DISSERTATION

A BIOECONOMIC AND GENERAL EQUILIBRIUM FRAMEWORK TO ADDRESS

FISHERY MANAGEMENT AND INVASIVE SPECIES

Submitted by

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In partial fulfillment of the requirements

For the Degree of Doctor of Philosophy

Colorado State University

Fort Collins, Colorado

Summer 2017

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ABSTRACT

A BIOECONOMIC AND GENERAL EQUILIBRIUM FRAMEWORK TO ADDRESS FISHERY MANAGEMENT AND INVASIVE SPECIES

Fisheries management is a complex issue that involves the management of people, fish populations and habitat. There are many facets to fishery issues including ownership, regulation, and environmental change. I address all three of these facets in the following work. I develop a general equilibrium model that incorporates fish stock and present two applications of it. I evaluate the change of a fishery under a regulated open access regime to an individual transferrable quota system. I apply the model to the Lake Erie yellow perch fishery, and I account for the different allocations of the value provided by the fish stock, and the potential changes in efficiency. I find that the change to an individual transferrable quota system results in welfare improvements but only if the individual transferrable quota system induces improved catchability and efficiency in fishery effort choices. I also develop an integrated bioeconomic model with the general equilibrium framework to evaluate the joint responses of a regional economy and lake food web to an environmental shock. The model is unique in that there are feedbacks between the economy and food web. The bioeconomic model is used to evaluate a potential Lake Erie Asian carp invasion. There are two primary results from the analysis; the Asian carp invasion leads to welfare improvements, and when invasion impacts are estimated with only the ecological food web model, without the consideration of changes in human choice, the impacts to some fish populations are overestimated while others are underestimated. In both

applications, I show that using a general equilibrium framework captures welfare impacts that would be missed by a partial equilibrium analysis.

ACKNOWLEDGMENTS

I am indebted to my committee members for their guidance and mentorship while working on this dissertation and completing my degree program. I am particularly grateful to my primary advisor, Travis Warziniack, for all that I have learned from him, and his continued support as I navigated this research and pursued both of my graduate degrees. I am also appreciative of the USDA Forest Service for the opportunity to serve as a Graduate Research Assistant while pursuing my Ph.D.

The simulations presented in this dissertation would not be possible without data. Thank you to the Great Lakes Fishery Commission for publishing and making their data so readily available. Thank you also, to Jeff Tyson and Travis Hartman from the Ohio Department of Natural Resources for providing me with the Lake Erie Status Reports. Data from both of these sources was crucial for my work.

I must also thank those that have helped me get here; Amy Galarowicz from UBMS at the College of St. Scholastica, Dr. Beam at the University of Wisconsin – Superior, Dr. Finnoff at the University of Wyoming, Katie, Zach and my ‘brothers’ from the University of Wyoming, my classmates (and now friends) from CSU: Jada, Brian, Aaron and Jason, and numerous friends and family members, who even when they didn’t know how to help, did in some way. It takes a village, and I’ve been lucky to have a great one. Thank you.

I must also acknowledge that part of the work presented in this dissertation was funded by Environmental Protection Agency Great Lakes Restoration Initiative, NOAA Center for Sponsored Coastal Ocean Research awards NA09NOS4780192 and NA10NOS4780218.

DEDICATION

I dedicate this work to my husband, Mitchell Apriesnig, and my mom, Kathy Bryce. There are no words that can express how grateful I am for the un-ending support you have given me while pursuing all my educational goals. I couldn't have done this without you. Thank you.

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CHAPTER 1: AN INTRODUCTION TO FISHERY ISSUES AND MODELING CONSIDERATIONS

1.1 Introduction

Fisheries are subject to a variety of management regimes, cross state and country boundaries, are susceptible to changes in the ecosystem, and can face various demands of commercial and recreational harvest; all of which makes evaluating fishery issues a complex, yet interesting, task. In the following chapters, I address these facets of fisheries while exploring two primary research questions: what are the implications of a fishery moving from a regulated open access management setting to an individual transferrable quota system? and how do the estimated impacts of an Asian carp invasion differ between models with linkages between the economy and the food web and models without? Human choice and response to changing conditions are present in both of these research questions, and make the questions interesting not only as impact analysis, but as a way to further investigate the trade-offs that occur when management changes or an invasion takes place. To evaluate each of these questions, I focus on Lake Erie of the Laurentian Great Lakes and develop place-specific bioeconomic models in a computable general equilibrium framework. In addressing both questions, I show that the general equilibrium framework captures more welfare impacts than a partial equilibrium model would. I introduce the modeling frameworks to evaluate those questions in Chapter 3 and 4. Here, I provide an overview of fishery issues and the common components involved in the economic modeling fisheries.

1.2 Background of Fishery Issues

Fisheries can play different roles within a community. Some fish species are targeted as a recreational activity, others are targeted by commercial fishers, and some are targeted by both. In addition to having multiple roles within a community, property rights, management strategies, and environmental changes are important fisheries issues. In this section I provide background on these three fishery issues.

1.2.1 Fishery Ownership

An integral component of the harvest and management of a fishery is its ownership. Clashes in property rights can occur among commercial fishers, but also between recreational and commercial fishers (Abbott, 2014; Kahn, 1995). On one end of the spectrum are open access fisheries for which no property rights are established, and the fishery is open to anyone. On the other end of the spectrum are private property fisheries for which ownership is clearly defined. Private property may happen naturally, such as a single owner of a lake, but more often property rights are not clearly defined, and need to be assigned and enforced by some entity in order to achieve an efficient outcome (Perman et al., 2012).

The issue of property rights is further complicated by variations in governing bodies. Not only are property rights called into question when deciding who has the right to fish, but the question also arises, "Who has the right to manage a fishery?" Consider, for example, differences between fisheries in the U.S. and Canada. The Fisheries Act in Canada establishes clear jurisdictions for fishery management by requiring permission from the minster to alter fish habitat or kill fish (Hubert & Quist, 2010). In the U.S., the jurisdictions for fishery management are not as clear, and ecosystem management is more difficult. Fishery managers have the authority to manage fisheries within the boundaries of districts they manage, but outside of those borders, managers must create partnerships with other agencies (Hubert & Quist, 2010). There are many different authorities that manage fisheries, from state departments of natural resources to the U.S. Fish and Wildlife Service, which can make boundaries and jurisdictions difficult to navigate.

1.2.2 Fishery Management

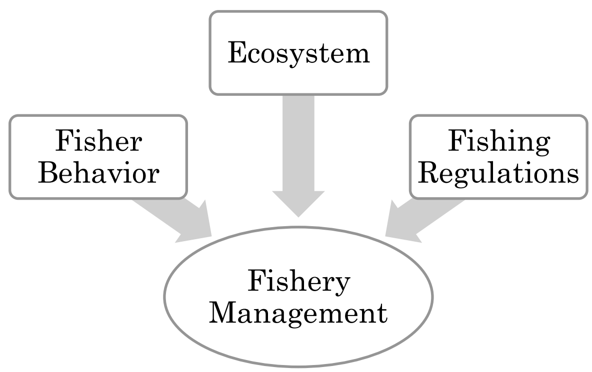

Figure 1: Components of Fishery Management

Successful fisheries management includes addressing fish populations, habitats, and human behavior (Hubert & Quist, 2010). Fisheries are managed in a variety of ways with a variety of goals, and some fisheries have no management. A fishery without management is not necessarily a problem. When private property exists, the rational owner is likely to operate the fishery in their own best interest, and over exploitation and the other issues associated with open access are not problems (Weitzman, 1974). However, if property rights are not established and the fishery is left unmanaged, the issues of an open access fishery will prevail; over exploitation and inefficient use of inputs will likely occur. When management is present in a fishery, the management practices employed can often be separated into two types; management that seeks to change or establish property rights, and management practices that place a limit or control on an aspect of the fishing process. The degree of involvement of managers depends on the management goal. I provide a brief overview of different management strategies here. Command and control is one classification of fishery management that typically limits or restricts an aspect of the fishing process. Examples include harvest limits, bag or size limits, fishing practices or season length. This type of regulation is present in both commercial and recreational fisheries. When it applies to a commercial fishery it is often referred to as regulated open access (ROA) because some aspect of the fishery is regulated, but entry and exit is free. In these management regimes, the government essentially maintains ownership of the fishery, or at least the right to manage the fishery, and tries to achieve the desired outcome of the fishery through their restrictions. The goals can vary, but often they are ecological in nature such as maintaining a sustainable fish stock. This type of fishery management usually requires a high level of involvement from fishery managers. Regulating fishing practices can be useful to achieve some goals. Setting net restrictions, for example, can help make fishing safer for non-target species, but it is hard to manage all aspects of fishing. When an effort input is restricted, fishers often expand effort in other ways, a common example is that when the number of boats allowed to fish is restricted, larger boats are used to increase harvests (Field, 2008). Limiting harvests can achieve the goal of maintaining a desired fish stock, but it often induces a race to fish and an over exertion of effort to be employed (Field, 2008).

The second category of fishery management strategies are those that seek to establish or impose property rights. The individual transferrable quota (ITQ) system is a management regime that has gained popularity as a way to achieve ecological goals in a way that is both efficient and sustainable (Field, 2008). While there is variation in how property rights are allocated and maintained, the common components of these types of strategies are that a TAC is established, the TAC is divided into individual catch quotas that are allocated to fishers, the quotas provide the right to fish and can be sold, leased or traded, and that quota rights are enforced (Conrad, 2010; Field, 2008; Perman et al., 2012). The combination of setting a TAC based on fish population data, and creating a market with clear property rights allows the fishers to arrive at an efficient outcome while still maintaining a safe fish stock (Perman et al., 2012). In practice, effectiveness of ITQ systems in encouraging efficient effort levels has been mixed (Fox et al, 2006; Walden et al., 2012).The effectiveness of an ITQ system depends on the manager's ability to set an ecologically sound TAC and maintain property rights, the initial allocation of quotas can also be a difficult process (Field, 2008). A criticism of the ITQ system is that it can create an exclusive group of wealthy fishers that has access to fishing permits, either from the initial allocation of quotas or through the market, although others have argued that this distribution of wealth is an aspect of the efficient outcome where fishers with the highest marginal cost exit the market (Conrad, 2010; Perman et al., 2012).

1.2.3 Environmental Changes

Another issue related to the use, management, and evaluation of fisheries are environmental changes. Any natural resource is susceptible to environmental changes from a variety of sources, and fisheries are no different. A source of environmental change (and the topic of part of this dissertation) that fish are particularly susceptible to is invasive species. An invasive species is a non-native species that causes or is likely to cause economic and or ecological damage

(Vásárhelyi & Thomas, 2003; Warziniack et al., *under review*). In order to become an invasive species, a sufficient number of the species must survive transportation from their native habitat, be introduced into new habitat and then reproduce, this means that there is often a lag time between when a species is introduced into a new habitat and when it is detected as an invasive species, which can make control of invasive species difficult (Hubert & Quist, 2010). Once a species has become invasive, control options become more limited and costly (Hubert & Quist, 2010; Lodge et al., 2016).

Climate can exacerbate the invasive species problem. Two aspects of the pathway for a species to become invasive are likely to be impacted by climate change: transportation and other human vectors of introduction, and natural dispersal into newly suitable habitats outside of a species native range. As climate change occurs, pathways of introduction that were previously non-existent may open up, such as shipping routes in the arctic (Countryman et al., 2016; Hellmann et al., 2008). Potential suitable habitats for non-native species may also expand as environmental conditions change, but habitats may also become un-suitable for native species that propagate there (Hellmann et al., 2008).

Invasive species are an important consideration for fisheries management in both ecological and economic contexts. Invasive species can change the dynamic of an ecosystem and put native species' survival at risk. Changes in the ecosystem can impact activities that humans have preferences for, such as commercial and recreational fishing. Understanding a fishery's susceptibility to an invasive species, and the tools available to limit introduction of or control of an invasive species leads to efficient management of both the targeted fishery and the invasive species (Lodge et al., 2016).

1.3 Fishery Modeling Considerations

Fishery models have progressed over the last fifty years, but for the most part, they all require standard choices about how the fishery will be represented, including how to best represent the fishery production function, the representation of the biological processes, and what the economic framework of the model should be. In this section I provide background on each of these modeling considerations.

1.3.1 Biological Growth Processes

The nature of modeling a fishery often requires using a bioeconomic model, that is, a model that includes the biology of a fish population, and the consequences that flow from the economic decisions made by humans (Field, 2008). Methods representing the biology that determines the size of the fish stock can vary from model to model, and depends on factors such as predation, foods sources, fecundity, and water quality (Field, 2008). Finding an effective way to generalize biological processes can be difficult, and many models simplify population changes down to the number of individuals in the population and the size of those individuals (Field, 2008). The size of the population can then be related to changes in the population through reproduction and mortality, and changes in individual size through maturation (Field, 2008). Simplifying the biological process in this way has given rise to wide-spread use of the logistic growth function as a way to represent fish stock in bioeconomic models. A general representation of the logistic growth function for a single species is:

$$S_{t+1} = S_t + rS_t\left(1 - \frac{S_t}{K}\right)$$

Where S_t is the stock biomass at time *t*, *r* is the maximum population growth rate, and *K* is the carrying capacity, or the maximum equilibrium biomass that the habitat can sustain (Hubert & Quist, 2010).

The growth equation written above assumes a discrete time component, but it can also be written with continuous time. Continuous growth functions represent a fish stock that responds instantaneously to harvest, and have been described as 'without memory' because they rely solely on the current population (Hanley et al., 2011). Hanley et al., (2011) explain that continuous-time growth models may not be appropriate for species who experience a delay between birth and the time that they are recruited to the harvestable stock.

The logistic growth function is a simple yet powerful equation that captures the tradeoffs within a species between population size and resource availability; however, it does have limitations. It is best at representing the growth process of fisheries contained with limited migration, and can be cumbersome to represent the relationships between many multiple species (Perman et al., 2012).

While the logistic growth function is a common representation of the biological process, other methods exist, from simple functional variations such as the Gompertz function (Conrad, 2010; Perman et al., 2012) to complex alternatives such as predator prey models (Eichner & Pethig, 2006) and ecological equilibrium models (Finnoff & Tschirhart, 2008). A major contribution of this dissertation is to integrate an ecological food web model with a model of the economy

(Chapter 4). The food web used here is developed by ecologists and more fully represents the intricacies that exist between multiple species in a food web than a logistic growth function alone can capture. Such integrated modeling of fully economic models has yet to be used in the field in the fishery economics and management. The Lake Erie food web is described in detail in sections 4.3.3 and 4.4.1.

1.3.2 Production Function

Another key aspect of modeling a fishery is how human choice is represented in the model, usually described by the production function, or as it is commonly referred to in fishery models, the harvest function. The harvest function relates output to the fish stock level and the fishing effort applied (Conrad, 2010). Harvest functions are generally of the form:

$$H_t = F(S_t, E_t)$$

Where H_t represents harvest, and E_t represents effort in time period *t*. There are multiple functional forms used in modeling the fishery, but the two most common are the Cobb-Douglas, or Schaefer, production function, $H_t = qE_t^{\alpha}S_t^{1-\alpha}$, and the exponential production function, $H_t = S_t(1 - e^{-qE_t})$, where q is a catchability coefficient and indicates how productive the inputs to harvest are (Conrad, 2010). Conrad (2010) argues that either production function should be viewed as an approximation to production. Conrad (2010) also argues that the choice of which production function to use in modeling the fishing firm, should be made based on the availability of time series data. Time series data is often used to estimate the parameters in a fishery production function, so choice of production function may depend on the available data (Conrad, 2010). In my models, I choose to use the Cobb-Douglas harvest function, as I believe it represents the relationship between stock, effort, and harvest in the Lake Erie fishery, and historical data is available to estimate the Cobb-Douglas parameters.

1.3.3 Model Scale

Another modeling consideration that receives less attention, but nonetheless becomes inherent in any model, is the choice to use a partial equilibrium model or a general equilibrium model to represent the economic framework. Which economic framework to use depends on if the research is concerned about the happenings in a single market, or how multiple markets interact.

The partial equilibrium approach is the traditional way that fishery issues were evaluated and follows from the fishery work done by Gordon, (1954) and work in the general natural resource literature by Weitzman, (1974) and Samuelson (1974). By its nature, partial equilibrium approaches to fishery modeling are generally more focused on a single fishery and often evaluate issues of ownership and management.

The general equilibrium approach takes on a broader perspective and is often used to evaluate connections between fisheries and other sectors within the economy. General equilibrium fishery models have been used to evaluate impacts of invasive species, tax policy changes, and changes to harvest quotas. Examples include work by Finnoff & Tschirhart (2008), Warziniack et al. (2011), Jin et al. (2012), and Manning et al. (2016) who use general equilibrium models to evaluate some aspect of the management, environmental changes, and ownership issues as they relate to both the fishery and greater economy. While these works address certain fishery related issues, they do not address changes in fishing regulation regimes or models with linkages between the economy and the ecosystem,

I use general equilibrium models in the work presented in later chapters to evaluate the impacts of an invasive species and the changes in management regimes. This allows me to capture the linkages between sectors, households, and producers in the economy. The general equilibrium model used in this dissertation offers a novel look at fishery management and regulations – primarily the potential outcomes of a transition from a regulated open access to a tradable quota system. I show that many of the impacts of the policy would be missed in a partial equilibrium setting, making a stronger case for general equilibrium models in fishery economics. A review of the literature that I build my research on, is developed in Chapters 3 and 4.

1.4 Contribution

To contribute to the current body of research, I focus on developing two aspects of the current literature; (1) evaluating the change of a fishery management regime from regulated open access to an individual transferable quota system in a general equilibrium setting, and (2) developing an integrated bioeconomic model that can be used to evaluate a variety of exogenous shocks, including invasive species. My work addresses three important fishery issues within this modeling framework: fishery ownership, fishery management, and fisheries' response to environmental changes. Figure 2 provides a visual of the flows in traditional general equilibrium model. In the traditional representation of the economy, producers interact through their participation in markets. The markets are assumed to capture all inputs that producers use and goods that households purchase. It is also assumed that for every physical flow (goods or inputs) a monetary flow exists. I expand the traditional representation by accounting for the role that fish stock plays in the economy.

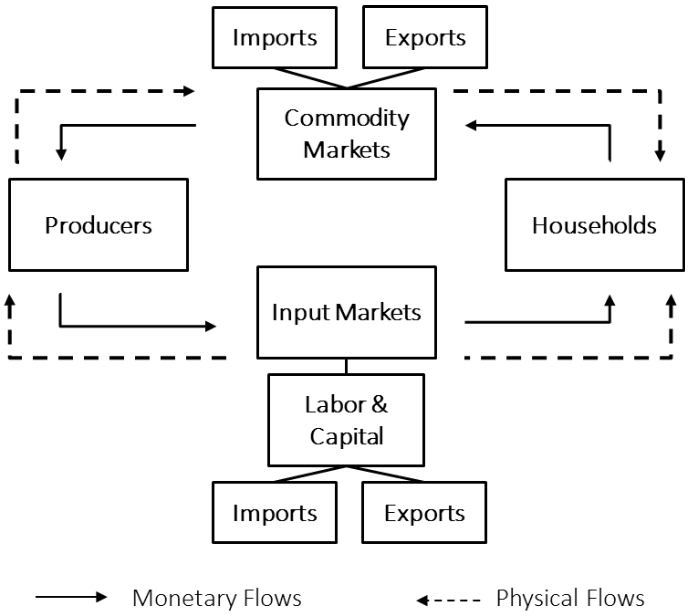

Figure 2: Traditional Circular Flow of General Equilibrium Economy
Shows the physical and monetary flows between households, producers, and the markets commodities and inputs to production are exchanged in.

My models include the standard general equilibrium flows between households, producers, and input and commodity markets, but they also include fish stock and the ecosystem services they provide. Figure 3 shows the circular flow of the economy when accounting for fish stock. In addition to physical and monetary flows between households and producers and the input and commodity markets, fish stock provides ecosystem services to fish producers and the households' consumption of non-market (ecosystem) goods.

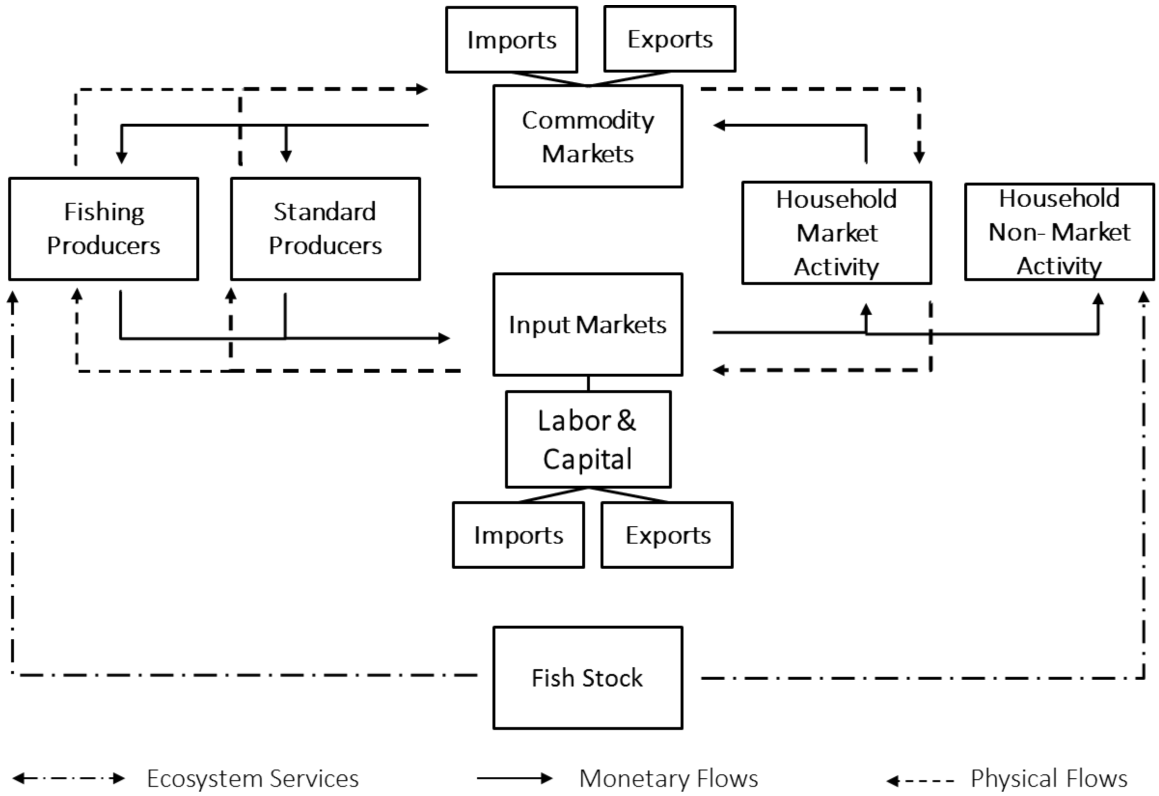

Figure 3: Circular Flow of General Equilibrium Economy with Natural Resource Input
Shows the physical and monetary flows between households, producers, and the markets commodities and inputs to production are exchanged in. In this circular flow, producers are separated by those in the commercial fishing sectors and those who are not, and household activity is separated by market activity and non-market activity. The flows between the fish stock and economy are also shown. Fish stocks provide ecosystem services to commercial fishing and household fishing.

One important aspect of natural resources and the ecosystem services they provide that is represented in the diagram is that typically no payment is allocated to the resources that provide the ecosystems services. In Figure 3, while fish stock provides ecosystem services to both households and producers, no monetary flow is transferred to the fish stock. Ecosystem services have only rarely been used directly in a general equilibrium model, and never fully integrated to the degree used here. Little guidance is offered in the literature on how to model tradeoffs between ecosystem services and traditional inputs to production like capital and labor. A number of innovations were required to adequately model these tradeoffs and the shadow prices that such tradeoffs imply. These innovations are highlighted throughout the dissertation. In the case of the fishery, ecosystem services, at the very least, can be described as the increases in harvest productivity from an increase in fish stock. In the first model, I present (Chapter 3), I use a fixed fish stock as an input to production, and describe the allocation of the value that input provides under different management regimes.

In Chapter 4, the model is developed further to connect the economic framework to a food web, so that I can analyze not only the impacts to the economy when a fish stock changes, but how human response to an exogenous shock further impacts the food web. The role of ecosystem services is expanded to include the input of fish stock into consumers' choices and utility. My modeling choices and the literature on which I base them on are described in detail in Chapters 3 and 4. Beyond my modeling techniques, my work shows that ITQ management systems lead to welfare improvements only if they lead to more efficient effort choices, ecological modeling systems alone do not fully capture the impacts of environmental shocks, and changes in fishing sectors can have welfare implications for the entire economy, even if those sectors are relatively small.

CHAPTER 2: LAKE ERIE BACKGROUND, DATA, AND CALIBRATION

2.1 Introduction

This dissertation explores two primary issues in relation to Lake Erie: (1) a change in the management regime of the U.S. Lake Erie yellow perch fishery, and (2) the impacts of an Asian carp invasion. To evaluate these issues for the Lake Erie fisheries and regional economy, I build computable general equilibrium (CGE) models. Details of the models are described in full in Chapters 3 and 4. Given that the study area is the same and much of the benchmark data used in each analysis overlaps, I provide background on the management of the Lake Erie fisheries and on aquatic invasive species in the Great Lakes region in this chapter to develop a common baseline framework for all subsequent chapters. Simulating a CGE model with biological inputs requires benchmark data on the economic activities in the region and benchmark fish stock data. These data are then used to calibrate parameters for all functions in the model. In this chapter, I also introduce the data I use in my simulations and explain how the available data was transformed into components required for the models I employ. I begin by defining the geographic scope of the study area used, then provide background on invasive species in the region, explain the benchmark economic data, describe the estimation of Cobb-Douglas harvest-function parameters, and finally describe the fishery regulation in the region.

2.2 Lake Erie Geographical Region

Lake Erie is the smallest of the Laurentian Great Lakes by volume and the second smallest by surface area. Lake Erie is bordered by Michigan, Ohio, Pennsylvania, New York and Ontario (Figure 4). I refer to the economy used in the simulations as the ‘Lake Erie Coastal Economy.’ In my specification of the economy, only counties that border Lake Erie are included. In addition to narrowing the scope of the economy to those counties with shores on Lake Erie, I omit the counties that are home to Detroit, Toledo, and Cleveland. Table 1 and Figure 4 show the counties included in the Lake Erie Coastal Economy.

Figure 4: Lake Erie Coastal Economy
Shaded counties are included in the Lake Erie Coastal Economy. Dashed line indicates Canadian and U.S. border.

Table 1: Counties in Lake Erie Coastal Economy

Ohio		Michigan	Pennsylvania	New York
Ashtabula	Lorain	Monroe	Erie	Chautauqua
Erie	Ottawa			Erie
Lake	Sandusky			

Lucas and Cuyahoga Counties in Pennsylvania, and Wayne County, Michigan are omitted because they are home to Toledo, Cleveland, and Detroit.

2.3 Lake Erie Yellow Perch Management Background

Commercial and recreational fishing has been present on Lake Erie as long as historical records have been kept (Cowan & Paine, 1997; Ebener et al., 2008; Kerr, 2010). Management of the fisheries has varied greatly over the years from both ecological and economic standpoints. The current regulations exist because of past experiences; lack of management, over-fishing, invasive species, and market changes have all had an impact on bringing the management regime to its current state (Kerr, 2010).

There are, in effect, two branches of regulators governing the fishery activities on Lake Erie: province and state departments, and a joint commission between managers in the U.S. and Canada. The GLFC is a joint committee between the United States and Canada responsible for estimating the stock of fish in the Lake and setting appropriate quota limits and season length. The GLFC was originally established in 1954 from the bilateral Great Lakes Fisheries Convention between the U.S. and Canada as a way to control sea lamprey, and has been utilized since as a way to coordinate Great Lakes management (Hubert & Quist, 2010). Each country is responsible for deciding how they allocate their portion of the TAC. The regulations include setting the requirements for fishing licenses, quotas, royalties, selling policies, and data recording requirements (Cowan & Paine, 1997; Ohio Department of Natural Resources Division of Wildlife, n.d.). The policies vary between the countries. In the case of the United States, each state has the ability to set their own regulations for their waters.

Ontario, the only Canadian province that borders Lake Erie, operates its commercial fishery under an ITQ system that was put in place in 1984 (Cowan & Paine, 1997). Under this system, to commercially fish the Canadian waters of Lake Erie one must possess an ITQ for the species being targeted and for possible by-catch species (Cowan & Paine, 1997). Fishers are able to trade, sell, or buy the ITQs. Ontario fishers are also able to use gillnets to harvest their catch. Historically gillnets have been the preferred net choice of fishermen because they are relatively inexpensive to obtain, but they can result in a lot of non-targeted by-catch (Johnson et al., 2004). Gillnets are still used in the Ontario waters of Lake Erie today.

On the U.S. side, commercial fishers are required to hold a fishing permit, but there is no limit on how much each fisher is allowed to catch (Ohio Department of Natural Resources Division of Wildlife, n.d.). While there are slight differences between each state's commercial fishing regulations, the overall idea is the same; licenses are required to fish, but there is no limit on how much each license holder can harvest, just that the industry as a whole must stay below the U.S. portion of the total allowable catch. To obtain an annual license in Ohio, applicants must put up a $1000 bond, be an Ohio resident for at least 90 days, and show two years of experience with commercial fishing equipment (Bill Analysis, 2007). License holders are required to submit records of daily catch (Ohio Department of Natural Resources Division of Wildlife, n.d.). Over the years, the Lake Erie states have implemented policies outlawing the use of gillnets.

Recreational fishers in the U.S. must also obtain a license to fish and are required to abide by the daily bag and size limits of the state who's waters they are fishing. Annual commercial fishing licenses in Ohio currently range from $19.00 to $40.00 depending on state resident status (Ohio Department of Natural Resources Division of Wildlife, 2017)

2.4 Great Lakes Invasive Species Background

The Great Lakes, in general, and Lake Erie, in particular, are under constant threat by invasive species. This section provides an overview of those threats, building on work in Johnson (2013).

