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Alberta's Fisheries Sustainability Assessment: A Guide to Assessing Population Status, and Quantifying Cumulative Effects using the Joe Modelling Technique

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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DEFINITIONS

Current Adult Density (CAD): A measure of population status that is the density of mature individuals in a fish population using the most recent information available across a defined spatial area.

Current Immature Density (CID): A measure of population status that is the density of immature individuals in a fish population using the most recent information available across a defined spatial area.

Fisheries Sustainability Assessment (FSA) Score: A reporting metric based on the population's proportion of maximum system capacity categorized from zero to five using criteria adapted from international conservation agencies (e.g., Williams et al. 2007; Faber-Langendoen et al. 2009). The transformation from proportion of system capacity to FSA score is not linear (see Table 1).

Historic Adult Density (HAD): A measure of historic population status that is density of mature individuals in a fish population across a defined spatial area. This represents the population in an undisturbed or lightly disturbed state.

Joe Model: A stable state cumulative effects model that combines multiple stressor-response curves to predict a common response metric that would be expected over the long term. Response metrics for each curve are multiplied together; the result is an additive cumulative effects model on the proportional (logarithmic) scale (Smit and Spaling 1995).

Recovery: A return to a state in which the population and distribution characteristics and the risk of extinction are all within the normal range of variability for the wildlife species (DFO 2014a).

Response metric: A dimensionless measure of population status used for cumulative effects modelling. The response metric K_{Joe} is defined $K_{Joe} = \frac{K}{K_{max}}$, K is system capacity and K_{max} is the reference condition.

Stressor-Response Curve: Describes the expected value (or range of values) over the long term for system capacity given a dose of the stressor. The definition has been adapted from DFO (2014b), but here includes both human-induced and natural stressors.

System Capacity: The expected long-term measure of population status under a given set of stressors. System capacity can either be: measured using field data; or, modelled using stressor-response curves. System capacity can be translated into a response metric (dimensionless) by division with the expected maximum system capacity when all stressors are at their optimal values. System capacity is a continuous variable that must not be confused with the FSA score that is categorical.

ABSTRACT

Managing fish and fisheries in Alberta is difficult and getting harder, with fewer fish, more stakeholders, and mounting stressors. These include both direct users of fish (Indigenous peoples, and recreational anglers) and indirect users of fish, such as forestry, municipalities, and agriculture, and their effects on fish populations through habitat changes. Effective communication of necessary trade-offs is fundamental in achieving the goal of long-term sustainability of Alberta's fish and fisheries.

For effective understanding and management of these complex effects on the status of fish populations, Alberta is adopting the principles of the United Nations Food and Agriculture Organization (FAO) Code of Conduct of Responsible Fisheries. These include the principles of Management Strategy Evaluation (MSE), as adapted to Alberta's complex issues of cumulative effects threats on multiple fish stocks. These adaptations have resulted in a transparent, easily communicated system of assessing status and threats. This system is called the Fisheries Sustainability Assessment (FSA) and is a two-part process: 1) Assess Status: current status scaled to a provincial reference condition, and contrast this to desired status, and 2) Assess Threats using simple cumulative effects models (called Joe Modelling): model the hypothesized threats to achieving desired status. From this process, effective mitigation actions can be designed, implemented, and tested.

A novel feature of the Joe Modelling process is its ease of design and communication. Built in workshop settings biologists can, in real time, include almost any stressor or management action the participants suggest. Sources of information can readily include academic knowledge, anecdotal descriptions from experienced stakeholders, and traditional knowledge. The model outputs and simulated trade-offs that alter system capacity are considered as hypotheses, not forecasts. As such, the purpose of the Joe Models can best be viewed as a tool for prioritizing impact hypotheses based on management actions and expected effects (i.e., changes in system capacity).

By combining a standardized population status assessment, and the Joe Modelling threats assessment, Alberta's Fisheries Sustainability Assessment is a logical, transparent process of determining fishery status and prioritizing mitigation actions. Learning and improvement on fisheries management and fish conservation are the ultimate objectives of this system.

ACKNOWLEDGEMENTS

Alberta Environment and Parks (AEP) biologists David Park, Dr. Michael Sullivan and Matthew Coombs developed the first iteration of this process during the mid-2000s, then called the “Fish Sustainability Index”. Laura MacPherson (AEP) and Jessica Reilly (AEP) then completed further refinements. Extensive developments over the past decade led to the amendments presented in this report. Further, we are indebted for the robust peer review provided by the Canadian Science Advisory Secretariat (CSAS) invited experts.

INTRODUCTION

Alberta has one of Canada's fastest growing human populations and economies, and consequently, faces rapidly changing issues of fisheries conservation (Schneider 2002, Post et al. 2002, Sullivan 2003, Schindler 2009). Achieving Alberta's legislated fisheries principles of conservation, Indigenous people's rights, sport fishing, and economic benefits requires a highly effective fisheries management process (AESRD 2014, Arlinghaus et al. 2016). The complexity of this task resulted in Alberta adopting aspects of the United Nations Food and Agriculture Organization (UN FAO) Code of Conduct of Responsible Fisheries (FAO 1995, FAO 2012, AESRD 2014). Integral to these principles is the formalized system of modelling and testing alternative management strategies known as the Management Strategy Evaluation (Holland 2010, Punt et al. 2014). These principles and techniques have been developed primarily for large commercial fisheries (Punt et al. 2008, Deroba and Bence 2012). Alberta fisheries biologists have developed a system to adapt these principles to the complex problem of managing multiple freshwater systems and species threatened by a variety of cumulative effects (including habitat loss, invasive species and overfishing). This adapted system is designed to rigorously meet the criteria of science-based natural resource management (Artelle et al. 2018). Alberta Environment and Parks (AEP) is moving toward managing all fish populations using this consistently applied, step-wise process that has measureable quantitative objectives, is based on empirical evidence with peer-review, and has public transparency.

This provincial-scale fisheries management process is conducted as hierarchal steps with three major components; Fisheries Management Objectives, Fisheries Sustainability Assessment, and Species Management Framework. The first component is a pair of policy-level decisions to quantify the Fisheries Management Objectives; what is the quantified scale of low to high risk for sustainability of this species in Alberta, and using that scale, what are the desired Fisheries Management Objectives for individual populations? The next component, termed the Fisheries Sustainability Assessment, is the assessment of the historic and current status using the same quantified scale, along with a quantification of stressors to each individual population. The remaining component is the Species Management Framework (or Species Recovery Plan if designated as a species at risk), which determines the appropriate actions and timelines to achieve the Fisheries Management Objectives for each population. Of these three steps, the Fisheries Sustainability Assessment is the key component to allow policy objectives to be realistically achieved through management actions.

The Fisheries Sustainability Assessment (FSA) is a two-part process: 1) *Assess Status*: current status scaled to a provincial reference condition, and contrast this to desired status, and 2) *Assess Threats using simple cumulative effects models (called Joe Modelling)*: model the hypothesized threats to achieving desired status. From this process, effective mitigation actions can be designed, implemented, and tested.

Assess Status: For assessment, fish populations are scored into density categories of 0 (extirpated) to 5 (very high density). Populations are defined as species-in-lakes, and species-in-watersheds, such as "Walleye in Lac Ste. Anne", or "Bull Trout in the Berland River watershed". Standardized index-netting in lakes and electrofishing in rivers are primarily used to determine density, as scored in relation to provincial-level thresholds. This simple score comparison, current versus undisturbed, provides a critical perspective to minimize shifting baselines, and standardizes the interpretation of fish status across species and watersheds. The current status is then compared to a socially and politically-determined desired status (Fisheries Management Objective). The difference in current versus desired score is therefore quantifiable, and clearly defines the management targets.

Assess Threats using Joe Modelling: Achieving the management target requires the assessment and effective mitigation of threats, such as habitat loss, invasive species, climate change, and overharvest. The threats for each individual population are examined using a novel modelling process termed “Joe Modelling” that uses a series of stressor-response models (e.g., impact of sediment on system capacity for fish), parameterized with watershed-specific data (e.g., what is sediment level in Berland River watershed?), and multiplies the scaled output of these individual stressor-response curves (additive effects on the proportional scale). The resultant cumulative threats assessment defines the capacity of the system to achieve a certain stable state; e.g., if threats are all high, the best state the population might achieve is low density as the model is additive. Therefore, by defining the cumulative effects of threats as a “system capacity”, Joe Modelling provides a useful bridge between listing all potential threats (as typically conducted in species at risk planning), and detailed quantitative fisheries population dynamics models that include threats of predicted high importance.

This document describes the second iteration of Alberta’s Fisheries Sustainability Assessment (FSA v2.0), first developed in 2010 as the Fisheries Sustainability Index, FSI (MacPherson et al. 2014). The key change is the quantification of impacts to a population using Joe Modelling. Multiple impacts are modelled as cumulative, using the model in a static format (i.e., one point in time) to prioritize hypotheses that quantify plausible key limiting impacts to the current status. This model can then be used in the next phase of management (Species Management Framework, or Species Recovery Plan) to develop management recovery scenarios that combine the effects of mitigating individual impacts in order to achieve the desired Fisheries Management Objectives. These management or recovery scenarios are hypotheses, which must then be tested using active adaptive management actions (Figure 1).

The following report is intended to act as an ‘operators manual’ and provide guidance to assist fisheries biologists in conducting consistent stock and cumulative effects impact assessments across the province and across fish species. These guidelines are expected to change over time, as problems are discovered and solutions found.

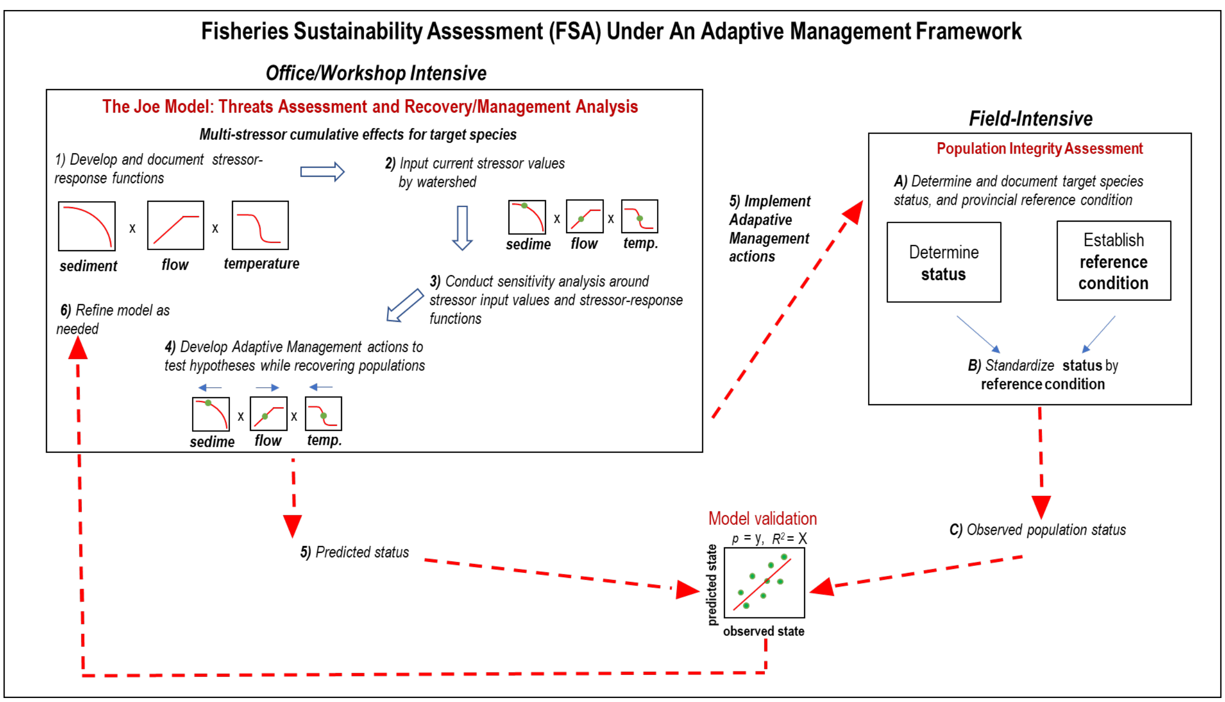

Figure 1. Diagram of Alberta’s Fisheries Sustainability Assessment (FSA) process that assesses population status and incorporates Joe Modelling under an adaptive management framework.

FISHERIES SUSTAINABILTY ASSESSMENT (FSA) RULE SET

To maintain consistency, this rule set provides instructions on how to 1) assess population status and 2) create a cumulative effects impact model (i.e., Joe Model) using the FSA methodology.

Fisheries Sustainability Assessments are applied to wild and naturalized (self-sustaining) stocked populations. Fish stocked into put-and-take fisheries do not require an FSA. For example, self-sustaining Walleye (*Sander vitreus*) stocked in prairie reservoirs would get an FSA even though they live in artificial habitat. On the other hand, Arctic Grayling (*Thymallus arcticus*) stocked in ponds in southern Alberta are generally not self-sustaining and do not require assessment.

Assessing Population Status

FSA Focal Species

FSA assessments of fish species in the province will be conducted in accordance with provincial objectives and priorities. At present, the following are priority FSA species:

1. Bull Trout (*Salvelinus confluentus*)
2. Lentic and lotic Walleye
3. Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*)
4. Lake Sturgeon (*Acipenser fulvescens*)
5. Athabasca Rainbow Trout (*Oncorhynchus mykiss*)
6. Arctic Grayling
7. Goldeye (*Hiodon alosoides*) and Mooneye (*Hiodon tergisus*)
8. Lake Trout (*Salvelinus namaycush*)
9. Lentic and lotic Northern Pike (*Esox lucius*)
10. Yellow Perch (*Perca flavescens*)
11. Mountain Whitefish (*Prosopium williamsoni*)
12. Sauger (*Sander canadensis*)
13. Burbot (*Lota lota*)

Many of the priority species have undergone an FSI assessment (version 1), but will need to be updated under the current FSA (version 2) format. For an up-to-date listing and results of all completed FSA species, refer to the Alberta Environment and Parks FSA webpage.

