Rocky Creek bull trout cumulative effects model and threat summary

APRIL 2023

This publication is available online at https://open.alberta.ca/publications/rocky-creek-bull-trout-cumulative-effects-model-and-threat-summary

For more information regarding this content visit: https://open.alberta.ca/documentation/pubguidedraft

This publication may be cited as: Government of Alberta. 2023. Rocky Creek Bull Trout Cumulative Effects Model and Threat Summary. Edmonton, AB. 32 pp.

Rocky Creek Bull Trout Cumulative Effects Model and Threat Summary | Environment and Protected Areas
 | Published: April 2023 | ISBN 978-1-4601-5716-9

Contents

Abstract

Alberta's fisheries cumulative effects modelling approach (Joe model) was used to assess the threats to bull trout in four Clearwater River watersheds (Rocky, Elk, Cutoff and Limestone creeks). The bull trout Joe model is a semi-quantitative cumulative effects model that combines 18 different threats and was populated with data specific for each of the four watersheds. This report documents the rationale for the relationship between the modelled stressors and predicted bull trout system capacity (stressor-response curves), describe the data used to populate the stressor input values, and provide a short summary of stressors and model outputs for each of the four study watersheds.

In all four Clearwater River watersheds, the most limiting, potential threats were from human-caused sediment, stream fragmentation, and angling mortality. The results from the Joe model indicated that habitat remediation in the Rocky Creek watershed would address the main threats limiting the bull trout population.

Introduction

Recovery of bull trout (*Salvelinus confluentus*) is a priority of the Alberta government. Many historical Alberta bull trout populations had high or very densities relative to a reference benchmark; however, higher density populations only remain in a handful of headwater watersheds (AEP 2013, Sinnatamby et al. 2020). Within the province, the species is listed as Threatened under the Alberta *Wildlife Act*, and as Threatened or Special Concern under the federal *Species at Risk Act* (The Canada Gazette 2019).

Understanding which threats limit bull trout populations and mitigating these threats is a critical component of the Alberta Native Trout Recovery program. The Alberta's fisheries cumulative effects modelling approach (Joe model) (MacPherson et al. 2019, MacPherson et al. in prep) is a helpful tool to synthesis information on threats, develop a hypothesis of the main limiting factors and explore the recovery potential of populations under various recovery action scenarios. The bull trout Joe model is a semi-quantitative static model composed of a series of stressor-response curves developed by the Alberta government and stakeholders that describe the relationship between bull trout system capacity and threats (MacPherson et al. 2019).

The bull trout Joe model was run on four study watersheds in the Clearwater River drainage with the intent of informing recovery actions in one watershed (Rocky Creek), and investigating the appropriateness of using Elk, Cutoff and Limestone creeks as controls in a monitoring program. Rocky Creek is a tributary to the Clearwater River that provides important spawning, rearing and overwintering habitat for bull trout. The Alberta government and our partners were interested in implementing recovery actions, specifically habitat remediation, in Rocky Creek, because habitat damage appeared substantial but mitigation measures were feasible in terms of access, cost and engineering. Rather than relying strictly on expert opinion, we wanted to use a science-based process (Joe model) to investigate if our recovery efforts would indeed address the main limiting factors in Rocky Creek. To evaluate the effectiveness of recovery actions in Rocky Creek, bull trout populations in three control watersheds (Elk, Cutoff and Limestone Creek) were also monitored for comparison purposes. Similar to Rocky Creek, we used the Joe model to understand key limiting threats in the control watersheds to test our assumption that conditions are similar among all study watersheds.

Objectives

The objectives of this report are to:

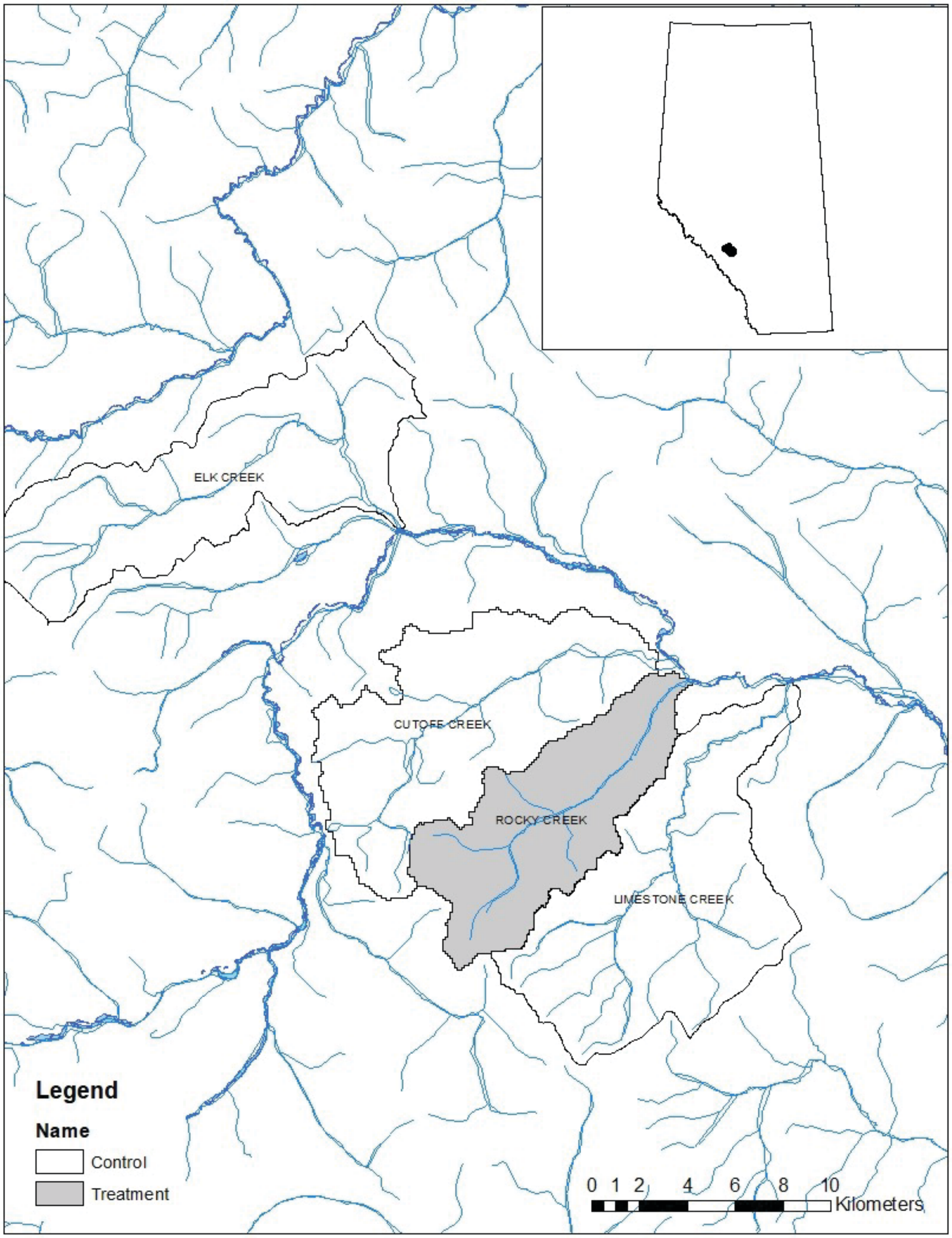
1. Document the rationale for the relationship between the modelled stressors and bull trout system capacity (stressor-response curves) and describe the data used to populate the stressor input values;
2. Provide the specific input parameter estimate (i.e., stressor) for each stressor-response curve for each study watershed; and
3. Provide a short summary of Joe model outputs for the study watersheds.

Methodology

Study area

The bull trout Joe model was run on four watersheds within the Clearwater River drainage, including Elk, Cutoff, Limestone and Rocky creeks (Figure 1). These watersheds are located within close geographic proximity, and are similar in terms of size (60-100 km^2), type and intensity of land use, and fish community composition. All contain a resident bull trout population. In the Rocky Creek watershed, there is substantial habitat degradation resulting from undesignated off-highway vehicle (OHV) use, including 31 OHV crossings within approximately 20 km of stream. Additional anthropogenic footprints include limited forestry and oil and gas activity, and several road crossings with culverts in the lower watershed. The watershed is also accessible via roads, cutlines and OHV trail networks and provides catch and release fishing for bull trout from April 1 to October 31.

FIGURE 1. Four study watershed (Elk, Cutoff, Limestone and Rocky creeks) within the Clearwater River basin. Results from the Alberta cumulative effects Joe model were used to inform recovery actions in the Rocky Creek watershed.

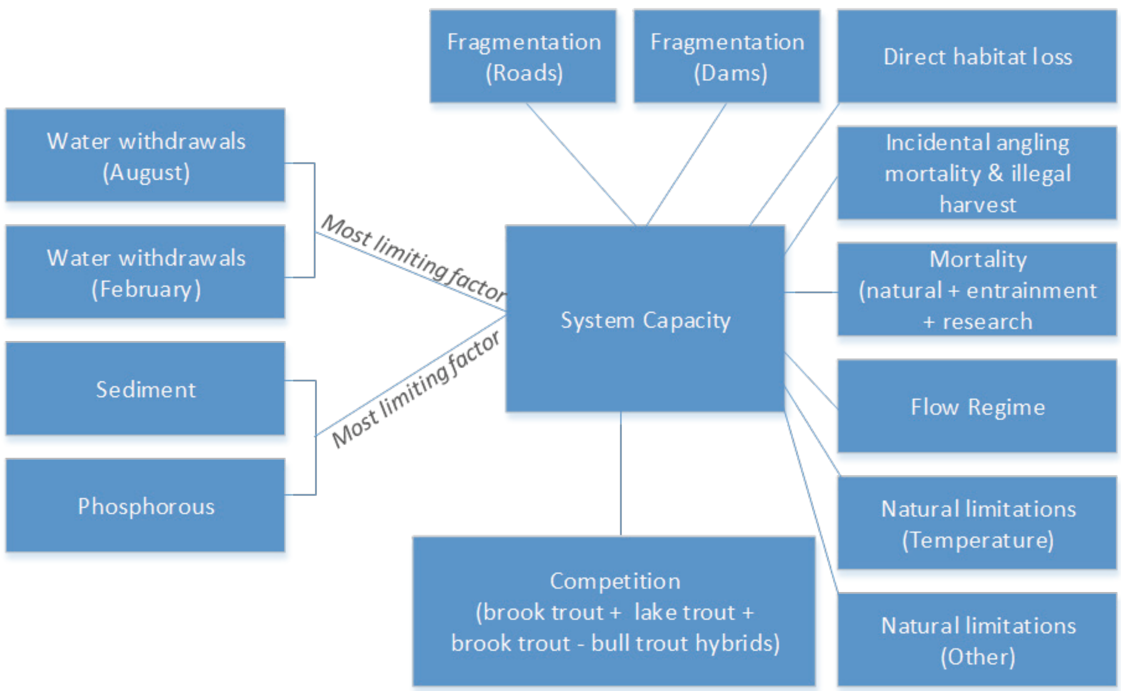


Bull trout cumulative effects Joe model

A fulsome explanation of Alberta's cumulative effects Joe model structure and methodologies can be found in MacPherson et al. (2019). Briefly, the bull trout Joe model is a series of stressor-response curves representing the impacts of threats that are combined to simulate and quantify the cumulative effects on the system capacity of a bull trout population. System capacity is defined as the potential for a system to support adult individuals. System capacity is a dimensionless proportion ranging from 0-100% of a pristine reference condition. If no threats are present, the system capacity will be 100% of the reference condition, whereas system capacity will be far lower if threats are all high.

Several broad categories of threats identified in the Bull Trout Conservation Management Plan (ASRD 2012) have been included in the model. Currently, there are 18 threats organized under 8 categories including natural limitations, direct mortality, fragmentation, competition with other species, changes to water quality, changes to water quantity, changes to flow, and habitat loss (Figure 2). It is expected that there will be refinements to the curves and potentially more (or less) complex interactions between threats as new information is generated during the adaptive management process.

FIGURE 2. Conceptual diagram of the bull trout cumulative effects Joe model.



The severity of each threat within the four study watersheds was determined through analysis of spatial data (i.e., GOA spatial layers or using ALCES Online©), fisheries data available in the provincial Fisheries and Wildlife Information Management System database (FWMIS), modelling exercises and consensus of professional opinion. Threat severity represents the current condition of the watershed.

