THE CONSEQUENCES OF CONNECTIVITY: INFORMING FISH PASSAGE AND RESTORATION DECISIONS WITH DECISION ANALYSIS

By

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ABSTRACT

Dams have dramatically altered riverine systems and are a major contributor to native fish population declines. Dam removal is now a common stream rehabilitation practice in the United States; however, many dams serve important ecological, social, and economic functions, such as flood control, invasive species control, and provision of recreational opportunities. Therefore, dam removal is often contentious among stakeholders and involves making tradeoffs among multiple competing objectives. Decision makers benefit from a thorough evaluation of the ecological, economic, and social consequences and tradeoffs of changes to connectivity using decision analysis and predictive models, yet few examples exist in the literature. The objectives of this research were to use a decision analytic framework and predictive models to evaluate the ecological, economic, and social consequences and tradeoffs of enhancing connectivity for migratory fishes in the Great Lakes basin.

In Chapter 1, I used structured decision making to engage a diverse group of stakeholders and rightsholders to evaluate the ecological, social, and economic consequences and tradeoffs of enhancing connectivity for migratory fishes in the Boardman-Ottaway River watershed. The optimal management alternative was passage of native fishes only; however, the optimal alternative varied based on the weight stakeholders might place on each objective. Four weighting scenarios were developed to evaluate the change in optimal management alternative with changes in objective weights.

In Chapter 2, an individual-based model framework was developed to forecast the response of migratory fishes to changes in connectivity and applied to predict the change in abundance and growth of six species under various fish passage scenarios on the Boardman-Ottaway River. Population response to barrier removal was species-specific and varied based on

initial population size and distribution within the watershed, the number of fish passed upstream, and species life history traits. Species that were found only below the barrier prior to removal benefitted most. Non-native species were found to have greater production potential than native Great Lakes basin species under full passage scenarios.

In Chapter 3, I evaluated stocking scenarios for lake sturgeon *Acipenser fulvescens* to aid decision making in the rehabilitation of an imperiled native Great Lakes basin species. Using the model from Chapter 2, several lake sturgeon stocking scenarios were simulated to forecast the potential time to reach a recovered lake sturgeon population. Sensitivity analysis and expected value of perfect information were used to elucidate key areas of uncertainty. The median time to reach the target abundance was estimated to take between 31 and 91 years, depending on the stocking strategy. The results of this research will help inform decision-makers in the Great Lakes basin on management alternatives for fish passage and restoration that are preferred by stakeholders and that are likely to achieve their objectives.

This dissertation is dedicated to Mom and Erika.
Thank you for always believing in me.

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PREFACE

The materials in chapters one, two, and three were developed with the intention of publication in peer-reviewed journals, and the language used there reflects the assistance of co-authors.

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INTRODUCTION

Barriers to stream connectivity, specifically dams, are ubiquitous throughout the United States and have dramatically altered stream ecosystems. Dams have contributed to decreases in migratory fish populations through habitat fragmentation and flow alteration. With increasing awareness of the negative ecological effects of barriers to stream ecosystems, barrier removal has become a common stream rehabilitation action (Pess et al. 2014; O'Connor et al. 2015). In the United States, 2,025 barriers have been removed since 1912 (American Rivers 2024). Oftentimes, increasing access to spawning habitat for migratory fishes is a major impetus for dam removal, and barrier removal has been shown to positively affect lotic fish communities by increasing species richness and promoting recolonization of upstream habitats (Catalano et al. 2007; Brenkman et al. 2019). Barrier removal also restores physical stream processes and conditions (i.e., temperature and instream flow) and increases access to critical life-stage habitats for migratory fishes, as well as energy and nutrient transport from lake and marine systems to stream habitats (Flecker et al. 2010; Childress and McIntyre 2016). Although increasing connectivity is beneficial to stream systems and migratory fishes, barrier removal and remediation decisions often have multiple, competing objectives that can lead to tradeoffs between the benefits to desirable species and the potential increased production of undesirable and invasive species, alongside myriad other concerns (Kuby et al. 2005; Zheng et al. 2009; Zheng and Hobbs 2013).

Dams fragment stream habitats, disrupt physical stream processes (i.e., stream flow and temperature), and have led to the decline of migratory fish populations; however, many dams simultaneously serve important ecological, social, and economic functions, such as flood control, invasive species control, and provision of recreational opportunities. Therefore, dam removal is often contentious among stakeholders and involves making tradeoffs among multiple competing objectives (Zheng and Hobbs 2013; Fox et al. 2016). Ding et al. (2019) noted a disparity in dam removal trends between developing and developed countries, due to developing countries prioritizing economic development over environmental conservation. In some situations intentional fragmentation could be preferred to produce positive ecological outcomes by reducing the production or spread of an invasive species (McLaughlin et al. 2013; Rahel 2013; Jensen and Jones 2018). In the Great Lakes, a "connectivity conundrum" (Zielinski et al. 2020) exists where barriers provide the most effective control for invasive sea lamprey *Petromyzon marinus* populations, yet limit access to spawning habitat for native fishes such as lake sturgeon *Acipenser fulvescens,* and walleye *Sander vitreus,* thereby reducing production and leading to population declines.

A solution to the connectivity conundrum is selective connectivity, or the ability to control which species—and how many—pass through a barrier to stream connectivity. Pratt et al. (2009) contains perhaps the first use of the term "selective connectivity" related to fish passage and the control of sea lamprey in the Great Lakes; however, what the paper describes as selective connectivity is manual sorting of fishes at a barrier. Similarly, translocation has been used to pass species such as white sturgeon *Acipenser transmontanus* (Jager 2006a) and lake sturgeon *Acipenser fulvescens* (Koenigs et al. 2019) over barriers, without barrier removal or improvements to passageways. However, for many years now, engineers, biologists, and others have been working to achieve selective connectivity without the need to manually sort and pass fishes. The Great Lakes Fishery Commission FishPass project will renovate the existing Union Street Dam on the Boardman-Ottaway River in Traverse City, MI and develop a bi-directional, selective fish passage facility to allow passage of desired species (e.g., lake sturgeon), while excluding non-desired species (e.g., sea lamprey; Zielinski et al. 2020).

The positive ecological outcomes of rehabilitation actions that improve stream connectivity (e.g., FishPass) are usually at the forefront of decision making; however, managers should also ideally consider social and economic consequences of changes to connectivity. Barrier removals have notoriously been contentious among stakeholders (Fox et al. 2016); therefore, a decision analysis framework such as structured decision making (SDM; Hammond et al. 1999; Clemen and Reilly 2001) can be particularly useful to increase transparency and improve stakeholder and rightsholder relations through engagement. Structured decision making is a facilitated and collaborative decision analysis framework that helps decision makers make more robust and transparent management decisions by incorporating the values and objectives of stakeholders. Stakeholders and rightsholders will value stream connectivity differently, have differing objectives, and potentially have species-specific preferences.

Migratory fishes have not accessed habitat upstream of Union Street Dam in over 150 years, and the response of fish populations is highly uncertain. A modeling framework to aid decision makers in predicting the response of fish populations to increases in connectivity before the change occurs would be an invaluable tool to fisheries managers, but is not currently available. There is a paucity of studies that predict fish population response to barrier removal before a removal occurs and much of the existing barrier removal literature documents changes to species assemblages post-barrier removal (Cooper et al. 2017; Whittum et al. 2023). Furthermore, monitoring efforts typically are not conducted over a long enough time frame to document the response of fish populations to changes to connectivity (Magilligan et al. 2016a, 2016b; Foley et al. 2017). However, available data suggest that fish populations are quick to colonize newly available habitats following barrier removal (Catalano et al. 2007; Burroughs et al. 2010; Whittum et al. 2023), but changes in abundance have been species-specific and the benefits of increased connectivity may take many years to develop (Sun et al. 2022). Prior to making changes to connectivity, decision makers should ideally weigh the ecological, economic, and social consequences of connectivity using predictive models and stakeholder engagement to make the most informed decision; yet few barrier removal or renovation projects include these considerations.

My dissertation used the Great Lakes Fishery Commission FishPass project as a case study to evaluate the ecological, economic, and social consequences and tradeoffs of enhancing stream connectivity in the Great Lakes region. The overarching goal of my research was to use structured decision making and predictive modeling to help inform fisheries managers when making management decisions surrounding novel connectivity regimes which have the potential to affect stakeholders and rightsholders. In my first chapter, I organized a working group of local stakeholders and rightsholders to work through the PrOACT framework (Hammond et al. 1999) of structured decision making. I elicited stakeholder and rightsholder objectives for enhancing connectivity on the Boardman-Ottaway River and estimated the consequences of different management alternatives to help decision-makers, stakeholders, and rightsholders make tradeoffs among multiple competing objectives. I used a simple multi-attribute ranking technique to determine the optimal management alternative under differing objective weighting schemes to better understand how the optimal management alternative may vary by stakeholder type. In my second chapter, I focused on predicting the response of fish populations to changes in connectivity and more thoroughly detail the modeling process I developed and used to model the consequences in the first chapter. Based on the desires of the working group, I developed an individual-based model (IBM) framework to predict the response of six species to various selective connectivity alternatives that were identified by the working group during the SDM workshops in Chapter 1. The modeling framework I developed is generalized and highly flexible; therefore, this framework can be improved upon and updated in the future, as well as applied to additional regions of the Great Lakes and beyond. Finally, in my third chapter, I consider what happens when connectivity is not expected to restore populations of some species. To address this question, I modified the IBM framework from Chapter 2 to forecast outcomes of different lake sturgeon stocking strategies and reduce uncertainty in lake sturgeon stocking decisions. The Grand Traverse Band of Ottawa and Chippewa Indians are interested in stocking lake sturgeon in the Boardman-Ottaway River to improve restoration efforts, and this model was used to identify potentially successful stocking strategies, elucidate areas of critical uncertainty in lake sturgeon life history, and help set expectations for the rehabilitation process.

In total, my dissertation is a comprehensive evaluation of the consequences of connectivity —or the possible outcomes of barrier removal or renovation— as well as the development of a decision tool to inform future fish passage decisions on the Boardman-Ottaway River and beyond. First, I used structured decision making to incorporate economic and social objectives of barrier removal into the decision process. I then developed a modeling framework to forecast changes to fish production pre-barrier removal to improve decision making and be iteratively improved upon in the future. Finally, I evaluated stocking scenarios for lake sturgeon to inform additional management action when enhanced connectivity is predicted to be inadequate in reaching objectives for population restoration. Overall, this research will aid decision-making when changes to stream connectivity are considered, which is timely and important because barrier removal and renovation will likely increase in the future in the United States and Europe (Ding et al. 2019).

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CHAPTER 1: ASSESSING THE OUTCOMES OF SELECTIVE FISH PASSAGE WITH STRUCTURED DECISION MAKING

Abstract

Dams have dramatically altered riverine systems and are a major contributor to native fish population declines. However, many dams serve important ecological, social, and economic functions, such as flood control, invasive species control, and provision of recreational opportunities. Therefore, dam removal is often contentious among stakeholders and involves making tradeoffs among multiple competing objectives. This research used Structured Decision Making to evaluate the ecological, social, and economic consequences and tradeoffs of enhancing connectivity for migratory fishes in the Great Lakes basin. We describe our efforts to engage a diverse group of stakeholders and rightsholders to elicit their objectives under various fish passage alternatives. We developed predictive models to help stakeholders weigh the costs and benefits of enhancing connectivity for several fish species with varying life history traits and initial distributions. We found that the optimal management alternative was passage of native fishes only; however, the optimal alternative varied based on the weight stakeholders might place on each objective. We created four weighting scenarios to evaluate the change in optimal management alternative with changes in objective weights. The results of this research will help inform decision-makers on fish passage alternatives that are preferred by stakeholders and that are likely to achieve their objectives.

Introduction

Dams have dramatically altered rivers throughout North America by changing flow and fragmenting habitat (Dynesius and Nilsson 1994; Cooper et al. 2017). Habitat fragmentation and flow alteration by dams have affected fish assemblages at least as much as other anthropogenic stressors (Cooper et al. 2017), and enhancing connectivity (e.g., dam removal) is important for rehabilitation of migratory fish populations (Pess et al. 2014; Watson et al. 2018). Enhancing connectivity can restore physical stream processes and conditions (i.e., temperature and flow regimes), improve accessibility of spawning habitat to migratory fishes, and increase nutrient transport from marine and lake systems to rivers. Due to increasing awareness about the negative effects of dams on migratory fish populations, a desire for free-flowing rivers, and aging infrastructure (Zheng and Hobbs 2013), many dam removal projects have recently been undertaken in the United States (Foley et al. 2017).

The majority of dam removals have occurred in North America and Europe (Ding et al. 2019), with migratory fish passage being a major impetus for these removals. For example, dam removals in the Pacific northwest USA have typically been aimed at rehabilitating salmonid populations *Oncorhynchus spp.* (Pess et al. 2008) and in the northeast USA at rehabilitating populations of Atlantic salmon *Salmo salar* (Nieland et al. 2015), American eel *Anguilla rostrata* (McCleave 2001), American shad *Alosa sapidissima* (Pess et al. 2014), and endangered shortnose sturgeon *Acipenser brevirostrum* (Johnston et al. 2019). Dam removals have been shown to be a viable stream restoration technique to restore fish populations (Birnie-Gauvin et al. 2018; Brenkman et al. 2019), and even small dam removals have resulted in positive shifts in lotic fish communities indicated by increased species richness, decreases in proportion of species tolerant to environmental disturbance (e.g., common carp *Cyprinus carpio*), increased abundance of intolerant species (e.g., brook trout *Salvelinus fontinalis*), and recolonization of upstream habitats by species that were previously found only downstream of—or rarely above—dams (Catalano et al. 2007).

Although dams can negatively affect river habitats and fish populations, the decision to remove them is not always straightforward or uncontentious (Fox et al. 2016; Brummer et al. 2017; Rahel and McLaughlin 2020). Dams serve important socio-economic functions and can provide recreation, hydropower generation, and flood control (Pejchar and Warner 2001; Kuby et al. 2005; Song et al. 2019). Furthermore, dam-affected river systems have 25-times greater economic output per unit water than undammed systems (Nilsson et al. 2005). Therefore, decision makers should ideally consider the values and objectives of a diverse group of stakeholders and rightsholders (hereafter "stakeholders," but we recognize some groups are rightsholders, including Indigenous communities and governments) and make potentially difficult tradeoffs among values and objectives related to maximizing abundance of fish populations, restoring ecological processes and cycles, dam safety, energy production, recreational uses, among others (Zheng et al. 2009; Zheng and Hobbs 2013; Song et al. 2019). Additionally, dams can block upstream movement of non-native and invasive species, which further complicates decision making surrounding dam removal (McLaughlin et al. 2013; Rahel and McLaughlin 2020; Zielinski and Freiburger 2021). In fact, in some instances it may be beneficial to keep an aquatic system intentionally fragmented to preserve native fish populations or stop the spread of invasive species (Peterson et al. 2008; Rahel 2013; Milt et al. 2018; Zielinski and Freiburger 2021).

The Laurentian Great Lakes of North America are among the planet's most invaded systems (Ricciardi 2006), and invasive species have profoundly affected Great Lakes fish communities. Sea lamprey *Petromyzon marinus* is an invasive species first detected in the Great Lakes in the 1830s that preys on economically important fishes, such as native lake charr *Salvelinus namaycush* and lake whitefish *Coregonus clupeaformis*, as well as non-native Pacific salmon. Sea lamprey populations are currently controlled by treatment of rivers with lampricide to kill larvae and barriers to prevent adults from accessing spawning habitat (Siefkes et al. 2021). Dams and natural barriers are the most effective control mechanism for sea lamprey in the Great Lakes because they prevent adult sea lamprey from accessing quality spawning habitat without the need for lampricide treatments that are costly and can be harmful to other desirable aquatic organisms (Hrodey et al. 2021). Jensen and Jones (2018) found that when new habitat becomes available to sea lamprey—given a fixed lampricide budget—sea lamprey populations would increase and control by lampricide treatment would become less effective. Therefore, when considering barrier removal and remediation in the Great Lakes, the benefit of opening new habitat to native migratory fishes must be weighed against the cost associated with increased sea lamprey abundance and subsequent harm they inflict on valuable fisheries.

An additional ecological concern when considering changes to connectivity is the existence of desirable non-native species. Native fish populations in the Great Lakes region such as lake charr, lake sturgeon *Acipenser fulvescens*, walleye *Sander vitreus*, white sucker *Catostomus comersonii*, smallmouth bass *Micropterus dolomieu*, and brook trout have been negatively affected by barriers (Bednarek 2001; Burroughs et al. 2010), but also by the introduction of non-native species such as Pacific salmonids (Fausch and White 1986; Huckins et al. 2008). Pacific salmon, the most common being Chinook *Oncorhynchus tshawytscha* and coho *O. kisutch*, were introduced to Lake Michigan in the 1960s to control invasive alewife *Alosa pseudoharengus* and provide open-water sport fishing opportunities. The stocking of non-native species has resulted in increased competition with native species for food and spawning habitat, increased predation on native species by non-native salmonids, introduction of non-native diseases and parasites, and genetic alteration such as hybridization (Crawford 2001;

Kornis et al. 2020). Furthermore, the construction of fish ladders has improved connectivity for some fishes, but their design allows for the passage of only strong swimming fishes with leaping ability (i.e., salmonids) and may still block the passage of weak swimming or non-leaping native fishes (Mallen-Cooper and Brand 2007). Therefore, in the Great Lakes, fish ladders may provide a negative feedback loop whereby native fishes are restricted to spawning habitat below a dam while non-native Pacific salmonids are allowed upstream, possibly resulting in further declines of native fishes (Mallen-Cooper and Brand 2007).

Ecological concerns, such as control of invasive species and native fish population rehabilitation, are usually at the forefront of decision making when enhancing connectivity in rivers, often called the "connectivity conundrum" (Zielinski et al. 2020); however, social and economic concerns should ideally also be considered (Zheng and Hobbs 2013; Fox et al. 2016; Brummer et al. 2017). Although enhanced connectivity is often beneficial for stream ecosystems, some stakeholders may value reduced connectivity for a variety of reasons (McLaughlin et al. 2013; Rahel 2013; Milt et al. 2018). Therefore, barrier removal and remediation decisions should incorporate local stakeholders' desires and fish community objectives. Complex issues such as these will likely be contentious among stakeholders and are best handled using a process that involves consideration of stakeholders' values and objectives through collaboration of diverse stakeholder groups.

Structured decision making (SDM; Hammond et al. 1999; Clemen and Reilly 2001) is a facilitated and collaborative decision analysis framework that helps decision makers make more robust and transparent management decisions by incorporating the values and objectives of stakeholders. It is built on the PrOACT acronym where decision makers and stakeholders frame the problem (Pr), determine objectives (O), develop alternative management actions (A), estimate the consequences of alternative management actions (C), and evaluate tradeoffs (T; Figure 1.1; Hammond et al. 1999). SDM is increasingly being used in natural resource management (Wright et al. 2020), including the management of recreational fisheries (Peterson and Evans 2003; Irwin et al. 2011), endangered species (Gregory and Long 2009), and invasive species (Blomquist et al. 2010); managing risk of disease (Sells et al. 2016); and creating regulations for wildlife harvest (Robinson et al. 2017). In the Great Lakes, an SDM process involving fisheries managers and stakeholders representing recreational and commercial fishing interests was used to develop harvest control rules for Lake Erie percids (Jones et al. 2016); more recently, SDM was used to identify methods for reducing densities of invasive grass carp *Ctenopharyngodon idella* in Lake Erie (Robinson et al. 2021).

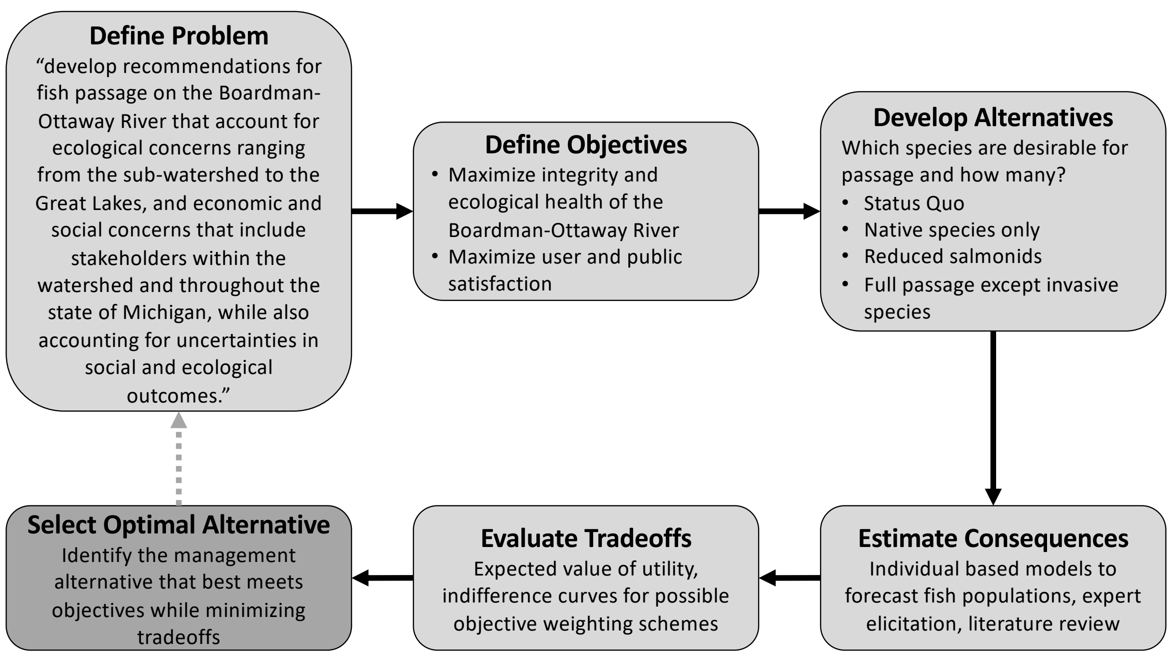

Figure 1.1. The structured decision making (SDM) process and PrOACT framework used to evaluate the decisions regarding fish passage at FishPass on the Boardman-Ottaway River, Traverse City, Michigan. The PrOACT framework begins at the top left (i.e., problem definition) and moves clockwise along the thick black lines. The dashed gray line is meant to emphasize the iterative nature of SDM and the ability to revisit previous steps as necessary.

Decision makers benefit from SDM because it is a thorough, deliberative process that considers uncertainty, which makes decision-making more transparent, defensible, and robust. In addition, it is a values-focused approach that directly considers objectives associated with management decisions (Keeney 1992). For these reasons, SDM is an ideal framework when considering barrier removals because these decisions involve diverse stakeholder groups and require making tradeoffs among multiple competing objectives (Lin et al. 2019a). Many studies have used decision analysis frameworks (e.g., decision support tools) to analyze optimization and prioritization strategies for barrier removals in a watershed (Kuby et al. 2005; Moody et al. 2017; Lin et al. 2019b); however, we are not aware of any published studies that use SDM to elicit the values and objectives of stakeholders, estimate consequences, and make tradeoffs for a single barrier removal or renovation project.

We present a case study describing how we used SDM to evaluate the ecological, social, and economic consequences and tradeoffs of different methods of enhancing connectivity for migratory fishes in the Great Lakes basin, using the Boardman-Ottaway River, Michigan, USA and Great Lakes Fishery Commission (GLFC) FishPass project (http://www.glfc.org/fishpass.php) as a case study, to better incorporate stakeholder views into the decision-making process for fish passage on the Boardman-Ottaway River. The goal of our SDM case study was to aid the Michigan Department of Natural Resources (MDNR) and Grand Traverse Band of Ottawa and Chippewa Indians (GTB) in their decision-making process for fish passage on the Boardman-Ottaway River. Barrier removal and renovations will probably continue and perhaps even increase in prevalence in the future. Therefore, the SDM case study described herein should be considered by fisheries managers and agencies as a way to defensibly and transparently make management decisions when multiple stakeholder objectives and goals should ideally be considered, which is a reality for nearly all barrier removal and renovation projects.

Study Area

The Boardman-Ottaway River is in the northwest lower peninsula of Michigan, USA (Figure 1.2). The watershed drains 743 km^2 of land in Grand Traverse and Kalkaska counties (Kalish et al. 2018), and the river discharges into West Grand Traverse Bay in Traverse City, MI. The Union Street Dam is a terminal barrier about 1.8 km upstream from the river's outlet to West Grand Traverse Bay. The Union Street Dam was constructed in 1867 (renovated in the 1960s) and soon after, three additional major dams were constructed on the river, Boardman (Keystone) Dam (1894), Sabin Dam (1906), and Brown Bridge Dam (1921). However, the Boardman-Ottaway River has recently been the site for multiple dam removals. Since 2012, Brown Bridge, Boardman (Keystone), and Sabin dams have been removed. The only remaining major dam on the mainstem of the river is the Union Street Dam, although there are about 20 dams with greater than six feet of head, and many small, private dams that still exist in the watershed (Kalish et al. 2018). The complete removal of Union Street Dam is not currently being considered because it operates as a barrier to invasive species, namely sea lamprey, as well as maintaining the water level in Boardman Lake (Kalish et al. 2018).

The Union Street Dam is the site for the GLFC FishPass project that aims to develop a bi-directional, selective fish passage facility to allow passage of desired species (e.g., lake sturgeon), while excluding non-desired species (e.g., sea lamprey; Zielinski et al. 2020). FishPass is coordinated by the GLFC, which is a binational organization established under the Canadian/U.S. Convention on Great Lakes Fisheries to coordinate fisheries research, control sea lamprey, and facilitate cooperative fishery management in the Great Lakes. Access to the habitat above Union Street Dam has been blocked for many migratory fishes for nearly 150 years, and there is substantial uncertainty regarding how fish populations will respond to selective passage at the Union Street Dam. Additionally, local stakeholder values vary among user groups of the watershed, and their concerns and desires regarding enhanced connectivity should ideally be considered in the decision-making process (Fox et al. 2016; Brummer et al. 2017). Therefore, a thorough evaluation of the ecological, social, and economic consequences and tradeoffs of enhanced connectivity was necessary to facilitate an informed decision-making process and measure success of the FishPass project.

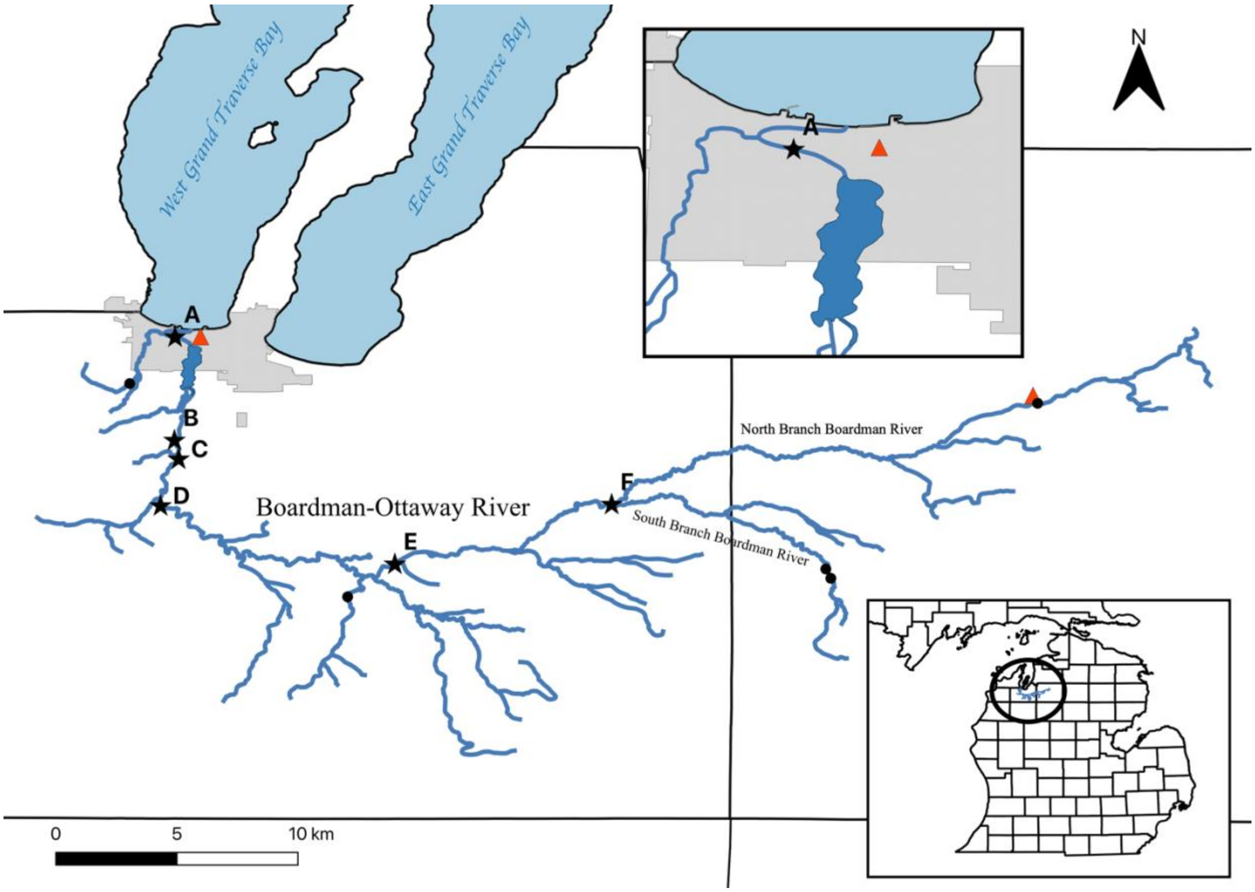

Figure 1.2. The Boardman-Ottaway River watershed and Grand Traverse Bays (Lake Michigan). Top inset showing the lower Boardman-Ottaway River, Traverse City (red triangle with city limits filled in gray), and Boardman Lake. Stars and letters (A–F) indicate points of interest within watershed; A=Union Street Dam; B= former Sabin Dam site; C= former Boardman Dam site; D= Beitner Rd.; E= former Brown Bridge Dam site; F= the Forks. Black dots are dams currently in the watershed with >6 ft of height. Red triangles are cities; Traverse City near West Grand Traverse Bay and Kalkaska along the North Branch. Bottom inset shows study area in relation to lower peninsula of Michigan, USA.

Methods

Overview of the FishPass SDM Process

To evaluate the ecological, social, and economic consequences and tradeoffs of enhancing connectivity for migratory fishes in the Boardman-Ottaway River via the FishPass project, we formed a group of local and regional stakeholders to participate in a series of SDM workshops (hereafter, the "working group"). Stakeholders defined the problem statement, defined their objectives, and developed potential management alternatives. We built simulation models that predicted changes in abundance, spatial distribution, and growth for six species to estimate ecological consequences and used constructed scales to predict social and economic consequences to evaluate the outcome of each management alternative. The results are used to better understand the potential consequences of each alternative and provide a means for choosing the optimal alternative. After the estimation of consequences, decision makers and stakeholders can make tradeoffs among objectives and determine which alternatives best meet their objectives.

Working Group

We included representatives of the decision makers in our case study, the MDNR and GTB. Additional stakeholders represented recreational fishing groups, local conservation groups, the Traverse City government, and the general public (Table 1.1). We aimed to have equal representation within recreation- and sustainability-focused groups versus all other groups (e.g., public), and furthermore, among native species (brook trout and walleye) and non-native species advocates (e.g., Pacific salmonids). Our final SDM working group was well-balanced, with stakeholders representing a variety of values and objectives (Table 1.1). In total, five virtual SDM workshops were held between October 29, 2020, and August 2, 2021. Workshop participation averaged nearly 13 people per workshop, with 12 people attending three or more workshops, and four people attending all five workshops (Table 1.1). The workshops with the working group were focused on scoping the problem, determining objectives, and describing a set of alternatives.

Table 1.1. The number of attendees and affiliations for seven focus groups of 21 total attendees of the five virtual structured decision-making workshops held between October 29, 2020 and August 2, 2021. NS = native species; NNS = non-native species.

Focus group	Number of attendees	Affiliation
Decision maker	4	Grand Traverse Band of Ottawa and Chippewa Indians and Michigan Department of Natural Resources
Facilitator	2	Michigan State University
General	2	Traverse City Public Services and Contractor with GLFC
Public	5	Traverse City Residents and Boardman-Ottaway River Property Owner
Recreation (NS)	4	Brook Trout Coalition, Adams Chapter Trout Unlimited, and NWMI Fishing Club
Recreation (NNS)	2	Traverse City Area Steelheaders Club and Michigan Steelhead and Salmon Fishermen's Associations
Sustainability	2	Grand Traverse Conservation District and Conservation Resource Alliance

Problem Statement

In SDM, the problem statement captures the overarching problem that the workshops are meant to address and guides the rest of the decision process (Hammond et al. 1999; Gregory et al. 2012). The problem statement for this case study contains information about important factors in the decision process for increasing fish passage at FishPass. For this project, the working group defined the problem statement as a need "to develop recommendations for fish passage on the Boardman-Ottaway River that account for ecological concerns ranging from the sub-watershed to the Great Lakes, and economic and social concerns that include stakeholders within the watershed and throughout the state of Michigan, while also accounting for uncertainties in social and ecological outcomes." The problem statement was revisited in later workshops but remained unchanged. Ultimately, decision-making authority regarding fish passage on the Boardman-Ottaway River lies with the MDNR and GTB; however, decisions made on the Boardman-Ottaway River may affect stakeholder objectives on a larger-scale, for example, if riverine production were to increase to a point that export to Lake Michigan occurred. Finally, stakeholders wanted to explicitly acknowledge uncertainty in potential outcomes. The uncertainties raised by stakeholders included the spread of invasive species, the effect of the nutrient-poor state of the Boardman-Ottaway River on native fish rehabilitation, the recreational user capacity of the watershed, and the potential for rare and unconsidered outcomes.

Objectives

After developing the problem statement, stakeholders described their fundamental and means objectives. Fundamental objectives are the outcomes that stakeholders specifically want to achieve, and means objectives are the ways to achieve those fundamental objectives (Keeney 1992; Gregory et al. 2012). Process objectives, which are objectives about how a decision is made, can also be included in an SDM framework. The working group also defined measurable attributes for their objectives to measure the level of achievement of each objective.

The working group identified two fundamental objectives and several means objectives regarding fish passage on the Boardman-Ottaway River that were arranged into an objectives hierarchy (Figure 1.3). The two fundamental objectives were to: (1) maximize the integrity and ecological health of the Boardman-Ottaway River, and (2) maximize user and public satisfaction. The means objectives for maximizing integrity and ecological health of the Boardman-Ottaway River were to maximize connectivity of the river, maximize native and coldwater species protection and rehabilitation, minimize the ecological effects to the river by increased human use, minimize negative human effects to the watershed as a result of tourism, minimize introduction of contaminants from migratory Great Lakes fishes, and minimize access to aquatic invasive species (Figure 1.3). The means objectives for maximizing user and public satisfaction were to maximize sustainable catch of brook trout and other naturalized resident trout (i.e., brown trout *Salmo trutta*), maximize sustainable catch of native species from Lake Michigan and

Boardman-Ottaway River for Tribal and non-Tribal fishers, maximize sustainable harvest of Pacific salmonids for recreational and commercial fishers, minimize user conflict, and minimize negative effects to resident riverfront homeowners (Figure 1.3).

The performance measures we used to measure the level of achievement of each means objective varied among the objectives but largely revolved around the abundance of six migratory species found in the watershed; brook trout, Chinook salmon, lake sturgeon, steelhead (migratory form of rainbow trout *Oncorhynchus mykiss*), sea lamprey, and walleye, as well as the quality (measured in average length) of brook trout and walleye. The working group chose these species because they were important to stakeholders, and they represented larger guilds of species found in the watershed. Some objectives required the use of constructed scales to relate the modeled outcomes to non-fish objectives (e.g., quality of life). The full list of performance measures for each means objective is given in Table 1.2. The working group also identified a number of process objectives including management by science, maximizing Tribal knowledge and perspective in the decision process, considering the seventh generation philosophy, accounting for unintended consequences, and considering the rights of the river itself (Figure 1.3).

Table 1.2. Consequence table with predicted outcomes for each management alternative of fish passage on the Boardman-Ottaway River at FishPass. Predicted outcomes associated with the 5 management alternatives are scaled from 0 to 1 (0 being worst and 1 being best). Bold numbers and blue filled cells indicate the best scoring management alternative for a given means objective and italics and magenta filled cells indicate the worst scoring alternative; orange filled cells are intermediate alternatives. W_{FO} = fundamental objective, W_{MO} = means objective. Description of management alternatives is given in Table 1.3. Min. = minimize, max. = maximize.

