. The fishway is 50-m in length, with an average width of 8-m. The upstream entrance is characterized by a concrete apron (8.5-m × 10.4-m), which was the foundation for the dam. This was left to ensure the integrity of the remaining structure. Eleven evenly spaced baffles (0.38-m width, 3.8-m length, 0.20-m spacing) were added to the concrete apron to disrupt high flow. The remaining length of the fishway was characterized by natural rock formations (e.g., cobble, boulders) that ranged from 2.5-cm to 71.3-cm in diameter (from intermediate axis), with boulder spacing that ranged from 1.8-m to 7.8-m formed from natural flow in the system (not engineered). The slope of the natural stream channel directly upstream of the fishway was 1.3%, the slope within the interior of the fishway (between the up- and down- stream fishway entrance) was 5.3% and the slope of the stream channel directly downstream of the fishway was 4.9% (see Figure 3.6.1).

### *3.3.2 Experimental Design*

This study was conducted over one year between the fall season of 2015 (October 27, 2015- Nov. 14, 2015) and into 2016 (March 14, 2015 – Oct. 30, 2016). All fish were captured with a pulsed DC backpack electrofisher (Smith Root, Vancouver, WA). Once caught, fish were temporarily held in a stream-side holding facility (diameter = 243-cm, depth= 90-cm and volume= 2839-L) supplied with ambient fresh water. Fish were anesthetized with clove oil (1 part clove oil to 10 parts ETOH) and then were measured, weighed and transferred to a v-shaped surgery trough in the supine position so that they could be implanted with a uniquely-coded radio transmitter (Sigma-Eight Inc., Markham, ON; 1.5V, 84 dB, 150-mHz). Fresh water was continuously pumped across their gills, maintained with a water pump. Transmitters were inserted into the body cavity through a 10-mm incision made on the ventral body surface of the fish, posterior to the girdle, using a scalpel (number 3 blade, rounded cutting point). The incision was closed with two simple interrupted sutures (PDS II, 3/0, Ethicon Inc). Fish were then returned to a recovery tank before their release. Since our target species, resident bull trout, are known to hybridize with non-native brook trout (*Salvelinus fontinalis*), it is possible that some of our tagged fish were hybrids (see Popowich et al., 2011). All transmitters were programmed to turn off during the winter (Nov. 15, 2015 – March 13, 2016) following the fall 2015 field season (Oct. 27, 2015 – Nov. 14, 2015). This was done as a method of conserving battery life and because large-scale movements were not expected in the winter upon the development of frazzle ice and based on previous studies of overwintering salmonid biology (Jakober et al., 2000; Muhlfeld et al., 2003).

Our study was conducted within a 12-km reach of Forty Mile Creek, using six fixed radio-telemetry receiver stations (Figure 3.6.2). Each fixed receiver station included one SRX 800 radio tracking receiver (Lotek Wireless, Newmarket, ON) and one or two 3-element yagi antennas (AF Antronics, Urbana, IL) (e.g., pointed up- or down- stream) powered by solar power (G2 Solar Corp, Carlgary, AB), to record passage events. Antennas were secured to a tree in both the up- or down- stream direction, except for the fishway antennas that were placed at a 90° angle, perpendicular to the stream bank. At the nature-like fishway, there were two fixed receiver stations; one was placed at the upstream entrance, while the other was placed at the downstream entrance, collectively referred to as Site 3 (S3). The downstream entrance station had one yagi antenna, pointed at a 90° angle towards the downstream entrance to detect fish entering (and approaching) the fishway. The second station at the upstream entrance had two yagi antennas, one pointed at a 90° angle towards the upstream entrance (concrete apron) and the other antenna pointed at a greater angle to detect fish exiting (or approaching) the fishway (see Figure 3.6.1). This provided the opportunity to track a fish's location as it passed through the fishway in a specified order based on relative signal strengths between the antennas. There were four additional fixed receiver stations, at both 2-km and 6-km up- and down- stream of the dam. The downstream sites were referred to as Site 1 (S1) and Site 2 (S2). The dam site was called Site 3 (S3) and the two upstream sites were Sites 4 and 5 (S4 & S5 respectively; see Figure 3.6.2).

Our study involved two groups of fish. Because we knew from historical sampling that bull trout existed at very low densities downstream of the dam, we enhanced the population by transporting upstream residents downstream (i.e., below the fishway; Parks Canada, unpublished data), in the hope that these individuals would exhibit homing behaviour which has been displayed in other salmonids (e.g., Halvorsen & Stabell, 1990). We also acknowledged that these fish were residents and may not move upstream at a high enough frequency to ensure adequate sampling size of fish if translocation was not used.

The translocated group of fish were caught upstream (~14-km upstream of the nature-like fishway) and transported within 1-km downstream of the fishway (S3) by helicopter in a Bambi bucket in the fall of 2015 (N = 52; Oct. 29 – 30, 2015) and spring of 2016 (N = 21; May 2 – 3, 2016; N=21, Fork Length (FL), 210-mm to 320-mm) where they were tagged and released. A control group of non-translocated fish (N=60; FL, 238-mm to 388-mm) were released within a few 100-meters of their capture sites in the reach upstream of the dam. The non-translocated fish release sites were located within 1-km to 2-km downstream of the upstream fixed receiver stations (S4 and S5). This was done to determine fishway permeability by comparing the number of approaches at the fishway by translocated fish (S3), with the number of approaches at upstream control sites by non-translocated fish (S4 and S5). The upstream fixed receiver stations (S4, S5) as well as the downstream fixed receiver stations (S1, S2) also provided a method of quantifying coarse-scale movement patterns of fish following a passage event through the fishway (2-km or 6-km in both directions).

To ensure the fixed receiver stations were working properly, range testing was conducted on a weekly basis at the nature-like fishway (S3) and bi-weekly at the remaining up- and down-stream stations (S1, S2, S4, S5). The translocated individuals were also manually tracked on a seasonal basis to understand their relative movements and spatial location outside of the fishway. This was especially important for the individuals that never approached the fishway. There were 7 translocated individuals in our study that were never recorded during our manual tracking sessions. It is possible that these individuals could have left the system, experienced tag failure, or were predated and removed from the system.