Non-native species become 'invasive species' when they cause or are likely to cause economic and or ecological damage (Vásárhelyi & Thomas, 2003; T. Warziniack et al., under review). The most notable invasive species in the Great Lakes are zebra mussels (*Dreissena polymorpha*) and sea lamprey (*Petromyzon marinus*). Sea lamprey are thought to have been introduced through the St. Lawrence Seaway and began to take hold in the Great Lakes in the 1940s and 1950s (Koonce et al., 1996). They are blamed for the collapse of native populations of lake trout and lake whitefish, among others, which allowed the populations of small marine invaders, alewife and rainbow smelt, to increase dramatically, disrupting the fish community further (Koonce et al., 1996). Sea lamprey control and stocking efforts have allowed the native fish population to rebound, but these controls cost the region approximately $20 million annually (Great Lakes Commission & St. Lawrence Cities Initiative, 2012).

It is expected that the first zebra mussels were introduced to the Great Lakes in 1986 through ballast water discharge (Griffiths et al., 1991). By 1991 the species had spread to most of the Great Lakes (Griffiths et al., 1991). Zebra mussels filter the water, clearing it of many of the nutrients that other fish depend on for survival. Zebra mussels also attach to underwater surfaces and can inhibit electric power generation, drinking water treatment and other water intakes (Connelly et al., 2007). In 1994 it was estimated that zebra mussels cost the Great Lakes Region $120 million for damages done to these water dependent industries (Connelly et al., 2007).

Current estimates indicate that zebra mussels cost the region $300-$500 million annually (Great Lakes Commission & St. Lawrence Cities Initiative, 2012).

In the past decade, Asian carp, a catchall term for a group of carp species, have started to gain attention as a potentially large-impact aquatic invasive species. Asian carp were introduced to the United States by humans as a way to control algae in fish ponds in the southern United States. In the 1990s flooding in the lower Mississippi River gave Asian carp access to the river. Within ten years of reaching the Mississippi river the carp had spread 1000 miles. Commercial fisheries in the Illinois River regularly catch up to 25,000 pounds of carp every day (Great Lakes Commission & St. Lawrence Cities Initiative, 2012). Asian carp are a concern because their ability to consume large quantities of zooplankton and phytoplankton, the foundation of the food web, allows the Asian carp to out compete native species, many of which support commercial and recreational fisheries (Just, 2011). Some ecologists suspect that Asian carp are currently established 55 miles from Lake Michigan (Great Lakes Commission & St. Lawrence Cities Initiative, 2012). Others have found evidence that Asian carp might already have crossed into Lake Michigan through the Chicago Area Water System (Jerde et al., 2011).

While not all non-native species introduced into the Great Lakes become invasive, those that do can have implications for the greater ecosystem. Of the non-indigenous species in the Great Lakes, approximately 10% have caused severe economic and ecological problems (Vásárhelyi & Thomas, 2003). Non-indigenous species cause problems through predation of native fish, by introducing disease or parasites, and by altering habitats for native species and recreational sue (Vásárhelyi & Thomas, 2003). Invasive species are usually difficult to eradicate and can have negative impacts on community composition, ecosystem function, and to human services such as water treatment (Briski et al., 2012).

While the ecosystem disturbances from invasive species are serious, the costs of control and prevention are also significant. In 1995, $17.75 million was spent on zebra mussel control in North America alone (Vásárhelyi & Thomas, 2003). O. Oemcke & van Leeuwen, (2004) report that between 1986 and 1995 zebra mussels cost the United States $5 billion, not just from treating the mussels, but from consequences of the mussels blocking intakes at power plants, water treatment plants, and fouling fishing nets, boat hulls, and buoys. These costs have increased over the years. In 2012, it was reported that in the Great Lakes alone, $20 million is spent annually to control sea lamprey, $150 million is devoted to managing introductions via ballast water discharge, and $300-$500 million is currently spent on zebra mussel control and damages (Great Lakes Commission & St. Lawrence Cities Initiative, 2012). In 2010, a chain-link fence was installed at a cost of $200,000 as a temporary measure to prevent movement of Asian carp between the Wabash and Maumee Rivers and Lake Erie during flood events (Hebert, 2010; Markey, 2012; O'Keefe, 2013). There is also a proposal to re-construct the Chicago Sanitary and Ship Canal to separate the Mississippi River Basin and Lake Michigan to prevent the spread of invasive species, including Asian carp to Lake Michigan, with estimated costs between $4 billion and $9.5 billion (Great Lakes Commission & St. Lawrence Cities Initiative, 2012).

2.5 Social Accounting Matrix

The foundation for the models used in this analysis is the social accounting matrix, or SAM. A SAM is a succinct and comprehensive way of representing economic activity in equilibrium of a region in a particular year (Wing, 2004). A SAM is a square matrix in which each account is represented by both a row and a column (Lofgren, Harris, & Robinson, 2002, p. 3). The accounts in the SAM are the sectors of interest, factors of production, households, government transfers, trade, and taxes to vary degrees of detail. The value in a specific cell within the matrix is the payment from the column account to the row account, so that receipts are shown along the rows and the payments are shown along the columns (Lofgren et al., 2002, p. 3). SAMs must be "balanced," in that the sum of an account's row is equal to the sum of its column, or that total expenditures are equal to total revenues. Because I am interested in the role of ecosystem services in the economy an ideal SAM would include these values. Such data, however, does not readily exist so innovative steps were taken. This section describes the construction of the SAM, how species-level sectors were created out of existing data, and how ecosystem services were included in the SAM.

The data for the Lake Erie SAM is taken from 2013 IMPLAN county-level data. Though deficient in some areas, it is more comprehensive than any other source and provides a decent building block on which to add. IMPLAN differentiates households by income class, and this specification was maintained. The federal and state government interactions were kept distinct. The differentiation of foreign and domestic trade was also maintained. The full Lake Erie Coastal Economy SAM can be found in the appendix.

Table 2: IMPLAN Social Accounting Matrix Sectors
Sectors provided by IMPLAN

Agriculture	Labor	Household 7
Commercial Fishing	Capital	Household 8
Air Transportation	Indirect Taxes	Household 9
Rail Transportation	Household 1	Federal Government
Water Transportation	Household 2	State Government
Truck Transportation	Household 3	Inventory
Power Generation Supply	Household 4	Foreign Trade
Water-Sewage and other Systems	Household 5	Domestic Trade
Seafood Processing	Household 6	

Table 3: Commercial Fishing Sectors
Sectors disaggregated from IMPLAN Commercial fishing sector.

Bigmouth Buffalo	Channel Catfish	Lake Whitefish	White Bass	Yellow Perch
Carp	Freshwater Drum	Walleye	White Perch	Other Species
Recreational Fishing				

2.5.1 Sector Disaggregation

Most of the industry sectors included in the SAM are aggregates of IMPLAN's 536 sectors, collapsed down to 20 sectors of interest in this analysis. These sectors are shown in Table 2. However, a few sectors, like recreational and commercial fishing had to be constructed using other data. Those sectors are shown in Table 3.

The species-specific commercial fishing sectors were disaggregated from IMPLAN's commercial fishing sector using USGS Lake Erie Landing report from 2013 (National Oceanic and Atmospheric Administration & U.S. Geological Survey, 2013). Of the 18 species of fish that the USGS reports the commercial landings of, nine were chosen to be disaggregated into separate sectors in the SAM; eight of these species account for the greatest proportion of the total commercial harvest. Walleye was included because it is a prized species and is heavily regulated. The 2013 harvest levels are shown in Table 4. To disaggregate these species from total commercial fishing, it was assumed that proportions of the of total harvest calculated using the USGS data were accurate and held true for the IMPLAN data. For example, because yellow perch account for 64% of total harvest in the USGS data, I attributed 64% of every activity in commercial fishing to the newly created yellow perch sector. This process was repeated for each of the species studied in detail. After these values were subtracted from the commercial fishing sector, I assume all remaining activities can be attributed to the fishing of other species both in Lake Erie and the surrounding lakes.

Table 4: USGS Lake Erie Commercial Landings
Highlighted species included in SAM.

	Species	Pounds Harvested	Dollar Value	Percent of Total Harvest
1	Yellow Perch	1,547,199	2,973,980	64.22
2	White Bass	741,959	420,384	9.08
3	White Perch	659,216	225,392	4.87
4	Channel Catfish	564,070	225,006	4.86
5	Bigmouth Buffalo	379,084	194,116	4.19
6	Freshwater Drum	526,894	121,183	2.62
7	Carp	402,925	109,089	2.36
8	Lake Whitefish	63,940	95,729	2.07
9	Quillback	284,375	83,595	1.81
10	Minnows	23,735	80,224	1.73
11	Goldfish	103,685	73,490	1.59
12	Brown Bullhead	50,796	14,623	0.32
13	Gizzard Shad	41,246	8,028	0.17
14	Suckers	28,790	3,613	0.08
15	Burbot	1,119	1,546	0.03
16	Walleye	271	925	0.02
17	Rockbass	129	0	0
18	Smallmouth Bass	111	0	0

NOAA & USGS. (2013). Great Lakes Commercial Fishery Landings.

Another important sector to this region that was not independently identified in the IMPLAN data is recreational fishing. The recreational fishing sector captures the market activities recreational fishing brings to the region. Recreational fishing was disaggregated from the composite miscellaneous sector using data from the 2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior, U.S. Fish & Wildlife Service, & U.S. Department of Commerce, U.S. Census Bureau, 2014). The survey reports daily expenditures of fishing the Great Lakes and the number of days fishing Lake Erie, from which I calculated annual expenditures of fishing Lake Erie of $9,972,180. These expenditures include the amount spent on trips and equipment. These expenditures were disaggregated from the miscellaneous sector in a manner similar to the commercial fishing disaggregation.

2.5.2 SAM and Biomass Reconciliation

One of the major contributions of this dissertation is the use of fish stock as an input to production. Traditionally, social accounting matrices only include marketed inputs. Little guidance for extending a SAM to include nonmarketed inputs is provided by the literature, so additional steps needed to be developed to reconcile the natural resource's contribution to production and the economic activities described in the SAM.

While fish stock biomass is an input to production in fishing sectors, it is not included in the original IMPLAN data. To account for biomass as an input to production and to maintain a balanced SAM, that is, a SAM in which demand is equal to supply across all accounts, I draw on three key assumptions about the values in the SAM. Those assumptions are: 1) the value in each cell represents the total dollar of activities transferred between accounts, 2) the initial price of each good or commodity without a natural resource input is $1, and 3) the output price of commodities with a natural resource input is not necessarily $1, but rather is the value that ensures that total revenue in the sector is equal to total costs. I use the yellow perch sector and its effort demand as an example to further describe how these values are reconciled.

Table 5 shows the yellow perch input demands from other sectors used in production from the Lake Erie Coastal Economy SAM. Yellow perch spends \$10,000 on both rail and water transportation, \$20,000 on truck transportation, \$9,090,000 on miscellaneous, \$320,000 on labor, \$3,080,000 on capital, and pays \$2,040,000 in business taxes. Summing across these payments gives the total effort payment by the yellow perch sector, equal to \$14.57 million. To maintain zero profit, it must be the case that total revenue, $P_f H_f$, where P_f is the output price of yellow perch, and H_f is the total harvest from either the Great Lakes Fishery Commission or the Ohio DNR depending on the species, is equal to total costs in the yellow perch sector of \$14.57 million. The benchmark price of yellow perch, P_f, can then be calculated as: $P_f = \frac{\$14.57 million}{H_f}$.

The key to this step is, while units of goods in a traditional SAM are generally thought to be arbitrary, and thus defined so benchmark prices are 1, this is not the case for fishing sectors in this model. Units of the fishing good are equal to the units in the harvest data.

Table 5: Effort Portion of Social Accounting Matrix
Yellow perch effort input demand from other sectors. Values measured in millions of dollars.

	Yellow Perch Payments
Rail Transportation	0.01
Water Transportation	0.01
Truck Transportation	0.02
Miscellaneous	9.09
Labor	0.32
Capital	3.08
Taxes	2.04

As stated earlier, an ideal SAM, for the purposes of the models used here, would include ecosystem services (species stock). This is not the case; the SAM created with IMPLAN data only includes monetary flows in the economy, with no mention of nonmarket inputs. The most straightforward way to think about the problem is that fish stock is an unpriced input to production, or that its benchmark price equals zero. The result is that the production function that combines effort and stock to produce harvest cannot be estimated using the SAM, as is traditionally done in computable general equilibrium modeling (note, this complication does not affect the aggregate of IMPLAN data to produce the human-provided input effort). In the case of fishing in Lake Erie, catch is constrained by policy, and the stock input is exogenously provided by nature. This requirement for additional data in estimating the production function is one of the primary reasons using harvest as the unit of measure in the fishery is necessary. The specifics of estimating the fishery production functions are described in the next section.

2.6 Harvest Function Parameter Estimates

Cobb-Douglas production functions are used to represent the harvest of all fish species in the model:

$$H_f = q_f E_f^{\alpha_f} S_f^{\beta_f} \qquad (1)$$

H_f is the fish species-specific harvest, E_f is the effort employed to catch species f, S_f is the species stock , q_f is the catchability or efficiency coefficient, and α_f and β_f are the species specific effort and stock elasticities of harvest. Stock is assumed to be exogenous to the fisher, leaving them to choose optimal effort to produce harvest. Because stock is not included in the SAM, these parameters cannot be estimated from the benchmark data and must be obtained via alternative means. I estimate Cobb-Douglas harvest function catchability coefficients and elasticities for yellow perch, walleye, white perch, white bass, and lake whitefish using data from the GLFC and the Ohio Department of Natural Resources. For all species, I use the natural log version of the Cobb-Douglas harvest function (Equation 2) in the estimate of the Cobb-Douglas parameters.

$$ln\, Harvest_{f,y} = \gamma_o + \gamma_1 ln\, Effort_{f,y} + \gamma_2 ln\, Stock_{f,y} + \epsilon_{f,y} \qquad (2)$$

In Equation 2, *y* represents the year, and *f* indicates the fishing firm and species. The coefficients of the natural-log version of the Cobb-Douglas harvest function are estimated, where γ_o is the natural log of the yellow perch firm's q_f, and γ_1, and γ_2 are the estimates of the yellow perch firm's α_f and β_f respectively. The assumption here is that observed effort is the result of some optimal decision in the background.

2.6.1 Yellow Perch Data

I estimate two production functions for yellow perch, one that represents U.S. fishers, and one that represents Canadian fishers. I estimate two production functions to capture how their differences impact their harvest. Recall from Chapter 1 that both countries share the TAC, but have different regulations in their gear and licensing practices. I explain the estimate of the Cobb-Douglas parameters first for the U.S. fishers.

Harvest, stock, and effort data for yellow perch and walleye were obtained from annual reports published by the GLFC. The U.S. yellow perch data includes millions of pounds of harvest and number of trapnet lifts in the Ohio regions of Lake Erie, and millions of pounds of yellow perch in Lake Erie from 1981-2015. The Ohio regions are used for the estimates of the U.S. yellow perch parameters because fishers in Ohio are consistent in their gear choice across the entire time frame, and they make up the greatest and most consistent proportion of the yellow perch fishery. Harvest and total allowable catch are not always equal (Lake Erie Yellow Perch Task Group, 2016). I believe that harvest is a better indicator of the fisher's behavior than the TAC, so for the purposes of this estimation I use harvest as the dependent variable; this approach follows the estimation techniques of Bjørndal & Conrad (1987), Campbell (1991) and Zhang & Smith (2011).

To estimate the coefficients of the Canadian harvest function, millions of pounds of harvest, kilometers of gillnet used in Ontario regions of Lake Erie, and millions of pounds of yellow perch are used. While data is also available for Ontario from 1981-2015, Canada enacted the individual transferrable quota system in 1984, so I restrict the time series to 1984-2015. Besides licensing regulations (a ROA system versus an ITQ system), another key difference between the Ohio and Ontario fisher are the nets they use. Ohio fishers use trapnets and Ontario fishers use gillnets. The effort measure in Ohio is the number of times a trapnet is lifted, the effort measure in Ontario is the kilometers of nets used. I address the different measures of effort before estimating the Cobb-Douglas parameters with the Ontario data.

Melville-Smith et al., (1999) demonstrate a methodology for converting gillnet effort to trap net effort. The authors used data on crab harvest and effort from Western Australia to estimate the effort ratio between gillnets and trap nets. Their method includes calculating the mean harvest for a given length of net and the mean harvest for a given number of lifts, they then divide the mean gillnet harvest by the mean trap net harvest. I follow their method to standardize the effort measures in the data set. I started by finding the grand mean of gillnet harvest, gillnet effort, trap net harvest and trap net effort. The grand mean entails finding the average over all time periods and units. Next these averages were used to find the average catch per average unit of effort. The mean catch per kilometer of gillnet is 1,415 lbs/km. The mean catch per lift of trap net is 81.73 lbs/lift. Dividing these means indicates that 17 trap net lifts is equivalent to one kilometer of gillnet. To generate a standardized effort measure gillnet effort is multiplied by 17. The standardized effort measures were then used in the Cobb-Douglas harvest function estimation. The same effort measure across countries would be ideal, but given the other similarities Ontario and Ohio have (same fish species, same lake, same region), this conversion allows a way to estimate the Cobb-Douglas parameters for the two regions.

2.6.2 Walleye Harvest Data

The walleye data includes number of fish harvested and kilometers of gillnet used in the Ontario region of Lake Erie and the estimated number of walleye in the entire lake from 1980-2016. The commercial harvest of walleye in the U.S. has become limited in recent years, so U.S. data is not available. While it is not ideal to use Canadian values for the estimation of the U.S. harvest of walleye, the production of walleye is limited, the impact in the potential error of estimation in the CGE simulations is expected to be small. Recall that walleye is included in the model because it is both highly sought after by recreational fishers and highly regulated, as opposed to being a large commercial fishery.

2.6.3 Other Species Data

Three other species I estimated Cobb-Douglas parameters for include white perch, white bass, and lake whitefish. The data for these species was obtained from the 1999, 2003, 2012, and 2015

Lake Erie Status Reports provided by Ohio's Department of Nature Resources (ODNR). Some of the data in the ODNR Lake Erie Status Reports need to be transformed to make it useable for estimating the Cobb-Douglas parameters. Annual harvest for each of the species was reported, but I only had access to the method of that harvest (i.e. net type) for 1999, 2003, 2012, 2015. I decided to focus on trapnet harvest to be consistent with the yellow perch fishery, and used the available data to calculate an average portion of total harvest attributed to trap net fishing for each species, and used that to derive an annual 'trap net harvest.' The reports also provided the annual trap net catch rates (pounds per lift) for each species, which allowed me to calculate the annual trap net effort (number of lifts) for each species. Unfortunately, the ODNR does not estimate lake abundance of species to the same extent that the GLFC reports provide, but they provide data from their trawl surveys. I use ODNR August trawl values, which provide the arithmetic mean catch-per hectare of age 1 and older fishes. To infer a relative abundance from the trawl surveys in the same units as the harvest data, the trawl data was transformed from number of fish to weight of fish. The reports included the mean length for white perch and lake whitefish for age-0 fishes, and age-2 fishes for white bass from the trawl surveys. While using the lengths of only one age class is likely not capturing the true mean length of the entire species population, it allowed me to calculate the average weight of each species by using the length-weight relationship represented by Equation 3, where for each species the log weight in grams is equal to some linear equation of length with a species-specific intercept and slope. The ODNR reports included their estimates of c_f and d_f for each species of interest.

$$\log(Weight_f) = c_f + d_f \log(length_f) \qquad (3)$$

Through these steps, I was able to derive an estimate of the total weight of species reported in the trawl surveys. While this is does not provide a true population estimate for the entire lake, it does provide a relative abundance for each year to use as a proxy for 'fish stock.' In estimating the Cobb-Douglas parameters for white perch, white bass, and lake whitefish, I used millions of pounds of harvest, number of trap net lifts, and relative abundance in millions of pounds from 1990-2015.

2.6.4 Harvest Parameter Estimates

Durbin-Watson and Breusch-Godfrey tests were performed for each species to test for auto-correlation. Both tests have a null-hypothesis of no autocorrelation. For each species the tests were performed for different lags, and in all cases except lake white fish, the null was rejected. The estimated coefficients and the lags of autoregressive integrated moving average models for each species is shown in Table 6. Parameter values from these estimated coefficients are used in the following chapters and are discussed further in those contexts.

Table 6: Cobb-Douglas Harvest Parameter Estimates

VARIABLES	Yellow Perch Ohio	Yellow Perch Ontario	Walleye	White Perch	Lake Whitefish	White Bass
γ_1	0.763***	0.275***	0.647***	0.307***	0.881***	-0.354
	(0.124)	(0.0595)	(0.041)	(0.0938)	(0.251)	(0.367)
γ_2	0.225*	0.186	0.464***	0.0662**	0.235**	0.185**
	(0.124)	(0.130)	(0.061)	(0.0330)	(0.116)	(0.0737)
γ_o	-8.288***	-2.629***	0.031*	0.391	1.107	-4.814**
	(1.397)	(1.000)	(1.043)	(0.545)	(1.386)	(1.947)
Observations	35	32	37	26	26	26
ARIMA Lags	AR2 MA1	AR2 MA1	AR3 MA3	AR4 MA2	MA1	AR2 MA4

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

2.7 Lake Erie Harvest Restrictions

I include fishery regulation in both of the models presented in Chapters 3 and 4. Two species of fish in Lake Erie are heavily monitored and regulated, yellow perch and walleye. In this section I describe the equations I use to represent the TAC of yellow perch and walleye in my models.

2.7.1 Yellow Perch Total Allowable Catch

Catches of yellow perch from Lake Erie are currently regulated by a TAC harvest limit. The TAC limits harvest annually for the entire lake and is set by managers in the region through the facilitation of the Great Lakes Fishery Commission (GLFC). The TAC is based on population estimates of the species (Lake Erie Yellow Perch Task Group, 2016). The Lake Erie Yellow Perch Task Group (YPTG) has set target fishing (harvest) rates, F_m, for each management unit that they believe "support viable sport and commercial fisheries without inviting excessive biological risk," (Lake Erie Yellow Perch Task Group, 2016, p. 8). Using the fishing rates and the U.S. proportions of Lake Erie surface area calculated by the YPTG (Table 7), I derive a function that approximates the TAC for the U.S. proportion of Lake Erie for a given biomass of the yellow perch (Equation 4), where the subscript yp represents yellow perch.

Table 7: Lake Erie Yellow Perch Fishing Rates and U.S. Management-Unit Proportions

Management Unit	U.S. Proportion of Surface Area (R_m)	Fishing Rate (F_m)
1	0.0886	0.670
2	0.1644	0.670
3	0.1704	0.700
4	0.0804	0.300

(Proportions and fishing rates taken from (Lake Erie Yellow Perch Task Group, 2016))

$$TAC_{yp} = S_{yp} \sum_m R_M F_m = 0.31291 S_{yp} \quad (4)$$

In this model, the U.S. TAC for Lake Erie yellow perch is determined by the fishing rate (F_m), where *m* indexes the Lake Erie management unit, the U.S. surface area in each unit (R_m), and the total stock of yellow perch in Lake Erie (S_f). The U.S. proportion of surface area is calculated against the entire lake in this case, so that the yellow perch biomass of the entire lake can be used. The Yellow Perch task group is more precise in their TAC calculations and calculates the TAC for each management unit, but for the purposes of this model, the entire lake TAC is used.

The yellow perch TAC is shared among commercial and recreational fishers. To my knowledge, there is no formal method for dividing the TAC for recreational and commercial fishing. To incorporate the harvest limit on both recreational and commercial fishing, I assume that each type of fishing gets a proportion of the TAC. I use historical harvest data from the Great Lakes Fishery Commission to find the mean proportion of harvest for each type of harvest, and set a TAC for each type of fishing. On average, recreational fishing harvests 62.8% of the total Lake Erie yellow perch harvest, and commercial fishing harvests 37.2%. The recreational fishing TAC ($TACR_{yp}$) is shown in Equation 5 and the commercial TAC ($TACC_{yp}$) is shown in Equation 6.

$$TACR_{yp} = .628\, TAC_{yp} = 0.19651 S_{yp} \quad (5)$$

$$TACC_{yp} = .372\, TAC_{yp} = 0.11640 S_{yp} \quad (6)$$

2.7.2 Walleye Total Allowable Catch

The harvest of walleye from Lake Erie is also regulated by managers through the facilitation of the GLFC. The Walleye Task Group (WTG) estimates walleye abundance and recommends a total allowable catch annually. The walleye TAC is based on different rules than the yellow perch TAC, and has varied over the years. The current TAC formulation sets the TAC such that the fishing mortality rate is equal to 60% of the maximum sustainable yield (MSY) fishing rate (Lake Erie Committee & Great Lakes Fishery Commission, 2015). The WTG uses statistical catch-at-age population estimates to estimate the walleye population and MSY every year. In order to represent walleye TAC with an equation in my simulations, I use the 'rule of thumb' equation for MSY (Hubert & Quist, 2010, p. 461) as shown in Equation 7 where w represents walleye, p_w is an empirical factor related to the age at which fish become susceptible to fishing, M_w is the instantaneous rate natural mortality of the fish species, and S_w is the fish stock.

$$MSY_w = p_w M_w S_w \tag{7}$$

In a typical MSY equation the stock level used is the maximum stock level attainable, in other words, the stock level that would exist if no fishing occurred. I do not have access to that information, so instead I use the projected fish stock. I calculated the average p_w from historical data reported by the GLFC from 2011-2017, so $p_w = 1.66$, which is likely higher than if the no-fishing level of biomass was available. In their projections, the WTG sets the instantaneous natural mortality rate, $M_w = .32$, so I do the same. With MSY calculated, I am able to solve for the fishing mortality rate that would achieve the MSY (F_{MSY}) and the fishing mortality rate that the WTG sets as the target rate, which is 60% of the MSY level, shown in Equations 8 and 9 respectively.

$$F_{MSY\ w} = \frac{MSY_w}{S_w} \tag{8}$$

$$F_{60MSY\ w} = .60\,\frac{MSY_w}{S_w} \tag{9}$$

In setting the TAC for walleye, the GLFC WTG uses the overall target fishing mortality rate (F_{60MSY}) and the selectivity of the different age groups (how susceptible different age fish are to harvest), to calculate the fishing mortality rates of each age group. The selectivity of the age groups change every year. For the purposes of my simulations I assume they are constant, and use the mean of the selectivity values presented to find the fishing mortality rate of walleye juveniles (ages 1-2) and adults (age three and older). This approach allows me to find the exploitation rate of each age group (assuming a constant survival rate). Finally, the TAC is found by multiplying the exploitation rate for each age group by its estimated stock size. The walleye TAC for all of Lake Erie fishing is defined by Equation 10, where *w* represents walleye, and SJ_W and SA_W are the stock levels for the juvenile walleyes and adult walleyes respectively (the targeted ages).

$$TAC_W = \frac{F_{60MSY}\,(.086SJ_W + .412SA_W)}{M_f + F_{60MSY}} \tag{10}$$

The WTG allocates the TAC to Ontario, Michigan, and Ohio based on the surface area they control. Michigan is allocated 5.83% of the TAC and Ohio is allocated 51.11%, so the TAC attributed to U.S. is shown Equation 11.

$$TACR_W = \frac{F_{60MSY}\,(.086SJ_W + .412SA_W)}{M_f + F_{60MSY}}.5694 \tag{11}$$

The GLFC and ODNR currently report no commercial fishing of walleye, however the USGS report some commercial landings of walleye that are included as a sector in the SAM, to account for the commercial harvest of walleye reported by the USGS, I set the commercial TAC of walleye equal to the 2013 commercial landings reported by the USGS. USGS reports the number of fish harvested. I convert this weight using the mean lengths reported by the ODNR for walleye age 5-6, and the length-weight regression equations reported by the ODNR in their 2015 status report. Equation 12 shows the commercial TAC of walleye.

$$TACC_W = 986.97\ lbs \tag{12}$$

2.7 Other Data

Up to this point, I have described the geographic region, estimation of elasticities for the commercial fishing sectors, and the walleye and yellow perch regulation. Other data required for the simulations includes elasticities for other sectors and agents, ecological values and parameters, and information on household fishing preferences. Those data and parameter values are model specific and are discussed in their respective chapters.

CHAPTER 3: GENERAL EQUILIBRIUM PRIVATIZATION OF A REGULATED FISHERY

3.1 Introduction

Economic theory and experience tell us that when fisheries are left unmanaged, the results are sub-optimal with inefficient levels of harvest and effort (Gordon, 1954; Samuelson, 1974; Weitzman, 1974). A number of methods have been proposed to correct the market failures that occur with the open-access fishery. Among those proposed, the ITQ system has been shown to be one of the most effective (Libecap, 2007). By assigning long-term enforceable property rights, a better outcome can be achieved with limited government intervention (Arnason, 2007; Grafton et al., 2006; Homans & Wilen, 1997; Libecap, 2007). The ITQ system aligns individual incentives with the goal of maximizing the aggregate value of scarce resources (Arnason, 2012; Libecap, 2007; Marchal et al., 2009). Fishers take into account the full cost of their harvest and choose more efficient levels of effort and harvest. The implementation of ITQs also correct the failure of too much labor being employed by establishing property rights and removing the 'race to fish' that exists in open access fisheries (Grafton et al., 2006). By ensuring more secure fishing rights, effort can be spread across a season rather than employed intensively in derby-style fishing. In practice, however, the most common regulations rely on restrictions on harvest, season lengths, or technology. Under these regulated open access management regimes, controls are put in place, but free entry and exit still exist. While in some cases these regimes can be effective at achieving ecological goals, they lead to economic inefficiencies (Boyce, 2004; Deacon, Finnoff, & Tschirhart, 2011; Grafton et al., 2006; Homans & Wilen, 2005). In fisheries regulated by aggregate quotas, fishers incorrectly attribute the production value of fish stock to factors of production, inducing a higher level of factors to be used than necessary, and distributing the resource value to them. When those quotas become tradeable, rents go to the owners of quotas – most often the owners of the vessels. The tradeable quota system requires owners of vessels to internalize their cost of harvest, so they demand less factors of production than under open access and regulated open access regimes. The release of factors used in the fishery allows the factors to be used in other sectors. Fishery management systems and the generation of resource rents have been studied extensively in models with one or few sectors starting with Samuelson (1974) and Weitzman (1974). However, less attention has been given to the welfare implications of fishery management systems in a general equilibrium setting with many commodity sectors. In a general equilibrium setting, the supply and demand of factors in any one sector depend on other markets, and while theory and partial equilibrium fishery models show that fishery effort will decrease under a transferable quota system, the economy-wide impacts of the change in effort employed has not been evaluated. Herein, I show that correcting the market failure can have welfare impacts and generate changes in other sectors and can have welfare implications beyond the rents that would be captured in a partial equilibrium model. In order to explore the impacts of moving from a regulated open access management regime to an individual transferrable quota system, a computable general equilibrium model of the Lake Erie region and yellow perch fishery is used.