			Management Alternatives						
Fundamental Objective	**Means Objective**	**Measurable Attribute**	**Status Quo**	**Native Only**	**Low Steel**	**Low Salmon**	**Full Salmon**	**W_{FO}**	**W_{MO}**
Max. integrity and ecological health of river	Max. connectivity of Boardman-Ottaway River watershed	Length of river km available for migratory fishes	*0*	**1**	**1**	**1**	**1**	0.5	0.166
	Max. native species protection	Lake sturgeon abundance	*0*	**1**	**1**	**1**	**1**		0.055
		Brook trout abundance	0.711	**1**	0.444	0.444	*0*		0.055
		Walleye abundance	*0*	0.415	0.5	0.5	**1**		0.055
	Max. coldwater species protection	Chinook salmon abundance	*0*	*0*	*0*	0.533	**1**		0.083
		Steelhead abundance	*0*	*0*	0.442	0.442	**1**		0.083
	Min. ecological effects on stream banks	Constructed scale for level of foot traffic and trash	**1**	**1**	0.5	0.25	*0*		0.166
	Min. negative human effects on watershed as a result of tourism and development	Constructed scale for level of tourism and outdoor recreation (e.g., fishing)	**1**	0.75	0.50	0.25	*0*		0.166
	Min. Great Lakes contaminant introduction	Abundance of species that carry contaminant loads	**1**	**1**	**1**	0.467	*0*		0.166
Max. rightsholder, stakeholder, and public satisfaction	Max. sustainable catch and quality of brook trout (and other naturalized resident trout)	Brook trout abundance	0.711	**1**	0.444	0.444	*0*	0.5	0.1
		Brook trout quality	0.487	**1**	0.488	0.488	*0*		0.1
	1) Max. sustainable catch and quality of native species 2) Max. sustainable harvest (Tribal)	Walleye abundance	*0*	0.415	0.5	0.5	**1**		0.1
		Walleye quality	*0*	0.059	**1**	**1**	0.875		0.1
	Max. sustainable harvest of Pacific salmonids (recreational and commercial)	Pacific salmon abundance	*0*	*0*	0.32	0.47	**1**		0.2
	Min. user conflict	Constructed scale for level of conflict with user groups (boaters, anglers, etc.)	**1**	**1**	0.5	0.25	*0*		0.2
	Min. effect to resident homeowners	Constructed scale for level of effect to homeowners, quality of life, and protection of property rights	**1**	**1**	0.75	0.25	*0*		0.2

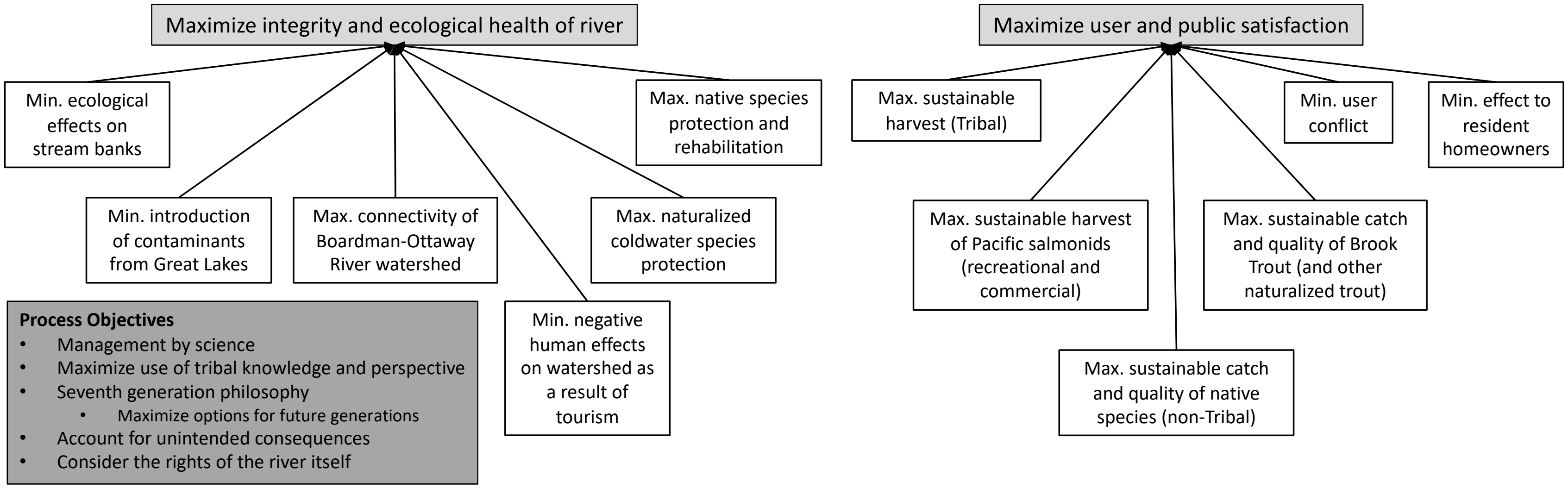

Figure 1.3. Objectives hierarchy showing the fundamental (light gray boxes), means (white boxes), and process objectives (dark gray box) for the decision analysis of the FishPass project, based on discussions with the structured decision making (SDM) working group during the first four virtual workshops (October 29, 2020 – April 8, 2021). The hierarchy represents the working group's input as well as refinement by the authors as part of the iterative SDM process. Min. = minimize, max. = maximize.

Alternatives

The MDNR, one of the decision makers for this project, requested that five management alternatives be included in the SDM evaluation (Table 1.3). Additional alternatives that were elicited from the working group were not alternatives that could be included in modeling. Examples of the types of additional alternatives suggested by stakeholders included the stocking of Atlantic salmon and Arctic grayling *Thymallus arcticus* into the watershed, and the construction of upstream barriers to allow Pacific salmonids partial access to the watershed. The implementation of these additional alternatives was highly uncertain (e.g., introducing new species) or improbable (e.g., building new barriers upstream), and therefore they were not included in modeling. The working group felt the five alternatives first suggested by the MDNR would provide measurably different outcomes. The five alternatives we evaluated for FishPass were: (1) no passage of any species (referred to as *status quo*); (2) allowed passage of only native Great Lakes basin species (referred to as *native only*); (3) allowed passage of 800 to 1000 steelhead and all native species (referred to as *low steel*); (4) allowed passage for 500 to 800 Chinook salmon, 800 to 1000 steelhead, and all native species (referred to as *low salmon*); (5) allowed passage for all species except aquatic invasive species (referred to as *full salmon*; Table 1.3).

As is common for many barrier removal projects, our case study included a great deal of conflict and differing opinions among stakeholders. Additional challenges were faced due to the Covid-19 pandemic, which forced us away from in-person workshops, instead requiring us to conduct a series of remote workshops. Ultimately, we were able to work through the first three steps of the SDM process (problem, objective, and alternatives) at which time the FishPass project was paused for litigation. The authors continued estimating consequences but determined constructed scales for some objectives and evaluated tradeoffs without input from the working group. Discussions with the working group during our SDM workshops were used to inform the tradeoffs step and evaluate the optimal decision under several different scenarios of preference.

Table 1.3. Set of management alternatives proposed by the Structured Decision-Making working group for fish passage on the Boardman-Ottaway River at FishPass.

Alternative	Action Type	Fish Passage Allowed
1–Status Quo	No passage	No passage, but FishPass facility is still constructed
2–Native Only	Native species passage	Allowed passage for only native Great Lakes basin species
3–Low Steel	Reduced salmonid passage	Allowed passage for 800 to 1000 steelhead and all native Great Lakes basin species
4–Low Salmon	Reduced salmonid passage	Allowed passage for 500 to 800 Chinook salmon, 800 to 1000 steelhead, and all native Great Lakes basin species
5–Full Salmon	Full salmonid passage	Full passage allowed for all species except aquatic invasive species

Consequences

The estimation of consequences involves predicting the relative achievement of each objective by each management alternative. Predictive models were developed for the objectives that had measurable attributes related to species' population changes, and constructed scales were developed to predict the effects of the management alternatives on social and economic objectives. We modeled six species: brook trout (resident or native trout), Chinook salmon (non-native semelparous Pacific salmonid), lake sturgeon (long-lived native species), steelhead (non-native iteroparous Pacific salmonid), sea lamprey (invasive species), and walleye (short-lived native species). Sea lamprey was included to evaluate the risk associated with the possibility of sea lamprey gaining access to habitat above the barrier but was not considered for passage among the management alternatives. Individual-based models (IBMs) were developed to predict each species' abundance, spatial distribution and density, and juvenile migration to West Grand Traverse Bay (Lake Michigan). Here, we give a brief overview of the IBMs; however, for more detail on the IBMs used for estimating the consequences of management alternatives see Chapter

2. The model outcomes, or consequences, were used to make tradeoffs among the different management alternatives and identify an optimal management action.

Population model.– We developed an IBM framework to evaluate the outcomes of the five fish passage alternatives on population abundance, length-at-age, and spatial distribution and density for six migratory species. In general, each of the species-specific IBMs began by initializing a simulated population; a pre-spawn mortality event then occurred, and spawners were allocated to stream reaches within the Boardman-Ottaway River watershed based on species-specific movement functions. Recruitment then occurred through species-specific Ricker stock-recruitment functions, after which a post-spawn mortality event occurred. Recruits hatched at spawning locations and moved downstream or stayed in stream reaches and grew until mature based on species life history. Simulations were run for 50 years and 1000 simulations for all species except lake sturgeon, which ran for 250 years due to their long life history (up to 154 years; Bruch et al. 2016). The IBM simulation and evaluation of results was performed using R (R Core Team 2023).

The IBMs were built in a hierarchy of complexity based on data availability, understanding of current species population statuses in the watershed, and the interests of stakeholders. Age- and sex-specific abundance were estimated for all species; spatial distribution and density were estimated for Chinook salmon, steelhead, and sea lamprey; and length-at-age were estimated for brook trout and walleye (see Table 1.2 for measurable attributes). Each species-specific IBM had unique variations. For example, the lake sturgeon model predicted age- and sex-specific abundance over 250 years from nine combinations of three initial population sizes and three demographic rates, and the walleye model included three distinct populations based on recent genetic work in the watershed (Gehri et al. 2021), with two Great Lakes walleye populations and one inland Boardman Lake walleye population.

The IBMs were single-species models but included inter-species interactions by varying demographic parameters among the management alternatives, based on literature review. We assumed that passage of Pacific salmonids would positively affect walleye abundance and length-at-age due to increased forage availability for walleye (Krueger et al. 2013). However, we assumed passage of Pacific salmonids would negatively affect brook trout abundance and length-at-age due to competition with juvenile Pacific salmonids (Fausch and White 1986; Peck 2001; Zorn et al. 2020) that restricted their populations towards the headwaters and tributaries (Janetski et al. 2011). Stochasticity was incorporated into the model by drawing demographic parameters (e.g., mortality rate) from assumed probability distribution functions across different iterations rather than assuming a single value.

Some of the ecological means objectives did not require modeling to estimate consequences. The objective for maximizing connectivity of the watershed was calculated as the amount of river kilometers available for fishes below the barrier and was 1.1 km for the *status quo* alternative and 131.7 km for all other alternatives. Some stakeholders expressed concern about the introduction of contaminants from Great Lakes migratory fishes to the Boardman-Ottaway River. To estimate contaminant introduction from Great Lakes species, we used the estimated Chinook salmon abundance as a proxy because they are considered a major contributor to contaminant transfer due to their semelparous life cycle (Janetski et al. 2012; Gerig et al. 2020). We assumed the other species would not greatly contribute to the transfer of contaminants.

Social and economic outcomes.– To incorporate social and economic considerations into the decision-making process, we used constructed scales that were tied to ecological outcomes. The objectives for minimizing ecological effects on the stream banks, minimizing negative human effects on the watershed as a result of tourism, minimizing user conflict, and minimizing the effect to riverfront resident homeowners were tied to fish passage with constructed scales from 1 (best) to 5 (worst) for each fish passage alternative. We created the scores for social and economic outcomes to approximate stakeholder concerns that were expressed during our workshops; however, we did not work through the consequences or tradeoffs step with the working group. These measures are provided as an example of how these objectives could be scored, and we expect that future applications of SDM for the Boardman-Ottaway River and other fish passage decisions would include discussions of these measures with the full working group.

The means objective to minimize ecological effects to the stream banks was measured in terms of the amount of foot traffic (a proxy for bank erosion) and trash left by those walking along stream banks. The means objective to minimize negative human effects to the watershed due to tourism was measured as the relative number of tourists, including all users of the watershed (e.g., fishers, kayakers, visitors of Traverse City but not specifically the river) and development of land in the watershed. Some stakeholders voiced concerns that increased connectivity and fish passage, as well as the FishPass site as an attraction, could increase tourism and development in Traverse City and reduce the amount of vacant land in the watershed. The means objective to minimize effects to riverfront resident homeowners was measured in the "quality of life" of riverfront homeowners and protection of their property rights. Stakeholders expressed concerns that passage of Pacific salmonids would reduce their quality of life, increase trespassing by anglers, and erode the sense of community by increasing the number of short-term rentals (e.g., AirBnB) and reducing the number of permanent residents. Finally, the means objective to minimize user conflict was an important objective for stakeholders. The two sources of user conflict that stakeholders were concerned about was among fishers, and between fishers and riverfront property owners. As more anglers become concentrated in a watershed, instances of arguing over fishing spots and trespassing on private property to access new areas may increase, whereas enjoyment of a fishing experience would probably decline. The fall Chinook salmon fishery is known to result in a high level of negative interactions among fishers, law enforcement, and riparian owners in Michigan (e.g., trespassing complaints; Lofton 2020a,b).

Tradeoffs

Due to the multi-objective nature of our case study and barrier removal problems more broadly, tradeoffs were considered among objectives to select an optimal management alternative. To make tradeoffs among the many objectives and identify an optimal alternative we arranged our modeling results (or consequences) into a consequence table with normalized scores ranging from 0 to 1 for each means objective (Table 1.2). Tradeoffs among objectives were made with multi-attribute tradeoff methods—specifically, the simple multi-attribute ranking technique, SMART—and additive utility value modeling (Equation 1.1) to calculate weighted scores for ranking each alternative (Keeney 1992; Gregory et al. 2012). The scores (U) were normalized from 0 to 1 for each objective and each of the *w* values represent the weight that stakeholders might place on a given objective (both fundamental and means). We assumed equal weighting between the two fundamental objectives (maximizing ecological health and maximizing user satisfaction) and among all of the means objectives within each fundamental objective. When a means objective had multiple performance metrics the objective weight was split equally among the performance metrics to create a weighted index (Gregory et al. 2012). The alternative with the greatest calculated expected utility value, E(U), was considered the optimal alternative for fish passage on the Boardman-Ottaway River.

Equation 1.1:

$$\begin{aligned} E(U) = w_{eco.\ health}\Big[&\big(w_{connectivity} * U_{connectivity}\big) + \big(w_{native\ spp} * U_{native\ spp}\big) \\ &+ \big(w_{coldwater\ spp} * U_{coldwater\ spp}\big) + \big(w_{stream\ banks} * U_{stream\ banks}\big) \\ &+ \big(w_{tourism} * U_{tourism}\big) + \big(w_{contaminants} * U_{contaminants}\big)\Big] \\ &+ w_{satisfaction}\Big[\big(w_{BKT\ catch} * U_{BKT\ catch}\big) \\ &+ \big(w_{native\ spp\ catch} * U_{native\ spp\ catch}\big) + \big(w_{salmon\ catch} * U_{salmon\ catch}\big) \\ &+ \big(w_{conflict} * U_{conflict}\big) + \big(w_{homeowner\ impact} * U_{homeowner\ impact}\big)\Big] \end{aligned}$$

Uncertainty and Sensitivity.– We were interested in evaluating how robust the decision was to uncertainty in the model results. There was uncertainty regarding demographic rates and environmental variables used in the IBM framework (see Chapter 2); however, we were only interested in the uncertainties that could influence which alternative was found to be optimal. Therefore, we varied the model results from the estimated median values to the 25th and 75th quantile values to evaluate the sensitivity of the decision with respect to the model results.

We used indifference curves (Conroy and Peterson 2013) to evaluate the sensitivity of the final decision to objective weights by varying the fundamental objective weights and recalculating the expected utility values to evaluate how sensitive the decision was to the weighting of objectives. Weights were not elicited directly from the working group during this case study, so we developed four scenarios to describe different weighting schemes and evaluated the sensitivity of the decision to these scenarios. The first scenario was the reference scenario with equal weights for fundamental and means objectives. The second scenario was a pro-salmon weighting scenario where weights for native species protection, contaminant introduction, brook trout catch, and native species catch were reduced to zero and the residual weight was distributed equally among the objectives for Pacific salmonid abundance. The weights for the other objectives were held at the values used for the reference scenario. The third scenario was the pro-trout scenario (e.g., resident trout such as brook trout) and the objective weights for lake sturgeon and walleye abundance, coldwater species protection, native species catch, and salmonid catch were reduced to zero and residual weight was distributed equally among the objectives for brook trout abundance and quality. The fourth scenario valued social objectives more than species abundance and quality. The objective weights were reduced to zero for all objectives except minimizing effects to stream banks, minimizing effects as a result of tourism, minimizing user conflict, and minimizing effects to resident homeowners and were distributed equally among these means objectives.

Results

Consequences

Population model.– Populations of all species increased as more individuals gained access to spawning and rearing habitat (see Chapter 2; Figure 1.4). Brook trout abundance declined with increasing passage of Pacific salmonids, and brook trout was predicted to be the dominant species for all alternatives except the *full salmon* alternative, in which steelhead became the dominant species (Figure 1.4). Walleye abundances increased for all passage scenarios; however, the increase was greater for Great Lakes populations than for the inland Boardman Lake population (Figure 1.4). For lake sturgeon, population abundance for the *status quo* was assumed to be 0 and for all other alternatives was predicted to between 0–856 in year-250, depending on demographic parameters. Estimated lake sturgeon abundance from “high growth, low initial abundance” lake sturgeon model scenario (N = 352 at year 250) was used in the consequences table. This was the lowest estimated abundance for all species considered in this study.

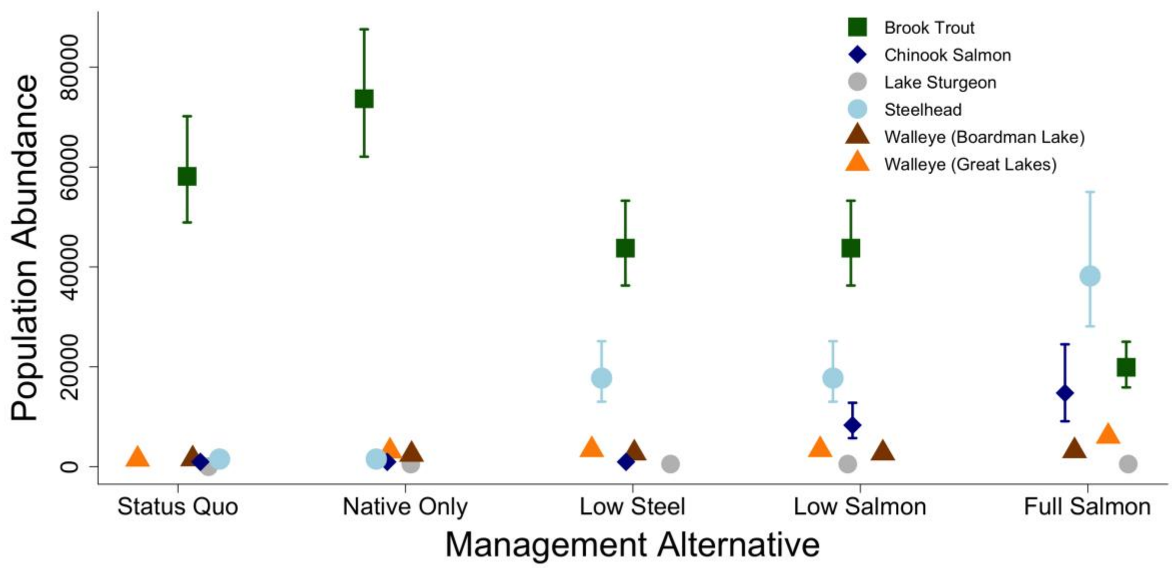

Figure 1.4. Predicted median population abundance and interquartile range in the last five years of simulation of lake sturgeon *Acipenser fulvescens*, Chinook salmon *Oncorhynchus tshawytscha*, steelhead *Oncorhynchus mykiss*, walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis* for each management alternative at FishPass on the Boardman-Ottaway River, Traverse City, Michigan, USA. Points are jittered along the x-axis.

The predicted length-at-age for brook trout and walleye were similar to the trends in abundance. Walleye length-at-age was predicted to be greatest for the alternatives that allowed Pacific salmonid passage, and lowest for the *native only* and *status quo* alternatives. For brook trout the opposite was true, length-at-age declined with increasing passage of Pacific salmonids. The highest brook trout length-at-age was predicted for the *native only* alternative and the lowest for the *full salmon* alternative (see Chapter 2).

Social and economic outcomes.– Based on discussions with stakeholders during the SDM workshops, the *status quo* was assumed to have the least amount of effects to stream banks, tourism and development, homeowners' quality of life, and user conflict, and the *full salmon* alternative was assumed to have the greatest effects (Table 1.4). The *status quo* performed best amongst all social and economic outcomes, and the total score of the constructed scale means objectives was equal to 4, the lowest possible score out of 20 (Table 1.4).

The *native only* alternative performed similar to the *status quo* except for the small increase in angling for native species and tourism as a result of a public perception of an improved watershed (total score = 5; Table 1.4). For the *native only* alternative, we assumed foot traffic and trash would not increase because many native fishes are protected by fishing closures while in the river (e.g., walleye) or are not highly sought after by anglers (e.g., white sucker). With *native only* passage, we assumed tourism would increase to the FishPass site and there would be a slight increase in the amount of development and outdoor recreation in the watershed. Novel fisheries of native fishes would probably attract new anglers to the watershed; however, many of the native migratory species in the watershed would either be protected by regulations from fishing pressure while in the river or are not highly desirable to fishers. Our constructed scale for minimizing the effect to resident homeowners assumed the lowest effect for the *status quo* and *native only* alternatives, and we assumed homeowners were unlikely to experience novel effects due to increased fishing pressure. For minimizing user conflict—for reasons previously described—we assumed that increases to fishing pressure and number of anglers would not occur under the *status quo* and *native only* alternatives.

Table 1.4. Scores for the means objectives that used constructed scales and were not modeled or tied to the modeling results. The best score is 1 and the worst score is 5 because the goal was to minimize each of the objectives.

Means Objective	Status Quo	Native Only	Low Steel	Low Salmon	Full Salmon
Minimize effects to stream banks	1	1	3	4	5
Minimize effect of tourism and development	1	2	3	4	5
Minimize effect to homeowners and quality of life	1	1	2	4	5
Minimize user conflict	1	1	3	4	5

For the alternatives allowing Pacific salmonid passage, we assumed increases in effects to stream banks, tourism, outdoor recreation, and user conflict. Additionally, we assumed that salmon and steelhead fishers would consider buying riverfront property as connectivity and number of salmonids passed increased, thereby increasing the amount of development in the watershed. For alternatives that included passage of Chinook salmon we assumed the highest effects. The *low steel* alternative performed moderately for all social and economic outcomes (total score = 11; Table 1.4). The upper Boardman-Ottaway River is currently closed to fishing during much of the steelhead run (October 1–last Saturday in April) and therefore, a novel steelhead fishery would probably result in increases in tourism and vacationers for a longer period of the year (an additional 7 months); however, much of the season is in the winter, which results in fewer fishers and less fishing pressure than the fall Chinook fishery. Additionally, due to the reduced number of fish passed we assumed user conflict and effect to homeowners would only moderately increase because nearby rivers (e.g., Betsie River) receive larger runs of steelhead that would attract more anglers.

The *low salmon* alternative performed worse than the *low steel* alternative for all objectives, because the Chinook salmon run attracts large numbers of anglers and observers (total score = 16; Table 1.4). The *full salmon* alternative performed worse than the *low salmon* alternative in terms of effect to homeowners, effects to stream banks, and user conflict because increasing fish passage would probably result in greater fishing pressure and thus, greater effects to resident homeowners, stream banks, and increases in user conflict (total score = 20; Table 1.4).

Tradeoffs

The *native only* alternative scored the highest (expected utility value, E(U) = 0.70; Figure 1.5) and the *full salmon* alternative scored the lowest (E(U) = 0.42) when all objectives were weighted equally. The *low steel* alternative was the second highest scoring (E(U) = 0.60), the *low salmon* alternative ranked third, and the *status quo* ranked fourth. When the expected utility value scores were decomposed by fundamental objective (Figure 1.5), maximizing integrity and ecological health of the river scored higher than maximizing user and public satisfaction for all alternatives. The *native only* alternative scored highest amongst all alternatives for maximizing ecological health (E(U)= 0.38) and user satisfaction (E(U)= 0.32). The lowest scoring alternative for both fundamental objectives was the *full salmon* alternative, and the *low steel* alternative scored higher than the *status quo* for both fundamental objectives (Figure 1.5).

The means objectives that were most influential on the fundamental objective for maximizing integrity and ecological health of the river were minimizing ecological effects to stream banks, minimizing effects of tourism, and minimizing the introduction of contaminants from the Great Lakes (Figure 1.5). The means objectives for maximizing connectivity and native species protection were similar across all alternatives, except the *status quo*. The *low salmon* and *full salmon* alternatives scored high on coldwater species protection; however, this was not enough to offset the losses in utility from the effects to stream banks, tourism, and contaminants (Figure 1.5).

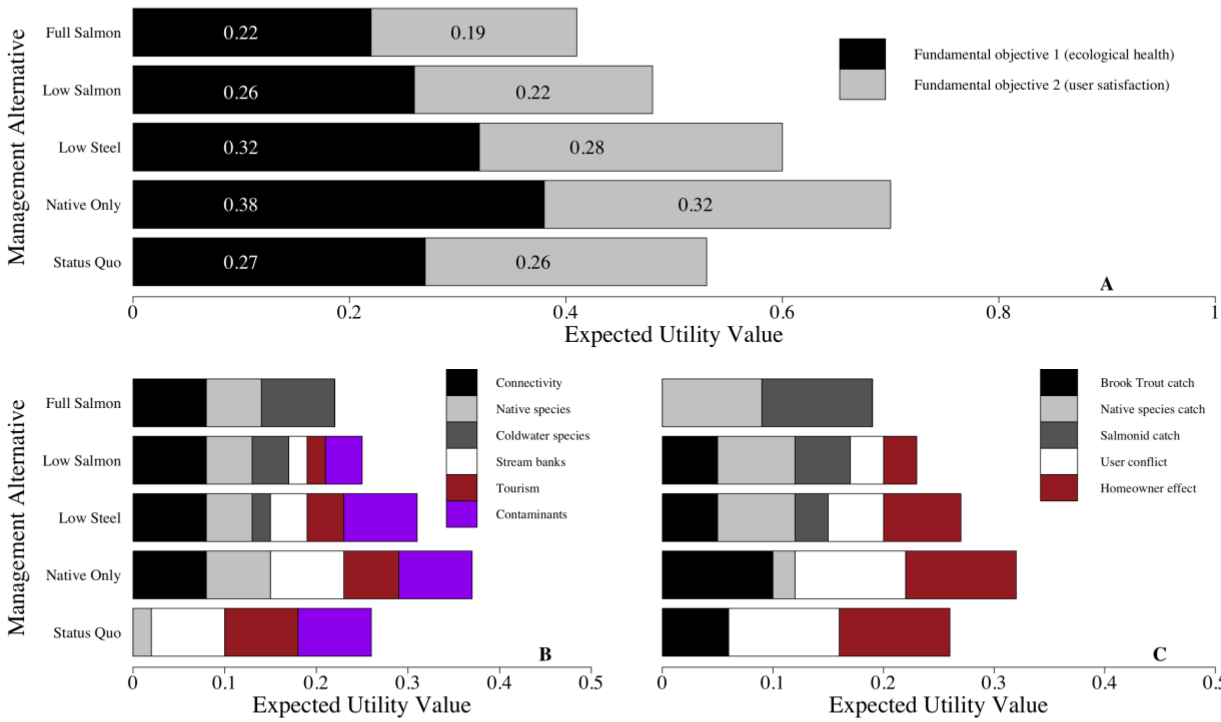

Figure 1.5. Expected utility values for the five management alternatives for fish passage on the Boardman-Ottaway River. The top panel (A) shows the two fundamental objectives. Fundamental objective 1 is to maximize ecological health and integrity of the Boardman-Ottaway River watershed and fundamental objective 2 is to maximize user and public satisfaction. Expected utility value scores comprise the weighted normalized score for each mean and fundamental objective. A score of 1 is perfect achievement of all objectives and 0 is achieving none of the objectives. Intermediate scores represent the tradeoffs among objectives. Values inside the boxes are the score for the fundamental objectives. The total expected utility value for each alternative is the sum of the two values. The bottom panels (B and C) decompose the means objectives expected utility value for both fundamental objectives and all five management alternatives. B = fundamental objective to maximize ecological health of the river, C = fundamental objective to maximize user and public satisfaction.

The means objectives that were most influential on the fundamental objective for maximizing user satisfaction were maximizing sustainable brook trout catch, minimizing user conflict, and minimizing effects to homeowners, and the *native only* alternative scored highest on all three (Figure 1.5). The *full salmon* alternative scored highest for maximizing sustainable native species catch and maximizing sustainable salmonid catch but had a score of zero for all other means objectives. The increases in utility of salmonid catch and native species catch for the *full salmon* alternative were not substantial enough to offset the losses from the objectives for brook trout catch, user conflict, and effects to homeowners (Figure 1.5).

Uncertainty and Sensitivity

The uncertainty among the 25th and 75th quantiles did not affect the choice of optimal alternative; all other rankings were the same as when the median values were used. Therefore, the decision is robust to uncertainty so long as the trends and relative difference among the alternatives were accurately captured.

When weights were varied for the fundamental and means objectives, the optimal alternative changed among the four weighting scenarios we developed. If weights were equally distributed among means objectives, the optimal alternative remained the *native only* alternative for all weights of fundamental objective 1 (Figure 1.6). If the means objectives weight was set for the pro-salmon scenario the optimal alternative was the *full salmon* alternative and all other alternatives ranked about the same across the fundamental objective weights. For the pro-trout scenario, the optimal alternative was the *native only* alternative, but the distance between the alternatives was much greater, meaning the *native only* alternative was strongly preferred over the other alternatives. For the pro-trout scenario, the second most optimal alternative was the *status quo*. Finally, for the pro-social scenario the optimal alternative was the *status quo* if the weight of fundamental objective 1 was greater than 0.2. If the weight was reduced below 0.2, meaning a stakeholder valued social objectives more than the integrity and ecological health of the river, the decision maker would become indifferent to the decision between the *status quo* and *native only* alternatives (Figure 1.6).

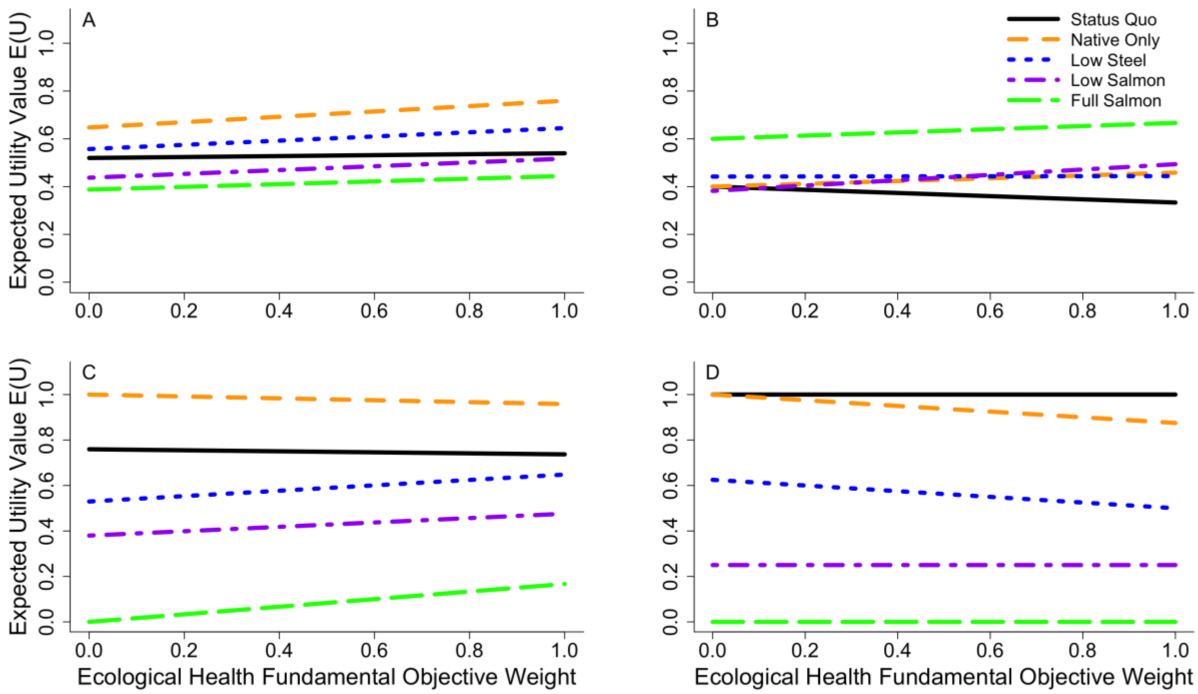

Figure 1.6. Indifference curves for four different mean objectives weighting scenarios showing how the expected utility value and optimal management alternative changes with the weights of fundamental objectives. A = equal weighting of all means objectives, B = increased weight for pro-salmon objectives, C = increased weight for pro-trout objectives, D = increased weight for pro-social objectives.

Discussion

Our research showed that selective connectivity and reduced fish passage regimes performed better than allowing full fish passage, suggesting that, at least in the Great Lakes, full connectivity may not always be the best alternative when considering barrier removals and renovations. In this study, a selective fish passage management alternative that allowed a reduced number of individuals upstream was preferable to alternatives that allowed a larger number of species and a higher abundance of individuals upstream when all objectives were equally weighted. In fact, the alternative with the highest degree of connectivity in our study ranked lowest among possible management alternatives. The *status quo*, which did not allow fish passage, was only slightly less optimal than the *low salmon* alternative which included passage of both Chinook salmon and steelhead. The most optimal alternative that included Pacific salmonid passage was the *low steel* alternative which allowed passage of a reduced number of steelhead while excluding Chinook salmon.

The optimal management alternative varied among the four stakeholder weighting scenarios. Although we attempted to capture the variety of stakeholder values through the four objective weighting scenarios we created, ultimately these scenarios represent extremes, and the truth probably lies somewhere in the middle. Science cannot tell us the correct answer; although we can predict the ecological effects of increases to connectivity, stakeholder values may shift and the optimal alternative may change (Tyre and Michaels 2011). However, this study did shed some light on management alternatives that stakeholders are likely to find preferable. First, the *status quo* alternative was identified as optimal for stakeholders valuing social effects most; however, passage of fishes is mandated by law (MCL 324.48301 *et seq.*) and so although we used this alternative as the baseline, in reality it is not a viable management option once the FishPass project has been completed. Second, the full passage of Pacific salmonids (*full salmon*) is only optimal for stakeholders wishing to fish for these species, otherwise the *low steel* alternative was the most optimal management alternative that allowed passage of Pacific salmonids. Finally, the passage of native species only (*native only*) was the most optimal management alternative across the four objective weighting scenarios. Unless the objective weights were heavily skewed towards Pacific salmonid objectives, salmon passage alternatives performed worse than the *native only* alternative. When ranking only viable management alternatives, *native only* was the most optimal, *low steel* was ranked second, *low salmon* ranked third, and *full salmon* ranked last.

