Prioritizing information needs in a well-studied recreational fishery: objectives over data

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Abstract

Research in social-ecological systems (SES) has utilized recreational fisheries, which are tightly coupled human and ecological systems, to study the complex interactions and feedbacks that exemplify SES. Simulation models are useful tools to characterize these interactions and predict patterns. I used an SES simulation model of a stocked rainbow trout (*Onchorynchus mykiss*) fishery in British Columbia, Canada to evaluate objectives, management alternatives, and uncertainty in this fishery. I also calculated the expected value of perfect information (EVPI) on model parameters to identify valuable directions for future data collection. These analyses revealed significant trade-offs between potential fishery objectives, but the EVPI was small, suggesting that quantifying objectives is a more valuable future direction for this fishery than additional data collection. Well-defined objectives would improve managers' ability to evaluate future decisions, but applying this model as a decision tool more generally would also require addressing outstanding uncertainty about its structure.

Keywords: Social-Ecological Systems; Value of Information; Recreational Fishery

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Chapter 1. Introduction

1.1. Introduction

The challenge of sustainably managing natural resources has led to the development of many approaches, theories, and frameworks for viewing and understanding human-nature relationships. Resources and their users are complex, interdependent, and adapt to one another, making it useful to view them as systems. Social-ecological systems (SES) are coupled human and natural systems that can act as a single system, exhibiting behaviours that are not seen in its separate components (Ostrom, 2007, 2009). SES theory applies to many natural resources and has motivated a shift in natural sciences toward interdisciplinarity (Fischer et al., 2015; Hunt et al., 2013; Lam et al., 2014). Recreational fisheries, where fish populations are exploited for sport or subsistence, have often been studied as SES (Arlinghaus et al., 2022; Carpenter & Brock, 2004; Hunt et al., 2011; Johnston et al., 2010, 2013; Solomon et al., 2020; van Poorten et al., 2011). Fishers and fish populations are tightly coupled: fishers directly influence fish survival, and the state of fish populations in turn dictates fishing patterns (Ward et al., 2016). Both fish and fishers are also influenced by governance systems, which represent the third subcomponent of a recreational fishery SES. These components and their interactions give rise to significant complexity, therefore it has been proposed that formalized models of SES are necessary to understand system dynamics and predict responses to change (Ward et al., 2016; Arlinghaus et al., 2017).

Quantitative models have historically contributed to managing fisheries by predicting fish population responses to harvest. It is increasingly recognized, however, that biological models alone do not sufficiently address the complex human and biological factors that managers of recreational fisheries deal with (Hunt et al., 2013). Social-ecological system modelling, where social and ecological subsystems and their interactions are explicitly simulated, has recently emerged to address the challenges of managing SES (Solomon et al., 2020). Applied to recreational fisheries, these models represent interactions between human behaviour and fish ecology that give rise to patterns of resource use (Hunt et al., 2011; Johnston et al., 2010; van Poorten et al., 2011; Wilson et al., 2020). Understanding and recreating the large-scale patterns that arise from interacting system subcomponents makes SES models attractive tools for fishery management (Cilliers et al., 2013). SES modelling has identified several unexpected outcomes, such as conflict among angler groups and potential for collapse (e.g., Johnston et al., 2010) and helped identify courses of action for management (e.g., Hunt et al., 2011); however, applications grounded in data from real fisheries have so far been uncommon (Solomon et al., 2020).

A fundamental challenge in managing recreational fisheries is making decisions with limited information and understanding of the system (Mcallister, 1999; Walters, 1986). Typically, fishery data fall short of characterizing all aspects of the system, requiring managers to make judgments despite a great deal of uncertainty. Decision-making frameworks that account for critical uncertainties have been applied in fisheries research (Fielder et al., 2016; Peterson & Evans, 2003; Robb & Peterman, 1998), but it remains uncommon for them to be applied by recreational fishery management agencies (Lackey 1998; Walters 1986). Decision analysis involves identifying management options, quantifying uncertainty, and developing performance measures or objectives, which allows alternative management paths to be evaluated in light of the uncertainty present (Morgan & Henrion, 1990; Robb & Peterman, 1998). This process can identify trade-offs among management actions and multiple management objectives (van Poorten, 2020; Varkey et al., 2016; Whitlock et al., 2021). Decision analyses also typically examine the robustness of management decisions to uncertainty, i.e., their ability to result in desired outcomes across many plausible states of an uncertain system component. With this information, the benefit of reducing uncertainty in key areas can be calculated (e.g. Prellezo, 2017). Such analyses, referred to as value of information (VOI), are especially relevant to management agencies because gathering information has opportunity costs, meaning that money spent to reduce uncertainty in one area is not available to fund lines of inquiry in other areas (Hansen & Jones, 2008).

Fishery goals and objectives are necessary for evaluating potential management pathways. Quantified and measurable fishery objectives allow for unbiased assessments of past and future management decisions and serve as guidelines for both long-term strategies and day-to-day decisions (Barber and Taylor, 1990). Objectives in recreational fisheries are often poorly defined or completely absent (Hilborn, 2007; Symes & Phillipson, 2009). However, conserving fish stocks and providing satisfying fishing opportunities to many people are common considerations. Various fishery objectives may be in conflict with one another, creating trade-offs where managing for one objective means poor performance in another (Camp et al., 2017; García-Asorey et al., 2011; Johnston et al., 2010). In fisheries where quantified objectives are lacking, understanding these trade-offs through modelling can be a useful step toward defining them.

British Columbia's rainbow trout (*Onchorynchus mykiss*) fishery is made up of lakes stocked with hatchery-raised fish that are fished by the province's 250,000 licensed anglers (rod-and-reel fishers). It is an exceptionally well-studied recreational fishery, having been a study system for foundational and current research (Askey et al., 2013; Cox & Walters, 2002; Mee et al., 2016). A social-ecological model of the fishery was recently developed by Carruthers et al. (2019) that accepts management changes as inputs and predicts angling effort across the landscape. Many of its model processes and parameters are empirically grounded through a series of experiments, and it is fit to fishing effort data from the fishery. However, it remains unclear how its predictive capability is influenced by uncertainty in the model's parameters. Parameter uncertainty can be reduced by funding appropriate data collection or research initiatives, but divergent research paths are likely in competition for limited funds. It is therefore beneficial to understand the value of improving precision in key parameters, which can be compared with costs to prioritize future data and research directions. The goals of this project are therefore to (1) evaluate the sensitivity of system performance under different sets of objectives, (2) calculate the value of information for key parameters to identify priorities for future monitoring and research, and (3) identify trade-offs among sets of candidate management policies and fishery objectives.

1.2. Background & study system

British Columbia (BC) is a large Canadian province with approximately 20,000 angling lakes, 675 of which are stocked annually with rainbow trout (Varkey et al., 2018). The fishery is under the jurisdiction of the BC Ministry of Forests, however, biologists and managers typically oversee the lakes in a single region. BC's nine management regions are diverse in their climate, productivity of fish stocks, and angler demographics. The Thompson-Nicola is a southern region with a large angling population, many accessible lakes, and relatively high rates of stocking. Its fisheries have been the grounds for testing leading-edge management science including adaptive management and decision analysis (Mee et al, 2016; Cox and Walters, 2002; Varkey et al., 2016), and there is generally an abundance of research conducted in the region.

Over four million rainbow trout are raised each year in hatcheries across BC to be stocked into small (<1000 ha) lakes (Province of British Columbia Ministry of Environment [BC MOE], 2007).Trout are stocked as fry (aged six months) or yearlings (aged one year), then grow in lakes until they become vulnerable to the fishery at approximately 2 years of age. Fish are also occasionally stocked as ‘catchables’, at a size where they are immediately vulnerable to the fishery. Managers aim to meet the expectations of anglers and maintain a desirable level of fishing effort primarily by manipulating the density and size of stocked fish. Data are collected to inform management decisions, which includes gill-net data for abundance and size structure, angler creel survey data, and effort estimates from aerial surveys.

The goals for managing BC’s freshwater fisheries include: 1) “establish governance approaches that are strategic, effective, and efficient”, 2) “conserve wild fish and their habitats”, and 3) “optimize recreational opportunities based on the fishery resource” (BC MOE, 2007, p.17). Most rainbow trout-stocked lakes are otherwise devoid of fish, and stocked with sterile hatchery-raised trout, therefore fish conservation interventions are rarely needed. The first and last goals, however, are highly relevant to the rainbow trout fishery and are explored in this work.