To gain insight on the environmental conditions that may influence fishway use, water level loggers (model U20L, Onset Hobo Inc.) were used to collect water depth (to the nearest cm) and water temperature (to the nearest 0.5 °C) at 30-min intervals. These loggers were installed within a few 100-m downstream of the fishway. Passage events were correlated with the closest water level and temperature measurements in our dataset.

### *3.3.3 Data Analysis*

A number of conditions were used to define movement activity in this study. At the fishway, an approach event was defined by a fish that was recorded at the entrance of the fishway (or upstream control sites) regardless if the fish went on to pass through the fishway. A "no approach" event was defined as a fish that was never recorded at the fishway. There were two options that defined the probability of passage. An "attempt" was defined as an event where a fish entered the fishway and was recorded at the up- or down- stream entrance of the fishway, but did not make a successful pass through the fishway. This meant that the fish was not recorded by the fixed receiver station at the opposite entrance from which the fish entered. Oftentimes studies will differentiate between the individuals that enter the fishway or remain within 3-m of the fishway entrance (Bunt et al., 2012). Based on these definitions we were unable to differentiate between the fish that “entered” the fishway in comparison to fish that were within a reasonable distance from the entrance (i.e., 5-m), and so we included both possible events as “attempts” in this study. A successful “passage” was defined as an event where a fish was recorded on all three antennas at the fishway (downstream entrance, fishway interior and upstream entrance) in consecutive order, and was dependent on passage direction. Passage duration was determined for each successful passage event, it began when a fish entered at one of the fishway entrances (up- or down- stream) and was completed when a fish reached the opposite entrance.

We recognize that using translocation could influence the standardized estimates for evaluating the effectiveness of fishways (such as attraction efficiency) by manipulating a fish’s behaviour in the hopes of motivating them to approach the fishway (i.e., homing; Hinch & Cooke, 2013). As such, we modified this efficiency estimate to reflect a more appropriate estimate given our study design. We term this estimate as “probability to approach”, which was calculated as the proportion of translocated fish (tagged and released downstream of the fishway) that subsequently approached the fishway, in comparison to the total number of translocated fish that were released in our study. We assumed that the non-translocated fish released upstream of the fishway were unlikely to move downstream as they were probably habituated to the dam being a barrier, and so they were not included in our estimate. Bunt et al. (2012), define passage efficiency by dividing the number of fish that entered the fishway by the number of fish that exited the fishway. In our study, individuals that passed through the fishway were likely to do so more than once and so we defined “passage efficiency” as the total number of successful passage events through the fishway compared to the total number of attempts at the fishway (regardless of fish ID). This provided a more conservative estimate because “multiple” attempts and passage events per fish occurred over various environmental conditions in our study period (i.e., late winter – fall).

### *3.3.4 Statistical Analysis*

To account for two separate release dates for translocated fish we ran a chi-square test to compare the proportion of individuals that approached the fishway by release date. This analysis failed to detect a significant difference ($X^2$=1.97, df=1, p=0.16) between the two release groups so they were grouped together for modelling. When multiple observations are made on the same individuals, the data are not independent (Heck et al., 2010). This was true for our study as fish that approached the fishway were likely to approach (and pass) more than once (multiple observations per fish). In recognition of this, we incorporated individual fish ID as a random effect in mixed effects regression models. We used backward model selection with Akaike’s information criterion (AIC, Akaike, 1974) to objectively compare model fits, and determine the most parsimonious model with the lowest AIC value. Prior to modeling, we used pairwise Spearman’s rank correlation plots and variance inflation factors (VIF) to assess if there was multicollinearity between predictor variables. It was found that fish weight and fork length were collinear (upon visual inspection), and so fish weight was removed from any further analyses. This was done as we recognized that fork length provided an equivalent metric to evaluate the influence of body size and has often been used in other studies related to fishway science (e.g., Stuart & Mallen-Cooper, 1999).

To determine the probability to approach the fishway (and upstream control sites), we ran a generalized linear mixed effects model (GLMM) with a binomial response (i.e., approach or no approach). Data were analyzed with glmer function in lme4 package in R statistical environment (R Studio 3.3.3; Bates et al., 2015). This model included presence/absence data for each fish (i.e., translocated and non-translocated) by season ("late" winter, spring, summer and fall) at the upstream control site or nature-like fishway (termed as "location") with fish ID as a random effect. Season was treated as a fixed factor with four levels, winter (March), spring (April & May), summer (June, July and August) and fall (September and October) and we used winter as baseline from which to generate the comparisons. Location was treated as a fixed factor with two levels that included the treatment group (i.e., the fishway (S3)) and control (S4 and S5). By comparing the probability to approach each site, we could compare whether fish approached the fishway at the same rate as the upstream control sites. This is effectively a measure of "permeability" under the assumption that a perfect fishway would be "invisible" to fish (i.e. fish would approach the fishway as often as any other area in the stream). We used predicted probabilities using the predict function in R statistical environment to describe relationships between predictor variables.

To determine the probability to pass through the fishway itself, we used a generalized linear mixed effects model (GLMM) with a binomial response (pass or attempt). This was done to test if water depth, water temperature, time of day and passage direction (up- or down- stream) influenced the probability of a fish to pass. In this model, we only included the fish that approached the fishway because we did not have environmental data for the fish that never approached the fishway. Time of day was included as a binary predictor of night or day. We also included both up- and down- stream passage as a binary predictor. Water temperature and water depth were included as continuous predictors and fish ID was included as a random effect.

To determine if passage duration was influenced by biotic and/or abiotic factors, we used a linear mixed model. The response (i.e., passage duration) was modeled as a continuous variable with a Gaussian error distribution. We included water depth and water temperature as continuous, time of day (as a factor with two levels) and fork length in our model, where fish ID was specified as a random effect (as there were more than one passage duration per fish). Data were analyzed using lme function in nlme package implemented in R statistical environment (R Studio version 3.3.3; Pinheiro et al., 2014). Residual plotting was used to test for model assumptions that included normality and homogeneity of fixed effects residuals (Zuur et al., 2009). Where heterogeneity of variance between fixed effects was observed, the variance weighting function varIdent from the nlme package (Pinheiro et al., 2014) was applied (Zuur et al., 2009).