The ecological consequences of enhanced connectivity are often at the forefront of barrier remediation decisions; however, decisions affecting stream connectivity should ideally consider the economic and social consequences, such as the cost of treating additional streams for invasive Sea Lamprey (McLaughlin et al. 2013; Jensen and Jones 2018) and the effect to recreational opportunities and stakeholder behavior (Fox et al. 2016). Currently, the upper Boardman-Ottaway River is closed to fishing from October to the end of April; however, salmon streams in Michigan are typically open all year to fishing. The effects of a longer fishing season could be positive economically with increased visitors in otherwise slow seasons; however, negative social effects to stakeholders (e.g., decreased user satisfaction) could offset these economic positives for some stakeholders. Melstrom et al. (2015) reported that increases to brook trout and walleye abundances returned the most economic value to Michigan's stream fisheries over brown trout, smallmouth bass, and panfishes; however, the study did not include Pacific salmonids. They further noted that of all the species considered in the study, brook trout and walleye had the lowest current biomass. There are several rivers nearby that have large migratory runs of salmon (e.g., the Betsie River, Platte River, Manistee River, Bear Creek, and Bear River); therefore, the Boardman-Ottaway River and FishPass project could provide a unique native species-focused fishery should the passage of Pacific salmonids be blocked. A unique and novel native-focused fishery could have substantial economic implications by creating an above average fishery for native species and drawing new anglers to the area. However, the economic considerations for the Boardman-Ottaway River and Union Street Dam might be comparatively less than barrier removals that also make tradeoffs among hydropower or public safety (Zheng and Hobbs 2013; Song et al. 2019). Our working group focused on economic outcomes related to increased tourism and recreational activities in the watershed; however, there remain substantial uncertainties in the economic outcomes related to selective fish passage. Ultimately the use of selective fish passage on a larger scale could allow managers in any system to fine-tune passage of desired species to provide the best outcomes socially, economically, and environmentally.

Selective fish passage facilities offer fisheries managers a new opportunity to enhance connectivity and restore watersheds while having control over the species which will inhabit the new system. The choice of species that ultimately are passed should ideally be informed by local stakeholders' values and objectives, and selective fish passage would give fisheries managers a lever they could pull if stakeholder objectives vary spatially. The species that were identified as desirable for passage by our working group would probably be similar for most barrier removal and renovation projects across the Great Lakes; however, regional variation would exist especially if they were in southern Great Lakes streams that lack resident brook trout populations. The tradeoff of potential increases in native populations versus introduced and invasive species is not unique to the Great Lakes and will be present in most locations; this tradeoff has been noted in the western United States (Rahel 2013; Rahel and McLaughlin 2020), the Mississippi River watershed (Cooper et al. 2021), and internationally in the United Kingdom (Kerr et al. 2021) and Australia (Stuart et al. 2006). While ecological objectives for barrier remediation projects are typically similar (i.e., pass desirable fish while blocking invasive and undesirable fishes), the social and economic values and objectives of stakeholders are likely to be more variable from case to case.

To incorporate social and economic objectives of stakeholders, we used constructed scales that were tied to ecological outcomes. Novel ways to incorporate social and economic considerations into the predictive modeling could be a future direction of this research but did not play an integral role in the initial development of the model. Discrete choice modeling (Train 2003) of stakeholder survey data could be particularly helpful to identify social and economic drivers of stakeholder preference and behavior. Although we were not able to measure some of the social and economic objectives as well as fish abundance because of the issues around virtual workshops and contention surrounding the project, we have identified needs for consideration when novel fish passage or fish community objectives are being considered and encourage using SDM for decisions which affect stream connectivity and consist of myriad economic and social objectives.

Unfortunately, our case study represents a decision in which stakeholder contention could not be overcome in a virtual setting. Our experiences were not unique; other similar work on barrier removal projects has been contentious and not able to reach the final stages (McLaughlin et al. 2013). The FishPass project led to litigation due to stakeholder objections, which has been typical of barrier removal projects throughout the United States. Several additional factors contributed to the difficulties we experienced; most importantly, we were not able to hold in-person meetings due to the Covid-19 pandemic, which led to an inability for the working group to build a sense of cohesion and trust in one another (Muir et al. 2023). In our experience, remote meetings are not a good way to conduct SDM workshops except in special circumstances such as a global pandemic or when there is already proper group cohesion, rather than when issues are still contentious.

Although we faced many challenges, there are many benefits to using a SDM process for decisions that result in changes to stream connectivity. First, SDM requires estimating the consequences of each alternative, which involves developing a model of the system. Developing a model increases fundamental understanding of the system and elucidates areas of critical uncertainty. These models are also research products themselves and can be used repeatedly to inform decisions or be improved upon or adopted by others. The individual-based model framework that we developed for this project could be applied to barrier removal projects in other systems and for different species. It could also be modified to serve several different uses, from hypothesis testing to further evaluating the effect of habitat or life history dynamics on population- or individual-level phenomena. Second, SDM engages stakeholders and is value-focused, meaning stakeholder values and objectives are explicitly stated and drive the rest of the decision process (Keeney 1992). SDM informs decision makers on stakeholders' preference for a range of management alternatives allowing them to better understand the consequences of their potential management actions and make robust management decisions that are likely to meet stakeholder objectives. We found that *more fish* is not always the best management alternative and when stakeholder objectives are explicitly stated and considered in the decision process, oftentimes selective fish passage alternatives outperformed the full passage alternative. Thirdly, SDM increases transparency in the decision-making process. For barrier removal projects, which have been notoriously contentious (Fox et al. 2016; Brummer et al. 2017), the inclusion of members from non-technical and non-fisheries expert groups can be especially beneficial in lending transparency to the decision making process. Our working group contained representatives from several different stakeholder groups including non-expert members of the general public. Although we faced challenges, we were able to lend transparency to the decision process by engaging with our working group in five workshops and working through the first three steps of the PrOACT framework. Finally, the iterative nature of SDM allows for revisiting and updating steps in the process, or moving to an adaptive management framework where management actions are aimed at reducing uncertainty in the decision process (Walters 1986). Silva et al. (2018) suggested that adaptive management will be required in future efforts to design selective fishways and minimize effects of passage on fishes. We add that adaptive management will also be a critical component of fine-tuning predictive models and meeting future stakeholder objectives related to fish passage. Future efforts to update the predictive model and validate it via monitoring, as well as reconvening the SDM working group to revisit the PrOACT framework and elicit additional stakeholder input for social objective scoring could be especially useful to improve this work. The MDNR has committed to a 10-year moratorium on passage of non-native species; therefore, the optimal management alternative we identified will be implemented for 10 years. Ideally, this SDM process would be repeated and models rerun with updated basin-specific data to evaluate the optimal alternative after the moratorium.

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CHAPTER 2: PREDICTING THE RESPONSE OF FISH POPULATIONS TO CHANGES IN RIVER CONNECTIVITY USING INDIVIDUAL-BASED MODELS

Abstract

Barrier removal and remediation restores physical stream processes and improves accessibility of critical habitats to migratory fishes. Although increasing connectivity is beneficial to stream systems and migratory fishes, barrier removal and remediation decisions often have multiple, competing objectives that can lead to tradeoffs between the benefits to desirable species and the potential increased production of undesirable and invasive species, alongside myriad other concerns. Few studies have predicted the response of migratory fish populations to enhanced fish passage prior to barrier removal or remediation, despite this being a critical step in the decision-making process. We developed an individual-based model framework to forecast the response of migratory fishes to changes in connectivity and applied the framework to predict the production of six species under various fish passage scenarios using the Boardman-Ottaway River FishPass project in Michigan as a case study. Population response to barrier removal was species-specific and varied based on initial population size and distribution within the watershed, the number of fish passed upstream, and species life history traits. Species that were found only below the barrier prior to removal benefitted most; non-native species were found to have greater production potential under full passage scenarios than native Great Lakes basin species. With increasing passage of non-native Pacific salmonids, steelhead *Oncorhynchus mykiss* (migratory form of rainbow trout) surpassed brook trout *Salvelinus fontinalis* as the system's dominant species. Our results will help inform decision-makers on management alternatives for fish passage on the Boardman-Ottaway River. Our model framework can be modified and updated as new monitoring data become available and applied to additional river systems as more barrier removal projects are conducted in the future.

Introduction

Barrier removal is increasingly being used to restore river connectivity and rehabilitate migratory fish populations (Pess et al. 2014; O'Connor et al. 2015; Watson et al. 2018). Enhancing connectivity restores physical stream processes and conditions (i.e., temperature and instream flow) and increases access to critical life-stage habitats for migratory fishes as well as energy and nutrient transport from lake and marine systems to stream habitats (Flecker et al. 2010; Childress and McIntyre 2016). Barrier removal and remediation have proven to be viable techniques for restoring stream fish populations (Birnie-Gauvin et al. 2018; Brenkman et al. 2019); even removals of small dams have positively affected lotic fish communities by increasing species richness and occurrence of intolerant species, decreasing occurrence of non-desirable species, and promoting recolonization of upstream habitats (Catalano et al. 2007).

Most previous research surrounding barrier removals and associated effects on fish populations has assessed changes in fish assemblages (Evans et al. 2015; Watson et al. 2018; Ding et al. 2019). These studies are usually field-based and occur post removal. However, making an informed decision on an action for dam removal or remediation for enhanced fish passage requires understanding how fish populations may respond to changes in connectivity prior to implementing the decision. Although increasing connectivity of a stream system through barrier removal or remediation generally benefits migratory fish populations (Brenkman et al. 2019; Whittum et al. 2023), standardized methods to predict changes in fish movement, genetic consequences, and population size or productivity to our knowledge have not been developed or are not widely available, especially when considering implementation of novel techniques for selective fish passage (Foley et al. 2017; Hayes et al. 2023). Additionally, quantifying the relationship between environmental variables and fish populations has been notoriously difficult (Rose 2000; Hayes et al. 2009).

Although rehabilitation of migratory fish populations is a common impetus for barrier removal, not all species will respond similarly to barrier removal due to variations in life history, initial abundances and distributions, or social and economic considerations (e.g., greater upstream abundance leads to higher fishing effort that hinders production of the target species). To forecast the response of fish populations to changes in connectivity from barrier removal or remediation, information on how demographics, dynamics, and life history of populations will respond to accessible new habitat and interactions with the biota in the newly accessible habitat are critical. Furthermore, variation at the individual level increases uncertainty in forecasting production, because individual-level processes (e.g., growth, reproduction, movement, and behavior) influence higher levels of complexity, such as population- and community-level phenomena (Huston et al. 1988). A simple and flexible modeling framework that can capture among- and within-species variation would be useful for forecasting potential consequences of changes to connectivity.

Individual-based models (IBMs) are a bottom-up approach to modeling, where simple governing rules, or submodels, are assigned to individuals and give rise to emergent system complexity because of individual stochasticity (Huston et al. 1988; Railsback and Harvey 2020). Individual-based models have been used in fisheries science since the late 1980s to link fish population dynamics to changes in environmental factors (Jager and Deangelis 2018). Individual-based model frameworks are beneficial because they consider variation at the individual level, can easily incorporate additional complexities in life history, behavior, or spatial and temporal variation in habitat conditions, and represent a natural feedback between the individual- and population-levels (Railsback and Harvey 2020). Further, IBMs have been identified as the best approach for addressing questions that link fish populations to changes in habitat (Rose 2000; Hayes et al. 2009). These attributes make IBMs an appropriate method to simulate anticipated changes in fish production resulting from barrier renovation, although many other methods also could be used (e.g., matrix models). We opted for an individual-based approach because we believed it was complex enough to handle individual variability, species interactions, and spatial variation while being conceptually straightforward enough to be understood and accepted by stakeholders that do not have extensive statistical training.

Individual-based models have previously been applied to evaluate the effects of temperature and flow regime on riverine fish populations (Van Winkle et al. 1998; Tyler and Rutherford 2007), ecological conditions during migration and individual fish fitness (Snyder et al. 2019), barrier density (Harvey and Railsback 2012), species interactions (Clark and Rose 1997), and genetic effects (Schueller and Hayes 2011; Lin et al. 2017). However, few examples exist of IBMs applied to assess changes in fish populations due to enhanced connectivity or from selective fish passage. An IBM was used to compare reconnection options and evaluate the effect of up- and downstream passage and river segment length on meta-population dynamics of white sturgeon *Acipenser transmontanus* on the Snake River, Idaho, USA (Jager et al. 2000; Jager 2006b). The same IBM was used to assess the demographic and genetic effects of translocation efforts on white sturgeon metapopulations (Jager 2006a); a similar methodology was employed to evaluate translocation options for humpback chub *Gila cypha* on the Little Colorado River, USA (Pine et al. 2013). Harvey and Railsback (2012) evaluated the effects of barrier density on population demographics and population stability and found that overall abundance and biomass declined at high and intermediate barrier densities.

We developed an IBM framework to evaluate potential consequences of selective connectivity (i.e., the passage of desirable species and exclusion of non-desirable species) on migratory fish populations, passage regimes that are most beneficial to certain life-history traits, and whether total barrier removal (i.e., complete connectivity) was always the best management action, which was found to not be the case by Harvey and Railsback (2021). Our model framework was constructed as part of a larger structured decision making (SDM; Hammond et al. 1999; Gregory et al. 2012) process that required forecasting possible intended and unintended consequences of enhancing connectivity to inform decision making. Understanding the population-level benefits and consequences of enhancing connectivity prior to enacting a decision is important for the management of fish stocks, yet few IBMs have been designed with such questions in mind (Jager and Deangelis 2018). Additionally, barrier removal and renovation decisions are often socio-economically complex and require making tradeoffs among multiple objectives (Zheng et al. 2009; Zheng and Hobbs 2013; Song et al. 2019) and differing stakeholder values (Fox et al. 2016; Brummer et al. 2017). Because stakeholders value some species (e.g., native species) over other species (e.g., non-native species), 'more fish' is not necessarily the best option. In the Great Lakes, non-native Pacific salmonids (*Oncorhynchus spp.*) and invasive sea lamprey (*Petromyzon marinus*) can have significant socio-economic consequences. Therefore, a modeling framework that can be used to forecast the ecological responses to changes in connectivity before a removal or renovation occurs, while making the decision making process transparent and defensible, is beneficial to decision makers despite being highly uncertain (Starfield 1997). None of the aforementioned IBMs were designed to evaluate selective connectivity regimes for a single barrier renovation, but previous research shows that IBMs are an appropriate framework to explore the response of fish populations to changes in connectivity (Hayes et al. 2009, 2023). Our IBM framework was designed to be iteratively improved upon, through an adaptive management (Walters 1986) process, to resolve important uncertainties (Runge et al. 2011) inhibiting decision making for changes in connectivity. It is important to note that our model framework was individual-based in that we explicitly modeled individuals; however, our model also employed traditional population-based approaches to simulate reproduction, growth, and movement, while incorporating individual variation thereby creating a hybrid of traditional population-based approaches and IBM. Our model framework can be updated to incorporate more individual-based elements as data availability improves. This process will require targeted research to better estimate life history parameters, understand patterns of movement, and resolve other important uncertainties currently inhibiting the development of a fully bottom-up approach that includes behavior and the ability of individuals to learn from and adapt to their environment.

We used the Great Lakes Fishery Commission (GLFC) FishPass project on the Boardman-Ottaway River, Michigan as a case study to evaluate the effectiveness of IBMs for predicting fish population response and changes to productivity under various selective fish passage regimes (Zielinski et al. 2020). We used the developed IBM framework to predict novel fish production for six individual migratory species under five different fish passage scenarios. First, we developed an IBM to predict lake sturgeon *Acipenser fulvescens* abundance and then built upon that framework to include additional layers of complexity in the submodels for individual movement, reproduction, mortality, growth, and consideration of genetically-distinct populations for several other species to create an IBM hierarchy (Appendix: Figure A2.1). Our research had three objectives: (1) develop a flexible modeling framework that can serve as a building block for others to iteratively improve upon and fine-tune in an adaptive management

(Walters 1986) process, (2) forecast the response of riverine populations of brook trout, Chinook salmon *Oncorhynchus tshawytscha*, lake sturgeon, steelhead (migratory form of rainbow trout) *Oncorhynchus mykiss*, sea lamprey and walleye *Sander vitreus* under five different management scenarios, and (3) evaluate the results of the IBMs for each species and management scenario to inform fisheries managers in determining a selective passage regime once the FishPass project is completed.

Methods

Study Area

The Boardman-Ottaway River is located in the northwest lower peninsula of Michigan, USA (Figure 2.1). The watershed drains 743 km^2 of land in Grand Traverse and Kalkaska counties (Kalish et al. 2018). The river flows into West Grand Traverse Bay (Lake Michigan) in Traverse City, MI. Since 2012, three major dams (Boardman, Brown Bridge, and Sabin dams) have been removed from the Boardman-Ottaway River. The only remaining major dam on the mainstem of the river is Union Street Dam, which was constructed in 1867 and is located approximately 1.8 km upstream from West Grand Traverse Bay. The complete removal of Union Street Dam is not currently being considered because it operates as a barrier to invasive species, namely sea lamprey, and maintains water level in Boardman Lake (Kalish et al. 2018).

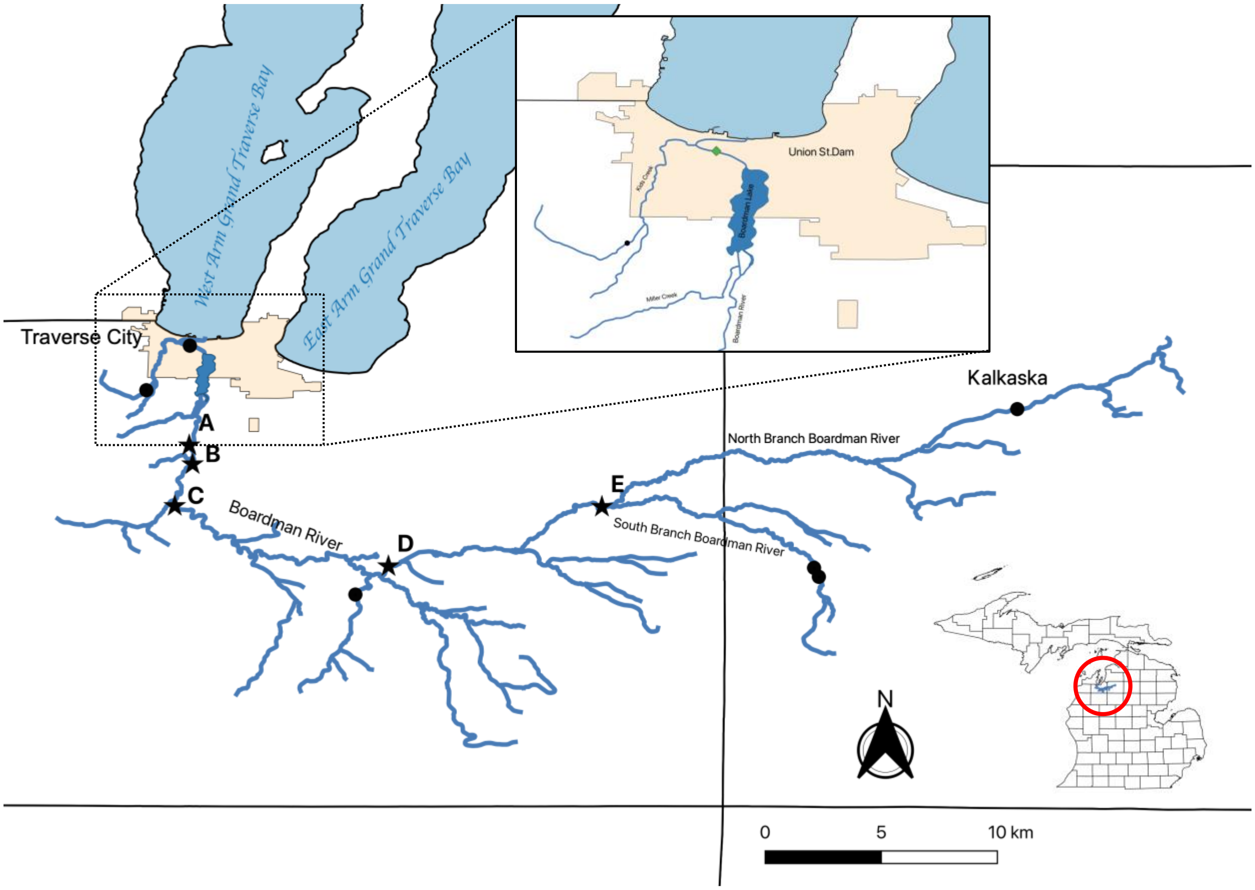

Figure 2.1. The Boardman-Ottaway River watershed and Grand Traverse Bays (Lake Michigan). Map inset showing the lower Boardman-Ottaway River and Traverse City (filled in beige), with green diamond at location of Union Street Dam. Black dots are dams currently in watershed with >6 ft of height. Stars and letters (A–E) indicate points of interest within watershed; A= former Sabin Dam site; B= former Boardman Dam site; C= Beitner Rd.; D= former Brown Bridge Dam site; E= the Forks.

Case Study

The Union Street Dam is the site for the GLFC FishPass project, which aims to develop a bi-directional, selective fish passage facility to allow passage of desired species (e.g., lake sturgeon) while excluding non-desired species (e.g., sea lamprey; Zielinski et al. 2020). FishPass is coordinated by the GLFC, which is a binational organization established under the Canadian/U.S. Convention on Great Lakes Fisheries to coordinate fisheries research, control Sea Lamprey, and facilitate cooperative fishery management in the Great Lakes. Access for many migratory species has been blocked to habitats above the former site of Sabin Dam for over 150 years, and there is substantial uncertainty among stakeholders and rightsholders (hereafter "stakeholders," but we recognize some groups are rightsholders, including Indigenous communities and governments) regarding how fish populations will respond to selective passage at the Union Street Dam. Additionally, values vary among stakeholders and their concerns and desires regarding enhanced connectivity should ideally be considered (Fox et al. 2016; Brummer et al. 2017). Therefore, a thorough evaluation of the ecological, social, and economic consequences and tradeoffs of enhanced connectivity was necessary to facilitate an informed decision-making process and measure success of the FishPass project. The FishPass facility provides an ideal case for testing predictive models because selective fish passage will allow for the evaluation of model predictions in a real-world system.

We held SDM workshops with local and regional stakeholders to elicit objectives, develop management alternatives (or scenarios), and identify performance metrics of objectives for evaluating and measuring success. The results from the SDM workshops informed the species and framework that were used for the modeling presented herein. The six species included for modeling were representative of six larger guilds of fishes: brook trout (resident or native trout), Chinook salmon (non-native semelparous Pacific salmonid), lake sturgeon (long-lived native species), steelhead (non-native iteroparous Pacific salmonid), sea lamprey (invasive species), and walleye (short-lived native species). Stakeholder objectives for fish passage decisions on the Boardman-Ottaway River included satisfaction for anglers targeting native and non-native species, as well as ecological integrity of the Boardman-Ottaway River. The group measured achievement of these objectives in terms of population abundance and spatial distribution for all six species, as well as a quality fishing experience (e.g., median fish length) of the five species that were targeted for angling.

Five different management scenarios were evaluated for FishPass: (1) no passage of any species (referred to as status quo); (2) allowed passage of only native Great Lakes basin species (referred to as native only); (3) allowed passage for 800 to 1000 steelhead and all native species (referred to as low steel); (4) allowed passage for 500 to 800 Chinook salmon, 800 to 1000 steelhead, and all native species (referred to as low salmon); (5) allowed passage for all species except aquatic invasive species (referred to as full salmon). Allowable passage for each of the scenarios was identified by the Michigan Department of Natural Resources (MDNR), which holds joint decision-making authority for fish passage on the Boardman-Ottaway River, along with the Grand Traverse Band of Ottawa and Chippewa Indians. Management scenarios that were assumed to have no effect on a species (e.g., no passage of the species) were not modeled (Table 2.1).

Table 2.1. Selective fish passage management scenarios for six fish species in the Boardman-Ottaway River. Management scenarios modeled for each species are shown with an X. The six species modeled were: lake sturgeon *Acipenser fulvescens*, sea lamprey *Petromyzon marinus*, steelhead *Oncorhynchus mykiss* (RBT), Chinook salmon *Oncorhynchus tshawytscha* (CHS), walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis*.

Fish Passage Scenario	Lake sturgeon	Sea lamprey	Steelhead	Chinook salmon	Walleye	Brook trout
Status Quo: No Passage		X	X	X	X	X
Native Only: Great Lakes Natives Only	X*				X	X
Low Steel: Natives + 800–1000 RBT	X*		X			
Low Salmon: Natives + 800–1000 RBT + 500–800 CHS	X*			X	X	X
Full Salmon: Full passage for all species except invasives	X*	X^	X	X	X	X

* Includes 9 parameter combinations of intrinsic growth rate, r, and initial abundance, N_1.
^ Includes reduced and full passage scenarios to make predictions for possible consequences of an inefficient barrier or barrier failure.

For sea lamprey, only the *status quo* was relevant to the management decisions that could be made at FishPass. However, we included additional scenarios for modeling sea lamprey production to weigh the consequences of non-desirable species production and forecast what-if scenarios. In addition to the *status quo*, which restricted approximately 800 sea lamprey below Union Street Dam, we simulated a *reduced passage* scenario that allowed around 50 sea lamprey access to the upper watershed annually, and a *full passage* scenario that allowed a full run of sea lamprey (~800 individuals) access to the upper watershed annually.

ODD Model Description

The model description follows the ODD (Overview, Design concepts, Details) protocol (Grimm et al. 2006, 2010). All IBMs were developed and simulations conducted in R (version 2022.12.0; R Core Team 2023). We report the percent change in median and interquartile range of abundance relative to the *status quo* scenario for the last 5 years of simulation for all species, spatial locations of fish in the final year of simulation for all species except lake sturgeon, and length frequencies in the final year of simulation for brook trout and walleye only. For annual abundance of spawners and recruits see Appendix.

Purpose.– The purpose of the IBM framework was to predict novel fish production for six migratory species (brook trout, Chinook salmon, lake sturgeon, sea lamprey, steelhead, and walleye) under five different fish passage scenarios to inform fisheries managers in determining a selective fish passage regime once the FishPass project is completed.

Entities, state variables, and scales.–

Individuals– The modeled individuals were fish of a single species. The state variables of individual fish were age (years), length (mm), sex, maturity, genetically distinct population group, spatial coordinates, and score of the habitat the entity is in. The six species were modeled separately, and some variables were only applied to a subset of the species.

Spatial Representation– The Boardman-Ottaway River was represented spatially as a series of connected cells on an (x, y) grid. Cell areas were heterogeneous in length and based on the National Hydrography Dataset Plus Version 2 (NHDPlusV2; USEPA and USGS 2005). The main channel of the Boardman-Ottaway River was represented by a continuous 1×26 grid while tributaries were represented by open cells adjacent to the main channel. Movement in the x-direction represented mainstem movement within the Boardman-Ottaway River while movement in the y-direction represented movement within a tributary. The state variables of the grid cells were spatial coordinates (x and y), length (m), width (m), area (m^2), network catchment area (km^2), network catchment area score (Dean 2023), species-specific habitat reach score (Dean 2023), species-specific larval area (fish per acre), and proportion of sea lamprey habitat type. Habitat suitability of the Boardman-Ottaway River was estimated for several migratory Great Lakes species including steelhead (Appendix: Figure A2.4), sea lamprey, (Appendix: Figure A2.5), walleye (Appendix: Figure A2.6), and brook trout (Appendix: Figure A2.7; Dean 2023). The results of these habitat suitability models included suitability scores for stream reaches (calculated in confluence-to-confluence segments) for each species and were attributed to habitat cells within the spatial framework. Network catchment area score (Dean 2023) indicated species preference for the size of a particular stream reach. All IBMs used HS data from Dean (2023), except for lake sturgeon because spatial aspects were not considered in that model. Dean (2023) did not score habitat suitability for Chinook salmon; therefore, we used the steelhead habitat suitability scores as a proxy for Chinook salmon habitat suitability. We estimated the proportion of sea lamprey habitat type (Dawson and Jones 2009) at 15 field sites and extrapolated to unsampled sites by assuming adjacent cells would be similar in habitat and unsampled tributaries most similar to the nearest sampled tributary (Appendix: Figure A2.3).

Time Step– One time step represents 1 year, and simulations were run for 50 years and 1000 iterations. The lake sturgeon IBM was run for 250 years due to the species' long life history and late age-at-maturity (Schueller and Hayes 2011; Nelson et al. 2022).

Process overview and scheduling.– The IBM used a sequential set of steps to represent the species-specific seasonal cycle of migration, spawning, and other life history events (Figure 2.2). At each process step, the values of attributes of individuals were updated. The attributes of grid cells were held constant. Time is modeled as discrete years and updated at the end of the model schedule.

For individual fish, the model proceeds as:

1. Pre-spawn mortality
2. Move to spawning grounds
3. Reproduce (add new recruits)
4. Post-spawn mortality
5. Grow
6. Mature, Transform, or Export (for species that leave the river as part of life cycle)

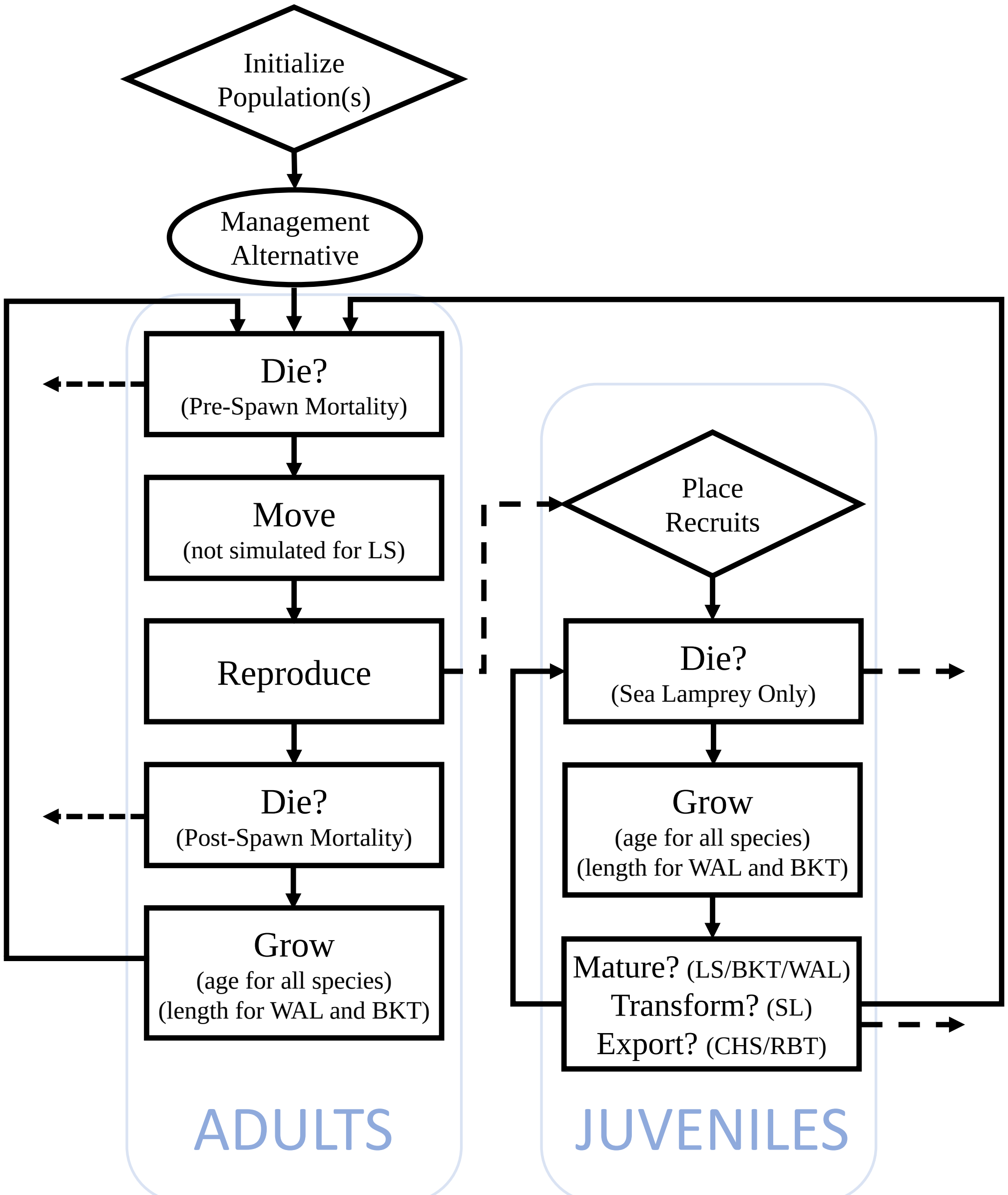

Figure 2.2. Conceptual diagram for the lake sturgeon *Acipenser fulvescens* (LS), Chinook salmon *Oncorhynchus tshawytscha* (CHS), steelhead *Oncorhynchus mykiss* (RBT), sea lamprey *Petromyzon marinus* (SL), walleye *Sander vitreus* (WAL), and brook trout *Salvelinus fontinalis* (BKT) IBMs. Dashed lines indicate where individuals are added to (e.g., recruitment) or removed from the model (e.g., death, transformation, and export to Lake Michigan).

Design Concepts.– Our model was built as part of a larger decision analysis process that required the forecasting of potential intended and unintended consequences of selective fish passage to inform decision making. Several species were desirable to stakeholders; therefore, due to variations in life history, data availability, initial distribution, and current understanding of species population statuses in the watershed, we opted for a simple and flexible modeling framework that can be improved upon as more information becomes available.

The IBMs for each species had unique aspects to account for key differences in species life history. For example, the lake sturgeon IBM evaluated how changes in intrinsic growth rate (r) and number of fish passed in the first year (N_1) affected abundance over time. The model included key aspects of lake sturgeon population dynamics, namely low initial abundances, sexually dimorphic late-age maturation, periodic spawning behavior, and low rates of intrinsic population growth and natural mortality (Table 2.2).

Table 2.2. Parameter values and references used in the lake sturgeon *Acipenser fulvescens* individual-based model developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity.

Parameter Description	Value	Reference
Non-varying parameters		
Male age-at-maturity	15	Schueller and Hayes (2011)
Female age-at-maturity	20	Schueller and Hayes (2011)
Annual mortality rate	5%	Schueller and Hayes (2011); Hayes et al. (2012); Colborne et al. (2021)
Annual female spawning probability	20%	Forsythe et al. (2012); Dammerman et al. (2019)
Annual male spawning probability	50%	Forsythe et al. (2012); Dammerman et al. (2019)
Carrying capacity of nursery grounds, K	200	Hayes et al. (2012)
Annual total recruitment, R_t		Calculated each year; Equation 2.5
Annual total spawners, S_t		Calculated each year
Recruitment error, ε	$\sim N(0, 0.25)$	Assumed
Varying parameters		
Initial population abundance, N_1	10	Assumed
	40	
	80	
Intrinsic growth rate, r	0.05	Bruch et al. (2016)
	0.1	
	0.3	

We used three intrinsic population growth rate values to represent a low growth (r= 0.05), medium growth (r= 0.10), and high growth scenarios (r= 0.30) and three values of adult spawner abundance passed upstream in the first year (N_1= 10, 40, and 80) to forecast abundance over 250-years, because there is high uncertainty about the current status of lake sturgeon in the watershed (see Appendix: Section A1). The Chinook salmon and steelhead IBMs included specific aspects of salmonid life history such as a riverine juvenile stage lasting from one to two years, high fishing mortality during migratory runs, and semelparous (Chinook salmon) or iteroparous (steelhead) spawning (Table 2.3). The sea lamprey IBM combined aspects of simulation models developed by Jones et al. (2009), Robinson et al. (2013), and Lin and Robinson (2019), and accounted for key aspects of sea lamprey life history including a protracted stream-dwelling larval stage (4–6 years), larval transformation to a lake-dwelling parasitic stage, and a semelparous life cycle (Table 2.4). The walleye IBM accounted for three genetically-distinct populations, sexually-dimorphic growth and maturation rates, and changes to carrying capacity, population growth rate, and length-at-age with the evaluated management scenarios (Table 2.5). The brook trout IBM incorporated aspects of the species' life history including high early life mortality and sexually-dimorphic maturation rates, and evaluated how changes to habitat suitability, carrying capacity, and growth in length rate affected the brook trout population over time (Table 2.6). See Appendix for additional details on species-specific models.

The emergent result of the model was population abundance. Individuals did not have adaptive traits; however, the rules used for modeling were intended to reproduce observed behaviors from other studies and were implicitly assumed to maximize individual fitness. No explicit sensing by individuals occurred, although individuals were assumed to know the habitat quality (i.e., habitat score).