Bull trout cumulative effects Joe model stressor-response curves

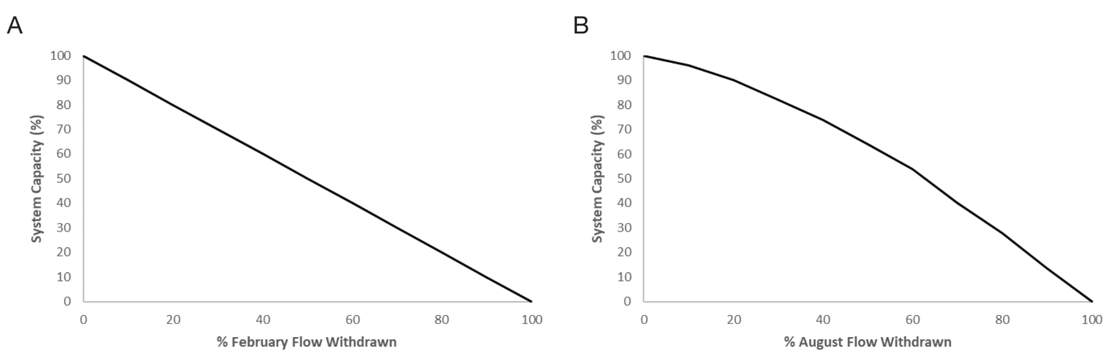
1. Water quantity: Surface water withdrawals

The effect of water withdrawals during February (winter) and August (summer) on bull trout was investigated using a multi-step analytical approach based on the low-flow habitat performance measures developed by Hatfield and Paul (2015). First, it was assumed there was a 1:1 relationship between the minimum available habitat (bottleneck effect) and bull trout population system capacity. To measure habitat, an index presented by Hatfield and Paul (2015) was used which: a) sets all flows >20% Mean Annual Discharge (MAD) to a habitat score of 1 (i.e., maximum suitability); b) has a habitat score of 0 at zero flow (i.e., no suitability); and, c) has a habitat score between 0 and 1 for flows between 0 and 20% MAD using a linear relation. This simple rating curve means that a flow of just under 20% MAD will score close to the maximum of 1, whereas a substantially lower flow will score proportionally less. The index was then used to determine the reduction in habitat scores from water withdrawals. Because withdrawals would have the greatest impact on the habitat score during low flows (i.e., < 20% MAD), percent withdrawal was determined for two periods of the year (August and February) and the lowest 10% of flows (i.e., Q_{90} or 90% exceedance flow) for these months. The approach was then applied to 37 rivers of varying size in Alberta that had year-round natural or naturalized (i.e., corrected for upstream water use) discharge and percent withdrawals ranging from 0–100% were modelled to assess the decrease in the habitat score from natural.

For February flow, all 37 rivers showed a similar linear response in the habitat score to water withdrawals. This average response was used as the basis for the stressor-response curve (Figure 3A). For August flows, the rivers showed a highly variable response in the habitat score to water withdrawals, ranging from linear (similar to February) to curvilinear with little initial response but increasing as withdrawals increased. The 75th percentile regression using a general additive model (Koenker 2017) was used to capture the curvilinear relationship (Figure 3B). The overall cumulative effects model only includes the season during which water withdrawals have the greatest effect on bull trout as physical habitat is assumed to limit populations by the minimum and not the combined product of February and August habitat.

As the Joe model accumulates cumulative effects multiplicatively (additive on a proportional scale), treating these two curves separately would inappropriately overemphasize the expected response. To overcome this issue, we treated February and August flow in the Joe model using a limiting factor approach. Simply, only the strongest, negative response from either the February and August stressor-response curves is used to calculate final system capacity. Anytime a watershed shows either February or August flow to be a hypothesized key driver, it must be acknowledged that the other stressor could be the driver given the collinearity.

FIGURE 3. Stressor-response curves depicting the expected relationship between changes in February (A) and August (B) flows and bull trout system capacity

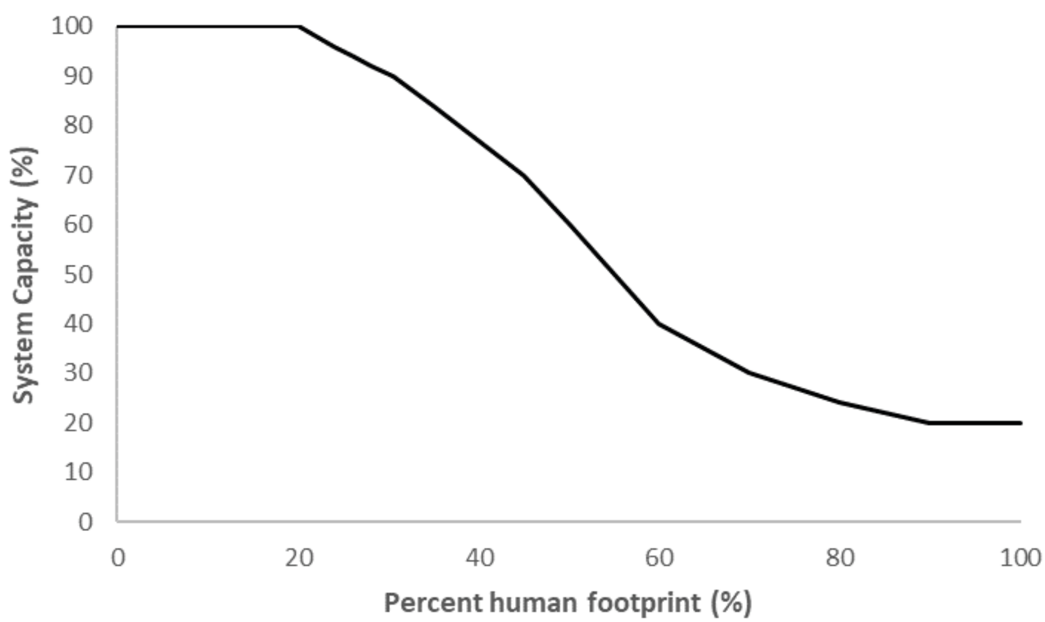


Data input: The percentage of water being withdrawn in the watershed of interest compared to natural low-flow discharge during February and August. This value was estimated using simple empirical relationships (Paul 2015, AEP, unpublished data) that relate mean annual discharge to the 90th exceedance flow (a measure of low flow) for either February or August and estimated water use derived from ALCES online©.

2. Flow regime: Modification of timing and frequency of peak-flow events

Removing forest cover and altering natural landscapes can result in changes in the magnitude and frequency of peak flow events which can impact the sustainability of fish populations. For instance, for trout species increased discharge during spring runoff and additional peak flow events throughout the year may result in downstream displacement of emerging fry (Ottaway and Clarke 1981) and also have negative effects on spring-spawning species that may be prey for trout (e.g., Seegrist and Gard 1972). Further, Jensen and Johnsen (1999) observed a negative correlation between year-class strength of two fall spawning salmonids and size of peak flood during the spring. There is also evidence that increased frequency of peak flow events can result in short and long term changes to river morphology that would impact trout, such as a reduction of habitat complexity and quantity of pool habitat (Lyons and Beschta 1983; Everest et al. 1985; Bonneau and Scarnecchia 1998) and the formation of an "oversized" channel. The potential for hydrologic change in watersheds was considered negligible when < 20% of the watershed was disturbed land (i.e., human footprint), low to moderate when 20–50% of the watershed was disturbed, and high when >50% of the watershed was disturbed (Figure 4). These thresholds are similar to Equivalent Clear-cut Area hazard categories recommended by Alberta Forestry and Agriculture (Stednick 1996; Guillemette et al. 2005; Mike Wagner pers. comm.). In the absences of other impacts, it was assumed that trout populations are resilient to a low degree of change and could persist, albeit at very low density, in watersheds where hydrologic change is high (Figure 4).

FIGURE 4. The hypothetical relationship between total human footprint area in a watershed and bull trout system capacity.



Data input: Total human footprint area (%) within the watershed unit of interest determined using ALCES Online ©.

3. Water quality: Phosphorus

Phosphorus is a major driver of primary production in aquatic ecosystems that affects other biotic and abiotic factors. Low-level inputs of phosphorus during oligotrophic stream fertilization projects in B.C. have resulted in increased fish size and abundance due to substantial increases in trophic productivity with limited impact to water quality (Koning et al. 1998). However, higher levels of nutrient inputs lead to stream eutrophication and degraded water quality, including reduced nocturnal dissolved oxygen in summer (Chung 2013; Jacobsen and Marin 2008) and overall anoxic conditions that can impair biodiversity (Meijering 1991). For example, degraded stream habitats and fish winterkill conditions in Alberta foothills were correlated with theoretical increases in phosphorus runoff due to land use at the watershed scale (Norris 2012).

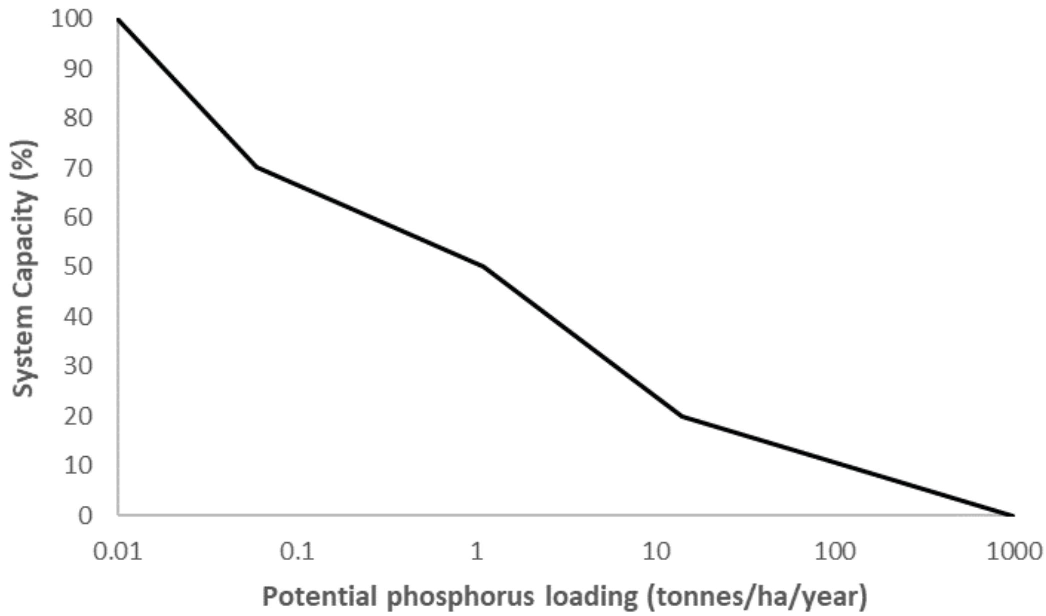
The phosphorus stressor-response curve was developed using potential phosphorous loading estimates and current Fish Sustainability Assessment (FSA) risk categories for bull trout (see MacPherson et al. 2019; AEP 2013) in 73 HUC 8 watersheds across the Alberta East Slopes. FSA risk categories vary from 0 (functionally extirpated) to 5 (very low risk). The potential phosphorous loading estimate for each watershed was obtained using ALCES© online. Phosphorus runoff was measured as a potential loading value for each watershed (tonnes/ha/year).

The stressor-response curve was derived by: a) using logistic regression to develop a statistical model relating probability of being within a given FSA category to the log-transformed phosphorous loading estimate; and, b) converting this statistical model into a stressor-response curve relating phosphorus to a percent reduction of bull trout system capacity from a pristine reference condition. Proportional-odds logistic regression was used since the response variable is a multinomial ordered variable (Venables and Ripley 2002). The proportional-odds assumption of independence among adjacent categories was assessed by comparing similarity of odds ratios among successive categories (Venables and Ripley 2002).

The stressor-response curve was derived from the proportional-odds logistic-regression models by estimating phosphorous loading estimates required for a 90% probability of falling within a given FSA category. This is similar logic to quantile regression (Cade and Noon 2003) that recognizes numerous unaccounted factors can be driving a response variable. FSA categories were converted to percent of reference condition using population percentages at transition points between adjacent FSA categories.

The FSA risk category for bull trout was rarely 3 or better when phosphorus loading potential was ≥0.1 tonnes/ha/year. No watersheds with a FSA risk category of 5 were observed. There was a significant phosphorus level effect (slope = -0.42, 95% profile confidence interval -0.65 to -0.21) with the probability of a watershed being within a lower FSA category increasing with phosphorus loading. From these probability distributions, the stressor-response for bull trout system capacity at a given phosphorus loading estimate was developed (Figure 5).