FSA Data Requirements

Each FSA represents a present-day snapshot in time of the current status of a fish population. Fisheries biologists should reassess their population's FSA score regularly as new data are collected on population status, as the severity of population impacts change and as new impacts appear, or management actions change. For any particular FSA, the most up-to-date data should be used to compare the focal fish population against the reference population described in the Fisheries Management Objectives. For high-profile fisheries, these data are collected using active monitoring protocols. These are generally index netting in lakes (index netting standard, Morgan 2002; ASRD 2010) and electrofishing in streams (river monitoring protocols, AEP 2018). For lower-profile lentic fisheries (e.g., remote, undisturbed, or less-used by humans), monitoring is conducted following the passive monitoring protocols (Brown 2017). Passive monitoring for lakes relies on assessing information relating to five factors: similarity to nearby actively monitored lakes, surface area, road access, citizen science, and government staff reports. Since reliability of the data is evaluated (detailed in a later section of the rule set), these assessments also aid in prioritization of future data collection. In instances where the data used may be imprecise or inaccurate, the quantity of data available is limited, or outdated data is used, this will be highlighted by low confidence scores in the field-based measure of current status.

As with other collected fisheries data, once finalized, FSA scores and supporting information are entered into the provincial Fisheries and Wildlife Management Information System (FWMIS) database to ensure that data is available for staff and stakeholders, is archived and data integrity is preserved.

Population Assessment Scale

In 2013, the Fisheries Management Branch in collaboration with Alberta Environment and Parks Data Management and Water Management, created a provincially comprehensive and aggregated collection of standard hydrologic units based on United States Geological Survey (USGS) standards and procedures. The Hierarchical Unit Code (HUC) watersheds establish a standardized baseline that covers all areas, and where successively smaller hydrologic units are nested within larger hydrologic units, creating a hierarchal watershed boundary dataset. Currently, four levels of nested watersheds have been delineated, 2, 4, 6 and 8-digit HUCs. 10-digit HUCs are delineated for Alberta's East Slopes. The 2-digit HUCs are the largest watersheds and 10-digit HUCs are the smallest, finer scale watersheds.

The HUC watershed dataset was delineated based on sound hydrologic principles to ensure that they are not created in favour of a specific department or program objective. As such, they can (and are) being used by a diverse group of government and non-government agencies, and will have a lasting value in water and watershed modelling and management programs, as well as resource inventory assessments. Given that the HUCs were delineated based on hydrologic principles, the scale may never perfectly fit our definition of a species' population, but the closest watershed scale will be chosen and used.

Given their widespread use, HUC watersheds have been chosen as the spatial assessment unit for the FSA when assessing lotic populations. However, the specific scale of HUC watershed used in a FSA evaluation will be species-specific and reflect life history traits and available genetic information. For the purposes of the FSA when appropriate genetic information is available, we define a population as a group of individuals that exhibit self-assignment rates of ~90% using multilocus genotypes. This means that, at a minimum, 90% of individuals captured within a certain spatial extent (e.g., lake, river, or river system) assign or 'belong' to the same population. However, in the absence of genetic data, telemetry information, or scientific literature can be used to determine at what scale to define a 'population'.

For instance, in a genetic analysis of Alberta Arctic Grayling, Reilly et al. (2014) found that self-assignment rates were relatively high when individuals were grouped at a spatial equivalent to 6-digit HUCs (average self-assignment rate = 86%). Conversely, when investigating Bull Trout population structure using genetic variation at nine microsatellite markers, Warnock (2008) found evidence of at least three populations in the Castle River 8-digit HUC. In this instance, an even smaller HUC (10-digit) would be the appropriate scale of assessment. Highly migratory species (e.g., Lake Sturgeon) populations may be defined at a larger spatial scale (e.g., an entire basin), but fisheries biologists may choose to separate results at smaller spatial scales where actions can realistically be applied.

All HUCs where the focal fish species exist, is suspected to exist, or has been extirpated must be identified and assessed. This can include areas of habitat that are seasonally or temporally unoccupied (IUCN 2008). In cases of extirpation or range contraction, the pre-disturbance distribution of species, irrespective of present day or historical human impacts, should be assessed.

To be consistent with the International Union for the Conservation of Nature's definition of extent of occurrence (IUCN 2008), "cases of vagrancy" should be excluded. For example, Lake Trout in Alberta are occasionally observed in rivers downstream from existing lake populations, but rivers would be excluded from their lentic assessment.

Lotic HUCs may include lakes, if the fish in these lakes are considered part of the larger lotic population. Alternatively, if lentic fish in the lake form a distinct fish population (or it is managed as such by lake-specific angling regulations), the level of assessment is the individual waterbody and the lentic population is defined separately within the lotic HUC as the boundaries of the lake and if applicable, any connected streams or lakes frequented by the lentic population.

Population Integrity

To capture changes in status of Alberta's fish populations through time, aspects of population integrity are summarized using three separate metrics: current adult density, current immature density and historic adult density.

Evaluations are made by comparing measured fish densities of focal populations to fish densities of an observed or a modelled-theoretical reference population covering the same areal extent but occurring in the most ideal habitat in Alberta, unaffected by anthropogenic influences (i.e., no fishing mortality, no habitat loss, and no competition with exotic species). Differences between focal and reference populations are translated to a scale of one to five, and represent five different risk categories (Table 1) (AESRD 2012). This is the FSA score. A score of one corresponds to a focal population that is least sustainable and very different from the reference population, and a five or higher corresponds to a focal population that is most sustainable and very similar to the reference population. This scoring system follows those used by international conservation agencies (e.g., Williams et al. 2007; Faber-Langendoen et al. 2009). Note that for cases where comparisons between the focal and reference population are not possible (e.g., extirpated, or no fish were detected), an additional rating of zero has been added to the Alberta FSA. A zero represents a functionally extirpated population (e.g., no fish were detected in recent history, or extirpation is suspected). While a few individuals may still occur in a functionally extirpated population, it is not thought to constitute a viable population. A FSA score of Current Adult Density (CAD) of four and five represent a population at low risk and very low risk, respectively. In practice, densities higher than the threshold of five are measured, and these are scored simply as a five (very low risk). A score of one is a population at very high risk. A zero represents a functionally extirpated population where extirpation is known or suspected (Table 1). In addition to allowing the FSA score to align with the Alberta species at risk framework, risk assessment ratings also provide broad categories, which allow fisheries biologists to more easily assign estimated FSA scores in the absence of intensive data.

Active monitoring for fisheries assessments should be conducted using scientifically robust and consistent methods. In Alberta, standardized stream and river electrofishing protocols have been developed (ASRD 2008; AEP 2018) and follow American Fisheries Society standards and sampling protocols (Bonar et al. 2009). For rivers and streams, boat and backpack electrofishing are currently the most cost-effective monitoring methods and are currently the primary flowing water assessment protocols (AEP 2018). In lakes, fisheries biologists primarily rely on standardized index netting procedures such as fall index netting (Morgan 2002; ASRD 2010) and North American Standard Index Netting (Bonar et al. 2009). Catchability of different techniques is quantified on an ongoing basis (Mogensen et al. 2014). Catch rates and population estimates should be as representative of the population-level system (i.e., entire HUC or lake) as possible, and avoid site-specific biases when sampling. Fisheries biologists must assess whether site-specific catch rates or population estimates are representative of the HUC or lake. For example, electrofishing for Bull Trout on a spawning tributary would likely be biased towards adult fish and higher densities than the HUC as a whole; or, electrofishing Arctic Grayling on the one tributary where they have been caught in the past may not be representative of the entire population in a HUC.

For passively monitored assessments, fish density is estimated using protocols relying on remote GIS-type data, and observational data (Brown 2017). These may be supplemented by other sources of information such as historical data (e.g., preliminary biological surveys, angler surveys, and commercial fishery data) and citizen science (e.g., Alberta typically uses data from anglers using iFish app on smart phones). Whether quantitative or qualitative, it is essential that fisheries biologists explain how they arrived at a score for density in the “Comments” section and if quantitative data are used, explain where these came from, how many sites were sampled, and how these sites were representative or adjust to represent the HUC or lake as a whole.

The only cases when zeros can be entered as FSA score values for adult (historical and present) density or immature density are when the focal species is a functionally extirpated population (none detected, extirpation suspected) in the HUC being considered.

As explained in subsequent sections, these measures of status are used as the common response metric (y-axis) in the cumulative effects Joe Model. While Alberta fisheries biologists have used the current status of mature fish to assess the sustainability of the population, users of the Joe Model could just as easily use the current status of immature fish or another measure of population status once scaled to a reference condition as the response variable.

Table 1. Proportion of a population remaining compared to a theoretical population undisturbed by anthropogenic influences, and the corresponding Alberta Fisheries Sustainability Assessment (FSA) scores.

FSA Score	Risk Assessment Rank	Percent (%) of Reference Population
0	Functionally Extirpated	0
1	Very High Risk	<20
2	High Risk	20–50
3	Moderate Risk	50–70
4	Low Risk	70–100
5	Very Low Risk	100

Density Metrics

Current Adult and Immature

Current Adult Density (CAD) and Current Immature Density (CID) metrics indicate the density of these two demographic classes of fish compared to the species-specific reference condition. If a population is undisturbed by human impacts, but is at the edge of its species range and at a very low density because of natural habitat limitations, it would still be scored as very low density. This is interpreted as being at very high risk and barely sustainable relative to the reference population occupying the same area of ideal habitat. This relative score is important, because the status score is not meant to imply that the assessed population has declined from a high density, nor that it could potentially recover to a high density. It does, however, imply that a lower density of fish is likely to be at a higher risk to sustainability than a population at higher density. When scoring populations fragmented by extensive human disturbance where there are no barriers to fish movement, consider the once-large, contiguous population and score each small fragment as the appropriate area-weighted fraction of the larger contiguous population. Species-specific CAD and CID density thresholds must be determined prior to the commencement of assigning a FSA score. General guidelines and the associated risk assessment scorings are found in Table 2.

A general note, while fisheries biologists have found that current adult and immature density are useful metrics used when monitoring and reporting on fish populations in the province, fisheries biologists also consider other key attributes such as population size structure, distribution, and genetic status.

Table 2. Alberta Fisheries Sustainability Assessment (FSA) scores and Risk Assessment ranks for adult and immature density.

FSA Score	Risk Assessment Rank	Current Adult Density	Current Immature Density
0	Functionally Extirpated	No adults observed	No immatures observed
1	Very High Risk	Lowest possible without extirpation, adults barely detectable	Lowest possible without extirpation, young barely detectable
2	High Risk	Low density, recruitment overfishing	Low density, recruitment overfishing
3	Moderate Risk	Moderate density, growth overfishing below maximum sustainable yield (MSY)	Moderate density growth overfishing below MSY
4	Low Risk	High density, population at or above MSY with minor growth overfishing	Highest possible density, population at or above MSY, potentially peak recruitment if exhibits Ricker stock-recruit curve, minor growth overfishing
5	Very Low Risk	Highest possible, adult population at reference carrying capacity	Very high density, peak or slight natural recruitment reduction possible (i.e., overcompensation)

Historic Adult

Historic Adult Density (HAD) uses the same criteria as the CAD metric described in the previous section, but for the undisturbed or lightly disturbed historical condition. The purpose of this metric is to capture information about the fish or fishery status at its earliest point of fishing or sampling, or the best available estimate of its undisturbed condition. Since fish species are often not naturally abundant across the entirety of their range, this metric can be used in the cumulative effects model to capture the natural limitations to the maximum productivity some fish populations may experience.

The time period associated with the historic condition will vary regionally. For example, watersheds and lakes in southern Alberta were described and documented in time periods before some northern portions of the province, and generally, these accessible fisheries were more heavily exploited than northern fisheries (although early cases of severe overexploitation in Lake Wabamun and Lac La Biche have occurred, Schindler et al. 2008). Records of fisheries from the fur trade period (late 1700s to late 1800s) can be very useful to assess if fish populations were either rare or abundant (e.g., Moberly 1929; Douglas 1977). Alberta municipal communities often have local histories published, which can be very useful in understanding abundance and distribution of fish in Alberta's early years of settlements. For example, to celebrate Alberta's 75th anniversary, Alberta Culture facilitated the writing of numerous local community histories, and these are increasingly becoming available in digital form (see digital collection, University of Calgary 2018). Other sources of historical fisheries status include interviews with early residents, such as collected in Chipeniuk (1975) or in historical surveys conducted by government biologists (Valastin and Sullivan 1997). The details of the sources of these historical assessments are captured in the description and comments. This metric will be used to help describe the original range and abundance of a species, and the changes in abundance relative to the current status.

For example, in the Tawatinaw River during the 1940s to 50s, a 1996 Local Environmental Knowledge (LEK) historical survey (Valastin and Sullivan 1997) summarized the following comments:

- Art Delancy: "always good for grayling, nice size, up to 1 lb"
- R.B. Miller: "abundant grayling below Meanook"
- M. Paetz: "grayling fishing by Colinton, but small fish by 1960s"
- John Kormendy: "by Perryvale, 1 lb. grayling easy to catch"
- Peter Marchuk: "fished in the 1950s and 1960s. Around the Rochester area the grayling were really nice "panners". Average size about 12–14 in."

Based on these stories, the Historical Adult Density in the Tawatinaw River was scored as FSA HAD status of 4 (low risk).

Conversely, in a historical LEK survey of Walleye in North Buck Lake (Valastin and Sullivan 1997), the following comments were recorded:

- Blake Smith fished from 1950 on and never caught any walleye there, but heard of people who did and said it was good.
- Hilaire Ladocoeur started fishing in 1940 and said he's never once caught a Walleye in this lake.
- John Gordey fished in 1960 and said that there used to be a lot of pike and Walleye here. "It would be nothing to get 5–6 Walleye here per day".

- M. Paetz a previous Superintendent of Fisheries Management said fish here consistently tasted “muddy”. This included the Walleye and it is rare for a Walleye to taste “muddy”.

Based on these conflicting stories and irregular Walleye catches in commercial fishing records, the Historical Adult Density would likely be scored as FSA HAD 3 (moderate risk; certainly present, not abundant for most anglers, perhaps patchy in distribution, low certainty of data quality).

In instances where a fishery was only recently established (e.g., Northern Pike naturally moving into a previously unoccupied reservoir), the year of establishment will be considered the historical condition of that fish species or fishery. Obviously, appropriate explanations should be added into the comments fields.