Anticipated interspecies interactions were included by varying parameters of growth, reproduction, and movement for some species, based on literature review and discussions with agency representatives, but due to high uncertainty no direct interactions were modeled. For walleye and brook trout, we varied carrying capacity, intrinsic population growth rate, and length growth rate among the management scenarios (Table 2.7). We assumed length growth rate of walleye increased for all management scenarios relative to the *status quo*, due to increased production and prey availability from passage of native catostomids, cyprinids, cottids, and non-native Pacific salmonids. We assumed that passage of Pacific salmonids would negatively affect brook trout populations because this is well documented in the literature (Fausch and White 1986; Zorn et al. 2020). Pacific salmonid density was found to be negatively related to brook trout density, and brook trout abundance was lower in Great Lakes accessible reaches (Zorn et al. 2020). Therefore, we assumed scenarios that included salmonid passage would result in lower population growth rate and carrying capacity, a restricted spatial distribution, and reduced length-at-age of brook trout (see Appendix: Section A5 for more details).

Stochasticity was included in the model by drawing parameters from statistical distributions to allow individual variability while keeping the parameters within a predetermined range. Stochastic parameters varied by species, but generally included mortality, number of fish passed annually, intrinsic population growth rate, and recruitment error. Bernoulli trials were used to draw mortality events, spawning probability (if a fish would spawn in a given year), maturation, and transformation.

The only collectives in the model were the walleye population groups that were used to make three distinct population groups that did not interact during reproduction. Fish from the different population groups grew at different rates, and moved to unique spawning grounds to create group-specific spawning aggregations.

Table 2.3. Parameter values and references used in the Chinook salmon *Oncorhynchus tshawytscha* (CHS) and steelhead *Oncorhynchus mykiss* (RBT) individual-based models developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity. SDM= structured decision making.

Parameter	Definition	Value	Reference
Biological			
F_P	Number of fish passed per year (varies by mgmt. scenario)	**Status quo, native only*:** CHS: ~U(500, 800) RBT: ~TN(800, 200, 0, ∞) **Low steel, low salmon:** CHS: ~U(500, 800) RBT: ~U(800, 1000) **Full salmon:** CHS: ~TN(1182, 489, 0, ∞) RBT: ~TN(2000, 200, 0, ∞)	SDM workshops and expert opinion
K	Maximum density of recruits	CHS: 0.2 fish/m^2 RBT: 0.1 fish/m^2	Cooper et al. (2020)
M_{PRE}	Rate of mortality of adults prior to spawning due to harvest	CHS: ~U(0.20, 0.30) RBT: ~U(0.40, 0.50)	Expert opinion
M_{POST}	Rate of mortality of adults after spawning	CHS: 1 RBT: 0.10	Assumed
P_A	Spawner allocation percentage of rule allocation belief	0.5	Assumed
r_i	Intrinsic growth rate of stream reach, *i* (varies by RS_i)	Low: ~TN(0, 0.05, 0, ∞) Med: ~N(1, 0.1) High: ~N(3, 0.2)	Assumed
$R_{i,t}$	Stream reach abundance of recruits at time, *t*	varies by reach, *i* and year, *t*	–
$S_{i,t}$	Stream reach abundance of spawners at time, *t*	varies by reach, *i* and year, *t*	–
$S_{T,t}$	Total number of spawners for entire stream	varies by year, *t*	–
ε	Recruitment error	~N(0, 2)	Assumed
Habitat			
LA_i	Larval habitat area (m^2) of stream reach	varies by reach, *i*	–
NC_i	Network catchment area (km^2) of stream reach	varies by reach, *i*	Dean (2023)
RS_i	Habitat suitability score for stream reach	varies by reach, *i*	Dean (2023)
v^i	Stream reach length (m)	varies by reach, *i*	NHDPlusV2 (USEPA and USGS 2005)
w^i	Stream reach width (m)	varies by reach, *i*	Field measured and supplemented with Google Earth measurements

* spawners restricted below Union Street Dam; fish passed upstream = 0.

Table 2.4. Parameter values and references used in the sea lamprey *Petromyzon marinus* individual-based model developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity.

Parameter	Definition	Value	Reference
Biological			
F_P	Number of fish passed per year (varies by mgmt. scenario)	**Status quo*:** ~TN(803, 327.3, 0, 3000) **Reduced passage:** ~TN(50, 100, 0, 200) **Full passage:** ~TN(803, 327.3, 0, 3000)	Boardman-Ottaway River trapping data and expert opinion
Alpha, α	Ricker recruitment parameter	4.346	Jones et al. (2009)
Beta, β	Ricker recruitment parameter	0.1573	Jones et al. (2009)
$L_{i,0,t}$	Larval stream reach abundance of age-0 recruits at time, t	varies by reach	–
M_{PRE}	Rate of mortality of adults prior to spawning	0.10	Assumed
M_{POST}	Rate of mortality of adults after spawning	1	Assumed
M_L	Annual mortality rate for larval fish	0.46	Jones et al. (2009); Johnson et al. (2016)
P_A	Spawner allocation percentage of rule allocation belief	0.50	Assumed
P_f	Proportion of females in population	0.50	Assumed
$S_{i,t}$	Stream reach abundance of spawners at time, t	varies by reach	–
$S_{T,t}$	Total number of spawners for entire stream	varies by year, t	–
T_P	Transformation probability	**Age-1 to -3:** 0 **Age-4:** 0.46 **Age-5:** 0.57 **Age-6:** 1	Haeseker et al. (2003); Robinson et al. (2013)
ε	Recruitment error	~N(0, 3.39)	Jones et al. (2009)
Habitat			
H_{1i}	Proportion of type I larval habitat of stream reach	varies by reach	Field measured and extrapolated
H_{2i}	Proportion of type II larval habitat of stream reach	varies by reach	Field measured and extrapolated
LA_i	Larval habitat area (m^2) of stream reach	varies by reach	–
NC_i	Network catchment area (km^2) of stream reach	varies by reach	Dean (2023)
r	Habitat suitability scalar	0.38	Jones et al. (2009)
RS_i	Habitat suitability score for stream reach	varies by reach	Dean (2023)

Table 2.4 (cont'd)

v^i	Stream reach length (m)	varies by reach	NHDPlusV2 (USEPA and USGS 2005)
w^i	Stream reach width (m)	varies by reach	Field measured, supplemented with Google Earth measurements

* spawners restricted below Union Street Dam; fish passed upstream = 0.

Table 2.5. Parameter values and references used in the walleye *Sander vitreus* individual-based model developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity. SD=standard deviation, BR=Boardman Lake walleye population, GL1=Great Lakes walleye population 1, and GL2=Great Lakes walleye population 2.

Parameter	Definition	Value	Reference
Biological			
N_1	Initial population size	See Appendix: Section A4	Little Traverse Bay Band of Odawa Indians/Grand Traverse Band of Ottawa and Chippewa Indians 2014 survey; Gehri et al. (2021)
M_{PRE}	Rate of mortality of adults prior to spawning due to harvest	~TN(0.11, 0.1, 0.05, 0.16)	Schneider et al. (2007); Roseman et al. (2010)
M_{POST}	Rate of mortality of adults after spawning	~TN(0.11, 0.1, 0.05, 0.16)	Schneider et al. (2007); Roseman et al. (2010)
r_i	Intrinsic population growth rate	varies by population, *i* and scenario; see Table 2.7	Assumed
ε	Recruitment error	~N(0, 0.5)	Assumed
T_{MAT}	Age-at-maturity	Female: age-4 & > 330mm Male: age-3 & >300mm	Schneider et al. (2007); Bozek et al. (2011)
L_∞	Mean asymptotic length (mm) (State average: 645mm)	**Female BR & GL1** Low: 768 (SD=20) Med. 779 (SD=20) High: 779 (SD=20) **Female GL2** Low: 779 (SD=20) Med: 779 (SD=20) High: 779 (SD=20) **Male BR & GL1** Low: 750 (SD=20) Med: 634 (SD=20) High: 634 (SD=20) **Male GL2** Low: 634 (SD=20) Med: 634 (SD=20) High: 634 (SD=20)	Schneider et al. (2000); Roseman et al. (2010)
k	Brody growth coefficient (State average: 0.24)	**Female BR & GL1** Low: 0.11 (SD=0.05) Med: 0.16 (SD=0.05) High: 0.21 (SD=0.05) **Female GL2** Low: 0.16 (SD=0.05) Med: 0.16 (SD=0.05) High: 0.26 (SD=0.05)	Schneider et al. (2000); Roseman et al. (2010)

Table 2.5 (cont'd)

		Male BR & GL1 Low: 0.10 (SD=0.05) Med: 0.23 (SD=0.05) High: 0.24 (SD=0.05) **Male GL2** Low: 0.23 (SD=0.05) Med: 0.23 (SD=0.05) High: 0.33 (SD=0.05)	
t_0	Age-at-length 0	0	Assumed
$R_{i,t}$	Population-specific abundance of recruits at time, *t*	varies by population, *i* and year, *t*	–
$S_{i,t}$	Population-specific abundance of spawners at time, *t*	varies by population, *i* and year, *t*	–
Habitat			
K_i	Population-specific carrying capacity	varies by population, *i* and scenario; see Table 2.7	–

Table 2.6. Parameter values and references used in the brook trout (BKT) *Salvelinus fontinalis* individual-based model developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity. SD=standard deviation.

Parameter	**Definition**	**Value**	**Reference**
Biological			
N_1	Initial population	See Appendix: Section A5	
M_{PRE}	Rate of mortality of adults prior to spawning due to harvest	~*N*(0.5, 0.05) age-1 ~*N*(0.776, 0.025) age-2	Zorn et al. (2020)
M_{POST}	Rate of mortality of adults after spawning	~*N*(0.5, 0.05) age-1 ~*N*(0.776, 0.025) age-2	Zorn et al. (2020)
r_i	Intrinsic population growth rate	See Table 2.7	Assumed
ε	Recruitment error	~*N*(0, 1)	Assumed
T_{MAT}	Age-at-maturity	Female: 80% @ age-1 Male: 100% @ age-1 100% at age-2	McFadden et al. (1967); Zorn et al. (2018)
L_∞	Mean asymptotic length (mm) (State average: ~360mm)	Low: 340 (SD=20) Med: 360 (SD=20) High: 380 (SD=20)	Kalish et al. (2018); Zorn et al. (2018); Hettinger (2020)
k	Brody growth coefficient (State average: ~0.24)	Low: 0.30 (SD=0.05) Med: 0.35 (SD=0.05) High: 0.40 (SD=0.05)	Kalish et al. (2018); Zorn et al. (2018); Hettinger (2020)
t_0	Age-at-length 0	–0.6	Assumed
$R_{i,t}$	Reach-specific abundance of recruits at time, *t*	varies by reach, *i* and year, *t*	–
$S_{i,t}$	Reach-specific abundance of spawners at time, *t*	varies by reach, *i* and year, *t*	–
$S_{T,t}$	Total number of spawners for entire stream	varies by year, *t*	–
YOY_{MI}	Mean density of young-of-year BKT in Michigan	265 BKT/hectare	Zorn et al. (2018)
P_A	Percentage belief in NSC_i spawner allocation rule	0.75	Assumed
$(1 – P_A)$	Percentage belief in RS_i spawner allocation rule	0.25	Assumed
Habitat			
K_i	Reach-specific carrying capacity	varies by reach, *i*; See Table 2.7	–
f_A	K_i habitat scalar factor	varies by scenario	–
RS_i	Habitat suitability score for stream reach	varies by reach, *i* and scenario	Dean (2023)

Table 2.6 (cont'd)

A_i	Area of stream reach (hectare)	varies by reach, *i*	NHDPlusV2 (USEPA and USGS 2005)
NCS_i	Network catchment area score	varies by reach, *i*	Dean (2023)

Table 2.7. Varying parameters for the walleye *Sander vitreus* and brook trout *Salvelinus fontinalis* individual-based models developed to predict effects of changes in connectivity on the Boardman-Ottaway River, MI, on fish productivity. BL=Boardman Lake, GL1 and GL2= Great Lakes populations 1 and 2.

	Management Scenario			
Parameter	**Status Quo**	**Native Only**	**Low Steel Low Salmon**	**Full Salmon**
Walleye				
K_i	**BL**=800 **GL1**=500 **GL2**=500	**BL**=1600 **GL1**=1000 **GL2**=1000	**BL**=1920 **GL1**=1200 **GL2**=1200	**BL**=2200 **GL1**=2200 **GL2**=2200
r_i	**BL=** ~TN(0.5, 0.3, 0, 1)	**BL=** ~TN(0.6, 0.2, 0, ∞)	**BL=** ~TN(0.6, 0.2, 0, ∞)	**BL=** ~TN(0.6, 0.2, 0, ∞)
	GL= ~TN(0.5, 0.3, 0, 1)	**GL=** ~TN(0.6, 0.2, 0, ∞)	**GL=** ~TN(0.6, 0.2, 0, ∞)	**GL=** ~TN(0.6, 0.2, 0, ∞)
Growth (length-at-age)	Low	Medium	High	High
Brook trout				
K_i	Statewide YOY_{MI} average	Increased 25% for entire watershed	Reduced 25% from status quo in BR mainstem	Reduced 25% for entire watershed
r_i	Varies based on RS_i Low r_i= ~TN(0.75, 0.2, 0, ∞); RS_i <=11 Medium r_i= ~TN(1.5, 0.2, 0, ∞); RS_i >=12 and <15 High r_i= ~TN(2, 0.2, 0, ∞); RS_i >=15			
Habitat Suitability Reach score (RS_i)	Reduced only below Sabin Dam	Reduced only below Sabin Dam	Reduced by 10% in BR mainstem	Reduced by 10% for entire watershed
Growth (length-at-age)	Medium; see Appendix	High; see Appendix	Medium; see Appendix	Low; see Appendix

Initialization.– All species models began by initializing a population and selecting a management alternative. All IBMs were initialized assuming a 1:1 male-female ratio and that all adults were sexually mature and capable of spawning. Initial abundances were spatially allocated based on field survey sampling when available or through discussions with agency representatives with knowledge of the Boardman-Ottaway River system (see Appendix). Annual spawning run sizes for Chinook salmon, steelhead, and sea lamprey were randomly drawn from uniform or truncated normal distributions (Tables 2.3 and 2.4). Walleye were assumed to have three genetically-distinct populations in the Boardman-Ottaway watershed: two Lake Michigan populations and one inland Boardman Lake population (Gehri et al. 2021). Monitoring studies suggest a spawning run of about 500–1000 walleye is likely to develop from Great Lakes populations (Reid Swanson, GLFC *personal oral communication, June 8, 2021*) and in 2014 the Boardman Lake population abundance was estimated to be approximately 807 (Brett Fessell, Grand Traverse Band of Ottawa and Chippewa Indians, unpublished data, 2021[1]). Of the six species modeled, only brook trout were initially present only above Union Street Dam, and only walleye were present both above and below Union Street Dam. Because migratory brook trout from Lake Michigan do not currently occupy the Boardman-Ottaway River, none of the scenarios included brook trout passage at the Union Street Dam.

Input data.– Habitat sampling (Dean 2023), which was used to inform the IBMs, was performed in the Boardman-Ottaway River watershed at 15 sites from 2019 to 2020 (Appendix: Figure A2.2) following the MDNR Streams Status and Trends Program sampling protocol for wadeable streams (Wills et al. 2008). Cell widths associated with the 15 sampling sites were field measured, and the remaining cell widths were determined by averaging five width measurements in Google Earth for each spatial cell in the NHDPlusV2 dataset. Hobo Water Temperature Pro v2 data loggers (Onset Computer Corporation, Bourne, MA) were deployed throughout the Boardman-Ottaway River system in 2019 and 2020 (11 sites in 2019 and 12 sites in 2020, with 4

[1] At the time of publication, data were not available from the Little Traverse Bay Band of Odawa Indians and the Grand Traverse Band of Ottawa and Chippewa Indians

sites duplicated each year). In addition to our field-collected data, other data were provided by partners throughout the region (Table 2.8).

Table 2.8. Description of the habitat and biological data used to inform the development and parameterization of individual-based models to describe the effects of selective fish passage on six native or introduced Great Lakes fish species in the Boardman-Ottaway River, Michigan. Fish species were: lake sturgeon *Acipenser fulvescens*, sea lamprey *Petromyzon marinus*, steelhead *Oncorhynchus mykiss*, Chinook salmon *Oncorhynchus tshawytscha*, walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis*.

Data	Years	Reference
Habitat		
Boardman-Ottaway River habitat sampling data (bank condition, substrate, cover, channel morphology, and sea lamprey habitat survey)	2019–2020	Field collected at 15 sites in watershed
Boardman-Ottaway River habitat suitability reach scores	2019	Dean (2023)
Network catchment area estimates	2020	Dean (2023)
Temperature	2019–2020	Field collected at 19 sites in watershed
Biological		
Species presence up- and downstream of Union Street Dam	NA	Kalish et al. (2018)
Boardman-Ottaway River weir catch data	1987–Current	Kalish et al. (2018)
Chapman-Pedersen population estimate for brook trout	1960–2019	Kalish et al. (2018); Hettinger et al. (2020)
Average total length-at-age and growth (relative to state average) for brook trout	2002–2010	Kalish et al. (2018)
Abundance (no./hectare) of brook trout at 5 long-term, fixed population survey sites	2005–2006	Kalish et al. (2018)
Brown Bridge removal evaluation survey for brook trout	2015	Burroughs (2016)
Boardman Lake walleye survey	2014	Little Traverse Bay Band of Odawa Indians/Grand Traverse Band of Ottawa and Chippewa Indians 2014
Adult sea lamprey trap catches	1980–2016	U.S. Fish and Wildlife Service, Sea Lamprey Control Program
Sea lamprey larval survey results (up- and downstream of Union Street Dam)	1960–2016	U.S. Fish and Wildlife Service, Sea Lamprey Control Program
Median larval sea lamprey production estimates by stream reach	2015	U.S. Fish and Wildlife Service, Sea Lamprey Control Program

Submodels.– Life history parameter estimates for survival, recruitment, maturation, growth, and movement were based on data (when available), literature review, or discussions with agency representatives participating in the SDM workshops. In most cases, distributional assumptions were made for parameters to include stochasticity and account for individual variability.

Mortality was separated into pre- and post-spawn events to represent semelparous and iteroparous life histories and the age of recruitment for all species was set to the species-specific age-at-maturity. We converted annual mortality to two additive instantaneous rates of pre- and post-spawn mortality. Only mature fish experienced mortality, except sea lamprey which also experienced larval mortality (see Appendix for details on species-specific models). Mortality was determined through a Bernoulli trial for each individual. Individuals that died were removed from the population.

Movement of spawning Chinook salmon and steelhead was simulated by two mechanisms that allocated adult spawners to stream reaches based on 1) the stream reach network catchment area (i.e., larger stream reaches were preferred over smaller stream reaches) and 2) the HS reach score (i.e., higher reach scores were preferred over lower reach scores; Dean (2023); Equation 2.1; see Table 2.3 for parameter definitions).

Equation 2.1: $S_{i,t} = S_{T,t} p_A \frac{NC_i}{\sum NC_i} + S_{T,t}(1 - p_A) \frac{RS_i}{\sum RS_i}$

For sea lamprey, we replaced habitat suitability reach score with the estimated larval habitat area (Equations 2.2 and 2.3; see Table 2.4 for parameter definitions), and for brook trout we replaced stream network catchment area with the stream reach network catchment area score, which scores smaller stream reaches higher (more preferred) than larger stream reaches (Equation 2.4; Dean (2023); see Table 2.6 for parameter definitions).

Equation 2.2: $S_{i,t} = S_{T,t} p_A \frac{NC_i}{\sum NC_i} + S_{T,t}(1 - p_A) \frac{LA_i}{\sum LA_i}$

Equation 2.3: $LA_i = v_i w_i (H_{1i} + H_{2i} r)$

Equation 2.4: $S_{i,t} = S_{T,t} p_A \frac{NCS_i}{\sum NCS_i} + S_{T,t}(1 - p_A) \frac{RS_i}{\sum RS_i}$

Movement was not included in the lake sturgeon model and only implicitly included in the walleye model by creating population-based spawning aggregations in pre-designated spatial cells. Larval and juvenile movement were not simulated for any species and recruits were allocated on the spatial grid depending on where adults were located during spawning.

After adults were allocated to stream reaches, recruitment was calculated with a Ricker function (Ricker 1954). For lake sturgeon and walleye, recruitment was calculated on a population-level for the river (one population for lake sturgeon, three for walleye; Equation 2.5; see Tables 2.2, 2.5, and 2.6 for parameter definitions).

Equation 2.5: $R_{i,t} = S_{i,t}e^{r_i(1-(S_{i,t}/K_i))-1} + \varepsilon$

For Chinook salmon and steelhead, the recruitment function was modified to allow intrinsic population growth rate, *r*, to vary among stream reaches by habitat suitability reach score, and we replaced carrying capacity, *K*, with estimated stream reach larval area, *LA* (Equations 2.6 and 2.7; see Table 2.3 for parameter definitions).

Equation 2.6: $R_{i,t} = S_{i,t}e^{r_i(1-(S_{i,t}/LA_i))-1} + \varepsilon$

Equation 2.7: $LA_i = (v_i w_i) * K$

For brook trout recruitment, intrinsic population growth rate, *r*, varied among stream reaches by habitat suitability reach score, and we allowed carrying capacity, *K*, to vary among stream reaches (Equations 2.5 and 2.8; see Table 2.6 for parameter definitions and Appendix for additional details). For sea lamprey, recruitment was calculated for each stream reach based on a Ricker function modified from Jones et al. (2009; Equation 2.9; see Table 2.4 for parameter definitions).

Equation 2.8: $K_i = A_i * \mathrm{YOY_{MI}} * f_A$; if $f_A = 1$ (status quo K_i)

Equation 2.9: $L_{i,0,t} = \frac{\alpha S_{i,t} p^f e^{-\beta \frac{S_{i,t} P_f}{LA_i} + \varepsilon}}{M_L}$

Following recruitment, recruits remained in natal stream reaches until maturation (lake sturgeon, brook trout, and walleye), transformation to juvenile parasitic stage (sea lamprey), or migration to Lake Michigan as juveniles (Chinook salmon and steelhead). While in natal stream reaches recruits aged but did not change size, except for brook trout and walleye which also increased in size. Length-at-age was modeled using a von Bertalanffy growth function (Beverton and Holt 1957) and updated each year based on age, sex, and population. We parameterized a low, medium, and high length growth rate scenario for walleye (Appendix: Figure A2.8) and brook trout (Appendix: Figure A2.9); therefore, growth in length-at-age varied among management alternatives but maturation schedules did not.

Results

Lake sturgeon

Median lake sturgeon abundance declined under the low and medium growth scenarios (r= 0.05 and 0.10) and increased for the high growth scenarios (Figure 2.3). When initial abundance was low, nearly 100% of simulations ended with a total abundance of 0, except when population growth rate was high (r= 0.3). Over 70% of the high growth scenario simulations ended in year-250 with population abundances above the minimum viable population size of 80 (Schueller and Hayes 2011; Hayes et al. 2012), regardless of initial abundance (Figure 2.3).

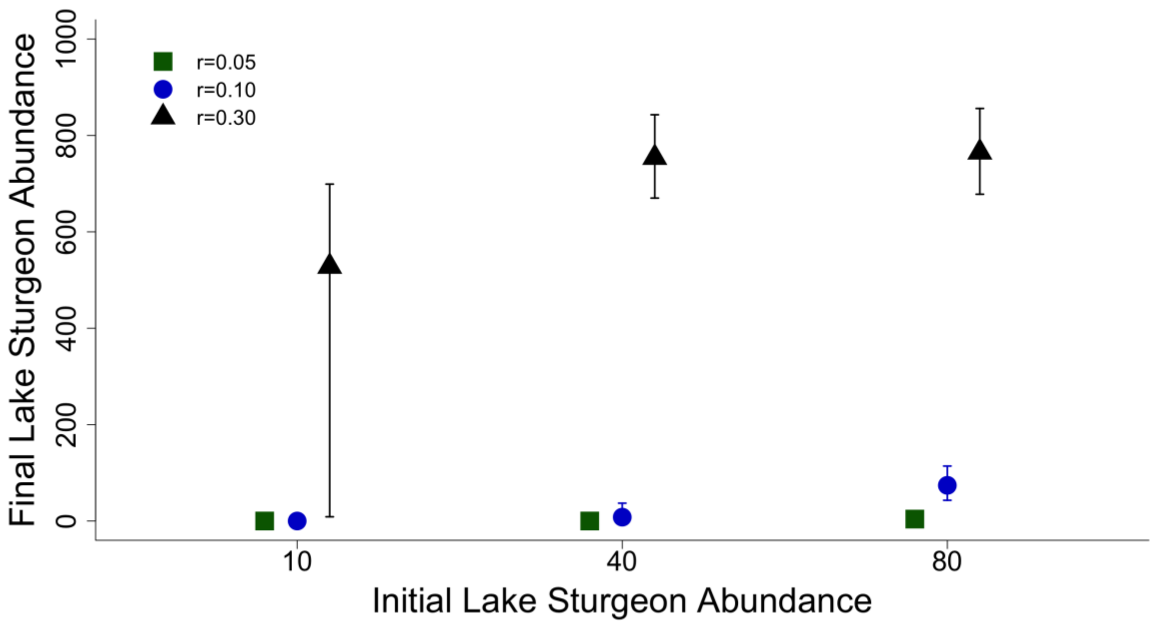

Figure 2.3. Predicted lake sturgeon *Acipenser fulvescens* population abundance in 250 years in the Boardman-Ottaway River for three initial abundances, N_I (10, 40, 80), and three intrinsic growth rates, r (0.05–green square, 0.1–blue circle, 0.3–black triangle). Points are the median of all simulations and error bars show the interquartile range. Points are jittered along the x-axis.

Median abundance in the last five years of the high growth–low initial abundance scenario was 528 and reached an equilibrium ($N \approx 765$) for the medium and high initial abundance scenarios (Figure 2.3). Median population abundances for all low and medium growth scenarios declined from the initial abundance over time (Figure 2.3); however, for the medium growth–high initial abundance scenario (r= 0.1, N_0= 80), 45% of simulations ended with an estimated population above the minimum viable population size of 80. Lake sturgeon abundance declined to zero for 76% of simulations when initial abundance was low and 29% when initial abundance was medium, but abundance was predicted to increase when initial abundance was high with 100% of simulations ending with population abundance above the minimum viable population size of 80 (Figure 2.3).

Chinook salmon and steelhead

Chinook salmon median abundance in the last five years of simulations increased 762% under the *low salmon* scenario, and 1430% under the *full salmon* scenario compared to the *status quo* (Figure 2.4). Steelhead abundance increased 1059% under the *low steel* and *low salmon* scenarios, and 2393% under the *full salmon* scenario compared to the *status quo* (Figure 2.4). For both species, the spatial distribution and abundance within the Boardman-Ottaway River watershed increased for all scenarios (relative to the *status quo* and *native only* scenarios) and was highest for the *full salmon* scenario. For the *status quo* and *native only* scenarios, average densities of Chinook salmon recruits in the lower Boardman-Ottaway River below Union Street Dam ranged from 0 to 150 recruits/ha and for Steelhead ranged from 0.8 to 320 recruits/ha. The highest density of both species was found in Kids Creek. For the *low steel* and *low salmon* scenarios, Chinook salmon densities ranged from 0 to 3200 recruits/ha, and for steelhead ranged from 0 to 400 recruits/ha. The highest densities of both species were found in Jaxon Creek. For the *full salmon* scenario, the estimated density of Chinook salmon ranged from 0.6 to 16600 recruits/ha and steelhead ranged from 0 to 3200 recruits/ha. The highest density of recruits for both species were found in the former Sabin Dam area.

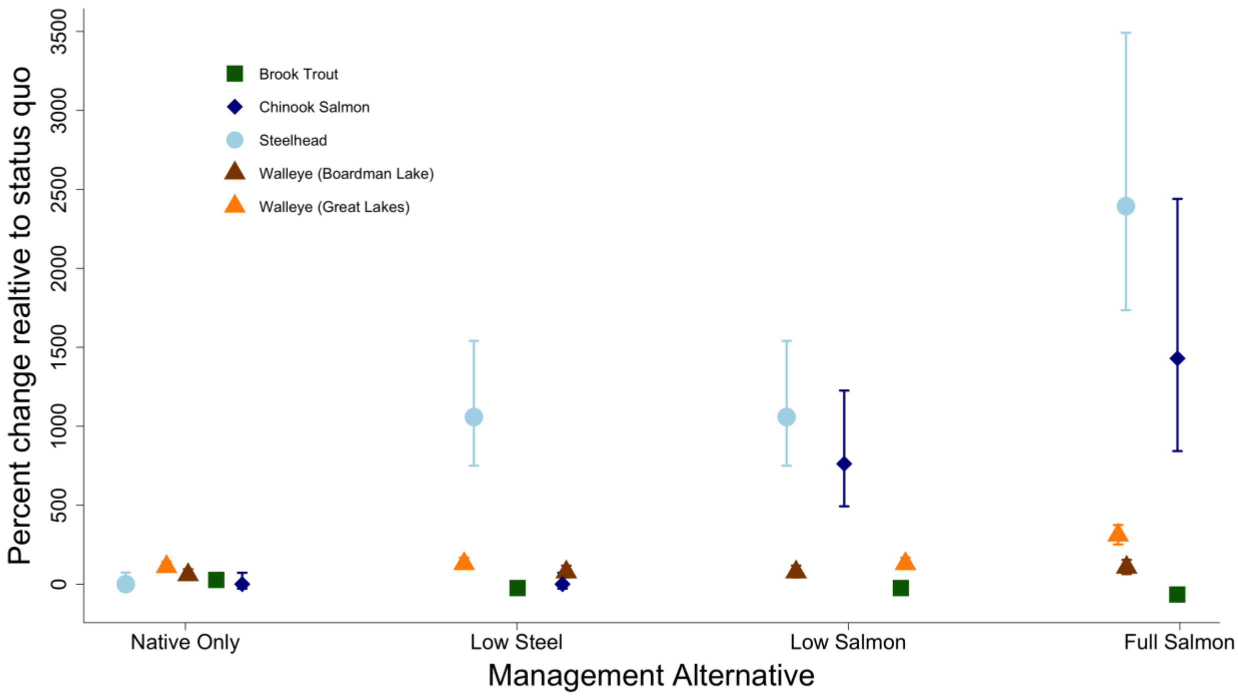

Figure 2.4. Predicted percent change in population abundance relative to the status quo in the last five years of simulation of lake sturgeon *Acipenser fulvescens*, Chinook salmon *Oncorhynchus tshawytscha*, steelhead *Oncorhynchus mykiss*, walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis* for each management scenario at FishPass on the Boardman-Ottaway River. Points are jittered along the x-axis.

Sea lamprey

We estimated the median abundance of sea lamprey in the last five years of simulation for the *status quo* with all spawners below Union Street Dam and compared the results with two scenarios that allowed passage of sea lamprey. Median population and larval transformer abundance were highest for the *full passage* scenario and lowest for the *reduced passage* scenario. The *reduced passage* scenario can be thought of as additional production in the upper river because under *status quo*, a spawning aggregation of sea lamprey would still be present below Union Street Dam, but production in the upper Boardman-Ottaway River would be equal to 0. The *reduced passage* scenario models approximately 50 sea lamprey in the upper river only, ignoring the larger spawning aggregation of ~750 sea lamprey that would still be blocked below

Union St. Dam. Therefore, the *status quo* abundance is higher than the *reduced passage* abundance when in reality, both scenarios would probably be occurring at the same time. For the *reduced passage* scenario, median abundance of larval sea lamprey in the Boardman-Ottaway River declined 71% and the number of transformers emigrating annually declined 81% compared to the *status quo* (Figure 2.5). For the *full passage* scenario, sea lamprey abundance increased 196% and the number of larval transformers annually emigrating from the Boardman-Ottaway River to Lake Michigan increased 119% (Figure 2.5).

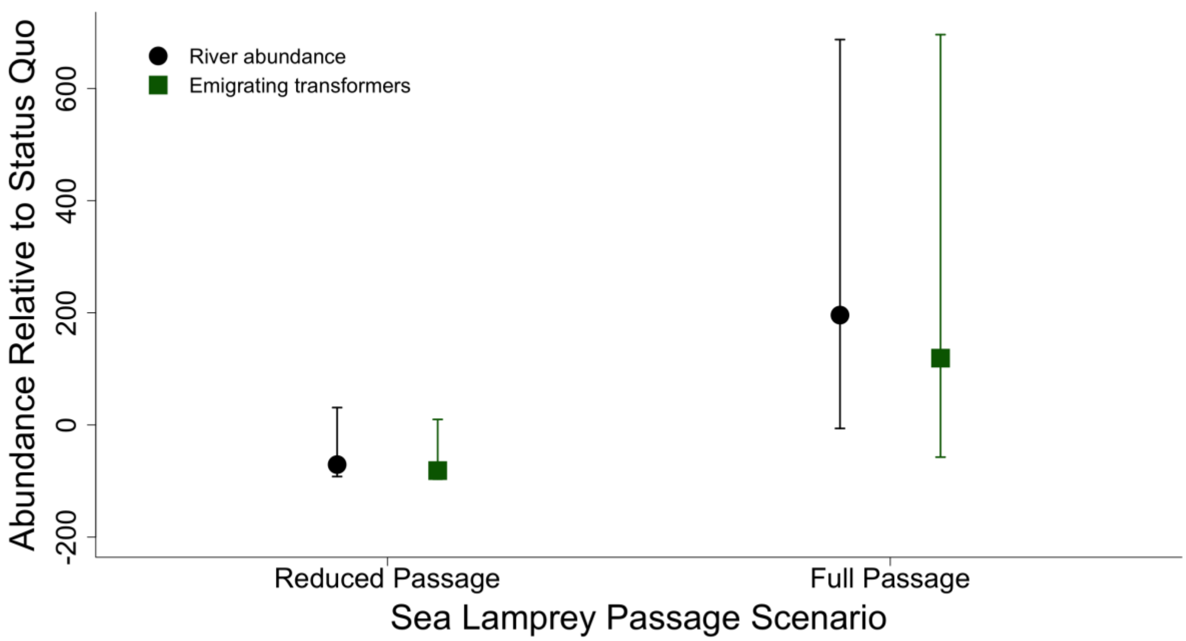

Figure 2.5. Predicted percent change in annual juvenile sea lamprey *Petromyzon marinus* abundance in the upper Boardman-Ottaway River (black circles) and emigration of larval transformers from the upper Boardman-Ottaway River to Lake Michigan (green squares) for the reduced and full passage management scenarios relative to the estimated status quo river abundance and emigrating transformers below Union Street Dam. Points are the median values and error bars are the interquartile ranges.

For the *status quo* scenario, larval densities in the lower Boardman-Ottaway River below Union Street Dam ranged from 10 to 20 fish/ha and were highest in Kids Creek. For the reduced passage scenario, larval densities were all < 20 fish/ha with the highest density in the Boardman-

Ottaway River above the former Brown Bridge impoundment. Fewer tributaries became infested in the *reduced passage* scenario compared to the *full passage* scenario. The spatial distribution of sea lamprey was greatest for the *full passage* scenario and larval densities ranged from 0 to 6500 fish/ha with the highest density in the North Branch Boardman River. Only two tributaries did not become infested by sea lamprey for the *full passage* scenario: Robbins Creek and an unnamed tributary.

Walleye

The median abundance for the last 5 years of simulations of all walleye populations increased for all scenarios compared to the *status quo* (Figure 2.4). The highest abundances for both the Boardman Lake and Great Lakes populations resulted under the *full salmon* scenario (Figure 2.4). Relative to the *status quo,* the estimated median abundance of Boardman Lake walleye increased 59% under the *native only* scenario and 106% under the *full salmon* scenario, while for the two Great Lakes walleye populations, median abundance increased 111% under the *native only* scenario and 310% under *full salmon* passage (Figure 2.4).

Length-at-age was highest for Boardman Lake walleye under the *low steel* and *low salmon* scenarios and lowest for the *status quo* (Figure 2.6). Among the evaluated scenarios, median length-at-age for Boardman Lake walleye ranged from 261 to 341 mm and Great Lakes walleye ranged from 280 to 420 mm (Figure 2.6). Length-at-age was similar for the *status quo* and *native only* scenarios, and among the three salmonid passage scenarios (Figure 2.6).