FIGURE 5. Stressor-response curve depicting the expected relationship between potential phosphorus loading (tonnes/ha/year) (log scale) and bull trout system capacity.



Data input: Total expected phosphorus export was calculated following the Event Mean Concentration method described in Donahue (2013) and is based on land cover type and annual precipitation within the natural region. Phosphorous runoff values were obtained from the Upper Bow River Basin Cumulative Effects Study (ALCES Group, 2012) and phosphorous delivery coefficients were obtained from Stelfox et al. (2008). Total estimated phosphorous export was calculated in ALCES Online© based on 2010 footprint within the spatial unit of interest.

4. Water quality: Sediment

Sedimentation can reduce the biological productivity of aquatic ecosystems and damage fish habitat (ASRD 2012). The amount of sediment a stream can transport is based on numerous factors including, but not limited to, precipitation, surface water transport, erosion, topography, geology and riparian vegetation (reviewed in Muck 2010). Anthropogenic disturbances (e.g., such as roads, Lachance et al. 2008) can produce substantial inputs of sediments into streams in excess of natural levels. These increased rates of sediment delivery can adversely affect bull trout habitat and have lethal and sub-lethal effects throughout trout life history from egg incubation to adulthood (reviewed in Muck 2010).

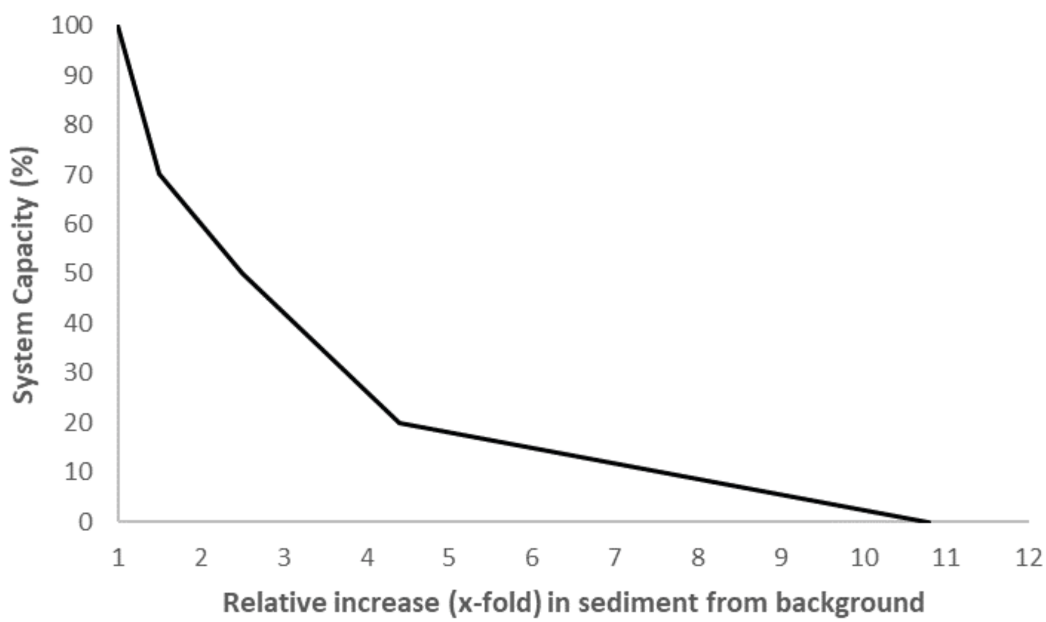
Potential impacts caused by excessive suspended sediments are varied, complex and often masked by other concurrent activities (Newcombe 2003), making it difficult to establish the specific effects of sediment impacts on fish (Chapman 1988). The sediment stressor-response curve was developed using sediment estimates obtained from ALCES online based on 2010 footprint and current FSA risk categories for bull trout (see MacPherson et al. 2019; AEP 2013) in 73 HUC 8 watersheds across Alberta. FSA risk categories vary from 0 (functionally extirpated) to 5 (very low risk). The dynamic pattern of sediment transport varies from watershed to watershed and aquatic ecosystems have adapted to the natural temporal and spatial pattern of this transport. As such, effects on fish from changes in sediment loading will be relative to natural conditions (Kemp et al. 2011). To capture relative change, sediment in the stressor-response curve was defined as the "sediment index" which is measured as potential sediment loading for 2010 (tonnes/ha/year) divided by potential sediment loading for 1910 (tonnes/ha/year).

The sediment index (2010 loading/1910 loading) stressor-response curves were derived by: a) using logistic regression to develop a statistical model relating probability of being within a given FSA category to the log-transformed sediment index; and, b) converting this statistical model into a stressor-response curve relating sediment to bull trout system capacity. Proportional-odds logistic regression was used as the response variable is a multinomial ordered variable (Venables and Ripley 2002). The proportional-odds assumption of independence among adjacent categories was assessed by comparing similarity of odds ratios among successive categories (Venables and Ripley 2002).

The stressor-response curve was derived from the proportional-odds logistic-regression models by estimating sediment index levels required for a 90% probability of falling within a given FSA category. This is similar logic to quantile regression (Cade and Noon 2003) that recognizes numerous unaccounted factors can be driving a response variable. FSA categories were converted to percent of reference condition using population percentages at transition points between adjacent FSA categories.

Compared to phosphorus, the data showed a much clearer separation among FSA categories with increasing sediment relative to background 1910 values. An adult FSA category of 4 existed in watersheds when relative sediment increases were negligible. FSA categories of 1 or 0 dominated when sediment increased more than 3 fold over background levels. Not surprisingly, there was a significant and strong negative sediment effect on association with FSA categories (slope = -2.8, 95% profile confidence interval –3.8 to -2.0). The stressor-response curve is shown in Figure 6.

FIGURE 6. Stressor-response curve depicting the expected relationship between relative increase in sediment loading from 1910 conditions and bull trout system capacity.



Data input: The sediment index is calculated as the total expected sediment export for 2010 divided by the total expected sediment export before substantial industrial activity (i.e., 1910). Total expected sediment export was calculated following the Event Mean Concentration method described in Donahue (2013) and is based on land cover type and annual precipitation within the natural region. Sediment runoff values were obtained from the Upper Bow River Basin Cumulative Effects Study (ALCES, 2012) and sediment delivery coefficients were obtained from Stelfox et al. (2008). Delivery coefficients for OHV trails were assigned a value of 6% based on the work of Welsh (2008). Total estimated sediment export was calculated in ALCES Online © within the spatial unit of interest.

5. Water quality: Collinearity of phosphorous and sediment

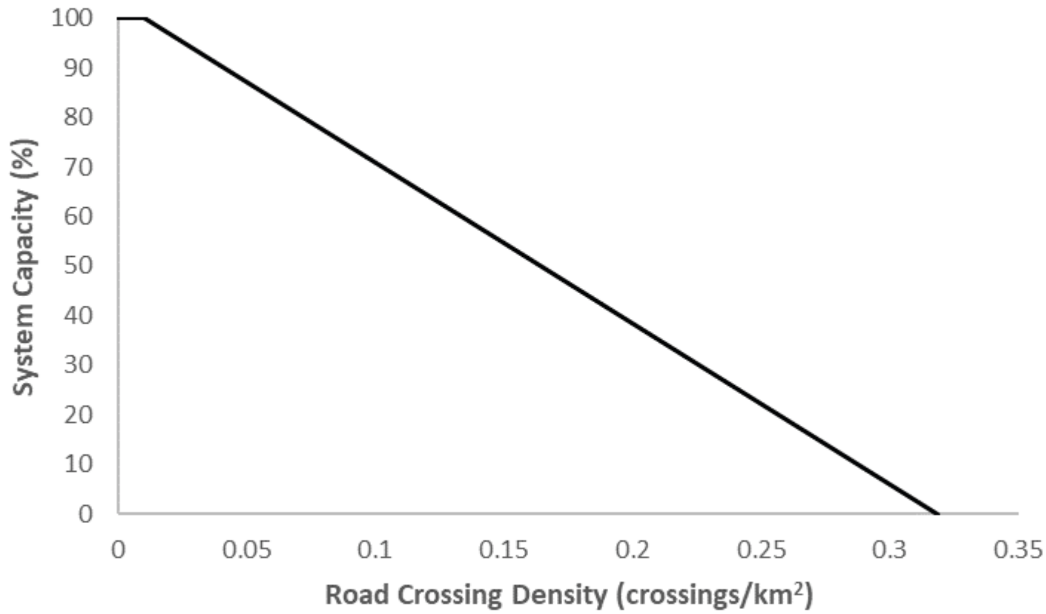
A major issue in assessing the importance of potential stressors in driving a response variable is collinearity amongst different stressors (Zuur et al. 2010). If different stressors are highly correlated, it is impossible to distinguish relative importance without further experimentation. There was a high degree of correlation between potential phosphorus loading (tonnes/ha/year) and the relative sediment increase across the 73 watersheds (Pearson R = 0.62, 95% confidence interval 0.45 – 0.74). Thus, it was difficult using the available data to separate the importance of phosphorus or sediment independently on bull trout system capacity. Our approach was to create two separate stressor-response curves (i.e., one for potential phosphorus loading independent of the sediment index and vice-versa) and acknowledge that the observed response could be driven by the other stressor. As the Joe model accumulates cumulative effects multiplicatively (additive on a proportional scale), treating these two curves separately would inappropriately overemphasize the expected response. To overcome this issue, we treated sediment and phosphorus in the Joe model using a limiting factor approach. Simply, only the strongest, negative response from either the phosphorus or sediment stressor-response curves is used to calculate final system capacity. Anytime a watershed shows either phosphorus or sediment to be a hypothesized key driver, it must be acknowledged that the other stressor (i.e., sediment or phosphorus, respectively) could be the driver given the collinearity.

6. Fragmentation: Road crossing structures

Bull trout are a migratory fish that require connectivity between key spawning, rearing, feeding, and overwintering habitats. Improperly installed road crossings can cause immediate and long-term effects on fish populations by altering habitat characteristics, fragmenting fish habitat and impeding fish movements necessary to complete life history processes (Warren and Pardew 1998; Gunn and Sein 2000; Harper and Quigley 2000; Morita and Yamamoto 2002; Park et al. 2008; Burford et al. 2009; MacPherson et al. 2012).

In the absence of a provincial road crossing status dataset, the assumption was that relatively high numbers of road crossings indicate a greater risk of habitat fragmentation. Audits of crossing structures in several northwestern Alberta watersheds reported that approximately half of assessed culverts were considered potential barriers to fish passage (Scrimgeour et al. 2003; Johns and Ernst 2007; Park et al. 2008). There is a paucity of studies directly measuring population-level impacts of fragmentation on bull trout specifically, although road density has been positively associated with reduced occupancy of the species (Ripley et al. 2005) and is correlated with road crossing densities within watersheds in the Alberta bull trout range (R^2=0.59, J. Reilly, pers. comm.). The hypothetical relationship between road crossing density and bull trout system capacity was determined following the risk threshold approach outlined in MacPherson et al. (2014) using the highest estimated road crossing density to indicate the greatest degree of extirpation risk (Figure 7).

FIGURE 7. Stressor-response curve depicting the expected relationship between road crossing density within a watershed and the system capacity of bull trout populations.