The Lake Erie yellow perch fishery is used as a case study because of its interesting regulatory context. Lake Erie has waters in both the United States and Canada, and the regulations the fishermen face differ by country. The Great Lakes Fishery Commission (GLFC), a committee with managers from both the U.S. and Canada, estimate Lake Erie fish stocks. In the case of yellow perch, they also set a TAC that is split between the countries. In addition to following the TAC for yellow perch, Ontario, the only Canadian province that borders Lake Erie, operates under an ITQ system that was put in place in 1984 (Cowan & Paine 1997). Under this system to fish commercially on the Canadian waters of Lake Erie, one must possess an ITQ for the species being targeted and for possible by catch species (Cowan & Paine 1997). Fishers are able to trade, sell, or buy the ITQs among themselves. On the U.S. side, fishers are required to hold a fishing permit, but there is no limit on how much each fisher is allowed to catch, or how many permits are issued (Ohio Department of Natural Resources Division of Wildlife, 2017). While the Canadian and U.S. management of the yellow perch fishery vary in more ways than one (gear restrictions and allocation of quota between commercial and recreational fishing are two examples), the ITQ system used by Canada is thought to have led to stronger Canadian fishery through its establishment of consistent assignment of property rights to fishers (Cowan & Paine, 1997). I use data from the U.S. yellow perch fishery to simulate the base scenario of a regulated open access. Data from the Canadian yellow perch fishery is used to inform parameter values when the movement to an ITQ scenario is simulated.

The paper proceeds as follows: I review the literature on general equilibrium modeling of fisheries in section two, in section three, the model and the variants in regulation will be described, in section four, Lake Erie's yellow perch fishery is introduced and the calibration and data is described, and finally section five presents the policy analysis.

3.2 Literature Review

To investigate welfare impacts of privatizing a fishery, I use a computable general equilibrium (CGE) model that simulates a small open economy. Using a CGE model allows for the full impacts of privatization to be estimated. The model incorporates multiple sectors, endogenous prices, and factors to represent the interactions that occur in agents' choices in the economy. A review of others' techniques of using CGE models to evaluate natural resource problems is presented here.

CGE models have been widely used to evaluate impacts to fisheries from exogenous shocks to either the ecosystem or in government policies. Warziniack et al. (2011), for example, use a CGE model to analyze the effectiveness of invasive species policies, and compare the welfare impacts across general equilibrium and partial equilibrium settings. While they do not focus specifically on commercial fishing, they find that the welfare losses of an invasive species are over estimated when using partial equilibrium models. In their follow up paper, Warziniack et al. (2013) look specifically at welfare effects of tax policies; their paper focuses on the spread of an invasive species through human-mediated vectors associated with recreational fishing, ignoring commercial fishing. In another paper, Warziniack et al. (2017), present a model of the consumer's recreational fishing choices that accounts for consumer preferences for fishing trips as an activity and their preference for specific species. This model also allows for impacts from exogenous shocks, either through a change to a specific species population, or through changes to the tax structure. In setting up the consumer model to include consumer preferences for species, the model goes beyond using only economic activity data to model consumer preferences for ecosystem services. Jin et al. (2012) show a method to analyze the welfare impacts of changes in the food web by integrating a CGE model, but do not address various fishery regulations or the allocation of rents. Manning et al. (2014) examine impacts of an open access fishery on the surrounding community with a general equilibrium model. In their model, the lack of regulation not only impacts the fishery, but also other sectors from which factors are drawn.

A number of papers focus on methods for modeling a fishery sector in a CGE model to solely estimate the impacts of changes in economic conditions, or government tax policies (Avila-Foucat et al., 2009; Pan et al., 2007). Avila-Foucat et al., (2009), for example, include a fishery in a CGE model to estimate the impacts that changing conditions in the agricultural sector have on the fishery sector of the economy. In their model the fishery is modeled with a Gordon-Schaefer harvest function and no fishery regulation is imposed. Pan et al. (2007) argue that each stage of bringing fish to market should be its own sector, but aggregate most other activities into a few sectors. They disaggregate different levels of fishery production, including the harvester and processor into their own sectors. They do not focus on fishery regulations, but rather how changes in other sectors can impact the fishing sectors. My approach differs from Pan et al. (2007) in that I aggregate all fishing activities into a single sector by species, and include more disaggregated sectors of other activities. Setting the sectors up in this way allows the species-specific activities to remain contained in a single sector, while highlighting the linkages between the fishing sectors and other activities. Responses to changes in the fishing sectors will likely come as indirect changes to other sectors through the input markets, rather than changes in the processing of fish.

To a lesser extent, CGE models have also been used to analyze the regulations specific to the fishing industry, such as harvest restrictions. In their papers, Finnoff & Tschirhart (2003, 2008) develop models that incorporate ecological models with a CGE; these GEEM models included multiple species and total allowable catch functions. They show the economy-wide impacts in a general equilibrium setting of a change in harvest quotas of target species. Waters & Seung (2010) use a CGE to examine the impacts of a change in total allowable catch of Alaskan walleye Pollock. These papers describe various modeling techniques for including a fishery in a CGE model and simulating changes to fishery regulation, but do not provide techniques for moving from one regulation mechanism to another.

Another segment of the fishery-CGE literature focuses on accounting for the impacts the natural resource has on the effort markets. Manning et al. (2016) and Finnoff & Tschirhart (2008) incorporate this natural resource externality into a CGE. While slightly different in their approaches, these authors present methods for incorporating a fishing industry into a CGE that accounts for a difference in the value of the inputs employed in that sector versus other sectors in the economy. Finnoff & Tschirhart (2008) develop a CGE model with a fishing sector that faces different factor prices. Their factor price differentials are justified by assumptions made about a restricted fishing season and a majority of factors being employed from out of the region (Finnoff & Tschirhart, 2008). Manning et al. (2016) develop a CGE model of a Honduran artisanal fishery. Their model also includes fishery and non-fishery sectors. They show that fishing firms overvalue the inputs they use relative to their true marginal products, and thus inefficient input allocations are employed. The model I present here most closely aligns with work that focuses on the relationship between the harvest of a natural resource and effort, but I also account for different fishery regulations in the set-up of the economy.

3.3 Model

Consider an economy with production, consumption, government, and trade. Each commercial sector is modeled with a representative firm. Let I be the set of sectors with a representative firm. The set H contains nine households, each representing an income classification. Government receipts and payments are accounted for on both the federal and state level, where g represents the level of government. It is assumed a social planner minimizes costs to allocate imports and exports with both domestic and foreign trade partners. Trade regions are indexed over the set *t*.

3.3.1 Producer Behavior

There are two types of firms, firms in the commercial fishing sector whose production depends on a biological component like the abundance of fish in a lake, and firms whose production does not. All firms use a multi-level nested production process to combine capital, labor, and intermediate inputs to produce an input to production called effort. The production of effort is similar to the 'standard' production process presented in De Melo & Tarr (1992). CGE models like that described in De Melo & Tarr (1992) do not include natural resource inputs. I include fish stock as an input to production. In the commercial fishing sector effort is combined with a regional fish stock to produce the economic good, harvest. The commercial fishing sector also abides by regulations that restrict harvest. The details of the production decisions of firms are described below.

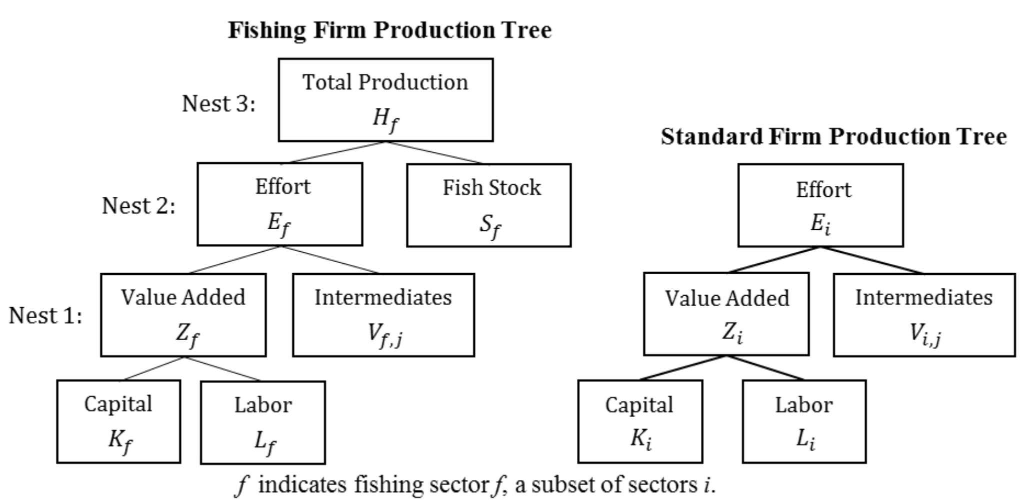

Figure 5: Firm Production Trees
The production nests for fishing firms and the standard firm from non-fishing sectors are shown. Fishing firms include stock as an additional input to production.

Regardless of sector, each firm behaves the same way in the lower two nests (Figure 5). For firms without a biological component in production there is no distinction between Nest 2 and Nest 3; their production is equal to the output of Nest 2. I assume constant returns to scale in each production nest, which allows me to solve for a firm's behavior one nest at a time. The inclusion of fish stock in Nest 3 is how this model incorporates the flow of ecosystem services depicted in Figure 3 (in Chapter 1). I begin by presenting the decision-making process in Nest 1 and Nest 2 which applies to all firms, and then proceed to the behavior in Nest 3 that is exclusive to the fishing firm.

Nest 1: Combining capital and labor to produce value-added

In Nest 1, the firm's objective is to minimize its cost of producing the value-added composite (Z_i), by substituting between labor and capital through a constant elasticity of substitution (CES) function. The firm has no factor market power, and so takes factor prices as given. The firm's cost minimization problem for Nest 1 is:

$$\min_{L_i,K_i} P_L L_i + P_K K_i \qquad s.t.\ \ Z_i = \varepsilon_{Zi}[\delta_{Li} {L_i}^{\rho_i} + \delta_{Ki} {K_i}^{\rho_i}]^{\frac{1}{\rho_i}}$$

L_i and K_i are the factor inputs of firm i, P_L and P_K are the prices of labor and capital, ε_{zi} is the efficiency parameter, δ_{Li} and δ_{Ki} are the firm's share parameters of each factor and sum to one due to the assumption that the function exhibits constant returns to scale. The parameter, ρ_i, is related to the firm's elasticity of substitution (σ_i) in the value-added function such that $\rho_i = \frac{\sigma_i - 1}{\sigma_i}$.

Taking first-order conditions of the firm's cost minimization and solving gives the firm's factor demand functions as a function of value-added:

$$L_i = (\frac{Z_i}{\varepsilon_i})(\frac{\varepsilon_i \delta_{Li} P_{Zi}}{P_L})^{\sigma_i} \quad (13)$$

$$K_i = (\frac{Z_i}{\varepsilon_i})(\frac{\varepsilon_i \delta_{Ki} P_{Zi}}{P_K})^{\sigma_i} \quad (14)$$

These factor demand functions represent cost minimizing labor and capital choices for production.

P_{Zi} is the firm-specific unit cost of value-added; for CES, the specification is:

$$P_{Zi} = \ (\frac{1}{\varepsilon_{Zi}})[\delta_{Li}{}^{\sigma_i} P_L{}^{(1-\sigma_i)} + \ \delta_{Ki}{}^{\sigma_i} P_K{}^{(1-\sigma_i)}]^{\frac{1}{1-\sigma_i}} \quad (15)$$

Equation 15 guarantees zero profits in the value-added nest of production.

Nest 2: Combining value-added and intermediate inputs to produce effort:

In Nest 2 of the production decision tree, all firms have a Leontief production function of their value-added (Z_i) and intermediate inputs from sector $j \in \mathrm{I}$ ($V_{i,j}$), so total value-added and intermediate inputs are used in fixed proportion to the total effort employed (E_i). The fixed ratio of effort needed for value-added and intermediate inputs are a_{Zi} and $a_{Vi,j}$ respectively. For any level of E_i, demand for value-added and intermediate inputs are:

$$Z_i = a_{Zi} E_i \quad (16)$$

$$V_{i,j} = a_{Vi,j} E_i \quad (17)$$

I allow for industry specific taxes to be paid by firm *i* at rate T_i on effort, which is calculated from the benchmark data. The Leontief assumption makes costs additively separable, so the unit cost of effort and the zero-profit condition for nest two of production is:

$$P_{E_i} = a_{Zi} P_{Z_i} + \sum_j a_{Vi,j} P_j + T_i \quad (18)$$

The sum of the firm-specific unit cost of value-added, the total cost of intermediate inputs, and taxes is equal to the marginal cost of effort.

Nest 3: Combining effort and stock of fish to produce harvest:

In Nest 3 of production, the choices of firms in fishing sectors and firms from other sectors diverge. The set of firms is divided such that, $o \subseteq I$, $f \subseteq I$, and $o \cup f = I$, where o represents firms from 'non-fishing' or other sectors and f represents firms from fishing sectors. Firms from other sectors do not have biological inputs, so their final output equals E_o, and their price of effort equals the market price of the good,

$$P_{E_o} = P_o \tag{19}$$

Up to this point, seven endogenous variables, L_o, K_o, P_{Zi}, E_o, Z_i, $V_{i,j}$, and P_{E_i} are identified by Equations 13 - 19 . The fishing firm's harvest and effort levels depend on the fishing regulation they face and are described in the next section.

3.3.1.1 Fish Sector Regulation

One goal of this analysis is to explore the economy-wide welfare effects of fishery regulations. To do so, the simulations are structured to assess a movement from a regulated open access (ROA) regime to a fishery with an individual transferable quota system. The choices of the fishing sector firms in nest three of production and rent allocation depend on the institutional setting and will be described in this section.

Regardless of the regulation, in Nest 3, each representative fishing firm targets a single species. For any given level of harvest, each representative fishing firm minimizes costs by choosing effort level E_f subject to its harvest function and taking the stock of species f, S_f, as given. The fishery has a Cobb-Douglas harvest function comprised of effort and stock, where q_f is the firm's catchability or efficiency coefficient, and α_f and β_f are the firm specific effort and stock elasticities of harvest:

$$H_f = q_f E_f^{\alpha_f} {S_f}^{\beta_f} \tag{20}$$

The Lake Erie yellow perch stock is highly regulated through an annual TAC. While the stock is subject to exogenous environmental shocks, the TAC limits the effect annual harvest has on the stock size at the beginning of the year. For this reason, I assume the stock to be the steady state of the fish population. Solving the constraint as a function of H_f and S_f gives:

$$E_f = \left(\frac{H_f}{q_f {S_f}^{\beta_f}}\right)^{1/\alpha_f} \tag{21}$$

3.3.1.1.1 Institutional Setting 1: Regulated Open Access

In the status quo scenario, the fishery is under a regulated open access regime. This is the type of regulation Lake Erie yellow perch fishers from the U.S. currently abide by. A ROA fishery can have different meanings and be modeled in various ways. For example, Homans & Wilen (1997) model a regulated open access fishery with a total allowable catch and a season length. This causes the fishing firm to choose effort where the value of average product is equal to cost, an inefficient outcome. In their model, the regulator anticipates the fisher's choices and sets the season length to achieve the desired quota. Here, I model a ROA fishery as one that allows free entry but that is restricted by a total allowable catch, as is the case for U.S. yellow perch fishers on Lake Erie. Importantly, despite the limit on the quantity of fish that can be harvested, free entry means that, over time, no input earns more than its opportunity cost. If this were not the case, there exists an incentive to enter the fishery (even if entry requires rental of other factor inputs). In a given year, the TAC is set by the Great Lakes Fishery Commission based on the estimated stock size, and thus is a function of the stock. Modeling the regulated open access scenario adjusts the specification of the model so harvest is constrained by the total allowable catch, $H_f = TAC_f(S_f)$. Substituting TAC_f for harvest in Equation 21(and dropping the functional notation for simplicity) the effort choice is given by:

$$E_f^{ROA} = \left(\frac{TAC_f}{q_f^{ROA} {S_f}^{\beta_f}}\right)^{1/\alpha_f} \qquad (22)$$

E_f^{ROA} and q_f^{ROA} are regulation specific.

The free entry assumptions into the yellow perch market implies zero-profits in the fishing sector. In the standard sectors, I defined the zero-profit condition as $P_{E_o}E_o = P_oE_o$, however for the fishing sector with non-market inputs, this zero-profit condition does not provide enough detail, and instead must be described differently. I turn to Euler's theorem to describe the zero-profit condition in an appropriate way for sectors with a natural resource input. Euler's theorem says that a function that is homogeneous of degree one can be expressed as the sum of the products of its inputs and their respective marginal products (Chiang & Wainwright, 2004). For the fishing sectors with constant returns to scale, this implies:

$$H_f = E_f^{ROA}\frac{\partial H_f}{\partial E_f} + S_f\frac{\partial H_f}{\partial S_f} \qquad (23)$$

$\frac{\partial H_f}{\partial E_f}$ and $\frac{\partial H_f}{\partial S_f}$ are the marginal products of effort and stock respectively. The Euler equation can be transformed to dollar value terms by multiplying by the output price of each good, P_f:

$$H_f P_f = E_f^{ROA} \frac{\partial H_f}{\partial E_f} P_f \; + S_f \frac{\partial H_f}{\partial S_f} P_f \qquad (24)$$

When written in this way, Euler's theorem shows that when each input is paid a wage or rental rate equal to the value of its marginal product (that is $\frac{\partial H_f}{\partial E_f} P_f = P_{E_f}$ and $\frac{\partial H_f}{\partial S_f} P_f = P_{S_f}$) the cost of inputs is exactly equal to the total revenue, and economic profit is zero (Chiang & Wainwright, 2004).

A fixed H_f and exogenous stock imply a fixed E_f. The market determines P_f such that market demand and supply are equal, and because stock is not paid for, value is shifted from stock to effort. In Manning et al. (2016), fishers incorrectly attribute the value of marginal product of the stock to other factors of production, and capital and/or labor earn more than the value of its marginal product. The same modelling approach is used here, but because of the nested production function, the extra value goes to effort and is divided between factors in a way consistent with cost minimization. Re-writing Equation 24,

$$H_f P_f = E_f^{ROA} \left[\frac{\partial H_f}{\partial E_f} P_f \; + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f \right] \qquad (24')$$

To maintain zero profits, when no payment is made to the natural resource input, it must be the case that the price of effort in a fishery under a regulated open access is (the right-hand term of Equation 24'):

$$P_{Ef}^{ROA} = \frac{\partial H_f}{\partial E_f} P_f \; + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f \qquad (25)$$

With complete and competitive factor markets, the value of marginal products are equated across inputs through wages. Under regulated open access, the result is different because the fish stock brings additional unpriced value to the fishery and that value is paid to effort. I call Equation 25 the "perceived effort" inverse demand curve, and it reflects that effort receives its value of marginal product plus the value of stock. Instead of the value marginal product of effort being equal to the unit price of effort, $P_{Ef}^{ROA} = P_{Ef}$ (recall P_{Ef} from Equation 18). Equations 22 and 25 identify the endogenous variables E_f^{ROA} and P_{Ef}^{ROA} respectively.

3.3.1.1.2 Institutional Setting 2: Individual Transferrable Quota

In rationalization, the fishery moves from the regulated open access regime to an ITQ system. Canadian yellow perch fishers of Lake Erie follow a similar system. The goal of the ITQ regulation is to induce fishers to internalize the value of the fish stock, and thus assign the proper value to the other inputs. Under the ITQ regulation, the regulator still sets the TAC, but each fisher is endowed with individual quotas that allow them to harvest a portion of the TAC. Each fisher has the right to keep their quotas to fish or sell their quotas at the market price P_I. The cost minimization problem for the representative fishing firm becomes:

$$\min_{E_f,\ H_f} \ P_{Ef}\, E_f + P_{If} H_f$$

$$s.t. \quad H_f = q_f^{ITQ} E_f^{\alpha_f} S_f^{\beta_f}$$

$$TAC_f \geq \ H_f$$

$$H_f > 0$$

$P_I H_f$ represents the additional cost to the firm to harvest, either as a direct cost for purchasing more quota to harvest, or as the opportunity cost of harvesting and not selling their quotas, this is similar to how quotas are included in the firm's decision in Hanley, Shogren, & White (2013).

Again, I focus on the binding case, and assume that all ITQs are used and the TAC is fully exploited, such that $H_f = TAC_f$. With the same binding TAC as in ROA, the firm's choice is limited to their effort level which must be:

$$E_f^{ITQ} = \left(\frac{TAC_f}{q_f^{ITQ} S_f{}^{\beta_f}}\right)^{1/\alpha_f} \qquad (22')$$

To describe the flow of payments under the ITQ scenario, first recall that Euler's equation requires that:

$$H_f P_f = E_f^{ITQ} \frac{\partial H_f}{\partial E_f} P_f + S_f \frac{\partial H_f}{\partial S_f} P_f \qquad (26)$$

The right-hand side of the equation represents the total payments to inputs. To reconcile the total payments to inputs with the cost function minimized by the firm, $P_{Ef}E_f + P_I TAC_f$, the prices of effort and quotas are defined by Equations 25' and 25a'.

$$P_{Ef}^{ITQ} = \frac{\partial H_f}{\partial E_f} P_f \qquad (25')$$

$$P_{If} = \frac{S_f}{TAC_f} \frac{\partial H_f}{\partial S_f} P_f \qquad (25a')$$

Defining prices as in equations 25' and 25a' ensure that $P_{Ef}E_f = E_f^{ITQ} \frac{\partial H_f}{\partial E_f} P_f$ and $P_I TAC_f = S_f \frac{\partial H_f}{\partial S_f} P_f$. In order for the quota market to capture the value of the stock, the ITQ price would need to be at least equal to that defined by 25a'. In doing so, the value of the stock is removed from the effort market and effort is paid only its value of marginal product. In this scenario, the fisher chooses effort so that its value of marginal product is equal to its marginal cost, and the perceived inverse demand for effort is now defined by Equation 25'. The introduction of the ITQ removes the value of marginal product of stock from being paid to effort.

ITQs are implemented as a policy tool to correct the inefficient level of effort employed. One possibility is that, by ensuring more secure fishing rights, effort can be spread across a season rather than employed intensively in derby-style fishing. For the same TAC as under ROA management, an ITQ system is used to induce the fishery to use less effort, which in turn would cause an increase in the total factor productivity (in practice, the long-term changes to productivity are mixed (Fox et al., 2006; Walden et al., 2012)). In this modeling set-up, factor productivity is measured by the Cobb-Douglas catchability coefficient, q_f. If an ITQ does increase efficiency through a decrease in effort, it must be true that $q_f^{ITQ} > q_f^{ROA}$. Since the harvest is the same, and effort is reduced, rents would exist in the fishery.

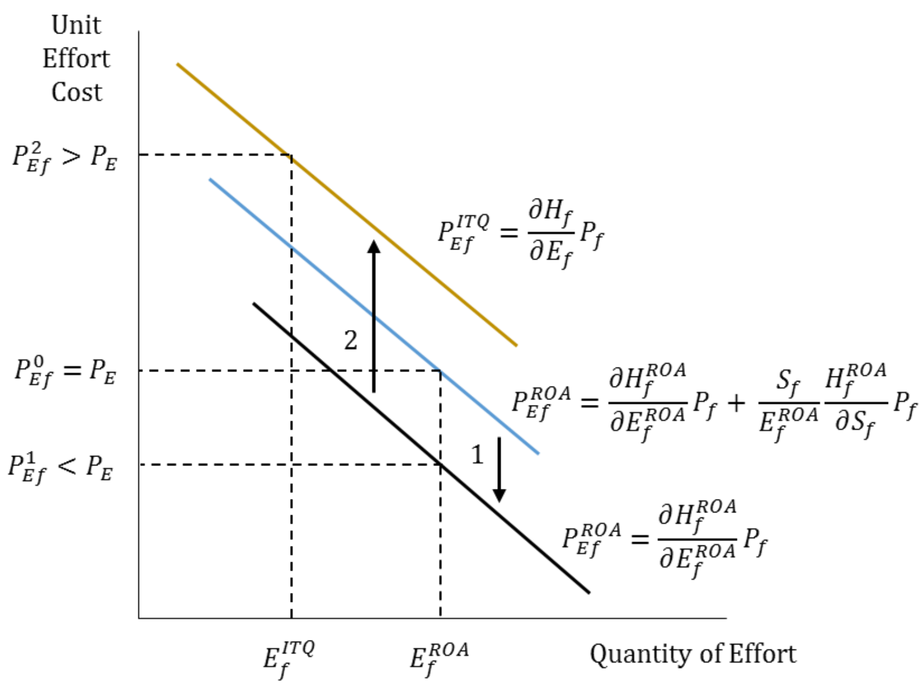

Figure 6: Fishery Effort Market
The shifts in the demand for effort in the fishery sector from going from ROA to ITQ are shown. Shift 1 shows the new defined demand curve under ITQ. The TAC is unchanged and the ROA harvest function and effort level is maintained. For the ROA effort level, the value of marginal product decreased. Shift 2 shows that with clearly defined property rights, the quantity demanded of effort is decreases, so given the fixed TAC, effort becomes more efficient, and increases its value of marginal product. Depending on how much more efficient effort becomes, the value of marginal produce may exceed the market wage. P_{Ef} represents the unit cost of effort in the fishery, and P_E represents the market unit cost of effort (market wage). The unit cost of effort in the final step, P_{Ef}^2 , is equivalent to P_{Ef}^{ITQ}.

Figure 6 shows the potential changes in the fishery effort market following the transition from ROA to an ITQ system when an efficiency gain is realized and when it is not. A change from ROA to ITQ is represented by a leftward shift in the demand curve for effort. Following this shift, at the initial level of E_f^{ROA}, the marginal value of product is below wage rates in other sectors. Too much effort is employed in the sector, representing something akin to an overcapitalization of the fishery. The ITQ serves as a mechanism to reduce this failure in the factor market – fisheries are not willing to pay market wage and factors are not willing to stay in an industry paying below market wage, causing effort to exit until marginal value product is at least equal to the economy-wide wage rate. Harvesting the same TAC with less effort implies an increase in efficiency, which is captured mathematically by an increase in the catchability coefficient.

If a large enough increase in fishing productivity occurs (as shown in Figure 6), harvest costs decrease and a resource rent is created. To define the magnitude of the resource rent generated by the ITQ program, let R_f^{ITQ} be the marginal rent earned by quota owners. Equation 27 defines the marginal rent in the ITQ scenario.

$$R_f^{ITQ} = P_{Ef}^{ITQ} - P_{E_f} \tag{27}$$

Multiplying this marginal rent by the harvest produces the total rent earned in the fishery. Most ITQ systems are organized so that vessel holders own the quota and receive the rent, however who owns the ITQ can vary by program (Arnason, 2007; Libecap, 2007) and is a crucial determinant of the distributional impacts of fishery rationalization. I simulate the allocation of the resource rent and ITQ payments in three ways: to the owners of capital, to the owners of capital and labor, and to the government. Capital ownership is representative of most ITQ systems, and government ownership is representative of a government imposed landings tax (Arnason, 2012; Hanley et al., 2011). The capital and labor ownership scenario is unlike the traditional ITQ set-up, but represents an alternative where rents can also be allocated to laborers in the fishery. In the ITQ scenario P_{Ef}^{ITQ}, P_{If}, and R_f^{ITQ} are identified by Equations 25', 25a', and 27, respectively.

3.3.2 Household Behavior

The model also includes consumer choices, and follows the work of De Melo & Tarr (1992). There are nine household income groups, here the representative household for each income group is modeled. Households maximize utility, a CES function, subject to their income constraint by choosing their consumption of goods:

$$\underset{X_{h_{i,h}}}{Max} \quad U_h = \left(\sum_i \alpha_{Xh_{i,h}} X_{h_{i,h}}{}^{\rho_h} \right)^{1/\rho_h} \quad s.t. \ \sum_i P_i X_{h_{i,h}} \leq Y_{D_h}$$

Households are indexed over set h, $X_{h_{i,h}}$ is the household demand of good i, $\alpha_{Xh_{i,h}}$ is the household's share parameter for good i, and ρ_h is the parameter based on the household's elasticity of substitution, such that $\rho_h = \frac{\sigma_h - 1}{\sigma_h}$. The household pays P_i for each good, and Y_{D_h} is the household's disposable income. The total amount spent on the goods must be less than or equal to the household's income. In this set-up, it is assumed that all income is spent and so the income constraint is binding.

Household income is derived through a two-stage process. Households are endowed with varying amounts of labor and capital, and in the case of privatization, ITQs. These factors are exchanged in factor markets, and through the production process generate value-added. Total value-added expenditures flow first to the factor "institutions", and are then redistributed to households based on factor ownership. Factor payment values are represented by:

$$A_L = \left(\sum_i L_i P_L\right)\left(1 - \sum_t \alpha_{L_t}\right) \qquad (28)$$

$$A_K = \left(\sum_i K_i P_K\right)\left(1 - \sum_t \alpha_{K_t}\right) \quad (29)$$

A_L is the total payment to labor and A_K is the total payment to capital. In the case of ITQs, A_R and A_I are the total rent payment from the effort market and the total payment to ITQs respectively.

$$A_R = \left(\sum_f R_f E_f\right) \quad (30)$$

$$A_I = \left(\sum_f H_f P_{If}\right) \quad (30')$$

I choose to define A_R and A_I separately, but in both are distributed to the owner of the ITQs. By defining the total effort rent payment and the total ITQ payment separately the implications of a change in productivity are more easily isolated. Not all value-added expenditures are paid to households in the region, a portion are payments to the rest of the world. The portion of labor and capital payments that are invested in the region are determined by a_{L_t} and a_{K_t} respectively, where *t* indicates the region. It is assumed that all rent and ITQ payments stay within the region.