Chapter 2. Literature Review

2.1. Managing recreational fisheries as social-ecological systems

2.1.1. Managing recreational fisheries

In recreational fisheries, fish are caught and consumed for leisure or as a means of food provisioning. In many parts of the world, coastal and freshwater fish caught recreationally are a significant source of food and provide food security that can buffer against the impacts of poverty (Cooke et al., 2018). In industrialized nations, fish stocks are exploited primarily for leisure. Millions of people globally participate in recreational fisheries, meaning their participation rate far exceeds that of commercial fisheries, but with less biomass caught (Arlinghaus et al., 2015). As fishers spend money on guided experiences, gear, and lodging, successful recreational fisheries provide economic benefits to surrounding communities. Because of this, recreational fishing rivals the economic impact and social relevance of commercial fishing despite its much smaller scale (Brown, 2016; Cooke et al., 2018).

It was previously believed that recreational fisheries did not require substantial management interventions. As fish abundance declined, it was assumed that fishers would seek opportunities elsewhere, thus reducing pressure on the declining population and creating a dynamic much like the fluctuations of a natural predator-prey system (Johnson & Carpenter, 1994). However, anecdotes and evidence of substantial abundance and size declines in some recreationally exploited stocks are now widely accepted (Hunt et al., 2011; Post, 2013). Governing bodies (typically government agencies) now actively manage most recreational fisheries by restricting gear and harvest, or increasing productivity through stocking or habitat enhancement. While recreational fishing includes many forms of harvest with a variety of gear, it has come to be associated most with angling (rod-and-reel fishing).

Fishery managers commonly use management interventions that involve either: 1) restricting the number of resource users; 2) restricting allowed gear to limit catch efficiency or mortality; 3) limiting the size and/or number of fish that can be retained; or a combination of these (Cox et al., 2003; Post et al., 2002; Van Poorten et al., 2013;

Walters, 1986). In North America, fisheries are typically managed as open-access resources, making it difficult to limit the total number of people fishing recreationally, called fishing effort (Daedlow et al., 2011). Instead, recreational fisheries are governed by a combination of size limits, harvest limits, and gear restrictions that aim to prevent excessive fishery-induced mortality without limiting public access to the resource, or imposing regulations that anglers would perceive as too restrictive (Aas et al., 2000; Dotson et al., 2013; Parkinson et al., 2018). As such, the trade-off between conservation and fishing opportunity is considered a central challenge of managing recreational fisheries (Cowx et al., 2010; van Poorten & Camp, 2019). In some cases, this fundamental trade-off can be avoided with stock enhancement, where popular sport fish are added to waterbodies to provide more angling opportunities and direct fishing effort away from waterbodies with naturally-occurring sport fish. This practice is somewhat controversial from an ecological perspective as it can negatively impact native biodiversity, including other sport fish (Arlinghaus et al., 2017, 2022; Hirner & Cox, 2007; van Poorten et al., 2011).

Another significant management challenge exists in governing what is often a large set of diverse ecosystems and resource users through a single, centralized management agency. For example, a size limit that is effective for a fish population in one lake may not be effective in a lake in a different area, where the climate does not allow fish to grow as large (Varkey et al., 2018). Furthermore, fishing effort may be orders of magnitude higher near urban centres than in rural areas, creating large-scale patterns of near-urban fishery depletion (Matsumura et al., 2019; Post et al., 2008; Wilson et al., 2020). Resource users are also diverse in their motivations, and not all anglers are satisfied with the same fishing experience (Beardmore et al., 2015; Fedler & Ditton, 1994; Ward et al., 2013). Creating attractive fisheries for a large, diverse user base likely warrants a diversity of policy responses, but due to the large number of fisheries that a single management agency may oversee, it is more efficient to apply 'one-size-fits-all' policies. However, these often do not achieve the fishery's goals (Carpenter & Brock, 2004; van Poorten & Camp, 2019).

2.1.2. Social-Ecological Systems

A recreational fishery can be thought of as a combination of at least two systems: a human (social) system and a natural (ecological) system. The human system reflects the characteristics and behaviour of anglers, but may also include the actions of fishery managers. The natural system includes the growth and reproduction of fish as well as their interactions with their environment. Fishing influences the ecosystem by altering the abundance and size structure of fish populations, and likewise, these same factors influence where and when people fish (Ward et al., 2016a). Social-ecological feedbacks and interdependencies are meaningful because they can create emergent properties, which cannot be seen in the SES's separate components, but may have large-scale impacts (Arlinghaus et al., 2017; Cooke et al., 2015). It has become increasingly common to adopt a social-ecological system perspective in recreational fisheries research (Solomon et al., 2020; Ward et al., 2016). Due to the tight coupling of the fishery resource and its consumers, recreational fisheries are excellent model systems in which to observe and study social-ecological dynamics (Hunt et al., 2011; Johnston et al., 2010). Because of the significant complexity that can arise from social-ecological dynamics in a recreational fishery, SES researchers have advocated for simulation modelling to predict system responses to change (Arlinghaus et al., 2017; Ward et al., 2016). SES models of recreational fisheries combine ecological and behavioural models to recreate system dynamics, and their use for management-focused research is gaining popularity (Solomon et al., 2020).

2.2. Human dimensions of recreational fisheries dynamics

2.2.1. Angler behaviour

Research in the 'human dimensions' of recreational fisheries addresses questions about human cognition and actions in both the exploitation and management of fisheries. Questions about the characteristics and behaviour of anglers are particularly relevant and have been explored for over 50 years (Hunt et al., 2013). Anglers often choose whether and where to fish in a landscape of fishing opportunities, and among other opportunities for recreation (Carruthers et al., 2019; Hunt et al., 2011). Motivations for choosing to fish at any given site vary from catching a trophy-sized fish to enjoying a state of relaxation in a natural setting – outcomes which are not easily comparable except in terms of their contribution to an individual's well-being. To compare diverse attributes on a common scale, it is useful to use the concept of utility. Broadly, utility is a unitless measure of value, or a measure of preference for aspects of an experience (Fenichel et al., 2013*b*). For example, the location, aesthetics, and catch rate of a given lake may each contribute a part-worth utility to the lake's overall utility. Utility theory assumes that anglers operate as if maximizing personally held objectives (Fenichel et al., 2013*a*), which dictates their choice of fishing locations and drives landscape-scale patterns. Models that predict a distribution of angling effort on a landscape often use utility theory, assuming that anglers fish where their utility is maximized. The process of distributing effort where utility is highest may be iterated until no further gains in total utility can be achieved, a state known as the ideal free distribution (IFD) of effort (Fenichel et al., 2013*a*). Where utility is used to predict angler distribution, it most directly corresponds to anglers' *expected* outcomes. Realized outcomes may differ from expectations (Arlinghaus, 2006), but both have been approximated by utility (Fenichel et al., 2013*a*).

Motivations for fishing recreationally are diverse, and defining them has been a focal point of research in this field (Arlinghaus, 2006; Beardmore et al., 2015; Hunt et al., 2019). Anglers may fish in pursuit of certain harvest expectations, a temporary change of routine, or relaxation (Beardmore 2011, Beardmore 2015, Arlinghaus 2016; Hunt et al 2013; Arlinghaus, 2006). Desirable attributes of a fishing experience may be catch-dependent (e.g., number of fish caught, number harvested) or catch-independent (e.g., degree of crowding, accessibility). While research has concluded that most anglers are not highly motivated by catch rates (Arlinghaus, 2006; Beardmore, 2011; Birdsong et al., 2021), they continue to be vocal in their opposition to restrictive catch and harvest regulations (Matlock et al 1988; Matlock 1991). An angler's perception of a fishing experience after it has occurred, or their satisfaction, may be more dependent on catch-related aspects of the trip than the factors that initially motivated them to fish (Arlinghaus, 2006; Birdsong et al., 2021). Angler satisfaction is of particular interest to fishery managers, as it may impact future license sales (Dabrowska et al., 2014).

2.2.2. Angler Heterogeneity

Anglers' preferences for different attributes of a fishing experience vary greatly among individuals. Anglers can be grouped by important motivations, for example, trophy-oriented anglers are primarily concerned with catching large fish and releasing most, while harvest-oriented anglers seek higher catch rates (Johnston et al., 2010; Matsamura et al., 2019; van Poorten et al., 2019; Varkey et al., 2016). Social-ecological research in recreational fisheries has begun recognizing angler heterogeneity as a key social process (Solomon et al., 2020), and anglers are sometimes grouped by their preferences using latent class models (Fenichel et al., 2013*a*; Hunt et al., 2019; Matsumura et al., 2019). Angler heterogeneity can add complexity to management by creating trade-offs when a management action would lead to gains in utility or satisfaction for one angler type but losses for another (Aas et al., 2000; Johnston et al., 2010; Ihde et al., 2011). This is particularly important when objectives include equitable or diverse participation in the fishery. While heterogeneity has been well-studied in the human dimensions literature, it has rarely been adopted as a consideration for fisheries biologists and managers, despite managers' vested interest in angler satisfaction (Fulton et al., 2011; Hunt et al., 2013, Ward et al., 2016).