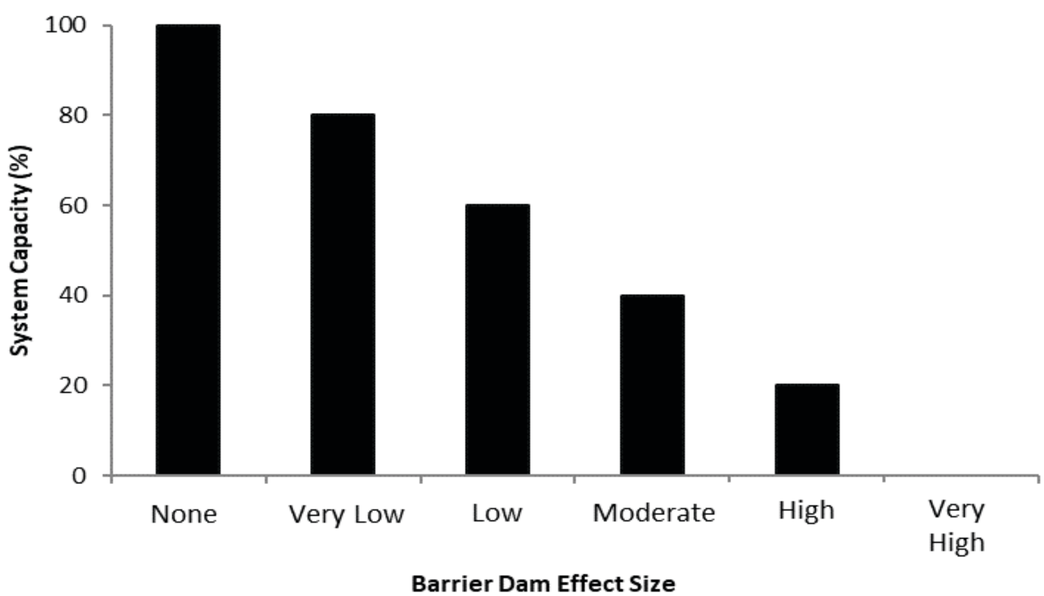
Data input: Number of road and stream intersections per watershed were estimated using the provincial road spatial layer, excluding winter roads and ferry crossings. Number of road and stream intersections per watershed were estimated using the provincial road spatial layer, excluding winter roads and ferry crossings. Only Order 2 and 3 streams were considered because Bull Trout occur infrequently in Order 1 streams and crossing structures used on streams Order 4 and greater are typically bridges that do not limit fish passage (e.g., Park et al. 2008).

7. Fragmentation: Large barrier dams

Dams can cause environmental changes such as alterations to water temperature, flow regime and sediment loading (Marmulla 2001). However, one of the most serious concerns for fish species is the fragmentation of once connected riverine habitats, as dams that are not equipped with fish passage facilities can act as complete barriers to upstream and downstream fish movements (Marmulla 2001; ASRD 2012). Many fish populations are likely sustained by metapopulation dynamics, where immigration from source populations ensures the persistence of surrounding subpopulations (Hanski 1998; Dunham and Rieman 1999). As a result, severing lotic connections may limit dispersal, and inhibit the recolonization of small isolated populations that can be at increased risk of extinction through demographic, environmental and genetic stochasticity (Lande 1998; Morita et al. 2009).While literature exploring population-level effects of dams is more plentiful for large migratory species such as salmon (e.g. Lawrence et al. 2016) and sturgeon (e.g. Jager et al. 2001), fewer studies have evaluated the effects of dams on stream-dwelling salmonids. Of available literature, Morita and Yokota (2002) found that extinction risk of white-spotted charr (*Salvelinus leucomaenis*) increased in small and isolated habitat fragments (Morita and Yamomoto 2002; Letcher et al. 2007). For cutthroat trout (*Onchorhynchus clarki*), Harig and Fausch (2002) noted that population success upstream of barriers was strongly linked to available habitat area. Letcher et al. (2007) simulated fragmentation scenarios in a stream resident brook trout (*Salvenlinus fontinalis*) metapopulation and found that extinction was likely unless immigration from above barriers was possible or fish exhibited demographic life history changes.

In Alberta, it is difficult to quantify the effects of fragmentation due to large dams on bull trout. The overall population level effect depends on a variety of factors such as patch size, habitat quality, and population size (Morita et al. 2009), as well as life history and behavioural traits (Letcher et al. 2007, Morita et al. 2009) and meta-population dynamics, and in most cases this information is not available for bull trout populations in the province. It is possible however, to provide a relative, qualitative estimate based on professional opinion considering available datasets. Several hypothetical situations will be described to elucidate this process. In the simplest case, if there are no dams in the system, then there is no effect. If a dam inhibits movement of migratory bull trout into a small portion of the lower watershed they once occasionally frequented (based on telemetry data), a low or very low effect size may be assigned. In contrast, a moderate effect would be assigned if the dam resulted in an isolated, resident bull trout population existing in a relatively small habitat patch, with no possible connections to other bull trout populations. Lastly, the severe decline or functional extirpation of migratory bull trout downstream of a dam soon after construction may indicate that suitable habitat to support all life history needs is no longer available (high to very high effect size). Until further information is available, the relationship between barrier dam effect size and sustainability of bull trout populations is assumed to be linear (Figure 8).

FIGURE 8. Stressor-response curve depicting the expected relationship between large barrier dams and the system capacity of bull trout populations.

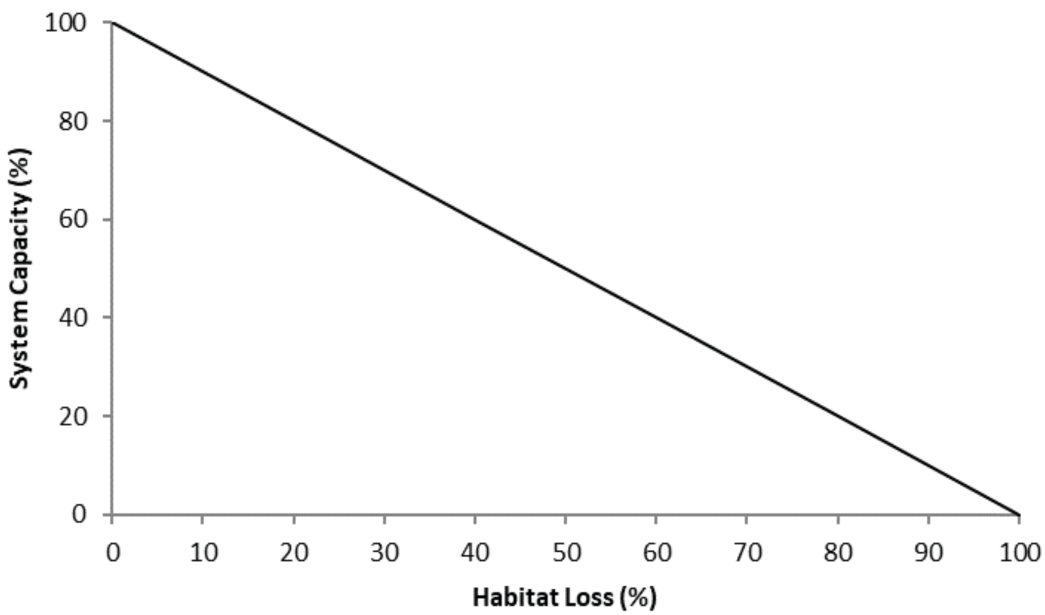


Data input: Qualitative estimate of barrier dam effect, based on expert opinion considering available data on bull trout movement, abundance, presence of life history type and change to bull trout populations before and after dam construction.

8. Direct habitat loss

Habitat loss and degradation is often cited as a major impact and limiting factor for fish populations (e.g., ASRD 2012). This stressor-response curve is exclusively meant to capture direct habitat loss. Direct habitat loss is defined as the removal of portions of a natural stream, or replacement of portions of a natural stream with a different landscape feature. For example, strip-mining for coal in parts of the bull trout range has deleted some stream sections, or has resulted in the replacement of streams sections with open-pit lakes or with channeled stream analogs (i.e., a ditch) that do not provide bull trout habitat. The stressor-response curve for habitat loss is depicted by a linear relationship between the percentage of stream habitat lost and system capacity (Figure 9).

FIGURE 9. Relationship between direct habitat loss and the effect on bull trout system capacity.



Data input: GIS-derived estimates of stream habitat lost or converted to different landscape features in the spatial unit of interest.

9. Competition by non-native species: brook trout, hybrids, lake trout

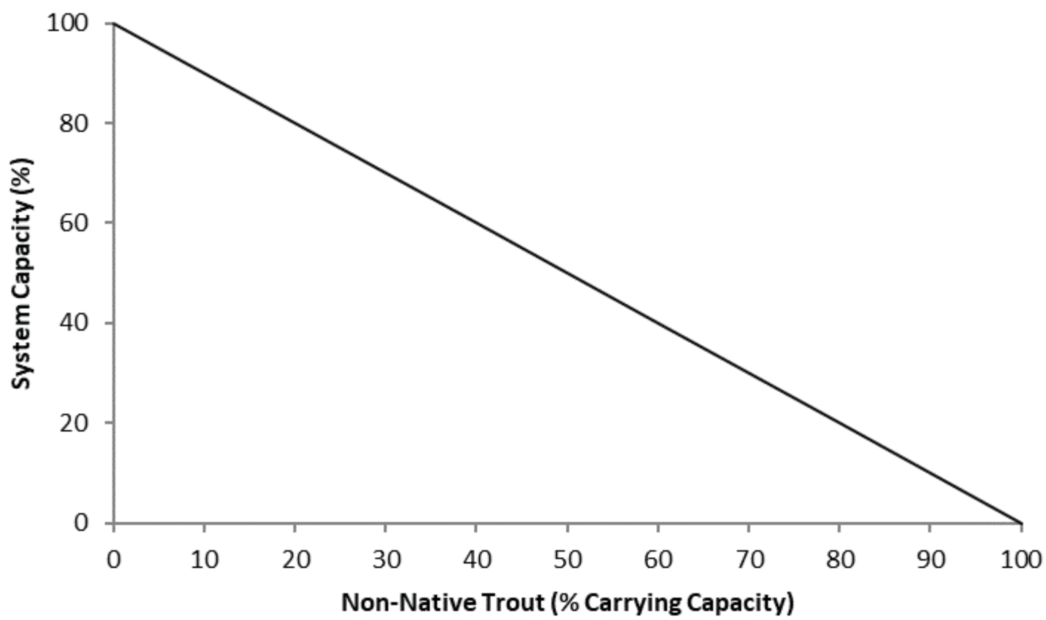
Brook trout is a wide-spread, invasive species that may compromise bull trout populations through competition (Warnock 2012, McMahon et al. 2007, Rieman et al. 2006). If successful, brook trout may displace or replace, native salmonids (Behnke 1992; Peterson et al. 2004; Fausch 2007; McGrath and Lewis Jr. 2007; Peterson et al. 2008; Earle et al. 2010a, b). Competition only occurs when resources are limited, or the system is near carrying capacity (Dunham et al. 2002b). Therefore, researchers should carefully examine available evidence to determine if brook trout are actually competing with bull trout, or if they are taking advantage of resources made available as a result of declining bull trout density due to other stressors (e.g., habitat changes, over-exploitation). Relatively high densities of brook trout may indicate that the system is near or at carrying capacity, and therefore, competition between brook trout and bull trout may be occurring.

Bull trout are known to hybridize with at least two other char species, brook trout (Kanda et al. 2002), and dolly varden (*Salvelinus malma*) (Baxter et al. 1997), though within Alberta hybridization has only been observed with brook trout (Earle et al. 2010a,b). Though bull trout -brook trout hybrids are fertile, they have not been observed to produce hybrid swarms in areas of overlapping range (Kanda et al. 2002), and hybridization itself is seen as less of a threat than competition with non-native species. Where hybridization occurs hybrid individuals will compete with bull trout for space and resources, similarly to brook trout, though the impacts are generally minor given the small proportion of the total fish community they comprise.

Populations of lake trout and bull trout rarely co-exist. Past introductions of lake trout have followed a pattern of dispersal, colonization, and establishment in natural lake systems and reservoirs followed by declines or extirpation of the bull trout population (ASRD 2012, Donald and Alger 1993, Fredenberg 2002). While the mechanisms of lake trout invasiveness have not been fully explored, it is assumed that niche overlap and direct competition result in the displacement or exclusion of bull trout. Where bull trout are sympatric with lake trout, they are generally relegated to a minor component of the fish community.

The stressor-response curve to depict the impacts of competition by non-native species on bull trout is a simple linear relationship between the ability of a system to hold adult bull trout (system capacity) and the capacity used by brook trout, hybrids, and lake trout (carrying capacity). Bull trout system capacity and non-native/hybridized trout carrying capacity must add to 100% (Figure 10).

FIGURE 10. Stressor-response curve depicting the expected relationship between non-native trout carrying capacity within a watershed and the system capacity of bull trout populations.



Data input: Competition will be considered as occurring in stream reaches (order ≥3) which once held or continue to hold bull trout, but now contain predominately brook trout or hybrids at densities that suggest the system is at or near carrying capacity (estimated at approximately 80 brook trout or hybrids /300m, but can be dependent on stream size). Stream electrofishing survey data from FWMIS identified locations with high brook trout and hybrid densities and then expert opinion was used to estimate the stream distance upstream and downstream from this location containing high brook trout and hybrid densities. Hybrids were determined strictly using external diagnostic features, as genetic information is extremely limited.