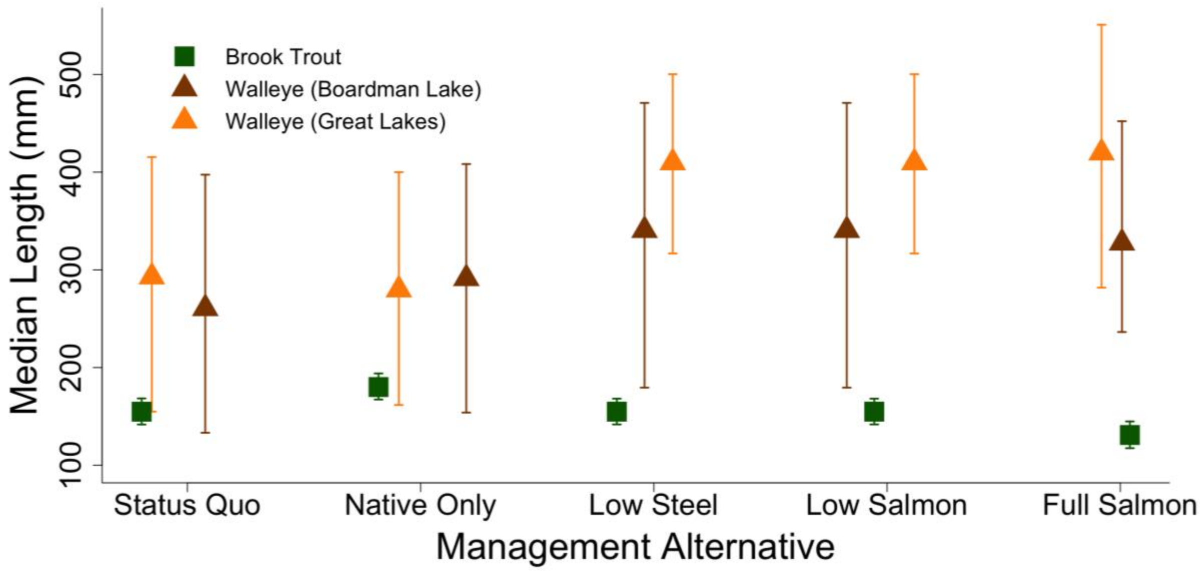

Figure 2.6. Predicted median length and interquartile range of walleye *Sander vitreus* and brook trout *Salvelinus fontinalis* populations in the final year of simulation of one iteration for each management scenario at FishPass on the Boardman-Ottaway River. Points are jittered along the x-axis.

Brook trout

The highest brook trout abundance occurred for the *native only* passage and the lowest brook trout abundance occurred when *full salmon* passage was allowed (Figure 2.4). Median brook trout abundance increased 27% compared to the *status quo* for the *native only* scenario, but declined 25% for the *low steel* and *low salmon* scenarios, and further declined 66% relative to the *status quo* with *full salmon* passage (Figure 2.4). Brook trout median length-at-age was highest for the *native only* scenario, lowest for the *full salmon* scenario, and similar among the *status quo*, *low steel*, and *low salmon* scenarios (Figure 2.6). Compared to the *status quo*, average length increased 16% to 180 mm for the *native only* scenario, but decreased 15% to 131 mm for the *full salmon* scenario (Figure 2.6).

The overall spatial distribution of brook trout did not vary among the scenarios. Brook trout were not found below the former Sabin Dam site for any of the evaluated scenarios;

however, all tributaries remained populated throughout the simulations. For the *status quo*, densities ranged from 10 to 4300 brook trout/ha, with the highest densities found in the tributaries and Boardman-Ottaway River above the former Brown Bridge site. For the *native only* scenario, densities ranged from 20 to 4800 brook trout/ha, with the highest densities occurring in Jaxon and East Creek. The former Sabin and Brown Bridge dam areas also had high densities of brook trout recruits for the native only scenario. The *low steel* and *low salmon* scenarios resulted in densities ranging from 6 to 4500 brook trout/ha, with the highest densities occurring in the mainstem Boardman-Ottaway River above the former Boardman Dam site and at the former Brown Bridge site (Figure 2.1). East Creek and Beitner Creek also had relatively high density of brook trout recruits when *low steel* and *low salmon* passage was allowed. When *full salmon* passage was allowed, densities ranged from 2 to 830 brook trout/ha, with the highest densities occurring in the Boardman-Ottaway River at the former Brown Bridge Dam site. Jaxon Creek had the highest density of brook trout recruits among tributaries for the *full salmon* scenario.

Discussion

Our IBM framework was developed specifically to forecast the potential consequences and tradeoffs of selective connectivity by elicitation of stakeholder objectives through SDM workshops. The species we chose to model were important to these groups and representative of larger guilds of species that are found throughout North America. Therefore, our framework should be highly exportable to other systems and represents perhaps the first attempt to build an "off-the-shelf" framework to predict the response of migratory fish populations to barrier removal and renovation (Hayes et al. 2023). Decision makers could use this modeling framework as a tool to forecast potential fish passage scenarios and assess stakeholder preferences and tradeoffs to aid in decision making for barrier removal or renovation projects. Additionally, this framework can be used to forecast potential unintended consequences of connectivity (McLaughlin et al. 2013; Milt et al. 2018), such as the spread of invasive species or possible declines in desirable species. Stakeholder engagement is an important part of effective natural resource management (Littell et al. 2017) and forecasting potential outcomes using the best available data prior to taking management action allows decision makers to evaluate tradeoffs among competing objectives, elucidate areas of critical uncertainty, and choose the optimal management action.

In many ways, the results of the species-specific models are much less important than the development of this modeling framework that can be iteratively updated and used as a decision-making tool to help managers make informed decisions for barrier removal and renovation projects. All barrier removal and renovation projects that make changes to stream connectivity are going to have unique needs for modeling based on the species present in the system, spatial aspects of the system, or other critical uncertainties. However, building a model, even with uncertainties, is a critical step which forces decision makers to be explicit about assumptions and can inform decisions given the best current information while acknowledging the need to improve in the future. It will be exceedingly rare for a barrier removal project to understand all of the inter-species interactions and demographic variables and how they might change with each different management scenario, while also understanding the relationship between the quality of habitat that has become available and changes in demographic parameters and other behavioral processes in response to gaining access to that habitat. Despite this unfortunate truth, managers can still benefit from an uncertain model built to inform decision making (Starfield 1997) and this is precisely what we have developed. This conceptual idea or exact model could be adapted to fit the needs of most if not all barrier removal or renovation projects. Additionally, this framework allows for learning about the system as we can test the assumptions in the model. The present study represents the first iteration of model development and an application of the model in the Boardman-Ottaway River case study.

This IBM framework allowed us to forecast changes in species dominance in a system under varying regimes of selective fish passage. We reported percent change relative to the *status quo*; however, when evaluating changes to the median abundance of each species we found that the Boardman-Ottaway River fish community could vary among the evaluated scenarios from a brook trout dominated system to a steelhead dominated system depending on the fish passage regime chosen. For the *status quo* and *native only* scenarios, the Boardman-Ottaway River remained a brook trout dominated system with walleye being the second most abundant species (of the species considered in the study) above Union Street Dam. The production of migratory species below Union Street Dam was relatively low for the *status quo* and *native only* scenarios. We found that the *low steel* and *low salmon* scenarios decreased the production of brook trout, but not enough to make the Boardman-Ottaway River a Pacific salmonid-dominated system. However, the *full salmon* scenario resulted in a system that was dominated by steelhead, with brook trout and Chinook salmon having similar abundances. Species that occurred below the barrier pre-removal benefitted more than species isolated above Union Street Dam. Of the species found below the barrier pre-removal, lake sturgeon abundance increased least, and Chinook salmon increased most when full passage of each was allowed. These results are dependent on the assumptions made in the model but are useful to visualize how a system might change.

A lake sturgeon population is unlikely to develop in the Boardman-Ottaway River given current population abundance and intrinsic growth rate estimates. The chances for successful restoration were most probable with initial populations greater than 40 and population growth rates greater than 0.1. These findings are comparable with Schueller and Hayes (2011) who found that initial lake sturgeon populations of greater than 80 had little chance of extinction, but extinction risk increased to 50% for initial populations of 40, and it was nearly 100% for initial populations of less than 15. Recently, Vaugeois et al. (2022) found that management actions which increase egg survival rate or stocking resulted in the fastest population growth and recovery compared to management actions that increased juvenile or adult survival rate; however, the study was focused on the population-level effects of contaminants. Finally, Nelson et al. (2022) found larger initial populations took less time to reach recovery target abundance, but they also assessed the effect of stocking yearlings into their model, which was shown to increase the probability of population recovery. Considerations for the effects of stocking lake sturgeon could be an important future use and area of development for this model (Baker and Scribner 2017).

As expected, we found Chinook salmon, steelhead, sea lamprey, and walleye abundances were likely to increase over time with increased passage and access to the upper Boardman-Ottaway River. Great Lakes walleye abundance increased more than Boardman Lake walleye abundance, and brook trout abundance declined in response to increasing non-native Pacific salmonid abundance in the upper watershed. This is not surprising because we parameterized our model to reduce habitat suitability, population growth rate, and individual growth in length-at-age with increasing Pacific salmonid passage; however, our model was parameterized with data from Michigan streams (Kalish et al. 2018; Zorn et al. 2018, 2020; Hettinger 2020). Despite relying on assumptions and parameters from studies of other Michigan streams, our model gives decision makers an estimate of *how much* abundance might change if one scenario is chosen over another, and *where* species might be most abundant, which can inform other riverine management decisions. Therefore, we expect our model results to conform to our assumptions and produce realistic projections when set in the context of nearby systems. Furthermore, understanding potential increases in non-native species stresses the importance of considering trade-offs for barrier removal or remediation projects (McLaughlin et al. 2013), especially after a previous barrier removal in the watershed was found to have facilitated the upstream spread of invasive New Zealand mud snail *Potamopygrun antipodarum* (Mahan et al. 2021).

We had little empirical evidence to parameterize novel interspecies interactions and increases to productivity based on changes to connectivity because post-removal monitoring efforts typically are not conducted over a long enough time frame (Magilligan et al. 2016a, 2016b; Foley et al. 2017). However, the data that exist suggest that fish populations are quick to colonize newly available habitats following barrier removal (Catalano et al. 2007; Burroughs et al. 2010; Whittum et al. 2023), but changes in abundance have been species-specific and the benefits of increased connectivity may take many years to develop (Sun et al. 2022). In the absence of standardized approaches to assess the response of migratory fish populations to barrier removal (Hayes et al. 2023) and some system-specific data, we developed an IBM framework and used information from other systems and expert knowledge to determine life history parameters, initial abundances, and changes to demographic parameters and species behavior following increases to connectivity. For these reasons, our results are difficult to validate and compare with other studies. However, this model was created as part of a larger decision analytic process for evaluating decisions for selective fish passage. We present the results of our models here, but note that gathering data to parameterize these models more fully would require implementation of a selective fish passage scenario, potentially through an adaptive management process (Walters 1986). An adaptive management process with consistent monitoring and iterative updating of this model could help reduce uncertainty in novel interspecies interactions and increases to productivity following changes to connectivity.

While developing the IBMs we faced numerous uncertainties, the first of which is a basic problem for modeling in general— determining the level of complexity needed in a model to accurately forecast the response of fish populations to barrier removal. While we opted for a model framework that has great time, effort, and data demands, the perceived accuracy of the model results is high (Hayes et al. 2023), and perception of accuracy can be important when working with stakeholders on decision making projects. We would caution, though, that that perceived accuracy might not always be warranted. Individual-based models could include many more considerations for individual variability and adaptive behavior than we have included in the present model and are generally considered to be the best approach for addressing questions that link fish populations to changes in habitat (Rose 2000; Hayes et al. 2009, 2023). However, including fine-scale considerations for food intake and bioenergetics (Madenjian and Carpenter 1991; DeAngelis et al. 1993; Sibly et al. 2013), changes in nutrient dynamics (Childress and McIntyre 2016), or predator–prey interactions (Giacomini et al. 2013), although important ecologically to fish growth and behavior, have not yet been shown to improve the predictive power of models used to forecast large-scale changes to fish populations following barrier removal or remediation. Our modeling framework, paired with a selective fish passage facility, provides a unique situation to evaluate the level of complexity needed in such a model. We developed our IBM framework to be easily modified by others with additional layers of complexity or simplified in instances when data are lacking for some individual-level processes, and based on the system-specific needs for model complexity.

The second uncertainty we faced was demographic parameterization, which might affect modeling results (i.e., parametric uncertainty; Williams et al. 2002). We were most uncertain about lake sturgeon parameters and relied heavily on data and parameters from outside the study area. Future data collected on lake sturgeon in the Boardman-Ottaway River could be used to improve parameterization of the model and increase the feasibility of including growth and movement submodels. Ideally, for all species, the movement submodel would be updated with telemetry data or empirical observations (e.g., redd counts) in the future because salmonids and sea lamprey aggregate in prime spawning locations to a greater or lesser degree than we have described in this model and these aggregations could have population-level effects. For walleye, the importance of considering the three different populations in the model is unknown. Genetic studies in the Boardman-Ottaway River suggest that three distinct populations exist (Gehri et al. 2021), but it is unknown if they would reproduce as an aggregate population under enhanced connectivity regimes. The genetic consequences of connectivity is an active area of IBM development in fisheries (Jager and Deangelis 2018) and could be a future direction for our model. Telemetry and genetic studies in the lower Boardman-Ottaway River are currently underway and those data could help improve considerations for the breeding ecology and spawning site choice of walleye in the model. For brook trout, we relied on assumptions about the relationship between intrinsic population growth rate and habitat quality to make predictions about possible outcomes. Because of their wide distribution in the watershed, we extrapolated abundance estimates to unsampled areas of the river and tributaries, but this model would be improved with additional spawner-recruit data and further monitoring and sampling of brook trout throughout the watershed. We used predicted Chinook salmon and steelhead smolt densities for a dam removal study on the Eel River in California (Cooper et al. 2020) as a good approximation of potential Chinook salmon and steelhead densities despite the study being from a western United States stream. A future improvement of this model would be to use smolt density estimates from Michigan streams following barrier removal or renovation. Finally, inter-species interactions were included in this model by varying demographic rates, but substantial uncertainty remains in how those interactions might affect population- and community-level outcomes under enhanced connectivity scenarios. Future research on interspecies interactions, especially between resident riverine fishes (e.g., walleye and brook trout) and novel migratory species (e.g., Pacific salmonids) could greatly improve parameterization of this model. However, it must be noted again that gathering data to parameterize these models more fully would require implementation of a selective fish passage scenario and long-term monitoring.

This model is not intended to be a final product, but rather an initial step in creating a framework that can be updated and modified as additional data are collected and more barrier removal projects are carried out. Future fish passage at FishPass will be controlled, and the predictions from these models can be evaluated in a real-world system and the parameters fine-tuned for future use on the Boardman-Ottaway River and other systems. Ideally, this framework will be used in the future to reduce uncertainty in the demographic parameters of the modeled species and improve predictive power of the model while addressing larger uncertainties such as the level of model complexity and data needed to forecast fish populations pre-barrier removal.

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APPENDIX: CHAPTER 2 SUPPLEMENTAL INFORMATION

Section A1: Lake Sturgeon IBM

The lake sturgeon IBM evaluated how changes in intrinsic growth rate (r) and number of fish passed in the first year (N_1) affected abundance over time. The model included key aspects of lake sturgeon population dynamics, namely low initial abundances, sexually dimorphic late-age maturation, periodic spawning behavior, and low rates of intrinsic population growth and natural mortality. There were only two management actions under consideration at FishPass that would influence lake sturgeon, the *status quo* and *full passage*, which occurred under all scenarios except the *status quo*; however, only the *full passage* scenario was modeled (Table 2.1) because the existing population is currently at low abundance and not expected to increase under the *status quo*. Therefore, we assumed that lake sturgeon abundance would only increase under enhanced connectivity scenarios.

Intrinsic growth rates (r) reported in the literature for lake sturgeon range between 0.049–0.138 in the Winnebago Lake system (WI) and 0.079–0.123 in the Great Lakes (Bruch et al. 2016). We used three intrinsic population growth rate values to represent a low growth scenario (r= 0.05), medium growth scenario (r= 0.10), and high growth scenario (r= 0.30) because there is high uncertainty about the population growth rate following increases to habitat availability. Similarly, we used three values of adult spawner abundance passed upstream in the first year (N_1= 10, 40, and 80). The value of 80 was chosen because this is the minimum viable population size for lake sturgeon suggested by Hayes et. al (2012). Currently, the lake sturgeon population in the Boardman-Ottaway River is estimated to be low with mature fish rarely observed below the Union Street Dam (Hay-Chmielewski and Whelan 1997; Travis 2019; UpNorthLive 2021); therefore, carrying capacity K, of the Boardman-Ottaway River watershed was set to 200, which is consistent with a small lake sturgeon population (Hayes et al. 2012). The model simulated a closed lake sturgeon population over a 250-year timeframe (about 10 generations), which is consistent with other lake sturgeon population models (Schueller and Hayes 2011).

The IBM tracked age, sex, recruitment, maturation, and mortality of individuals. A 5% annual mortality rate was assumed (Schueller and Hayes 2011; Hayes et al. 2012; Colborne et al. 2021) with all mortality occurring during the pre-spawn period (post-spawn mortality=0). Recruitment was calculated using a Ricker (1954) stock-recruitment function (Equation 2.5). Females spawned each year with a probability of 20% (equating to an average female spawning once every five years), and males spawned with a probability of 50% (equating to an average male spawning once every two years; Forsythe et al. 2012; Dammerman et al. 2019). If no females were present in the Boardman-Ottaway River and actively spawning, the recruitment for that year was set to zero, which was a possible occurrence because of low population abundances. Males and females were assumed to mature at age 15 and 20, respectively (Schueller and Hayes 2011).

Section A2: Chinook and Steelhead IBM

The Chinook salmon (CHS) and steelhead (RBT) IBMs included key aspects of salmonid life history such as a riverine juvenile stage of one to two years, high fishing mortality during migratory runs, and semelparous (CHS) or iteroparous (RBT) spawning. Initial abundances of CHS and RBT in the Boardman-Ottaway River were assumed to be zero and annual spawner abundances were determined by the management scenario. We evaluated three scenarios for CHS and RBT; the *status quo*, that restricted CHS and RBT to the lower Boardman-Ottaway River below Union Street Dam, the *low steel* and *low salmon* scenarios that allowed some passage of a reduced number of CHS or RBT, and the *full salmon* scenario that allowed full passage of CHS and RBT to the upper Boardman-Ottaway River (Table 2.3). To determine a full passage spawning run size, we used data from a weir operated on the Boardman-Ottaway River by the MDNR. For CHS, we averaged fall weir catch from 2016 to 2020. The Boardman-Ottaway River weir is only operational during the fall, so no catch data were available for RBT. Consequently, we used expert opinion to inform the initial RBT abundance parameters (Gary Whelan, MDNR, *personal written communication, March 16, 2021*). Annual spawning run sizes for CHS and RBT for each simulation were randomly drawn from uniform and truncated normal distributions, respectively (Table 2.3).

The CHS and RBT IBMs accounted for age, sex, recruitment, mortality, and migration of juveniles to Lake Michigan. At model initiation, all adult CHS and RBT were assumed to be age-4 and mature. Pre-spawn mortality accounted for the harvest of adults during spawning migration and was randomly drawn from a uniform distribution with lower and upper limits of 20 and 30% for CHS and 40 and 50% for RBT (Gary Whelan, MDNR, *personal written communication, March 16, 2021*). The post-spawn mortality was set to 100% for CHS (semelparous), and 10% for RBT (iteroparous). Movement of adult spawners was simulated by two mechanisms that allocated adult spawners to stream reaches based on 1) the stream reach network catchment area (i.e., larger stream reaches were preferred over smaller stream reaches) and 2) the habitat suitability reach score (i.e., higher reach scores are preferred over lower reach scores; Dean (2023); Equations 2.1 and 2.7). Each spawner allocation mechanism was given equal weight (P_A= 0.5). Maximum smolt density (*K*) was set to 0.2 fish/m^2 for CHS and 0.1 fish/m^2 for RBT (Cooper et al. 2020).

After adults were allocated to stream reaches, recruitment was calculated for each stream reach based on a modified Ricker function (Ricker 1954; Equation 2.6) where the intrinsic population growth rate, *r* varied among stream reaches by habitat suitability reach score (Table 2.3). For habitat suitability reach scores < 14, we assumed a low population growth rate of $r \sim TN(0, 0.05, 0, \infty)$, for reach scores 14 to 16, we assumed a medium growth rate of $r \sim TN(1, 0.10, 0, \infty)$, and for reach scores >16, we assumed the highest population growth rate, $r \sim TN(3, 0.20, 0, \infty)$. Reach scores were adjusted from Dean (2023) in the lower river spatial grid cells to account for observed low natural reproduction of migratory salmonids in the lower Boardman-Ottaway River below the site of former Sabin Dam, especially below Union Street Dam (Kalish et al. 2018). Therefore, reach scores below Union Street Dam were reduced to have the lowest rates of *r*.

After recruitment was calculated for each stream reach and recruits were allocated to the spatial grid, juveniles remained in the Boardman-Ottaway River and aged. Juveniles undergo smoltification and migration to Lake Michigan in the first (CHS) or second (RBT) year. All juveniles were expected to become smolts and migrate to Lake Michigan (i.e., no partial migration) and the number of juveniles leaving the river each year was recorded. Annual juvenile migration to Lake Michigan was equal to abundance but time-lagged based on life history (e.g., 1 year for CHS and 2 years for RBT). Once in Lake Michigan, they were lost to the model. Adults were not tracked in the lake, and each year new adult spawners were drawn from a distribution according to the management scenario. Although Pacific salmonids exhibit natal homing, annual spawner abundance would still be dictated by the management scenario (e.g., the number of fish allowed to pass upstream) and spawner abundance in the lower river is reduced via a weir operated by the MDNR. None of the considered management actions included unregulated passage of Pacific salmonids.

Section A3: Sea Lamprey IBM

The sea lamprey IBM combined aspects of simulation models developed by Jones et al. (2009), Robinson et al. (2013), and Lin and Robinson (2019). The model tracked annual spawner abundance, recruitment, larval growth and transformation, and migration to Lake Michigan. The model accounted for key aspects of sea lamprey life history including a protracted stream-dwelling larval stage (4–6 years), larval transformation to a lake-dwelling parasitic stage, and a semelparous life cycle. Initial abundance of sea lamprey in the upper Boardman-Ottaway River was assumed to be zero. To estimate the number of sea lamprey that might be present at the Union Street Dam in spring, we used the Boardman-Ottaway River sea lamprey trapping data from 2000 to 2020 (Table 2.8). The 20-year mean abundance of spawners from the Boardman-Ottaway River trapping index was 814 (SD=327.3); therefore, a full spawning run size of sea lamprey each year was drawn from a truncated normal distribution $\sim TN(814, 327.3, 0, 3000)$.

We evaluated three passage scenarios for sea lamprey (Table 2.4). The first scenario was the *status quo*, which restricted all sea lamprey to the lower Boardman-Ottaway River below Union Street Dam. For the second scenario, we assumed that sea lamprey were not completely blocked by the barrier, allowing a small number of spawning adults upstream each year, distributed as $\sim TN(50, 100, 0, 200)$. The third scenario was an open barrier scenario, which allowed the full spawning abundance of sea lamprey unrestricted access to the upper Boardman-Ottaway River.

Pre-spawn mortality was set to 10% whereas post-spawn mortality was set to 100% to represent semelparous spawning. Larvae and transformers were subjected to an annual mortality of 45% (Jones et al. 2009). The movement of adult spawning phase sea lamprey was simulated by two mechanisms which allocated spawners to stream reaches based on 1) the stream reach network catchment drainage area, such that larger stream reaches are preferred over smaller stream reaches, and 2) the estimated larval habitat area (Equation 2.2 and Equation 2.3). Each spawner allocation mechanism was given equal weight ($P_A = 0.5$). Our method of allocating spawners to stream reaches was modified from Jones et al. (2009) which used drainage area and larval abundance to allocate spawning sea lamprey to stream reaches. Once adults were allocated to stream reaches for spawning, recruitment was calculated for each stream reach based on a Ricker function modified from Jones et al. (2009; Equation 2.9).

After recruitment was calculated for each stream reach and recruits placed on the spatial grid, larval sea lamprey grew, transformed, or died. Growth was tracked in age (years) and the probability of transformation (T_P) increased with age (Table 2.4). No larvae remained in the river beyond 6 years, all age-6 sea lamprey transformed that year. Once a sea lamprey transformed, it left the Boardman-Ottaway River, migrated to Lake Michigan, and was lost to the model. Adults were not tracked in the lake and each year the number of adult spawners was drawn from the probability above distributions, because sea lamprey do not exhibit natal homing (Waldman et al. 2008). The spatial locations and individual attributes of fish in year 50, abundance, recruitment, number of spawners, and number of larval transformers migrating to Lake Michigan over time were recorded.

Section A4: Walleye IBM

The walleye IBM accounted for several genetically-distinct populations, sexually-dimorphic growth and maturation rates, and changes to carrying capacity, population growth rate, and length-at-age with the evaluated management scenarios. We evaluated four management scenarios for walleye (Table 2.7). The *low steel* and *low salmon* scenarios were considered equal because we assumed walleye production would not differ significantly between these scenarios.

Walleye were simulated as three genetically-distinct populations based on results from Gehri et al. (2021). Two distinct populations in Lake Michigan (Muskegon and Little Bay de Noc) were identified, as was one population above Union Street Dam in Boardman Lake that originated from an out-of-state source (Oneida Lake, NY). Estimated abundance and length frequency of walleye from surveys conducted in 2014 by the Little Traverse Bay Band of Odawa Indians and Grand Traverse Band of Ottawa and Chippewa Indians fishery management agencies were used to initialize the model (Brett Fessell, Grand Traverse Band of Ottawa and Chippewa Indians, unpublished data, 2021[2]). Based on a mark-recapture assessment in 2014, an abundance of 807 (SE=160) adult walleye was estimated for Boardman Lake with a density of 1.03 walleye/hectare. Walleye length in Boardman Lake ranged from 120 mm to 560 mm ($\bar{x} = 331$ mm). Gehri et al. (2021) found the Little Bay de Noc (GL2) population to have significantly larger lengths than the Muskegon (GL1) and Boardman Lake (BL) populations; however, the lengths of the GL1 & Boardman Lake populations were not significantly different. Little information exists regarding walleye abundance in the Great Lakes (near-Traverse Bays) and number of spawning adults. Preliminary data from the GLFC monitoring studies suggest a

[2] At the time of publication, data were not available from the Little Traverse Bay Band of Odawa Indians and the Grand Traverse Band of Ottawa and Chippewa Indians

spawning run of about 500–1000 individuals is likely to develop from Great Lakes populations (Reid Swanson, GLFC *personal oral communication, June 8, 2021*). Initial population size and average length of the three walleye populations are given in Table S1.

Annual natural mortality of adult walleye in Michigan is believed to be around 20%, with total annual mortality estimates ranging from 20 to 65% (Schneider et al. 2007; Roseman et al. 2010). Fishing mortality is believed to be around 10–30% for Great Lakes walleye populations, and the total annual mortality around 30–50% (Schneider et al. 2007). We therefore assumed an average annual adult mortality rate for adult walleye of 40%, with a possible range from 20 to 60%. We converted annual mortality to two additive instantaneous rates of pre- and post-spawn mortality (Table 2.5). Mortality rates did not vary among populations or the evaluated scenarios.

Movement of walleye was only implicitly modeled due to large uncertainty in future movement in the upper Boardman-Ottaway River. Walleye form spawning aggregations in specific areas of rivers and have been found to re-use spawning sites year after year (Schneider et al. 2007). However, there are currently no empirical data with which to determine future spawning locations and recently completed habitat suitability show walleye habitat is generally homogeneous throughout the watershed (Dean 2023; Figure S6). Based on expert opinion, migration of walleye was not expected to occur above the Forks region, at the confluence of the North and South Branch Boardman Rivers (Figure 1; Reid Swanson, GLFC, *personal oral communication, June 8, 2021*). Therefore, movement was included in the model, but governed solely by the population and assumed spawning aggregation sites. The movement of the Boardman Lake population was restricted to two spatial cells below the former Sabin Dam site, based on observed natural reproduction in this area (Brett Fessell, Grand Traverse Band of

Ottawa and Chippewa Indians, unpublished data, 2021[3]). The two Lake Michigan populations were randomly assigned to spatial cells from the former Sabin Dam site upstream to the Forks (Figure 1).

Recruitment for the genetically-distinct spawning aggregations was based on a Ricker (1954) stock-recruitment function (Equation 2.5) for each population separately on a river-wide scale. Following recruitment, recruits were spatially allocated using the same population-based functions; therefore, Boardman Lake recruits were placed below former Sabin Dam, and GL1 and GL2 recruits were randomly placed above former Sabin Dam. To simulate larval drift, recruits were placed either in Boardman Lake or Lake Michigan as a function of the population; Boardman Lake juveniles drift to and remain in Boardman Lake until maturation, and GL1 and GL2 juveniles drift to and remain in Lake Michigan until maturation.

Maturation schedules were sexually dimorphic and length-based but the same across all populations and evaluated scenarios. Females mature at age-4 and lengths > 330mm whereas males mature at age-3 and lengths > 300mm (Schneider et al. 2007; Bozek et al. 2011). We used Gehri et al. (2021) and the Little Traverse Bay Band of Odawa Indians and Grand Traverse Band of Ottawa and Chippewa Indians 2014 survey length frequencies, von Bertalanffy parameter estimates (L_∞ and k) for Muskegon and Little Bay de Noc walleye (Roseman et al. 2010), and statewide averages for walleye length-at-age (Schneider et al. 2000) to parameterize a low, medium, and high growth scenario (Table 2.5; Figure S8). Length-at-age was modeled using the von Bertalanffy growth function (Beverton and Holt 1957) and updated each year based on age, sex, and population.

[3] At the time of publication, data were not available from the Little Traverse Bay Band of Odawa Indians

Carrying capacity, intrinsic population growth rate, and length growth rate varied among the management scenarios (Table 2.7). We assumed length growth rate of walleye increased for all management scenarios, except the *status quo*, due to increased production and prey availability from passage of native catostomids, cyprinids, cottids, and non-native Pacific salmonids. The *status quo* restricts GL1 and GL2 walleye to the lower Boardman-Ottaway River below Union Street Dam, while the Boardman Lake population has access to the entire upper watershed and spawns between Boardman Lake and the former Sabin Dam site. For the *status quo*, we assumed the slowest growth in length rate (Appendix: Figure A2.8) and that all populations were at carrying capacity (Boardman Lake = 800; GL1 = 500; GL2 = 500). The average population density estimate for Michigan inland lakes is 0.89 walleyes/hectare (range= 0–3.72; Schneider et al. 2007). Boardman Lake is approximately 136 hectares (Kalish et al. 2018); therefore, a carrying capacity, *K*, of approximately 800 (0.97 walleyes/hectare) is consistent with the State average density. For the Great Lakes walleye populations, we assumed the density in West Grand Traverse Bay was much lower (0.04 walleyes/hectare). Grand Traverse Bay is 71,742 hectares and West Grand Traverse Bay is roughly half that area (35,871 hectares). Therefore we calculated a carrying capacity, *K*, of approximately 8900 walleye for West Grand Traverse Bay; however, we assumed stocking would reduce that carrying capacity by about 90% to 1000, or 500 for each population. From 2018 to 2022, between 75,000 and 166,667 walleye fingerlings were stocked annually into Grand Traverse Bay (https://www.dnr.state.mi.us/fishstock/). We assumed that all three genetically-distinct populations were near carrying capacity and intrinsic growth rate was randomly generated from a truncated normal distribution, $r \sim TN(0.5, 0.3, 0, 1)$.

For the *native only* scenario we assumed that the carrying capacity for all populations would increase due to improved access to spawning habitat and greater productive potential of the entire system due to the increase in productivity of other native fishes such as catostomids (Swanson et al. 2021). We assumed the carrying capacity of all populations would double so that K=1600 (~1.9 walleye/hectare) for the Boardman Lake population and K=1000 for each of the Lake Michigan populations. The intrinsic growth rate, r for all populations increased by 20% (Table 2.7) and growth in length was assumed to follow the medium scenario (Figure S8).

For the *low steel* and *low salmon* scenarios, we assumed the increased passage and production of RBT and CHS in the Boardman-Ottaway River would increase the carrying capacity of all populations by 20% (Table 2.7) and length-at-age would increase above the state-average for all populations, due to feeding on smolts and other novel native fish production (Appendix: Figure A2.8). We assumed no change in the walleye populations between the *low steel* and *low salmon* scenarios.

For the *full salmon* scenario, we assumed that full connectivity would result in the highest K and highest length-at-age growth scenario for all populations (Table 2.7; Figure S8). Carrying capacity increased to 2200 (~2.6 walleye/hectare) for the Boardman Lake population and for each Great Lakes population. This carrying capacity for Great Lakes populations assumed 50% of the West Grand Traverse Bay carrying capacity (calculated at 0.04 walleyes/hectare) was allotted to natural reproduction and therefore, we assumed stocking was reduced in West Grand Traverse Bay because of increased natural reproduction.

Section A5: Brook Trout IBM

The brook trout (BKT) IBM incorporated aspects of the species' life history including high early life mortality and sexually-dimorphic maturation rates, and evaluated how changes to habitat suitability, carrying capacity, and growth in length rate affected the BKT population over time. We evaluated four management scenarios for BKT (Table 2.7). Because no migratory BKT currently occupy the system, none of the scenarios included BKT passage at the Union Street Dam. We assumed that the passage of Pacific salmonids would negatively affect BKT populations. This is well documented in the literature (Fausch and White 1986; Zorn et al. 2020). Pacific salmonid density was found to be negatively related to BKT density, and BKT abundance was lower in Great Lakes accessible reaches (Zorn et al. 2020). Therefore, we assumed scenarios that included salmonid passage would result in lower population growth rate and carrying capacity, a restricted spatial distribution, and reduced length-at-age. We assumed BKT production would not differ between the low steel and low salmon scenarios.

To determine BKT initial abundance, we used data from the MDNR Boardman River Assessment (Kalish et al. 2018) and the 2015 Brown Bridge removal evaluation from Michigan Trout Unlimited (Burroughs 2016; Table 2.8). The MDNR and Michigan Trout Unlimited estimated BKT density at nine sites from 1960 to 2019 (Table S2), and we used these values to initialize BKT populations for the sampled sections and extrapolated to unsampled areas half-way up- and downstream to the next sampled section. The mean young-of-year density for BKT reported in Michigan was 265 BKT/hectare (Zorn et al. 2018) and Boardman-Ottaway River density estimates ranged from 0 to 166 BKT/hectare (Burroughs 2016; Kalish et al. 2018; Hettinger 2020). Brook trout densities are known to be low in the lower river (Kalish et al. 2018) and we assumed no BKT were present below Sabin Dam. Density estimates from the Beitner

Road site were extrapolated downstream to above Sabin Dam (Figure 2.1; Table S2). We assumed the density of all unsampled tributaries was lower than the mainstem Boardman-Ottaway River and averaged 50 BKT/hectare (SD=15). Initial abundance and spatial distribution did not vary among the evaluated management scenarios.

Previous research has found that annual survival rates of age-0 BKT decreased following RBT introductions in a Lake Superior tributary (Peck 2001; Zorn et al. 2018); however, more recent research reported BKT survival was not significantly different for Great Lakes accessible and non-accessible streams (Zorn et al. 2020). Zorn et al. (2020) reported annual survival rates of BKT to age 1 and age 2 of approximately 25% (10–40%) and 5% (2–10%), respectively, in land-locked streams when Brown Trout were present. Although the Boardman-Ottaway River was not included in this study, surveys have found BKT older than age-2 are very rare in the Boardman-Ottaway River (Kalish et al. 2018; Tonello 2022). Annual mortality of age-1 BKT was assumed to average 75% and range from 60 to 90%. Age-2 mortality was assumed to average 95% and range from 90 to 98%. Annual mortality rates were converted to two additive instantaneous rates for pre- and post-spawn mortality events. Age-1 pre- and post-spawn mortality was normally distributed with a mean of 0.50 and standard deviation of 0.05. Age-2 morality was normally distributed with a mean of 0.78 and standard deviation of 0.025. Mortality rates did not vary between sexes or among the evaluated management scenarios.

The movement of adult spawning BKT was simulated by two mechanisms that allocated spawners to stream reaches based on 1) the stream reach network catchment drainage area score, which scores smaller stream reaches higher (more preferred) than larger stream reaches, and 2) the habitat suitability reach score (Equation 2.4).