2.3. Management objectives

Desired outcomes for a fishery are represented by objectives, which may be biological, social, or economic in nature. Biological objectives have historically pertained to desired yields, but are increasingly replaced with objectives that better represent the role of fisheries in society (Barber & Taylor, 1990; Fenichel et al., 2013*b*). Social objectives may consider individual rights, community needs, and equity (Symes & Phillipson, 2009), but often the well-being of resource users is simply represented by an angler satisfaction objective (Carpenter & Brock, 2004; Johnston et al., 2010). Economic objectives consider that angling creates significant revenue and jobs; seeking to increase participation in the fishery is a common objective that reflects economic priorities (Cooke et al., 2015). Choosing between these is a normative judgement about what recreational fisheries provide to society (Fenichel et al., 2013*b*). Where multiple objectives are necessary, they may conflict with one another, for example, increasing angler participation and conserving a fish population. Understanding trade-offs between multiple potential objectives has been called a central challenge to recreational fisheries management (García-Asorey et al., 2011).

Fishery objectives can be categorized as either 1) means objectives or 2) fundamental objectives (Clemen, 1996; Peterson & Evans, 2003). Fundamental objectives reflect underlying values, while means objectives are those whose realization is expected to contribute to achieving a fundamental objective in turn. For example, a manager of a trophy fishery may have a size target (means objective) that they believe to be conducive to angler satisfaction (fundamental objective). It is uncommon for management agencies to have well-defined fundamental objectives (Barber and Taylor, 1990, Hilborn, 2007; Symes & Phillipson, 2009), often using vague goals of 'wise' or 'sustainable' management instead. While these may be acceptable to the public, they are not conducive to effective management. Without quantifiable objectives, progress cannot be measured, alternative states of the fishery cannot be valued, and goals can shift over time (Barber and Taylor, 1990). Reluctance to define fundamental objectives may be due to the challenge of identifying common values in society and the management institution, or that most fishery managers are biologists with limited social science literacy (Hilborn, 2007; Hunt et al., 2013). However, normative criteria are used to judge any potential decisions, and individual decision-makers will apply their own criteria in the absence of well-defined objectives for the fishery (Fenichel et al., 2013*b*). Researchers have called for management institutions to define quantifiable objectives for decades (Barber & Taylor, 1990; Lackey, 1998), but the problem is still pervasive.

2.4. Frameworks for dealing with uncertainty

2.4.1. Uncertainty

Uncertainty is always present in science, especially when attempting to understand and model complex natural resource systems. There are three sources of uncertainty in modelled natural systems: 1) background variation, the inherent randomness occurring in nature; 2) parametric uncertainty, about the values and functional relationships of parameters in a model; and 3) structural uncertainty, about which parameters to include in the model or which mechanisms to explicitly or implicitly include (Walters, 1986). The first two can alternatively be described as process error and observation error, with error being a mathematical representation of uncertainty. Process error is a product of nature and the way it is modelled, while observation error results from inaccuracies in collecting data and subsequently estimating parameters (Hilborn & Mangel, 1997). In other words, uncertainty arises because humans cannot perfectly replicate nature with a model, nor can they observe nature perfectly. While observation error can usually be reduced by collecting more or higher quality data, other sources of uncertainty cannot be resolved with additional data (Walters, 1986).

Traditional science generally seeks to reduce uncertainty (Halpern et al., 2006). Treatment of uncertainty in research is often limited to reporting a single metric of error around a value, despite the availability of analytic methods for addressing and incorporating uncertainty, including decision analysis (Hansen & Jones, 2008; Morgan & Henrion, 1990). If not dealt with explicitly, uncertainty may be ignored in decision-making. Complex social-ecological systems have many sources of uncertainty and may exist on large landscapes where extensive data collection would be exceptionally costly. Increasing data collection as way to address uncertainty falsely assumes that a gain in information about the system always leads to a gain in decision-making capability. Reducing observational uncertainty is often not sufficient to improve management decisions (Falcy, 2021). Instead, decision analysis can identify management decisions that are robust to uncertainty and aligned with objectives.

2.4.2. Decision analysis and value of information

Decisions in natural resource management of publicly held resources involve significant ecological risk, and make use of public monetary resources. Even so, they often rely on the judgement of one or a few experts (Morgan & Henrion, 1990; Powers, 1975). With extensive knowledge of the system, experienced managers may be able to predict outcomes resulting from different courses of action and make decisions accordingly. However, research has shown that even when equipped with extensive data, expert judgement tends to be biased due to differences in individuals' implicit understanding of the system and its objectives (Morgan & Henrion, 1990). Conversely, decision analysis processes require decision makers to make their assumptions about the system explicit. In general, decision analysis proceeds as follows: 1) identify quantified, measurable objectives, 2) identify alternative management actions, 3) identify key uncertainties in the system, 4) estimate the probability distributions associated with key uncertainties, 5) predict outcomes and their expected value in terms of objectives, and 6) identify the best management actions or identify trade-offs (Mcallister, 1999). The structure provided by decision analysis helps managers and stakeholders identify decision components and create a transparent decision-making process (Irwin et al., 2011). It may also reveal trade-offs among potential management actions and identify optimal courses of action (Halpern et al., 2006; van Poorten, 2020; Varkey et al., 2016; Woodruff et al., 2021).

Extensions of decision analysis address questions about the value of reducing uncertainty, called value of information (VOI) approaches. VOI concepts come from information economics, a branch of microeconomic theory (Quirk, 1976), however, information can be valued in non-monetary terms, including in terms of utility (Link & Peterman, 2011; van Poorten, 2020). When applied in natural resource management, VOI approaches challenge the idea that collecting large amounts of data necessarily confers good management (Halpern et al., 2006; Hansen & Jones, 2008). Two phenomena about information gathering and uncertainty provide rationale for VOI analyses. First, funds used to address a key uncertainty are not available for other uncertainties or management actions, therefore gathering information has opportunity costs. Second, in gaining additional information, a limit to the certainty that data can provide is reached, therefore information has diminishing returns (Hansen & Jones, 2008). Valuing information allows for comparing benefits and costs of information, and prioritizing information that has the greatest impact on management decisions.

The value of information can be approximated by several metrics. The value of accounting for uncertainty in a decision analysis can be calculated, and is referred to as the expected value of including uncertainty (EVIU). EVIU has been used to demonstrate the value of decision analytic approaches, especially in their early applications. The value of reducing uncertainty through data collection is known as the expected value of perfect information or expected value of sample information (EVPI/EVSI) (Walters, 1986). The EVPI assumes that all uncertainty can be eliminated with perfect information. As such, the EVSI is more realistic, but more complicated to calculate, and therefore EVPI is more common (Prellezo, 2017; but see Williams & Brown, 2020). The EVPI represents a theoretical upper bound to the EVSI, and it can be interpreted as the maximum that a decision maker should spend to resolve a given uncertainty, beyond which the cost of information gathering exceeds the maximum value of the benefits it can provide (Walters, 1986).

Decision analysis has been applied in many natural resource management contexts, including watershed management (Ohlson & Serveiss, 2007), marine reserves (Halpern et al., 2006), and fisheries. In recreational fisheries, decision analysis has been used to evaluate potential fishery regulations in systems with multiple objectives that may be in conflict with one another (Peterson & Evans, 2003; van Poorten and MacKenzie, 2020; Varkey et al., 2016). Value of information applications have included examining the value of additional biomass or abundance surveys (Link & Peterman, 2011; Prellezo, 2017). Mäntyniemi et al. (2009) calculate the value of information about the functional form of a stock-recruitment relationship, and in doing so they address a key uncertainty in many fisheries, but also provide an example of VOI that addresses structural uncertainty. Despite these notable applications, VOI analyses remain relatively uncommon in fisheries.

Chapter 3. Methods

3.1. SSES model

3.1.1. Model overview

The Spatial Social-Ecological Systems (SSES; Carruthers et al. 2019) model predicts landscape-scale fishing effort using parameters and processes defined through previous research in this system. It is spatially explicit, comprising of 584 rainbow trout-stocked lakes and nine modelled angler population centres (Figure 1). Four distinct types of anglers are modelled, differing in the utility they attribute to multiple aspects of fishing, and in their catchability. Management changes can be simulated by altering stocking rates or fishing regulations (bag limits, gear restrictions, boat and engine restrictions). It is also possible to simulate uncertainty in parameters relating to fishing, angler utility, and population dynamics. The model predicts an equilibrium effort prediction for each lake, average catch rates and fish sizes, and angler utility for each lake. The SSES model is made up of three interdependent submodels: 1) an angler behaviour model, 2) a biological model of growth and mortality, and 3) a method for predicting angling effort at equilibrium using ideal free distribution (IFD) theory. The model is described in detail by Carruthers et al. (2019).