Labor and capital payments to households are then found as the net of total value-added payments for labor net of these adjustments. Equations 31 and 31' correspond to household income under ROA and ITQ respectively. Household income includes the portion of the labor, capital, and rent payments the household receives plus an exogenous payment, Y_{E_h} that includes transfer payments form the government, interest income, and earnings from outside the region.

$$Y_h = \theta_{K_h} A_K + \theta_{L_h} A_L + Y_{E_h} \quad (31)$$

Where θ_{K_h} and θ_{L_h} are the share of the capital and labor payments household *h* receives. The rent payment is allocated to households based on the proportion of capital and labor they own.

When moving to an ITQ system that generates rent, the household's income depends on the ownership and allocation of ITQ payments and rents. When ITQ payments and rents are allocated to owners of capital the household's income is defined as:

$$Y_h = \theta_{K_h}(A_K + A_R + A_I) + \theta_{L_h}A_L + Y_{E_h} \qquad (31')$$

When ITQ payments and rents are allocated to owners of capital and labor the household income is:

$$Y_h = \theta_{K_h}(A_K + A_R + A_I) + \theta_{L_h}(A_L + A_R + A_I) + Y_{E_h} \qquad (31'')$$

When the government receives the ITQ payments and rents, the household's income is defined the same as in Equation 31.

The household pays taxes on its income to state and federal governments, at the rate $t_{H_{h,g}}$, where g indicates the state or federal government. The household also saves a proportion of their income in both domestic and international markets at rates $a_{Dh_{h,t}}$ and $a_{h_{h,t}}$ respectively. After saving and paying taxes the household is left with disposable income, Y_{D_h}.

$$Y_{D_h} = Y_h\left(1 - \sum_g t_{H_{h,g}} - a_{Dh_{h,t}} - \sum_t a_{h_{h,t}}\right) \qquad (32)$$

From the utility maximization and the disposable income budget constraint, the households choose their consumption of each type of good, per the following demand equation:

$$X_{h_{i,h}} = \left(\frac{\alpha_{Xh_{h,i}}}{P_i}\right)^{\sigma_h}\left(\frac{Y_{D_h}}{\sum_i \alpha_{Xh_{h,i}}{}^{\sigma_h} P_i{}^{1-\sigma_h}}\right) \qquad (33)$$

In the household portion of the model the endogenous variables Y_{D_h}, Y_h, A_L, A_K, A_R, A_I and $X_{h_{h,i}}$ are identified by Equations 28 - 33.

3.3.3 Trade

Trade occurs in the region with both domestic and foreign partners. Transfer of goods from outside the region but within the United States is considered domestic trade, and the transfer of goods from outside the United States is considered foreign trade. Trade of goods is assumed to occur in perfectly competitive markets. The supply blend of regional and imported goods is allocated through the "Armington assumption" (Armington, 1969). It is assumed that a social planner minimizes the cost of supply over the regional price (P_i) and the exogenous international market price P_{M_i}. The total supply of goods (Q_i) is defined by a CES function comprised of regionally produced goods (Q_{D_i}) and imports from both domestic and foreign sources $(Q_{M_{t,i}})$, where *t* identifies the domestic or foreign trade region.

$$Q_i = \varepsilon_{Q_i}\left(\delta_{QD_i} Q_{D_i}{}^{\rho_{Q_i}} + \sum_t \delta_{QM_{t,i}} Q_{M_{t,i}}{}^{\rho_{Q_i}}\right)^{1/\rho_{Q_i}} \tag{34}$$

In the total supply function, the efficiency parameter is represented by ε_{Q_i}; δ_{QD_i} and $\delta_{QM_{t,i}}$ are the distribution parameters of domestic supply and imports respectively, and ρ_{Q_i} is the parameter based on the elasticity of substitution, σ_{Q_i}, such that $\rho_{Q_i} = \frac{\sigma_{Q_i}-1}{\sigma_{Q_i}}$. The first-order conditions of the cost-minimization define the mix of imports to regional production.

$$Q_{m_{t,i}} = \left(\frac{Q_i}{\varepsilon_{Q_i}}\right)\left[\frac{\delta_{QM_{t,i}}\left(\delta_{QD_i}{}^{\sigma_{Q_i}} P_i{}^{1-\sigma_{Q_i}} + \sum_t \delta_{QM_{t,i}}{}^{\sigma_{Q_i}} P_{M_i}{}^{1-\sigma_{Q_i}}\right)}{P_{M_i}}\right]^{\sigma_{Q_i}} \tag{35}$$

The same process is followed to find the mix of regional demand and exports. The cost of demand is minimized by a social planner over the regional price (P_i) and the international market price P_{M_i}. Total demand is also represented by a CES function, and is comprised of domestic demand and exports.

$$X_i = \varepsilon_{X_i}\left(\delta_{XD_i} {X_{D_i}}^{\rho_{X_i}} + \sum_t \delta_{XE_{t,i}} {X_{E_{t,i}}}^{\rho_{X_i}}\right)^{1/\rho_{X_i}} \quad (36)$$

The demand efficiency parameter is represented by ε_{X_i}; δ_{XD_i} and $\delta_{XE_{t,i}}$ are the distribution parameters of domestic demand and exports respectively, and ρ_{X_i} is the parameter based on the elasticity of substitution, σ_{X_i}, such that $\rho_{X_i} = \frac{\sigma_{X_i}-1}{\sigma_{X_i}}$. The first-order conditions of the cost-minimization define the mix of exports and regional demand.

$$X_{E_{t,i}} = \left(\frac{X_i}{\varepsilon_{X_i}}\right)\left[\frac{\delta_{XE_{t,i}}\left({\delta_{XD_i}}^{\sigma_{X_i}} {P_i}^{1-\sigma_{X_i}} + \sum_t {\delta_{XE_{t,i}}}^{\sigma_{X_i}} {P_{M_i}}^{1-\sigma_{X_i}}\right)}{P_{M_i}}\right]^{\sigma_{X_i}} \quad (37)$$

Regional demand and supply must also be defined. In addition to the supply from firms, H_i , the supply from government, $Q_{G_{i,g}}$, where *g* indicates the state or federal level, and stored inventory, Q_{V_i}, also need to be included in the regional supply measure (Equation 38).

$$Q_{D_i} = H_i + \sum_g Q_{G_{i,g}} + Q_{V_i} \quad (38)$$

Government and inventory supply fixed proportion of total supply, at rates $a_{QG_{i,g}}$ and a_{QV_i}.

$$Q_{G_{i,g}} = a_{QG_{i,g}} Q_i \quad (39)$$

$$Q_{V_i} = a_{QV_i} Q_i \quad (40)$$

Regional demand includes the demand of final goods by households and the government demand, $X_{G_{i,g}}$, intermediate demand from firms in other sectors, and the amount of goods the firms save for inventory, X_{V_i}.

$$X_{D_i} = \sum_h X_{h_{i,h}} + \sum_g X_{G_{i,g}} + \sum_j V_{i,j} + X_{V_i} \quad (41)$$

Government demand and savings inventory are a fixed proportion of total supply at rates $a_{XG_{i,g}}$ and a_{XV_i}.

$$X_{G_{i,g}} = a_{XG_{i,g}} X_i \quad (42)$$

$$X_{V_i} = a_{XV_i} X_i \quad (43)$$

Equations 34 - 43 identify the endogenous variables Q_i, Q_{D_i}, $Q_{M_{t,i}}$, X_i, X_{D_i}, $X_{E_{t,i}}$, $Q_{G_{i,g}}$, Q_{V_i}, $X_{G_{i,g}}$, and X_{V_i} in the trade portion of the model.

3.3.4 Model Closure

The model is closed with equations 44, 45, and 46 and by defining international market price, P_{M_i}, of all goods as the numeriare. Demand for labor and capital cannot exceed the initial endowments in the region, $\bar{L}$ and $\bar{K}$, and total demand and total supply of factors must be equal. These closure equations identify the endogenous factor prices P_L, P_K, and the output price, P_i respectively.

$$\sum_i L_i = \bar{L} \quad (44)$$

$$\sum_i K_i = \bar{K} \quad (45)$$

$$Q_i = X_i \quad (46)$$

3.4 Application: Lake Erie Yellow Perch

To simulate the movement from an ROA to an ITQ management system, data from the Lake Erie yellow perch fishery and the surrounding regional economy is used. Fish biomass and the economic data is from 2013. The yellow perch stock size is taken from estimates provided by the Great Lakes Fishery Commission's Lake Erie – Yellow Perch task group; they estimate the 2013 yellow perch stock in Lake Erie to be 47.621 million pounds (Lake Erie Committee Yellow Perch Task Group, 2013). The economic data was collected from IMPLAN Group LLC, (2013) and is used to build a social accounting matrix. A full description of the total allowable catch, geographical region, social accounting matrix, and harvest function can be found in Chapter 2. I describe model-specific calibration in the section to follow.

3.4.1 Calibration of Parameters

The exogenous parameters in the model can be classified into two types: those that can be calibrated directly from the SAM and those that cannot. Most parameters used in the simulations are calibrated from the benchmark data. Elasticities cannot be calibrated from the benchmark SAM, so they need to be accounted for in other ways. One set of elasticities, the elasticities of substitution for each of the constant-elasticity of substitution functions is taken from the literature. In their 2008 work, Finnoff & Tschirhart present elasticity of substitution values for a variety of sectors including fishing sectors. Most of the elasticities presented by (Finnoff & Tschirhart, 2008) are averages of elasticities presented in the literature, and given the similar fishery focus, CES technology in the standard sectors, and the economic model set-up, I use their values in this analysis (Table 8).

Table 8: Elasticity Values

Elasticity parameter values by type and sector

Parameter	Elasticity Type	Sector & Value
σ_{Q_i}	Total Supply	Fishing Sectors: 3.90 Recreational Fishing: 2.79 All Other Sectors: 2.79
σ_{X_i}	Total Demand	Fishing Sectors: 1.42 Recreational Fishing: 1.42 All Other Sectors: 2.12
σ_i	Production Value -Added	All Sectors: 0.8672
σ_{h_h}	Household Consumption	All Households: 0.8672

Elasticities taken from Finnoff & Tschirhart 2008

Because fish stock levels are not presented in the SAM, the elasticities in the Cobb-Douglas harvest function for yellow perch are a second set of parameters that cannot be calibrated from the benchmark data. Instead, they are estimated using historical fishing data, discussed in Chapter 2. The results of the estimation of the Ohio yellow perch Cobb-Douglas harvest function parameters are shown in Table 9.

Table 9: Cobb-Douglas Harvest Parameter Estimates

COEFFICIENTS	Ohio Estimates
γ_1	0.763***
	(0.124)
γ_2	0.225*
	(0.124)
γ_o	-8.288***
	(1.397)
Wald χ^2	265.42
Observations	35

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

The estimate of effort elasticity of harvest is significant at a p-value of less than 0.001. However, the stock elasticity of harvest is not estimated at a high significance, so I assume constant returns to scale exists and that $\beta_f = 1 - \alpha_f$. Assuming constant returns to scale also makes the fish sector consistent with the other sectors in the model.

The calibration of the catchability coefficient warrants additional discussion. The catchability coefficient changes between the ROA and ITQ simulations. However, the catchability coefficient in the ROA base scenario is calibrated from the benchmark data in a way to ensure that the total allowable catch equals the harvest produced with the stock and the effort levels present.

$$q_f = \frac{TAC_f}{E_f^{\alpha_f} S_f^{\beta_f}}$$

In some simulations when moving to the ITQ scenario, an increase in the catchability coefficients is also imposed. A range of percent increases are simulated.

Once the initial level of effort has been determined in the yellow perch sector, the output price of effort can then be addressed. In the base case, all prices except the output prices of yellow perch are assumed to equal one. To reconcile the economic IMPLAN data with the GLFC harvest data for yellow perch, it is assumed that the initial price of yellow perch, P_f, is such that Euler's equation holds, and there are zero profits, $P_f TAC_f = P_{Ef}^{ROA} E_f$. The initial price of yellow perch therefore equals $P_f = \frac{E_f P_{Ef}^{ROA}}{TAC_f}$. This price applies to both the demand and supply side of the yellow perch sector.

3.5 Policy Analysis

Beyond the baseline scenario of the ROA, the simulations I perform consist of three components; 1) moving from a ROA to an ITQ system, as described by the changes in the modeling structure, 2) simultaneously increasing the catchability coefficient over a range from zero to a hundred percent, and 3) performing components one and two under three different ITQ ownership scenarios: where owners of capital own the ITQs, where both owners of labor and capital own the ITQs, and where the government owns the ITQs. Under each ownership scenario the owners of the ITQs also own the rights to any rents generated in the fishing sector. Recall that the primary difference between the ROA and ITQ scenarios is how the 'extra value' in the fishery is distributed. In the ROA scenario labor and capital receive the value stock brings to production, which is then passed on to households. In the ITQ scenario the value of marginal product of the stock is routed through an ITQ market, and if any rent exists it is allocated to the owners of the ITQs.

The increase in catchability coefficient (q_f) is used to evaluate the possibility that an ITQ system improves the catchability. A change in the catchability means that the amount of effort employed can also change (recall that it remains the same between scenarios otherwise). In practice, a reduction in effort has primarily occurred in two ways, through the dismantling of the 'race to fish' (Conrad, 2010) or through the transfer of quotas from fishers that sold their ITQs and exited the market, as was the case when Canada implemented the ITQ system for its Lake Erie fisheries (Cowan & Paine, 1997). Because each fisher has a right to harvest a proportion of the TAC, there is less of a need to race to harvest. This has been seen in Pacific halibut fishery (Conrad, 2010). On the other hand, others have shown that ITQs do not result in the long term efficiency gains in the amount of effort employed in the fishery, Walden et al., (2012) show that efficiency actually decreased after the ITQ system was implemented in the Mid-Alantic quahoq fishery. To capture all of these possibilities, I simulate a range of possible catchability changes. By doing so, instead of only seeing a shift in the perceived demand curve, the perceived demand curve and the level of the effort can change. The increase in catchability can cause effort to be more productive, and in turn create rents. The welfare implications of the rent generation depend on who receives the rent.

3.5.1 Response of Effort Variables

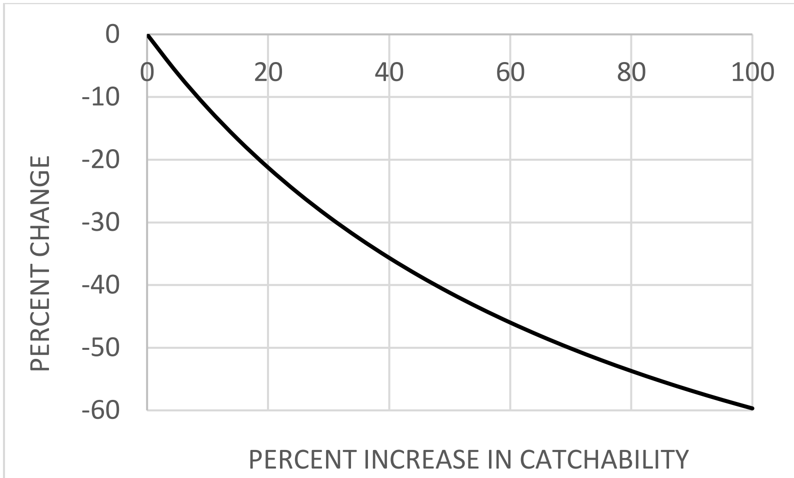

Figure 7: Percent Change in Fishery Effort Inputs as Catchability Increases
Percent decrease in effort (E_f), value-added (Z_f), factor demands (K_f, L_f), and intermediate demands ($V_{i,f}$) in the yellow perch sector as catchability increases.

The first set of results I look at are how effort and inputs to effort in the fishing sector respond to changing to the ITQ system across the different catchability coefficient values. Across all simulated catchability increases, effort, value-added, factor demands, and intermediate demands of air transportation, rail transportation, water transportation, truck transportation, recreational fishing, and the miscellaneous sector of the yellow perch sector decreased. For a given catchability increase, these variables change by the same percent, which is due to the nesting structure of the yellow perch sector. An increase in catchability reduces the amount of effort needed to harvest the TAC, value added and intermediate goods are used in a fixed ratio of effort, so they also decrease by the same percent, and value added passes the decrease into factors. Figure 7 shows the percent change in effort-input variables in the fishery sector as catchability increases. The highest simulated increase in catchability, a hundred percent increase, reduces the firm’s effort choice by 59.69%, and the smallest simulated increase to catchability, a five percent increase, reduces the firm’s effort choice by 6.19%, suggesting that returns to an increase in catchability decrease as catchability improves. There is no change in effort or its components when the catchability coefficient does not change.

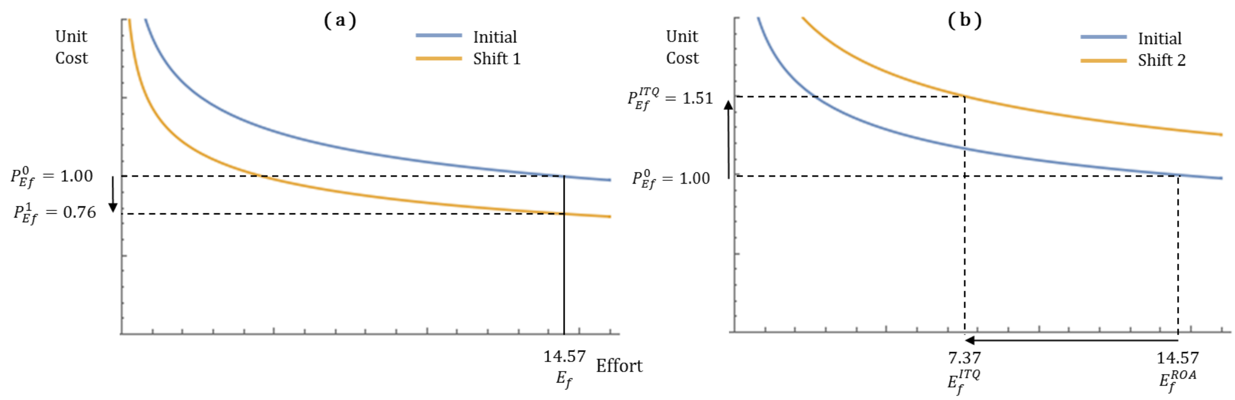

Figure 8: Fishery Effort Perceived Inverse Demand Curves
Response of inverse demand curve for effort in the fish sector when moving from ROA to ITQ system with 68.28% increase in catchability. Panel (a) shows the initial shift in the demand curve after the value of the fish stock is removed from the effort market. Panel (b) shows the increase in value of marginal product of effort and decrease in quantity of effort employed when catchability increases by 68.28%.

In addition to looking at how effort changes as catchability changes, I also look at the demand for effort. Figure 8 is related to Figure 6 above and shows the response of perceived effort demand curves for the yellow perch fishery. The initial shift in the effort demand curve after moving to the ITQ system is shown in Panel (a). Removing the value of the fish stock from the effort market decreases perceived unit cost of effort by approximately 24%. However the reduction in the unit payment to effort would induce effort to leave the market until the value of marginal product of effort was at least equal to the price effort is paid elsewhere in the economy. Reducing the amount effort employed by fishers, but harvesting the same TAC, causes a simultaneous increase in the catchability, which causes an additional shift in the effort demand curve, as shown in Panel (b). Panel (b) is drawn for an effort decrease that occurs with a 68%

increase in catchability. The actual decrease in the amount of effort employed and the corresponding change in the catchability is unknown.

Table 10: Marginal Products and Value of Marginal Products
The marginal product and value of marginal produces in the base ROA.

Parameter		Base Scenario
Marginal Product Effort	$\frac{\partial H_f}{\partial E_f}$	0.78
Marginal Product Stock	$\frac{\partial H_f}{\partial S_f}$	0.07
Value Marginal Product Effort	$\frac{\partial H_f}{\partial E_f} P_f$	0.76
Value Marginal Product Stock	$\frac{\partial H_f}{\partial S_f} P_f$	0.07

Table 10 shows the marginal products and the value of marginal products of effort and stock in the baseline scenario. The marginal products when there is no change in catchability were the same as in the baseline. Figure 9 shows how the value of marginal products change as catchability changes. The increase in catchability induces an increase in the marginal product of effort. A hundred percent increase in the catchability coefficient increases the marginal product of effort to 1.93, a 148% increase from its original value of 0.78. The marginal products of factors employed in other sectors do not experience a change of the same magnitude from the increase in yellow perch catchability (factor prices decrease across the economy by 0.011% at the most). The increased productivity of effort in the yellow perch sector is not captured by the market wages, and so the rent paid to factors by the yellow perch firm increases. Even though the ITQ market captures the value of marginal product of stock, the extra value of effort from its increase in productivity causes the per-unit rent of effort to increase. For the scenario depicted in

Figure 8, the increased in catchability is enough to increase the value of marginal product of effort beyond the market price for effort, which would result in rent generation. Rent changes and distribution are discussed further in the next section.

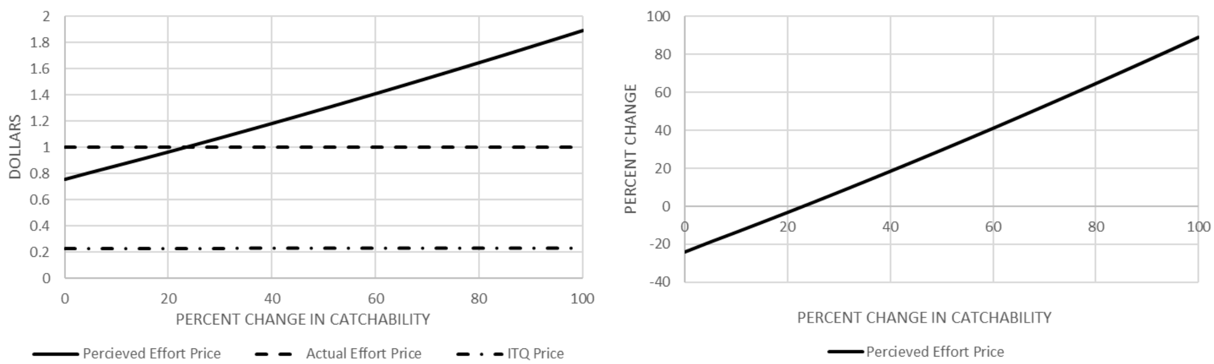

Figure 9: Changes in Perceived Effort Price, Market Effort Price, and ITQ Price
The left panels show the level values of perceived effort price (P_{Ef}^{ITQ}), market effort price (P_{Ef}), and ITQ price (P_{If}) as catchability increases. The right panel shows the percent change in perceived effort price as catchability increases.

3.5.2 Response of Fishery Payment Variables

How the value of effort changes and is distributed is an important aspect of the model and the simulations, so I focus on those changes in this section. Figure 9 shows how the perceived effort price, market effort price, and ITQ price change as catchability increases. The perceived effort price increases as catchability increases, while the market price of effort and ITQ price remain relatively constant as catchability increases. The constant market price supports the hypothesis that the small yellow perch sector is unlikely to induce substantial changes in the rest of the economy. Recall the ITQ price, $P_{If} = \frac{S_f}{TAC_f}\frac{\partial H_f}{\partial S_f}P_f$, the constant value in the ITQ price as catchability increases suggests that any gains an increase in q_f^{ITQ} adds to the marginal product of stock was cancelled out by the impact a decrease in effort had on the marginal product of stock.

A key feature of Figure 9 is the point when the perceived effort price and the market price are equal. The perceived effort price is less than the market price of effort until an increase in catchability of at least 23% is realized. With a catchability increase of at least 23%, the value of marginal product of effort increases enough so that effort employed in the fishing sector generates rents. When the catchability increase is less than 23%, the result is similar to that of Scenario 1, the value of marginal product of effort is less than the going wage in the economy. In this situation, the amount of harvest would decrease, until the value of marginal product was at least equal to the going wage in the economy. In the simulations I perform, I do not allow for the case when the TAC is not binding, so I am not capturing this change.

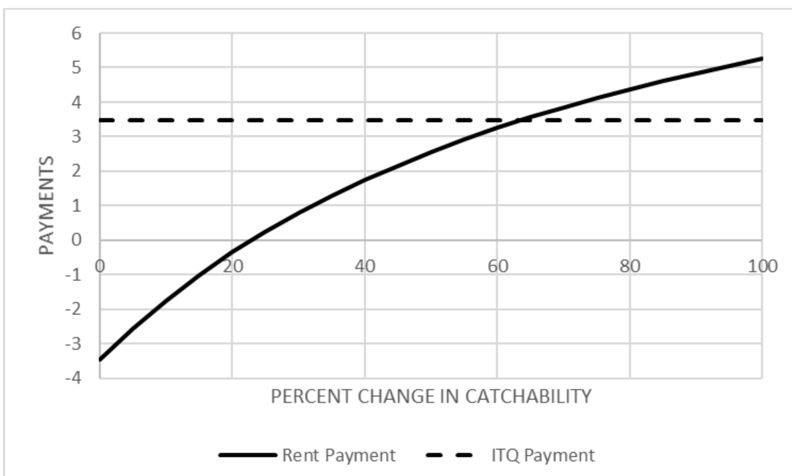

Figure 10: Total Rent and ITQ Payments as Catchability Increases
The total rent payment (A_r) and total ITQ payment (A_I) as catchability increases

Figure 10 shows the total rent and ITQ payments as catchability increases. The total payments for the ITQ remains constant as catchability improves. The constant ITQ payment is due to the constant ITQ price and harvest level. Recall that both of these payments are distributed to the ITQ owner. The total rent payment increases as catchability increases, but does not reach a positive value until catchability increases by at least 23%. As in the case of the perceived effort price and market price, this is a function of imposing a fixed TAC in the modeling structure. To harvest the same TAC, effort must be paid less in the fishing sector than in the rest of economy. The negative rent is how the lower payment is facilitated in the model. Again, these simulations ignore the possibility of a reduction in the harvest and corresponding level of effort used.

3.5.3 Welfare Analysis

Welfare is also compared between the scenarios. Welfare is measured using an indirect money metric utility function as defined by Varian (1992) and calibrated by Rutherford (2008) and equivalent variation. Equivalent variation (EV) is used to find the income required at benchmark prices to attain the utility level possible at the new prices and income. The indirect money metric utility is a way to assign monetary value to utility, so that EV can be calculated. Rutherford (2008) describes a method to calibrate the money metric utility function defined by Varian (1992) in a general equilibrium framework, and I follow his technique. There are three components to defining money metric utility, the first is benchmark share for each good *i* of the household's initial disposable income, represented by $\theta_{i,h}$.

$$\theta_{i,h} = \frac{\overline{P_i}\,\overline{X_{i,h}}}{\overline{Y_{D_h}}} \tag{47}$$

$\overline{P_i}$, $\overline{X_{i,h}}$, and Y_{D_h} are the benchmark price, household demand, and household disposable income, respectively. The second component is the unit expenditure function, P_{U_h}.

$$P_{U_h} = \left[\sum_i \theta_{i,h} \left(\frac{P_i}{\overline{P_i}}\right)^{1-\sigma_h}\right]^{1/(1-\sigma_h)} \tag{48}$$

Rutherford describes P_{U_h} as the unit expenditure function, and it serves as a way to relate the benchmark prices with prices that occur after the simulation. The final component defines indirect money metric utility, V_h, which is found by dividing the counterfactual disposable income by the unit expenditure function.

$$V_h = \frac{Y_{D_h}}{P_{U_h}} \tag{49}$$

Equation 49 defines the indirect money metric utility for household. For welfare analysis, the initial value is subtracted from the counterfactual value to calculate the EV. A positive value indicates a welfare improvement. Because money metric utility is in dollar terms, aggregate welfare measures are found by summing across the households.

Figure 11 shows the change in money metric utility summed over all households for each ITQ ownership scenario in Scenario 2. Welfare increased across all households regardless of the ownership of the ITQs. The largest welfare increase occurred when households were allocated ITQs based on their ownership of both capital and labor with an increase to total money metric utility of 88.86 million dollars. The smallest welfare increase occurs when the ITQ payments and resource rents are allocated to the government. Recall that government transfer payments are assumed fixed, so rents and ITQ payments are not given back to the households. Looking at the percent change in money metric utility (shown in panel b), provides perspective on the magnitude of the changes; the largest percent increase occurred in the capital and labor ownership, and only increased total money metric utility by 0.11%. It is worth noting that even though the percent change in welfare is smaller, the implications of welfare measured with money metric utility, are larger than if we just looked at the rent generation alone. For example, when catchability increases by 100%, $8.36 million is generated in payments between the rent and ITQ payments. In that same simulation, money metric utility increases by $88.89 million dollars. The difference in the payments from the fishery and money metric utility value, suggests that if rent payment were used as a welfare measure, a part of the picture would be missing.

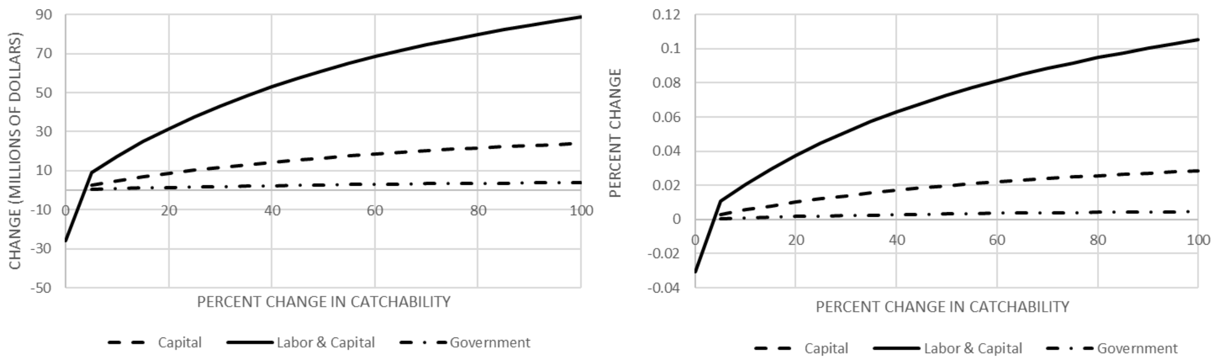

Figure 11: Percent Change in Total Money Metric Utility for Each Ownership Scenario
Change in total money metric utility as catchability increases. The left panel shows change in millions of dollars and the right panel shows percent change.