The catchment drainage area score spawner allocation mechanism was given 75% of the weight (P_A= 0.75) and the habitat suitability reach score allocation mechanism was given 25% weight (1–P_A=0.25). For all scenarios the downstream limit of BKT movement was the former Sabin Dam site. After movement was simulated, recruitment was calculated for each spatial cell (Ricker 1954; Equation 2.5). We varied K and r among the scenarios assuming changes in these parameters would occur following changes to connectivity. For the *status quo* scenario, the carrying capacity of each spatial cell was calculated as the product of the area and mean statewide estimated BKT young-of-year/hectare (265 BKT/hectare; Zorn et al. 2018; Equation 2.8).

For the *native only* scenario, we assumed the carrying capacity would increase 25% above the statewide average meaning the carrying capacity in the model was increased by a factor of 1.25 relative to the *status quo* (Equation 2.8, f_A = 1.25). For the scenarios that included salmonid passage, the carrying capacity and habitat suitability reach scores were decreased. For the *low steel* and *low salmon* scenarios, the carrying capacity was reduced by 25% of the status quo (Equation 2.8, f_A = 0.75) and habitat suitability reach scores were reduced by 10% in the mainstem Boardman-Ottaway River up to the Forks region. For the *full salmon* scenario, the carrying capacity was reduced by 25% of the status quo (Equation 2.8, f_A = 0.75) and habitat suitability reach scores were reduced by 10% for the entire watershed. The habitat suitability reach scores were used to simulate BKT movement and therefore, the reduction in habitat suitability reach scores affects BKT movement.

We related BKT habitat suitability reach scores (Table 2.7) to r such that reach scores <=11 have a low r= ~TN(0.5, 0.05, 0, ∞), reach scores >11 and <15 have a medium r= ~TN(1, 0.1, 0, ∞), and reach scores >=15 have a high r= ~TN(1.6, 0.2, 0, ∞). Because the habitat suitability reach scores are reduced for the salmonid passage scenarios, the *r* values are also reduced accordingly. Following recruitment, recruits remained in natal stream reaches until maturation. Maturity was assumed to be sexually dimorphic and age-based with 100% of males and 80% of females becoming mature by age-1 (McFadden et al. 1967; Zorn et al. 2018). All age-2 fish were assumed to be mature. The maturation schedule did not change among the management scenarios.

Length-at-age of BKT was assumed to vary by management scenario. Zorn et al. (2020) reported that length-at-age was significantly lower for age-0 BKT in Great Lakes accessible reaches (Zorn et al. 2020). This has been attributed to BKT being competitively inferior to Brown Trout (Fausch and White 1981; Waters 1983) and Pacific salmonids (Fausch and White 1986; Fausch 2018). Although BKT >200 mm and age-2 are very rare in the Boardman-Ottaway River watershed, reported length-at-age has been near or above the statewide average (Kalish et al. 2018). We parameterized a low, medium, and high growth scenario (Appendix: Figure A2.9; Kalish et al. 2018; Zorn et al. 2018; Hettinger 2020). Under the status quo, BKT length-at-age followed the statewide average (medium growth scenario). For the native passage only scenario we assumed BKT length-at-age increased to above the statewide average (high scenario). For the low steel and low salmon scenarios length-at-age followed the medium growth scenario, and for the full salmon scenario BKT length-at-age was assumed to decrease below the statewide average (low growth scenario; Appendix: Figure A2.9). Parameters used in the BKT IBM are given in Table 2.6.

Table A2.1. Initial adult abundance and mean length (mm), and juvenile abundance for three distinct walleye *Sander vitreus* populations. Juvenile abundance listed is the number in each year class 1–4. Standard deviation of adult length in parentheses.

	Adult		Juvenile
Population	**Abundance**	**Length**	**Abundance**
Boardman Lake	800	430 (40)	200
Great Lakes 1 (GL1) – Muskegon	200	495 (70)	50
Great Lakes 2 (GL2) – Little Bay de Noc	200	660 (50)	50

Table A2.2. Brook trout *Salvelinus fontinalis* survey data and average densities used to parameterize initial abundance and spatial distribution. Stream segments are listed from downstream (top) to upstream (bottom).

Stream segment	Years Sampled	Ave. Density (SD)	Reference
Below Sabin Dam	2006	0	Kalish et al. (2018)
Beitner Rd.	2005	328 (NA)	Kalish et al. (2018)
Shumsky's Landing	1985–2005	2.8 (1.3)	Kalish et al. (2018)
Below Brown Bridge	1985–2010; 2015	10.2 (3.5)	Kalish et al. (2018); Burroughs (2016)
Brown Bridge Impoundment	2015	410 (NA)	Burroughs (2016)
Scheck's	1985–2005; 2015	265.4 (299.5)	Kalish et al. (2018); Burroughs (2016)
Ranch Rudolf	1960–2019	286.1 (167.5)	Hettinger et al. (2020)
Forks	1985–1994	37 (42.9)	Kalish et al. (2018)
South Branch	1985–2002	144.3 (101.1)	Kalish et al. (2018)
North Branch	1985–2002	260.1 (219.0)	Kalish et al. (2018)

Table A2.3. Predicted median lake sturgeon *Acipenser fulvescens* abundance (and interquartile range), percent increase in abundance, percent of simulations that ended in year-250 with populations equal to zero, and the percent of simulations that had abundances above the minimum viable population (MVP) size of lake sturgeon (MVP= 80) for each scenario for the final 5 years of simulation.

Scenario	**Abundance**	**% increase**	**% sims. with N_{250}= 0**	**% sims. with N_{250} >= MVP**
r=0.05, N_1=10	0 (0–0)	–100%	99.9	0
r=0.1, N_1=10	0 (0–0)	–100%	98.5	0.1
r=0.3, N_1=10	528 (9–699)	5180%	22.6	70.2
r=0.05, N_1=40	0 (0–0)	–100%	75.9	0
r=0.1, N_1=40	8 (0–37)	–80%	29.0	9.1
r=0.3, N_1=40	754 (670–843)	1785%	0	100
r=0.05, N_1=80	4 (0–16)	–95%	27.8	1.5
r=0.1, N_1=80	74 (43–114)	–7.5%	1.1	45.3
r=0.3, N_1=80	765 (678–856)	856.3%	0	100

Table A2.4. Predicted median population abundance with interquartile range in parentheses for lake sturgeon *Acipenser fulvescens*, Chinook salmon *Oncorhynchus tshawytscha*, steelhead *Oncorhynchus mykiss*, walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis* for the last five years of simulation.

FishPass Scenario	Lake Sturgeon	Chinook Salmon	Steelhead	Walleye Boardman Lake	Walleye Lake Michigan	Brook Trout
Status Quo	–	965.0 (209–1170)	1531.5 (978–2656)	1536.0 (1274.0–1846.0)	1477.0 (1261.0–1717.0)	58139.0 (48905.0–70175.0)
Native Only	528* (9–699)	965.0 (209–1170)	1531.5 (978–2656)	2438.0 (1947.5–2992.3)	3108.5 (2714.0–3587.0)	73678.5 (62070.8–87578.3)
Low Steel	528* (9–699)	965.0 (209–1170)	17743.5 (13017–25127)	2710.0 (2184.0–3343.0)	3387.5 (2893.0–3954.0)	43762.0 (36275.5–53254.8)
Low Salmon	528* (9–699)	8320.0 (5265–12304)	17743.5 (13017–25127)	2710.0 (2184.0–3343.0)	3387.5 (2893.0–3954.0)	43762.0 (36275.5–53254.8)
Full Salmon	528* (9–699)	14760.5 (8351–23370)	38187.0 (28106–55013)	3166.0 (2514.8–3902.0)	6057.5 (5195.0–7010.3)	19905.0 (15874.8–25018.3)

* high growth–low initial abundance scenario.

Table A2.5. Predicted percent change in median population abundance compared to the *status quo* scenario with interquartile range in parentheses for lake sturgeon *Acipenser fulvescens*, Chinook salmon *Oncorhynchus tshawytscha*, steelhead *Oncorhynchus mykiss*, walleye *Sander vitreus*, and brook trout *Salvelinus fontinalis* for the last five years of simulation.

FishPass Scenario	Chinook Salmon	Steelhead	Walleye Boardman Lake	Walleye Lake Michigan	Brook Trout
Native Only	–	–	58.7% (26.8–94.8)	110.5% (83.8–142.9)	26.7% (6.8–50.6)
Low Steel	–	1058.6% (750.0–1540.7)	76.4% (42.2–117.6)	129.4% (95.9–167.7)	-24.7% (-37.6 – (-8.4))
Low Salmon	762.2% (445.6–765.4)	1058.6% (750.0–1540.7)	76.4% (42.2–117.6)	129.4% (95.9–167.7)	-24.7% (-37.6 – (-8.4))
Full Salmon	1429.6% (1175.0–2321.8)	2393.4% (1735.2–3492.1)	106.1% (63.7–154.0)	310.1% (251.7–374.6)	-65.8 (-72.7– (-56.7))

Table A2.6. Median and interquartile range of juvenile sea lamprey *Petromyzon marinus* abundance and larval transformers for the last five years of simulation for each scenario.

Scenario	Abundance	Transformers
Status Quo	8147 (4331 – 12575)	183 (78 – 406)
Reduced Passage	2384 (653 – 10671)	34 (6 – 201)
Full Passage	24106 (7640 – 64145)	401 (78 – 1457)

Table A2.7. Median length (mm) and interquartile range of walleye *Sander vitreus* and brook trout *Salvelinus fontinalis* for each scenario in the final year of simulation.

FishPass Scenario	Walleye Boardman Lake	Walleye Lake Michigan	Brook Trout
Status Quo	260.6 (133.2–397.4)	293.0 (154.9–415.4)	154.8 (141.7–168.4)
Native Only	291.8 (153.9–408.3)	279.7 (161.7–400.0)	180.0 (167.3–194.0)
Low Steel	340.5 (179.4–470.9)	409.8 (316.7–500.3)	154.9 (141.8–168.2)
Low Salmon	340.5 (179.4–470.9)	409.8 (316.7–500.3)	154.9 (141.8–168.2)
Full Salmon	327.6 (236.4–452.1)	419.9 (281.8–550.9)	131.0 (117.5–144.9)

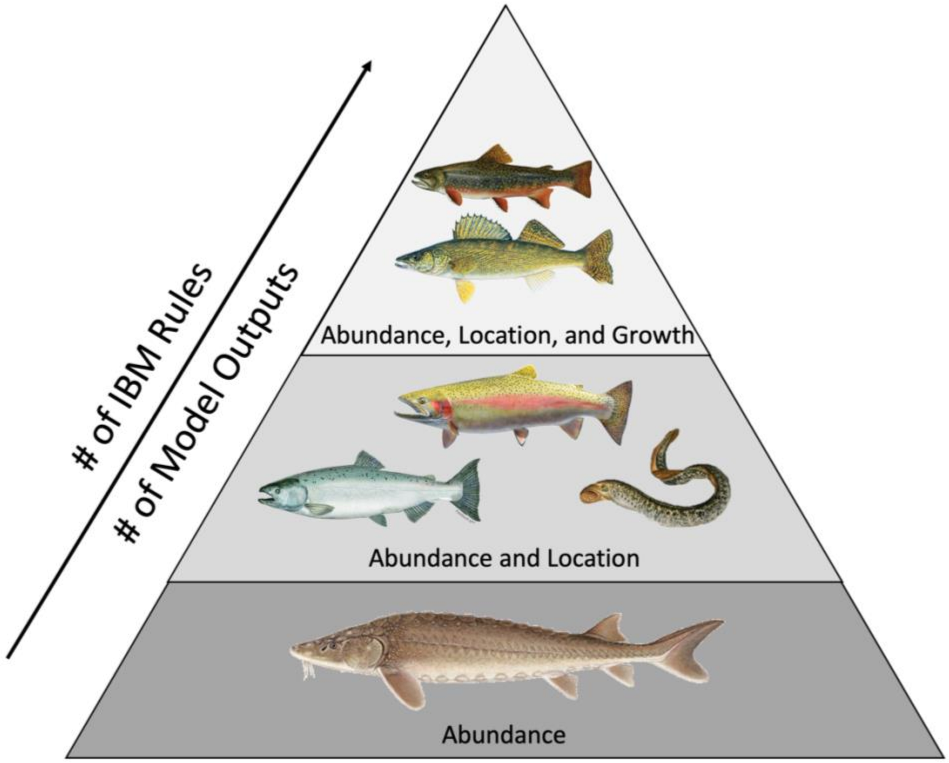

Figure A2.1. Individual based model hierarchy conceptual diagram, which depicts the range of complexity in species-specific models, for six species. Models were created to predict the changes in abundance (all species), location (Chinook salmon, rainbow trout, sea lamprey, walleye, and brook trout), and growth (walleye and brook trout) under different scenarios of selective fish passage at the Union Street Dam on the Boardman-Ottaway River in Michigan, USA.

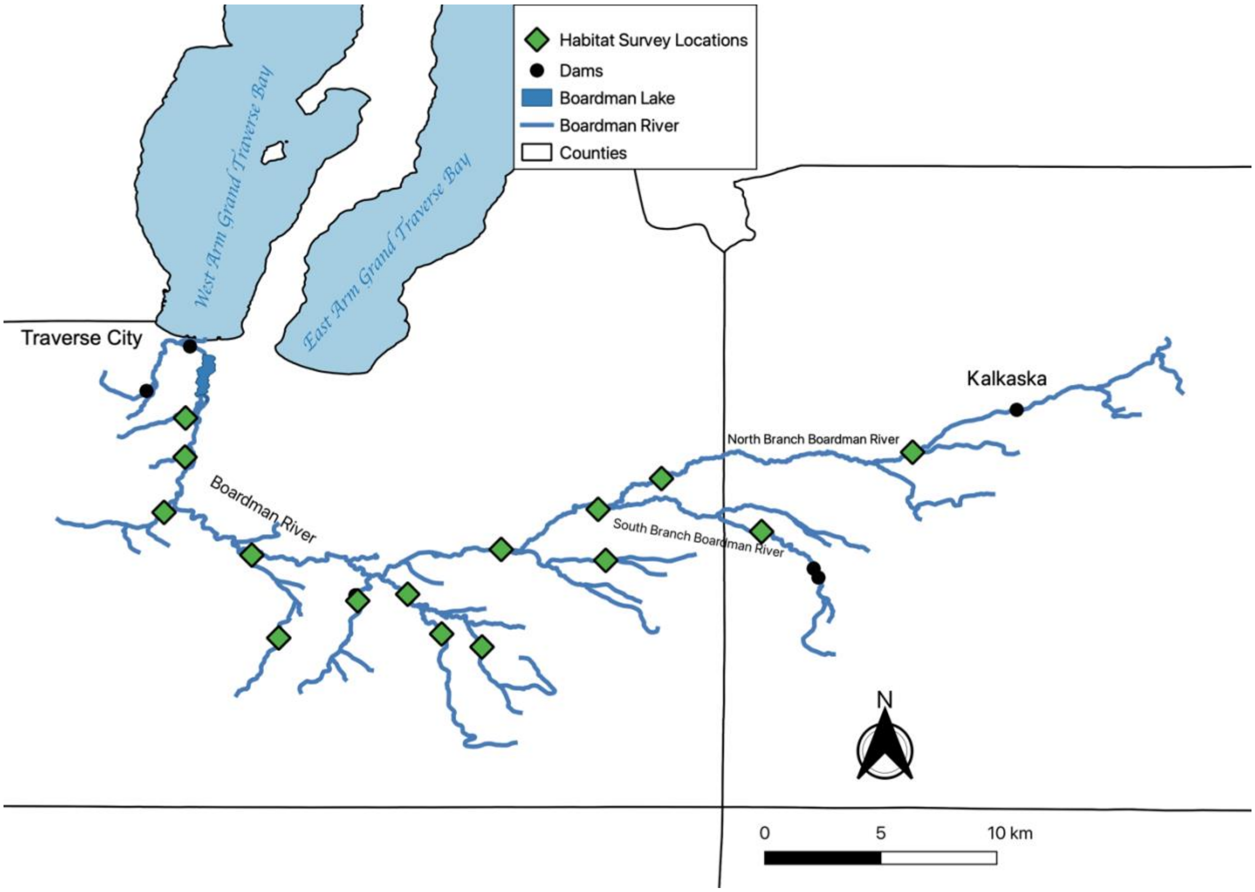

Figure A2.2. Habitat sampling field sites within the Boardman-Ottaway River watershed in Michigan, USA. Habitat was sampled in 2018 and 2019.

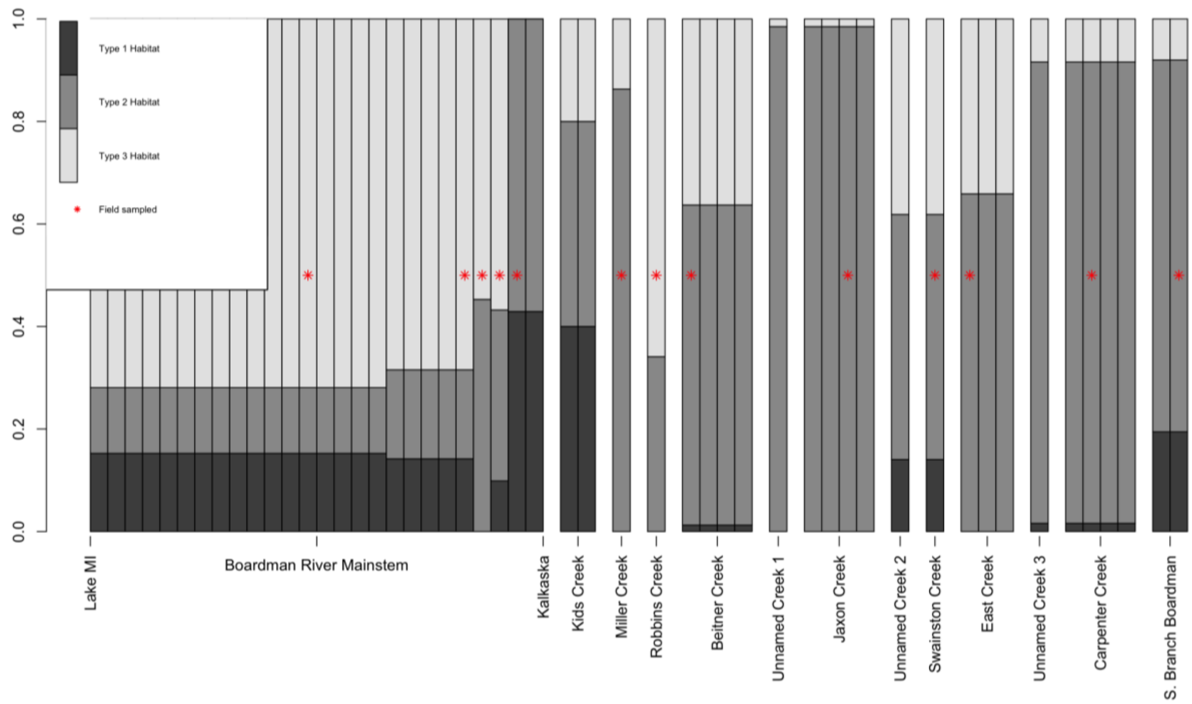

Figure A2.3. Sea lamprey *Petromyzon marinus* habitat types and proportions by stream reaches within the Boardman-Ottaway River watershed in Michigan, USA, used for an individual based model. Figure shows the number of cells per tributary and how sea lamprey habitat data were extrapolated among sites. Jackson and Bancroft creeks are not shown but were sampled; however, those creeks are not included in the spatial grid used for modeling because they are tributaries of East Creek.

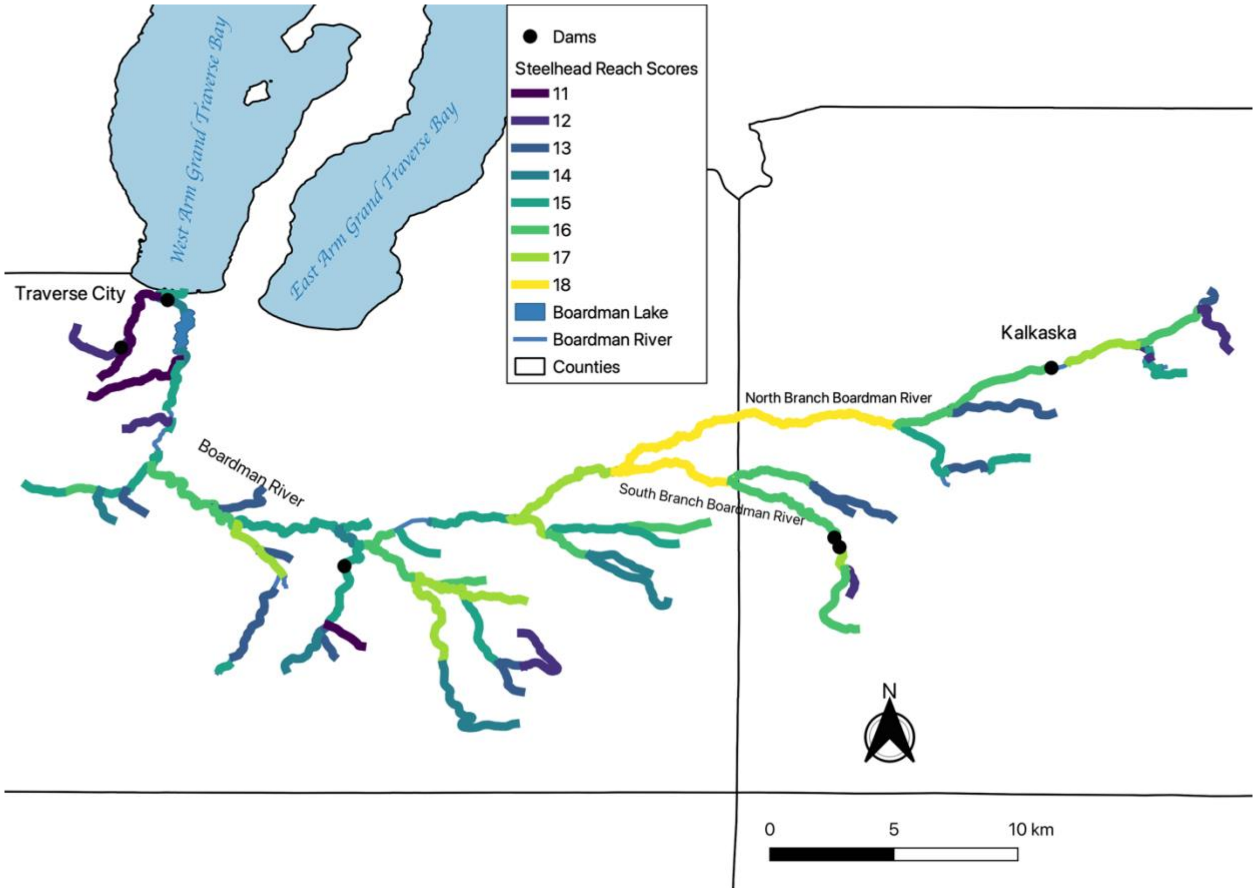

Figure A2.4. Rainbow trout (steelhead) *Oncorhynchus mykiss* habitat suitability reach scores for the Boardman-Ottaway River, Michigan, USA (Dean 2023).

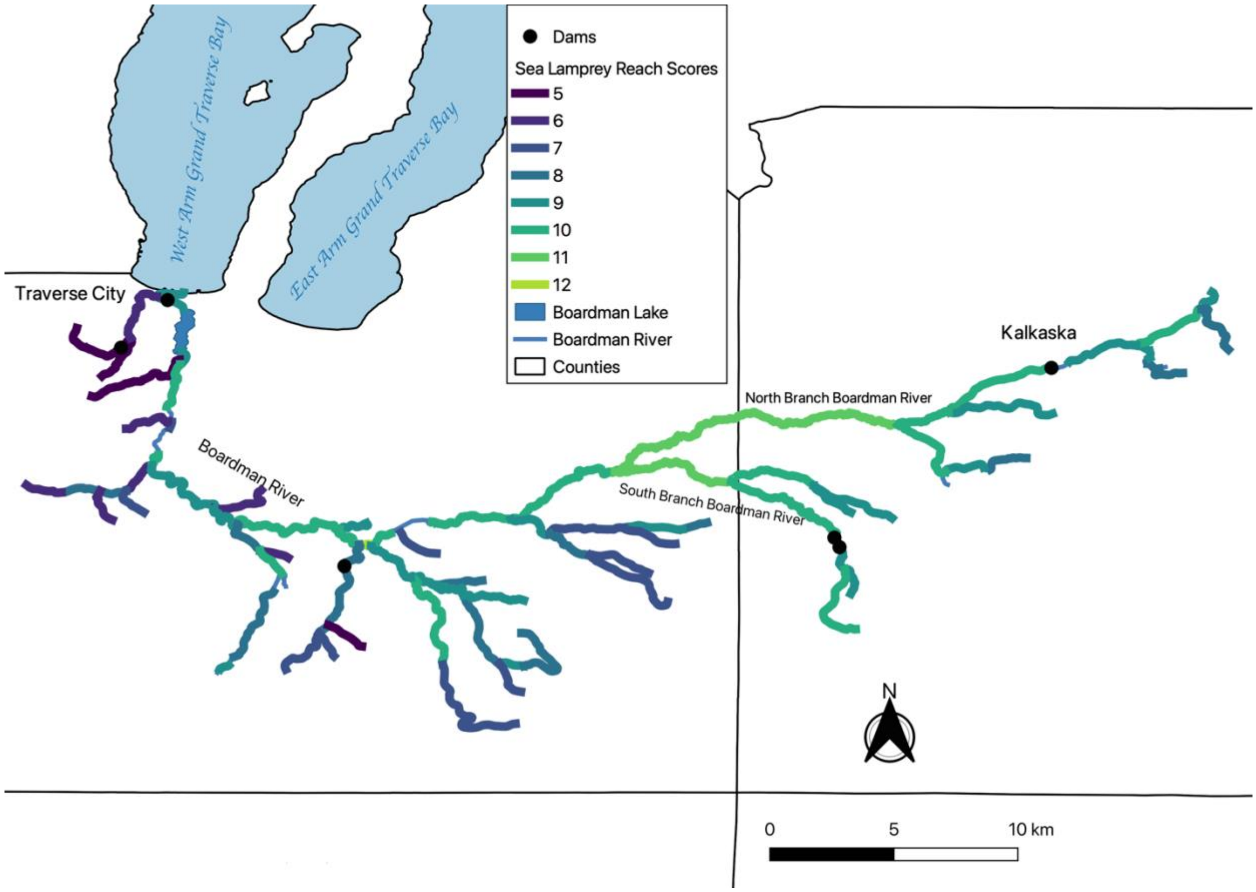

Figure A2.5. Sea lamprey *Petromyzon marinus* habitat suitability reach scores for the Boardman-Ottaway River, Michigan, USA (Dean 2023).

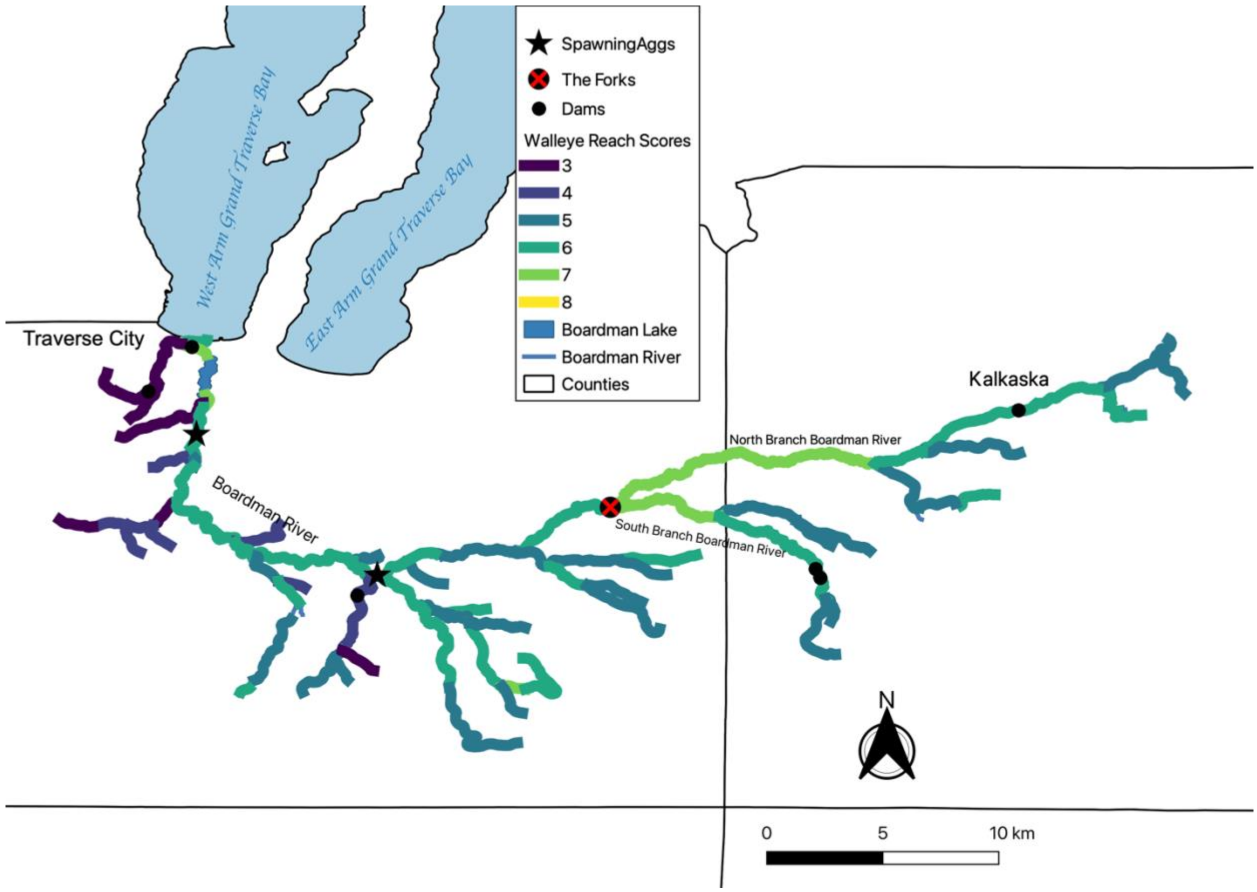

Figure A2.6. Walleye *Sander vitreus* habitat suitability reach scores for the Boardman-Ottaway River, Michigan, USA (Dean 2023).

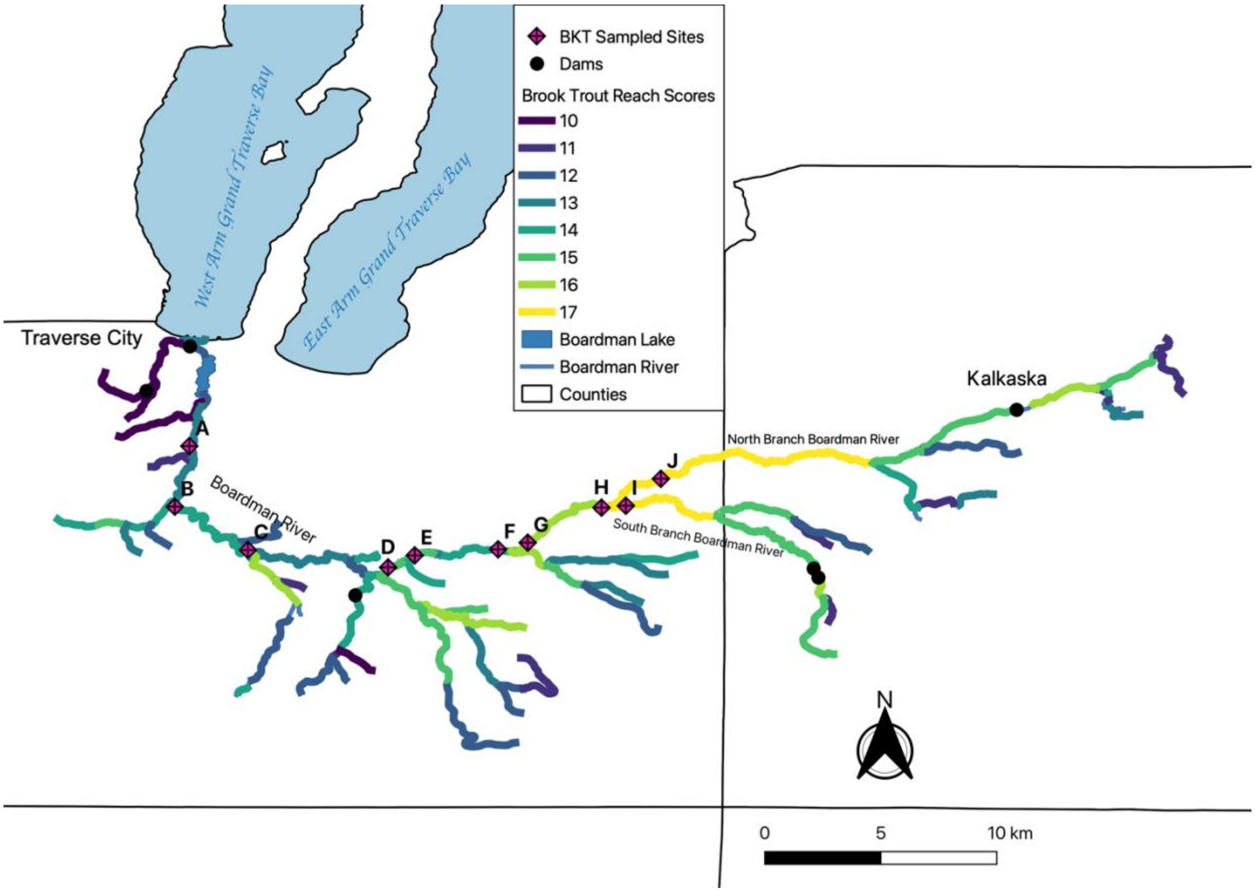

Figure A2.7. Brook trout *Salvelinus fontinalis* habitat suitability reach scores for the Boardman-Ottaway River, Michigan, USA, from Dean (2023). Letters (A–J) indicated brook trout (BKT) survey sites. A= below Sabin Dam; B= Beitner Rd.; C= Shumsky's Landing; D= below Brown Bridge; E= Brown Bridge impoundment; F= Scheck's campground; G= Ranch Rudolf; H= the Forks; I= South Branch; J= North Branch.

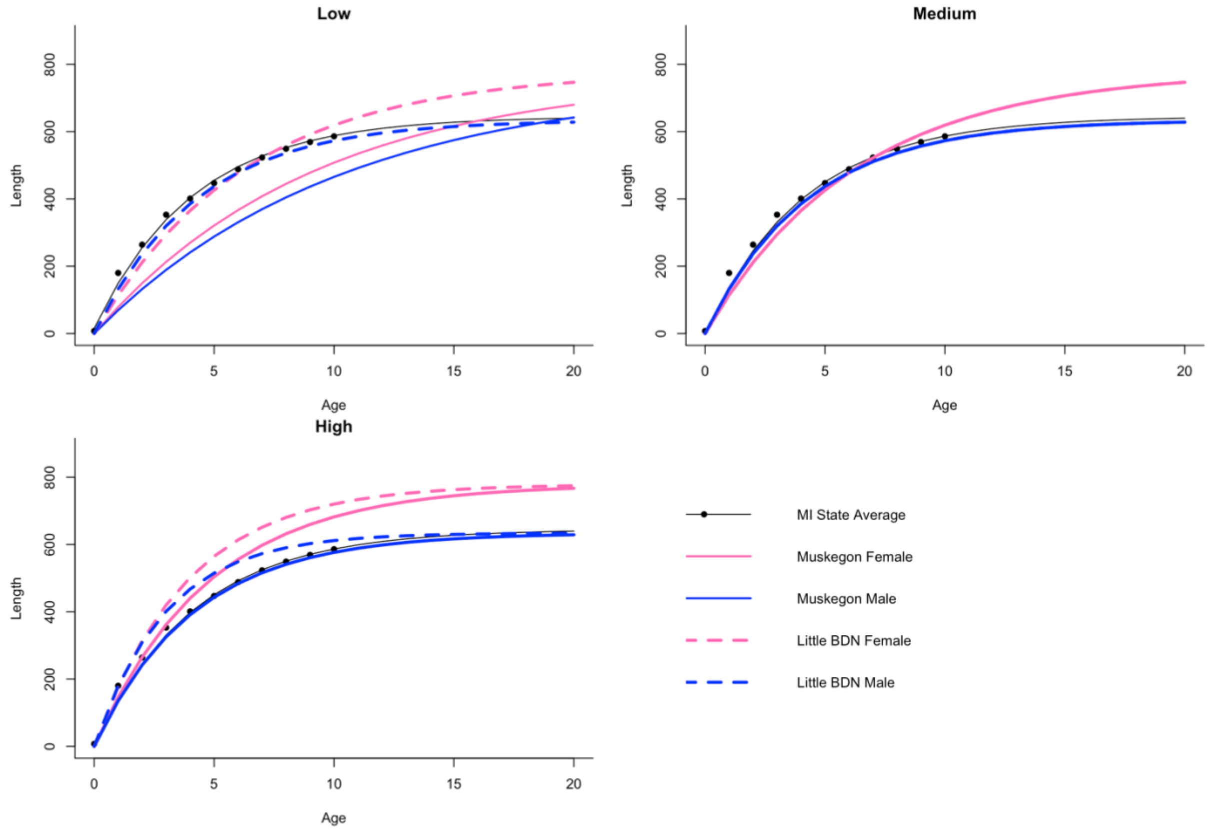

Figure A2.8. Walleye *Sander vitreus* length-at-age growth scenarios used to parameterize an individual based model for predicting changes in walleye productivity under different fish passage scenarios in the Boardman-Ottaway River, Michigan, USA. Boardman Lake and Great Lakes 1 (GL1) populations grow at Muskegon rates and Great Lakes 2 (GL2) grow at Little Bay de Noc rates.