**3.4 Results**

The probability for translocated fish to approach the fishway was low (37%; 27 of 73 translocated individuals) and consistent with upstream control sites (33%; 20 of 60 non-translocated individuals). However, most of translocated fish that approached the fishway passed through the fishway (21 of 27 translocated individuals), as well as two non-translocated individuals that were released upstream of the fishway. Fish were likely to pass the fishway more than once, with a passage efficiency of 78% (54 of 69 passage events). There was only 22% (N=15) attempt-only events at the fishway. These were likely not failed attempts but rather a lack of motivation of a fish to ascend (or descend) the fishway.

Our best ranked model for probability to approach (either the fishway or control sites) included all predictor variables ($R^2$=0.17; Table 3.7.1). Our results suggest that the fishway was indeed as permeable as the upstream control sites (p=0.21) and that movement at both the control site and fishway differed among seasons. There was a relatively low number of approaches during late winter of 2016 (when transmitters turned back on) to early spring of 2016 (p=0.56). However, the frequency of approaches increased in late spring and into the summer (p=0.003), and was followed by a decline into late fall (p=0.28; see Figure 3.6.3). There was a significant interaction between location (i.e., the fishway and upstream control sites) and fork length (p=0.001), where large non-translocated individuals were more likely to approach the upstream control sites, while small translocated individuals were more likely to approach the fishway (Figure 3.6.4). For example, the probability to approach the upstream control sites increased by 0.40, for a large non-translocated individual (with FL 320-mm) when compared to a small non-translocated individual (with FL 240-mm). In contrast, the probability to approach the fishway increased by 0.30 for a small translocated individual (with FL 230-mm) when compared to a large translocated individual (with FL 310-mm).

The best ranked model for the probability to pass through the fishway included both water depth and time of day ($R^2$=0.25; Table 3.7.2). Although fish could pass during the day, the probability of passage was higher at night (p= 0.005) and at greater water depths (>0.40-m; p= 0.0007; Figures 3.6.3 & 3.6.5). For example, at the mean water level (0.40-m), the probability of passing through the fishway at night increased by 0.30, in comparison to during the day. In addition, if water level rose by 10 cm, the probability to pass at night also increased 0.30. Fork length (p=0.34), temperature (p=0.31), and direction of passage (p=0.61) were not included in the top ranked model, with insignificant p-values. This suggests that although small individuals were more likely to "approach" the fishway, there was no size limitation for those that passed the fishway (p=0.34).

Passage duration through the fishway varied between 5-min to 13-days with an average of 1.80-days ± 2.87-days (± SD; Table 3.7.3). Passage duration was not significantly influenced by water depth (p=0.74), time of day (p=0.10), water temperature (p=0.25), or fork length (p=0.87). There were a number of translocated individuals that underwent large-scale movements following a passage event through the fishway by travelling 2-km upstream (N=11), or 2-km downstream (N=2; Table 3.7.4 & Figure 3.6.6). There were no translocated individuals in our study that exhibited homing behaviour, as they did not pass 6-km (S5) upstream of the fishway which would have been required to ultimately reach their upstream capture sites. In addition, we did not see any translocated (or non-translocated) individuals pass the 6-km downstream site (S1) towards the Bow River.

### 3.5 Discussion

Nature-like fishways provide an innovative way of restoring fragmented systems and have improved connectivity for both resident (e.g., Calles et al., 2007; Steffensen et al., 2013) and migratory species (e.g., Calles & Greenburg, 2009; Franklin et al., 2012) when a land-management agency does not have the capacity to completely remove a dam. In the present study, although the probability to approach the nature-like fishway was low (37%), most of the translocated individuals that approached also passed through the fishway (78%). It is likely that the fishway did not act as a velocity barrier to stream-dwelling resident bull trout and so the limited number of attempt-only events (22%) at the fishway were probably not "failed" attempts but rather a lack of motivation by the fish to ascend (or descend) the fishway. These findings are remarkably different from the impassable barrier that was once present in this system. This provides evidence that a nature-like fishway with minimal in-stream engineering can restore connectivity for resident salmonids.

Water temperature has often been shown to influence both up- or down- stream movement of fishes (e.g., Hohausova et al., 2003; Prchalova et al., 2006; Taylor et al. 2013), and fishway use (Castro-Santos et al., 2009). Previous studies on resident bull trout have shown an increase in downstream overwintering movements (>1-km) with declining temperatures in the fall (e.g., Jakober, 1998). Our study found that temperature did not influence passage success. This is likely because our system did not experience a wide range of temperatures ($0.5$-$9.0 \pm 1.8$ °C) during our study period. A similar study (and study system) documented by Jakober et al. (1995) in a high elevation stream (1408-m), found that the direction and extent of movement by bull trout in the fall was limited (2-m to 316-m) but that large-scale movements were instead triggered in winter (i.e., November), and involved searching for downstream overwintering habitat to settle into (which would not have been captured in this study).

Changes in water level have also been associated with large-scale fish movements (Alabaster et al., 1970; Egglishaw et al., 1992; Taylor and Cooke 2012), and are often an important predictor of passage success through fishways (e.g., Mallen-Cooper et al., 2007; Cahill et al. 2016). We observed a relatively low number of passage events in early spring, followed by an increase in passage events from late spring to early summer and then a decline until late fall. The increase in the number of passage events that began in late spring corresponded with greater water depths (>0.40-m), which was likely a proxy for discharge in this system (as seen in Aarestrup et al., 2014), with point discharges that ranged from (1.91-$m^3$/s to 2.33-$m^3$/s) at high water levels. The movement activity recorded at the nature-like fishway by season (i.e., late winter, spring, summer, fall) was comparable to the upstream control sites. This suggests that the lack of passage events at low water levels was not characteristic of the fishway, but rather the biology of the fish in this study system (see Figure 3.6.3). Therefore, higher water levels likely provided motivation for a fish to pass, but did not limit overall passage success.