ASSESSING THREATS: INTEGRATING CUMULATIVE EFFECTS USING JOE MODELLING

The challenge of conserving and managing fish given a myriad of stressors and complex cumulative effects is daunting (Dudgeon et al. 2006; Hansen et al. 2015; Hunt et al. 2016). Rather than recognizing the importance of integrating these multiple complex drivers into the theory and management of fisheries (Beard et al. 2011; Schindler and Hilborn 2015), too often management agencies have responded by creating long lists of potential impacts and actions with a lack of a coordinated strategy for implementation. For instance, in Alberta’s Athabasca Rainbow Trout Recovery Plan (Athabasca Rainbow Trout Recovery Team 2014), over 30 impacts were listed. Understanding which impacts were limiting the trout populations and developing mitigation for these impacts was a goal proposed by the Recovery Team, but was stated without specifying direct actions to achieve this understanding.

Alberta’s cumulative effects modelling process (Joe Modelling) is designed as a strategic tool to address these complexities. The model is aptly named in honour of Dr. Joe Nelson (Murray et al. 2012), because ‘if you wanted to know anything about Alberta fish, ask Joe’. The process of building and using these cumulative effects models is designed to include stakeholders and directly incorporate local, traditional, and academic knowledge. Ideally designed in an interactive workshop setting, and refined through data analysis, and extensive exploration of the literature, a completed model results in clear statements of hypotheses of impact mitigation (e.g., given our assumptions, what is the predicted outcome from proposed actions?). Modelled results are treated as hypotheses needing testing, rather than forecasts. The results from these models are emphasized as simply the mathematical representation of the participants’ best available understanding of threat quantity, effect and combination on the particular population. As such, the Joe Models serve two related strategic purposes: 1) they quantify existing impacts to identify the hypothetical key drivers of population status; and 2) they allow scenario modelling of mitigation actions to explore and optimize potential combinations of recovery actions.

The Conceptual Model

Alberta cumulative effects Joe Models are a series of stressor-response curves representing impacts and limiting factors that are combined to simulate and quantify the cumulative effects on the system capacity of a fish population. Each model consists of a series of stressor-response curves where each impact is treated as independent, with the identified impact as the stressor and output from the curve is system capacity. Prior to using this output for the Joe Model, system capacity must be scaled to the reference condition for the response metric. The response metric is a fraction of the putative reference Current Adult Density. The response metrics for each stressor are then multiplied together to develop the cumulative adult response (Figure 2). System capacity (i.e., the response metric) is the resulting proportion (0–100%) of the reference condition. System capacity must not be confused with the FSA score that is categorical. However, system capacity can easily be assigned an FSA category using criteria in Table 1.

The Joe Model output predicts the proportion of system capacity relative to the reference condition achieved over the long term if input threats (stressors) remain at specified levels. The model does not predict the temporal or spatial trajectory a system would follow in reaching the predicted system capacity or dynamic patterns that may occur in the long-term. Regardless, proportion of system capacity is a measurable quantity that can be explicitly tested.

For example, if temperature was not limiting densities of a Mountain Whitefish population, the input “stressor” parameter would be a temperature value within the optimal range for Mountain Whitefish. The output from the stressor-response curve for temperature for that population would be the maximum system capacity. If however, the stream temperatures in the watershed were too warm, it would be expected to cause the population to be at a depressed status, perhaps caused by factors such as increased mortality, or concentration in a few pockets of thermal-refuge habitat. The output system capacity “response” for this population would be depressed.

Each impact is initially developed and modelled independently through a stressor-response curve that predicts system capacity. The combination of impacts (i.e., system capacity from each stressor-response curve) are: a) represented as a fraction of the reference condition (i.e., response metric); and, b) response metrics for each stressor are multiplied to get an overall proportion of the reference condition (Figure 3). Although this sounds somewhat complicated, it simply describes an additive cumulative effects model on a proportional scale, which is the sensible biological scale if each impact influences survival independently. Weighting of individual impacts is not necessary because each impact is quantified as acting on the same output parameter: the common and dimensionless response metric. Weighting impacts has long been a difficulty of traditional cumulative effects models (Walters 1997). The novel approach of Joe Modelling simplifies that difficulty.

To take our example further, if the Mountain Whitefish population was occupying warm fringe waters (resulting in an expected system capacity of 2.1 units, out of 5.0[1]), was also experiencing high direct angling mortality (resulting in an expected system capacity of 3.3 units), and the reference condition is set at a maximum system capacity of 5 units[1], the cumulative effect of warm streams and high mortality on system capacity would be 2.1/5 x 3.3/5 x 5 =1.4 units or 28% of the reference condition (i.e., 1.4/5 x 100). This would be categorized as a FSA score of 2 (high risk; Table 1). In this example, the impact of overfishing would be hypothesized to have a smaller effect than the impact of warm temperatures. In a recovery scenario considering only these two parameters, the theoretical action of closing the fishery would only result in a CAD increase from 1.4 to 2.1 units.

Predictions of system capacity from the Joe Models are quantitative in nature. Predicted system capacity is therefore directly testable using field data collected at appropriate scales.

[1] The use of a 5 units as maximum system capacity is a hold over from when the Joe Models worked directly with FSA numbers. However, as FSA is categorical, it should not be treated as continuous. Furthermore, attempts at translating FSA into a response metric is not intuitive given non-linearity of the FSA categories (Table 1). Thus, the 5 unit scale in the example does not equate to FSA values but rather can be simply and intuitively converted to percent of reference condition through division by 5.

The novel and effective value of the Joe Modelling concept is in these two points: 1) any number of impacts or “stressors” can be efficiently added to the model; and 2) the potentially complex weighting of impacts is accomplished by simply using one easily measured output response metric. With respect to the first point, the additive (on the proportional scale) nature of the model means increasing stressor numbers necessarily lowers the predicted cumulative system capacity unless added stressors are at optimal conditions. Although this may impact predictions of cumulative system capacity, it does not detract from the prioritization of stressor importance or strategic selection of watersheds to test management actions.

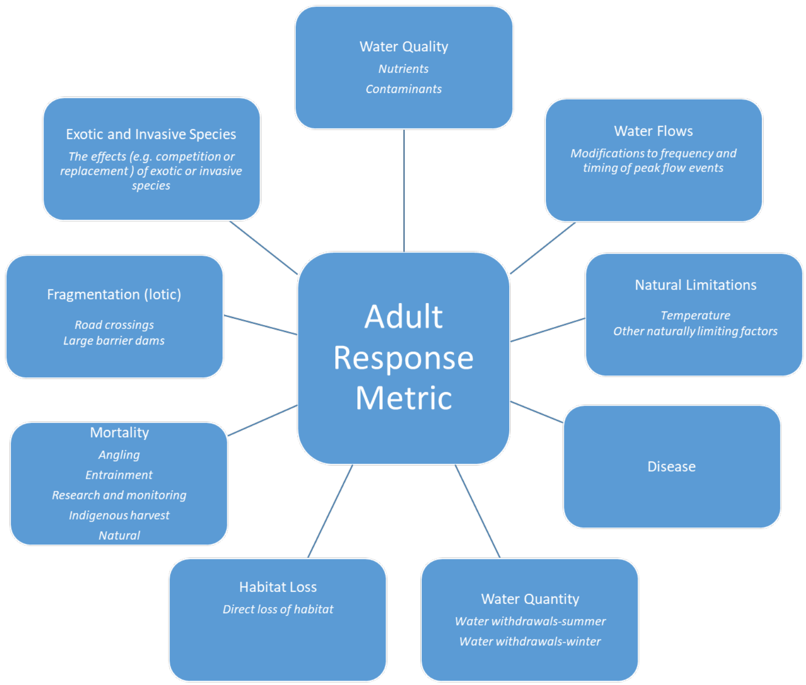

Figure 2. Conceptual diagram of an example of a cumulative effects Joe Model and broad impact categories. Adult response metric is system capacity scaled to the reference condition.

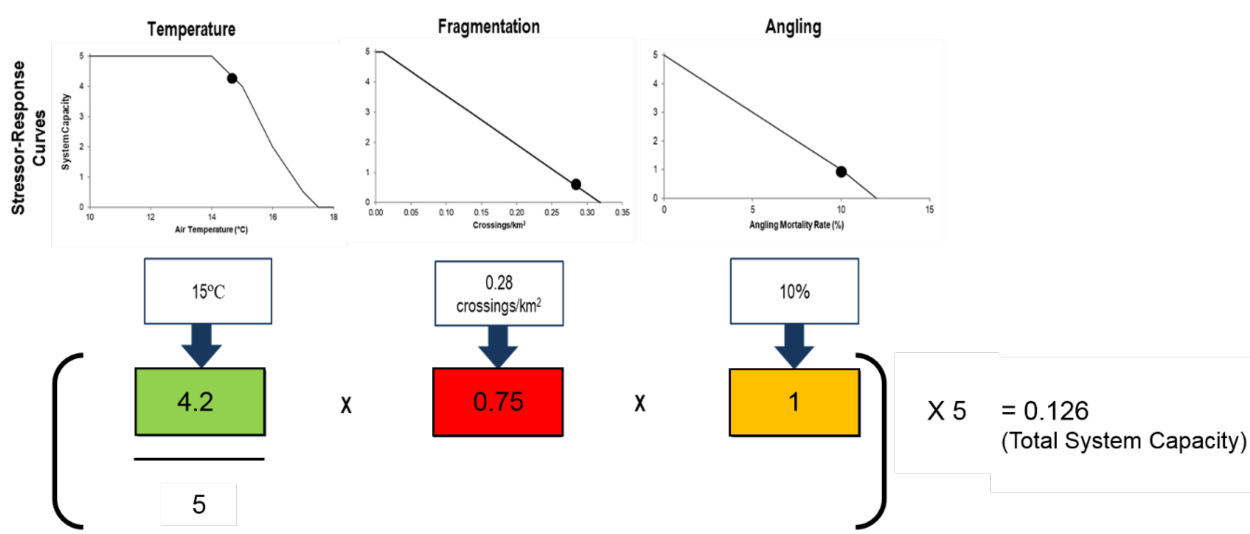

Figure 3. Illustration of the multiplicative effect of three hypothetical stressor-response curves (impacts) on predicted cumulative adult status. Figure adapted from Reilly and Johnston (pers. comm.).

Alberta's cumulative effects Joe Models are not designed to be complex ecosystem-level models that capture synergistic or antagonistic interaction among impacts. They are also not meant to replace localized action plans requiring tactical, fine-scale, specific-site data. Rather, these are strategic, population-level models using the best available science to create reasonable hypotheses of cumulative effects and management actions. As such, the output from these models are treated not as forecasts, but as best available hypotheses whose predictions need to be tested and validated.

Modelling Software

Although many computer programs can be used to create a cumulative effects model and quantify population responses, Alberta fisheries biologists generally use STELLA® modelling software (Richmond 2004). STELLA® allows users to quickly create system models in an interactive, real-time setting, such as stakeholder-participation workshops. Further, it easily allows the creation of models to run scenarios (e.g., climate change, complex mitigation actions). For Alberta fisheries biologists this provides a unique opportunity to create a cumulative effects model of impacts in a user-friendly platform that can be readily explained to stakeholder groups. This software and modelling process has been successfully used within Alberta Fisheries Management for over 20 years.

How to Build a Cumulative Effects Joe Model

Building the Stressor-Response Curves

As described above, Alberta's cumulative effects Joe Models rely on a series of combined stressor-response curves to characterize the effects of impacts and limiting factors on system capacity. While the model structure is quite simple, often participants have initial difficulty quantifying the expected complexities of individual stressor-response curves. A common initial response is, for example, "We don't have all the data to know the exact shape of the stressor-response curve for August flow".

Alberta fisheries biologists have found it useful to build out the preliminary model in a workshop-style setting, and then make refinements and finalize details of the model by analyzing data and performing an in-depth search of any available and relevant scientific literature. Workshop attendees could then further comment on, and possibly refine the model in follow-up meetings. Dependent on objectives, workshop invitees could include provincial fisheries biologists, species experts, and select stakeholders. This allows all participants to list and explore the stressors they feel are important and then collaboratively build stressor-response curves.

For instance, to create the Joe Bull Trout model, participants at a 2017 workshop included stakeholders concerned with key impacts listed in the Bull Trout Conservation Management Plan (ASRD 2012). Participants discussed these, as well as other known limiting factors found in the literature, and some advocated for including impacts they were personally passionate about. Participants were then challenged to quantify the degree to which each of the stressors (acting independent of other stressors) was assumed to affect the status of an adult Bull Trout population. If for example, a participant was adamant that Bull Trout declines are largely due to summer water withdrawals from creeks by industry water trucks holding TDL's (temporary diversion licenses) they would need to appropriately support their claims by drawing the quantified population-level stressor-response curve. After this preliminary workshop bounding the stressors to be included in the model, fisheries biologists begin the lengthy task of combing through data and scientific works to support, refine, identify key uncertainties, and document the rationale for individual stressor-response curves. A second, lengthy and equally important task is required to quantify the stressor, including its expected value and precision (e.g., allocated water volume for all TDL licenses in the above example). Quantifying the stressor and its uncertainty can be completed through a variety of methods including measured field data, regulatory documentation (e.g., licensing), GIS analysis, or modelling.

Using Heuristics and Fearing Cognitive Bias in Building Stressor-Response Curves

It is common for fisheries biologists and stakeholders to initially view building a stressor-response curve as impossibly daunting, e.g., “How can we possibly know the exact shape of this curve? It will take years of research!” A useful technique to solve this impasse is to break the complex question into smaller, simpler, and focused questions. These simpler questions designed to help answer complex issues are termed ‘heuristics’ (Tversky and Kahneman 1974).

For example, in a modelling workshop, a toxic chemical was proposed to have a potential population-level impact. The participants understood its lethal effects but felt that the precise intricacies of a stressor-response curve were too complex to understand. Instead of this complex question, they were asked if the absence of the chemical implied no effect. This was accepted as obvious, and the corresponding stressor-response was graphed as a point at “0 chemical concentration = maximum system capacity”. The next heuristic question was “At what concentration is complete death of all fish expected?” Participants knew from laboratory experiments that 20 ppm was toxic, so that corresponding stressor-response was graphed as a point at “20 ppm = no system capacity”. The next heuristic question was “What might the potential effect of 10 ppm be on a population, more specifically, is it likely higher, lower or equal to the two expected points?” The consensus answer was that it was likely higher than the linear line, but the exact value was uncertain. The resulting stressor-response curve was hyperstable, with clear endpoints (Figure 4). Depending on the level of the stressor (e.g., toxic chemical ppm), this level of detail for the stressor-response curve may be sufficiently adequate.