Competition with lake trout was considered as occurring in lakes and reservoirs that once held or continue to hold bull trout, but now contain predominately lake trout at densities that suggest the system is at or near carrying capacity based on expert opinion. There are relatively few HUC10 watersheds where lake trout densities suggest competition may be occurring; these include the Middle Waterton River, Spray River and Abraham Lake watersheds.

The input value was the percentage of habitat occupied by brook trout, hybrids and lake trout at hypothesized carrying capacity relative to the available bull trout habitat within a watershed.

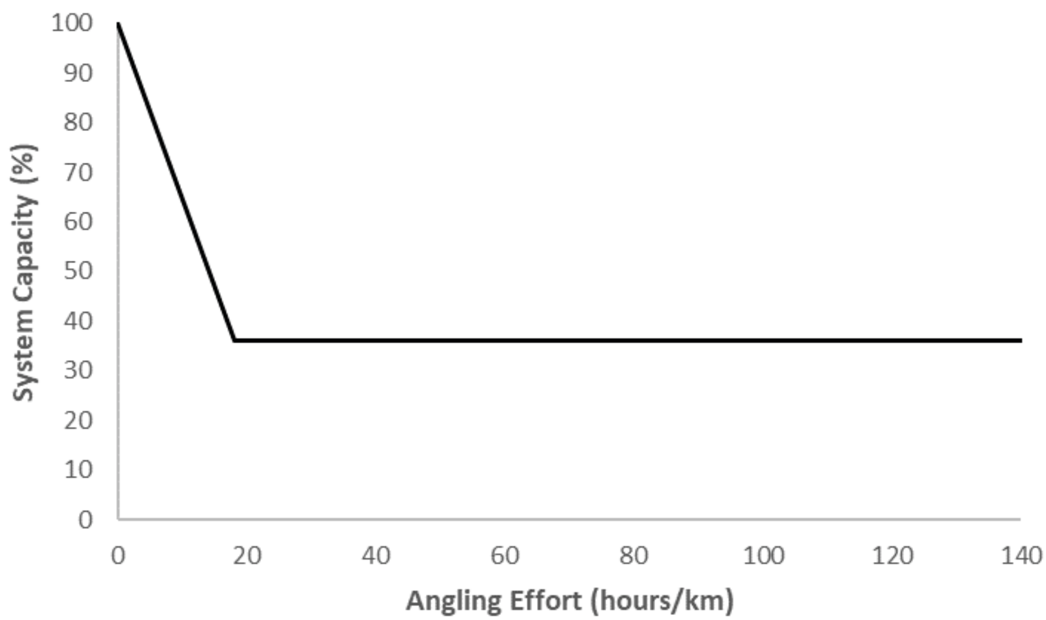
10. Incidental angling mortality and illegal harvest

Bull trout were legally harvested throughout the eastern slopes prior to the implementation of the province-wide zero harvest regulation in 1995 (ASRD 2012). Angling may still represent a major threat to population sustainability due to incidental mortality (i.e., mortality due to stress or physical damage from hooking or handling) associated with catch and release fishing if angling effort and catchability is high (e.g., Post et al. 2003). Illegal harvest, either intentional or due to misidentification, occurs at an unknown frequency and may also contribute to population declines. Past case studies demonstrate that some bull trout populations are capable of recovering relatively quickly (5-10 years) from an over-exploited state under zero harvest regulations/reduced angler effort or complete angling closures (Johnston et al. 2007; Sullivan 2014).

In the absence of information on population parameters, such as catchability and abundance, angling effort (annual rod hours per km) was used as the stressor input value. Angling effort is considered a surrogate for fishing mortality; it is assumed that there is greater mortality at high levels of effort, and lower mortality and low levels of effort. The approximated relationship between angling effort and system capacity encompasses a linear decline in system capacity with increasing angler effort until an inflection point is reached where additional effort has no further impact on system capacity. The assumption is that there is a proportion of fish in a population that are less vulnerable or invulnerable to anglers, resulting in population resiliency albeit at lower abundance, i.e., fishing mortality alone is not expected to extirpate a population. This may occur because of reduced or variable catchability due to learned hook avoidance, heterogeneity among individual fish, and environmental factors (Askey et al. 2006, van Poorten & Post 2011). Additionally, some fish may not be caught because anglers are unable to access all areas of a river system. These inaccessible areas provide a similar function to marine protected areas by reducing the proportion of the total population available to fishing (Cox and Walters 2002).

The specific shape of the stressor-response curve for bull trout was based on a series of assumptions. First, we assumed that there was no impact to system capacity at zero angling effort (0 annual rod hours/km, 100% system capacity). Next, the x-coordinate of the inflection point was set at 18 annual rod hours/km, based on Post et al. (2003) that predicted that angling effort above this value would not maintain a viable population of bull trout under catch-and-release regulations. The y-coordinate of the inflection point was set as the 95th percentile of system capacity, calculated using all populations experiencing angling effort above the critical value. The 95th percentile was applied because it is expected that the relationship between system capacity and angling effort is better represented by the rate of change near the maximum response rather than the mean response, as system capacity is affected by multiple factors that may vary in their effect. This is similar logic to quantile regression that recognizes numerous unaccounted factors can be driving a response variable (Cade and Noon 2003). This produced inflection point coordinates of 18 annual rod hours/km and 36% system capacity for bull trout (Figure 11).

FIGURE 11. Stressor-response curve depicting the expected relationship between annual angler effort (hours/km) within a watershed and the system capacity of bull trout populations.



Data input: The amount of angler effort was summarized as annual angling hours/km. When available, creel information was used to determine angling effort. In the absence of creel data, a model that incorporated the annual temporal and spatial availability of angling opportunities for bull trout was used. This was achieved by estimating the hours of angling on weekdays and weekends for relevant lengths of bull occupied streams (km) in a watershed. This was then summarized as an annual total by accounting for the number of weeks in an angling season.

11. Mortality

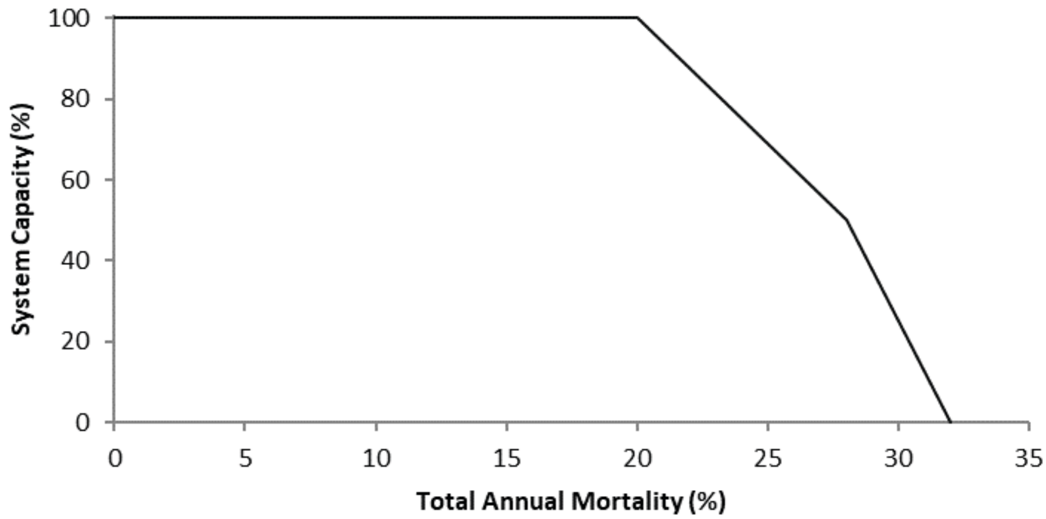
In the bull trout Joe model, direct mortality was separated into natural causes, entrainment and research and monitoring, although more variables can be added as required. Using these three mortality sources, the total annual mortality rate (A) can be calculated using the conditional rates of natural mortality (n), entrainment mortality (en) and research and monitoring mortality (r), by applying the following equation adapted from Ricker (1975):

$$A=1-[(1-n)\times(1-en)\times(1-r)]$$

The stressor-response curve for direct mortality (Figure 12) is based on the results from modelling using a modified version of the bull trout model of Post et al. (2003). Assuming a conditional mortality rate of 20% from natural causes (Post et al. 2003) a bull trout population was shown to switch from growth overfishing to recruitment overfishing (assumed to occur at ½ of maximum system capacity) if the combined conditional rate of mortality from other sources exceeded 8% and extirpation was expected when additional mortality exceeded 12%.

Classification: Public

FIGURE 12. Stressor-response curve depicting the expected relationship between total annual mortality and the system capacity of bull trout populations.



Entrainment mortality

Bull trout can become entrained in irrigation canal headworks and killed if not rescued before the canal is dewatered at the end of the irrigation season. Entrainment rates are expected to be variable between canals, however, there have been no recent studies to determine the total number of entrained bull trout and the overall effect on population sustainability. The primary data source to inform the potential severity of this threat is the Trout Unlimited Canada annual fish rescue program, which includes most but not all canal headworks within the current bull trout range. Typically, no or small numbers (<10) of entrained bull trout are rescued (Lindsey et al. 2015); however, it should be noted that the rescues are not designed to estimate entrainment rates so these numbers should be viewed as minimum values only. In contrast, entrainment at the Belly River canal prior to screening to exclude large fish was estimated at 15 – 20% of annual mortality (Clayton 2001). The majority of canal headworks within Alberta's eastern slopes do not have fish exclusion devices.

Bull trout can also become entrained in powerhouses for hydroelectric reservoirs where a portion are killed as they pass through the turbines. Various aspects of bull trout entrainment in hydroelectric reservoirs have been widely studied in both the U.S. and British Columbia (B.C.) (Martins et al. 2013, Ma et al. 2012, Underwood and Kramer 2007, Salow and Hostettler 2004, and FERC 1995). Entrainment and mortality rates are highly site-specific, varying with physical factors including reservoir size, dam height, fore bay configuration, depth of intake, turbine type and operational timing as well as biological factors including fish size, seasonal and diurnal movements and density-dependent influences on fish movement. While few generalizations can be made, entrainment rates of adult, sub-adult or juvenile bull trout can and do impact populations typically as annual losses (i.e., direct morality and permanent loss to downstream reaches) of <5%. For example: 1) Kinbasket Reservoir, B.C., adult bull trout–3.4% loss (Martins et al. 2013); 2) Arrowrock Reservoir, ID, adult bull trout–4% loss, 11% loss when drawn down for maintenance (Salow and Hostettler 2004), and 3) Rimrock Reservoir, WA, sub-adult bull trout–1.4% loss (Underwood and Cramer 2007). Substantially greater mortality rates (9 – 42%, size dependent) are anticipated for the Peace/Halfway River bull trout population during the operation of the Site C Dam in B.C. (Ma et al. 2012).

Data input: In general, watersheds containing irrigation canal headworks were assigned an entrainment conditional mortality rate of 1% and those containing hydroelectric dams were assigned a rate of 4%, unless other data were available.

Research and monitoring

Standard scientific methods for monitoring bull trout populations typically involve the non-lethal capture, handling and release of individual fish. Methods used to capture bull trout include electrofishing, angling, trapping and netting, with backpack and boat electrofishing being the most widely used in Alberta. After capture, fish are held for processing, often anesthetized, and measured. Depending on the project objectives, fish may also be marked (tagged), surgically implanted with telemetry transmitters, and/or have a small portion of a fin removed for genetic analysis. Lethal sampling of bull trout is uncommon, but may occur if information that cannot be collected using non-lethal means (e.g., maturity and age data) is required for management and assessment purposes. In these cases, the potential impacts on population sustainability are thoroughly reviewed by the appropriate regulatory agency(s) prior to project approval.