In addition to comparing welfare impacts across scenarios, welfare impacts can also be compared across households. Figure 12 shows the changes in money metric utility for each ITQ ownership in Scenario 2 across households. When ITQs are allocated to capital owners, Household 9 saw the biggest improvement in welfare, and Household 2 saw the smallest. When ITQs and rent were allocated to household based on their ownership of both capital and labor, Household 8 had the biggest improvement to welfare, and Household 1 had the smallest. When ITQs and rents were given to the government, Household 6 experienced the largest welfare improvement and Household 2 the smallest. When ITQs are owned by the government, the welfare changes were very close across all of the households, particularly Household 1 and Household 9, which is a different outcome than the other scenarios. Part of the variation in impacts is driven by the shares of capital and labor each household owns, and part is driven by the household preferences. Table 11 shows the households' share of factors. Household 8 owns the largest share of labor, and Household 9 owns the largest share of capital. Overall, we see that shares of labor are larger than shares of capital, and less capital is owned by households in the region. We also see that 34.8% of the capital and 11.8% of labor in Lake Erie Coastal Economy is provided by outside sources. The presence of capital and labor provided by outside sources has welfare implications for the Lake Erie Coastal Economy. When moving from the ROA to the ITQ value is being shifted from labor and capital to owners of the ITQ. Depending on the ownership scenario, wealth could be distributed outside of the region, potentially making households worse off.

Table 11: Household Share of Factors
Proportion of total factor payments that each household receives.

Household	Labor	Capital
1	0.003	0.001
2	0.004	0.001
3	0.020	0.006
4	0.034	0.009
5	0.074	0.018
6	0.165	0.038
7	0.158	0.040
8	0.215	0.063
9	0.209	0.116

The money metric utility is capturing two aspects of the households' welfare, their income level and the prices of goods they purchase. The scenario where ITQ and rents are allocated to households based on their ownership of both capital and labor is the only scenario where households experienced an increase in disposable income; all other simulations resulted in a decrease in income from the base scenario. On the other hand, prices in all sectors besides yellow perch decreased in all ITQ scenarios except the Labor & Capital ownership scenario. That the combination of decreased income and decreased prices cause a welfare improvement suggests that price changes, even though relatively small (decreases of less than 0.01% in most cases) are driving the welfare changes.

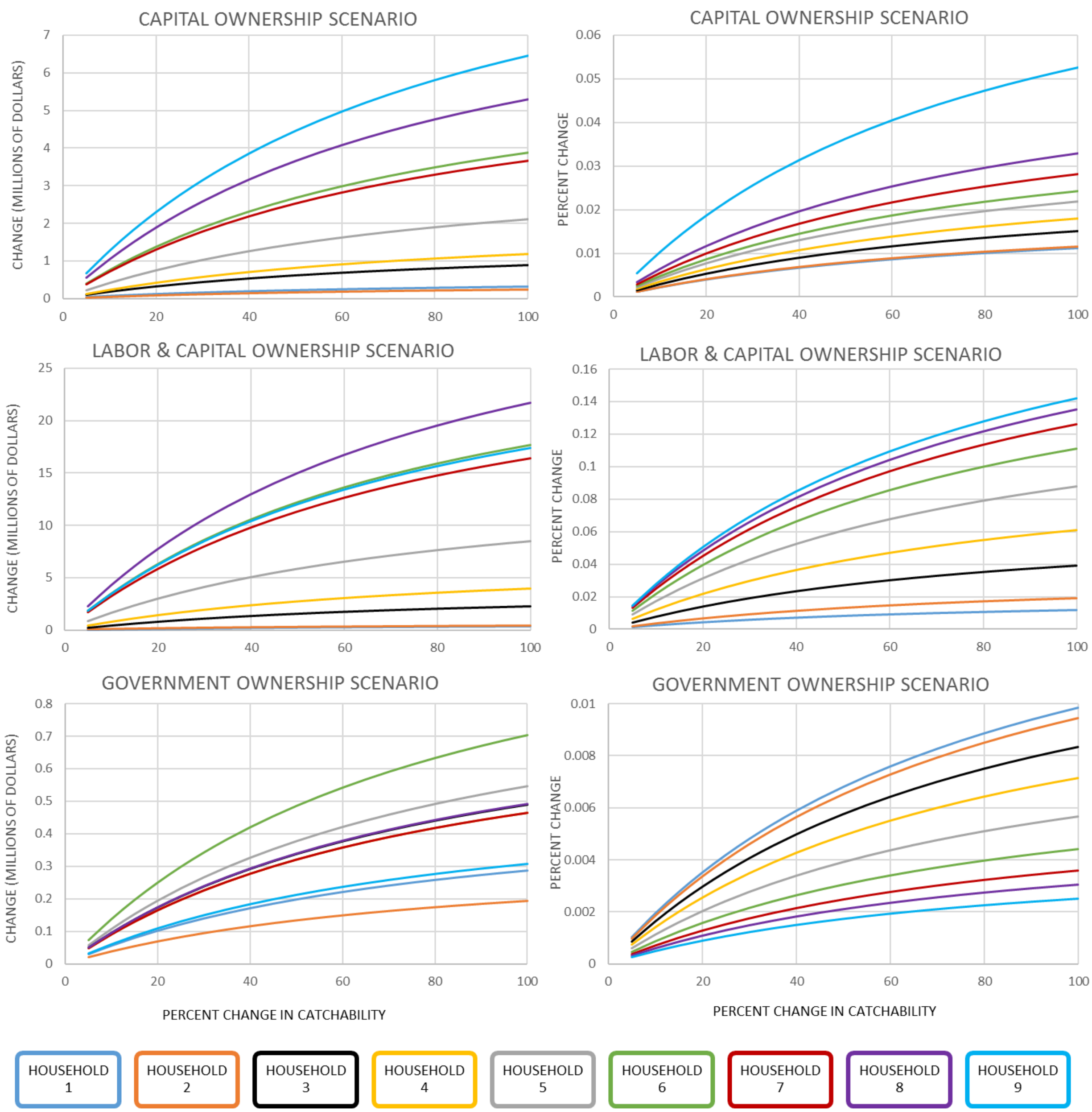

Figure 12: Change in Money Metric Utility by Household and Ownership Scenario
Panels on the left show the percent change in money metric utility by household for each ITQ ownership scenario as catchability increases. Panels on the right show the change in money metric utility in millions of dollars as catchability increases.

Comparing the graphs of the percent in individual household welfare in the government ownership scenario and the labor & capital ownership scenario in Figure 12 also provides additional insight. The order of the household's percent change in welfare is exactly opposite between the two scenarios. This suggests that while price decreases are improving the household's welfare, the ownership of ITQs and thus the right to the rent and ITQ payments impact the magnitude of the welfare change. The higher income households (households 6-9) are the largest owners of capital and labor, and as Figure 12 shows, they have the largest percent increases in welfare when they own the ITQs, as opposed to the government ownership scenario where the ITQ rents and payments are not redistributed to owners of capital and labor.

The variation in welfare improvements by ownership is in line with the observation Weitzman (1974) drew about efficiency and distribution between private ownership and open (free) access. Weitzman (1974) observed that while private ownership of a natural resource is always more efficient than open access, the welfare implications depend on how the rents and tolls from private ownership are distributed, and in the case of no-distribution, private ownership leads to welfare losses. My government ownership scenario is like the no distribution scenario as rents and ITQ payments are collected by the government, but no additional redistribution takes place. If the fishery was not modeled in a general equilibrium frame work, the rents and ITQ payments transferred to the government without redistribution would likely lead to welfare losses, however I see welfare improvements, even in that scenario. This is because, while the redistribution of income disadvantages the households, the price decreases across other sectors from the redistribution of effort, outweigh the households' decreases in income. The distribution of rents matter, but so does the consideration of impacts to price across sectors. This result further supports the need to model fisheries in general equilibrium frameworks.

The results are presented over a range of catchability increases, which begs the questions, "how would the catchability change in the Lake Erie yellow perch fishery under and ITQ system?" In

Chapter 2 I estimated the catchability coefficient for the Ontario yellow perch firm. While there are some limitations to that estimation, primarily the different type of net used in their harvesting, it does provide a possible measure of the expected catchability coefficient. Recall the estimates of the constants in natural log equation of the Cobb-Douglas harvest function for the ROA and ITQ scenario as $\gamma_0^{ROA} = -8.288$ and $\gamma_0^{ITQ} = -2.629$. These values are the natural log of the catchability coefficient, and if used to find the percent change in catchability would suggest an increase of about 6 percent. An increase in catchability under an ITQ system of about 6 percent would fall in the range of catchability coefficients where the marginal product of stock would not increase enough to outweigh the loss in payment to effort that gets allocated to the ITQ market. If an increase in catchability of about 6 percent occurred the firm would harvest less. The welfare impacts of reducing the harvest level needs to be further explored. Fox et al., (2006) found catchability improvements of 26% to 35% depending on boat size in the south-east trawl fishery of Australia after moving to an ITQ system. If these increases were experienced in the Lake Erie yellow perch fishery, very small welfare improvements would occur.

3.6 Conclusion

Herein I analyzed the movement from a regulated open access management system to an individual transferrable quota system in the general equilibrium setting. I examine the outcomes over a range of possible catchability improvements, and for different ITQ ownership scenarios.

When the ITQ program does induce an increase in catchability across the fishery that is coupled with a decrease in effort employed, rents are generated, and there is a welfare improvement. The magnitude of the welfare improvement depends on who owns the ITQs and how the rents are allocated. In the Lake Erie Coastal Economy, when rents are allocated to both capital and labor owners (opposed to just labor owners or the government), the greatest improvement to welfare is seen. It is important to note that the 34.8% of the capital and 11.8% of labor in Lake Erie Coastal Economy is provided by outside sources, which impacts the welfare measurements that are based on households in the region.

When the ITQ system did not induce a reduction in effort and an increase in catchability high enough to increase the value of marginal product of effort in the fishing sector to at least the market price of effort, there was a welfare loss. In that situation, the households lose income as the production value of the stock is transferred from factors to ITQ owners, and are worse off because of it. This suggests that ITQs only produce welfare improvements if they actually cause efficiency gains and generate rents.

I also saw that a decrease in prices of most goods in the region was driving the welfare improvements. In the cases when prices decreased, households increased their consumption of their preferred goods, and make themselves better off overall. This result suggests that the management of one sector can have economy-wide impacts, even when that sector is small relative to the rest of the economy, as is the case with yellow perch and the Lake Erie Coastal economy. The magnitude of the welfare improvements was different than size of the rent and ITQ payments that were distributed to the ITQ owners. This outcome shows that while the partial equilibrium analysis of fishery issues may be useful for some analysis, it misses welfare impacts that can reverberate through the economy a change in the fishing sector, even if that sector is relatively small. For more complete impact analysis, the relationships between sectors and agents, as are represented in a GE framework, need to be considered. Management in the fishing sector examined here, has impacts beyond the fishing sector.

Beyond addressing the welfare implications of moving from a regulated open access regime to an individual transferrable quota system, I have displayed a method for incorporating fishing sectors into a general equilibrium model with different management settings and capturing the potential rents that exist. The magnitude of my findings are true for the Lake Erie yellow perch fishery, and will likely differ by setting and make-up of the economy. The Lake Erie yellow perch sector is small in size relative to the regional economy it is a part of. Changes in the management of fishing sectors, or other natural resource sectors, in other regional economies, may produce different welfare results. However, even though the welfare implications can vary depending on the situation and focus, the main point remains the same, to truly understand the impacts of management regimes they need to be evaluated in a general equilibrium setting where the relationships between sectors, factors, firms, households, and market prices are more fully modeled and better accounted for.

CHAPTER 4: A BIOECONOMIC MODEL OF A REGIONAL ECONOMY AND AN AQUATIC FOOD WEB

4.1 Introduction

Economic and ecological systems are interconnected and interdependent. Humans use the resources from the ecosystems they are part of for both commercial and recreational activities, and when those ecosystems change, so must human behavior. Changes in human behavior, in turn, cause further changes in the ecosystem. In both research and management, questions often arise about how a new policy, a change in species abundance, or some other shock will impact the economy and ecosystem. To fully analyze the impacts to these systems from a disturbance, a model that integrates the two systems is needed. This paper presents a model that integrates the regional economy surrounding Lake Erie with the Lake Erie food web. The model can be used to analyze a variety of policy changes and system shocks, but here I focus on the ramifications of an Asian carp invasion.

Two species of Asian carp, bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*H. molitrix*), and their potential threat to the Great Lakes' ecosystem have been of primary concern to stakeholders and policy makers (Great Lakes Commission & St. Lawrence Cities Initiative, 2012). The carp were intentionally introduced to North America in the 1970's to improve pond water quality and support aquaculture in the southern United States, and have since invaded the entire Mississippi River basin (Kolar et al., 2007). Should they successfully enter the Great Lakes, many worry their presence will cause irreversible impacts to the Great Lakes ecology, commercial fishing, and recreational boating (Rasmussen, Regier, Sparks, & Taylor, 2011).

Others predict that they are already present in Lake Michigan (Jerde et al., 2011). Food web models predict an Asian carp invasion would decrease walleye populations, with mixed results for yellow perch populations (Zhang et al., 2016). Bioenergetics modeling, however, has indicated that bighead and silver carp likely cannot colonize or grow in the Great Lakes with a few exceptions in warm productive habitats, such as western Lake Erie (Cooke and Hill 2010), and some Great Lakes fisheries experts believe that if bighead and silver carp were to establish in Lake Erie, there would be little negative impact to sport fish such as walleye and yellow perch (Wittmann et al., 2015). The uncertainty and disagreement about the ecological consequences of Asian carp in the Great Lakes makes the impacts difficult to evaluate. To address the uncertainty of how Asian carp will behave in the Great Lakes, Zhang et al. (2016) incorporate the expert opinions presented in Wittmann et al. (2015) to develop an Asian carp scenario in their Ecopath with Ecosim (EwE) food web model of Lake Erie. Their approach combines a mathematical modeling system with the opinions of experts in the field, and in doing so creates a more robust estimate of what a Lake Erie food web with Asian carp would look like.

While previous work in the economic and ecological literature has looked at the economic impacts of a change in the ecosystem, there has not been as much focus on how human behavior will further influence the ecosystem and cause future impacts. In many cases, outputs from an ecological model are used as inputs to the economic model, and that is where connections between the ecosystem and the economy end (Avila-Foucat et al., 2009; Byron, Jin, & Dalton, 2015; Eichner & Tschirhart, 2007; Jin et al., 2012; T. Warziniack et al., 2017). While this sort of analysis does a good job of evaluating the initial impacts of a change in the ecosystem, it does not capture the impacts on the ecosystem from changes in human behavior, and is therefore missing an important feedback and potentially incorrectly measuring the impacts from a shock to either the economic or ecological system.

This paper presents a technique for integrating the economic and ecological systems into a single model. I link a computable general equilibrium model of a Lake Erie regional economy with a food web model developed by Zhang et al., (2016) to include an Asian carp presence. The linked economic and ecological model can be useful for a variety of applications, but for the purposes of demonstration, this paper will focus on analyzing the impacts of an aquatic invasive species on Lake Erie and the surrounding region. There are two primary objectives of this paper: to present the economic and ecological impacts of a Lake Erie Asian carp invasion, and to show how the magnitude of these impacts differ when only evaluated with either the economic or ecological system independent of the other. I find that the introduction of Asian carp to the Lake Erie food web has mixed ecological impacts, some species' populations increase, while others decrease. The increase in species populations leads to overall welfare gains. While understanding the impacts of an invasive species is important, another, perhaps more interesting, result is how the ecological consequences differ when evaluated with and without the economic feedbacks. When not accounting for human choices, the food web model overestimates the populations of some species and underestimates the populations of others. I attribute the difference in impacts to species populations to the lack of human choice and substitutability in ecologically based models.

This paper proceeds as follows: an review of the literature of general equilibrium bioeconomic is presented in the next section , followed by a description of the economic components of the model, the EwE methodology, the Lake Erie case study, and finally the results and discussion.

4.2 Literature Review

Traditionally in economics, fisheries are modeled with a logistic growth function. While the simplicity of a single-species Schaefer model of a fishery can be useful in analysis, it leaves something to be desired when it comes to modeling the complexity of fisheries and relationships between species in the real world. Linkages are present between species and people, and these linkages and feedbacks need to be considered when estimating the impacts from a change in either the economic or ecological system. Examining the economic impacts of changes in the ecosystem is not a new concept, and it has been established that the feedbacks between ecosystems and human behavior are important in measuring those impacts (Bossenbroek, Finnoff, Shogren, & Warziniack, 2009; Finnoff, Settle, Shogren, & Tschirhart, 2009; Finnoff & Tschirhart, 2003; Jin et al., 2012). Armstrong (2007) argues that for best fisheries-management practices, economist need to not only incorporate multiple species into their fishery models, they also need to collaborate with ecologists to develop models that are complex enough in both the economic and ecological sense to better capture the reality of the fishery. The methods and the extent in which those feedbacks are modeled vary; bioeconomic and integrated models can refer to a variety of methods from the micro-focused predator-prey models such as Eichner & Pethig, (2006) to large scale general equilibrium bioeconomic models such as Finnoff & Tschirhart, (2008). The work presented here is on the general equilibrium end of the spectrum, and builds off of similar work, discussed in this following section.

Much of the large-scale work on feedbacks between economic and ecological systems uses outputs from an ecological model as inputs in an economic model, and that is where connections between the ecosystem and economy end. Byron et al. (2015) uses the simulated stock biomass from an Ecopath model as an input to their regional input-output model, a linear representation of the transactions in an economy. While this enables them to account for ecological changes in their economic model, the use of the input-output model does not allow them to account for any substitutability in human behavior. Jin et al. (2012) takes the simulations a step further by modelling the relationships between an economy and the ecosystem by using outputs from a marine food web model in a CGE model. Avila-Foucat et al. (2009) also use output from a food web model, but they use the Ecopath system, and model only the production side of the economy with production functions of various sectors including fishing, ecotourism, and agricultural that would presumably all be impacted by changes in the food web. In the same vein, Warziniack et al. (2017) focus on a household model that includes both marketed and non-marketed goods. While there is value in this type of analysis, such as evaluating the initial impacts of a change in the ecosystem, it does not capture the impacts on the ecosystem from changes in human behavior. Humans are adapters that make their choices given a set of conditions and constraints. When the ecosystem changes, humans adjust and their choices change. Their new choices may have additional impacts on the ecosystem which could in turn have further repercussions on the economy and the resources available to people.

Finnoff & Tschirhart (2003, 2008) take the feedback between the economic and ecological systems a step further in their development of the general equilibrium ecosystem model (GEEM), a model that takes advantage of the similarities between economies and ecosystems, and models individual firms, households, and organisms together. While Finnoff & Tschirhart (2003, 2008) use the GEEM to evaluate how the Alaskan economy and ecosystem respond to policy changes, the model is complex to develop and parameterize for different regions and scenarios and is not an established model in the ecology field.

The model I present in this paper captures the additional feedbacks between humans and the ecosystem by modeling how changes in human behavior further produce changes in the ecosystem. The economy is represented with a computable general equilibrium model, and the established ecological model Ecopath with Ecosim (EwE) is used to represent the food web. CGE models have been utilized for trade and development problems for decades, and their usefulness for environmental problems in adaptive systems has been shown more recently (Finnoff & Tschirhart, 2008; Jin et al., 2012; Pan et al., 2007; Warziniack et al., 2013). CGE models are powerful because they include maximizing agents that respond to changes in the economy (Shoven & Whalley, 1992). Prices in the economy change and agents adjust their behavior accordingly. It has also been shown that the balancing relationships present in a CGE model determine the values of all variables that, along with the utilization of consumer and producer data, create a tool useful for policy analysis (Tschirhart, 2004). The bioeconomic model presented here is also novel in that the economic model incorporates both recreational and commercial fishing activities, harvest quotas, and fish biomass values with an ecosystem model in a truly integrated way.

4.3 Model

To jointly estimate the ecological and economic impacts from an exogenous shock to either the ecosystem or the economy, a computable general equilibrium (CGE) model is combined with the ecological food web model Ecopath with Ecosim (EwE). This approach is unique in that it allows for feedback between the two models. Figure 13 shows the model overview and connections between the economy and food web. The economy includes households and producers and the physical and monetary flows between them. The food web includes the piscivorous fish (fish who primarily prey on other fish) and omnivorous fish and the groups of species they target. Households and producers make their recreational and commercial harvest choices based on the biomass values of their target species, which are primarily piscivorous fish.

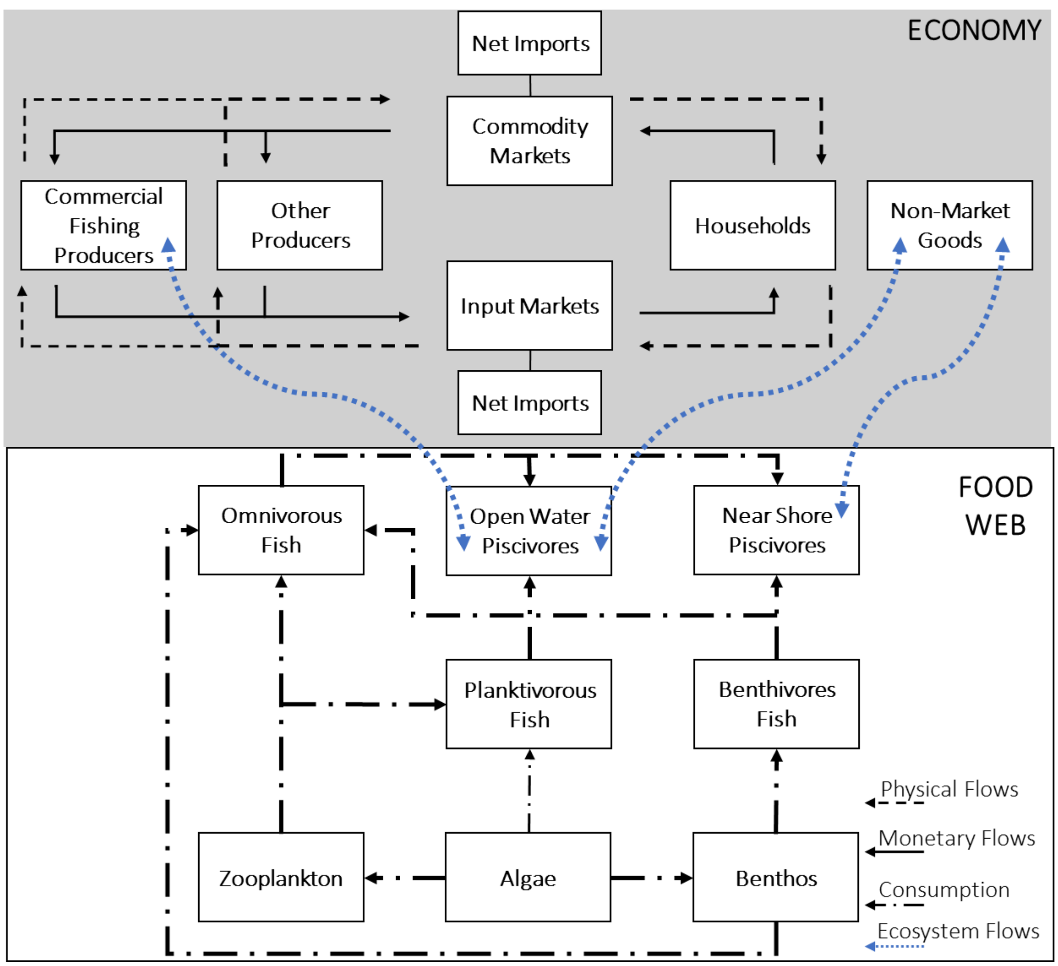

Figure 13: Bioeconomic Model Overview
The figure shows the flows between the CGE and the food web. The biomass of piscivorous fish (fish that primarily consume fish prey) are considered in both commercial fish production and household consumption of non-market goods. In turn, human consumption of fish impacts the food web. Similar figure presented in (Lodge et al., 2016)

The EwE models a Lake Erie food web and is fully described in Zhang et al., (2016). The base CGE used in this analysis is the same as the model presented in the previous chapter, with two exceptions: open access commercial fishing sectors are included, and households consume non-market recreational experiences in addition to marketed goods. Other aspects of the model, production of non-fishing goods, trade, and the government's role, are the same framework as presented in Chapter 3. In this chapter, I only show the aspects of the models that are unique to this specification. The methods section proceeds as follows: the variation in commercial fishing is explained, the description of the household non-market choices are presented, an overview of the ecological model is given, then the link between the economic and ecological sides of the model is described.

4.3.1 Fishing Sector Regulations

Fishing sectors in this model can be further categorized into two types, those that are limited by a total allowable catch and those that are not. Lake Erie yellow perch and walleye harvest (both commercial and recreational) are restricted by a total allowable catch, while the other fish species, bigmouth buffalo, common carp, channel catfish, freshwater drum, lake whitefish, white bass, and white perch do not have restricted harvests. The different management practices influence the behavior of the firms, so they are each explained separately.

4.3.1.1 Regulated Open Access Fishing Sectors

I begin by describing the behavior of fishing firms in sectors regulated by harvest restrictions, referred to as the regulated open access (ROA) fishing sectors. These sectors include yellow perch and walleye and are modeled in the same way as the ROA in the previous chapter. The representative firm from sector $r \in F$, is constrained by a TAC. Recall the regulated open access result from Chapter 3, harvest is constrained by the total allowable catch, so $H_r = TAC_r(S_r)$, and firms do not pay for stock, so $P_{S_r} = 0$. The effort choice for regulated fishing sectors is given by:

$$E_r = \left(\frac{TAC_r}{q_r {S_r}^{\beta_r}}\right)^{1/\alpha_r} \tag{50}$$

4.3.1.2 Non-Regulated Fishing Sectors

Other fishing sectors do not face a harvest restriction and are referred to as the non-regulated fishing sectors. The non-regulated representative firm $n \in F$, also minimizes costs in harvesting by choosing its aggregate effort level E_n, but is not constrained by a TAC. The specification of the effort choice for non-regulated firms is the same as in the general specification in Chapter 3 above, but again, firms do not pay for stock, and the effort choice is described by:

$$E_n = \left(\frac{H_n}{q_n {S_n}^{\beta_n}}\right)^{1/\alpha_n} \tag{51}$$

4.3.1.3 Fishery Zero-Profit Condition

Free entry into the fishery sectors implies zero-profits for all fishing firms regardless of if they are regulated by a TAC or not. Euler's theorem says that a homogenous of degree $\alpha_f + \beta_f$ function can be expressed as the sum of the products of its inputs and their respective marginal products (Chiang & Wainwright, 2004). For the fishing sectors, this implies:

$$(\alpha_f + \beta_f)\, H_f = E_f \frac{\partial H_f}{\partial E_f} + S_f \frac{\partial H_f}{\partial S_f} \tag{52}$$

$\frac{\partial H_f}{\partial E_f}$ and $\frac{\partial H_f}{\partial S_f}$ are the marginal products of effort and stock respectively. The Euler equation can be transformed to dollar value terms by multiplying by the output price of each good, P_f:

$$(\alpha_f + \beta_f)\, H_f P_f = E_f \frac{\partial H_f}{\partial E_f} P_f + S_f \frac{\partial H_f}{\partial S_f} P_f \tag{53}$$

When written in this way, Euler's theorem shows that when each input is paid the value of its marginal product (that is $\frac{\partial H_f}{\partial E_f} P_f = P_{E_f}$ and $\frac{\partial H_f}{\partial S_f} P_f = P_{S_f}$) the cost of inputs is exactly equal to the total revenue adjusted by the degree of homogeneity, thus economic profit is zero (Chiang & Wainwright, 2004).

The market determines P_f and because stock is not paid for, effort is paid more than its marginal product. Manning et al., (2016) show that the fishers incorrectly attribute the value of marginal product of the stock to factors of production, and pays capital more than the value of its marginal product. The same idea is used here, but because of the nested production function, the extra value goes to effort first and then to factors. Re-writing 53,

$$(\alpha_f + \beta_f)\, H_f P_f = E_f \left[\frac{\partial H_f}{\partial E_f} P_f + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f\right]$$

To maintain zero profits, it must be the case that the price of effort in a fishery is:

$$P_{Ef} = \frac{\partial H_f}{\partial E_f} P_f + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f \tag{54}$$

Each firm has a different aggregate of inputs, so they will each have a unique unit cost of effort. With complete and competitive factor markets, the value of marginal products are equated across inputs through wages. In the fishery sectors the result is different because the fish stock value is attributed to effort. The fish stock creates value in the fishery that firms attribute to effort. Equation 54 reflects that effort receives its value of marginal product plus the value of stock.

4.3.2 Household Behavior

Recreational fishing can be a large part of the relationship between humans and the ecosystem, as is the case in the Lake Erie region where U.S. recreational fishing typically exceeds commercial fishing in fish landings (Lake Erie Committee Yellow Perch Task Group, 2013). It is important to account for the recreational component of fishing in policy analysis, and for this reason the household and its behavior is emphasized in this model. To be comprehensive about the choices that the household faces, the household consumes both marketed and nonmarketed goods. Goods acquired through markets are referred to as consumption goods, non-market goods are attained through recreation trips. The model of household behavior for non-marketed goods follows that of Warziniack et al. (2017), a model that describes the household's consumption of both market and non-market goods in a general equilibrium setting. Here, I apply the Warziniack et al. (2017) model specifically to recreational fishing.