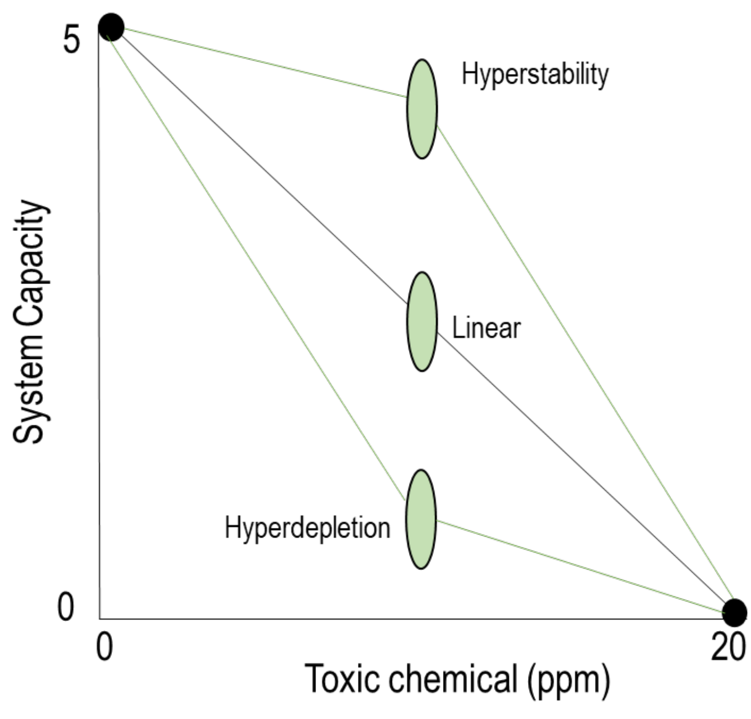

Figure 4. Theoretical stressor-response curve of a toxic chemical, with known endpoints and uncertain mid-points. Three possible types of curves are presented and participants can decide on the most likely relationship. By answering a few simple questions (heuristics), a more complex question can be logically addressed (Tversky and Kahneman 1974). The reference condition (maximum system capacity) is represented in this example as 5.

Although highly useful, reliance on heuristic questions to resolve complex issues can sometimes lead to severe systematic errors (Tversky and Kahneman 1974). Cognitive biases of representativeness, availability, and anchoring will undoubtedly influence stressor-response curves derived largely from opinion and heuristics. Was the curve influenced by representativeness (e.g., is the fish population under consideration similar or different from the population that was thought of to define the curve)? How does availability influence the curve (e.g., was a recent news event such as a major oil spill in a distant land influencing the participants view of oil spills in the local river under question)? Have the participants fallen under the bias of anchoring (e.g., did the first value that was mentioned create an artificial baseline for scaling the rest of the curve)? Active consideration and avoidance of these common (if not ubiquitous) cognitive biases is necessary to avoid making judgements that seem logical during the discussion at the moment, but may be markedly different with different participants, or under different conditions. Documentation of the reliability and confidence of stressor-response curves derived from heuristics is absolutely necessary in later stages of Joe Modelling, to define how inherent uncertainty affects the robustness of conclusions.

The importance of allowing participants adequate time and opportunities to describe their concerns, their world-view, and their concerns about consequences cannot be overstated. The value of the Joe Modelling process is that it can be built with the full participation all peoples, needing no expertise in western science or mathematics. Beliefs about cause and effects can transparently and clearly be used to influence the initial model output. Nonetheless, a functional model must adhere to best available mathematical and scientific relationships of cause and effect, and not unsupported opinions.

Causation vs. Correlation

An important challenge is that the effects of each added stressor need to be thought of as independently acting on a population, and must be a causative hypothesis rather than a correlative relationship. That is, when developing the cumulative effects model, fisheries biologists should be careful to include the underlying mechanism hypothesized to be the cause. Analogous to impact statements, impact models and pathways diagrams of cumulative effects assessment (Hegmann et al. 1999), this is to ensure they do not 'double dip' and accidentally capture the effect of a single stressor multiple times. For example, road crossings can cause immediate and long-term effects on fish populations by providing increased angler access, 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). However, this is correlative and cannot be captured in a single stressor-response curve. Rather, stream fragmentation from hanging culverts (MacDonald and Davies 2007; Bouska and Paukert 2009; Norman et al. 2009), increased sedimentation (Wellman et al. 2000; MacPherson et al. 2012), and higher angling mortality given road access (Gunn and Sein 2000) all likely contribute to declining fish populations. Therefore, when developing the stressor-response curves for each of these causative factors, the effect on the fish population is independent from the other two stressors. Management actions must then be connected back to their effect on stressor levels. Continuing the example, a management action that was to reduce road crossings must then separately account for the change in each stressor related to road crossings (i.e., fragmentation, sedimentation, and angling mortality).

The benefit of developing the cumulative effects Joe Model using this approach is that it forces all participants to be very clear about their conceptual models of how the system functions, the key impacts to fish populations, and what type of information they are based upon (i.e., personal anecdotal data versus empirical evidence) (Reilly and Johnston pers. comm.). This tends to move the process away from being argumentative to more of a structured exploration and learning exercise.

It's important to note that when stressor-response curves are designed, they need to be developed at the appropriate scale in space and time. Data describing a specific stressor needs to be available to populate the model. The stressor of the impact will be at the chosen population scale (i.e., watersheds or individual lakes) and the response to that impact is the effect on an entire population. For example, a single high silt event on a Walleye spawning shoal in the spring could be catastrophic for any eggs deposited, but may have no population-level effect if there are many other undisturbed spawning shoals in the lake. Similarly, an invasive species may be pervasive and a major impact to a native trout species within a single isolated stream reach, but at a larger watershed-scale the effect on the native trout population is relatively minimal. As fisheries biologists work through the development of impact stressor-response curves, the importance of maintaining the appropriate population-level response in space and time is crucial.

All stressor-response curves are developed using the best available scientific information at the time, including analysis of spatial data (i.e., in-house GIS or using ALCES Online©), fisheries data available in the provincial Fisheries and Wildlife Information Management System database (FWMIS), and consensus of professional opinion developed during workshops. For each curve, a formal sensitivity analysis of both the input data and the hypothesized stressor-response relationship should be conducted (for more details see the modelling uncertainty and making robust conclusions section of this report). It is expected that there will be refinements to the curves and potentially more (or less) complex interactions between impacts as new fisheries information is collected and hypotheses are challenged during adaptive management and recovery actions.

Incorporating Traditional Knowledge

Incorporating Traditional Ecological Knowledge (TEK) into the complexities of cumulative effects management and resource management holds considerable difficulties and important benefits (Berkes et al. 2000; Usher 2000; Houde 2007). These include the complexity of meeting important Canadian legislative requirements when dealing with species at risk (Mooers et al. 2010). A major difficulty of using modelling in a context of Aboriginal Traditional Knowledge (ATK) has been the perceived contrast between the holistic, interrelated cosmology of Indigenous peoples versus the isolated, mechanistic focus of western science (Tsuji and Ho 2002; Berkes 2012).

Using Joe Modelling has, in our limited experience, conformed more closely to systems-approach, holistic view of human and fish ecosystem relationships, as contrasted to our experiences with single-species population dynamics modelling. In this sense, we have used Joe Modelling workshops to integrate useful parts of western science into ATK, rather than the opposite path of attempting to bring fragments of ATK into western science. As Moffa (2017) describes, an unfortunately dominant theme assumes traditional knowledge is subordinate to western science. Our use of the process of Joe Modelling instead initially assumes the holistic, interrelated view of a system is the underlying basis for understanding threats to that system. We ask, "What is it about fish you value?", and then ask, "What do you believe threatens those values?" The value of listening to participants, and being able to immediately and transparently connect their world view and understanding of threats to the cumulative system capacity is a key benefit of the Joe Modelling process. Participants can take pride of ownership of the process, and have a clear understanding of their results.

In spite of the apparent benefits and simplicity of using this modelling process in workshops with Indigenous groups, we stress the critical importance of respect and professional involvement in these workshops. Ensuring ATK is collected and used in a respectful and rigorous manner is mandatory for successful understanding and knowing (Ellis 2005; CEAA 2015). Experienced facilitators and social scientists should be involved where possible. Techniques such as the Delphi method (Sutherland et al. 2013; Mukherjee et al. 2015), semi-directive interviews (Briggs 1986; Ferguson and Messier 1997; Huntington 1998) or other structured decision-making tools are useful, but require experienced practitioners for application to Indigenous knowledge (Gregory et al. 2012).

Why Joe Modelling is not Population Dynamic Modelling

Dynamic models describe change in a system over time or space such that its future state depends on some aspects of its current state (Gurney and Nisbet 1998). Because the predicted response metric of the Joe Model is entirely independent of the model's current state, the model is static. Simply introducing a time-varying stressor does not make the model dynamic as there is no state dependence. Joe Modelling is similar to Habitat Suitability Index modelling, where the environmental habitat conditions are combined to form an index of population suitability (Hirzel and Le Lay 2008). Joe Models represent the population status under the simulated conditions of stressors, and not the time- or space-dependent dynamic process where the current state updates the future states.

Dynamic modelling is important at tactical-level recovery and management planning for at least two reasons. First, dynamic models provide a prediction on the recovery trajectory of a population in time or space. That is, a dynamic model addresses the question of "how quickly will a population respond?" Second, the dependence of future states on current, or even lagged, states in a dynamic model captures situations that would be impossible to predict from a static model. The simplest example of this is population extirpation. A dynamic model would predict an extirpated population remains extirpated regardless of stressor levels (unless there was immigration); in contrast, the Joe Model erroneously predicts recovery of an extirpated population if stressor levels are simply reduced. In Alberta, tactical-level issues of recovery under different management actions are explored using detailed models of species-specific population dynamics (Post et al. 2003; Sullivan 2003).

The strength of strategic-level static modelling is to rank stressors that are most likely affecting population recovery and focus management actions to test those predictions. Dynamic models should be used to address tactical level questions regarding recovery trajectories or depensatory processes that may arise from proposed management actions. Thus, there remains a level of onus for users of a Joe Model to ensure they are focusing tactical-level recovery efforts to support meaningful changes to fish population status.

Adding population dynamics to the Joe Models is simple. Because the response metric of Joe Models is analogous to population carrying capacity (system capacity) scaled by the reference condition (called the response metric K_{Joe} such that $K_{Joe} = \frac{K}{K_{max}}$, K is system capacity and K_{max} is the reference condition), a simple logistic growth model can be used to introduce population dynamics:

$$Z_{t+1} = \frac{K_{Joe} Z_t}{Z_t + e^{-r}(K_{Joe} - Z_t)}$$

where Z_t is the scaled population state ($Z_t = \frac{N_t}{K_{max}}$) in year t; Z_{t+1} is the scaled population state the following year; and, r is the intrinsic rate of population growth (Gurney and Nisbet 1998).

Although simple to implement, we have found dynamic versions of the model can reduce the transparency and broad understanding held by stakeholders for the model. Furthermore, the perceived benefits of increased accuracy with a dynamic version are, in our experience, not usually justified. Complex details (and major uncertainties) of population dynamics are often best addressed in a different phase using a Management Strategy Evaluation process (Holland 2010; Punt et al. 2014). However, a dynamic version of the Joe Model may be warranted given certain situations, most notably if stressors do not follow a chronic but rather a pulsed pattern (e.g., periodic droughts, oil spills) or feedback loops occur (e.g., dependence of the fishing mortality rate on population status).

Climate Change Modelling

The potential effects of climate change on the system capacity of a fish population can easily be simulated using Joe Modelling in a static format. In Alberta, the two main aspects of climate change that we have modelled as potentially affecting fish populations are temperature and precipitation; specifically increases in mean warmest month temperature, and changes in precipitation with associated effects on flows (including drought).

The primary data source we are using for climate change projections is from the software Climate WNA (Western North America) (Wang et al. 2016). We use the most recent version as published on the UBC Forestry climate data website. As of June 2019, this version is Climate WNA v.6.00. This program downscales large climate data sets such as PRISM and WorldClim to the local areas defined by the user. The climate data includes both historical data and future projections based on GCMs (general circulation models) corresponding to the Fifth Assessment Report of the IPCC (IPCC 2013). The output from these data and projections is a wide range of biologically relevant climate variables, such as growing degree-days, mean warmest month temperature, monthly precipitation, and frost-free period.

Climate projections for future periods are selected from 15 GCMs from Climate Model Intercomparison Project 5 (IPCC 2013). Typically, the scenario used in Alberta projections is the RCP 8.5 trajectory, which assumes that greenhouse gas emissions will continue to increase throughout the modelled period (2018 to 2080). A more optimistic scenario of emission peaking in 2040 and declining (RCP 4.5) can also be modelled. Future projections of selected climate variables are usually displayed as groups in 30-year time periods (i.e., 2011–2040 = 2025, 2041–2070= 2055, and 2071–2100 = 2085). Selecting the appropriate GCM is dependent on the purpose of the simulation, and can be the mean of the ensemble suite of GCMs, or a distribution of projections in the Joe Model to simulate a range of potential outcomes.

Increases in Mean Warmest Month Temperature

The Joe Models used in Alberta typically have an input variable of temperature, usually defined as mean warmest month temperature (MWMT). As such, the linking of climate change to the Joe Model simulation is trivial; the GCM projection becomes the new input. This may be done as a static value for a particular future time (Figure 5).