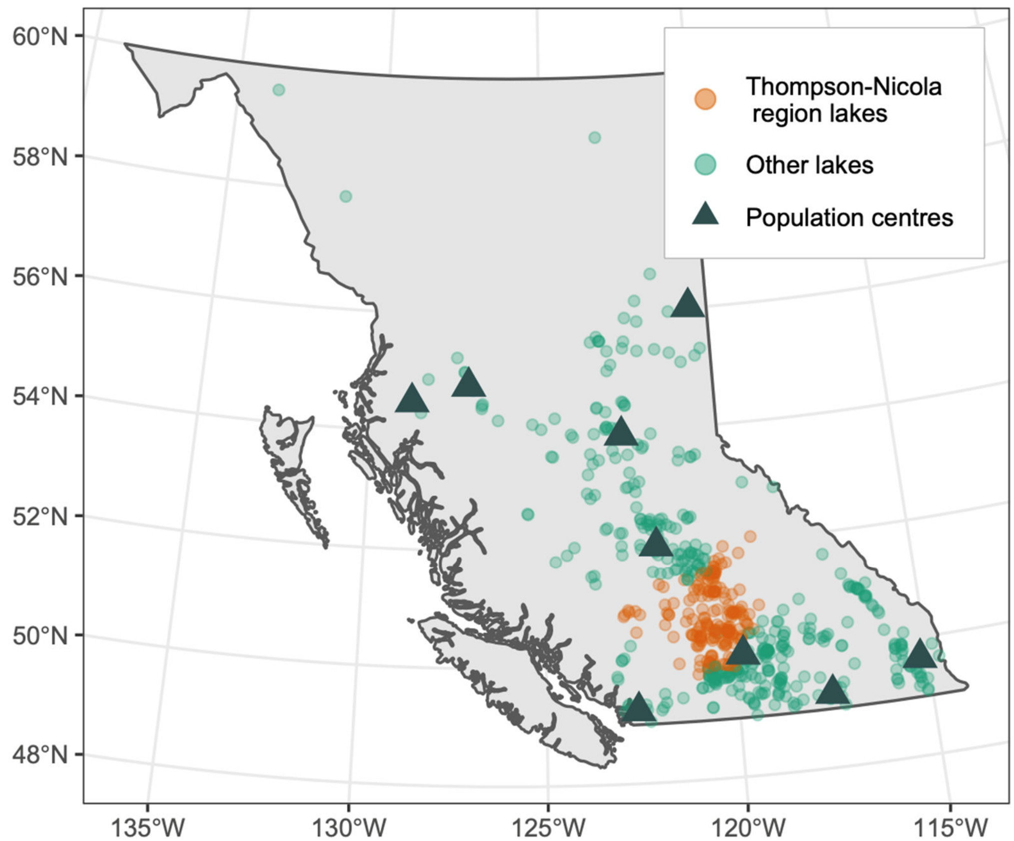

Figure 1. **Map of British Columbia (BC) showing locations of lakes and population centres in the SSES model. There are 584 modelled lakes, 184 of which (Thompson-Nicola region lakes; in orange) were subject to simulated stocking changes in the proceeding analyses. Modelled population centres are aggregations of several smaller population centres, and do not necessarily spatially coincide with real population centres in BC.**

3.1.2. Angler behaviour submodel

The angler behaviour model includes multiple angler types to account for heterogeneous preferences and behaviours within the angling population for this fishery (Johnston et al., 2010; Matsumura et al., 2019; Mee et al., 2016). The angler classes identified and the latent class model used are described in detail in Dabrowska et al. (2011). Log-transformed angler utility for a given lake is calculated as a sum of part-worth utilities for aspects of the catch and angling experience as well as a set of lake-specific attributes:

$$(1)\quad \log(U) = U_{size} + U_{catch} + U_{dist} + U_{crowd} + U_{attr}$$

where U_{size} and U_{catch} are the expected size and number of rainbow trout caught, U_{dist} is the travel distance, U_{crowd} is the angler crowing, and U_{attr} is a set of lake attributes including the bag limit, boat launch facilities, and engine restrictions. This lake-specific utility is then used to estimate angler effort distribution across the fishing landscape. Effort originating from a given angler class and population centre is given by:

$$(2)\quad E_L = m \cdot l \cdot \frac{U_L}{c + \sum_L U_L}$$

where *m* is the maximum angling days per year, *l* is the number of licenses sold, U_L is the utility of a given lake, and *c* is the utility of not angling.

3.1.3. Biological submodel

Fish growth is modelled as biphasic, whereby growth is linear until maturity and then becomes asymptotic, following a study by Ward *et al* (2017). Length before maturity and after maturity, respectively, are as follows:

$$(3)\quad L = \left\{ \begin{array}{cc} h \cdot Q + L_s & Q \leq Q_{mat} \\ L^{\infty} \cdot (1 - e^{-K(Q-t)}) & Q > Q_{mat} \end{array} \right\}$$

Length before maturity is determined by the immature growth rate *h*, thermal age *Q*, and length at stocking L_s. Mature growth ($Q > Q_{mat}$) follows a Von Bertalanffy growth model (Lester et al., 2004). Abundance is a function of the number of stocked fish, their survival, and the number removed; the model does not include naturally occurring rainbow trout recruitment. Fishing mortality is calculated from the fraction of fish removed by anglers *R*, proportion captured that are within the bag limit *b*, voluntary release rate *V*, and post-release mortality rate *P*:

$$(4)\quad F_L = (1 - V)(1 - b_L)\, R + [1 - (1 - V)(1 - b_L)\,]R \cdot P$$

3.1.4. Angling effort prediction

The IFD of angler effort distribution is approximated by initializing with fishing effort estimates for each lake, then iteratively calculating expected catch rates, size, and crowding, their resulting utility (Eq. 1) then effort (Eq. 2). The model is considered to have converged on an IFD prediction when total landscape effort fluctuates by no more than 0.1 angler-days between iterations. The model is then fit to observed effort data to estimate voluntary release rate, accessibility, and participation rate parameters. This is done with effort data from a small set of data-rich lakes first, then with all 584 lakes (Carruthers et al., 2019).

3.2. Sensitivity analysis and parameter selection

Stocking-effort and stocking-utility relationships were established to ensure that angler effort and utility responded predictably to simple management changes. Total stocking in Region 3 was varied between no stocking and a 150% increase from the current (baseline) stocking rate, then resulting effort and utility were calculated. The stocking-effort relationship then served as a basis from which to assess parameter sensitivity. Four of the parameters that exposed the most model sensitivity were selected for further analysis: 1) The maximum growth rate, Δ, which governs growth during the immature stage of biphasic growth (Eq. 3); 2) the voluntary release rate V, or proportion of fish released before the harvest limit requires it (Eq. 4); 3) the intercept of the linear function for angler utility for travel distance to a lake, U_{dist}, and; 4) the linear intercept of utility for fish size, U_{size} (Figure 2). The predicted effort response was highly sensitive to these parameters, and together they represent at least one key parameter from each submodel, as well as one that links angler and population dynamics.