Passage success has often been associated with body size for many fishway types (e.g., Denil or vertical slot fishways; see Schwalme et al., 1985; Noonan et al., 2012; Podgorniak et al., 2016). Here, we observed that small translocated individuals were more likely to approach the fishway than their larger counterparts (see Figure 3.6.4). It is plausible that the limited use of the fishway (37%) could be a result of their life history as residents and from being displaced, rather than size selectivity of the fishway itself. This is reasonable to assume given that the fishway was short in length (50-m) and had a relatively low gradient (5.3%), which likely did not serve as a physical impediment to bull trout movement. It is possible that large translocated individuals were able to secure a "new" home range further downstream of the fishway (i.e., and thus did not exhibit homing behaviour). In doing so, this likely required smaller individuals to use remaining (potentially sub-optimal) downstream habitat or encouraged them to move upstream. In contrast, large non-translocated bull trout (i.e., control fish) were more likely to approach upstream control sites in comparison to their smaller counterparts (see Figure 3.6.4). We suggest that these large non-translocated (i.e., control) individuals could use their home range more effectively by frequently transitioning between home sites (see Clap et al., 1990). In contrast, their smaller non-translocated counterparts may not have had access to same opportunities, limiting their home range use (see Clap et al., 1990).

In this study, there was a number of translocated individuals that underwent large-scale movements 2-km up- (N=11) and down-stream (N=2) of the fishway following a successful passage event (See Table 3.7.4 & Figure 3.6.6). It is plausible that these fish were exhibiting searching behaviour by attempting to locate and ultimately secure a new home range, which could have been triggered if they were unable to locate optimal foraging habitat from their downstream release site. This behaviour likely occurs when foraging conditions become sub-optimal (potentially occupied by larger translocated individuals; Gowan & Fausch, 2002; Rodriguez, 2002). We suggest that non-translocated individuals did not frequent the fishway (N=2), as they were likely habituated to the dam being a prominent downstream barrier for over 100-years.

Stream-dwelling bull trout tend to exhibit diel habitat partitioning and often emerge from cover at night (e.g., large woody debris, boulder crevices and deep pools), where they shift towards using low cover and/or shallow water habitats (Jakober et al., 2000). Although passage events occurred during daylight, there was an increase in passage events at night. Indeed, nocturnal activity patterns have been observed in several species for fishway use (e.g., Thiem et al., 2011; Cahill et al., 2016). In our study, movements often occurred after dusk when this species would be less vulnerable to predation (e.g., birds; Alvarez & Nicieza 2003; Railsback et al., 2005), and would potentially have more foraging opportunities (e.g., Metcalfe et al., 1999; Furey et al., 2015). This also explains the extended periods of time that fish spent in the fishway (upwards of 13-days). Indeed, it is possible that once a fish navigated into the fishway they may have been able to access profitable foraging opportunities and/or ample cover to hide under and settle into, before moving up- or down- stream at a later time.

Our study has provided novel information on fishway use by a threatened salmonid through a nature-like fishway in Banff National Park, Alberta. The probability to approach the fishway was size dependent (favoring small translocated individuals), and relatively low (37%) but comparable with upstream control sites (33%). Passage success was not influenced by passage direction, temperature or fork length (for those that successfully approached). Increased water depths (>0.40-m) corresponded with a high probability of passage success (which was interchangeable with season) and passage events often occurred at night during the summer months. Here, we provide the groundwork for future studies to explore how fishway use and dam removal can be combined through an innovative study design. The aim of this study was to aid researchers looking to expand the science beyond dam removal and recognize the opportunities of combining these two concepts to understand system connectivity as a whole.

### 3.6 Figures

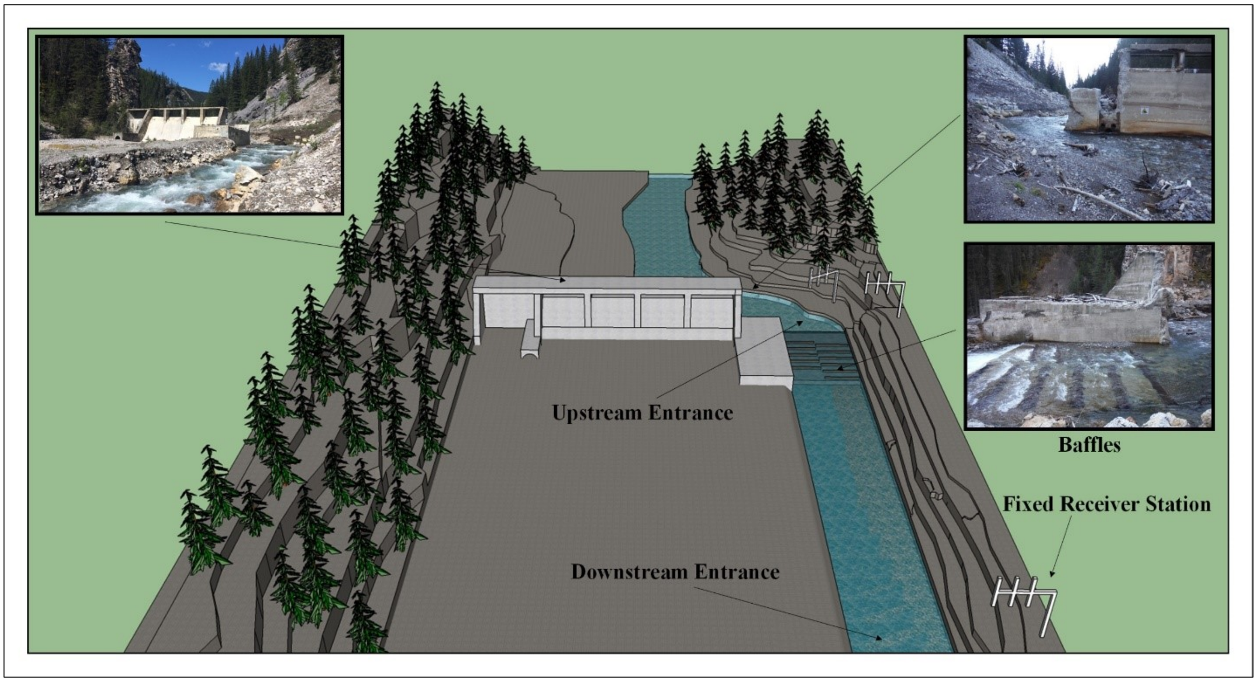


**Figure 3.6.1** Schematic drawing of nature-like fishway in Forty Mile Creek, Banff National Park, the upstream entrance of the fishway is characterized by 11 evenly spaced baffles to control flow and reduce potential bank erosion, the interior of the fishway and downstream entrance are characterized by natural rocky substrate (e.g., cobble, boulders). Fixed receiver stations are represented by antennas at the up- and down- stream entrance of the fishway. The two antennas depicted at the upstream entrance account for one fixed receiver station, whereas the single antenna at the downstream entrance accounts for a separate fixed receiver station.