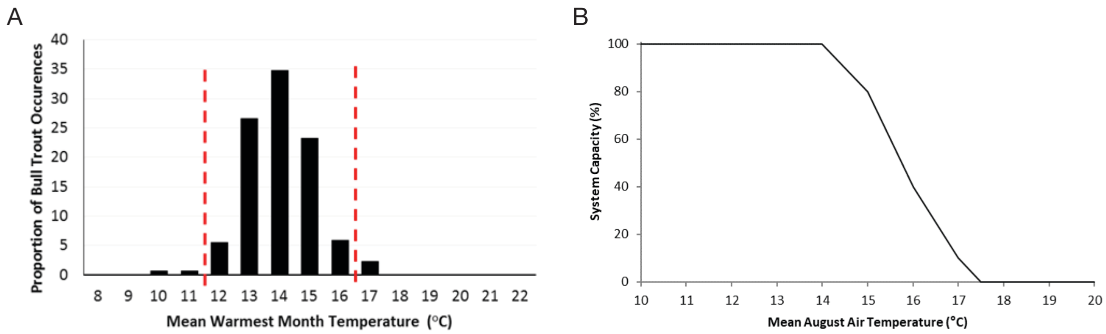
Alberta Fisheries Management has developed a series of standards including the Standard for the Ethical Use of Fish in Alberta (AESRD 2013a), Standard for Sampling Small Streams in Alberta (AESRD 2013b) and Electrofishing Policy Respecting Injuries to Fish (AFMD 2004) in order to minimize fish injury, stress and mortality during non-lethal collection and handling by research crews. These standards are included as conditions on Research Licences, which are mandatory licences issued to all agencies and organizations conducting fisheries-related work in the province. Research Licences also include a section detailing Best Management Practises relating to the processing of fish in cold and hot weather, proper handling techniques, and the use of anaesthetic. While the application of standards and best management practices does minimize fish injury, stress and mortality, some incidental mortality during fish collection and handling may occur. Incidental mortality is assumed to have negligible to very low population-level effects because the majority of bull trout surveys are limited to small representative areas of a watershed and project time periods are typically short (1-5 years). Therefore, mortality due to scientific research and monitoring will not be included in the total annual mortality calculation unless there is evidence of a population-level impact within a particular watershed. Similarly, the U.S. Fish and Wildlife Service analyzed the effects of scientific research through a biological opinion survey (USFWS 2000) and determined that scientific collection does not jeopardize bull trout populations, and is therefore not identified as a threat factor in the U.S. Bull Trout recovery plan (USFWS 2014).

Data input: Conditional rate of mortality due to research and monitoring was set to 0% unless other data were available. Values will be adjusted if new information becomes available suggesting otherwise.

12. Natural limitations: Temperature

Bull trout is a thermally sensitive species vulnerable to increased water temperature resulting from land disturbance and climate change (ASRD 2012). The thermal characteristics of bull trout habitat in Alberta were explored by comparing mean warmest month temperature (MWMT) derived using the program Climate WNA© (Hamann and Wang 2005; Wang et al. 2016) to all locations where bull trout have been captured between 1946–2013 (FWMIS query, Nov. 2013; Figure 13A). Air temperature was used in this analysis because there is currently no province-wide water temperature dataset or model available. In addition, air and water temperatures are typically correlated over time scales >1 week (Mohseni et al. 1998). The minimum and maximum air temperature thresholds (10°C and 17°C; Figure 13A) were similar to those reported in previous laboratory and field studies investigating the effects of water temperature on bull trout growth and survival (Selong et al. 2001) and occupancy (Dunham et al. 2003; Wenger et al. 2011). The rapid decline in the number of occurrences between 10°C to 13°C is likely due to sampling bias (i.e., there are fewer sampling events in cold, high-elevation areas that are difficult to access). The findings of this analysis were used to inform the shape of the stressor-response curve below, which characterizes the expected influence of warm temperatures on the system capacity of bull trout populations (Figure 13B).

FIGURE 13. A) Thermal range of occupied bull trout waters within historic bull trout range. B) Stressor-response curve informed using thermal range data. This curve depicts the expected relationship between relatively high air temperatures and system capacity of bull trout. Potential influence of cold temperature is not included in this curve.

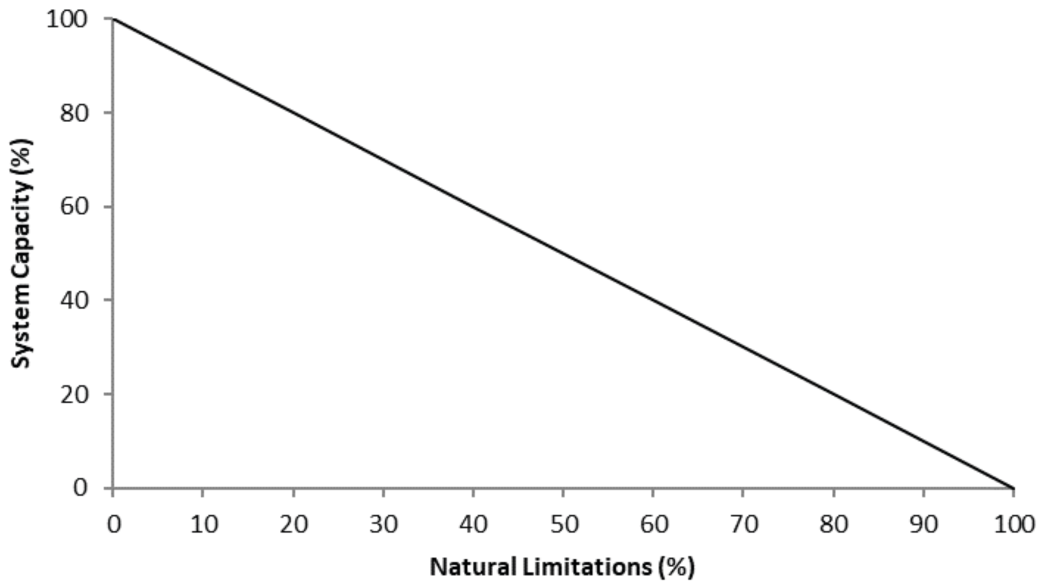


Data input: Mean August air temperature (°C) provided by ALCES Online© within the spatial unit of interest. Mean August air temperature (from ALCES Online©) and MWMT (from ClimateWNA) are expected to be highly correlated because the highest air temperatures in Alberta typically occur in August.

13. Natural limitations: Other

Bull trout are naturally limited by other environmental variables besides water temperature. These limitations occur at varying spatial scales and include both biotic (e.g., productivity, fish community, etc.) and abiotic features (amount of woody debris, substrate composition, water velocity, groundwater input, natural sediment loads, natural flow regimes, etc.). Further, persistence in suitable, but isolated habitats may be limited if connectivity with neighbouring populations is naturally restricted and supplementation or re-colonization following local population decline or extirpation is not possible. These naturally limiting factors mean that a specific bull trout population may never and would not be expected to achieve the reference system capacity of 100%, which is set using the most abundant, pristine bull trout populations in the province. Understanding the influence of natural limitations on bull trout populations today is difficult due to the confounding spatial and temporal effects of human disturbance and harvest. However, the relative status of bull trout prior to extensive human disturbance and harvest (i.e., early 1900s) was assessed by the Historic Adult Density (HAD) FSA score, which will be used as a surrogate measure of natural limitations within each watershed in the species historic range. The HAD score is ranked from 1 (very low bull trout density) to 5 (very high bull trout density), but for this purpose was converted to a percentage-based natural limitations value. It was assumed there is a linear relationship between system capacity and natural limitations (Figure 14).

FIGURE 14. Stressor-response curve depicting the expected relationship between natural limitations and the system capacity of bull trout populations.



Data input: The natural limitations % was based on Historic Adult Density FSA score within the spatial unit of interest (i.e., HAD 1 = 10%, HAD 2 = 35%, HAD 3 = 60%, HAD 4 = 85%, and HAD 5 = 100%). There is no historic (i.e., >100 years) fisheries survey data; therefore, anecdotes, photographs, local environmental knowledge (LEK), traditional environmental knowledge (TEK) and information on fish barriers were used to inform HAD FSA scores. This included historical accounts from warden reports and angler interviews. Please note, these natural limitations rankings should only reflect limitations other than temperature.

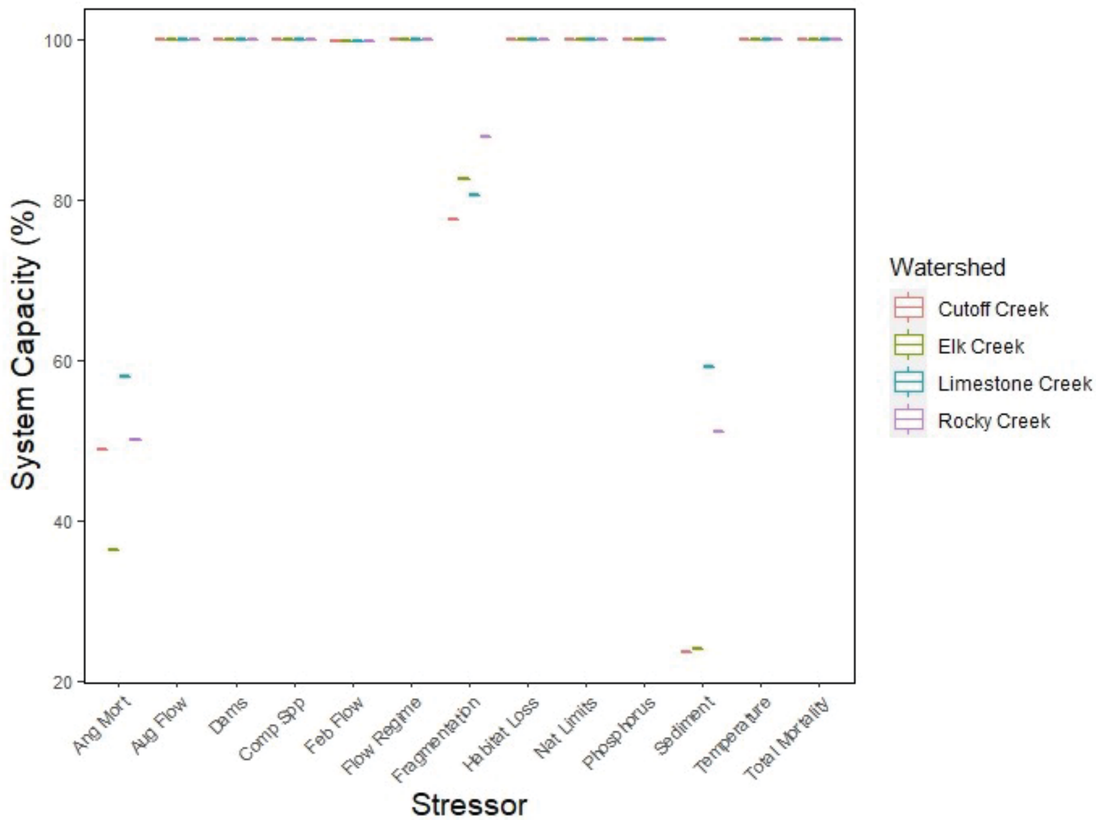
Bull trout cumulative effects model study watershed input values

All input values for each of the four study watersheds were summarized (Table 1) and run in Alberta Environment and Parks (AEP) Shiny interactive web-based app for the bull trout Joe model (https://fw-habitat-aep.shinyapps.io/BLTR_Dashboard/). We expect there to be continued improvements to the features of this interactive web application through time.

Results

The overall system capacity values and main threats for the four study watersheds were comparable (Figure 15). For the Rocky Creek bull trout population, the overall system capacity was 22%, and the putative, major threats were sedimentation, small stream fragmentation and angling mortality. Similarly, the overall system capacity values for Elk Creek, Limestone Creek and Cutoff Creek were 7%, 27%, and 9%, respectively, with sedimentation, small stream fragmentation and angling mortality identified as the main limiting factors. Bull trout populations in all watersheds are thought to be naturally limited by factors other than water temperature (Figure 15).

FIGURE 15. Output of a Joe model depicting impact to system capacity (%) of each threat impact category. The bars at the top of the graph suggest little to no effect on the population of interest, and the lowest bars suggest the strongest negative effect. Stressors are Ang Mort (angling mortality), Aug Flow (August flow), Comp Spp (competing species), Dams (barrier dams), Feb Flow (February flow), Fragmentation (small stream fragmentation), Habitat Loss, Flow Regime (modification of timing and frequency of peak-flow events), Nat Limits (other natural limitations), Phosphorus, Sediment, Temperature, and Total Mortality. A small amount of jitter was added to the x axis to visualize points for all stressors.



Discussion

The major threats to bull trout populations in all four Clearwater River watersheds were sedimentation, small stream fragmentation and angling mortality.