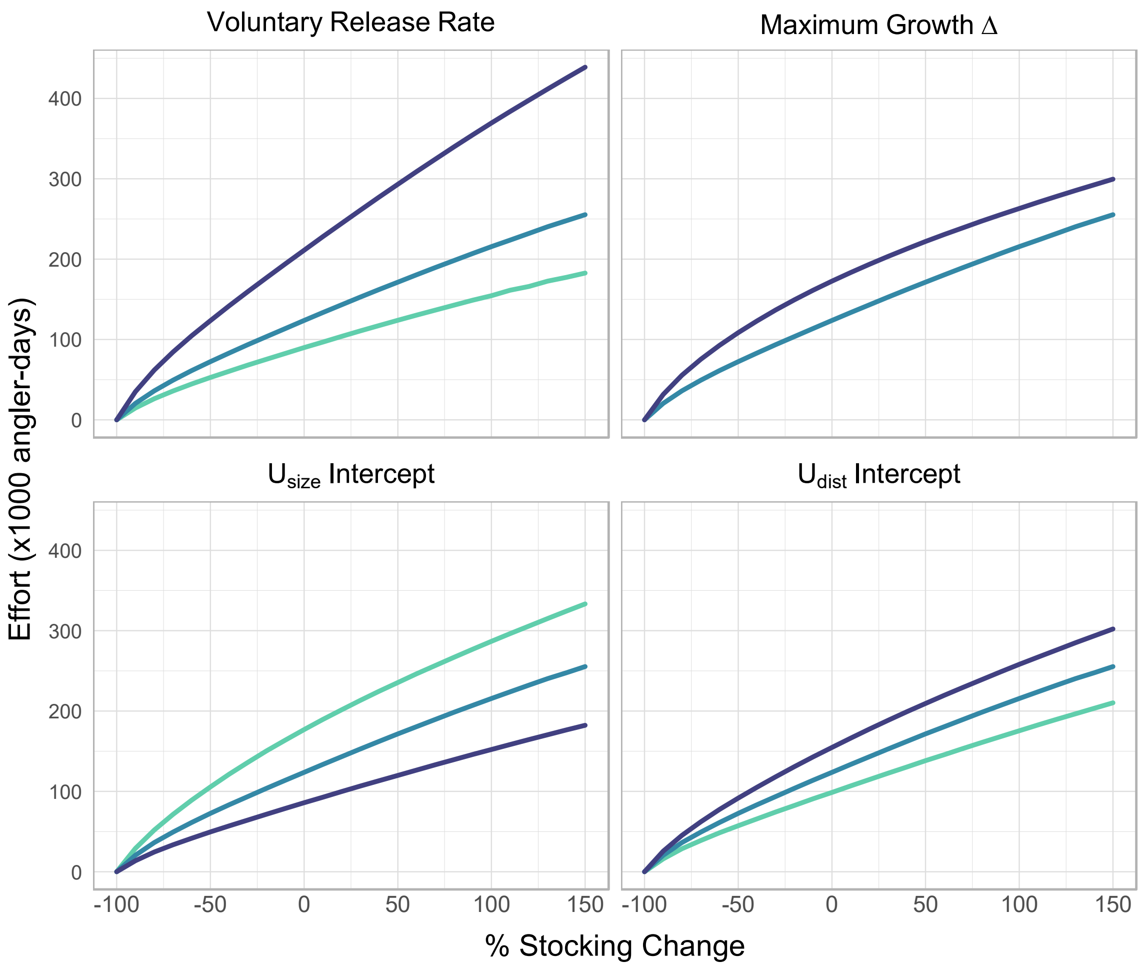

Figure 2. Sensitivity plots for each of four model parameters, showing the resulting regional effort across a range of stocking rates. The regional stocking rate is varied along the x-axis from -100% (no fish stocked) to a 150% increase (more than doubled) from baseline stocking. Three lines represent parameter value manipulations by an arbitrary multiplier of 0.5 (lightest line), 1 (middle line), and 1.5 (darkest line). Note that the maximum growth rate parameter lacks a light-coloured line due to a restricted range of values that can be accommodated by the growth calculations.

3.3. Development of alternative management regimes

To evaluate the key parameters identified above in a decision analysis framework, it was necessary to develop simulated management alternatives to serve as 'decisions'. These alternatives reflected strategic decisions a manager could make that would impact all lakes within a single region. The Thompson-Nicola region was used as a study system due to its many lakes, large angling population, and abundance of previous research on the region's rainbow trout fishery (Cox & Walters, 2002; Mee et al., 2016; Varkey et al., 2016; Ward et al., 2013). The distribution of stocked fish was the only difference among the management alternatives, as stocking is a major management lever in this fishery that frequently changes from year to year. The total quantity of stocked fish was held constant to view the influence of strategic changes separately from the broad-scale numerical effort response to greater fish availability (Figure 2).

Two criteria determined how stocking would be redistributed in a given management alternative. Due to the well-understood importance of travel distance in angler dynamics (Post & Parkinson, 2012; Wilson et al., 2020), stocked fish were redistributed between lakes which are near to population centres (<150 km) and far from population centres (≥ 150 km). Alternatively, stocked fish were redistributed between lakes which already supported high angling effort and those that did not. In this case, donating and receiving lakes were split equally by the median annual effort. All management alternatives are described in Table 1.

Table 1. Descriptions of simulated management alternatives. Fish were redistributed from near-urban lakes (n=41) to distant lakes (n=143), or from high-effort lakes (n=92) to low-effort lakes (n=92). Note that management alternative 9 is the status quo, where no changes were made to the original stocking pattern.

Management Alternative ID	Donating Lakes	Receiving Lakes	Amount redistributed (yearling-equivalents)
1	Close lakes	Far lakes	19,657 (10% from donating lakes)
2	Far lakes	Close lakes	69,830 (10% from donating lakes)
3	Close lakes	Far lakes	39,315 (20% from donating lakes)
4	Far lakes	Close lakes	139,660 (20% from donating lakes)
5	Low effort lakes	High effort lakes	44,088 (10% from donating lakes)
6	High effort lakes	Low effort lakes	45,399 (10% from donating lakes)
7	Low effort lakes	High effort lakes	88,176 (20% from donating lakes)
8	High effort lakes	Low effort lakes	90,798 (20% from donating lakes)
9	None	None	0

3.4. Development of objectives

To evaluate and compare alternative management regimes, objectives for the management of this system needed to be established. The goals of BC's rainbow trout fishery are to "conserve wild fish and their habitat", "establish governance approaches that are strategic, effective, and efficient", and "optimize recreational opportunities based on the resource" (BC MOE, 2007, p.17). The first is not applicable as no wild stocks are included in the SSES model. The second and third are not adequately quantified for this analysis. I assumed that the stocked rainbow trout fishery has two fundamental goals: 1) maximize angling effort, and 2) maximize angler utility, consistent with the government fisheries program plan (Freshwater Fisheries Society of BC [FFSBC], 2017). As Johnston et al (2010) demonstrated, optimal fishery outcomes may depend on underlying assumptions about the relative importance of the angler classes' utilities. Since the BC freshwater fishery aims to maximize utility, but has no stated method for integrating across heterogeneous angler classes, I evaluated all three utility aggregation methods described by Johnston et al (2010). As such, modelled landscape outcomes were evaluated with respect to maximizing four objectives: 1) total effort; 2) total utility, with angler classes weighted equally; 3) egalitarian utility, weighted by the proportion of each angler type in the population; and 4) Rawlsian utility, which considers only the lowest-utility angler class.

3.5. Quantifying parameter uncertainty

The SSES model is deterministic and therefore does not include parameter uncertainty. To include parameter uncertainty in simulations, estimates of uncertainty were obtained from the literature. For the maximum growth rate parameter Δ, the coefficient of variation of the lognormal posterior probability distribution from Ward et al (2017) was used such that it centered on the model Δ value. The intercept parameters for U_{dist} and U_{size} were assumed to be normally distributed with a standard deviation equal to that of the average part-worth utility across all angler classes (Dabrowska et al., 2011). Finally, the voluntary release rate V was compared directly to data from experiments done on stocked rainbow trout lakes in the Thompson-Nicola management region system (Cox, 2002). The uncertainty distributions of these four parameters are shown in Table 2. In all analyses that included parameter uncertainty, discrete values spanning two standard deviations from the mean were simulated.

Table 2. **Uncertainty distributions around each of four model parameters chosen for analysis. Distributions were approximated from reported uncertainty in the literature that contributed to original parameter estimates in the SSES model. Discrete values within two standard deviations of the mean were used to simulate uncertainty in these SSES model parameters.**

Parameter	Distribution	Source
Voluntary release rate *V*	Observed frequencies	Cox, 2002
Maximum growth Δ	$\mathrm{Ln}(\Delta) \sim N(207.70, 8.74)$	Ward et al., 2017
U_{dist} Intercept	$U_{dist} \sim N(0.46, 0.078)$	Dabrowska et al., 2011
U_{size} Intercept	$U_{size} \sim N(-0.75, 0.49)$	Dabrowska et al., 2011

3.6. Value of information

The SSES model was run to convergence on IFD angling effort for each combination of management alternatives and simulated parameter values. The values calculated in each scenario are the predicted regional fishing effort, total utility, egalitarian utility, and Rawlsian utility, as defined in section 3.4. The expected value of perfect information (*EVPI*) for a given parameter is the difference between the expected value with perfect information and the expected value under uncertainty (i.e., without perfect information):

$$(5)\quad EVPI = EVwPI - EVU$$

where *EVwPI* is the expected value with perfect information, which assumes the true parameter value is known and the management alternative with the best outcome is chosen, and *EVU* is the expected value under uncertainty, which assumes a single management path must be chosen that maximizes the probability-weighted mean of possible outcomes. These can be written as:

$$(6)\quad EVwPI = \sum_x \max\left(M[d]_x\right) \cdot p_x$$

$$(7)\quad EVU = \max\left(\sum_{x} M[d]_x \cdot p_x\right)$$

Where M is the model-predicted outcome in terms of an objective, d is a decision from a discrete set of decisions D, x is a discrete parameter value from set X, and p_x is the probability of that value.