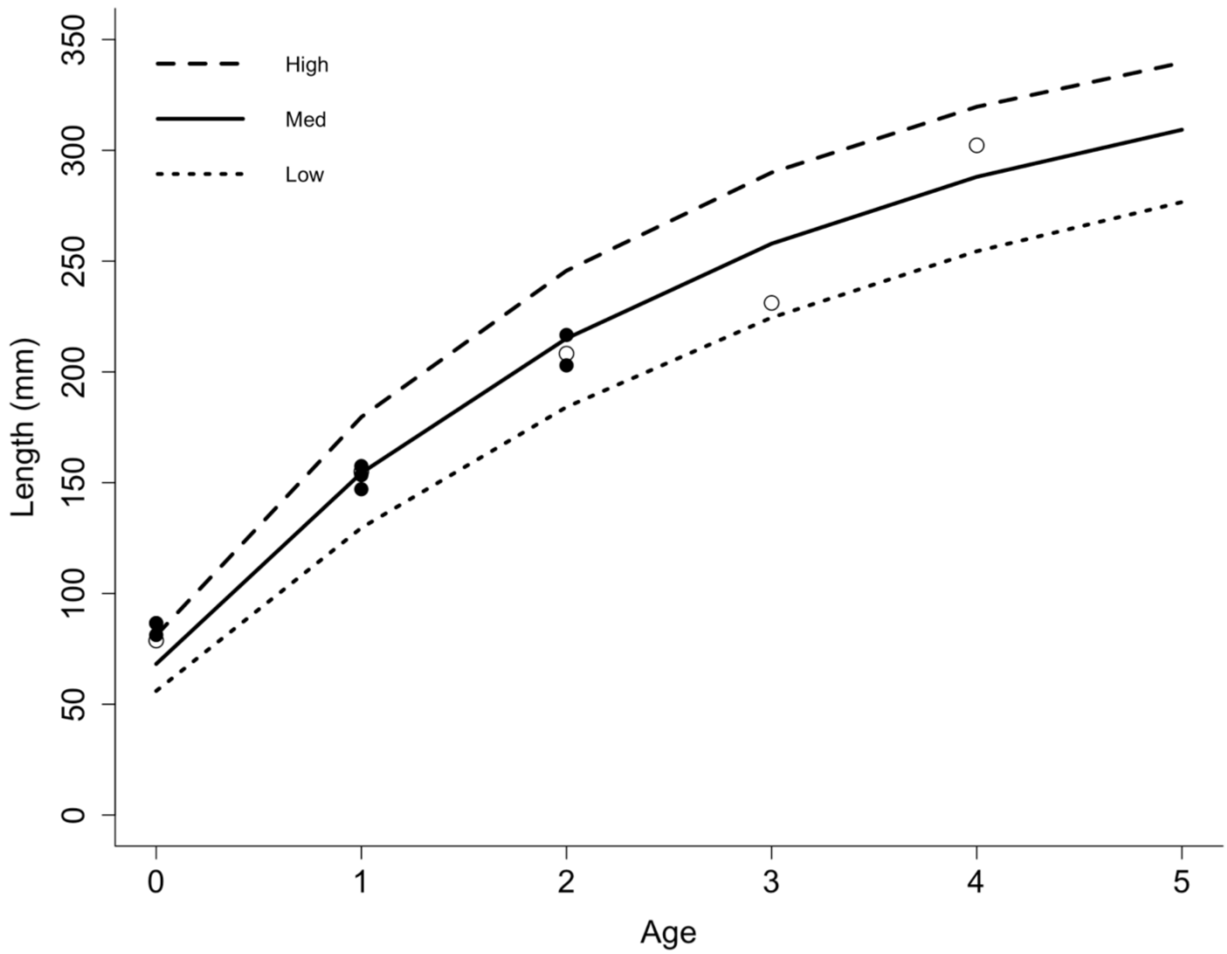

Figure A2.9. Brook trout *Salvelinus fontinalis* length-at-age growth scenarios used to parameterize an individual based model for predicting changes in brook trout productivity under different fish passage scenarios in the Boardman-Ottaway River, Michigan, USA. Open circles are estimated length-at-age from (Zorn et al. 2018), and closed circles are length-at-age estimates from (Hettinger 2020).

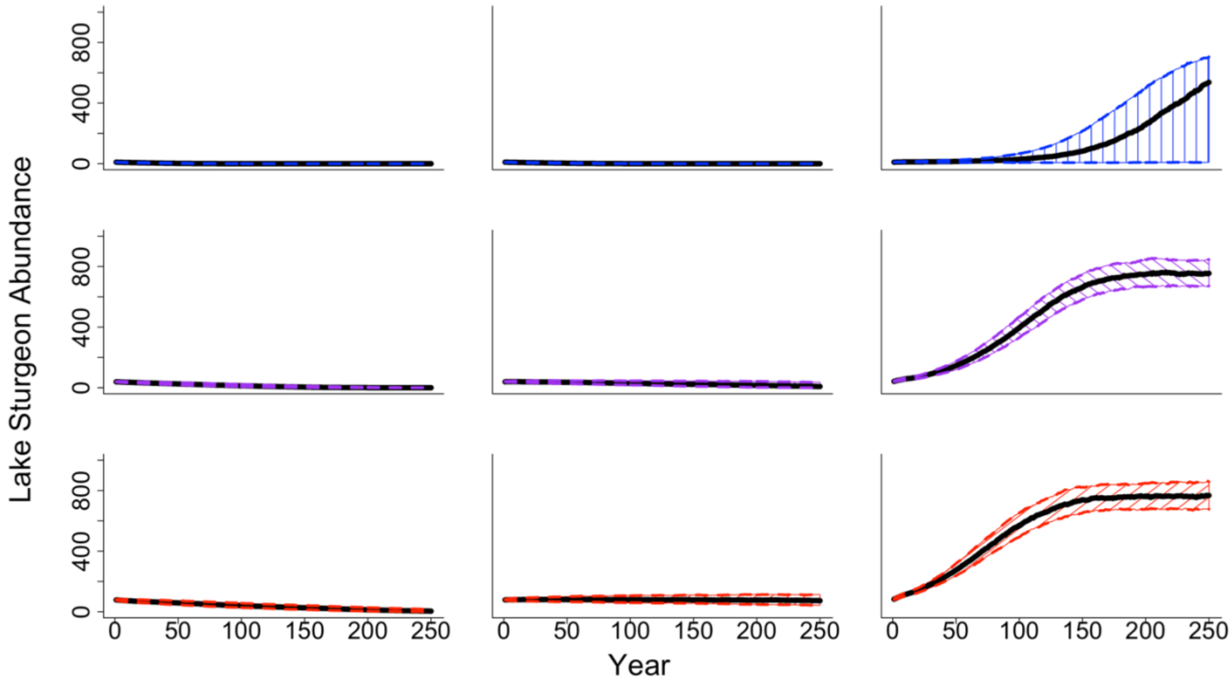

Figure A2.10. Predicted lake sturgeon *Acipenser fulvescens* population abundance over 250 years in the Boardman-Ottaway River, Michigan, USA. Panels are arranged top to bottom with increasing initial abundance, N_1 (10–blue, 40–purple, 80–red), and left to right with increasing intrinsic growth rate, *r* (0.05, 0.1, 0.3). Black lines are the median of all simulations while shaded portions show the interquartile range.

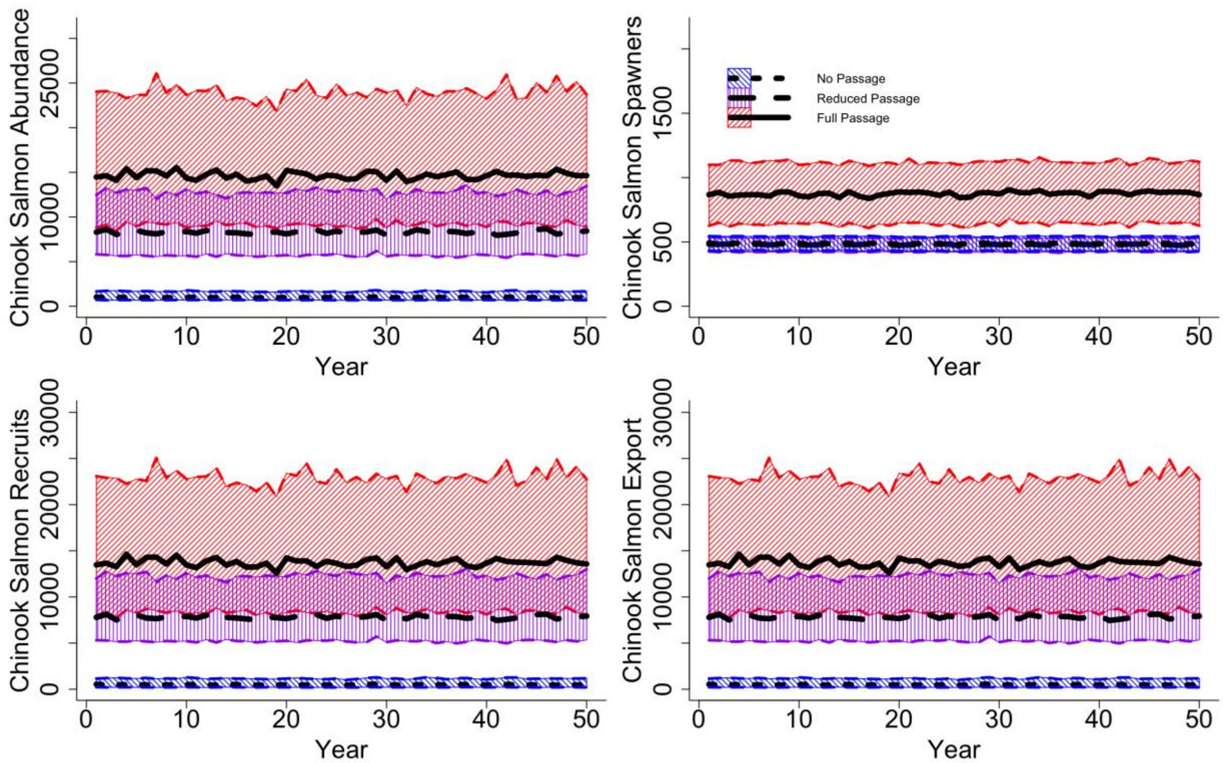

Figure A2.11. Median and interquartile range of annual abundance, spawners, recruits, and juvenile export of Chinook salmon *Oncorhynchus tshawytscha* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no passage of Chinook salmon above the Union Street dam (“no passage”), passage of some Chinook salmon above the dam (“reduced passage”; n=500), and full passage of Chinook salmon above the dam (“full passage”; n=1200).

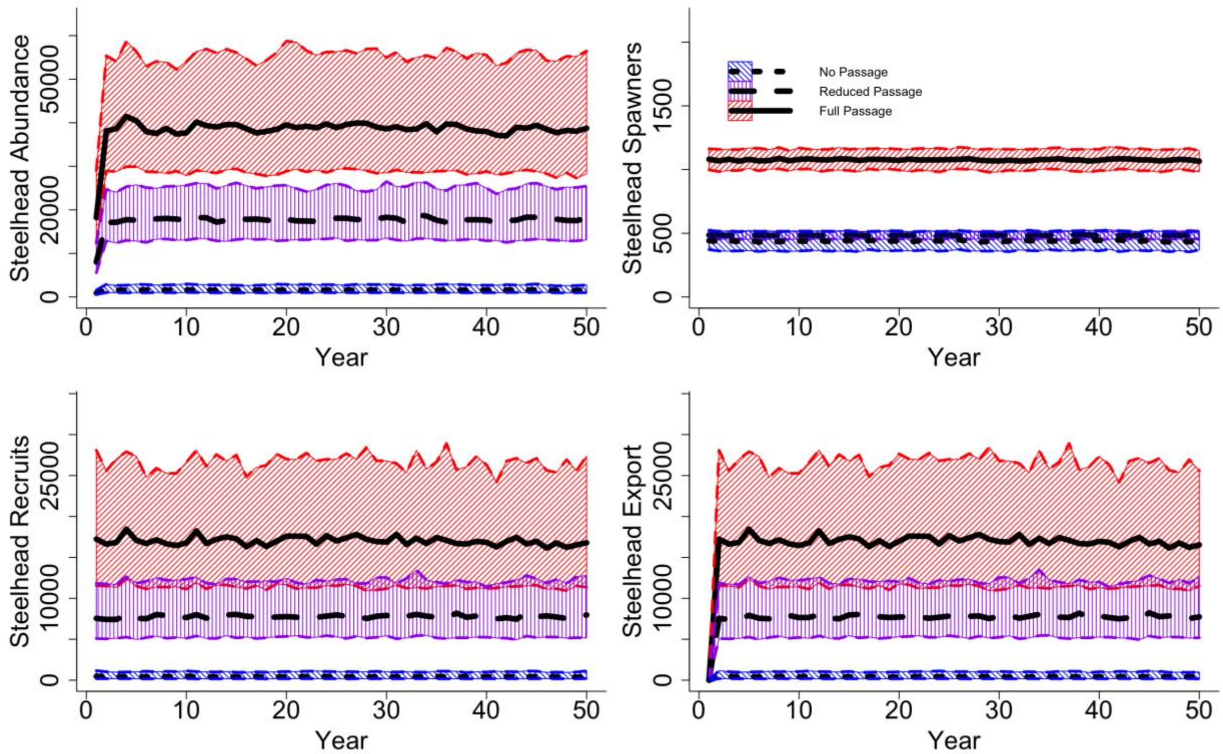

Figure A2.12. Median and interquartile range of annual abundance, spawners, recruits, and juvenile export of steelhead *Oncorhynchus mykiss* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no passage of steelhead above the Union Street dam (“no passage”), passage of some steelhead above the dam (“reduced passage”; n=800), and full passage of steelhead above the dam (“full passage”; n=2000).

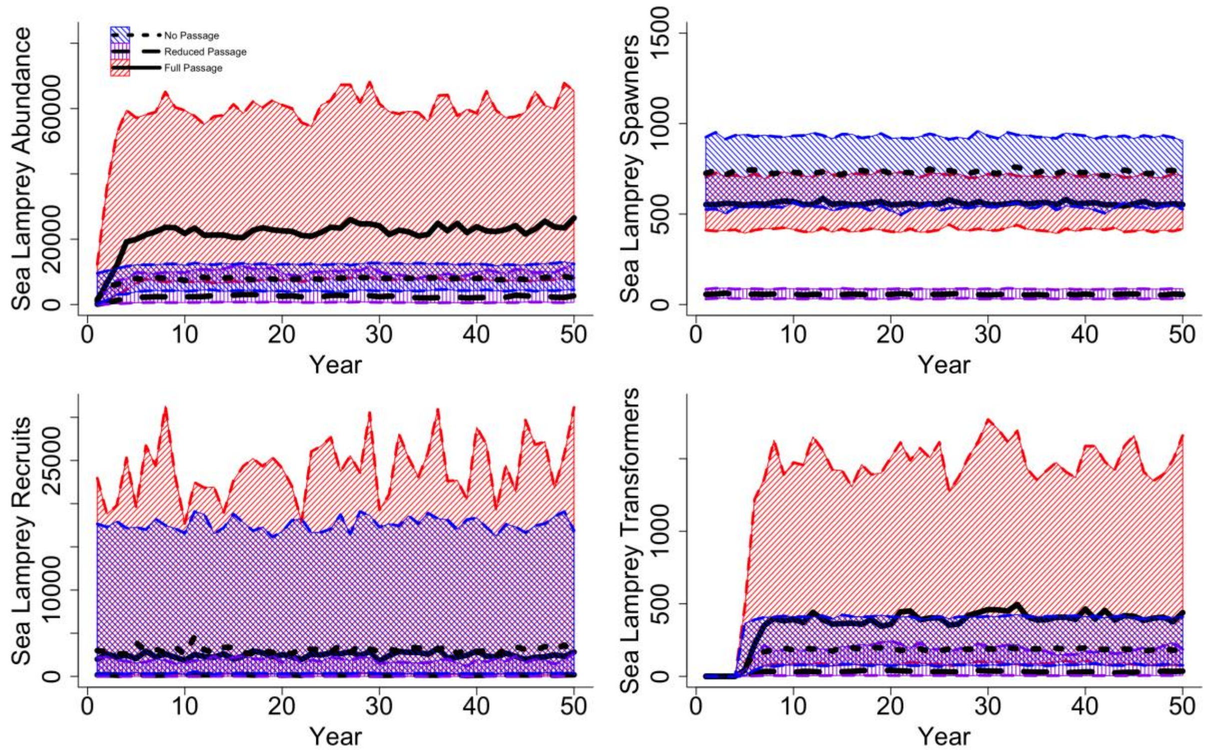

Figure A2.13. Median and interquartile range of annual abundance, spawners, recruits, and larval transformers of sea lamprey *Petromyzon marinus* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no passage of sea lamprey above the Union Street dam ("no passage"), passage of some sea lamprey above the dam ("reduced passage"; n=50), and full passage of sea lamprey above the dam ("full passage"; n=800).

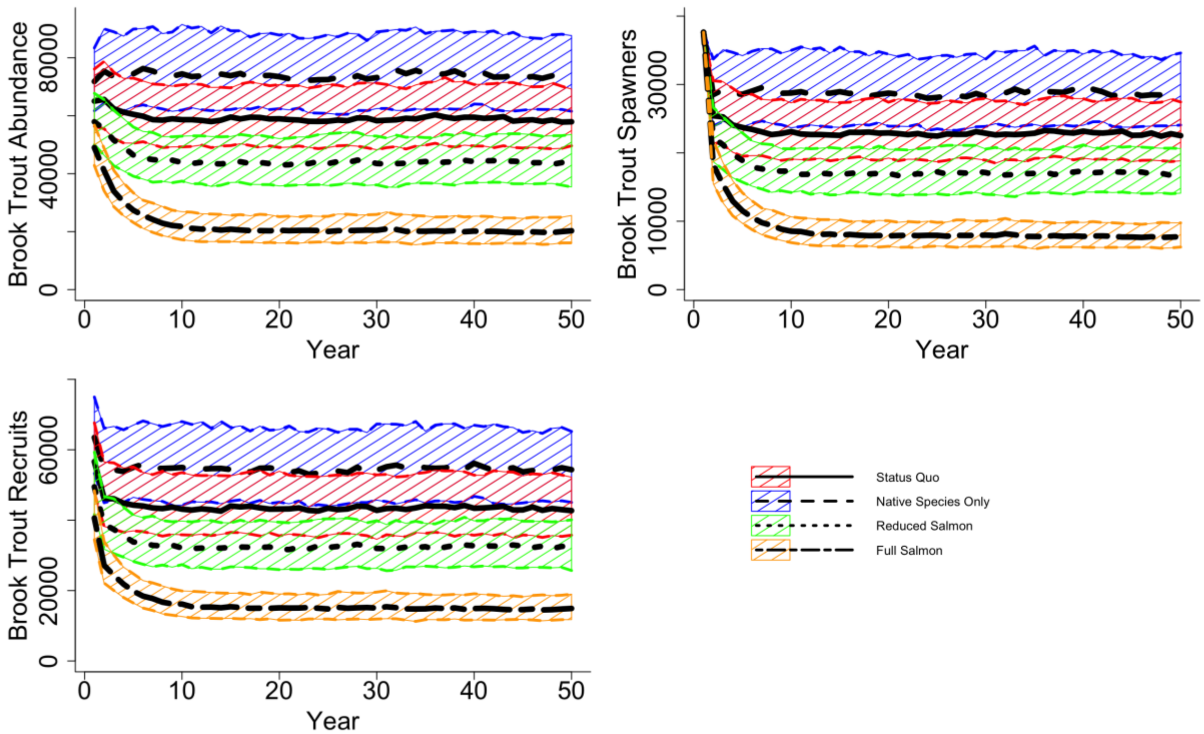

Figure A2.14. Median and interquartile range of annual abundance, spawners, recruits of brook trout *Salvelinus fontinalis* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no change in the current fish community in the river (“status quo”), passage of only native Great Lakes species (“native species only”), some passage of Pacific salmonids (“reduced salmon”; 500 Chinook salmon, 800 steelhead), and full passage of Pacific salmonids (“full salmon”; 1200 Chinook salmon, 2000 steelhead).

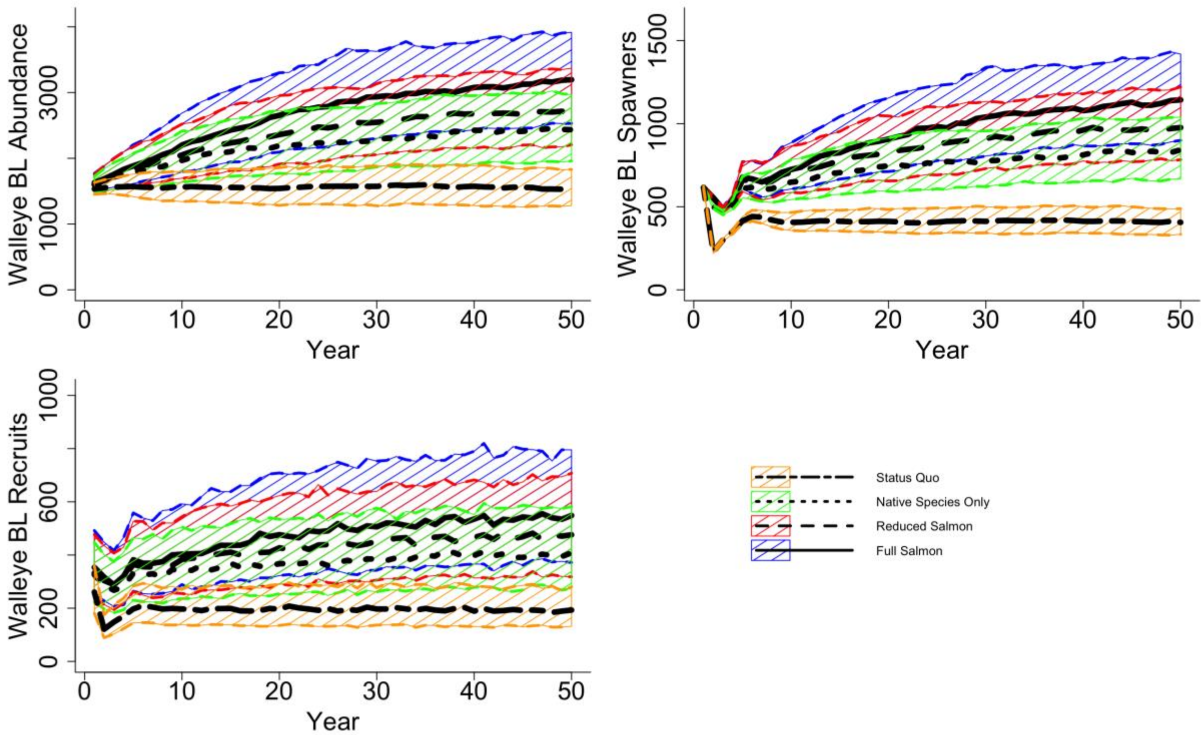

Figure A2.15. Median and interquartile range of annual abundance, spawners, recruits of Boardman Lake walleye *Sander vitreus* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no change in the current fish community in the river ("status quo"), passage of only native Great Lakes species ("native species only), some passage of Pacific salmonids ("reduced salmon"; 500 Chinook salmon, 800 steelhead), and full passage of Pacific salmonids ("full salmon"; 1200 Chinook salmon, 2000 steelhead).

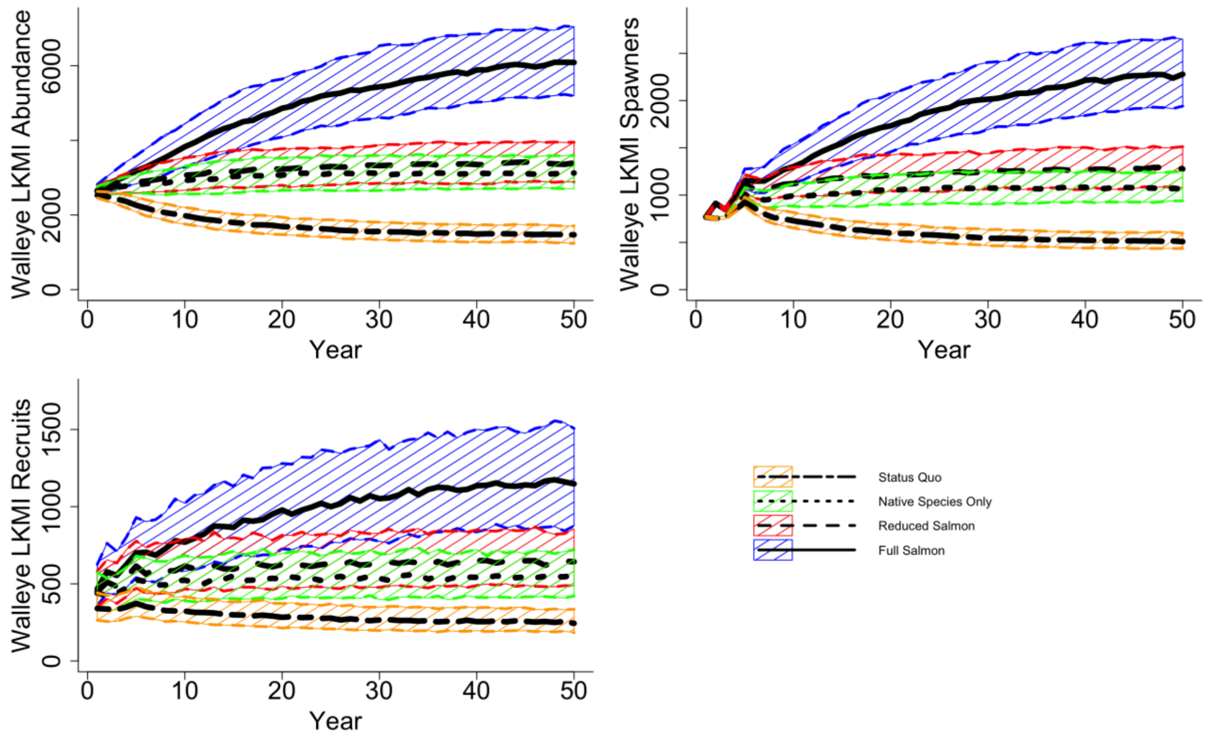

Figure A2.16. Median and interquartile range of annual abundance, spawners, recruits of Great Lakes (GL1 and GL2 combined) walleye *Sander vitreus* for scenarios of fish passage evaluated in the Boardman-Ottaway River, Michigan, USA, over a 50-year time period. Scenarios included no change in the current fish community in the river ("status quo"), passage of only native Great Lakes species ("native species only), some passage of Pacific salmonids ("reduced salmon"; 500 Chinook salmon, 800 steelhead), and full passage of Pacific salmonids ("full salmon"; 1200 Chinook salmon, 2000 steelhead).

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CHAPTER 3: EVALUATING STOCKING SCENARIOS FOR THE RESTORATION OF LAKE STURGEON IN THE BOARDMAN-OTTAWAY RIVER

Abstract

Lake sturgeon *Acipenser fulvescens* is an IUCN Red List endangered species and native to the Laurentian Great Lakes. Anthropogenic barriers have blocked lake sturgeon migration to stream spawning grounds and contributed to significant population declines. With barrier removals increasing in North America, lake sturgeon may once again have access to spawning grounds; however, because of low current population abundance, long life history, and slow population growth rate, additional management actions may be necessary. The Grand Traverse Band of Ottawa and Chippewa Indians (GTB) are considering a lake sturgeon stocking program in the Boardman-Ottaway River to enhance restoration efforts. To aid the GTB in decision making, we modified an existing individual-based model to include various stocking rates and demographic parameters to evaluate the potential time to reach a target population (n= 750 mature adults). We also evaluated the sensitivity of the model to certain demographic and habitat parameters to elucidate key areas of uncertainty that might influence restoration actions. We found the median time to reach the target abundance was between 31 and 91 years, depending on the stocking strategy. The time to reach the target population size decreased as number of fingerlings stocked annually increased and as duration of stocking increased. When the number of fingerlings stocked annually randomly varied, median time to reach the target population was 51 years. This research shows that the rehabilitation of Great Lakes lake sturgeon populations will take time and likely require multiple targeted rehabilitation actions including stocking, a harvest moratorium, protection to spawning individuals, and habitat restoration.

Introduction

Increasing connectivity of rivers via barrier removal and renovation is an important step in rehabilitating populations of migratory fishes (Foley et al. 2017; Bellmore et al. 2019). In the Great Lakes, a "connectivity conundrum" (Zielinski et al. 2020) exists where barriers act as a control method for invasive sea lamprey *Petromyzon marinus* but simultaneously limit access for desirable native species such as lake sturgeon *Acipenser fulvescens*, walleye *Sander vitreus*, and suckers *Catostomid spp.* Selective fish passage, in which desirable species are passed above a barrier and undesirable and invasive species are blocked, is a novel approach to enhancing connectivity. The FishPass project is a Great Lakes Fishery Commission (GLFC) project (http://www.glfc.org/fishpass.php) which aims to provide selective fish passage on the Boardman-Ottaway River, Michigan, USA, to allow passage of desirable species while simultaneously blocking undesirable and invasive species. The renovation of Union Street Dam via FishPass is the capstone of a nearly 20-year, watershed scale rehabilitation project. A major goal of the FishPass project is to allow passage of lake sturgeon to spawning habitat to aid in population restoration, and stakeholders and rightsholders are eager to see lake sturgeon return to the watershed. However, for species like lake sturgeon that have long life spans and low population growth rates (Bruch et al. 2016), immediate population-level responses to barrier removal or renovation may be unrealistic and targeted management actions to enhance these populations could be warranted.

Lake sturgeon, a slow growing, long lived fish (up to 154 years; Bruch et al. 2016), which matures at late age (age-15 for male, age-20 for females; Schueller and Hayes 2011), is native to the Great Lakes basin and is an IUCN Red List endangered species (Haxton and Bruch 2022). The species travels long distances in its lifetime, and it is a potamodromous, fluvialdependent species that uses Great Lakes tributaries for spawning. In the Great Lakes, males are believed to spawn every other year and females are believed to spawn every three to seven years (Hay-Chmielewski and Whelan 1997). Lake sturgeon exhibit high levels of natal homing and low levels of straying (Homola et al. 2012). The Boardman-Ottaway River is probably a historical spawning ground for lake sturgeon (Hay-Chmielewski and Whelan 1997), but the current population status in the this river is uncertain and probably very low (Hay-Chmielewski and Whelan 1997; Hayes et al. 2012). This can at least in part be attributed to the existence of the Union Street Dam which blocks upstream migration and access to spawning habitat in the upper river. Occasionally, lake sturgeon have been observed below the Union Street Dam in spring (Travis 2019; UpNorthLive 2021) indicating a spawning run could develop. Furthermore, recent fisheries surveys below Union Street Dam captured a reproductively mature female lake sturgeon, which further provides evidence that a spawning run of this species could develop if fish could pass above the dam (B. Fessell, Grand Traverse Band of Ottawa and Chippewa Indians, *personal communication*).

Lake sturgeon is an important species to stakeholders and rightsholders in the Boardman-Ottaway River watershed, especially the Grand Traverse Band of Ottawa and Chippewas Indians (GTB), who hold joint decision-making authority for fish passage along with the Michigan Department of Natural Resources (MDNR). Despite high levels of uncertainty in the current status of a lake sturgeon population in the watershed and future response of lake sturgeon populations to barrier removal or renovation, the rehabilitation of lake sturgeon is an important objective for Boardman-Ottaway River stakeholders and rightsholders (see Chapter 1). Therefore, an individual-based model (IBM) was developed to evaluate the possible outcomes of increased lake sturgeon passage on the Boardman-Ottaway River (see Chapter 2). The results of this modeling effort indicated that lake sturgeon rehabilitation was not likely to occur given current population abundance and intrinsic growth rate estimates. The chances for successful restoration were most probable with initial populations greater than 40 and intrinsic population growth rates (r) greater than 0.1. Schueller and Hayes (2011) found that in the Great Lakes, initial lake sturgeon populations greater than 80 had little chance of extinction, but extinction risk increased to 50% for initial populations of 40, and it was nearly 100% for initial populations of less than 15. Recently, Vaugeois et al. (2022) found that in the Great Lakes, management actions which increase egg survival rate or included stocking resulted in the fastest population growth and recovery compared to management actions that increased juvenile or adult survival rate; however, the study was focused on the population-level effects of contaminants. Finally in Manitoba, Canada, Nelson et al. (2022) found somatic growth rate of lake sturgeon affected the time to reach recovery target abundance and the stocking of yearlings was shown to increase the probability of population recovery. Therefore, management actions which increase lake sturgeon abundance in the watershed or increase population growth rate are likely to expedite recovery to target levels.

Due to the importance of lake sturgeon to stakeholders and rightsholders in the Boardman-Ottaway River watershed, management actions are being considered to enhance lake sturgeon rehabilitation. The GTB is exploring lake sturgeon stocking scenarios to increase the chances and reduce the time to recover the population. To aid the GTB in identifying potential outcomes of various lake sturgeon stocking scenarios, we modified the lake sturgeon IBM (see Chapter 2) to evaluate the population-level effects of stocking fingerlings and yearlings into the Boardman-Ottaway River. The objectives for this research were to evaluate the potential response of lake sturgeon populations in the Boardman-Ottaway River to various stocking scenarios, better understand uncertainties in that response, and help decision makers understand the commitment and resources needed to rehabilitate the long-lived, slow growing lake sturgeon. Furthermore, this work serves as an example of the flexibility and potential future applications of the IBM framework, and ease of modifications to incorporate new submodels and refine existing submodels.

Methods

Stocking Scenarios

We developed several stocking scenarios based on genetics stocking guidelines (Welsh et al. 2010) and discussions with representatives of the GTB. The stocking guidelines provide recommendations for annual stocking rates assuming 25 years of stocking to achieve a target population of 750 adult individuals. The guidelines base the number to stock on annual survival rates of fingerlings, yearlings, and adults. Based on discussions with the GTB, the management informed annual survival rates were set at 50% for fingerlings, 75% for yearlings, and 95% for adults (Brett Fessell, Dan Mays, and Sean Leask, Grand Traverse Band of Ottawa and Chippewa Indians, *personal communication*), which according to the stocking guidelines, equates to annual target stocking of 280 fingerlings or 140 yearlings. The GTB has a goal to stock 1000 fingerlings annually and has a 'sturgeon in the classroom' program which results in 0–6 yearlings for stocking annually.

We developed several additional stocking scenarios (Table 3.1) that allowed us to evaluate the sensitivity of the outcomes to uncertainties in availability of fish for stocking and stage-based survival. The first scenario is the "goal scenario" of 1000 fingerlings annually. The second scenario is half of the goal scenario (500 fingerlings annually). The third scenario is the recommended stocking level based on Welsh et al. (2010) given the management informed estimates for stage-based survival rates in the Boardman-Ottaway River. The fourth scenario is half of the recommended level. Four additional scenarios were designed to evaluate how variation in the duration of stocking affected population response to reach the target population level. Lastly, one scenario was included to evaluate the stochasticity in outcomes when the number stocked annually was randomly drawn from a uniform distribution from 0 to 1000.