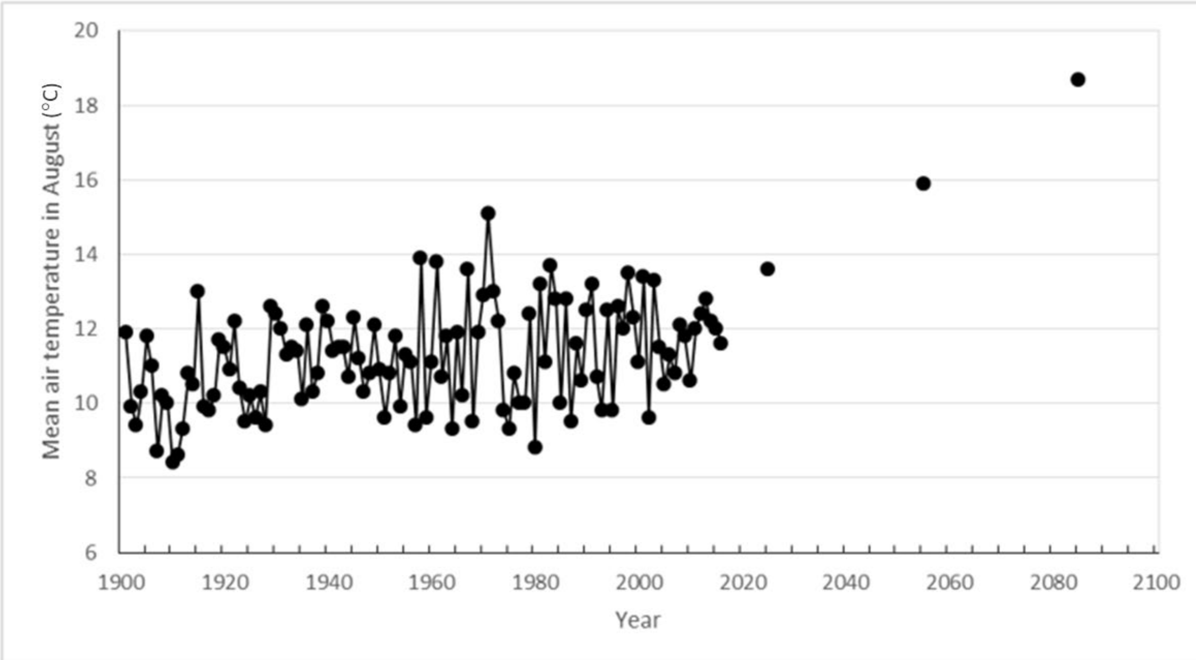

Figure 5. Historical and future temperature data for headwaters of Racehorse Creek (Oldman River watershed). Data was derived from Climate WNA v5.51, for 1901 to 2016, and is mean August air temperature. Future projections from the ensemble of 15 GCMs suggest that increases in air temperature may result in conditions unsuitable for the two species of native salmonids in this watershed (Westslope Cutthroat Trout and Bull Trout).

Linking the climate data and projections of MWMT to stream temperature is more complex. If the original Joe Model stressor is air temperature, obviously no conversion is necessary. If stream temperature is required, the MWMT must be related to water temperature.

Changes in Precipitation and Flow

The GCM projections in Climate WNA include potential changes to precipitation, scaled to season or month. In the Joe Models, input variables often include changes to summer or winter flow. In order to explore the possible changes to flow from climate change, the fisheries biologist must make assumptions relating precipitation to stream flow for the particular watershed of interest. This may be done empirically by relating the historical precipitation data from Climate WNA to historical flow data for that watershed. Future projections assume similar relationships persist. Fisheries biologists will need to explore the relationships between climate data and flow data to determine the most likely cause-effect correlations, such as seasonal precipitation patterns, influences of upstream watersheds, and the complexities of evapotranspiration with temperature changes.

A useful feature of Climate WNA is the historical precipitation data as used to understand the frequency of drought events. For example, a close relationship between spring and summer precipitation and low flow (drought) conditions in the stream might be assumed for a simulation. The historical data from Climate WNA for a specific watershed can be analyzed to determine the frequency of these events (Figure 6). The frequency of these drought events may be related via a stressor-response curve to adult fish density. Future projections of drought frequency would be associated with two factors: 1) precipitation; and 2) a trend of increased (or decreased) variance in drought events.

It is important to appreciate the high uncertainty associated with all these relationships and convey that uncertainty to any conclusions.

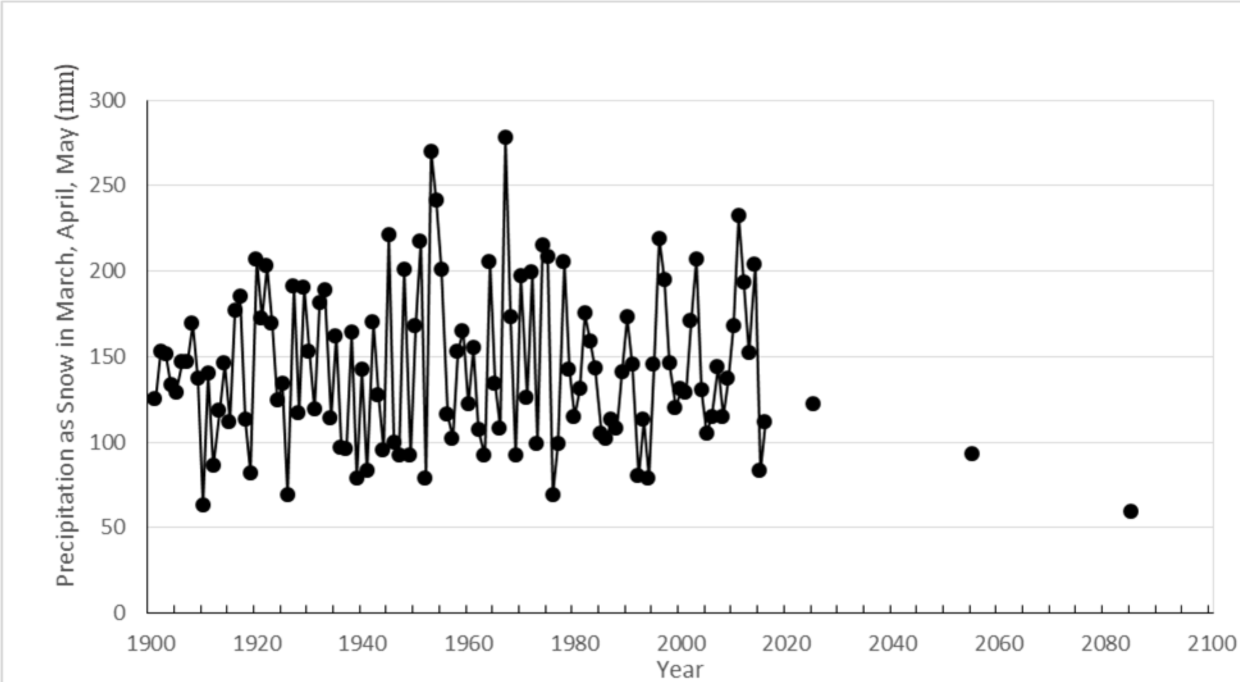

Figure 6. Historical and future precipitation data for headwaters of Racehorse Creek (Oldman River watershed). Data was derived from Climate WNA v5.51, for 1901 to 2016, and is precipitation as snow for March, April and May. Low flow events (droughts) that may have affected stream fish populations were observed in 2015 and 2016. Future projections from the ensemble of 15 GCMs suggest that decreases in precipitation may result in severe drought events.

Modelling Uncertainty and Making Robust Conclusions

Uncertainty is inherent in all aspects of managing fisheries (Ludwig et al. 1993; Hilborn and Walters 1992; Fulton et al. 2010). How many fish are in a lake? How many anglers will attend? How many fish will they catch? What catch is acceptable to anglers, and to sustainability? Each step of almost every fisheries management decision process relies on answers that have ranges of possibilities. This intrinsic uncertainty, however, is seldom considered when assessing the strength of a decision (Harwood and Stokes 2003), or if considered, is usually defined as vaguely qualitative, e.g., “Given all the unknowns, that seems like a reasonable quota.” Mathematical modelling and analysis, however, allows uncertainty to be addressed quantitatively.

A highly useful aspect of the Joe Modelling process is the ease of explicitly defining ranges of uncertainty in the stressor-response relationships as well as in the ranges of uncertainty inherent in the input parameters. By explicitly acknowledging this uncertainty and formalizing sensitivity analyses in Joe Modelling, fisheries biologists can improve their understanding of the potential consequences of uncertainty on the robustness of their conclusions. Critically, however, the model outputs are never “predictions with certainty”. They are simply the logical result of explicitly stated inputs. Key drivers might be entirely unknown, stressor-response curves may be unique and unknown, and input variables may change significantly from year-to-year. The value of modelling is to demonstrate the effect of uncertainty, not eliminate it.

The formal process of Management Strategy Evaluation (Holland 2010; Punt et al. 2014) is an excellent framework for exploring the uncertainty inherent in the variety of potential alternate management actions. These potential actions must be selected with stakeholder involvement and should include social, economic and legal considerations. Testing and proposing the potential range of outcomes of the selected and proposed strategies, however, fully involves the role of the modellers and the quantitative aspects of the model.

Demonstrating uncertainty by using Joe Modelling has, in our experience, taken two related pathways; formal sensitivity analysis, and robustness analysis. Sensitivity analysis is done internally by the model builders and is primarily used to understand and strengthen the model structure. Robustness analysis is generally done externally, and primarily used in decision-making settings (e.g., public meetings, workshops, regulation design meetings).

Sensitivity Analysis

Formal sensitivity analysis can be conducted during the model design phase to determine which stressors cause the most sensitive changes to the model output. The objective is to focus work, such as literature reviews or field studies, on parameters where reducing uncertainty can be most efficient and effective. In brief, understanding the sensitivity of Joe Modelling to stressor levels and stressor-response curves could be done by:

1. Adjusting stressor levels (input parameters) by a constant proportional amount to determine which have the largest output effect. Stressors with the largest effect require particular attention to uncertainty in their input values.
2. Redefine the stressor-response curves to produce a range of curves around the most likely curve to determine its effect on output. However, this is better tackled through more formal methods such as Bayesian networks (Scutari and Denis 2015) and is beyond the scope of the current Joe Models.

Within the modelling software of STELLA®, sensitivity analysis involving multiple input parameters (stressor levels) can be conducted under the “Run Sensitivity Spec” options. Interpretation of multiple runs of multiple variables, however, is complex. A logical, step-wise process of assessing variables and parameters is necessary.

Robustness Analysis

Ultimately, the key objective of uncertainty analysis is to determine if a management action suggested by the model output is robust to the uncertainty in critical input parameters and curves (e.g., will the suggested action increase fish by the desired amount *under most situations* of uncertainty?). We refer to this as robustness analysis. It directly addresses the requirements quantifying uncertainty within the Management Strategy Evaluation process. Generally, more uncertainty results in the necessity of more precaution, and therefore more costs in management actions. Costs can be social, political, and economic. The value of robustness analysis in Joe Modelling is that this trade-off (uncertainty and costs) can be made explicit to decision-makers.

A simple example of demonstrating the effects of uncertainty on management robustness, and its inherent trade-off is shown in Figure 7. Joe Bull Trout is used to simulate a typical Alberta east slope watershed with multiple stressors, but with sediment, stream fragmentation, and angling effort as the key and controversial stressors. The first simulation shows the range in potential output (i.e., system capacity, a measure of adult Bull Trout density), when one variable, sediment, is modelled with variance. Possible outcomes for system capacity range from approximately 0.5 to 1.5, with a reference condition maximum of 5. The second simulation shows a slight increase in the range of possible outcomes when two variables (sediment and fragmentation) are each modelled with variance. The third simulation shows a larger range of outcomes (from approximately 0.2 to 1.8) when all three variables, sediment, fragmentation, and angling effort, are each modelled with variance (in this example, all variables are modelled with variance represented by standard deviation of 25% of the mean).

The next analyses shown in Figure 7 deal with two management actions; a minor versus a major reduction in angler effort. The management question to be asked is, “What is the least action needed for obvious improvement?” In this sense, “obvious improvement” is the management necessity of having fish populations increase enough to be detected in a few watersheds (e.g., low sample size of experimental units), be obvious to anglers and politicians, and be likely to occur in spite of unexpected environmental variation. All management actions have social, political, and economic costs. Justifying these costs with the expected outcomes therefore cannot be entirely science-based. The results in Figure 7 demonstrate a visual method of explaining these trade-offs. The red line shows the possible outcome of a minor reduction (25%) in angler effort, while the blue line shows outcomes from a major reduction (90%) in effort. Fisheries biologists must decide if the possible range of outcomes is worth the costs of minor versus major angling reductions. The value of a clear visual explanation of robustness of a management action cannot be overstated.

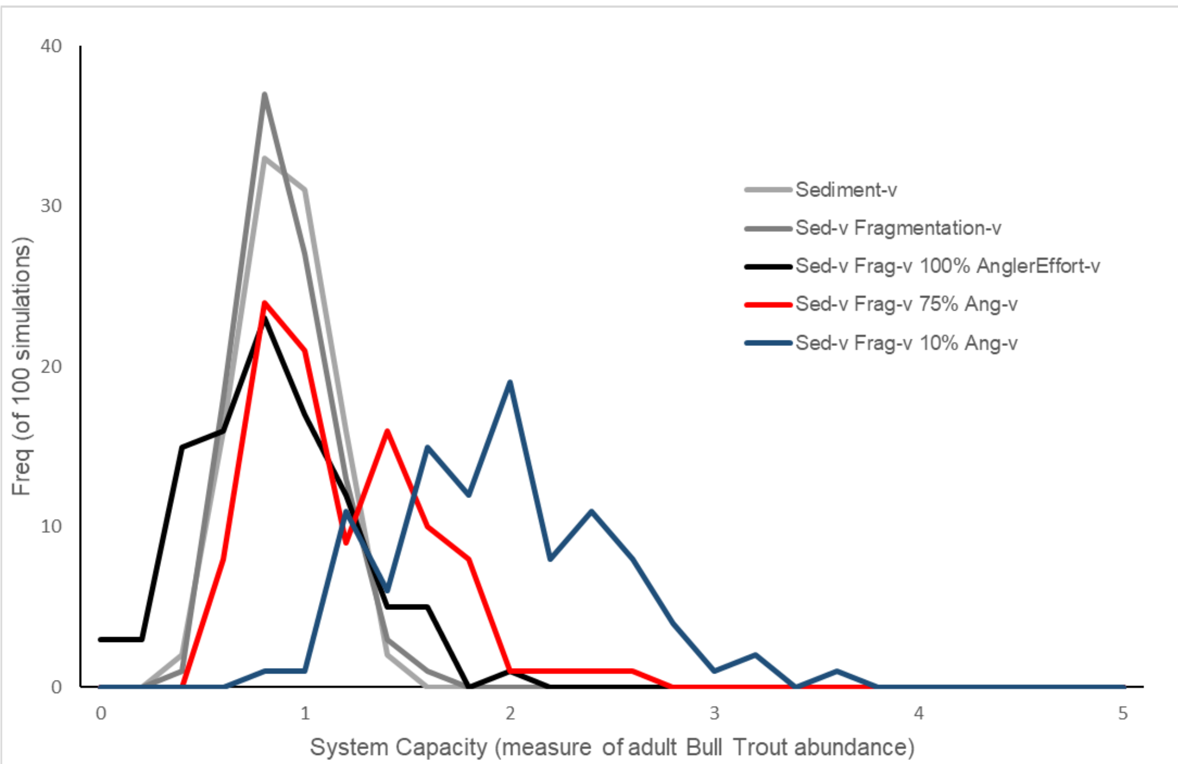

Figure 7. Robustness analysis of three threats (sediment, fragmentation, and angling) and their variance (-v) in a Joe Bull Trout model to two management actions; a minor (25%) and a major reduction (90%) in angler effort. The simulation represents a typical east slope watershed with multiple stressors and a heavily stressed Bull Trout population.