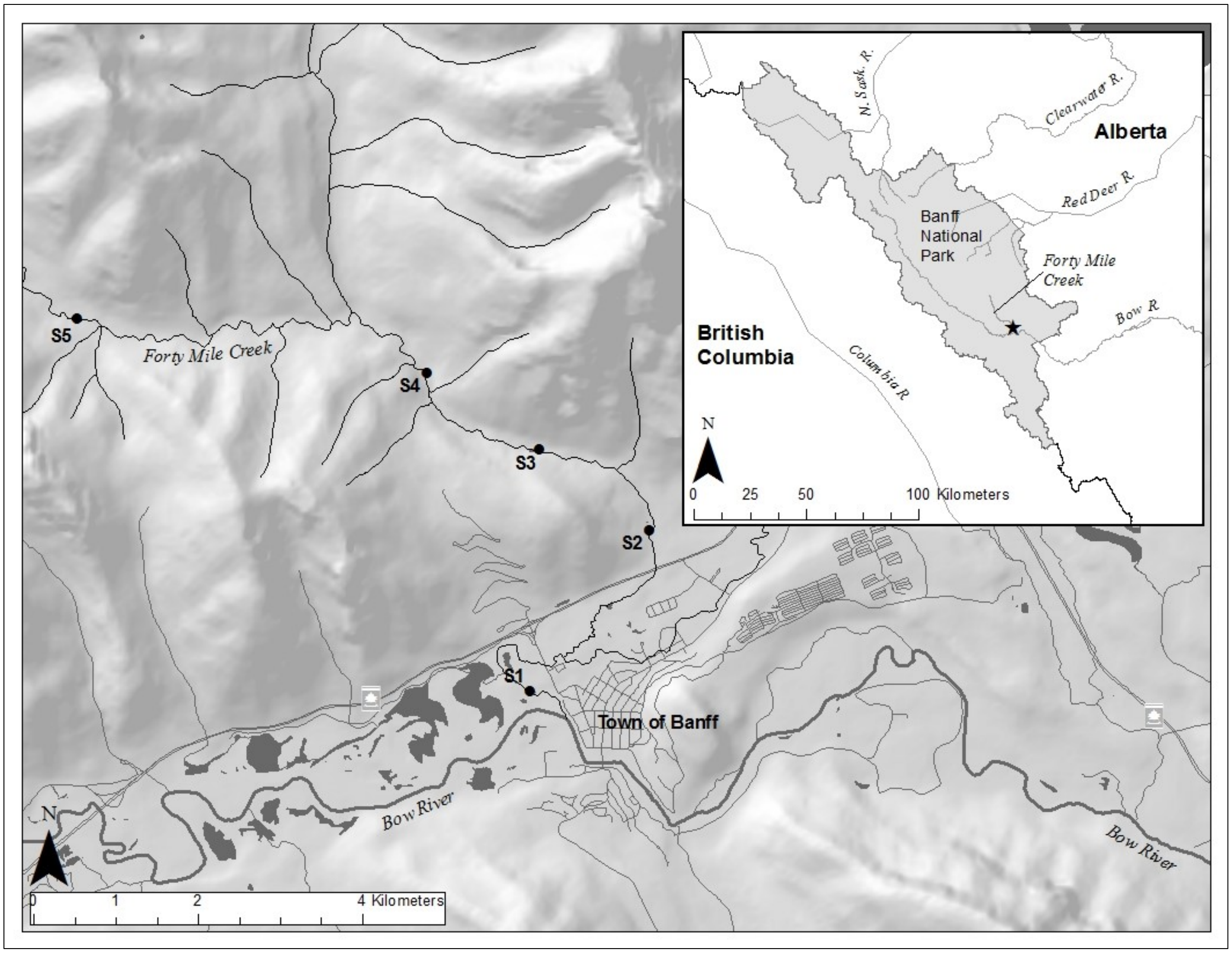


**Figure 3.6.2** Radio telemetry fixed receiver stations along Forty Mile Creek in Banff National Park. S3 represents the site of the nature-like fishway, S4 and S2 are stations positioned 2-km up- and down- stream of the fishway respectively, while S1 and S5 are 6-km up- and downstream of the fishway respectively