An individual household, h, maximizes utility by taking recreational trips and by consuming a composite of other goods. The household makes decisions in three steps: 1) The individual decides how to divide its income between recreation R_h and a composite of all other consumptive goods C_h, 2) For the portion of its income allocated to recreation, the individual decides which species to target on trip $T_{h_{f,h}}$, where f continues to represent the fish species targeted, and 3) The individual minimizes the cost of the recreational experience on trips, where the recreational experience is produced by a combination of individual household effort, $E_{h_{f,h}}$, and the level of fish stock, S_f. Figure 14 shows the nesting structure for household consumption. This model also considers the ecosystem services depicted in Figure 3 that the fish stock provides to the household and their consumption of non-market goods.

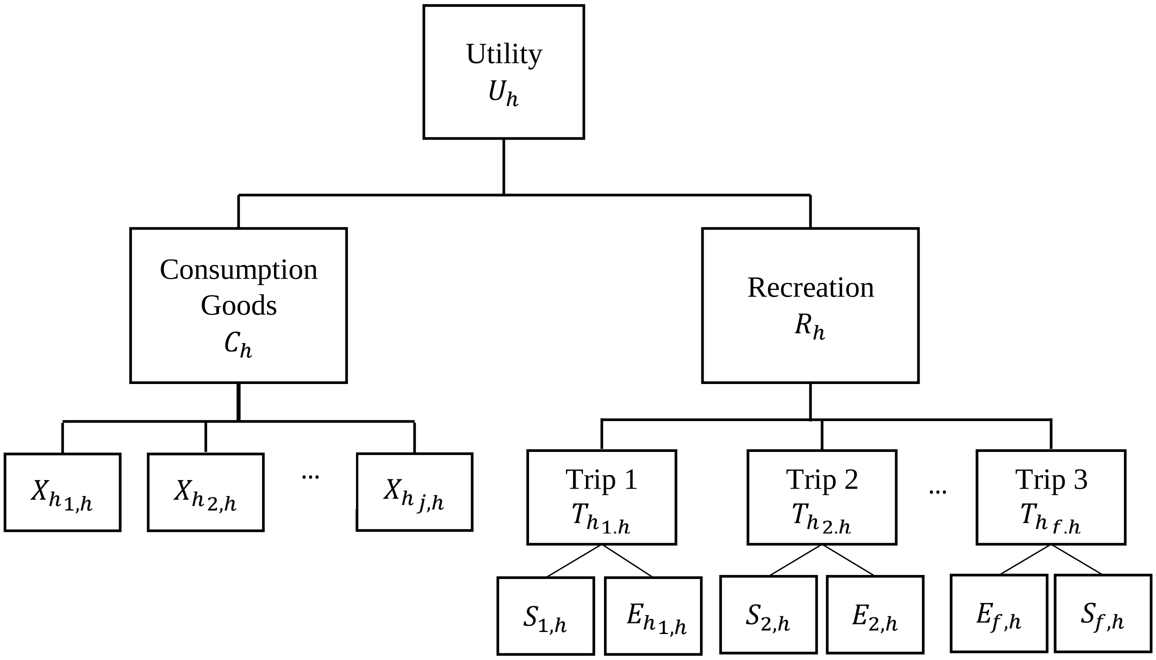

Figure 14: Nesting Structure for Household Behavior
Substitution possibilities at each level in the nest described by constant elasticity of substitution (CES) functions.

Step One: Income allocation between recreation and other consumption goods

Let P_R be the price of recreational trips, P_C be the price of other goods, and Y_{D_h} be the household's level of disposable income. The budget constrained utility maximization problem is

$$\max_{R_h, C_h} U_h(R_h, C_h) \quad s.t \quad Y_{D_h} - P_R R_h - P_C C_h = 0$$

Assume the utility function *U*(.) is well behaved with $U_R, U_C > 0$, $U_{RR}, U_{CC} < 0$, and $U_{RC} \geq 0$.

Defining λ_{YD_h} as the marginal utility of disposable-income, the first-order conditions are:

$$\frac{\partial U_h}{\partial R_h} = \lambda_{YD_h} P_R \qquad (55)$$

$$\frac{\partial U_h}{\partial C_h} = \lambda_{YD_h} P_C \qquad (56)$$

$$Y_{D_h} = P_R R_h - P_C C_h \qquad (57)$$

These conditions require the household to demand a mix of consumption goods and recreation so that the marginal rate of substitution equals the price ratio of the two types of goods, while maintaining the budget constraint. Equations 55-57 identify R_h, C_h, and λ_{YD_h}.

Step Two: Allocate trip income across species-specific trips

After spending $P_C C_h$ on consumption goods, the consumer has $Y_{D_h} - P_C C_h$ to spend on recreation. Recreation expenditures are divided among trips, $T_{h_{f,h}}$, targeting specific species. Let the price of trips targeting fish species *f* be P_{T_f} and the well behaved sub-utility function $R_h(T_{h_{1,h}}, T_{h_{2,h}}, \ldots, T_{h_{f,h}})$. The second stage optimization problem becomes:

$$\max_{T_{h_{f,h}}} R_h\left(T_{h_{1,h}}, T_{h_{2,h}}, \ldots, T_{h_{f,h}}\right) \quad s.t. \quad Y_{D_h} - P_C C_h - \sum_f P_{T_f} T_{h_{f,h}} = 0$$

If the marginal utility of trip income is λ_{R_h} the necessary conditions for an interior solution are

$$\frac{\partial R_h}{\partial f} = \lambda_{R_h} P_{T_f} \quad for\ all\ f \epsilon [1, \ldots, n] \tag{58}$$

$$Y_{D_h} - P_C C_h = \sum_f P_{T_f} T_{h_{f,h}} \tag{59}$$

The system defined by equations 58 and 59 requires the individual to take trips targeting each activity until the marginal rates of substitution for all pairs of activities are equal to the price ratio of trips, and trip income available is exhausted. The simultaneous solution of equations 58 and 59 implicitly defines demand functions for targeted trips $T_{h_{f,h}} = T_{h_{f,h}}(P_{T_1}, \ldots, P_{T_f}, P_C, Y_{D_h})$. Given the demand functions, the indirect sub-utility function can be found (Varian, 1992) with its associated unit expenditure function $e_{R_h} = e_{R_h}(P_{T_1}, \ldots, P_{T_f}, P_C, Y_{D_h})$. The unit expenditure function provides the link to Step (1) as it provides the price index for trips, $e_{R_h} = P_R$.

Step 3: Minimize the cost of the recreational experience by combining effort and stock

Species – targeted trips are a good, which could be measured in terms of actual outcome or just enjoyment of the activity. The recreational experience on a targeted trip is a good produced by individuals combining human effort $E_{h_{f,h}}$and species stock size, S_f. While individuals choose the amount of effort they exert, they take the level of stock as given. Though the stock is beyond the control of any one individual, they are an input to their production of the recreational experience, and so have nonmarket value.

It is assumed that the individual behaves as if they minimize the cost of recreating where costs flow from the market costs of effort, P_{Eh_h} (generally taken as a fraction of the market wage), and the virtual costs of ecosystem services P_{S_f}.

$$\min_{E_{h_{f,h}}} \; P_{Eh_h} E_{h_{f,h}} + P_{S_f} S_f \quad s.t.\; T_{h_{f,h}} = T_{h_{f,h}}(E_{h_{f,h}}, S_f)$$

Letting the Lagrange multipliers associated with the recreational experience production function be λ_{T_h} the necessary conditions for an interior solution are (for all $s \in [1,..,n]$)

$$P_{Eh_h} = \lambda_{T_h} \frac{\partial T_{h_{f,h}}}{\partial E_{h_{f,h}}} \quad (60)$$

$$P_{S_f} = \lambda_{T_h} \frac{\partial T_{h_{f,h}}}{\partial S_f} \quad (61)$$

$$T_{h_{f,h}} = T_{h_{f,h}}(E_{h_{f,h}}, S_f) \quad (62)$$

Simultaneous solution of equations 60-62 provides input demand functions for effort $E_{h_{f,h}} = E_{h_{f,h}}(P_{Eh_h}, S_f, T_{h_{f,h}})$ and inverse demand functions for the nonmarket values of ecosystem services $P_{S_f} = P_{S_f}(P_{Eh_h}, S_f, T_{h_{f,h}})$ (as in Carbone & Smith, 2013). From these functions, a cost function for the targeted recreational activity can be constructed

$$C_{T_{f,h}}(P_{Eh_h}, S_f, T_{h_{f,h}}) = P_{Eh_h} E_{h_{f,h}} \left(P_{Eh_h}, S_f, T_{h_{f,h}}\right) + P_{S_f}(P_{Eh_h}, S_f, T_{h_{f,h}}) S_f \qquad (63)$$

We assume $T_{h_{f,h}}(E_{h_{f,h}}, S_f)$ is homogenous of degree one, so the cost function is homogenous of degree one in $T_{h_{f,h}}$. Defining $c_{T_{f,h}}(P_{Eh_h}, E_{h_{f,h}})$ as the unit cost of the targeted activity (and equivalent to the price of the targeted trip P_{T_f}) allows the statement and link between Step (2) and (3) to be

$$C_{T_{f,h}}\left(P_{Eh_h}, S_f, T_{h_{f,h}}\right) = c_{T_{f,h}}\left(P_{Eh_h}, E_{h_{f,h}}\right) T_{h_{f,h}} = P_{T_f} T_{h_{f,h}} \qquad (64)$$

The conditions above define the optimal trips targeting each activity given the unit cost of targeting each activity.

The allocation of expenditures between consumptive goods follows standard CGE (De Melo & Tarr, 1992) procedures. Households choose consumption levels $X_{h_{i,h}}$ to minimize the cost of achieving sub-utility level $\bar{C}_h$. The mathematical expression of this optimization is

$$\min_{X_{h_{i,h}}} \sum_i P_i X_{h_{i,h}} \quad s.t. \quad \bar{C}_h = \left(\sum_i \alpha_{Xh_{i,h}} {X_{h_{i,h}}}^{\rho_h}\right)^{1/\rho_h}$$

The household's sub-utility function is CES, where $\alpha_{Xh_{h,i}}$ is the household's share parameter for good *i*, and ρ_h is the parameter based on the household's elasticity of substitution, such that $\rho_h = \frac{\sigma_h - 1}{\sigma_h}$. The first-order conditions require:

$$\frac{\partial C_h / \partial X_{h_{i,h}}}{\partial C_h / \partial X_{h_{j,h}}} = \frac{P_j}{P_i} \qquad (65)$$

Equation 65 identifies the household's demands of market consumption goods.

The other key aspect of the household's decision-making process is the income available for consumption. Household income is derived through the same two-stage process as described in Chapter 3.

4.3.3 Ecopath with Ecosim Model

The food web is modeled with the Ecopath with Ecosim (EwE) food web modeling system. The EwE model is an established model in the ecology literature that incorporates species populations, trophic levels, and energy (food) availability to model a specified food web, and is used to analyze ecosystem responses to past and future perturbations to aquatic ecosystems (Christensen & Walters, 2004; Cox & Kitchell, 2004; Kao, Adlerstein, & Rutherford, 2014; Langseth, Rogers, & Zhang, 2012; Plagányi & Butterworth, 2004). EwE consists of two parts, Ecopath, a modeling system used to represent the flows between elements in an ecosystem (Christensen, Walters, & Pauly, 2005), and Ecosim, which allows for the simulation of potential disturbances to the systems through its dynamic modeling capabilities (Christensen & Walters, 2004). Once a model Ecopath has been built and parameterized for the desired food web, it can be used with the Ecosim interface to simulate disturbances to the benchmark. Ecopath models are built on the assumption that mass balances, that is, any input to the system must be accounted for. Over a given time period mass eithers leaves the system or accumulates in the system (Christensen et al., 2005). In Ecopath, mass balances on the species level.

There are two core equations that Ecopath models are parameterized on, an equation that describes the rate of generation for a specific group in the food web, and one that describes the energy balance of each group in the food web (Christensen et al., 2005). I describe the model below using the same variables as Christensen et al., (2005). Note, some of the variables in this section have the same combination of letters as the economic model but have a different interpretation in the EwE context.

In the Ecopath model the production (rate of generation of a species) is equal to the sum of the fishing mortality rate, predation mortality, biomass accumulation, net migration and other mortality (Christensen et al., 2005).

$$P_i = Y_i + B_i M2_i + E_i + BA_i + P_i(1 - EE_i) \quad (66)$$

where P_i is the total production rate of group i, Y_i is the total fishery catch rate, $M2_i$ is the total predation rate for group *i* (rate that group *i* is consumed by other groups), B_i is the biomass of the group, E_i is the group's net migration rate, BA_i is the biomass accumulation rate, and $P_i(1 - EE_i)$ is the 'other mortality' rate (Christensen et al., 2005). Equation 66 shows that the production, or increase to the species group, exactly equals the loss to the species, either through mortality or migration.

The second equation in the Ecopath model ensures that mass balances between groups. Equation 67, the mass balance equation, states that for each species group, their consumption (Q_i) is equal to the sum of production (P_i), respiration (R_i) and unassimilated food (U_i) (Christensen et al., 2005; Christensen & Pauly, 1992; Christensen & Walters, 2004).

$$Q_i = P_i + R_i + U_i \quad (67)$$

Ecopath requires that values be given for B_i, P_i, and Q_i, which can be used to calculate the other mortality rate, $P_i(1 - EE_i)$ (Byron et al., 2015). Additionally, for every species group, the mortality rate due to fishing and the diet composition must be provided for every variable to be identified in the Ecopath parameterization (Byron et al., 2015).

4.3.4 Linking the EwE and CGE Models

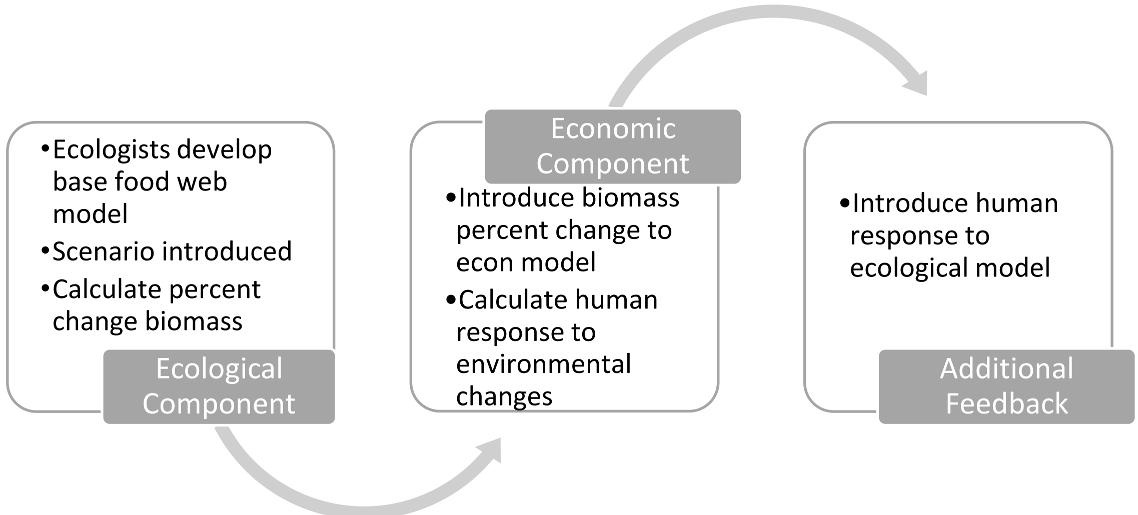

Figure 15: CGE and EwE Link

In this analysis, I further the work on current bioeconomic models by truly integrating the CGE and EwE models. Changes in species' populations (modeled by EwE) change human behavior, and changes in human behavior (modeled by the CGE) change the harvest of species. In performing a simulation, the feedback between the two systems is best described as recursive dynamic. In the first stage, the EwE interface models changes in species' biomass for a given disturbance scenario to the food web. In the second stage, the new biomass levels are fed into the CGE model as a counterfactual scenario to the benchmark, and the agents make their new optimal choices based on that biomass level. In the third stage, the commercial and recreational harvest from the CGE model is reported and biomass levels are adjusted in the EwE model (Figure 15), and the food web's response to those changing populations is reported. The CGE model is built in the General Algebraic Modeling System (GAMS).

4.4 Application: Lake Erie Asian Carp Invasion

The model and methods presented thus far can be applied to a variety of regions and scenarios. I choose to focus on a potential Asian carp invasion of Lake Erie. Lake Erie is an interesting focal point because of its susceptibility for invasive species and the variety of species it is home to that are targeted in both commercial and recreational fishing.

In developing and applying the combined CGE and EwE bioeconomic model, I worked with an interdisciplinary team that facilitated the development of a truly bioeconomic model that takes into account the many facets of environmental and natural resource problems. The team includes Marion Wittman, director of Natural Reserve System at University of California – Santa Barbara, who led a team that gathered expert opinions on the likely impacts of Asian carp on other species in Lake Erie presented in Wittmann et al., (2015), Hongyan Zhang and Ed S. Rutherford, from the National Oceanic and Atmospheric Administration, Great Lake Environmental Research Laboratory, who developed the Ecopath with Ecosim model specific to the Asian carp invasion scenario, and Jason T. Breck from the Department of Computer Sciences at the University of Wisconsin who developed the code that allows the CGE and EwE systems to communicate with each other. Travis Warziniack from the USDA Forest Service, and David C. Finnoff from the Department of Economics and Finance at the University of Wyoming, reviewed the economic modeling components and helped refine the scope and research questions of the project. The interdisciplinary approach is one of the strengths of this work; by working with experts from other fields, I can more accurately integrate the ecological system with the economic system. In the following sections I provide ecological background on the study area and describe how the economic and ecological modeling components are calibrated.

4.4.1 Ecological Background

I begin by providing ecological background including information on Asian carp, characteristics of Lake Erie and why it is susceptible to invasions, and how the EwE model used here is calibrated.

4.4.1.1 Invasive Species and Asian Carp

In the Great Lakes there are at least 184 established invasive species (Ricciardi, 2006; Zhang et al., 2016). Two species of concern, bighead carp (*Hypophthalmichthys nobilis*) and silver carp (*H. molitrix*), commonly referred to as 'Asian carp,' have received increased attention in recent years (Great Lakes Commission & St. Lawrence Cities Initiative, 2012). The carp were intentionally introduced to North America in the 1970's to improve pond water quality and support aquaculture in the southern United States, and have since invaded the entire Mississippi River basin (Kolar et al., 2007). The carp are filter feeders and consume large quantities (20-40 percent of their body weight) daily of species at the base of the food web such as phytoplankton, small zooplankton, detritus, and bacteria (Zhang et al., 2016; Great Lakes Commission & St. Lawrence Cities Initiative, 2012). Asian carp are also known to have high growth and fecundity rates, and can reach up to 110 pounds (Zhang et al., 2016; Great Lakes Commission & St. Lawrence Cities Initiative, 2012). These characteristics of Asian carp create concern that Asian carp will out-compete forage fish and planktivorous larval fish, and consequently decrease important commercial and recreational fish species (Wittmann et al., 2015; Zhang et al., 2016). In the Illinois River, silver carp also impede recreational boating by leaping out of the water, creating a nuisance and sometimes causing bodily harm (Great Lakes Commission & St. Lawrence Cities Initiative, 2012).

In the ecological community, there is debate about potential impacts of Asian carp on the Great Lakes. In this analysis, I follow the lead of my collaborators and use their estimates of what an Asian carp population would look like in Lake Erie and what the potential associated impacts are (Wittmann et al., 2015; Zhang et al., 2016).

4.4.1.2 Ecological Characteristics of Lake Erie

With a volume of 480km^3 and a surface areas of 25,657km^2, Lake Erie is the smallest of the Great Lakes by volume and the second smallest of the Great Lakes by surface area, and in terms of hosting species populations, is the most productive (Hubert & Quist, 2010; Zhang et al., 2016). Lake Erie has been exposed to multiple anthropogenic stressors such as contaminants, wetland destruction and invasive species introduction, and is vulnerable to invasion of non-indigenous species through multiple vectors (Zhang et al. 2016). The ballast water discharge from sea-faring ships and the connection to the St. Lawrence Seaway are two well-known pathways of introduction of non-native species and are credited with introducing the well-known invasive species zebra mussels (*Dreissena polymorpha),* and sea lamprey (*Petromyzon marinus)* (Griffiths et al., 1991; Koonce et al., 1996). When it comes to Asian carp, two waterways have been identified with high likelihood for passage into the Great Lakes – the Chicago Sanitary and Ship Canal (CSSC) that connects the Mississippi water basin and Lake Michigan and the Wabash River which occasionally connects to Lake Erie (Kocovsky et al., 2012). The Wabash River is already host to Asian carp and there is concern that carp could be transferred to Lake Erie during occasional flood events that connect the Wabash River to the Maumee River (Great Lakes Commission & St. Lawrence Cities Initiative, 2012).

4.4.1.3 Parameterization of EwE

In applying the model to Lake Erie, the disturbance to the ecological system comes through the parameterization of the Ecopath model. The EwE model used in this analysis is described in full in Zhang et al., (2016). An observable population of Asian carp does not currently exist in Lake Erie, so their characteristics in the Lake Erie food web was initially unknown. To determine the required parameters needed for the EwE simulation (total production, consumption, biomass, other mortality, and diet composition) expert opinions from a structured expert judgement (SEJ) elicitation by Wittmann et al., (2015) were used. SEJ is a performance-based technique used to elicit and combine expert judgments in order to quantify uncertainty for variables of interest (Cooke 1991). Expert elicitation is a useful methodology for considering potential impacts or outcomes of complicated problems when the empirical data required for decision-making may be inexistent, sparse or debated (Rothlisberger, et al. 2012). SEJ has been widely used in applications ranging from nuclear regulation to food safety (Cooke 1991).

Wittmann et al. (2015) elicited the judgments of 11 Great Lakes fisheries and Asian carp experts to quantify the impact of Asian carp establishment in Lake Erie. The experts were asked to give their estimates for walleye and yellow perch biomass in a scenario in which Asian carp (bighead and silver) were assumed to be established in Lake Erie. Experts provided 5^{th}, 50^{th}, and 95^{th} uncertainty percentile ranges of their subjective probabilities distributions for these variables of interest. Expert assessments were then combined using a weighted arithmetic mean in which individual assessments were weighted according to their (relative) expertise based on performance on a set of calibration variables. Calibration variables were from the experts' field of specialization, closely resembling the variables of interest, and whose values were not known to the experts at the time of elicitation, but were realized *post hoc* (Cooke & Goossens, 2000; Cooke, Mendel, & Thijs, 1988). For example, experts were asked to quantify the whole-lake biomass of yellow perch in Lake Erie in 2011, a value measured annually by regional agencies and for which results were released in 2012, after the elicitations were conducted. Performance-based weights were determined by calculating a calibration score and information score based on the calibration variables. The calibration score represents statistical accuracy, and is the probability that the divergence between the expert's probabilities and the calibration variable realizations might have arisen by chance. The information score is the degree to which the expert's probability distribution is concentrated. Of these two, statistical accuracy is more important, and informativeness was used to discriminate between statistically accurate assessments. Zhang et al. (2016) used the SEJ to estimate median values and uncertainty for the EwE input variables total production, consumption, biomass, and diet composition.

The food web model of Lake Erie was first developed without Asian carp with a balanced Ecopath model of year 1999 and a calibrated Ecosim model from 1999 to 2010, and includes 47 species groups (Mason, 2003; Zhang et al., 2016). Once the base EwE model was calibrated, the 'counterfactual' scenario was developed and Asian carp were added to the model. Two age groups of each species of Asian carp were included, age 0 to 12 months, and age 1 and older, to account for the vulnerability to predators of age 0 Asian carp (Zhang et al., 2016). Most Asian carp age 1 and older are too large to be prey for other species (Zhang et al., 2016). This model setup allows for potential direct and indirect effects of an Asian carp on the food web as both prey and predators to be accounted for.

4.4.2 Economic Parameters

This section provides background and parameterization of the economic components in the model that were not already described in Chapter 2. I begin by describing the initial values for household recreation and non-market consumption and then describe the parameterization of production components.

4.4.2.3 Initial Household Recreation Values

There are multiple variables in the households' recreational choices that required initial values outside of those supplied by the IMPLAN data and the SAM. These variables include the household's WTP for target species and total trip costs. Initial values for the households' consumption of non-market goods are taken from Besedin et al., (2004) in which the authors use a random utility model to estimate angler's average cost of fishing trip (including the opportunity cost of missing work) and the angler's willingness to pay for catching a specific species. Besedin et al., (2004) focus on four species specific groups: bass, perch, walleye-pike, and salmon-trout. The recreational species of interest I focus on include smallmouth bass, white bass, perch, walleye, lake trout, rainbow trout, and lake whitefish. For species that are not specifically defined in the Besedin et al., (2004) work, I use a willingness to pay from a species group with similar characteristics. The initial willingness to pay values are shown in Table 12. The WTP values provide benchmark values for the initial price of species targeted trips (P_{T_f}). Besedin et al. (2004) also provide estimates on the total number of days people targeted each species in the Great Lakes region which are used to find aggregate trip costs for each species.

Table 12: Initial Willing to Pay Values for Recreational Targeted Species

Species	Willingness to Pay ($/fish)
Smallmouth Bass	12.86
White Bass	12.86
White Perch	2.47
Yellow Perch	2.47
Walleye	18.43
Lake Trout	20.13
Rainbow Trout	20.13
Lake Whitefish	2.47

Values from Besedin et al., (2004)

4.4.2.4 Calibration of CGE Parameters

Table 13: Elasticity Values
Elasticity parameter values by type and sector

Parameter	Elasticity Type	Sector & Value
σ_{Q_i}	Total Supply	Fishing Sectors: 3.90 Recreational Fishing: 2.79 All Other Sectors: 2.79
σ_{X_i}	Total Demand	Fishing Sectors: 1.42 Recreational Fishing: 1.42 All Other Sectors: 2.12
σ_i	Production Value -Added	All Sectors: 0.8672
σ_{h_h}	Household Consumption	All Households: 0.8672

Elasticities taken from Finnoff & Tschirhart 2008

The exogenous parameters in the economic side of the model can be classified into two types: those that can be calibrated directly from the SAM and those that cannot. Most parameters used in the simulations are calibrated from the benchmark data. Elasticities cannot be calibrated from data and instead need to be collected from alternative means. The elasticities for each of the constant elasticity of substitution functions are taken from the literature. In their 2008 work, Finnoff & Tschirhart present elasticity of substitution values for a variety of sectors including fishing sectors. Most of the elasticities presented by Finnoff & Tschirhart (2008) are averages of elasticities presented in the literature, and given the similar fishery focus and economic model set-up, I use their values in this analysis (Table 13).

The elasticities present in the Cobb-Douglas harvest functions also need to be found with alternatives means, and they are estimated using historical data. The estimation of those parameters is presented in Chapter in 2, the estimates for the relevant species are presented in Table 14. A discussion of the use of the estimated Cobb-Douglas parameters follows.

Table 14: Cobb-Douglas Harvest Parameter Estimates

VARIABLES	Yellow Perch	Walleye	White Perch	Lake Whitefish	White Bass
γ_1	0.763***	0.647***	0.307***	0.881***	-0.354
	(0.124)	(0.041)	(0.0938)	(0.251)	(0.367)
γ_2	0.225*	0.464***	0.0662**	0.235**	0.185**
	(0.124)	(0.061)	(0.0330)	(0.116)	(0.0737)
γ_o	-8.288***	0.031*	0.391	1.107	-4.814**
	(1.397)	(1.043)	(0.545)	(1.386)	(1.947)
Observations	35	37	26	26	26
ARIMA Lags	AR2 MA2	AR3 MA3	AR4 MA2	MA1	AR2 MA4

Standard errors in parentheses
*** p<0.01, ** p<0.05, * p<0.1

The estimates of both the stock elasticity and effort elasticity of harvest for walleye, white perch, and lake whitefish have p-values of less than 0.05, and so I accept their values, and set $\alpha_f = \gamma_1$ and $\beta_f = \gamma_2$ for those species. Yellow perch and white bass only have one coefficient with a p-value of less than 0.05, so I assume that constant returns to scale exists and that $\beta_f = 1 - \alpha_f$, and use the coefficient estimated at the highest significance to calculate the other, so for yellow perch $\alpha_f = \gamma_1 = 0.763$ and $\beta_f = 1 - 0.763 = 0.237$, and for white bass $\beta_f = \gamma_2 = 0.185$ and $\alpha_f = 1 - \beta_f = 0.815$. Stock and effort data is not available for the other species of interest in the model, so I use the average of the elasticities of yellow perch, walleye, white bass, white perch, and lake white fish. The effort and stock elasticities of channel catfish, bigmouth buffalo, freshwater drum, and carp are $\alpha_f = 0.683$ and $\beta_f = 0.234$.

The catchability coefficient is calibrated from the benchmark data in a way to ensure that the initial harvest level is consistent with the stock and the effort levels present.

$$q_f = \frac{H_f}{E_f^{\alpha_f} {S_f}^{\beta_f}}$$

Once the initial level of effort has been determined in the yellow perch sector, the output price of effort can then be addressed. In the base case, all prices except the output prices of the various fish are assumed to equal one. To reconcile the economic IMPLAN data with the GLFC and ODNR harvest data for the fish species, it is assumed that the initial price of fish, P_f, is such that Euler's equation holds, and there are zero profits, $(\alpha_f + \beta_f)\, P_f H_f = P_{Ef}^{ROA}\, E_f$. The initial price of fish equals $P_f = \frac{E_f P_{Ef}^{ROA}}{(\alpha_f + \beta_f)\, H_f}$.

4.5 Ecological Disturbance Analysis

To evaluate the response to the Asian carp disturbance, the model performs 120 iterations of feedbacks. When a disturbance is introduced into the food web in EwE, it can take several iterations for the food web to reach a new equilibrium among the species groups. Performing the feedbacks 120 times allows the model to reach an equilibrium and move past the initial noise the introduction of a new species causes. Throughout the 120 iterations, it is assumed that no additional exogenous shocks occur in the system.