In Figure 7, the grey to black lines show how the range of possible outcomes increase by combining variances of stressors. The light grey line of “Sediment-v” represents the possible outcomes of Bull Trout system capacity with variance in the sediment stressor, and all other stressors simulated with no variance. The darker grey line represents outcomes with variance in sediment plus variance in fragmentation. The black line represents the increase in outcome uncertainty with the addition of variance in Angler Effort.

The robustness analysis is shown by the red and blue lines. The red line represents Angler Effort being reduced by a minor amount (to 75%), with expected system capacity being higher than 95% of the outcomes from original Angler Effort (black line) in only 21% of cases. The blue line represents Angler Effort being reduced by a major amount (to 10%), with expected outcomes being higher than 95% of the outcomes from original Angler Effort (black line) in 80% of cases. The major reduction in Angler Effort (blue line) was also higher than 95% of the minor reduction in Angler Effort (red line) in 58% of the cases.

Scoring Data Reliability

Reliability of Joe Modelling input stressors are evaluated in the previously described robustness analysis, while for each of the three metrics assessing population integrity (CAD, CID and HAD) the overall quality of the data used to complete the population status assessment is evaluated. This addresses how likely the assessment is erroneous due to incorrect, inaccurate, or lacking data. The three monitoring metrics considered are monitoring quality, monitoring quantity, and monitoring timeliness. Specific definitions for each of the data reliability scores will depend on scale and the species being assessed. These should be defined and summarized for each species assessed. For example in the Bull Trout FSA, for data quality, standardized sampling programs with a randomized sampling design throughout the watershed would receive a score of 5, while an assessment made primarily on professional opinion would receive a 1. For data quantity, greater than 50 surveys sites from standardized sampling programs would be scored a 5, while less than 25 surveys sites and/or professional opinion would get a score of 1. Lastly, for data timeliness, data collected in the last 5 years was scored the highest quality while older data (>20 years) was scored the lowest (1). Note that in cases where the focal fish species was never surveyed in an HUC, a not applicable score or 'n/a' is permitted for the timeliness metric. Table 3 can be used as a guideline for evaluating data reliability.

Table 3. Alberta Fisheries Sustainability Assessment rankings for the quality, quantity, and timeliness of monitoring data used to assess population status.

FSA Scores for Monitoring Quality (Is the data precise and accurate?)
1 = Imprecise and inaccurate 2 = Precise but inaccurate data 3 = Accurate but imprecise 4 = Likely OK 5 = Precise and accurate
FSA Scores for Monitoring Quantity (Is sufficient data available to evaluate this metric?)
1 = No data 2 = Insufficient data 3 = Moderately sufficient data 4 = Nearly sufficient data 5 = Sufficient data

FSA Scores for Monitoring Timeliness (How likely is it the population being assessed is functionally different from when the last field data were collected?)
n/a = The focal fish species has never been surveyed
1 = Extremely different
2 = Very different
3 = Moderately different
4 = Slightly different
5 = Not different

Cumulative Effects Model Template

While the creation of a cumulative effects Joe Model may seem daunting at first, several general themes of impacts and limitations can be used to guide the creation of a new model. To date, Alberta fisheries biologists have generated models for several fish species (e.g., Bull Trout, Westslope Cutthroat Trout, and Athabasca Rainbow Trout) and work is underway to complete other priority species. After completing this work, we have found that stressor-response curves can generally group into the following broad categories: habitat impacts (water quantity and flows, water quality, fragmentation, habitat loss, temperature), harvest impacts (angling mortality, by-catch mortality, research mortality, Indigenous fisheries), and invasive species impacts (disease, non-native species and hybrids) (Figure 2). A fourth category of limitations that is not human-caused impact, but can be considered a factor that “impacts” the sustainability of a population is natural limitations.

In this section, we use Bull Trout as an example and discuss each one of these broad impact categories and provide rationale and stressor-response curves. While not all categories may be applicable and some additional categories will need to be included, this section can be used as a general guiding template for the creation of a cumulative effects Joe Model. After the completion of individual models, we suggest writing individual summary reports that provide the species-specific stressor-response curves, impact rationale, and data reliability.

The Joe Model uses a proportional response metric (e.g., 0–100) which is system capacity (a measure of population density) scaled to a maximum reference condition density. The output from the Joe Models is the proportional response metric that can be scaled back to the population density (system capacity) by multiplying with the reference condition. The use of a 5 unit scale, with 5 being the maximum reference condition density, in this section is a hold over from when the Joe Models worked directly with FSA scores. However, as the FSA score is categorical it should not be treated as continuous, and thus, not used directly in the Joe Models. Furthermore, translating FSA scores into a continuous response metric is not intuitive given non-linearity of the FSA categories (Table 1). Thus, the 5 unit scale in stressor-response curves below do not equate to FSA scores but rather can be simply and intuitively converted to the dimensionless response metric (i.e., proportion of the maximum reference condition) through division by 5.

Habitat Impacts

Water Quantity and Flows

Fish rely on water to complete various life history processes, and dependent on species, the required quantity and flow regime will vary. Here we present the rationale supporting the inclusion of changes to water quantity and flow regime.

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 8A). 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 8B). 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.

A) B)

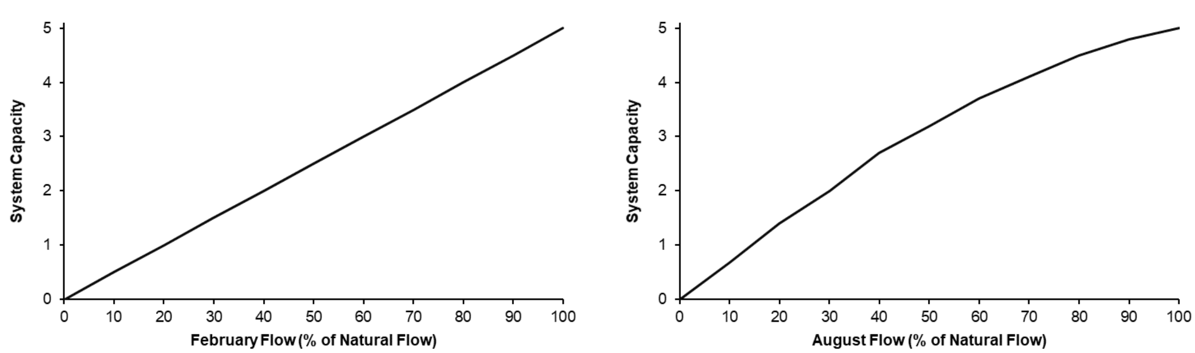

Figure 8. Stressor-response curves depicting the expected relationship between changes in February (A) and August (B) flows and the system capacity of Bull Trout populations. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Flow Regime: Modification of Timing and Frequency of Peak-Flow Events

For fish in flowing waters, changes in the magnitude and frequency of peak flow events may impact the sustainability of 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. In several lotic trout Joe Models, fisheries biologists captured changes to flow in an index of potential hydrologic change to provide a qualitative description that captures the differences in the magnitude and frequency of peak flow events relative to the historic condition of the watershed. 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 9A). 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 9B).

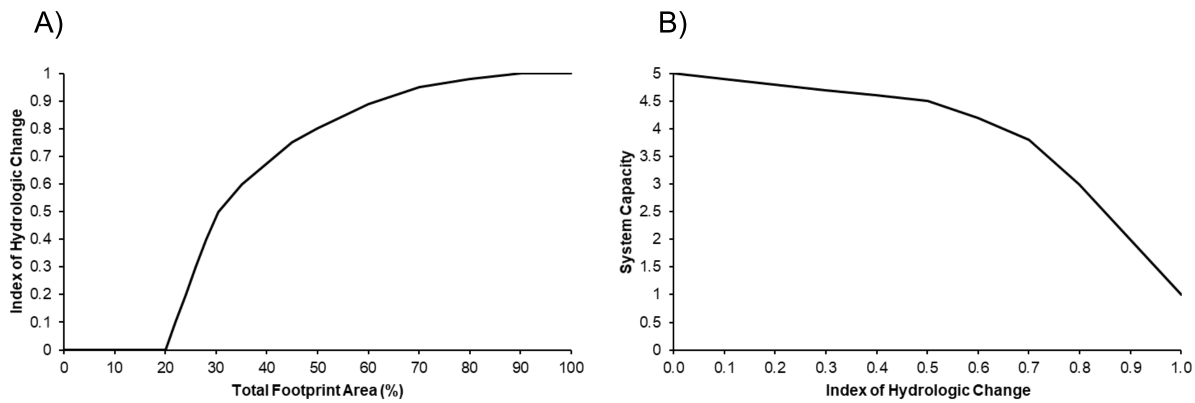

Figure 9. The hypothetical relationship between total human footprint area in a watershed and the Index of Hydrologic Change (A) and the predicted effect of hydrologic change on Bull Trout population system capacity (B). System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Water Quality

This category of impacts was included to capture changes to water quality (from the natural state) that would affect fish populations. Among others, this could include nutrient inputs (phosphorus or nitrogen), sedimentation and/or contaminants. For brevity, we only provide rationale for the inclusion of phosphorus.

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 British Columbia 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 (Jacobsen and Marin 2008; Chung 2013) and overall anoxic conditions that can impair biodiversity (Meijering 1991).

For example, degraded stream habitats and winterkill conditions in Alberta foothills were correlated with theoretical increases in phosphorus runoff due to land use at the watershed scale (Norris 2012).

Potential impacts caused by excessive phosphorous may be masked by other concurrent activities, making it difficult to establish the specific effects of phosphorous on fish. High variance is often observed when investigating complex ecological relationships (e.g., Cade and Guo 2000; Dunham et al. 2002a) and can be an indication that the dependent variable (i.e., fish system capacity) is affected by more than one factor (i.e., phosphorous) and that these other factors have not been measured or considered in the model (Cade and Noon 2003). In such cases, the relationship between the dependent variable (i.e., response) and factor of interest (i.e., stressor) is better represented by the rate of change near the maximum response using quantile regression, rather than the average response (Cade and Noon 2003; Figure 10A).

To quantify this relationship in the Bull Trout cumulative effects model, a stressor response curve for phosphorous was derived from the 90% quantile regression between current adult Bull Trout FSA score converted to the proportion of maximum system capacity (Table 1) and an index of phosphorus (i.e., the ratio of current export to undisturbed phosphate export, see description below). A range of quantiles was evaluated with the 90% quantile selected as it was statistically significant and most likely to exclude the effect of unmeasured factors on adult status (Cade and Noon 2003). To account for non-linear patterns in the stressor-response curve, additive quantile regression (Koenker 2017) was utilized. The smoothing parameter (lambda) in the quantile regression model was adjusted to produce the minimum AIC with the curve still intersecting with the x-axis. A stressor-response curve was derived from the quantile regression results by adjusting the statistical curve so a system capacity of 5 occurred at a phosphorous index score of 1 or less and the inflection point and x-intercept preserved (Figure 10B).

A) B)

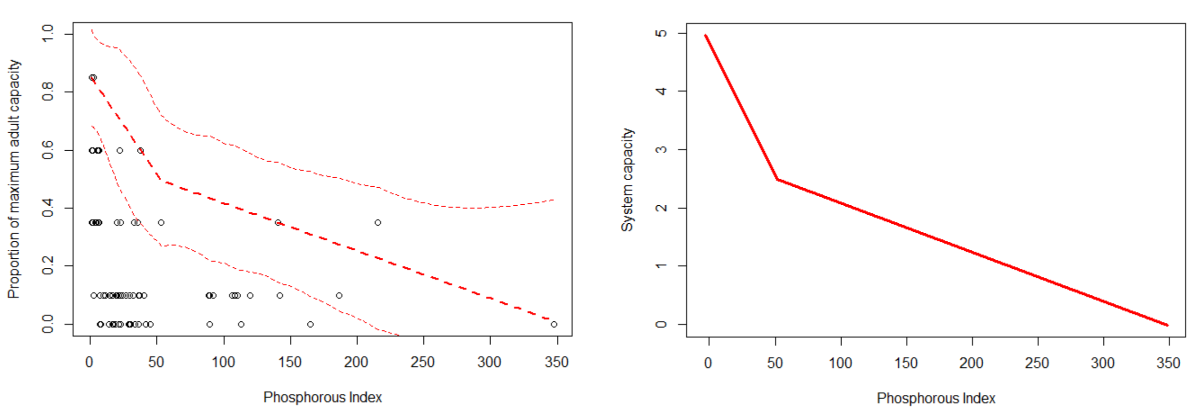

Figure 10.The observed relationship between adult Bull Trout status (proportion of maximum adult density) by HUC 8 watershed (AEP 2013) and phosphorous index (A) and the derived stressor-response curve used for modelling (B). Phosphorus index represents the ratio of current export to undisturbed phosphate export. The red line in the observed data (A) is the 90th quantile regression using general additive models, which provides a statistically based non-linear fit to the data. The 95% confidence intervals on the 90th quantile regression line are also shown. The derived stressor-response curve (B) was based on the statistical curve but adjusted so that a system capacity of 5 occurred at a phosphorous index score of 1 and an FSA score of 0 occurred at a phosphorous index score of 350. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Fragmentation

Culverts can fragment fish habitat and impede fish movements necessary for growth, survival, reproduction, gene flow, and colonization. Below we present the stressor-response curve for the effect of stream fragmentation from crossing structures on Bull Trout populations.