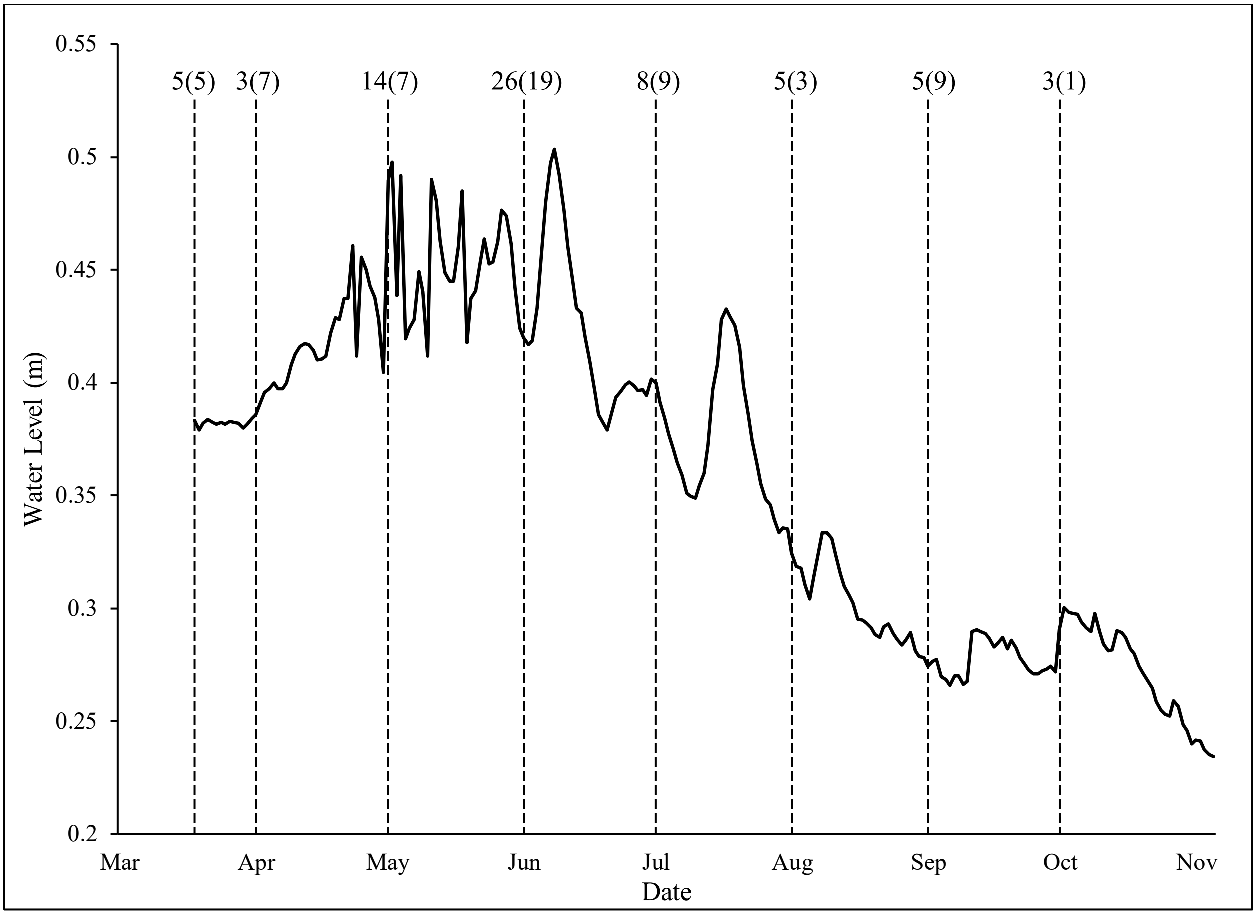


**Figure 3.6.3** Water level measurements for the nature-like fishway in Forty Mile Creek, Banff National Park (March – October 2016). The number of approach events are provided on a monthly basis at the fishway and the (upstream control sites) in brackets to show the relative changes in movement activity in the study system over varying water levels (interchangeable with season).

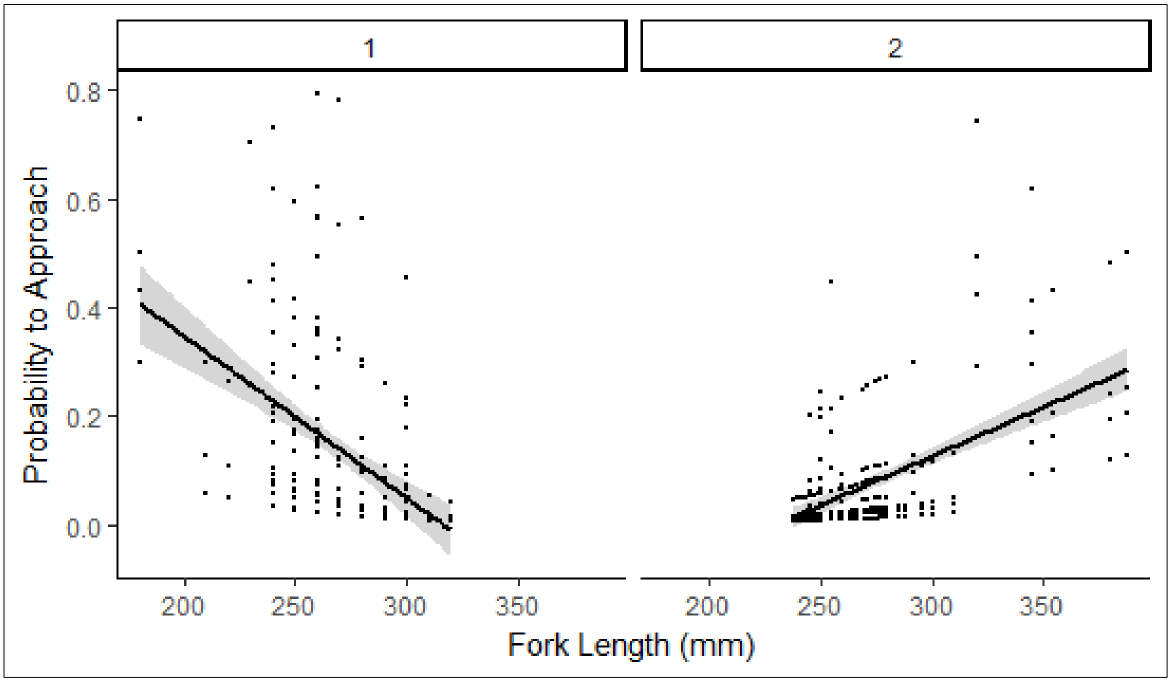


**Figure 3.6.4** Probability curve depicting the probability to approach the fishway (1) by translocated individuals and the probability to approach the upstream control sites (2), by non-translocated (ie., control) individuals based on fish fork length in Forty Mile Creek, Banff National Park. Shaded area accounts for 95% confidence intervals.