Chapter 4. Results

4.1. Sensitivity analyses

Modelled outcomes corresponding to four objectives (total effort, total utility, egalitarian utility, and Rawlsian utility) were measured across a range of stocking rates to establish a baseline from which to understand model-predicted responses to stocking changes. Stocking-effort and stocking-utility relationships are shown in Figure 3. Effort, total utility, egalitarian and Rawlsian utility all increase continually with stocking. Total effort has a nearly linear relationship with the rate of stocking (Figure 3a), while total utility is approximately logarithmic (Figure 3c) despite predicted effort being calculated from utility (Eq. 2). Large differences in the scale of utility among angler classes are not seen in predicted effort, in fact the class with the lowest utility ('occasional' anglers) sustains very high effort at high stocking rates. This drives the differences among total utility and effort patterns. Egalitarian utility differs from total utility in its scale, because it is a weighted mean, but does not differ in its functional form (Figure 3e). Finally, Rawlsian utility is the utility of the occasional angler class, which is unique in that it increases nearly linearly with stocking.

Sensitivity analyses were conducted and four parameters to which the model was highly sensitive were selected to be the focus of the value of information analyses (Figure 2). For highly sensitive parameters, a 50% increase in the parameter value resulted in differences of over 50,000 angler-days of regional effort. The model-predicted effort was not highly sensitive to most biological parameters, except the maximum growth rate parameter, Δ, and natural mortality. Most fishery-related parameters such as angler selectivity and catchability had low to medium sensitivity. However, the model was most sensitive to the voluntary release rate (Figure 2). Among part-worth utility parameters for fishery attributes, the model was sensitive to distance utility and size utility intercepts, but less so to angler crowding and catch rate utility functions.

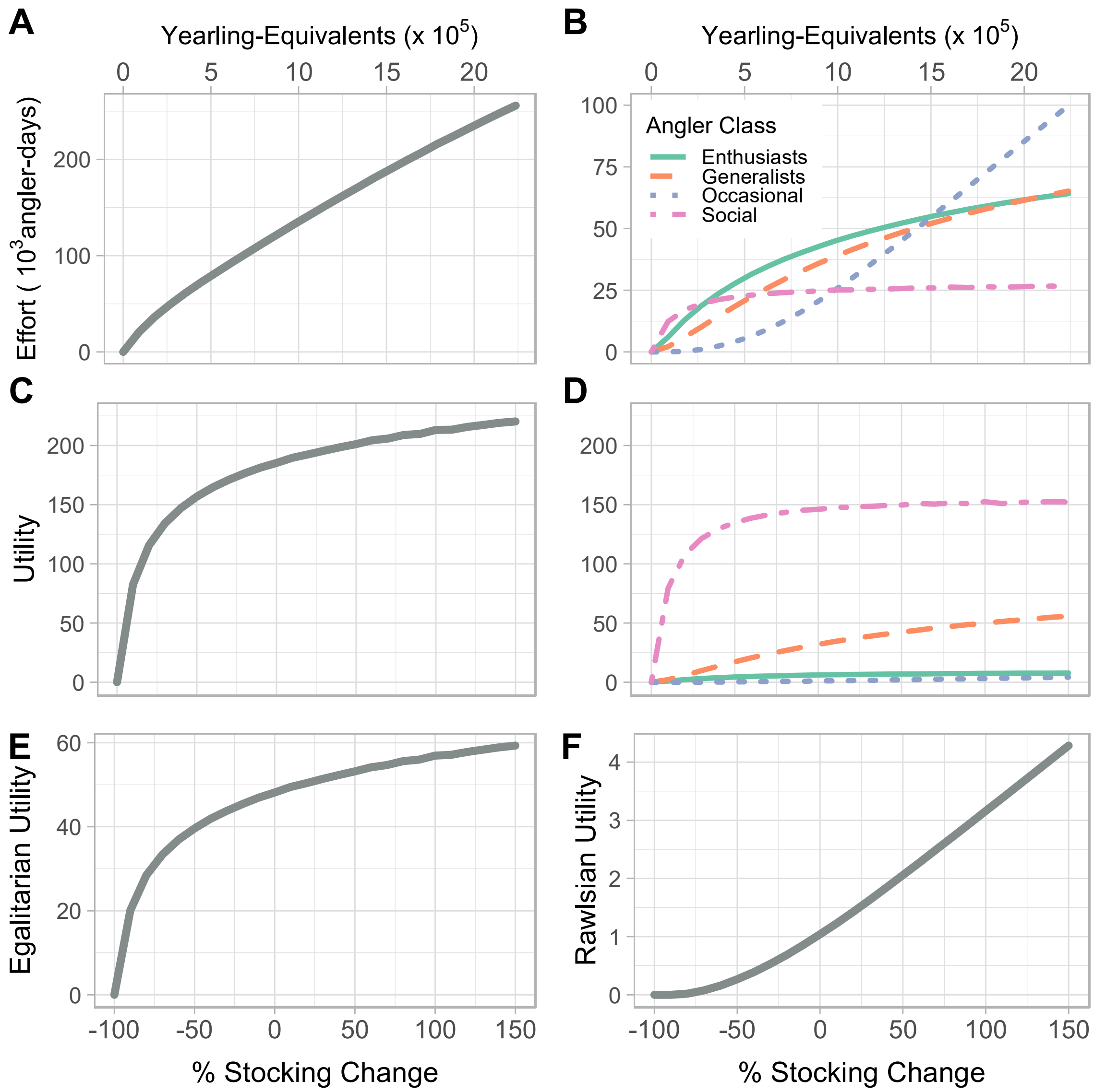

Figure 3. **Response of each of four hypothesized management objectives to stocking changes. Regional stocking rate varied from no fish to over 20 million yearling-equivalents, or a 150% increase from baseline. Total effort (A) is the sum of effort from four angler classes (B), and likewise for total utility (C, D). Egalitarian utility (E) weighted angler classes' utilities by their population size. Rawlsian utility (F) includes only the lowest-utility angler class. Note the differing y-axis scales.**

4.2. Landscape outcomes

The strategic stocking changes modelled as management alternatives for the Thompson-Nicola region resulted in between 850 and 3,650 additional angler-days of fishing effort compared to the ‘status quo’ scenario. The largest effort increase resulted from redistributing fish from low-effort lakes to higher-effort lakes (management alternative 7). Paradoxically, the reverse redistribution (from high-effort to low-effort lakes, management alternative 8) had the highest total utility and egalitarian utility. The effort-maximizing alternative scored relatively low in utility, and vice versa (Table 3). The highest Rawlsian utility always resulted from the status quo management scenario, which had the lowest total effort and total utility.

I calculated effort and utility outcomes for each combination of management alternative and uncertain parameter value and then ranked the alternatives from highest to lowest expected effort or utility. The rank order of management alternatives was sensitive to uncertainty in the selected parameters, however, the highest-ranked alternative rarely changed. For the U_{dist} and U_{size} intercept parameters, the highest effort alternative changed at the upper and lower ends of the range of values analyzed (Figure 4). There was no change in the highest-utility (total, egalitarian, or Rawlsian) management alternative across the range of simulated values.

Table 3. **Resulting effort, total utility, egalitarian utility, and Rawlsian utility on lakes in the Thompson-Nicola management region from nine simulated management alternatives. The highest outcome for each objective is highlighted with a black border. Shading represents the rank order of management alernatives within each objective, with the darkest shade for the highest values. Outcomes listed are calculated under original parameter values without uncertainty.**

Management Alternative ID	Description	Total Effort (x10^3 angler-days)	Total Utility	Egalitarian Utility	Rawlsian Utility
1	Close to far lakes (10%)	124.63	190.70	49.51	1.03
2	Far to close lakes (10%)	125.41	195.45	50.73	0.92
3	Close to far lakes (20%)	125.34	194.63	50.49	1.02
4	Far to close lakes (20%)	125.42	196.73	51.07	0.82
5	Low to high effort lakes (10%)	125.85	187.63	48.88	0.95
6	High to low effort lakes (10%)	124.72	200.91	51.97	1.01
7	Low to high effort lakes (20%)	127.42	188.38	49.17	0.87
8	High to low effort lakes (20%)	125.31	208.76	53.92	1.00
9	Status quo	123.78	185.49	48.22	1.04