Increased rates of sediment delivery from anthropogenic sources can adversely affect bull trout habitat and have lethal and sub-lethal effects throughout trout life history (reviewed in Muck 2010). Incidental catch and release mortality and illegal harvest, either intentionally or through misidentification, may still be sufficiently high to limit population recovery (Joubert et al. 2020) if angling effort and catchability is also high (e.g., Post el al. 2003). Additionally, poorly constructed or maintained road crossing structures can fragment fish habitat and impede fish movements necessary to complete life history processes (Warren and Pardew 1998; Gunn and Sein 2000; Harper and Quigley 2000; Morita and Yamamoto 2002; Park et al. 2008; Burford et al. 2009; MacPherson et al. 2012).

Within the Rocky Creek watershed, a primary source of sedimentation and access to the fishery is provided by an extensive network of Off Highway Vehicle (OHV) designated and undesignated trails. The effects from OHV use can include soil compaction, erosion and displacement, vegetation loss and destruction, and degraded water quality due to sediment inputs (e.g. Marion et al. 2014, Graziano 2018). For instance, in Colorado, Welsh (2008) compared sediment delivery along native surface forestry roads and OHV trails and found that sediment production was six times higher on OHV trail segments than road segments largely because they are often found in valley bottoms. While a healthy stream could buffer from infrequent OHV disturbances (Meyer 1997), frequent OHV stream crossings may be problematic. In Rocky Creek, there were 31 highly eroded stream crossings within a 20 km section of stream and no crossing structures or engineering to mitigate sedimentation; in some reaches, crossing density exceeded 5 crossings/km. Rolling back the OHV trail network would also remove all motorized access to the majority of the Rocky Creek watershed, likely reducing angling effort and therefore incidental mortality. The opportunity for catch and release fishing for bull trout would still be available via foot-access. Although fragmentation was identified as a potential limiting threat, field inspection of culverts demonstrated that small stream fragmentation was not an issue at identified road crossings and did not need to be addressed.

Overall, the results of the Joe modelling process indicated that habitat remediation of the OHV trail network would address the main factors limiting bull trout in Rocky Creek. Additionally, the similarities between overall system capacity values and main threats supported the use of bull trout populations in Elk Creek, Limestone Creek and Cutoff Creek as controls in a monitoring program to evaluate if recovery actions improved the Rocky Creek bull trout population.

TABLE 1. Bull trout cumulative effects Joe model input values for the four Clearwater basin watersheds (Elk, Limestone, Cutoff and Rocky Creeks).

Watershed	Area (km²)	Mean August Air Temperature (°C)	Other Natural Limitations (%)	Angling Effort (annual hours/km)	Entrainment Mortality	Research Mortality	Natural Mortality	Phosphorus Loading (tonnes/ha/yr)	Relative Sediment Increase	February Flow (% withdrawn)	August Flow (% withdrawn)	Total Footprint (%)	Brook Trout (%)	Hybrid Brook Trout x Bull Trout (%)	Lake Trout (%)	Road Crossing Density (crossings/km²)	Barrier Dam Effect	Habitat Loss (%)
Cutoff Creek	88.2	11.7	100	14.4	0	0	0.2	0.0038	4.2	0.18	0.03	1.62	0	0	0	0.07	0	0
Elk Creek	94.4	11.1	100	30.9	0	0	0.2	0.0046	4.1	0.26	0.05	1.72	0	0	0	0.02	0	0
Limestone Creek	100.1	11.4	100	11.9	0	0	0.2	0.0024	2.0	0.32	0.06	0.64	0	0	0	0.1	0	0
Rocky Creek	62.7	11.5	100	14.1	0	0	0.2	0.0010	2.4	0.33	0.06	0.54	0	0	0	0.02	0	0

Literature cited

Alberta Environment and Parks (AEP). 2013. Bull Trout Fish Sustainability Index. Alberta Fish and Wildlife Policy Branch, Edmonton, Alberta. https://www.alberta.ca/bull-trout-fsi.aspx.

Alberta Sustainable Resource Development (ASRD). 2012. Bull Trout Conservation Management Plan 2012-17. Alberta Sustainable Resource Development, Species at Risk Conservation Management Plan No. 8, Edmonton, AB, 90 p.

Alberta Environment and Sustainable Resource Development (AESRD). 2013a. Standards for the ethical use of fishes in Alberta. 5 p.

Alberta Environment and Sustainable Resource Development (AESRD). 2013b. Standard for sampling of small streams in Alberta. 18 p.

Alberta Fisheries Management Division (AFMD). 2004. Electrofishing Policy Respecting Injuries to Fish. 3 p.

Baxter, J.S., E.B. Taylor, R.H. Devlin, J. Hagen, and J.D. McPhail. 1997. Evidence for natural hybridization between Dolly Varden (*Salvelinus malma*) and Bull Trout (*Salvelinus confluentus*) in a northcentral British Columbia watershed. Canadian Journal of Fisheries and Aquatic Sciences 54(2): 421-429.

Behnke, R.J. 2002. Trout and salmon of North America. The Free Press, New York. 359 p.

Bonneau, J.L., and D.L. Scarnecchia. 1998. Seasonal and diel changes in habitat use by juvenile Bull Trout (Confluentus salvelinus) and cutthroat trout (Oncorhynchus clarki) in a mountain stream. Canadian Journal of Zoology 76:783-790.

Burford, D.D., T.E. McMahon, J.E. Cahoon, and M. Blank. 2009. Assessment of trout passage through culverts in a large Montana drainage during summer low flow. North American Journal of Fisheries Management 29: 739–752.

Cade, B.S. and B.R. Noon. 2003. A gentle introduction to quantile regression for ecologists. Frontiers in Ecology and the Environment 1: 412–420.

Chapman, D. W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society 117: 1-21.

Chung, C. 2013. Diel oxygen cycles in the Bow River: Relationships to Calgary's urban nutrient footprint and periphyton and macrophyte biomass. MSc Thesis, University of Calgary, AB. 92 pp.

Clayton, T.B. 2001. Movements and status of Bull Trout (*Salvelinus confluentus*) in the Belly River, Alberta and Montana. Pages 141-145 in Brewin, M.K., A.J. Paul, and M. Monita, editors. Bull Trout II conference proceedings. Trout Unlimited Canada, Calgary, Alberta, Canada.

Donahue, W.F. 2013. Determining Appropriate Nutrient and Sediment Loading Coefficients for Modelling Effects of Changes in Landuse and Landcover in Alberta Watersheds. Water Matters Society of Alberta, Canmore, AB. 52pp.

Donald, D.B., and D.J. Alger. 1993. Geographic distribution, species displacement and niche overlap for lake trout and Bull Trout in mountain lakes. Canadian Journal of Zoology 71:238-247.

Dunham, J., S.B. Adams, R.E. Schroetr, and D.C. Novinger. 2002. Alien invasions in aquatic ecosystems: Toward an understanding of brook trout invasions and potential impacts on inland cutthroat trout in western North America. Reviews in Fish Biology and Fisheries, 12: 373-391.

Dunham, J.B. and B.E. Rieman. 1999. Metapopulation structure of bull trout: influence of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9:642–655.

Dunham, J., B. Rieman, and G. Chandler. 2003. Influences of temperature and environmental variables on the distribution of Bull Trout within streams at the southern margin of its range. North American Journal of Fisheries Management, 23:894-904.

Earle, J.E., A.J. Paul and J.D. Stelfox. 2010a. Quirk Creek population estimates and one-pass electrofishing removal of Brook Trout – 2009. Unpublished report, Fish and Wildlife Division, Alberta Sustainable Resource Development, Cochrane, Alberta.

Earle, J.E., J.D. Stelfox and B.E. Meagher. 2010b. Quirk Creek Brook Trout suppression project – 2009. Unpublished report, Fish and Wildlife Division, Alberta Sustainable Resource Development, Cochrane, Alberta.

Everest, F.H., N.B. Armantrout, S.M. Keller, W.D. Parante, J.R. Sedell, T.E. Nickelson, J.M. Johnston, and G.N. Haugen. 1985. Salmonids. In Management of wildlife and fish habitats in forests of western Oregon and Washington Edited by E.R. Brown. USDA Forest Service, Portland, Oregon, 199–230.

Fausch, K. D. 2007. Introduction, establishment and effects of non-native salmonids: considering the risk of rainbow trout invasion in the United Kingdom. Journal of Fish Biology 71:1-32.

Federal Energy Regulatory Commission (FERC). 1995. Preliminary assessment of fish entrainment at hydropower projects, a report on studies and protective measures, volumes 1 and 2 (appendices). FERC Office of Hydropower Licensing, Washington, D.C. Paper No. DPR-10. June 1995 (volume 1) and December 1994 (volume 2).

Fredenberg, W. 2002. Further evidence that lake trout displace Bull Trout in mountain lakes. Intermountain Journal of Sciences, Vol. 8. No.3:143 -152.

Graziano, S.J. 2018. Impacts of off-road vehicle (ORV) stream crossings: turbidity, sediment distribution, and management. M.Sc thesis. Alaska Pacific University, Anchorage, Alaska.

Government of Alberta (GOA). 2015. Fish Conservation and Management Strategy for Alberta. Fish and Wildlife Division, Edmonton, Alberta. 56 p.

Guillemette, F., A.P. Plamondon, M. Prévost and D. Lévesque. 2005. Rainfall generated stormflow response to clearcutting a boreal forest: peak flow comparison with 50 world-wide basin studies. Journal of Hydrology 302:137-153.

Gunn, J.M, and R. Sein. 2000. Effects of forestry roads on reproductive habitat and exploitation of lake trout (*Salvelinus namaycush*) in three experimental lakes. Canadian Journal of Fisheries and Aquatic Sciences 57: 97–104.

Hamann, A., and T. Wang. 2005. Models of climatic normals for genecology and climate change studies in British Columbia. Agricultural and Forest Meteorology 128:211–221.

Hanski, I. 1998. Metapopulation dynamics. Nature 396:41-49.

Harig, A.L. and K.D. Fausch. 2002. Minimum habitat requirements for establishing cutthroat trout populations. North American Journal of Fisheries Management 12: 535–551.

Harper, D.J., and J.T. Quigley. 2000. No net loss of fish habitat: an audit of forest road crossings of fish-bearing streams in British Columbia, 1996–1999. Canadian Technical Report of Fisheries and Aquatic Sciences 2319: 43 pp.

Hatfield, T., and A.J. Paul. 2015. A comparison of desktop hydrologic methods for determining environmental flows. Canadian Water Resources Journal, 40:303-318.

Hedrick, R.P., T.S. McDowell, K. Mukkatira, M.P. Georgiadis, and E. MacConnell. 1999. Susceptibility of selected inland salmonids to experimentally induced infections with *Myxobolus cerebralis*, the causative agent of whirling disease. Journal of Aquatic Animal Health 11(4): 330-339.

Jacobsen, D., and R. Marin. 2008. Bolivian Altiplano streams with low richness of macroinvertebrates and large diel fluctuations in temperature and oxygen. Aquatic Ecology 42:643-656.

Jager, H.I., J.A. Chandler, K.B. Lepla, and W.V.Winkle. 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. Environmental Biology of Fishes 60:347-361.

Jensen, A.J., and B.O. Johnsen. 1999. The functional relationship between peak spring floods and survival and growth of juvenile Atlantic Salmon (*Salmo salar*) and Brown Trout (*Salmo trutta*). Functional Ecology 13:778-785.

Johns, T., and T. Ernst. 2007. Culvert crossings as potential barriers to fish movement in the Kakwa River Watershed, Alberta. Alberta Conservation Association, Peace River, AB.

Johnston, F. D., J. R. Post, C. J. Mushens, J. D. Stelfox, A. J. Paul, and B. Lajeunesse. 2007. The demography of recovery of an overexploited Bull Trout, *Salvelinus confluentus*, population. Canadian Journal of Fisheries and Aquatic Sciences 64:113–126.

Joubert, B.A., M.G. Sullivan, B.C. Kissinger, and A.T. Meinke. 2020. Can smartphones kill trout? Mortality of memorable-sized Bull Trout (*Salvelinus confluentus*) after photo-releases. Fisheries Research 223:105458.

Kanda, N., R.F. Leary, and F.W. Allendorf. 2002. Evidence of introgressive hybridization between Bull Trout and Brook Trout. Transactions of the American Fisheries Society 131(4): 772-782.