Table 3.1. Lake sturgeon *Acipenser fulvescens* stocking scenarios. Bold and italic text is intended to highlight the variation among the different stocking scenarios compared to the goal scenario.

			Number stocked		
Scenario	**Assess**	**Description**	**Fingerling**	**Yearling**	**Duration**
Stock.1000	Number Stocked	Goal	1000	~U(0, 6)	25 years
Stock.500	Number Stocked	½ goal	***500***	~U(0, 6)	25 years
Stock.280	Number Stocked	Recommended	***280***	~U(0, 6)	25 years
Stock.140	Number Stocked	½ recommended	***140***	~U(0, 6)	25 years
Time.5	Time of Stocking	Recommended 5 yrs.	1000	~U(0, 6)	***5 years***
Time.10	Time of Stocking	Recommended 10 yrs.	1000	~U(0, 6)	***10 years***
Time.15	Time of Stocking	Recommended 15 yrs.	1000	~U(0, 6)	***15 years***
Time.20	Time of Stocking	Recommended 20 yrs.	1000	~U(0, 6)	***20 years***
Stoch.Stock	Uncertainty	Stochastic Stocking	***~U(0, 1000)***	~U(0, 6)	25 years

ODD Model Description

The model description follows the ODD (Overview, Design concepts, Details) protocol (Grimm et al. 2006, 2010). The model was developed and simulations conducted in R (version 2022.12.0; R Core Team 2023). We report the median and interquartile range of time to reach the target population abundance (n= 750 adults) and the median and interquartile range of adult abundance for the last 5 years of simulation. We evaluated the time to reach target abundance at two levels of certainty: time for 50% (median) and 95% of simulations to reach the target population abundance.

Purpose.– The purpose of this model was to predict possible outcomes of lake sturgeon stocking scenarios to inform the GTB decision-making process for lake sturgeon rehabilitation in the Boardman-Ottaway River watershed.

Entities, state variables, and scales.–

Individuals– The modeled individuals were lake sturgeon. The state variables of individual fish were age (years), sex, maturity, the origin of the individual (i.e., wild or stocked), and if an individual would spawn each year.

Time step– One time step represents 1 year, and simulations were run for 250 years (about 10 generations) and 100 iterations due to the species' long life history and late age-at-maturity (Schueller and Hayes 2011; Nelson et al. 2022).

Process overview and scheduling.– The model used a sequential set of steps to represent the species-specific seasonal cycle of migration, spawning, and other life history events (Figure 3.1). At each process step, the values of attributes of individuals were updated. Time is modeled as discrete years and updated at the end of the model schedule.

For individual fish, the model proceeds as:

1. Stocking
2. Reproduce (add new recruits)
3. Post-spawn mortality
4. Grow (age) and mature

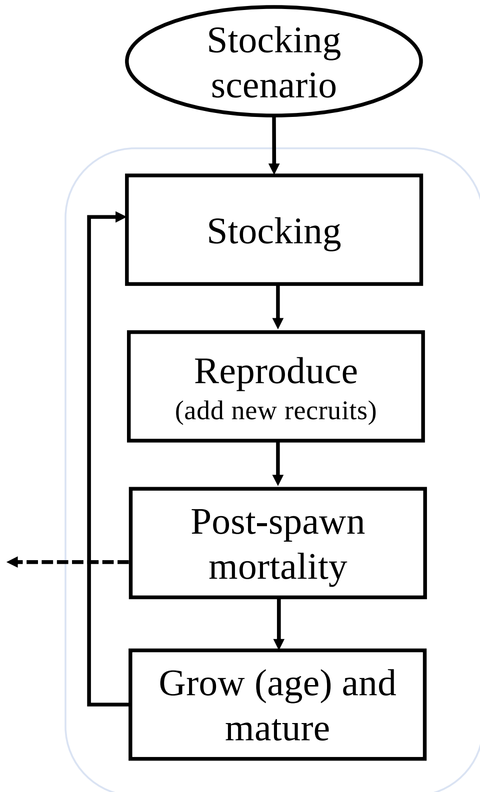

Figure 3.1. Conceptual diagram for the lake sturgeon *Acipenser fulvescens* population model to aid decision making on stocking in the Boardman-Ottaway River, Michigan, USA.

Design Concepts.– The lake sturgeon population model was modified from Chapter 2 to evaluate how changes in number of fish stocked, duration of stocking, and stage-specific mortality rates affect the time for lake sturgeon populations to reach the target abundance in the Boardman-Ottaway River. The model included key aspects of lake sturgeon population dynamics, namely sexually dimorphic late-age maturation, periodic spawning behavior, and low rates of intrinsic population growth and natural mortality. Except for the addition of individuals through stocking, the model simulated a closed lake sturgeon population.

The emergent result of the model was population abundance. Individuals did not have adaptive traits, no explicit sensing by individuals occurred, and only indirect interactions among individuals were modeled. Stochasticity was included in the model by drawing parameters from statistical distributions to allow individual variability while keeping the parameters within a predetermined range. Stochastic parameters included stage-based mortality rates, number of fish stocked annually, intrinsic population growth rate, and recruitment error. Bernoulli trials were used to draw mortality and spawning events (if a fish would spawn in a given year). There was substantial uncertainty regarding stage-specific survival rates, intrinsic population growth rate (r), the number of fingerlings that would be stocked each year, and the duration (in years) of stocking. For all model runs, stage-specific mortality rates were drawn annually from a uniform distribution based on Welsh et al. (2010), and intrinsic population growth rate (r) was drawn annually from a uniform distribution of rates from Great Lakes lake sturgeon populations reported in the literature (Table 3.2; Bruch et al. 2016).

Initialization.– The lake sturgeon population in the Boardman-Ottaway River is currently estimated to be very low with mature fish rarely observed below the Union Street Dam (Hay-Chmielewski and Whelan 1997; Travis 2019; UpNorthLive 2021); therefore, we assumed there was no initial population in the watershed in year-1. The model began in year-1 with the first stocking event and the number stocked was dependent on the stocking scenario (Table 3.1). Stocked individuals were initialized assuming a 1:1 male-female ratio, fingerlings were age-0, yearlings were age-1, and all individuals were immature.

Input data.– The model does not use input data to represent time-varying processes.

Submodels.– The first submodel was the stocking submodel (Figure 3.1). Fingerlings and yearlings were stocked annually based on the scenario being evaluated (Table 3.1). Next, recruitment was calculated using a Ricker (1954) stock-recruitment function (Equation 3.1). See Table 3.2 for parameter values and definitions.

Equation 3.1: $R_t = S_t e^{r(1-(S_t/K))-1+\varepsilon}$

Only mature individuals were able to reproduce each year. Mature females spawned each year with a probability of 20% (equating to an average female spawning once every five years), and mature males spawned with a probability of 50% (equating to an average male spawning once every two years; Forsythe et al. 2012; Dammerman et al. 2019). If no females were present in the Boardman-Ottaway River and actively spawning, the recruitment for that year was set to zero. We allowed the annual intrinsic population growth rate to vary uniformly each year within simulations among published estimates of Great Lakes lake sturgeon populations (r=0.079–0.123; Bruch et al. 2016), because there is high uncertainty about the population growth rate in the watershed. We assumed carrying capacity K, of the Boardman-Ottaway River watershed was equal to 200, which is consistent with a small lake sturgeon population (Hayes et al. 2012).

Mortality was stage-based and varied annually within the range used to set stocking recommendations (Welsh et al. 2010); however, the management informed estimates for the Boardman-Ottaway River is given in Table 3.2. Wild offspring (non-stocked individuals) which had not yet matured were not subjected to natural mortality due to a maturity lag in the recruitment function (Ricker 1954). Age-2 fish were considered juveniles and their mortality followed the annual adult mortality, but individuals were immature. All fish were subjected to the adult mortality rate once mature. At the end of each year, growth of individuals was modeled by age in years. Maturity was then updated; females and males were assumed to mature at age 20 and 15, respectively (Schueller and Hayes 2011).

Table 3.2. Parameter values, statistical distributions, and references used in the lake sturgeon *Acipenser fulvescens* population model and sensitivity analysis. "Informed" parameter values were used in the sensitivity analysis and statistical distributions were used for stocking scenario model simulations (where listed). Asterisk (*) indicates parameter values that were varied by +/- 20% of the informed value for sensitivity analysis.

Parameter Description	Value	Reference
Age-at-maturity – female	20	Schueller and Hayes (2011)
Age-at-maturity – male	15	Schueller and Hayes (2011)
Annual mortality rate – adult*	0.05 (informed) ~U(0.02, 0.10)	Personal communication, GTB; Welsh et al. (2010)
Annual mortality rate – fingerling*	0.50 (informed) ~U(0.50, 0.90)	Personal communication, GTB; Welsh et al. (2010)
Annual mortality rate – yearling*	0.25 (informed) ~U(0.10, 0.25)	Personal communication, GTB; Welsh et al. (2010)
Annual spawning probability – female	20%	Forsythe et al. (2012); Dammerman et al. (2019)
Annual spawning probability – male	50%	Forsythe et al. (2012); Dammerman et al. (2019)
Annual total recruitment, R_t	–	Calculated each year
Annual total spawners, S_t	–	Calculated each year
Carrying capacity of nursery grounds, K*	200	Hayes et al. (2012)
Intrinsic growth rate, r*	0.1 (informed) ~U(0.073, 0.123)	Bruch et al. (2016)
Recruitment error, ε	~N(0, 1)	Assumed
Number of fingerlings stocked annually*	140, 280 (informed), 500, and 1000	Personal communication, GTB
Stocking duration (years)*	5, 10, 15, 20, and 25 (informed)	Personal communication, GTB

Sensitivity Analysis and Expected Value of Perfect Information

We performed a one-way sensitivity analysis of the lake sturgeon population model by varying one parameter by 20% above and below the management informed value while holding all others steady (Table 3.2). We evaluated the sensitivity of the response in median time (years) to reach the target population level for variations in stage-based survival rates, intrinsic population growth rate (r), number of fish stocked annually, duration of stocking (i.e., years), and carrying capacity (K; Table 3.2).

Based on the results of the sensitivity analysis, we evaluated the expected value of perfect information (Equation 3.2; Runge et al. 2011) to determine how much management could improve if some of the structural uncertainty was resolved.

Equation 3.2: $EVPI = E_s[max_a U(a,s)] - max_a E_s[U(a,s)] \equiv EVPI = E_C - E_U$

The EVPI is calculated as the difference between E_C and E_U and is the expected improvement in management outcome as a result of obtaining perfect information. In Equation 3.2, a is a management action taken, s is a model of the system, and $U(a,s)$ is the utility associated with taking action a, under model s. The first term in Equation 3.2 represents the expected value of making the decision if the uncertainty was resolved (E_c), and the second term in Equation 3.2 represents the expected value of making the decision without first resolving the uncertainty (E_U; Runge et al. 2011). We evaluated the EVPI for lake sturgeon fingerling mortality to determine how resolving that structural uncertainty could improve the time to reach a restored lake sturgeon population. We simulated a low, medium, and high fingerling mortality scenario with stocking rates of 280, 500, and 1000 fingerlings per year for each mortality scenario, and evaluated the time to reach a restored population.

Results

The median time to reach the target abundance for lake sturgeon populations (n= 750 mature adults) was estimated to take between 31 and 91 years, depending on the stocking strategy. The time for 95% of the simulations to reach the target abundance ranged from 34 to 120. The fastest time to reach the target population occurred when the goal (n= 1,000 fingerlings) was stocked annually for 25 years. No difference in median and interquartile recovery time was found for the model which stocked the goal for only 20 years, compared to 25 years (Figure 3.2), although the time estimated for a 95% chance for recovery increased slightly from 34 to 39 years (Figure 3.2). The slowest time to reach the recovery level was when only 140 fingerlings were stocked annually. When the recommended number was stocked annually (n= 280) the median time to reach recovery was 68 and the 95% chance for recovery was 85 years. When the number stocked annually and duration of stocking declines, the median time to reach the recovery target declines almost linearly (Figure 3.2). When the number stocked annually was allowed to randomly vary and drawn from a uniform distribution from 0 to 1,000 the median time to reach recovery level was 51 years with a 95% chance of recovery in 69 years. In the final 5 years of simulation (years 246–250), median abundance ranged among stocking scenarios from 4,148 to 5,021 and 95% of simulations ended with abundances ranging from 1,814 to 2,200 (Table 3.3).

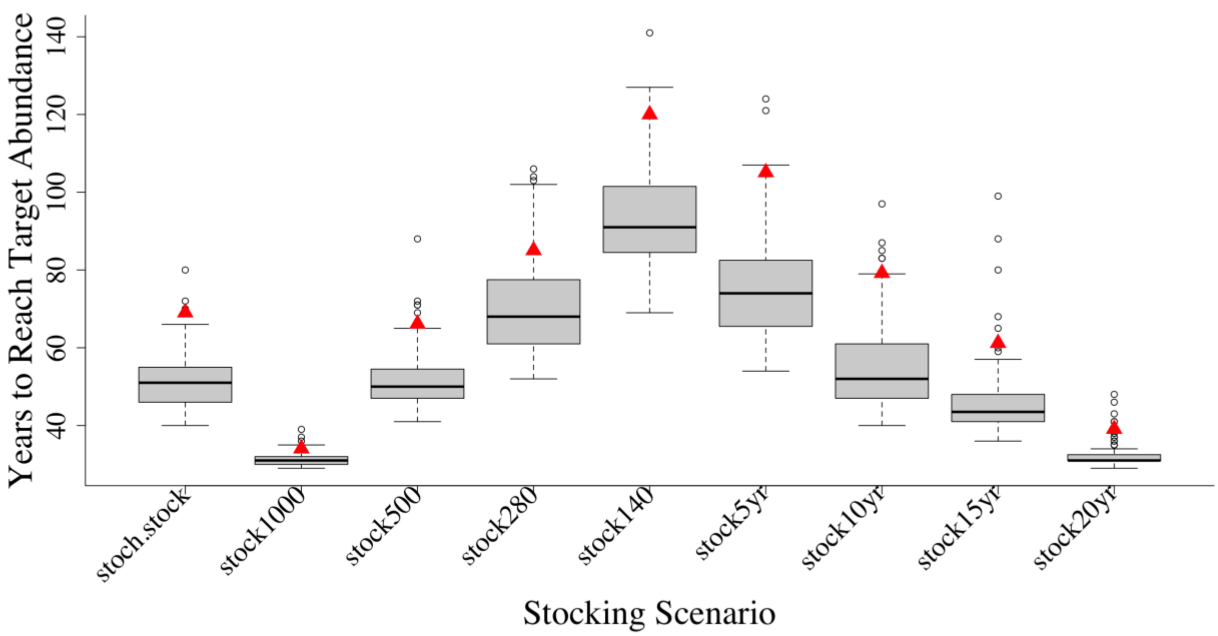

Figure 3.2. Boxplot showing the median (black line) and interquartile range (gray shaded region) of time (years) to reach the target population abundance of lake sturgeon *Acipenser fulvescens* (*n*= 750 adults) for each of the nine stocking scenarios. Red triangles are the time for 95% of simulations to reach the target abundance. Stocking scenarios on x-axis are described in Table 3.1.

Table 3.3. Median, quartiles, and 95th percentile of lake sturgeon *Acipenser fulvescens* population abundance in the final 5 years of simulation.

Scenario	Median (25 – 75%)	95th percentile
Stock.1000	5021 (3655–6696)	1908
Stock.500	4964 (3648–6900)	2197
Stock.280	4819 (3730–6401)	2200
Stock.140	4417 (3499–5807)	1831
Time.5	4594 (3527–5976)	2082
Time.10	4614 (3497–6122)	1917
Time.15	4732 (3454–6178)	2180
Time.20	4148 (3126–5745)	1814
Stoch.Stock	4838 (3544–6310)	1868

The sensitivity analysis found the time to recovery was most sensitive to fingerling mortality, particularly increases in the parameter value (Figure 3.3). The time to recovery was sensitive to the number stocked annually and the duration of stocking and was more sensitive to lower values (Figure 3.3). The time to recovery was also sensitive to adult mortality, especially decreases in the adult mortality rate (Figure 3.3). The time to recovery had comparably low sensitivity to yearling mortality, intrinsic population growth rate (r), and carrying capacity (K). Increases in the stocking duration beyond 25 years resulted in no change in time to reach population target level (Figure 3.3).

We calculated the EVPI for resolving the uncertainty in fingerling mortality rate because the sensitivity analysis showed the time to reach target abundance was most sensitive to changes in fingerling mortality rate (Figure 3.3). The EVPI analysis showed that by resolving uncertainty in fingerling mortality rate, management could expect to improve and reach the target abundance about 3 years earlier (Figure 3.4).

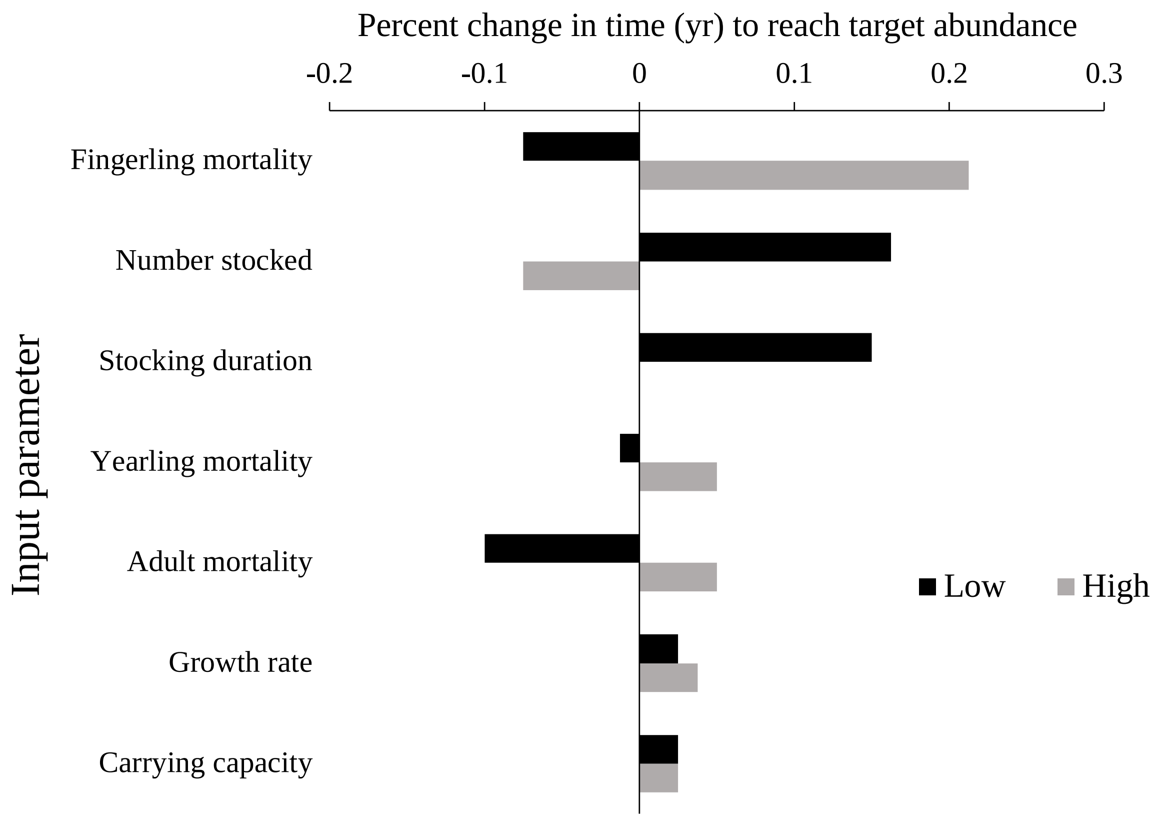

Figure 3.3. Tornado plot showing the sensitivity of parameters used in the lake sturgeon *Acipenser fulvescens* population model. A positive percent change is an increase in time to reach target population level. Black and gray bars show the change when parameter was varied 20% lower or higher, respectively than the base value.

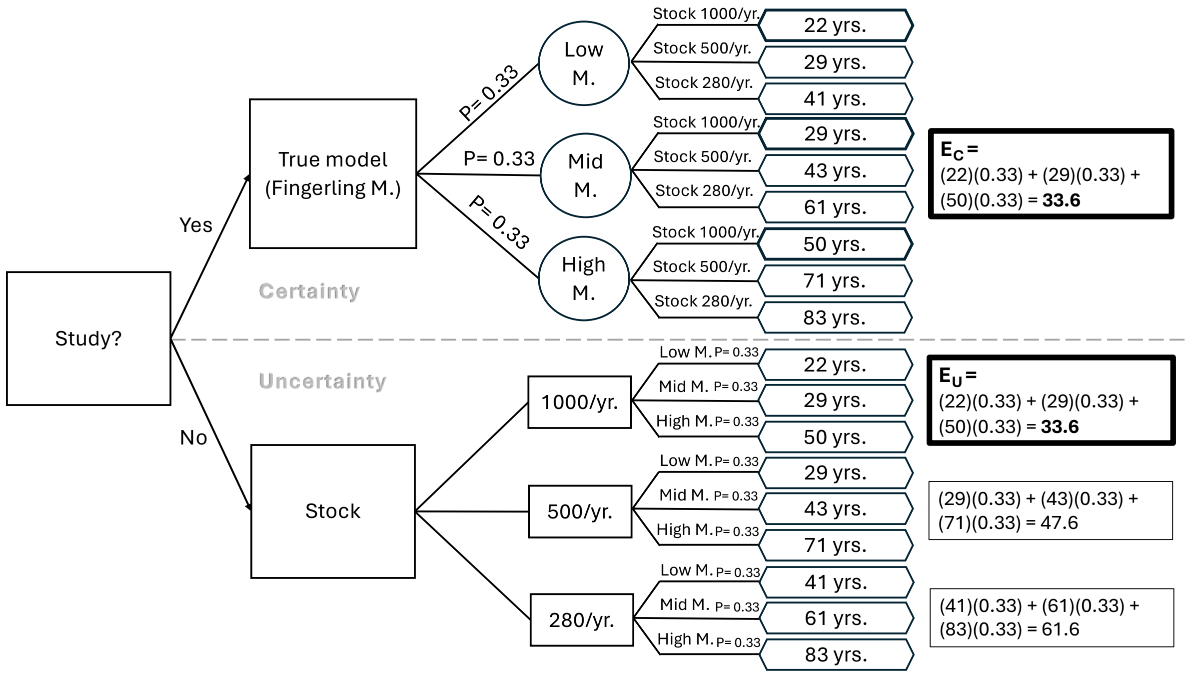

Figure 3.4. Decision tree and expected value of information for resolving uncertainties around lake sturgeon *Acipenser fulvescens* fingerling mortality rates and the outcomes of stocking scenarios. Gray dashed line represents the division of the expected value of certainty (E_C) and uncertainty (E_U) in decision making.

Discussion

Rehabilitation of lake sturgeon is an important objective for stakeholders and rightsholders in the Boardman-Ottaway River watershed. Lake sturgeon is native to the Great Lakes basin and is a threatened species. Furthermore, the species is especially important to Native American Tribes—such as the Grand Traverse Band of Ottawa and Chippewa Indians, a decision-maker for FishPass alongside the Michigan Department of Natural Resources—due to their ecological and cultural value. Mature lake sturgeon have been observed below Union Street Dam (Travis 2019; UpNorthLive 2021), and the Boardman-Ottaway River is a historical spawning ground (Hay-Chmielewski and Whelan 1997); however, a current population does not exist (Hayes et al. 2012). Our results show that a lake sturgeon stocking program on the Boardman-Ottaway River could potentially restore populations to target levels in under 50 years, with a very high likelihood of restoration in 100 years. If 1,000 lake sturgeon fingerlings were stocked for only 5 years, the population could be restored in 105 years. Without stocking, Lake Sturgeon populations in the Boardman-Ottaway River were unlikely to be restored (see Chapter 2), and in the best-case scenario restoration may take over 180 years without targeted rehabilitation efforts. The 95% chance for population recovery was at 240 years for only one scenario which included the highest intrinsic population growth rate and highest initial abundance ($n = 80$). All other scenarios that did not include stocking never reached the target population level by year-250 (see Chapter 2). Our results show that stocking could enhance rehabilitation of lake sturgeon populations; however, our model assumed that natural reproduction occurred, and Nelson et al. (2022) found that lake sturgeon recovery was highly sensitive to recruitment failure. There has been no documented lake sturgeon spawning in the Boardman-Ottaway River; however, a gravid female was captured during spring monitoring by the GTB (Brett Fessell, personal communication).

Due to the difference in the GTB goal for stocking (1000/year) and the genetic guideline recommendations (280/year; Welsh et al. 2010) there are several tradeoffs for decision-makers among the scenarios. For example, stocking the goal for only 5 years would result in a similar median time to recovery as the recommended stocking scenario, and stocking the goal for 10 years led to a similar 95% time to recovery as the recommended stocking scenario. The longest time to reach the target population was for the half recommended stocking scenario (140/year for 25 years). If the GTB goal is stocked for 25 years, and natural reproduction occurs, the model suggests a high likelihood of reaching a target population in under 40 years. If stocking does not occur a population is unlikely to be restored (see Chapter 2); however, if stocking of any of the scenarios occurs a lake sturgeon population would probably be restored in under 120 years. As long as some stocking occurs, the exact stocking scenario chosen might not be critical when considering the predicted long-term response. For example, in years 246–250, 95% of simulations of all scenarios had an estimated abundance between 1,814 and 2,200. However, in the short-term (<100 years) the decisions made regarding the numbers of fingerlings stocked and the duration of the stocking program will affect the time to reach the target population.

Previous lake sturgeon research has focused on population viability (Schueller and Hayes 2011; Nelson et al. 2022), modeling known populations (Bruch et al. 2021; Vaugeois et al. 2022), and translocation (Koenigs et al. 2019). This research is relatively unique in that we model a species currently not found in the watershed. Despite this, our results compare well with other lake sturgeon research. Schueller and Hayes (2011) used an IBM to perform a population viability analysis of lake sturgeon to evaluate the population-level effects of inbreeding and determine the risk of extinction and minimum viable population size. We used several parameters from Schueller and Hayes (2011) including sex-specific age-at-maturity and adult mortality. Because of the similarities between the two models, the population-level results compare well, although Schueller and Hayes (2011) focused additionally on the genetic consequences of inbreeding on population viability, which was not a focus of this study.

Recently, Nelson et al. (2022) used a population viability analysis to evaluate the effects of recruitment variability and adult harvest rate on lake sturgeon populations in Canada and found that slow somatic growth rates and higher rates of adult harvest inhibited lake sturgeon population viability. Nelson et al. (2022) evaluated three somatic growth rates and harvest rates between 5–9%; however, considerations for somatic growth and adult harvest were omitted from the present model. In our model, annual adult mortality was randomly drawn from a uniform distribution from 2% to 10% for all scenarios to account for natural mortality. It is likely that a lake sturgeon restoration effort in the Boardman-Ottaway River would include management protections to reduce adult mortality; however, lake sturgeon are occasionally caught in Tribal nets in Grand Traverse Bay (Brett Fessell, personal communication). Similar to this study, Nelson et al. (2022) found that stocking yearling lake sturgeon increased the probability of population recovery. The target population was much larger in that study (*n*= 5,384 females) and thus, took much longer to reach (about 250 years); however, stocking increased the probability of recovery within 100 years. Nelson et al. (2022) found stocking 1,000 yearlings annually in combination with harvest reductions would yield high probabilities of recovery. Although our stocking scenarios included a much smaller number of yearlings (0–6 annually), we found the stocking of 1,000 fingerlings annually would also dramatically decrease the time to reach the recovery target.

We found the short-term population response (<100 years) was heavily dependent on the duration of stocking; however, the long-term population (>200 years) response was similar across stocking scenarios regardless of duration. Nelson et al. (2022) simulated stocking for 50 years, which was longer than any of our scenarios. The longest duration of our stocking scenarios was 25 years, but we found little difference in the time to reach recovery if stocking occurred for only 20 years. Additionally, for the sensitivity analysis we increased stocking duration by 20% (from 25 to 30 years) which resulted in no change in time reach a recovered population. When the duration of stocking decreased from 20 years, declines in time to reach the target population occurred. Vaugeois et al. (2022) developed an IBM for lake sturgeon in the Great Lakes basin to evaluate the effect of contaminants on population abundance and recovery for various management actions. To simulate stocking, they allowed 50 spawners to reproduce annually for the first 15 years of a 200-year simulation. They found management actions which increased egg survival rate or the stocking of fingerlings resulted in the fastest population growth and recovery compared to management actions that increased juvenile or adult survival rate; however, the study was focused on the population-level effects of contaminants. Nevertheless, our results compare well with Vaugeois et al. (2022); we found that stocking fingerlings reduces the time to reach target population abundance. Our study did not include an egg stage, but we found our model to be sensitive to fingerling mortality, especially to increased fingerling mortality.

Uncertainties in lake sturgeon habitat use and movement in the study system required us to make some assumptions and modifications to the model. For example, our model does not account for habitat; however, based on recent Boardman-Ottaway River lake sturgeon habitat suitability (Dean 2023), habitat likely is not limiting for lake sturgeon in the upper Boardman-Ottaway River. Additionally, we did not include individual movement or growth in the model due to large uncertainty in these metrics. Lake sturgeon have not had access to the upper Boardman-Ottaway River in over a century and, although habitat is likely suitable (Dean 2023), areas of spawning are not known, as is the case for some better studied lake sturgeon rivers in Michigan (e.g., Black River, Cheboygan County, MI). Similarly, since lake sturgeon have been absent from the upper river, we do not have information available to parameterize individual growth and movement in the model. However, Nelson et al. (2022) reported that faster somatic growth rates led to faster lake sturgeon population recovery. Finally, our model does not include immigration and emigration; however, lake sturgeon is known to exhibit low rates of straying (Homola et al. 2012) and therefore, immigration and emigration are unlikely.

For some species, barrier removal may not restore populations of migratory species in an acceptable timeframe for stakeholders and rightsholders. Lake sturgeon will likely need additional management actions beyond enhancing connectivity to restore populations in the

Great Lakes basin. This lake sturgeon population model is a simple, yet effective tool to aid decision makers in choosing lake sturgeon stocking strategies and forecasting possible outcomes of different stocking strategies for stakeholders, rightsholders, and decision makers. Using realistic life history parameter estimates and a modeling approach with sex-specific maturation schedules, periodic spawning events, and a protracted juvenile stage of up to 20 years, our model captures important dynamics of lake sturgeon life history. Our results indicate rehabilitation of lake sturgeon will take time and will likely require targeted rehabilitation actions including stocking, a harvest moratorium, protection to spawning individuals, and habitat restoration. Future data collected on lake sturgeon in the Boardman-Ottaway River could be used to improve parameterization of this model and increase the feasibility of including predictions of growth and movement. Furthermore, future studies might attempt to estimate fingerling and adult mortality, because we found our model was sensitive to these two life history parameters. However, because the expected value of resolving uncertainty in fingerling mortality was so zero, delaying the beginning of a stocking program to resolve this uncertainty would not be warranted. Nevertheless, our model is an important contribution to aid in the decision making and rehabilitation of a highly desired, yet imperiled native Great Lakes species in a watershed they are currently not found.

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CHAPTER 4: CONCLUSION

My dissertation evaluated the ecological, economic, and social consequences and tradeoffs of enhancing stream connectivity in the Great Lakes region. The overall goal of my research was to use decision analysis and predictive modeling to help inform fisheries managers on management decisions surrounding novel connectivity regimes. I used the Great Lakes Fishery Commission FishPass project as a case study; however, the results of my research have management implications beyond my case study.

In Chapter 1, I formed a working group of stakeholders and rightsholders to elicit their objectives and values for enhancing connectivity under various fish passage alternatives using structured decision making. The two fundamental objectives identified by the working group were to maximize the integrity and ecological health of the river and maximize user and public satisfaction. Many of the objectives and values from this study will be applicable to decision making elsewhere in the Great Lakes, especially for invasive and introduced species. However, streams that do not have resident trout populations isolated upstream of barriers might have less contention surrounding the passage of non-native Pacific salmonids than was found on the Boardman-Ottaway River. The optimal management alternative was passage of native fishes only; however, the optimal alternative varied based on the weight stakeholders might place on each objective. Four weighting scenarios were developed to evaluate the change in optimal management alternative with changes in objective weights. In general, the passage of native fishes only was the optimal alternative unless a stakeholder heavily valued salmon or social objectives. Interestingly, I found *more fish* was not always the best management alternative and when stakeholder objectives were explicitly stated and considered in the decision process, oftentimes selective fish passage alternatives outperformed alternatives with more fish passage.

The results of this research will help inform decision-makers on fish passage alternatives that are preferred by stakeholders and that are likely to achieve their objectives. This research is a rare example of structured decision making being applied to a barrier removal or remediation project; therefore, these results likely will be highly valued by decision makers working on similar barrier problems in the Great Lakes and beyond.

In Chapter 2, I developed an individual-based modeling framework to forecast changes to species abundance and distribution for varying levels of enhancing connectivity. Few studies have predicted the response of migratory fish populations to enhanced fish passage prior to barrier removal or remediation, despite this being a critical step in the decision-making process. The response of fish populations to changes in connectivity was species-specific and affected by the initial abundance and spatial distribution in the watershed, number of fish passing the barrier, and species life-history traits. Due to the increase in access to novel habitat, species found only below the barrier prior to barrier remediation were predicted to have larger production potential than species found both above and below the barrier. Non-native species were predicted to have more production than native Great Lakes basin species.

The modeling framework I developed is flexible and can accommodate several species types and spatial arrangements; therefore, it could be adapted to fit the needs of most if not all barrier removal or renovation projects. Additionally, this framework allows for learning about the system as assumptions are tested in the model. This work is a first step in developing a modeling framework that can be a standard approach to evaluate management alternatives, make tradeoffs among objectives, and inform decision making for enhancing connectivity. Prior to this work, a modeling framework to aid decision makers in predicting the response of fish populations to increases in connectivity before the change occurs was not available; however, it has now been developed and is available for further use. Additional updating and application of the modeling framework to new systems and connectivity research will improve the ability to forecast changes in the future.

In Chapter 3, I evaluated stocking scenarios for lake sturgeon in the Boardman-Ottaway River. Lake sturgeon is an imperiled species and native to the Great Lakes, but they are currently rarely found in the Boardman-Ottaway River; therefore, restoration of the species is an important objective to many stakeholders and rightsholders. I found in Chapter 2 that lake sturgeon populations are unlikely to be restored in under 250 years through enhanced connectivity alone. Therefore, additional management actions such as stocking, might be needed to increase the chances of an expedited population recovery.

I modified the lake sturgeon model from Chapter 2 to include various stocking rates and demographic parameters to evaluate the potential time to reach a target lake sturgeon population. The median time to reach the target abundance (n= 750 adults) was estimated to take between 31 and 91 years, depending on the stocking strategy. The model suggests that a lake sturgeon stocking program on the Boardman-Ottaway River could potentially restore populations to target levels in under 50 years, with a very high likelihood of restoration in 100 years. The time to reach the target population decreased as number of fingerlings stocked annually increased and as the duration of stocking increased. Sensitivity analysis found the two most sensitive life history parameters were fingerling mortality and adult mortality. The expected value of perfect information was calculated for the most sensitive parameter, fingerling mortality, but there was low value in resolving that structural uncertainty in terms of the decision of how many fingerlings to stock.

Decision making is the job of the natural resource manager. Decisions to enhance stream connectivity are complex and potentially contentious due to multiple competing objectives. As barrier removals continue and probably increase in the future, managers will repeatedly be faced with decisions surrounding enhanced connectivity. In some instances, enhancing connectivity might be the only management action needed to restore migratory fish populations. Other times, additional management actions might be necessary to restore populations of some fishes. Based on the wide range of ecological, social, and economic values and objectives for enhancing connectivity of stakeholders and rightsholders, any number of passage alternatives might be optimal for a given stream system. Natural resource managers can increase the chances of making decisions that meet the approval of stakeholders and rightsholders by using decision analysis and predictive modeling to incorporate their values and objectives into the decision process, while forecasting the potential consequences of each alternative and making tradeoffs among objectives explicitly and transparently.