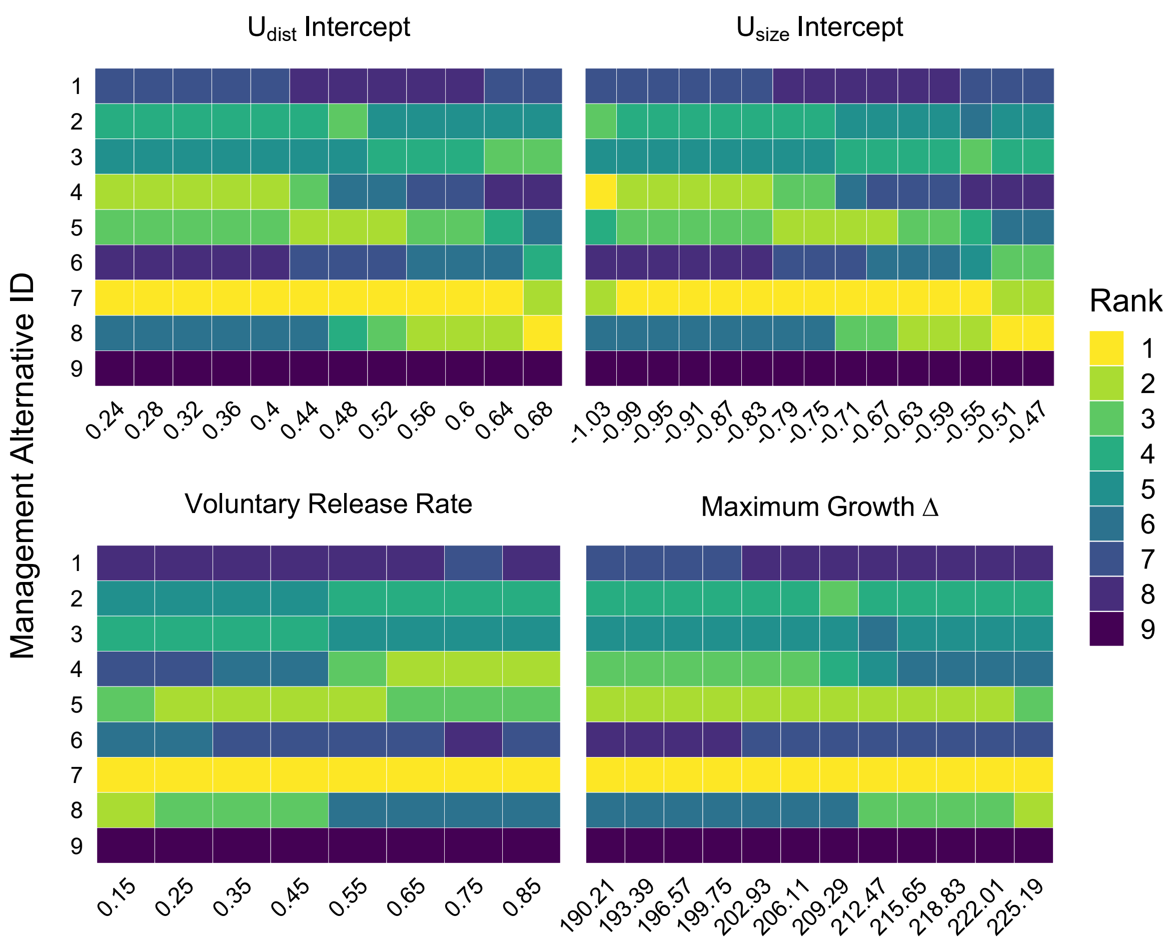

Figure 4. **Heatmap of management alternatives resulting in the highest region-wide effort (rank 1; yellow) to the lowest effort (rank 9; purple), across simulated values of four parameters. Highest and lowest ranks are relatively stable to uncertainty in parameter values, while middle-ranked alternatives shift readily. Management alternatives are described in Table 1.**

4.3. Value of information

The expected value of perfect information was zero for all measures of utility, because the highest-utility decisions were robust to all parameter values tested. For effort, however, there were small positive EVPI values for the utility intercept parameters: 6.22 angler-days for U_{dist} and 39.35 angler-days for U_{size}. The monetary value of a day of angling effort is approximately $127 (Southwick Associates, 2020), making the monetary value of perfect information on these parameters $789.94 and $4,997.45, respectively.

Chapter 5. Discussion

I found that optimal management policies were robust to uncertainty in four key parameters after simulating nine management regimes under a range of values for these parameters. As such, the EVPI was small or none, indicating that there is little value in seeking to improve estimates of model parameters. I found trade-offs among objectives, with management changes that maximized fishing effort entailing significant losses in utility (total and egalitarian). These key results suggest that defining objectives should be a priority for managers of this fishery.

5.1. Effort and utility responses to stocking

My analysis revealed that regional angling effort and aggregate utility measures (total and egalitarian) have different functional relationships with the regional stocking rate (Figure 3). Despite lake-specific utilities being the driver for lake-specific fishing effort, effort and utility at the landscape scale are not proportionally related in the SSES model. Rather, effort for a given lake is related to its fraction of the total utility of the landscape, i.e., the utility of all lakes and the utility of not angling (Eq. 2). A small amount of utility can therefore result in a large effort response, as seen in the occasional angler class, which drives the increase of total effort at high stocking rates (Figure 3). Effort and utility also have key conceptual differences in the SSES model - while utility is a theoretical measure of lakes' attractiveness to anglers, effort is the realized state of the fishery. The distribution of effort also depends on the population and spatial distribution of anglers, the proportion of each angler type present, and their propensity to participate in the fishery. Previous work with SES models has similarly demonstrated the influence of population size, composition, and spatial distribution on regional effort patterns (Hunt et al., 2013; Matsumura et al., 2019; Wilson et al., 2020).

Another potentially significant driver of the overall utility and effort patterns is the relationship between the stocking rate and the resulting catch rates that largely influence utility and effort. The SSES model assumes that fish abundance continues increasing with stocking, i.e. there is no density dependence in the survival of stocked fish. Although catch rates differ between angler types, all are non-decreasing, despite evidence suggesting that a stocking increase of the scale we simulated would decrease fish abundance and therefore catch rates in at least some lakes (Post et al., 1999). This structural assumption in the SSES model likely had a large influence on the non-decreasing stocking-effort and stocking-utility relationships we observed.

The nearly linear increase of total angling effort (Figure 3) resulted from the aggregation of four distinct responses to stocking – with social, generalist, and enthusiast anglers contributing most of the effort at low stocking rates, and occasional anglers driving the pattern at high stocking rates after the effort of the other classes had largely saturated. The effort response of each angler type reflects the utility they associate with catch-related aspects of fishing, i.e., catch rates and average size of catch, which are inversely related (Wilson et al., 2016). Occasional anglers gain utility from the high catch rates that result from intensive stocking and lose little utility from catching small fish. However, other angler types have strong preferences against small fish (i.e., enthusiasts) or gain almost no utility from catching more than one fish per day (i.e., social anglers). The ability for these individual-scale preferences to result in landscape-scale patterns of exploitation has been observed by others, and is a compelling reason to model heterogeneous human behaviour as part of the fishery system (Arlinghaus et al., 2017; Hunt et al., 2013; Ward et al., 2016; Wilson et al., 2020).

The extent to which anglers are motivated by expected catch rates, called catch-orientation, is often highly variable across anglers, which can lead to difficulty in achieving management objectives (Arlinghaus, 2006; Beardmore et al., 2011; Dabrowksa et al., 2017; Hunt et al., 2019; Stoeven, 2014). Less catch-oriented anglers may sustain higher fishing effort at low abundances, because a relatively small proportion of their utility comes from catch (Hunt et al., 2011; Stoeven, 2014). The social angler class in the SSES model has the characteristically stable effort associated with low catch-orientation (Hunt et al., 2011; Stoeven, 2014). The occasional angler class, however, has effort nearly proportional to stock size, characteristic of highly catch-oriented anglers. These angler types also differ in their skill, and therefore the mortality they inflict. Differences in effort responses and catchabilities can interact to create unexpected and potentially undesirable responses to stocking. Simulation experiments have revealed, for example, that an angling population likely to maintain high effort at low catch rates often contributed to collapsing a fishery (Golden et al., 2022). Managers can use information about the behaviour of various angler classes along with demographic information to better predict management outcomes, even without the use of a model. Angler demographics and preferences are known from previous work in this fishery, but thus far this information has not been formally integrated into management decisions.