There are two objectives of the policy analysis; evaluating the impacts of a potential Asian carp invasion in Lake Erie, and comparing how the estimated impacts from such a disturbance depend on feedbacks between the economic and ecological systems. To address these objectives, I first describe the ecological responses over the entire time frame, then describe the economic responses, and finally I compare how these impacts would differ if the potential invasion was evaluated with only the economic or ecological system independent of the other.

4.5.1 Ecological Responses

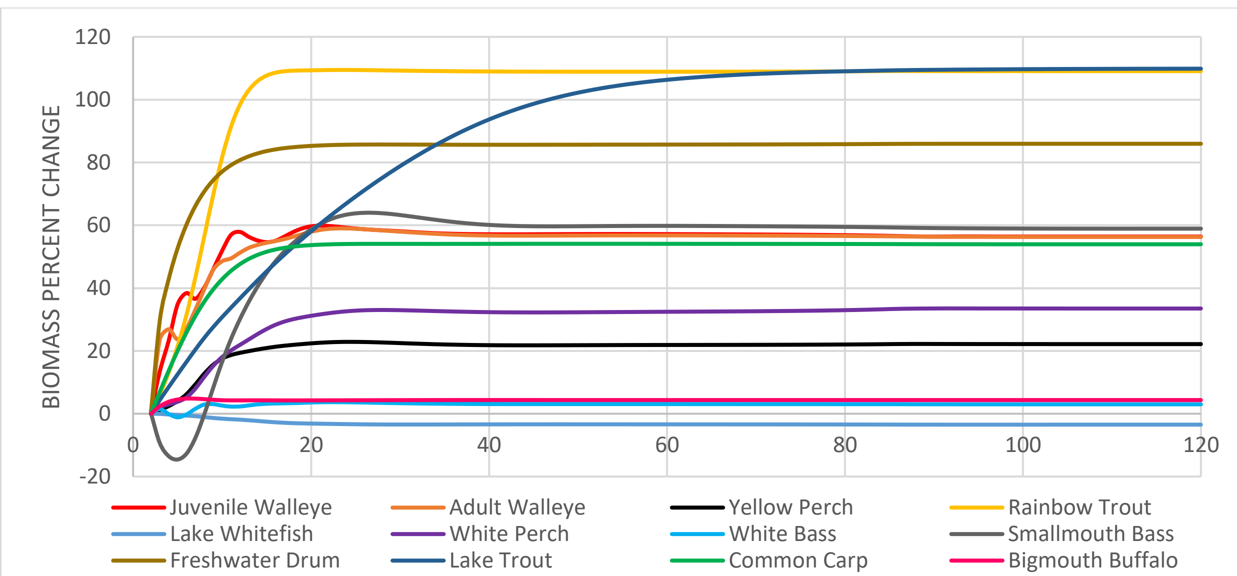

Figure 16: Percent Change in Species Biomass
Percent change in biomass from initial stock value. Positive percent change indicates increase from initial value.

Figure 16 shows the percent change in biomass of primary species from the initial value over the 120 time steps. Over the time frame the biomass of many target species increase. However, the population of Lake Whitefish decreases in the long run. The population of all species fluctuate in the first ten years, due to the population responding to the introduction of Asian carp into the food web. It's also important to note that adding Asian carp to the food web has more impacts than those presented in the in Figure 16. While many of the target species happen to increase following an invasion by Asian carp, other trade-offs of a new species entering the lake must occur somewhere. In these simulations, alewife, gizzard shad, detritus, rainbow smelt, and shiners are among some of the species that experience population declines due to the additional species entering the system. Further exploration of the ecological impacts of non-target species is warranted, but for the purposes of the analysis in presented in this paper, I focus only on target species at this time.

The decrease in the population of lake whitefish is likely due to competition with Asian carp, either for its own prey, or through the diets of its prey. For example, there is some overlap in the diets of lake whitefish and the simulated Asian carp, but there is also competition between the species that lake whitefish prey on and Asian carp, these two forms of competition cause a slight decrease in the lake whitefish population.

Some of the species' populations that saw increases from an Asian carp invasion also have diets that overlap with Asian carp. However, the increase in species populations is attributable to the consumption of age-0 Asian carp by those species. For example, the adult and juvenile walleye both saw population increases. While the diet of walleye does not overlap with the diet of Asian carp, walleye can prey on young Asian carp, and therefore use Asian carp as an additional food supply.

4.5.2 Economic Responses

There is a variety of variables in the CGE model that can be evaluated in response to the Asian carp invasion. In this section I focus on the changes in commercial harvest and recreational catch of fish species, production effort, prices, supply of goods in other sectors, and household welfare. I start by focusing on responses in the marketed goods sectors, then move to responses in non-market goods (fishing trips), and finally discuss the welfare implications.

4.5.2.1 Market Reponses

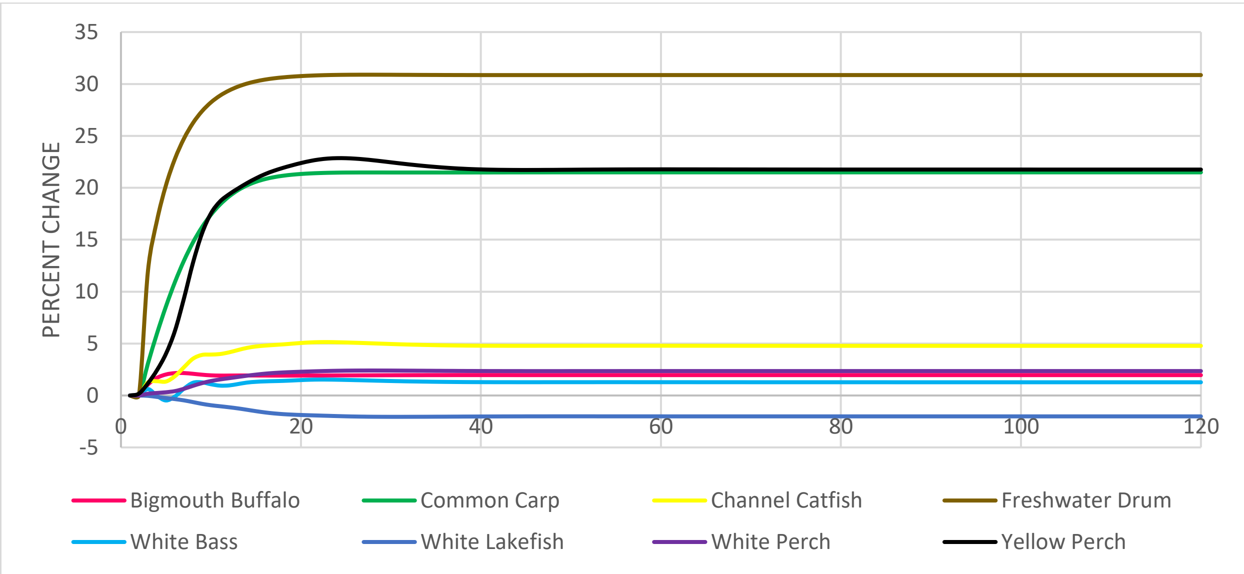

Figure 17: Percent Change in Commercial Harvest
Percent change in commercial harvest, calculated initial value minus new value. Positive value indicates that harvest increased after Asian carp invasion.

I begin by exploring the responses of market goods to an Asian carp invasion, including both the commercial fishing and non-fishing sectors. Figure 17 shows the percent change in commercial harvest. The commercial harvest of lake whitefish was the only fishing sector that experienced a decrease in commercial harvest. The commercial harvest of all other species increased, with freshwater drum experiencing the largest percent increase. Commercial harvest of walleye has a fixed total allowable catch (not a function of changing stock), so it does not experience changes in harvest and is not included on the graph. The percent changes in commercial harvest follow a similar form as the percent change in stock, but have different magnitudes. Freshwater drum for example, sees an increase of about 80 percent increase in biomass, but harvest only increased by about 30 percent.

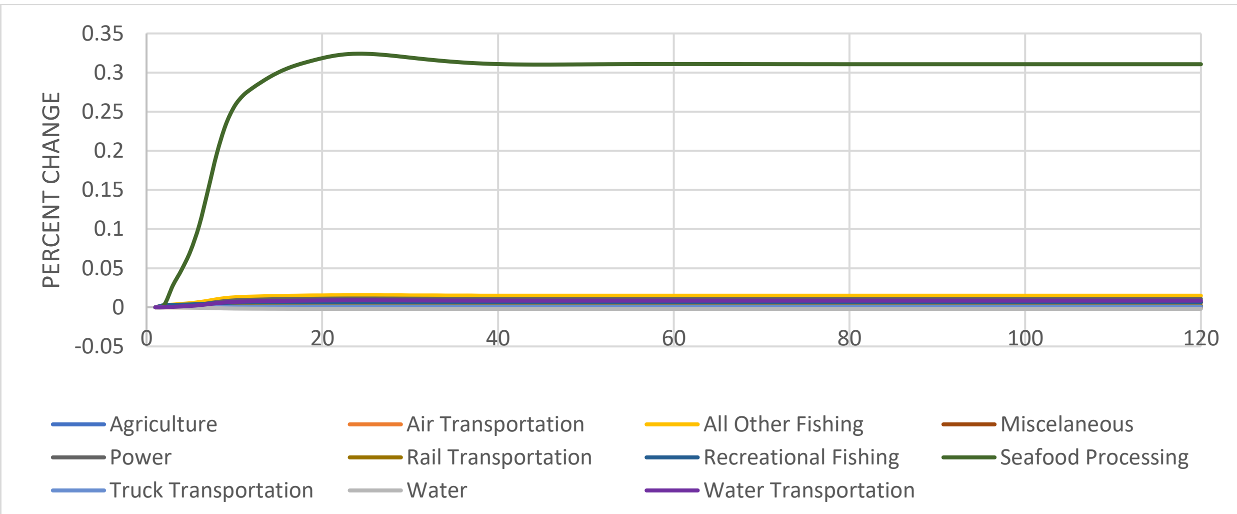

Figure 18: Percent Change in Supply of Non-Fishing Sectors
Percent change of supply in non-fishing sectors. Positive value indicates increase after Asian carp invasion.

We can also look at how supply in the non-fishing sectors responds over the time frame. Figure 18 shows the percent change of supply in non-fishing sectors. The response of other sectors to the Asian carp invasion is minimal with less than 0.35% change in all sectors, and just slight increases in most. Of the small changes experienced by the non-fishing sectors, seafood processing experienced the largest changes in supply. The higher response of the seafood processing sector to an Asian carp invasion is not surprising, seafood processing is a small sector that relies relatively heavily on the commercial fishing sectors for its production, particularly yellow perch, and thus is sensitive to changes within the fishing sectors.

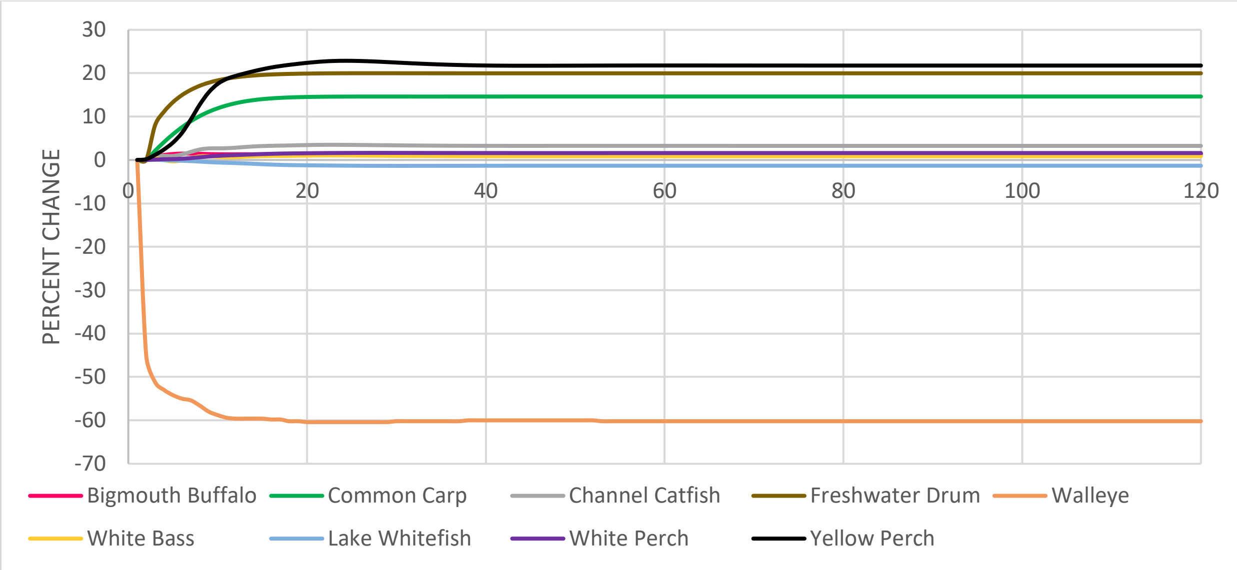

Figure 19: Percent Change in Effort - Fish Sectors
Percent change of effort employed in commercial fishing sectors. Positive percent change indicates increase after Asian carp invasion.

Another aspect of production response is the amount of effort used by firms. Figure 19 shows the percent change of effort in the commercial fishing sectors. Again, for most species, the effort choices follow the same shape as the changes in stock. Walleye, however, is an exception. Recall that the commercial TAC of walleye is fixed, and so as the stock of walleye increases, harvest remains the same, but the amount of effort required to harvest the fixed TAC decreases, as we see in Figure 19. When comparing the magnitude of the percent change in harvest and the percent change in effort, we see that even though the choices follow a similar path, the magnitudes are different for some species. A 30% increase in the harvest of freshwater drum occurs with a 20% increase in effort in that sector. In the yellow perch sector, however, there is a 22% increase in both harvest and effort. The different relationships between harvest and effort choices across the fishing sectors can be related back to the returns to scale in each sector. The estimated Cobb-Douglas harvest function for yellow perch exhibited constant returns to scale, while the freshwater drum exhibited slightly increasing returns to scale, so more harvest is gained from a smaller increase in inputs. Yellow perch and white bass have constant returns to scale in their estimated Cobb-Douglass harvest function, white perch has decreasing returns to scale, and all other fishing sectors have increasing returns to scale.

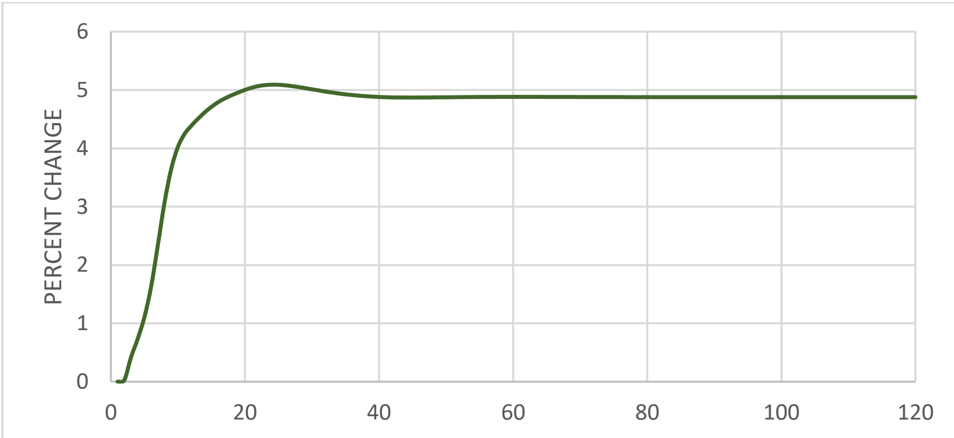

Figure 20: Percent Change in Effort - Seafood Processing
Percent change of effort in the seafood processing sector. A positive value indicates an increase following a simulated Asian carp invasion.

Figure 20 shows the percent change of effort in the seafood processing sector which is the only non-commercial fishing sector that exhibited changes in effort. The seafood processing firm increases their effort as their production increases. The total supply of seafood processing increased by only 0.35 percent at its highest, but their effort level increased by nearly 5 percent. Again, this is due to their reliance on yellow perch and the other commercial sectors in their production process. The effort measure includes intermediate inputs from other sectors, so the increase in effort reflects the increase in commercial fishing inputs.

Market price is another variable of interest. Figure 21 shows the percent change in fish sector output prices and Figure 22 and shows the percent change in the market price of seafood processing output. As we would expect, the price changes are inverses of the harvest and supply graphs. For example, as the population of lake whitefish decreases, its price increases, and as the supply of freshwater drum increases, its price decreases. The same relationship holds true in the non-fishing sectors as well, which is shown by the percent change in price of seafood processing.

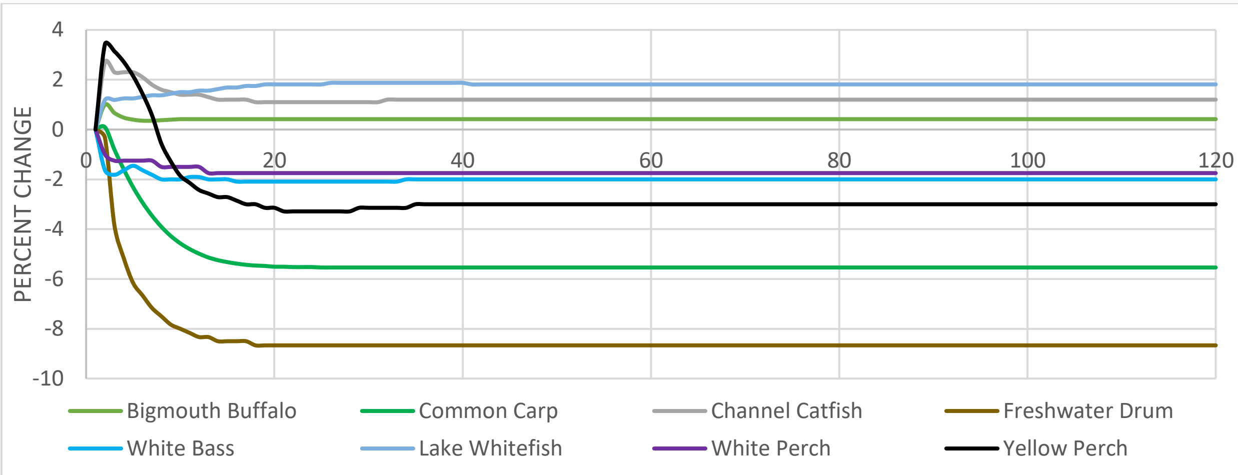

Figure 21: Percent Change in Fish Sector Output Prices
Percent change of prices in fishing sectors. Positive percent change indicates increase after Asian carp invasion.

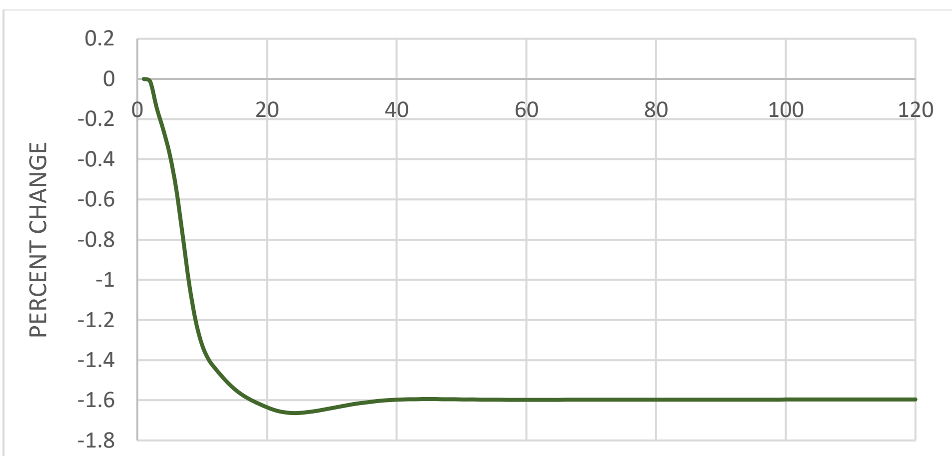

Figure 22: Percent Change in Prices in Seafood Processing Sector
Percent change of prices in seafood processing shown. Positive percent change indicates increase after Asian carp invasion. All other sectors experienced price increases of less than 0.01%

4.5.2.2 Non-market Reponses

I now evaluate how the consumer responded to the Asian carp invasion through their choice of fishing trips and recreational fishing catch. Figure 23 shows the percent change in the recreational fish catch. The recreational catch of lake trout and smallmouth bass increase. The catch of rainbow trout, white bass, walleye, lake whitefish, and perch all decrease even though lake whitefish is the only one of those species that experiences a decrease in biomass in the long run.

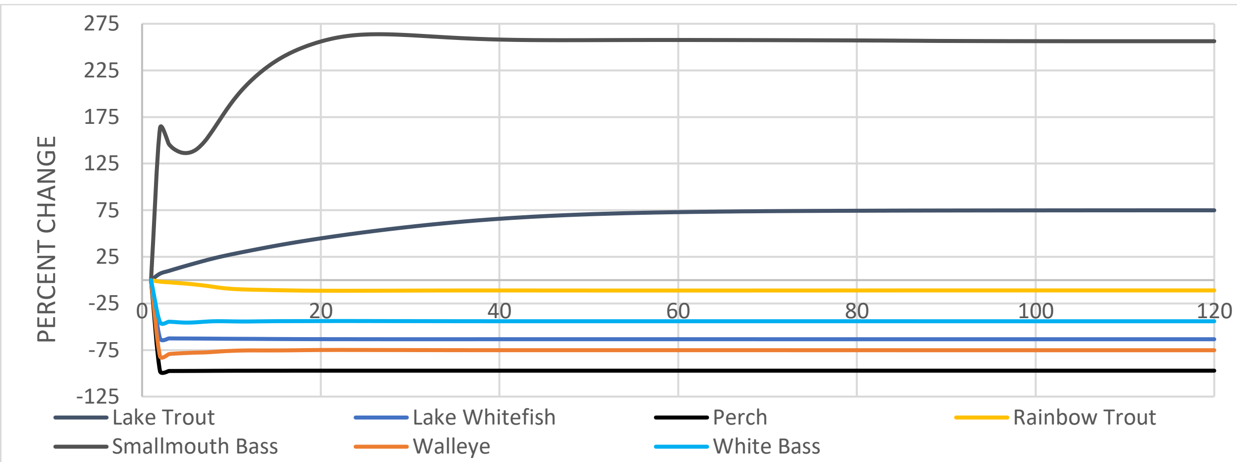

Figure 23: Percent Change in Recreational Fishing Catch
Percent change in recreational catch, positive value indicates increase in catch after introduction of Asian Carp.

The changes in recreational fishing catches do not mirror the changes in biomass like the commercial harvest does. The variation in results between the commercial harvest and recreational fishing is due to the difference between the firm and the household decisions and model set up. The firm in this model only targets a single species, so they cannot substitute between species and their production decisions only take into the account the stock of the fish they are targeting. Their production decisions, therefore, closely match the changes in stock of the species they target. The household, on the other hand, is assumed to substitute between species with relative ease and makes their recreational choices over a variety of species and can target more than one species. Their decision on which fish species to target considers the stock of multiple species, and they make their choices based on both their preferences and opportunity cost. Catch of species may decrease even if its stock has increased, the decrease would occur if the relative marginal utility per dollar of opportunity of another species has increased.

I next turn to the prices and consumption of non-market goods. Figure 24 shows the percent change in species-specific trip prices by household. Recall that a household's trip price takes into account the cost of effort, including the opportunity cost of lost wages. The price of trips targeting lake trout and smallmouth bass decrease for all households. The price of trips targeting all other species, perch, walleye, white bass, rainbow trout, and lake whitefish increase for all households. Figure 23 showed the percent change in total recreational catch, and Figure 25 shows the percent change in recreational catch for each species by household, in both graphs we see that the households increase their catch of species with a reduced trip cost and reduce their catch of species with an increased trip cost, which is what we would expect.

Lake trout and smallmouth bass are two of the species whose biomass is projected to increase by the greatest percent with an Asian carp invasion. The increase in biomass implies that less effort is required to catch a fish, which in turn reduces the price of a trip. Walleye, perch, rainbow trout, and white bass also experienced population increases from the Asian carp invasion, but the household recreational catch of these species decreased. The different responses to species populations by the household is a function of the different trip cost of obtaining a fish of a specific species. The cost of trips targeting smallmouth bass and lake trout decrease, while all other species trips costs increase, so the household moves towards the species with the lower trip cost, and spends more time targeting smallmouth bass and lake trout.

Notice across all species, Household 2 experiences the smallest percent change in price. This is related to the opportunity cost of missed wages, Household 2 experiences the smallest percent increase in income and thus also has the smallest additional loss of missing work. The percent change in recreational catch of specific species is fairly similar across households, although we do see some differences in smallmouth bass, lake trout, and white bass, and rainbow trout. Household 2 experiences a smaller percent increase in the catch of smallmouth bass and lake trout than the other households, but a larger percent decrease in white bass and rainbow trout. This suggests that they are more constrained by their income than the other households.

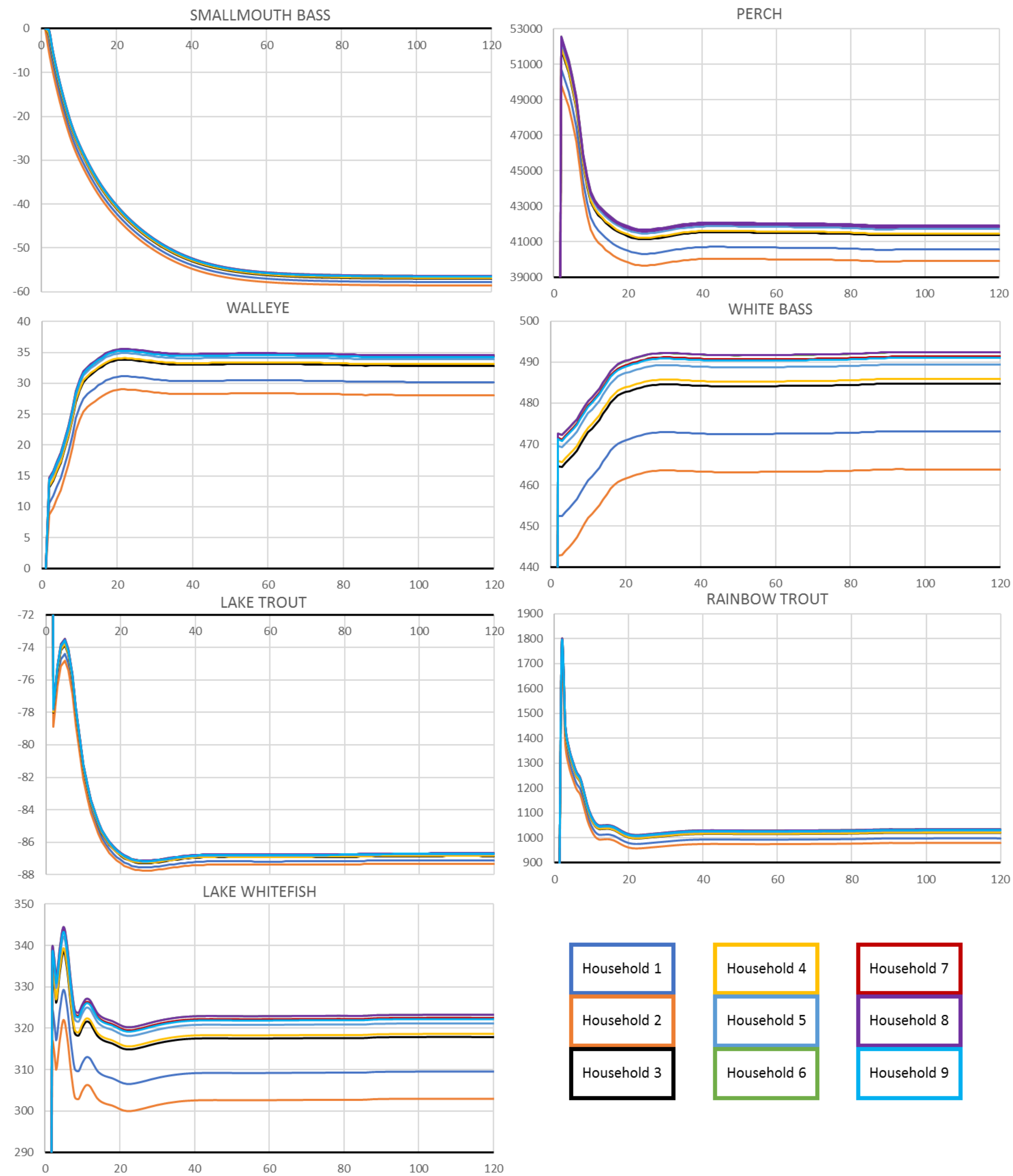

Figure 24: Percent Change in Species Specific Trip Prices
Percent change in species specific prices. Positive value indicates increase following Asian carp invasion. Note that y-axis values of percent change differ from panel to panel.