Kemp, P., D. Sear, A. Collins, P. Naden, and I. Jones. 2011. The impacts of fine sediments on riverine fish. Hydrological Processes 25:1800-1821

Koenker, R. 2017. quantreg: Quantile Regression. R package version 5.33. https://CRAN.R-project.org/package=quantreg

Koning, C., K. Ashley, P. Slaney, and A. Paul. 1998. Stream fertilization as a fisheries mitigation technique for perturbated oligotrophic trout streams in British Columbia. Pages 109-120 in M.K. Brewin and D.M.A. Monita, tech. Cords. Forest-fish conference: land management practices affecting aquatic ecosystems. Proc. Forest-Fish Conf., May 1-4, 1996, Calgary, AB. Nat. Resource Canada, Edmonton, AB, Inf. Rep. NOR-X-356.

Lachance, S., M. Dube, R. Dostie, and P. Berube. 2008. Temporal and spatial quantification of fine-sediment accumulation downstream of culverts in brook trout habitat. Transactions of the American Fisheries Society 137: 1826-1838.

Lande, R. 1998. Anthropogenic, ecological and genetic factors in extinction and conservation. Researches on Population Ecology 40:259–269.

Letcher, B.H., K.H. Nislow, J.A. Coombs, M.J. O'Donnell, and T.L. Dubreuil. 2007. Population response to habitat fragmentation in stream-dwelling brook trout population. PLoSONE 2:1-11.

Lindsay, E., L. Peterson, H. Tunna, J. Dubnyk, and T. Urquhart. 2015. Late Fall Fisheries Investigations in Irrigation Canals of Southern Alberta, 2014 Trout Unlimited Technical Report No. AB-037.

Lyons, J.K., and R.L. Beschta. 1983. Land use, floods, and channel changes: Upper Middle Fork Willamette River, Oregon (1936–1980). Water Resources Research 19:463-471.

Ma, B., E. Parkinson, and D. Marmorek. 2012. Using single species population models of Bull Trout, kokanee and Arctic grayling to evaluate Site C passage alternatives. Site C Clean Energy Project Technical Data Report: Vol. 2, App. Q3, Attachment B.

MacPherson, L.M., M.G. Sullivan, A.L. Foote and C.E. Stevens. 2012. Effects of culverts on stream fish assemblages in the Alberta foothills. North American Journal of Fisheries Management 32: 480–490.

MacPherson, L., Sullivan, M., Reilly, J., and Paul, A. 2020. Alberta's Fisheries Sustainability Assessment: A Guide to Assessing Population Status, and Quantifying Cumulative Effects using the Joe Modelling Technique. DFO Can. Sci. Advis. Sec. Res. Doc. 2019/058. vii + 45 p.

Marion, D.A., J.D. Phillips, C. Yocum, and S. Mehlhope. 2014. Stream channel responses and soil loss at off-highway vehicle stream crossings in the Ouachita National Forest. Geomorphology 216:40-52.

Martins, E., L. Gutowsky, P. Harrison, D. Patterson, M. Power, D. Zhu, A. Leake, and S. Cooke. 2013. Forebay use and entrainment rates of resident adult fish in a large hydropower reservoir. Aquatic Biology 19: 253–263.

Marmulla,G. 2001. Dams, fish and fisheries. Opportunities, challenges and conflict resolution. FAO Fisheries Technical Paper No. 419. 166 pp.

McGrath, C. C., and W. M. Lewis Jr. 2007. Competition and predation as mechanisms for displacement of greenback cutthroat trout by brook trout. Transactions of the American Fisheries Society 136:1381-1392.

McMahon, T.E., A. V. Zale, F. T. Barrows, J.H. Selong, and R. J. Danehy. 2007. Temperature and Competition between Bull Trout and Brook Trout: A Test of the Elevation Refuge Hypothesis. Transactions of the American Fisheries Society 136:1313–1326.

Meyer, J.M. 1997. Stream health. Incorporating the human dimension to advance stream ecology. Journal of North American Benthological Society 16(2):439-447.

Morita K., S.H. Morita, and S. Yamamoto. 2009. Effects of habitat fragmentation by damming on salmonid fishes: lesson from white-spotted charr in Japan. Ecological Research 24:711-722.

Morita K., and S. Yamamoto. 2002. Effects of habitat fragmentation by damming on the persistence of stream-dwelling charr populations. Conservation Biology 16:1318–1323.

Morita K., and A. Yokota. 2002. Population viability of stream-resident salmonids after habitat fragmentation: a case study with white-spotted charr (*Salvelinus leucomaenis*). Ecological Modelling 155:85-94.

Mohseni, O., H.G. Stefan, and T.R. Erickson. 1998. A nonlinear regression model for weekly stream temperatures. Water Resources Research 34:2685-2692.

Muck. J. 2010. Biological effects of sedimentation on Bull Trout and their habitat – guidance for evaluating effects. U.S. Fish and Wildlife Service. Washington Fish and Wildlife Office. Lacey, WA.

Newcombe, C. P. 2003. Impact assessment model for clear water fishes exposed to excessively cloudy water. Journal of the American Water Resources Association 39: 529-544.

Norris, A.P. 2012. Cumulative effects thresholds for rctic grayling in the Wapiti River watershed. MSc Thesis, Royal Roads University, BC. 53 p.

Ottaway, E.M., and A. Clarke. 1981. A preliminary investigation into the vulnerability of young trout (*Salmo trutta*) and Atlantic salmon (*S. salar*) to downstream displacement by high water velocities. Journal of Fish Biology 19:135-145.

Seegrist, D.W. and R. Gard. 1972. Effects of floods on trout in Sagehen Creek, California. Transactions of the American Fisheries Society 101:478-482.

Park, D.J. 2006. Stream fragmentation by hanging culverts along industrial roads in Alberta's boreal forest: assessment and alternative strategies. M.Sc. Thesis, University of Alberta, Edmonton, AB, 97p.

Park, D., M. Sullivan, E. Bayne, and G. Scrimgeour. 2008. Landscape-level stream fragmentation caused by hanging culverts along roads in Alberta's boreal forest. Canadian Journal of Forest Research. 38: 566–575.

Paul, A.J. and J. Reilly. 2016. Whirling disease –risk mapping for the Bow River Basin. Technical report for Fisheries Management, Alberta Environment and Parks. Cochrane, AB. 1 Sept 2016. 14 pp.

Peterson, D.P., K.D. Fausch, J.Watmough, and R.A. Cunjak. 2008. When eradication is not an option: Modeling strategies for electrofishing suppression of non-native Brook Trout to foster persistence of sympatric native Cutthroat Trout in small streams. North American Journal of Fisheries Management 28:1847-1867.

Peterson, D.P., K.D. Fausch, and G.C. White. 2004. Population ecology of an invasion: Effects of brook trout on native cutthroat trout. Ecological Applications 14(3):754-772.

Post, J., C. Mushens, A. Paul, and M. Sullivan. 2003. Assessment of alternative harvest regulations for sustaining recreational fisheries: model development and application to Bull Trout. North American Journal of Fisheries Management 23:22–34.

Reilly, J., R. Konynenbelt, and S. Herman. 2016. Pinto Lake Bull Trout Recovery Action Plan 2015-2020. Alberta Environment and Parks, Fisheries Management, Rocky Mountain House, AB. 13 p + 1 appendix.

Rieman, B., J. Peterson, and D. Myers. 2006. Have brook trout (*Salvelinus fontinalis*) displaced Bull Trout (*Salvelinus confluentus*) along longitudinal gradients in central Idaho streams? Canadian Journal of Fisheries and Aquatic Science 63:63-78.

Ripley, T., G. Scrimgeour, and M.S. Boyce. 2005. Bull Trout occurrence and abundance influenced by cumulative industrial developments in a Canadian boreal forest watershed. Canadian Journal of Fisheries and Aquatic Sciences 62: 2431–2442.

Salow, T. and L. Hostettler. 2004. Movement and mortality patterns of adult adfluvial Bull Trout (*Salvelinus confluentus*) in the Boise River basin Idaho. U.S. Bureau of Reclamation, Denver, Colorado.

Scrimgeour, G., P. Hvenegaard, J. Tchir, S. Kendall and A. Wildeman. 2003. Stream fish management: cumulative effects of watershed disturbances on stream fish communities in the Kakwa and Simonette River Basins, Alberta. Alberta Conservation Association, Peace River and the Alberta Research Council, Vegreville, AB. Northern Watershed Project Final Report No. 3.

Selong, J.H., T.E. McMahon, A.V. Zale, and F.T. Barrows. 2001. Effect of temperature on growth and survival of Bull Trout, with application of an improved method for determining thermal tolerance in fishes. Transactions of the American Fisheries Society 130:1026–1037.

Sinnatamby, R.N., A. Cantin, and J.R. Post. 2020. Threats to at-risk salmonids of the Canadian Rocky Mountain region. Ecology of Freshwater Fish 29:477-494.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. Journal of Hydrology 176:79-95.

Stelfox, B., M. Anielski, M. Carlson, and T. Antoniuk. 2008. Alberta Southern East Slopes Integrated Land Management Pilot Project. Prepared for Alberta Environment and Alberta Sustainable Resource Development by the ASPEN Group. 55 p.

Sullivan, M. G. 2014. The Bull Trout of Jacques Lake, Jasper National Park: a population in recovery. Alberta Cooperative Conservation Research Unit, Edmonton AB. 18p.

Sullivan, M.G. 2018. Can fishing mortality at catch-and-release Bull Trout fisheries be a potential threat to recovery? AEP technical memo, 11 July 2018. Alberta Environment and Parks, Fish and Wildlife Policy Branch, Edmonton AB. 7pp.

Sullivan, M.G., and S. Spencer. 2016. Cumulative effect of whirling disease and fishing mortality on Alberta trout populations. Technical report, Fisheries Management, Alberta Environment and Parks. Edmonton, AB. 29 August 2016. 10 pp.

The Canada Gazette. 2019. Species at Risk Act: Order Amending Schedule 1 to the Species at Risk Act. Canada Gazette Part II, 153.

U.S. Fish and Wildlife Service (USFWS). 2000. Revised section 7 programmatic consultation on issuance of section 10(a)(1)(A) scientific take permits and section 6(c)(1) exemption from take for Bull Trout (*Salvelinus confluentus*) (6007.2100). Memorandum from Acting Supervisor, Snake River Basin Office, Boise, Idaho, to Regional Director, Region 1, Portland, Oregon. February 14, 2000. 22 p.

U.S. Fish and Wildlife Service (USFWS). 2014. Revised draft recovery plan for the coterminous United States population of Bull Trout (*Salvelinus confluentus*). Portland, Oregon. xiii + 151 p.

Underwood, K., and S. Cramer. 2007. Simulation of human effects on Bull Trout population dynamics in Rimrock Reservoir, Washington. American Fisheries Society Symposium 53:191-207.

Venables, W. N., and B. D. Ripley. 2002. Modern Applied Statistics with S. Springer. https://doi.org/10.1007/978-0-387-21706-2_1

Volpe, J.P., B. R. Anholt, and B. W. Glickman. 2001. Competition among juvenile Atlantic Salmon (*Salmo salar*) and steelhead (*Oncorhynchus mykiss*): relevance to invasion potential in British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 58:197-207.

Wang, T., A. Hamann, D. Spittlehouse, and S.N. Aitken. 2006. Development of scale-free climate data for western Canada for use in resource management. International Journal of Climatology 26:383–397.

Warnock, W. 2012. Examining brook trout invasion into Bull Trout streams of the Canadian Rockies. PhD Thesis, University of Lethbridge, AB. 184 p.

Warren, M.L., Jr., and Pardew, M.G. 1998. Road crossings as barriers to small stream fish movement. Transactions of the American Fisheries Society 127: 637–644.

Welsh, M. J. 2008. Sediment Production and Delivery from Forest Roads and Off-Highway Vehicle Trails in the Upper South Platte River Watershed, Colorado. MSc thesis, Colorado State University.

Wenger, S.J., D.J. Isaak, J.B. Dunham, K.D. Fausch, C.H. Luce, H.M. Neville, B.E. Rieman, M.K. Young, D.E. Nagel, D.L. Horan, and G.L. Chandler. 2011. Role of climate and invasive species in structuring trout distributions in the interior Columbia River Basin, USA. Canadian Journal of Fisheries and Aquatic Science 68: 988–1008.

Zuur, A.F., E.N. Leno and C.S. Elphick. 2010. A protocol for data exploration to avoid common statistical problems. Methods in Ecology and Evolution 1:3-14.

Alberta