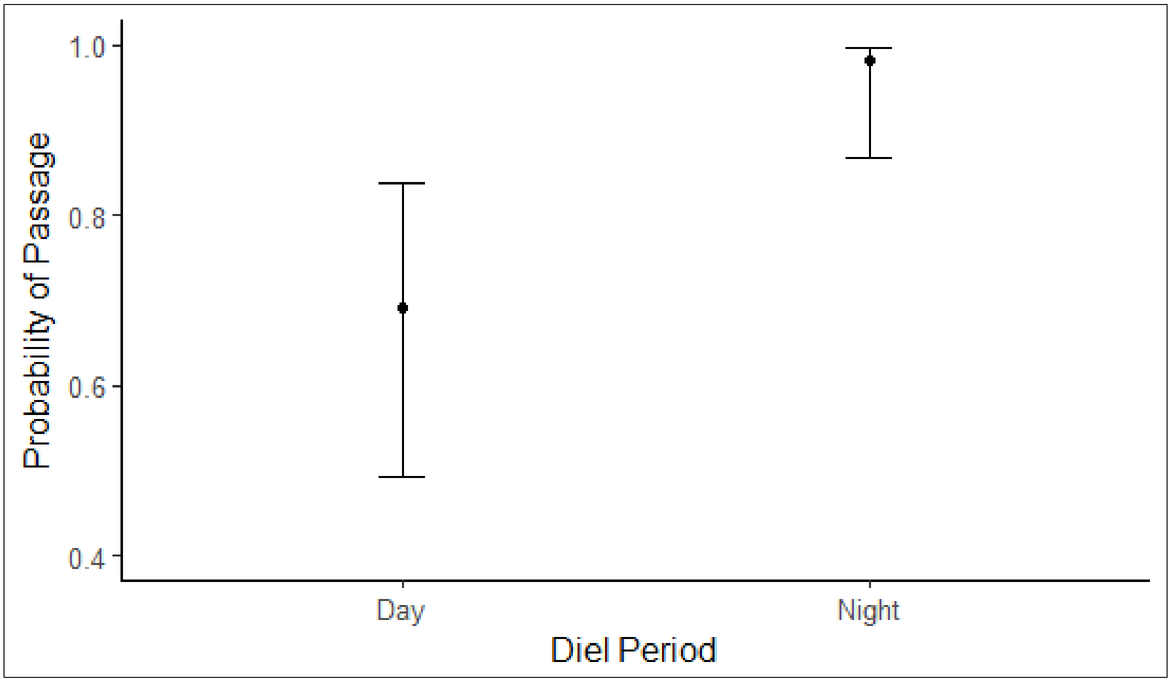


**Figure 3.6.5** Predicted probabilities of passage through the nature-like fishway by bull trout in Forty Mile Creek, Banff National Park with a total of 69 events by translocated (N=27) and non-translocated (N=2) individuals with standard error bars (± SE), night and day were categorized based on local sunset and sunrise times during our study period while water level was held at it's mean (0.40-m).

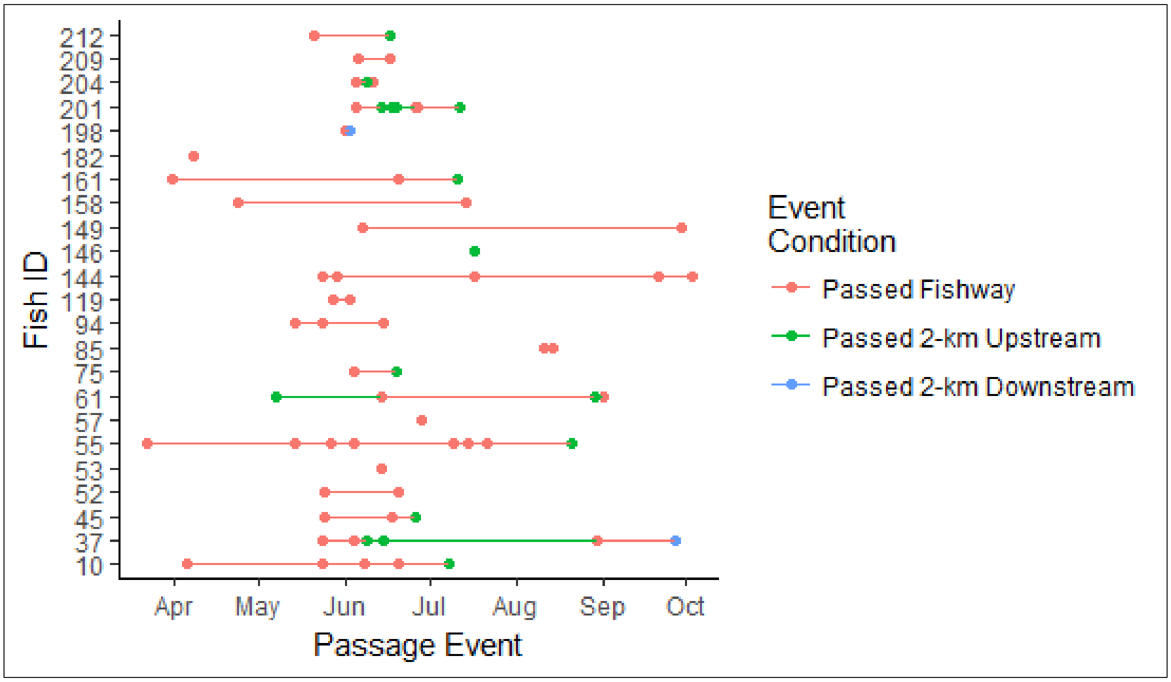


**Figure 3.6.6** Passage success by both translocated (N=21) and non-translocated (N=2) individuals at the fishway and/or 2-km up- or down- stream of the fishway defined by month. One passage event at the fishway was not recorded for ID 61 which likely passed when tags were turned off overwinter and therefore was first captured descending the fishway in this study.

### 3.7 Tables

**Table 3.7.1** A subsample of the individuals that used the fishway for multiple up- and down-stream passes (S3) and/or made large-scale movements 2-km up- or down- stream (S4 and S2 respectively), transit times are provided as dd:hh:mm:ss, adopted from (Cahill et al., 2016).

| Fish ID | Station Passed | Number of Passes | Median Transit Time (S3) | (Min) Max Transit Time (S3) |
|---|---|---|---|---|
| 10 | S3 | 4 | 00:14:13:19 | (00:03:02:23) |
| | S4 | 1 | | 8:22:17:17 |
| 144 | S3 | 5 | 00:01:35:44 | (00:00:33:20) 00:04:44:08 |
| 55 | S3 | 7 | 02:21:16:21 | (00:01:31:30) |
| | | | | 07:23:33:40 |
| 161 | S3 | 2 | 0:3:04:55 | (0:0:54:26) 1:1:38:51 |
| | S4 | 2 | | |
| 61 | S3 | 2 | 4:0:09:40 | (0:0:20:12) |
| | S4 | 1 | | 7:23:59:08 |

**Table 3.7.2** Parameter estimates for a generalized linear mixed effects model using glmer function in lme4 package in R statistical environment (R Studio 3.3.3), to understand the probability of approach for translocated bull trout through a nature-like fishway and non-translocated bull trout at upstream control sites in Forty Mile Creek, Banff National Park.