5.2. Analysis of management alternatives

The best-performing management option differed among objectives. Redistribution of fish to low effort lakes (alternative 8) resulted in the highest total utility and egalitarian utility across most parameter values, while the highest regional effort resulted from redistributing to lakes which already sustained high effort (alternative 7). Unexpectedly, the highest effort alternative scored relatively low in all measures of utility. Trade-offs among objectives are common in recreational fisheries, which often have conflicting goals of conserving fish stocks and providing fishing opportunities (Camp et al., 2017). However, effort and angler utility are highly related, and are not generally considered to be in conflict. This trade-off suggests that even seemingly complementary objectives are achieved by different strategies. Because small differences in the definitions and formulation of objectives can lead to different optimal outcomes (Johnston et al., 2010), it is important that governance agencies carefully consider the desired outcomes for the fishery, and the values that they reflect (Fenichel et al., 2013*b*). Reluctance to define objectives has been a widespread issue in fisheries since the recognition that biological objectives, such as maximum sustainable yield, are inadequate (Lackey, 1998; Larkin, 1977). This may be due to the increased complexity of formulating social, biological, and economic objectives, or an unwillingness to confront value-laden questions about the relative importance of each of these (Lackey, 1998; Symes & Phillipson, 2009). Even so, well-defined objectives are crucial for unbiased evaluations of past and future management decisions, which allow for constant improvement and accountability (Barber & Taylor, 1990).

In testing four postulated objectives for BC's stocked rainbow trout fishery, I found that current stocking practices in the Thompson-Nicola region are most consistent with an objective of maximizing Rawlsian utility. This strategy implicitly values not alienating any angler group, but comes at the expense of total welfare provided by the fishery, because the stocking strategies preferred by the lowest-utility angler group (occasional anglers) did not result in high utility for any other group. Managing to maximize utility or angler satisfaction implies valuing the social benefits that the fishery provides, and requires carefully defining a measurable metric that reflects how various angler groups' utilities are valued. Alternatively, an economic objective may be deemed more appropriate. It has been suggested that sustainable effort can be used as a performance metric for this fishery (Askey et al., 2013). Unlike in naturally recruiting fisheries, it may be possible to manage for high effort, providing maximum economic benefits, without negative consequences for the sustainability of fish populations. Despite trade-offs between satisfaction and effort, it has been suggested that large losses in satisfaction could be avoided with careful stocking strategies (Askey et al., 2013). Objectives could be formed that aim to achieve high effort while maintaining satisfaction, for example, maximizing effort within some threshold angler utility. However, managers should recognize that both may not be achieved by the same strategies, and avoid assuming that stocking is a panacea that can allow for trade-offs to be bypassed.

5.3. The value of information

The best-performing alternative for each objective in this analysis was relatively robust to uncertainty in all four parameters (Figure 4). As such, the expected value of perfect information was either zero (V and Δ) or very small relative to the scale of the fishery (39 angler-days and 7 angler-days for U_{dist} and U_{size} intercepts; <0.03% of regional effort). The EVPI is interpreted as the maximum benefit of reducing uncertainty about the true value of these parameters. These results suggest additional information pertaining to these parameters would not change the decision on how to manage the fishery. Further, the cost of any information collecting activities would exceed the monetary value of the potential effort gain (~ $900 - $5,000). The EVPI may be low because the model predicted similar effort and utility for each management scenario, resulting in relatively little value that could be gained (Walters, 1986). Furthermore, symmetry in the probability distributions of the parameters caused the expected values to trend toward the baseline value, making the *EVwPI* and the *EVU* similar. Although sensitivity analyses showed that the model is sensitive to the parameters analyzed, that sensitivity was only observed in the decision analysis at the limits of the range of U_{dist} and U_{size} values. Because I used estimates of uncertainty from the literature, the range of values analyzed may not have been large enough to observe the model's sensitivity. In this case, it may be fitting to interpret the EVPI results as indicating that previous research in this fishery has led to enough certainty about these parameters to allow for optimal decision-making. Information about model parameters may have already reached a state of diminishing returns, where further reducing uncertainty may not change management decisions (Hansen & Jones, 2008). This suggests that future work with these complex models should focus on structural uncertainty in the model and defining management objectives.

5.4. Insights from SES model application

The SSES model includes several processes known to influence fishery outcomes, such as angler heterogeneity, a utility-based choice model, a management model, and spatial effort patterns (Johnston et al., 2010; Matsumura et al., 2019; Wilson et al., 2020). It also is fitted to data from the fishery, placing it at the frontier of social-ecological modelling in this field (Solomon et al., 2020). The rising popularity of social-ecological models, and the expansion of processes that are included, follow a widespread trend in modelling toward increasing complexity (Cilliers et al., 2013; Solomon et al., 2020). However, complex models have trade-offs. For example, complexity may inhibit the use of a model by others, including managers of the fishery it represents. Modellers are tasked with interpreting the system into a formalized model and are often the only ones familiar with this interpretation (Cilliers et al., 2013). Understanding how model results can be interpreted back to information about the system can be especially difficult for the non-modellers, and may lead to incorrect assumptions and interpretations. Further contributing to challenges with interpretation, large and complex models have many parameters, some of which may be redundant (Raick et al., 2006; Walters, 1986). Establishing causal links between model inputs and their outcomes is challenging with many parameters and a complex structure. As a result, fishery managers may misinterpret results, or become frustrated and reluctant to make use of complex models. Studies in model complexity have found that simpler models can make predictions nearly as accurately as complex ones, suggesting the rule of thumb that the ideal model size is the minimum size that produces sufficiently accurate predictions (Walters, 1986). However, it is worthwhile to consider the objectives of the model and whether generating accurate predictions is the intended purpose

(Raick et al., 2006). Alternatively, exploratory modelling could be considered valuable for its ability to formally represent system complexities and identify emergent properties (Moallemi et al., 2020*a*; Ward et al., 2016).

5.5. Limitations

This work has limitations pertaining to the SSES model and methods used in applying decision analysis to it. First, decision analyses typically involve identifying objectives, management alternatives, and key uncertainties collaboratively with stakeholders (Mcallister, 1999). A model of the fishery system is often then constructed to predict outcomes from the pathways identified in initial discussions with managers, scientists, and industry members. Much of the power of decision analytic techniques lies in this collaborative approach, and especially in the co-production of objectives (Irwin et al., 2011). In contrast, this project began with an existing model, and objectives, management alternatives, and key uncertainties compatible with a model-based analysis were then identified. A downfall of this approach is that the model became the basis for the management options and uncertainties explored, rather than the issues that fishery managers face. For example, the SSES model outputs are fishery-related and do not include any ecological metrics. Although stocked rainbow trout rarely compete with other sport fish in this fishery, they may pose a threat to other species at risk (Hirner & Cox, 2007). Ecological objectives were ignored in this analysis but are likely implicitly considered by managers of this fishery. Another limitation of the decision analysis is the *ad hoc* approach used to identify management alternatives and uncertainties. Management alternatives should be co-produced with managers and stakeholders, however for the purposes of this project, I largely developed them independently. However, management alternatives applied in this analysis reflected realistic broad-scale policy considerations as much as possible.

The structure of the SSES model likely also influenced the results of this analysis. For example, the decision not to model density-dependent survival may have had a significant influence on the resulting model predictions, and in turn on the EVPI results. With density dependence included, very high stocking rates would result in die-offs that would lower catch rates, and therefore utility and effort. The responses of the objectives to stocking would be different, possibly altering the rank order of outcomes, and leading to different results and key findings. The SSES model's structure and how it influences predictions should be explicitly examined before it is used to evaluate real management decisions. Structural uncertainty in models can be explored through decision analysis in the same way I explored parameter uncertainty (e.g., Mäntyniemi et al., 2009), but it is much less common, and model structure is usually an inflexible component of the analysis (Moallemi et al., 2020*a*). By only examining parameter uncertainty, this analysis overlooked important sources of structural uncertainty in the model.

5.6. Conclusions

The objectives of this project were to (1) evaluate the sensitivity of system performance under different sets of objectives, (2) calculate the value of information for key parameters to identify priorities for future monitoring and research, and (3) identify trade-offs among sets of candidate management policies and fishery objectives. Although the value of information did not point to a direction for prioritizing future research, it suggested that additional data may not be a priority for this fishery at all. The BC rainbow trout fishery has a relatively extensive data collection program, and is one of the most frequently studied recreational fisheries worldwide (Solomon et al., 2020). While calls for more research and data are ubiquitous in science, no amount of research can eliminate uncertainty, and reducing uncertainty does not necessarily result in better management (Hansen & Jones, 2008). In exploring optimal management for this fishery, I found that seemingly complementary objectives were met by different management strategies. Taken together, these results show that this fishery would benefit more from defining quantifiable management objectives than from any additional data collection or research initiative.

Effective long-term natural resource planning requires clear goals and objectives with which to evaluate past and future actions. While the original intent of this work was to influence day-to-day management decisions and perhaps some longer-term strategy, the results that emerged point to a need for additional rigor needed in the process of identifying and implementing values and objectives. One of the goals of this fishery is to “establish governance approaches that are strategic, effective and efficient” (BC MOE, 2007, p.17) - establishing measures of success is a necessary precursor.

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