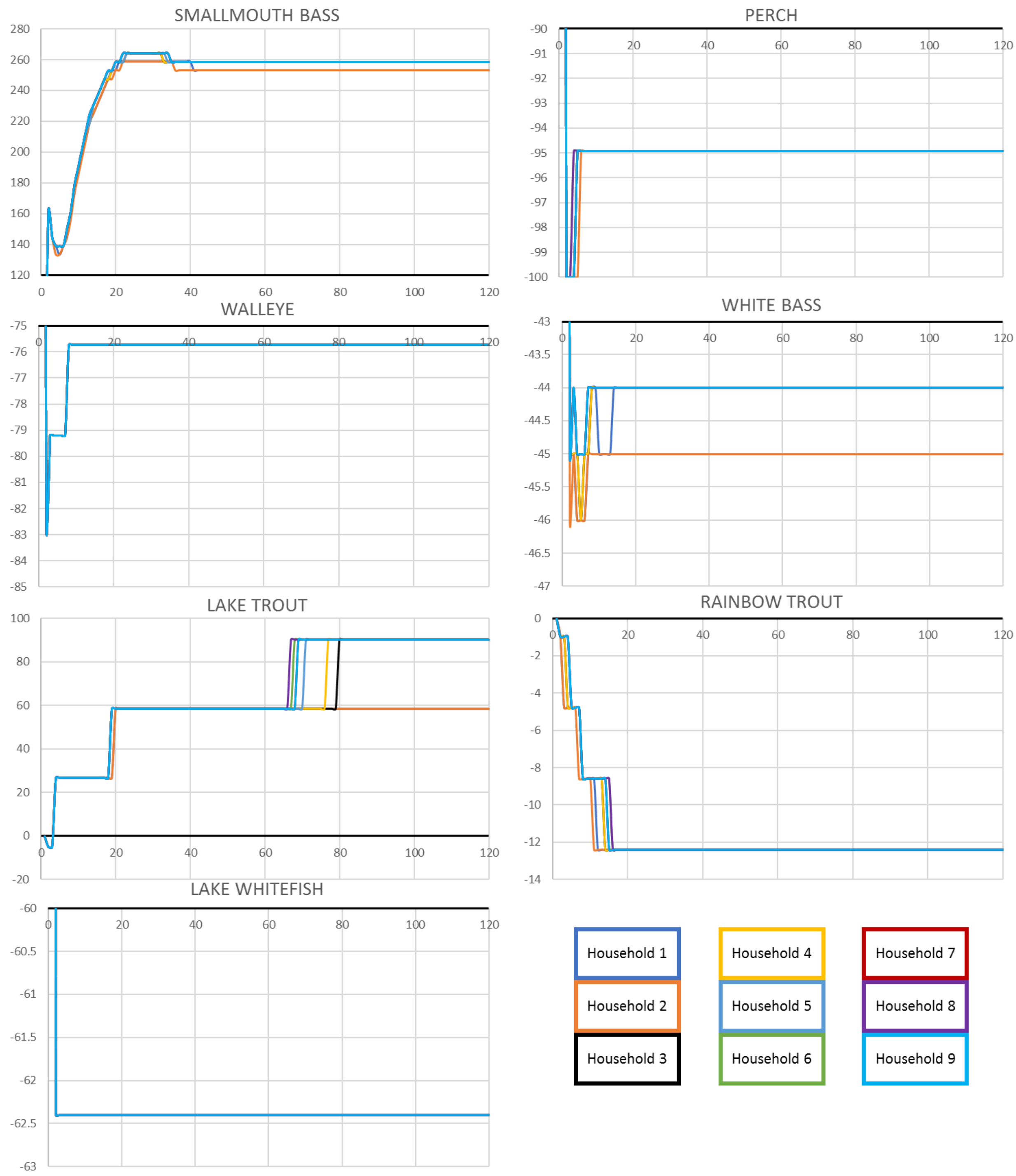

Figure 25: Percent Change in Household Recreational Catch by Species
Percent change in species specific recreational catch by household. Positive value indicates increase following Asian carp invasion. Note that y-axis values of percent change differ from panel to panel

4.5.2.3 Welfare Analysis

For the next point of discussion, I turn to the household welfare implications of the Asian carp invasion. Welfare is measured using a money metric indirect utility function as defined by Varian (1992) and calibrated by Rutherford (2008) as described in Chapter 3. To compare welfare over the 120-year timeframe, I calculate the benchmark money metric utility and then calculate its percent change for each time step. The money metric utility includes both market and non-market goods. Figure 26 shows the percent change in money metric utility for each household and the money metric utility summed over all households. Each household experienced improved welfare over the entire time frame. Household 2 experienced the largest percent increase in their welfare, and Household 8 experienced the smallest percent gain in welfare.

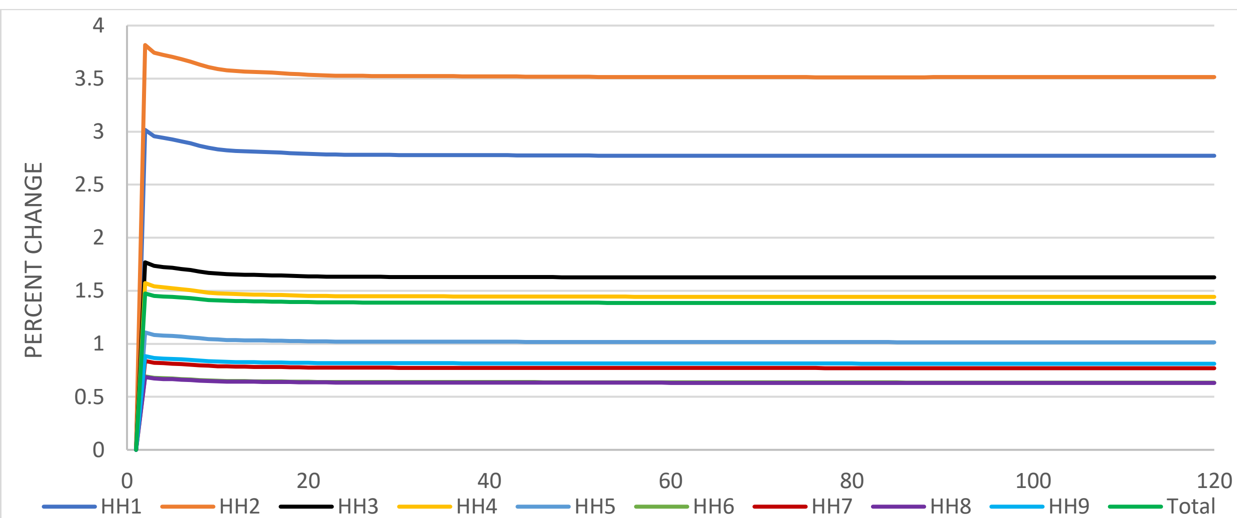

Figure 26: Percent Change in Money Metric Utility
Percent change of money metric utility. Positive value indicates increase following Asian carp invasion.

The changes in household welfare is driven by their purchasing power and can be broken down into two parts, changes in income and changes in prices. Figure 27 shows the percent changes in disposable income for each household. Each household experienced slight increases in disposable income. Household 3 displayed the largest percent increase income. While all households experienced increased incomes, the percent changes in disposable income are quite a bit smaller than the increase in money metric utility, which indicates that purchasing power, rather than absolute change in income is driving the welfare improvements.

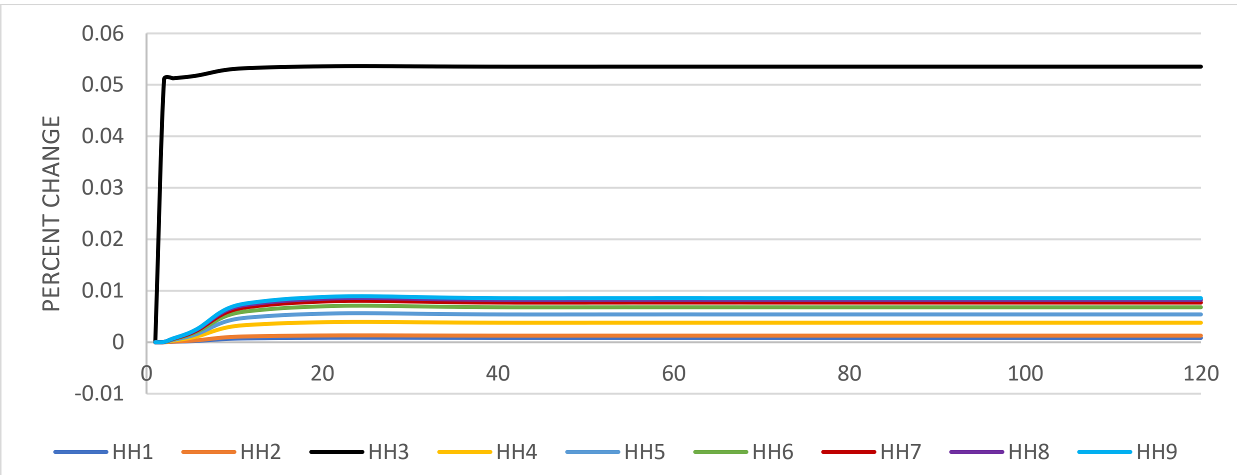

Figure 27: Percent Change in Disposable Income
Percent change of disposable income for each household. Positive value indicates increase following Asian carp invasion.

Household welfare depends on household disposable income and their consumption of both market and non-market goods. There are two sets of prices the households face in their decision making, price of marketed consumption goods and the price of the non-market recreational goods. We saw in Figure 21 there was some movement in the prices in the commercial fishing sectors, however, household consumption of commercially harvested fish is fairly limited (initial household demand shares are shown in Table 15), so even though some prices of commercial fish increase and some decrease, it is likely to have little impact on the households and their welfare. The seafood processing sector had the largest percent change in price of the non-fishing sectors with an average decrease from the benchmark of 1.06%, however seafood processing only makes up approximately 0.02% of the household demand. The largest share of household consumption is from the miscellaneous sector, with an average of 97 percent of household consumption coming from the sector. The price of goods from the miscellaneous sector only increased by a maximum of 0.007 percent over the 120-year time frame, so while the households may be sensitive to changes in the miscellaneous sector, the welfare implication are also likely to be small from the non-fishing sector of marketed goods.

Table 15: Household Benchmark Percent Shares of Market Consumption Bundle
Initial percent shares of household demand of marketed goods.

Sectors	Households								
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
AGR	0.300	0.300	0.300	0.300	0.300	0.300	0.300	0.300	0.200
EFISH	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.000
BUFF	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.000
CARP	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
CAT	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
DURM	0.001	0.001	0.001	0.000	0.000	0.000	0.000	0.000	0.000
WHITE	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
WALL	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
WBASS	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.001
WPERCH	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001
YPERCH	0.020	0.021	0.018	0.016	0.017	0.016	0.016	0.014	0.011
AIRT	0.500	0.300	0.400	0.400	0.400	0.400	0.400	0.500	0.700
RAILT	0.026	0.016	0.016	0.017	0.019	0.024	0.025	0.032	0.048
WTRT	0.073	0.048	0.054	0.056	0.056	0.064	0.068	0.083	0.100
TRKT	0.300	0.200	0.400	0.400	0.400	0.400	0.400	0.400	0.500
POW	1.300	1.600	1.300	1.200	1.100	0.900	0.800	0.700	0.500
WATR	0.200	0.300	0.200	0.200	0.200	0.200	0.200	0.200	0.100
SEAF	0.021	0.021	0.018	0.018	0.019	0.020	0.021	0.021	0.018
MISC	97.30	97.20	97.40	97.50	97.50	97.60	97.70	97.70	97.70
RFISH	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.001

When looking at the non-market fishing trips, we saw that while all households were consistent in which species they targeted more or less of, Household 2 experienced the smallest percent increases and largest percent decreases in their catch. Household 2 also experienced the smallest percent increase in income, but had the largest percent gain in welfare. Even though Household 2 experiences the lowest percent increase in income and recreational catch, the increase gives them larger returns in terms of utility. The additional income expands their consumption options without increasing their opportunity cost to the same level as the other households. Household 1 had the second highest percent gain in welfare. This suggests that while an Asian carp invasion may induce welfare improvements across all households, they may be particularly beneficial to households in the lowest income brackets.

4.5.3 Comparison of Outcomes

The integrated model used here allows us to see how a disturbance in the ecosystem impacts both the food web and the regional economy it is a part of, and better captures the interconnectedness of people and the ecosystem. In this section I compare the estimated outcomes of an Asian carp invasion in a system with feedbacks and without feedbacks.

To compare how impacts would differ if only estimated with an ecological system, such as EwE, I find the percent difference in species biomass when estimated with an integrated system and when estimated with only EwE. Figure 28 shows the percent difference in the species populations when an Asian carp invasion is modeled with and without economic feedbacks. Positive percent changes indicate that when an Asian carp invasion is modeled without feedbacks (solely in the EwE system) the species populations are under estimated. A negative percent change indicates that species populations are overestimated when evaluated without economic feedbacks. Some species had similar population sizes when modeled with and without feedbacks, as indicated with the percent change near zero.

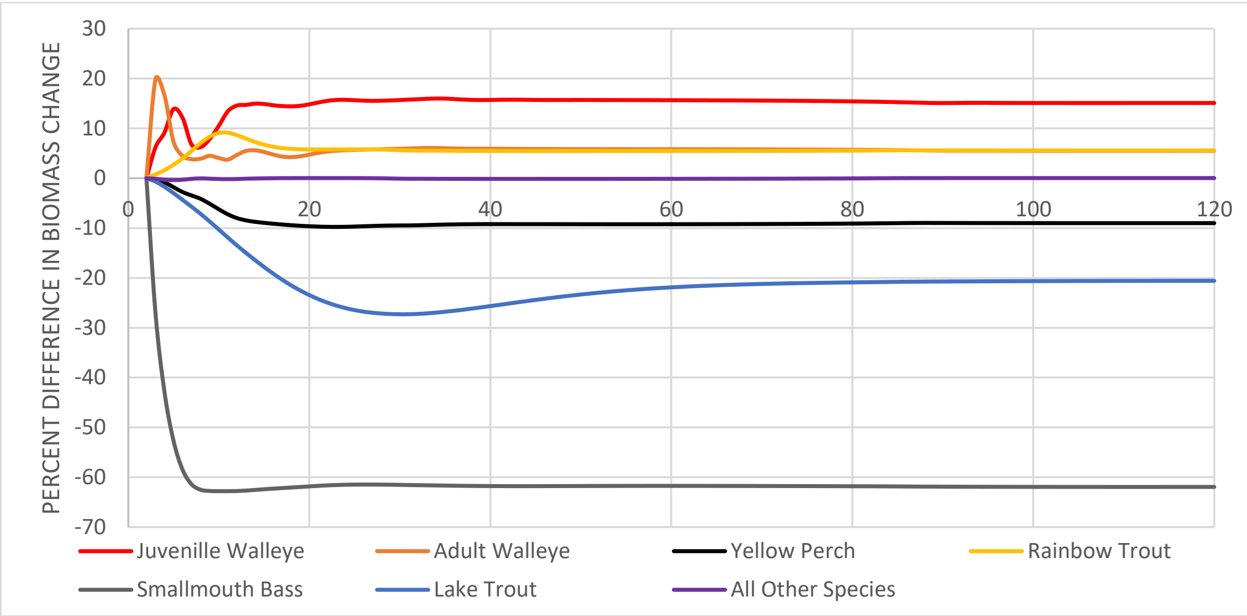

Figure 28: Percent Difference in Species Biomass with and without Economic Feedbacks
Responses in biomass of Lake Erie fish species under an Asian Carp invasion scenario, expressed as the percentage difference between scenarios including feedbacks between the ecological and economic systems relative to scenarios without feedbacks [100(biomass with feedbacks – biomass without feedbacks)/biomass without feedbacks]. Species with similar population estimates with and without feedbacks are included in the 'all other species' category.

The over and under-estimates of species response to an Asian carp invasion support the hypothesis that ecological systems alone do not fully capture the impacts of environmental shocks, and that human response needs to be accounted for when evaluating ecological disturbances. Adult walleye, juvenile walleye, and rainbow trout are species that EwE simulations would underestimate their population sizes. Yellow perch, smallmouth bass, and lake trout are species whose population sizes are overestimated with EwE alone.

When looking at both the commercial and recreational changes of these species following the invasion, we see a pattern. Yellow perch experienced increases in commercial harvest as the total allowable catch increased with stock increases, smallmouth bass and lake trout both saw increases in recreational catch. Walleye and rainbow trout on the other hand, both saw large decreases in recreational catch. This suggests that when estimating the impacts of the Asian carp invasion with only the ecological system, key components, human choice and substitutability, are being missed, and the impacts to species populations may be incorrectly estimated.

4.6 Concluding Remarks

In this paper, I presented a bioeconomic model that integrates the ecological and economic systems that are responsive to disturbances in the ecosystem. By using a model where the economic and ecological systems feed into each other, we are better able to estimate the impacts of an exogenous shock to either system. In this analysis, I chose to focus on a potential Asian carp invasion to Lake Erie, but the modeling framework could be used for a variety of scenarios. If impacts were only estimated with the ecological framework, in this case an EwE model, the populations of some species would be overestimated and some species would be underestimated. If the economic impacts were only estimated with the initial response from the EwE system the long-run ecological impacts would not be accounted for and the initial noisy response of the species may be given too much credence.

While an Asian carp invasion is generally thought to be a negative shock, here I showed that it would lead to welfare improvements across all representative households. The welfare improvements are driven by the positive ecological responses to Asian carp, namely the additional food source that young Asian carp would provide to current fish inhabitants. In both the economic and ecological components of the model there are numerous parameters in which these results are dependent on. Since an observable population of Asian carp do not currently exist in Lake Erie, parameters on what an Asian carp population would look like in Lake Erie were taken from experts' opinions. If Asian carp do colonize in Lake Erie, and more data becomes available, parameters can be updated to see if Asian do in fact cause increases in certain species populations. I do not perform a sensitivity analysis in the work presented here, but further analyzing the response of the results to different parameter values is warranted. Zhang et al., (2016) highlight the proportion of detritus in the Asian carps' diet and Asian carp larval consumption as parameters the results could be sensitive to, and would be parameters to analyze further for the results presented here. As for economic parameters, I believe the most important parameters to explore further would be the household's elasticity of substitution between species. The modeling framework assumes that households can substituted relatively easily between species, and that assumption has the capability of driving the results. Another area that could use further attention is the assumption made about in-region versus out of region fishing. The modeling framework assumes that recreational fishers will continue to fish in the region, more exploration of fishers moving out of the region is warranted.

While one objective of this paper was to understand the impacts of an Asian carp invasion on the Lake Erie regional economy and food web, it does not include the possible government expenditures for management and prevention of the invasive species. It also does not include Asian carp as possible commercial or recreational fisheries. These two considerations are certainly two interesting aspects, and require further analysis.

CHAPTER 5: CONCLUSION

Through the work presented in this dissertation, I addressed three important fishery issues: fishery ownership, fishery management, and fisheries' response to environmental changes as I sought to contribute to the fishery-economics' literature by answering two questions: what are the implications of a fishery moving from a regulated open access management setting to an individual transferrable quota system? and how do the estimated impacts of an Asian carp invasion differ between models with linkages be the economy and the food web and models when they are not? In addressing these questions, I developed a novel general equilibrium framework that integrates fish stock and the ecosystem services that fish stock provides to both firms and households in the region. The general equilibrium model I developed allows me to more fully capture the linkages between sectors, households, and producers in the economy. In evaluating the effects of both fishery management and an Asian carp invasion, I show that many impacts would be missed in a partial equilibrium setting, which suggests the importance of developing fishery models in the general equilibrium setting.

I first used the bioeconomic model with flows of ecosystem services to the fishing firm to evaluate the implications of a movement from a regulated open access management regime to an individual transferable quota system. I showed that under a regulated open access regime, the value of stock is incorrectly attributed to the effort market, and the ITQ system corrects this failure. The ITQ market captures the value of the fish stock, and fishing firms no longer attribute the value of the fish to effort and over employ the input. When simulating the transition from the regulated open access to an ITQ system, I also considered three different ownerships scenarios. I found that the change to an ITQ system results in welfare improvements, if the ITQ system induces improved catchability and efficiency in fishery effort choices. However, if the ITQ system does not result in an efficiency improvement, a welfare loss may be experienced as payments to factors are re-distributed to vessel owners. When the original capital and labor owners in the fishery were given the rights to the ITQ payments and rents, the largest welfare improvements were seen. Another key result of this analysis, is that welfare impacts were not the same size as the rent generated in the fishery. This provided evidence that using a partial equilibrium analysis to measure policy impacts in a fishery would miss the impacts felt by agents across the region, and potentially lead to a different decision on which policy instruments to use.

In the second application, I develop an integrated bioeconomic model to evaluate the joint responses of the Lake Erie regional economy and food web to an Asian carp invasion. This model included the role of ecosystem services from fish stock to both fishing firms and households. The model is unique in that there are feedbacks between the economy and food web. The bioeconomic model is used to evaluate a potential Lake Erie Asian carp invasion. There are two primary results from the analysis: 1) the Asian carp invasion leads to welfare improvements; and, 2) when invasion impacts are estimated with only the ecological food web model, without the consideration of changes in human choice, the impacts to some fish populations are overestimated, while others are underestimated, suggesting that ecological models alone will not correctly capture the impacts of an environment change. Economists and ecologists need to work together to fully understand responses to changes in either system.

Fisheries management includes addressing fish populations, habitats, and human behavior, and to successfully manage a fishery, all components must be considered. Traditionally, fisheries management has based policy decisions on ecological population estimates and current fishing levels. However, humans adapt to changing conditions and those adaptations need to be accounted for in order to achieve successful management. The bioeconomic model presented in this work integrates the economy and ecosystem, thus creating a tool that fully captures all aspects of fisheries management. Incorporating the components of successful fishery management into a general equilibrium framework allows for the linkages between producers, households, input sectors, and the ecosystem to be more fully captured. Both applications presented in this dissertation showed the importance of these linkages, when any aspect is missing, regardless of if it’s the linkage between sectors or between the economy and ecosystem, important impacts are missed. Whether the modeling tool is used by economists to measure impacts, or by fishery managers to make policy choices, all linkages are important.

Throughout the modeling and evaluation of the management and invasive species issues presented here, I have strived to represent the systems in the most accurate way possible, however limitations do exist. The results depend on the parameter values, both economic and ecological, and I would stress that the results presented here should be taken as direction of movements rather than putting too much stock into their absolute values. As more data becomes available, parameter estimates can be validated and refined. The data used in the simulations is specific to the Lake Erie region, and while I am confident in the modeling framework, it would be interesting to see how the results differed with a different study area. For example, when comparing ownerships scenarios, the welfare impacts of changes in regulation were driven by the amount of labor and capital provided from sources outside of the region. It would be interesting to study an area with a different supply structure of factors. Further work is also warranted in evaluating the sensitivity of the results to parameter values, the households' elasticity of substation between recreational species, the diet composition of Asian carp, and the vulnerability of Asian carp to predators would be an initial set of parameters to conduct a sensitivity analysis on. The data used to estimate to estimate the Cobb-Douglas parameters for white perch, white bass, and lake whitefish, is also lacking, and to make the results more accurate, obtaining better data, and better estimating the stock size of white perch, white bass, and lake whitefish for production function estimation purposes, would be needed. Another limitation of my model is the binding TAC. Allowing the TAC to be non-binding could provide additional insights on the welfare implications in both the management and invasive species application.

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APPENDIX

Table 16: Species Groups in EwE Food Web

Abbreviations and names of species groups in Lake Erie food web.

Group	Species	Group	Species
DCCM	Double-crested cormorant	SUK	White Sucker, Quillback, Bigmouth Buffalo
MERG	Red-breasted merganser	EMS	Emerald Shiner, Spottail Shiner
WAE_L	Walleye, Larval	PAN	Panfish
WAE_Y	Walleye, age 0	Other	Silver Chub, Trout-perch, Common Log Perch
WAE_1	Walleye, juvenile (age 1-2)	DREI	Zebra mussel and quagga mussel
WAE_2	Walleye, adult (age 3+)	AMPH	Amphipoda and Isopoda
YEP_L	Yellow Perch, larval	CHIR	Chironomidae
YEP_Y	Yellow Perch, age 0	OLIG	Oligochaeta
YEP_1	Yellow Perch, juvenile (age 1)	MOLL	Emphemeropter
YEP_2	Yellow Perch, adult (age 2+)	EPHE	Gastropoda, Sphaeriidae, Bivalvia
GIZ	Gizzard Shad	OtherB	Other Benthos
RBT_Y	Rainbow Trout, stocked yearlings	CLAD	Herbivorous cladocerans
RBT_A	Rainbow Trout, adult	COPE	Calanoida and Cyclopoida
LWF	Lake Whitefish	ROTI	Rotifera
BBT	Burbot	PRED	Spiny water flea
WHP	White Perch	PROT	Ciliates and heterotrophic flagellates
WHB	White Bass	BACT	Bacteria
SMB	Smallmouth Bass	PIO	Picoplankton
FWD	Freshwater Drum	EDIB	Edible algae
ALW	Alewife	CYAN	Cyanophyta
LKT_Y	Lake Trout, stocked yearlings	DETR	Suspended and sediment detritus
LKT_A	Lake Trout, adult	CMP	Common Carp
RBS	Rainbow Smelt	RGB	Round Goby

		1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37
1	AGR	90.7																	0.05	1879	0.06				15.7	11.4	29.8	31.5	47.9	78.7	64.1	74.6	50.9	0.01	26.6	39.1	222	656
2	FISH																		0.01	3.72					0.07	0.05	0.13	0.14	0.21	0.33	0.27	0.30	0.19		0.12		0.55	0.49
3	BUFF																		0.01	2.73					0.05	0.04	0.10	0.10	0.15	0.24	0.20	0.22	0.14		0.09		0.40	0.36
4	CARP																		0.01	1.53					0.03	0.02	0.05	0.06	0.09	0.14	0.11	0.12	0.08		0.05		0.23	0.20
5	CAT																		0.01	3.16					0.06	0.05	0.11	0.12	0.18	0.28	0.23	0.25	0.16		0.10		0.47	0.41
6	DURM																		0.01	1.70					0.03	0.02	0.06	0.06	0.10	0.15	0.12	0.14	0.09		0.05		0.25	0.22
7	WHITE																		0.00	1.34					0.03	0.02	0.05	0.05	0.08	0.12	0.10	0.11	0.07		0.04		0.20	0.18
8	WALL																			0.01									0.00	0.00	0.00	0.00	0.00				0.00	0.00
9	WBASS																		0.02	5.91					0.11	0.08	0.21	0.22	0.33	0.53	0.42	0.47	0.30		0.19		0.87	0.77
10	WPERCH																		0.01	3.17					0.06	0.05	0.11	0.12	0.18	0.28	0.23	0.25	0.16		0.10		0.47	0.41
11	YPERCH																		0.15	41.8					0.81	0.60	1.49	1.53	2.33	3.75	3.00	3.36	2.10		1.34		6.16	5.45
12	AIRT	0.86											0.01	0.13	1.03	10.8	5.42	0.01	0.00	457	0.02				23.5	11.7	38.1	43.0	62.5	114	97.7	144	146	58.2	33.5	10.6	26.1	9.46
13	RAILT	4.17										0.01	0.07	0.93	0.20	17.6	23.7	0.05	0.00	557	0.02				1.69	0.77	2.27	2.64	4.27	8.49	7.44	11.2	12.3	7.15	42.9	9.97	42.6	55.7
14	WTRT	3.18										0.01	0.06	0.14	0.17	0.59	1.28	0.03		160	0.01				4.27	2.07	6.61	7.51	11.2	20.9	18.0	26.7	27.8	2.78	7.85	3.37	194	95.7
15	TRKT	10.2										0.02	0.37	1.97	2.48	24.0	2.85	0.04	0.02	1485	0.05				15.5	8.19	36.5	42.3	60.6	106	92.3	120	99.8	20.9	49.5	89.2	138	105
16	POW	4.70											0.15	0.09	4.15	2.28	1132	0.31	0.00	1493	0.05				56.4	47.3	112	115	164	233	168	177	107	5.92	58.0		7.81	4091
17	WATR	5.53											0.01	0.05	3.31	0.23	19.8	0.16		144					10.2	10.1	25.4	25.8	39.8	59.8	45.7	50.6	31.3	6.61	82.0		1.00	6.38
18	SEAF																		0.03	45.0					1.37	0.96	2.45	2.68	4.28	7.48	6.38	7.65	5.14	0.14	15.6		3.36	6.50
19	MISC	474	0.81	0.59	0.33	0.69	0.37	0.29		1.29	0.69	9.09	84.4	187	368	923	1451	6.50	0.13	90346	3.03				2847	2000	5839	6418	9300	15390	12596	15582	11862	4252	18950	15122	12728	70753
20	RFISH	0.02												0.01	0.01	0.03	0.05			3.03					0.10	0.07	0.20	0.22	0.31	0.52	0.42	0.52	0.40	0.14	0.64	0.51	0.43	2.37
21	LAB	127	0.03	0.02	0.01	0.02	0.01	0.01		0.05	0.02	0.32	31.1	134	48.2	505	793	5.91	0.12	53058	1.78																	
22	CAP	433	0.27	0.20	0.11	0.23	0.13	0.10		0.44	0.23	3.08	16.5	48.5	51.9	193	2345	12.1	0.07	41171	1.38																	
23	INDT		0.18	0.13	0.07	0.15	0.08	0.07		0.29	0.15	2.04	14.6		5.97	25.2	701	3.08	0.02	7669	0.26																	
24	HHD1																					160	54.5											426	170	2090		15.6
25	HHD2																					192	43.4											827	319	652		18.7
26	HHD3																					1068	241											2028	779	1706		104
27	HHD4																					1861	381											1939	738	1513		181
28	HHD5																					4030	787											2481	929	1338		393
29	HHD6																					8993	1638											3236	1178	1525		877
30	HHD7																					8636	1703											2249	794	790		842
31	HHD8																					11731	2685											2286	778	1067		1144
32	HHD9																					11444	4975											1969	653	1811		1116
33	FGOV	44.3																0.56		19.9		6461	378	983	-85.3	-53.3	-228	-170	15.3	800	1230	2029	5297	2418		9290		
34	SGOV	2.13	0.12	0.09	0.05	0.10	0.05	0.04		0.19	0.10	1.33						13.5		3137	0.11	131	166	7439	24.3	10.9	59.0	92.4	243	622	682	987	1544	4163	9344	6288		
35	INV	55.7	0.01			0.01				0.01	0.01	0.07								1246	0.04		29694									475	2782	53.7		4595	5580	3458
36	FT	240	4.49	3.29	1.85	3.81	2.05	1.62	0.02	7.12	3.82	50.4	220	3.53		29.9	39.2		20.3	17196	0.58		-76.45		20.5	16.5	49.3	56.4	87.8	152	131	168	140	79.2	83.4	216		
37	DT	1822	0.67	0.49	0.27	0.57	0.31	0.24		1.06	0.57	7.49	927	437	108	780	1466	526	88.0	77366	2.59		1607		-20.5	-16.5	-49.3	-56.4	-87.8	-152	-131	-168	-140	-79.2	-83.4	-216		

Figure 29: Lake Erie Coastal Social Accounting Matrix
Millions of dollars of activities.