| | Estimate ± SE | z-value | P-value |
|---|---|---|---|
| **Intercept** | -2.94 ± 0.53 | -5.55 | $2.93e^{-08}$ |
| **Spring** | 0.28 ± 0.48 | 0.58 | 0.56 |
| **Summer** | 1.36 ± 0.45 | 3.02 | 0.003 |
| **Fall** | -0.58 ± 0.54 | -1.08 | 0.28 |
| **Receiver** | -0.57 ± 0.46 | -1.25 | 0.21 |
| **FL** | -0.86 ± 0.35 | -2.44 | 0.02 |
| **FL * Receiver** | 1.47 ± 0.45 | 3.30 | 0.001 |

**Table 3.7.3** Parameter estimates for a generalized linear mixed effects model using glmer function in lme4 package in R statistical environment (R Studio 3.3.3), to understand the probability of passage for bull trout through a nature-like fishway in Forty Mile Creek, Banff National Park.

| | Estimate ± SE | z-value | P-value |
|---|---|---|---|
| **Intercept** | -9.60 ± 3.02 | -3.17 | 0.002 |
| **Water Depth** | 26.00 ± 7.62 | 3.41 | 0.0007 |
| **Time of Day** | 3.15 ± 1.11 | 2.84 | 0.005 |

**Table 3.7.4** Parameter estimates for a linear mixed effects model using lme function in nlme package in R statistical environment (R Studio 3.3.3), to understand biotic and abiotic factors influencing passage duration for bull trout through a nature-like fishway in Forty Mile Creek, Banff National Park.

| | **Estimate ± SE** | **t-value** | **P-value** |
|---|---|---|---|
| **Intercept** | -2.07 ± 6.38 | -0.32 | 0.75 |
| **FL** | 3.23 ± 20.60 | 0.16 | 0.88 |
| **Temperature** | -0.16 ± 0.25 | -0.66 | 0.52 |
| **Water Depth** | 8.11 ± 7.01 | 1.16 | 0.26 |
| **Time of Day** | 1.24 ± 0.78 | 1.59 | 0.12 |

## Chapter 4: General Discussion

### 4.1 Findings and Implications

As an effort to improve dam removal science around the globe and expand the knowledge base on restoring connectivity in streams (i.e., nature-like fishways), this thesis began with a review that synthesized the methodologies and trends in fish responses to dam removal. It was found that most studies lacked an appropriate experimental design (including replicability, reliability, and relevance) to properly test whether dam removal is achieving fish restoration objectives. These findings will hopefully stimulate discussion and action towards using data in creative ways through extracting information from long-term monitoring programs or comparative watersheds to understand the system prior to dam removal, as well as putting monitoring programs in place for systems that will likely undergo dam decommissioning in the coming years. By allocating resources that focus on well-defined research and monitoring efforts over long-term timescales (using multiple end points), it will be possible to grow a high-quality evidence base for future research in an era where dam decommissioning is likely to occur more than ever before (Poff & Hart, 2002), and the importance of evidence based decision-making is at large (Sutherland et al., 2004).

Chapter 3 set out to investigate the effectiveness of a nature-like fishway in supporting the up- and down- stream movement of bull trout following the partial removal of a small-scale dam. The biological evaluation of the nature-like fishway explored both abiotic and biotic factors that may influence the probability to approach, probability of passage and passage duration. It was found that the probability to approach the fishway was low when individuals were translocated downstream of the fishway (37%), but for those that approached the fishway, their passage success was high, with a passage efficiency of 78%. Movement captured at the nature-like fishway by translocated individuals was similar to non-translocated individuals at upstream control sites. Passage success was determined by water depth and time of day, while passage duration ranged from 5-min to upwards of 13-days, showing that the fishway could have other purposes for fish (e.g., foraging). This research also suggests that resident bull exhibit size dependent movement that may be triggered from displacement (i.e., translocation), or level of habitat use (i.e., non-translocated).

### 4.2 Future Directions

As part of Chapter 2, we provided guidance for improving the evidence base on dam removal in meeting fish restoration objectives. When future research pertaining to dam removal cannot effectively follow these guidelines due to time limitations (i.e., possible dam failure, liability) or lack of sufficient baseline data (e.g., including watershed reference data), we encourage researchers to evaluate the study system itself and proceed accordingly. For example, instead of quantifying fish response to dam removal directly, it can be combined with other topics to answer pressing questions that will enrich the evidence base in other ways (i.e., nature-like fishways), in doing so we will gain a better understanding of system connectivity as a whole.

As part of Chapter 3, we noted size dependent movement patterns in non-translocated (control) individuals, in which large individuals were more mobile than their smaller counterparts. Gerking (1959) proposed a theory concerning the restricted movement of stream fishes, that was later termed a paradigm for salmonid biology. The so-called "restricted movement paradigm" (RMP) proposed that resident stream-dwelling salmonids are sedentary and stay within deep pools with little movement outside of these small home ranges (less than a few 100-m; defined in Gowan et al., 1994). However, most of the studies that support this paradigm rely on mark re-capture estimates that have been recognized as biased for detecting movement (e.g., Gerking, 1953; Berra & Gunning 1972). In recognition of this, researchers have moved towards using telemetry to depict movement patterns in resident fishes. By using telemetry, it has been found that many resident species undergo extensive up- and down-stream movements (Walker et al., 2012), and furthermore these movements have the potential of being an important indicator of habitat enhancement (Deboer et al., 2015).

Future research should focus on understanding the mechanisms that influence size dependent movements for bull trout on a continuous scale (i.e., fixed receiver stations) with supplemental fine-scale manual tracking to assess habitat use (as a potential trigger for movement). This will provide clarity on the patterns (or processes) defining these movements, from which an evidence-based approach to fisheries management (e.g., regulations, habitat modifications) can be enacted that accounts for the "size dependent" mobile and sedentary components of this threatened salmonid's population.

**4.3 Conclusion**

We hope this thesis advances dam removal science across the globe and connects the topic of dam removal with other important areas of research (i.e., nature-like fishways). With case studies on the topic of dam removal (that follow guidelines/advice provided in this thesis) as well as those that combine dam removal with nature-like fishways together (as shown in Chapter 3), we will be able to continue to enrich the knowledge base for restoring connectivity in riverine systems in the years to come.

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