Bull Trout are a migratory fish that require connectivity between key spawning, rearing, feeding, and overwintering habitats. Habitat connectivity permits the exchange of individuals between populations, facilitating gene flow and the re-establishment of depleted populations. Habitat fragmentation and loss occurs when culverts and other stream crossing structures are improperly constructed or maintained. These structures may represent complete upstream and downstream movement barriers or partial barriers depending on stream flows. Several audits of crossing structures in 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).

In the absence of a provincial crossing status dataset, the assumption was that relatively high numbers of road crossing and densities indicate a greater risk of habitat fragmentation. There is a paucity of studies directly measuring population-level impacts of fragmentation on Bull Trout, 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 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 (0.257 crossings/km^2) to indicate the greatest degree of extirpation risk (Figure 11).

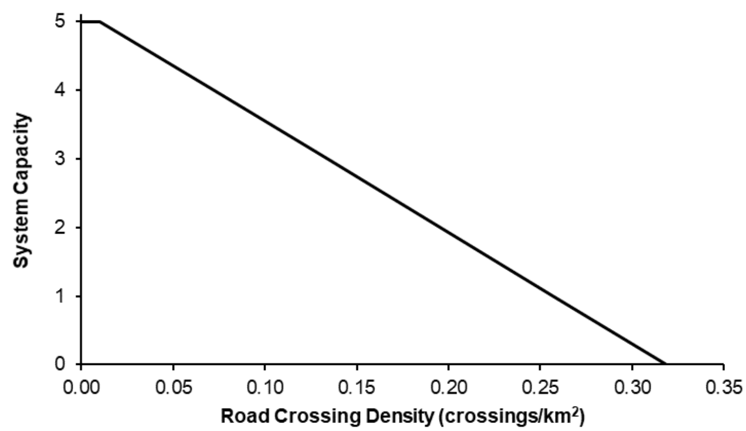

Figure 11. Stressor-response curve depicting the expected relationship between road crossing density within a watershed and the system capacity of Bull Trout populations. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Habitat Loss

Habitat loss and degradation is often cited as a major impact and limiting factor for fish populations (e.g., ASRD 2012). Here, we present the rationale for the inclusion of habitat loss in the Bull Trout cumulative effects model. Of note, the effects of anthropogenic habitat degradation are captured in other stressor-response curves, so this category is exclusively meant to capture direct habitat loss.

Direct habitat loss can occur in part of the Bull Trout range. This type of 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. Strip-mining for coal in this region has deleted some stream sections. They may be replaced with open-pit lakes, or with channeled stream analogs (i.e., a ditch).

The stressor-response curve for habitat loss is simply the percentage of stream habitat lost or converted to non-Bull Trout habitat (Figure 12).

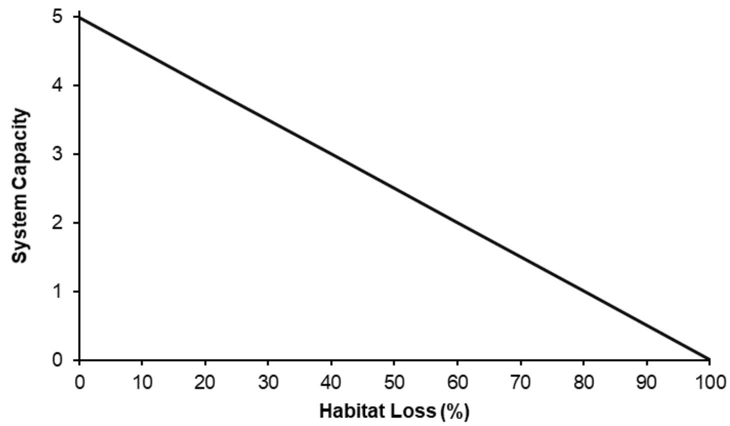

Figure 12. Relationship between habitat loss and the effect on Bull Trout system capacity. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Invasive Species Impacts

Disease

Dependent on fish species and location, there are numerous diseases that could affect a fish population. As an example, we discuss whirling disease (*Myxobolus cerebralis*) and its effect on Alberta Bull Trout populations.

In 2016 whirling disease was confirmed as present throughout the Bow River drainage in Alberta and it has since been confirmed in other provincial watersheds. Bull Trout are susceptible to whirling disease, though they are thought to have higher resistance than some other species, such as Rainbow Trout (Hedrick et al. 1999). Populations in the Bow River watershed are assumed to be exposed to whirling disease and it is likely that populations in other nearby watersheds will be exposed as well if whirling disease spreads. Sullivan and Spencer (2016) developed an age structured cohort model to estimate the effects of whirling disease on adult Bull Trout density. Juvenile annual mortality was estimated at 80% apart from the effect of whirling disease, and mortality was increased as the severity (scaled to combination of burden and prevalence) of whirling disease increased from low (82% mortality) to moderate (85% mortality) to high (87% mortality). These mortality rates reflect the lower sensitivity of Bull Trout to whirling disease than is noted for other species of trout. The structured cohort model was used to determine the effects of increased juvenile mortality on system capacity. The resulting system capacity was compared to system capacity with no whirling disease effect and scaled to a maximum of 5 to create a stressor-response curve (Figure 13). This is a strategic-level stressor-response, and as such does not include indirect effects of whirling disease such as growth rate or reproductive changes.

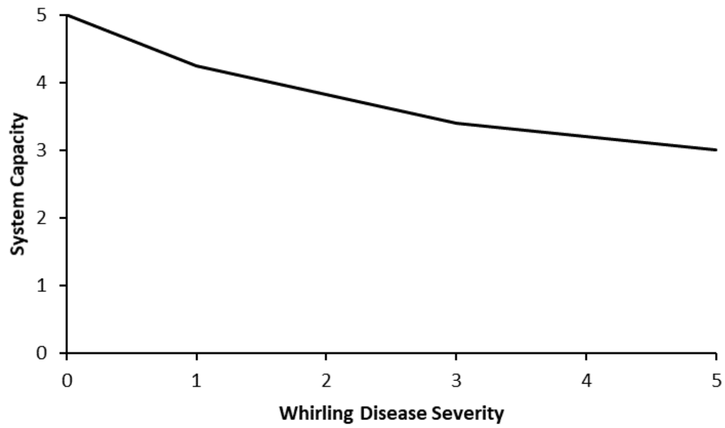

Figure 13. Stressor-response curve depicting the expected relationship between whirling disease effect (none = 0, low = 1, moderate = 3, and high = 5) and Bull Trout system capacity. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Exotic and Non-Native Species

Exotic species invasion is often identified as one the major impacts on North American inland fisheries as a result of human activities, either through replacement or displacement of native fishes (Volpe et al. 2001; Dunham et al. 2002b). An example of how Bull Trout competition with Brook Trout (*Salvelinus fontinalis*) was modelled follows.

Competition/Replacement: Brook 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). In this latter and expected case, the Brook Trout are therefore replacing niche vacancies from missing Bull Trout, rather than displacing existing trout. The stressor-response curve (Figure 14) evaluates the expected impact of Brook Trout on Bull Trout sustainability relative to the overall carrying capacity of the system.

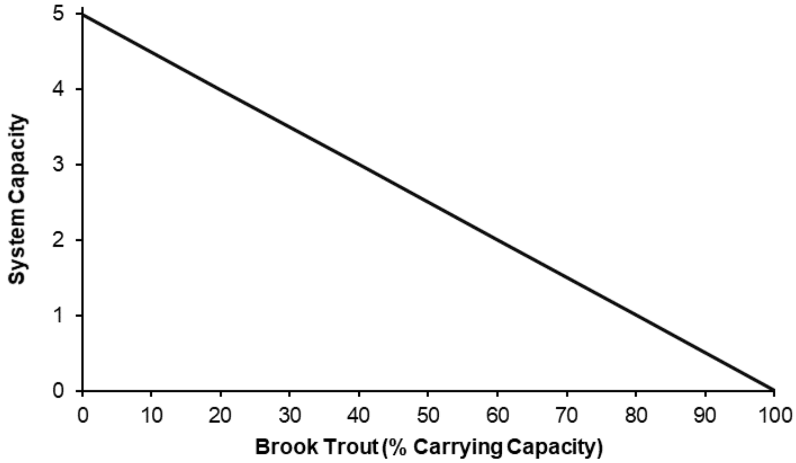

Figure 14. Stressor-response curve depicting the expected relationship between Brook Trout carrying capacity within a watershed and the system capacity of Bull Trout populations. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Hybridization: Brook Trout

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. 2010b). 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. The stressor-response curve (Figure 15) evaluates the possible impact of Bull Trout hybrids on Bull Trout sustainability relative to the carrying capacity of the system.

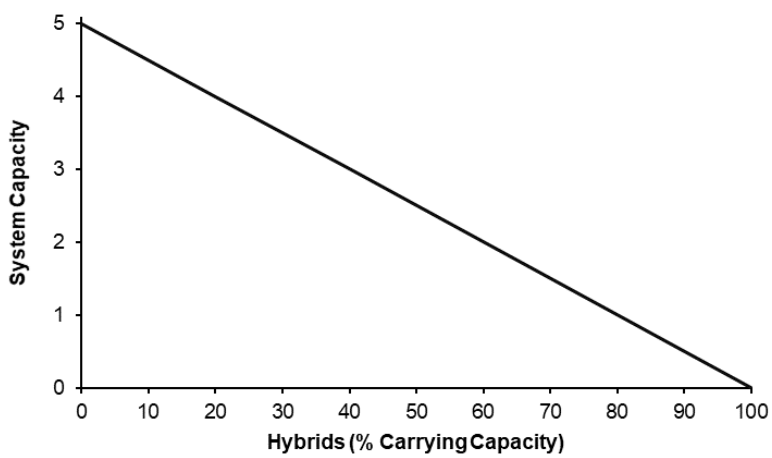

Figure 15. Stressor-response curve depicting the expected relationship between hybrid Bull Trout occupancy (%) within a watershed and the system capacity of Bull Trout. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Harvest Impacts

Mortality

In the Joe Models, direct mortality is often separated into natural causes, angling, entrainment, research and monitoring, and Indigenous harvest, although more variables can be added as required. Using these five mortality sources, the total annual mortality rate (A) can be calculated using the conditional rates of natural mortality (n), angling mortality (m), entrainment mortality (en), research and monitoring mortality (r), and Indigenous peoples harvest (i), by applying the following equation adapted from Ricker (1975):

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

The stressor-response curve for direct mortality (Figure 16) 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 exceeds 8% and extirpation occurring when additional mortality exceeds 12%.

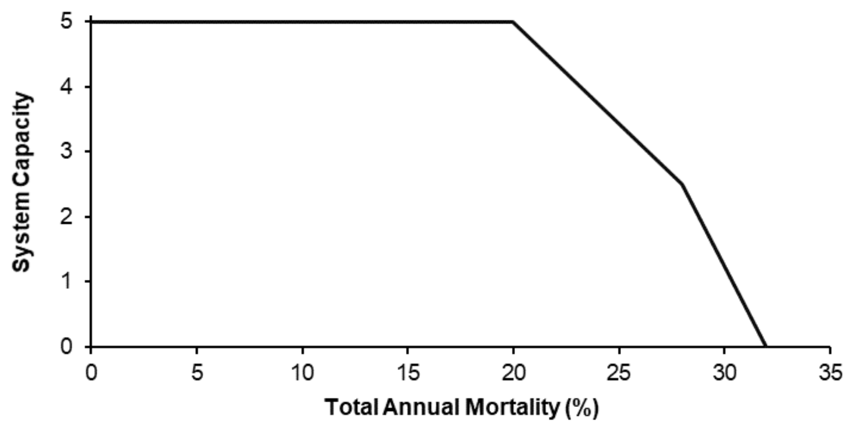

Figure 16. Stressor-response curve depicting the expected relationship between total annual mortality and the system capacity of Bull Trout populations. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Using Bull Trout as an example, we briefly discuss one of the sources of direct mortality.

Incidental Angling Mortality and Illegal Harvest

Bull Trout were legally harvested throughout the eastern slopes in accessible watersheds prior to the implementation of the province-wide zero harvest regulation in 1995 (ASRD 2012). Angling, however, may still represent a major impact to population sustainability from incidental mortality (i.e., mortality due to stress or physical damage from hooking or improper handling) in spite of catch-and-release regulations. Illegal harvest, either intentional or due to misidentification, may also contribute to population declines. The combination of incidental mortality and illegal harvest will be unsustainable if angling effort is sufficiently high (Post et al. 2003; Sullivan 2018). Past case studies demonstrate that in east slope streams and lakes, some but not most Bull Trout populations are capable of recovering relatively quickly (5–10 years) from an over-exploited state under zero harvest regulations and complete angling closures (Johnston et al. 2007; Sullivan 2014; Reilly et al. 2016). However, widespread recovery of Bull Trout populations has not occurred in the situation of catch-and-release regulations and high angler effort.

Impacts from Natural Limitations

In the absence of anthropogenic influences, fish species are naturally limited by other environmental variables. These limitations occur at varying spatial scales and include both biotic (e.g., productivity, fish community) and abiotic features (e.g., temperature, substrate composition, lake depth, lake size, stream velocity). These variables need to be captured in the cumulative effects model. As an example, we explore how stream temperature naturally limits Bull Trout.

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 17A). 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 17A) 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 13°C to 11°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 temperature on the system capacity of Bull Trout populations (Figure 17B).

A) B)

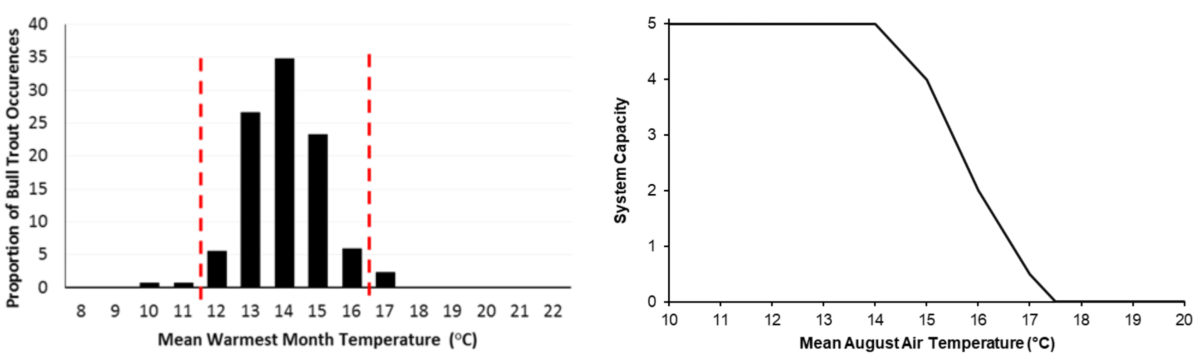

Figure 17. A) Thermal range of occupied Bull Trout waters within historic Bull Trout range. B) Stressor-response curve informed using thermal range data. This curve depicts the expected relationship between air temperature and system capacity of Bull Trout. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5. Not pictured: thermal profile for entire Bull Trout range.

Summarizing Model Results

Once populated, the output of a cumulative effects Joe Model is a predicted system capacity for a population of interest. While this is the primary model output, Alberta fisheries biologists have found that the model can also be used to summarize impacts and limiting factors into three major impact categories: habitat (loss and degradation), hybrids (non-native and exotic species) and harvest (sources of direct mortality). This can be a very useful graphic to quickly convey a complex message to the public (Figure 18).

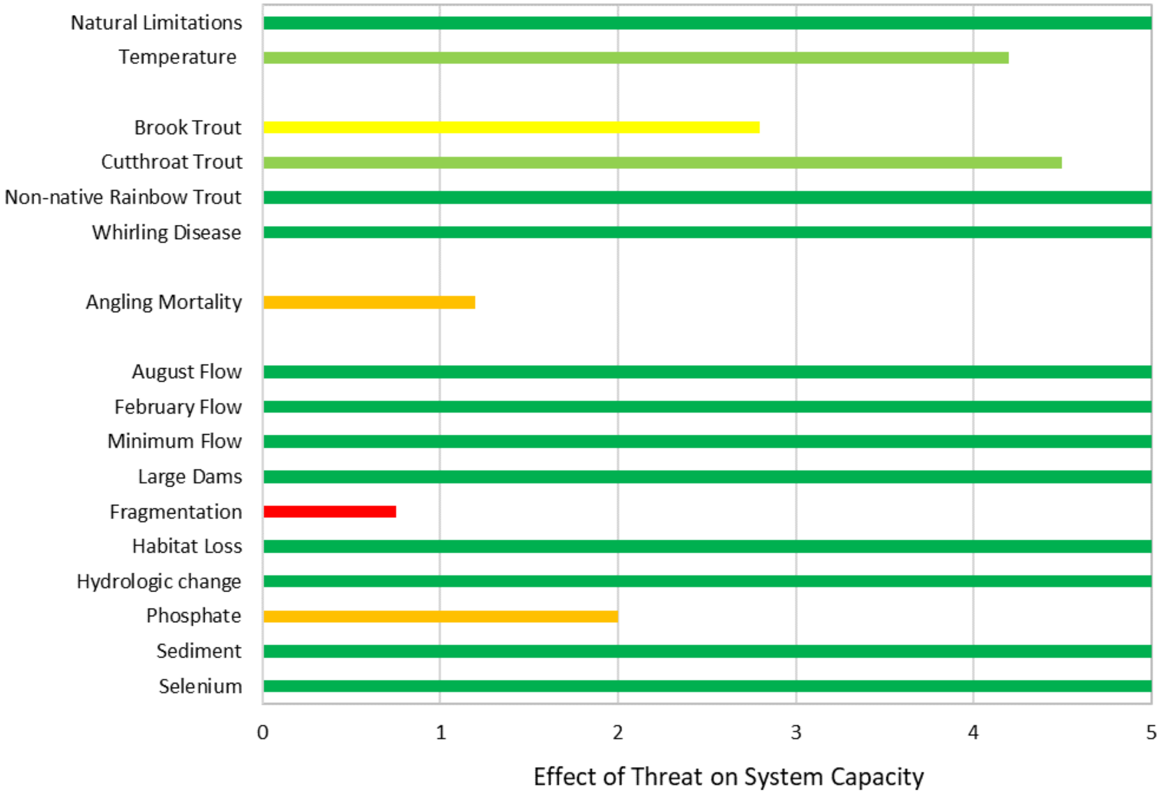

Figure 18. Output of a Joe Model depicting adult system capacity of each impact category. The bars reaching the top of the graph suggest little to no effect on adult density, and the lowest bars suggest the strongest negative effect. System capacity (depicted from 0–5) is a measure of adult density relative to a maximum attainable capacity of 5.

Model Validation and Adaptive Management: an Example

The fundamental goal of the FSA (assess status and assess threats) is to provide a framework for learning how we can recover populations at risk. Although this might seem overly pedagogical and the fundamental goal should be recovery, without learning our actions become inseparable from unexplained variance and their efficacy lost. FSA lends itself to an adaptive management approach with the Joe Models evaluating hypotheses and the FSA score providing a common means to publicly report status.

Bull Trout management in the Clearwater River Watershed of west central Alberta provides an example of this adaptive management approach (Figure 19). Several management actions or combinations of management actions were hypothesized and assessed using the Joe Model for the watershed (Table 4). Actions with the largest predicted increase in system capacity were given higher priority for implementation and testing.

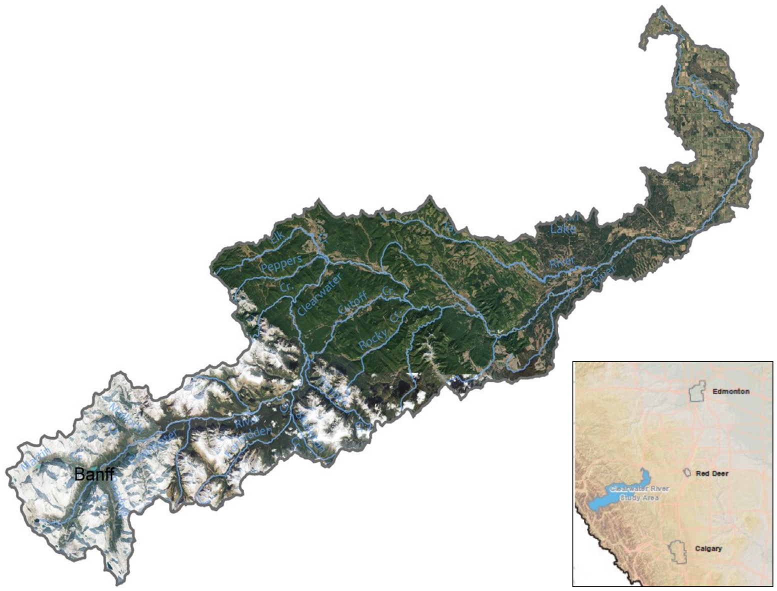

Figure 19. Clearwater River watershed in west central Alberta.

The benefit of Joe Modelling quickly becomes apparent. Statements like “water quality is the problem” or “continued angling threatens the population” no longer make sense. A one-size-fits-all recovery action does not apply across all sub-basins (Table 4). Bull Trout in sub-basins toward the more pristine headwaters (e.g., Clearwater River above Elk Creek) were not predicted to see a benefit from improved water quality. In contrast, sub-basins in more developed areas of the watershed (e.g., Seven Mile Creek) were predicted to see improvements from water quality. These are testable predictions.

The proposed management action from Alberta’s fisheries biologists for the larger (HUC 8) Clearwater River watershed was: a) close the watershed to fishing; b) remediate road crossings; and, c) improve water quality through erosion control and reestablishment of riparian zones (Table 4). The predicted result of these actions was to be a large and detectable increase in the watershed’s adult Bull Trout population. Furthermore, the proposed action for the Clearwater River was one of several HUC 8 watersheds across the province for which a combination of management actions were proposed as part of a replicated before-after control-impact study design. Not surprisingly, the political palatability of such a large-scale experiment was low and fisheries biologists faced either abandoning efforts entirely or retooling their proposal (Schneider 2019). An unforeseen benefit from the Joe Modelling was the rapid and transparent ability for fisheries biologists to propose smaller-scale experiments for actions that would be supported politically, while also capturing lost recovery potential. In the Clearwater, this consisted of road crossing remediation and water quality improvements in sub-basins where these actions were predicted to have a detectable effect but with a trade-off in: lost recovery potential; and, continued uncertainty in the efficacy of angling closures to population recovery.

Table 4. Example of current predicted system capacity for adult Bull Trout in sub-basins (HUC 10 watersheds) of the larger Clearwater River watershed (HUC 8) and the predicted change to the system capacity from implementation of several recovery actions.

HUC8	HUC10	HUC10 Name	Predicted System Capacity	Recovery Action(s): Fishing Regs	Road Crossing Remediation	Improve WQ	Fishing Regs & Road Crossing Remediation	Fishing Regs & Improve WQ	Road Crossing Remediation & Improve WQ	Fishing Regs & Road Crossing Remediation & Improve WQ
				Predicted Change to System Capacity[1]						
11010301	1101030101	Clearwater River – Banff National Park	3.2	1.5	0.0	0.0	1.5	1.5	0.0	1.5
11010301	1101030102	Clearwater River above Elk Creek	3.1	1.5	0.0	0.0	1.5	1.5	0.0	1.5
11010301	1101030103	Forbidden Creek	3.2	1.5	0.0	0.0	1.5	1.5	0.0	1.5
11010301	1101030104	Timber Creek	3.2	1.5	0.0	0.0	1.5	1.5	0.0	1.5
11010301	1101030105	Washout Creek	3.2	1.5	0.0	0.0	1.5	1.5	0.0	1.5
11010301	1101030106	Elk Creek	1.8	0.8	0.0	1.3	0.8	2.8	1.3	2.8
11010301	1101030107	Middle Clearwater River	1.6	0.7	0.1	0.9	0.9	2.1	1.1	2.4
11010301	1101030108	Limestone Creek	1.9	0.9	0.4	0.3	1.5	1.3	0.8	2.1
11010301	1101030109	Seven Mile Creek	0.6	0.3	0.1	1.6	0.5	2.6	2.1	3.3
11010301	1101030110	Tay River	0.7	0.3	0.3	1.2	0.8	2.1	2.0	3.3
11010301	1101030111	Lower Clearwater River	0.2	0.1	0.2	0.9	0.3	1.4	1.9	2.9

[1] *Incremental change in predicted system capacity indicated by green highlighting.*

SUMMARY AND CONCLUSIONS

Alberta's Fisheries Sustainability Assessment is a two-step, standardized process to 1) assess status (i.e., estimate current status of fish populations in relation to desired status), and 2) assess threats (i.e., quantitatively assess potential cumulative threats and effects of mitigation to that status). It builds upon important international principles of fisheries management and conservation, including the United Nations Code of Conduct of Responsible Fisheries and Management Strategy Evaluations. Further, it adapts these important principles to the specific issues of multiple stocks of freshwater fish as threatened by a complexity of cumulative effects. This has become an efficient and effective method to consistently manage many different species and fish stocks in a large and diverse landscape such as Alberta. Importantly, the FSA allows for structured decision making by contrasting current to desired status, prioritizing potential management actions and trade-offs necessary to achieve management objectives and, thus, identify actions for active adaptive management experiments (Walters 1986; Walters 1997; Irwin et al. 2011).

This represents a novel advancement from previous fish management and recovery plans in Alberta, where potential impacts were simply listed and qualitatively discussed for a species as a whole. Little attention was given to quantifying each impact in terms of severity or in terms of effect on the species. Virtually no attention was directed to identifying specific impacts at the population level.

The Joe Model, however, requires fisheries biologists to consistently, and using best-available-science, quantitatively describe their concerns, both at the species-scale (i.e., explicitly define the stressor-response curves), and at the local population-scale (i.e., explicitly quantify the stressor at that scale). This has had a major change to meetings and discussions on species recovery in Alberta. Prior to having these tools available, fisheries decisions were often based on anecdotal examples or small-scale unreplicated studies extrapolated to provincial scales. Qualitative arguments and statements were the norm, e.g., “I’ve seen sediment right on spawning beds in Moose Creek, therefore, we need to increase buffer widths on all forestry operations”. Statements such as these will undoubtedly continue (and can be very important and useful as observations), but their over-riding influence on provincial-level policy and decisions is fading. When such statements are now made, the proponent is usually challenged to draw the stressor-response curve, explain the mechanism, and quantify the stressor.

Addressing the uncertainties from any modelling simulation are key. We consider the model outputs as hypotheses, not predictions. We favour using an adaptive management approach to explicitly test hypotheses (Walters 1986; Sullivan 2003; Arlinghaus et al. 2017). Further, evaluation of stressor-response relationships from the Joe Model can be done by comparing predicted adult system capacity against the observed adult system capacity following adaptive management experiments.

As with any major change, there have been (and will continue to be) difficulties. The adoption of the FSA represents a major shift in Alberta’s fisheries management culture. Fisheries biologists are required to be proficient and conversant in computer modelling and simulation. Data derived from remote sensing is as important as field sampling. These skills are not trivial to learn, and understand. Professional judgement based on opinion can become discounted in favour of empirical data and relevant experience. The change has not been easy for some fisheries biologists, nor is it appreciated by long-term stakeholders versed in value-based, non-empirical arguments.

In spite of the difficulties, the availability of a standardized assessment of status and impacts has resulted in a much more consistent and rigorous approach to fisheries management in Alberta, across all species and populations. Field projects and techniques are consistent over all areas, and regional staff are able to move and assist seamlessly across multiple projects, as manpower and priorities change. Data is consistent and can be shared and compared across a species’ Alberta range. Regulations and management actions are more consistent and aligned at a provincial-scale. Communication of the fishery status and objectives to all stakeholders has been simplified. Justification of regulations and management actions is more consistent and logical.

As identified by Artelle et al. (2018), four fundamental and interrelated characteristics of an effective science-based natural resource management agency are: measureable objectives; quantitative information of populations and impacts including uncertainty; transparency; and independent review. Alberta’s FSA specifically addresses and meets these objectives. The FSA continues to be an evolving, rational, quantitative process that brings consistency to individual fish stocks and cumulative effects assessments province-wide. These changes offer an effective approach to the recovery and sustainability of Alberta’s fish and fisheries.

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