THE CONSEQUENCES OF MISREPRESENTING FEEDBACKS IN COUPLED HUMAN AND ENVIRONMENTAL MODELS

Jenny L. Apriesnig[a,*], Travis W. Warziniack[b], David C. Finnoff[c], Hongyan Zhang[d], Katherine D. Lee[e], Doran M. Mason[f] and Edward S. Rutherford[f]

Author Affiliations:
[a] College of Business, Michigan Technological University, 1400 Townsend Drive, Houghton, MI 49931, USA, jlapries@mtu.edu
[b] Rocky Mountain Research Station, US Forest Service, 240 West Prospect Road, Fort Collins, CO 80526, USA, travis.w.warziniack@usda.gov
[c] Department of Economics, College of Business, University of Wyoming, 1000 E. University Avenue, Laramie, WY, 82071, USA, finnoff@uwyo.edu
[d] Eureka Aquatic Research LLC, 6245 Wilson Road, Ann Arbor, MI 48108, USA, hongyanzhang2012@gmail.com
[e] Agricultural Economics and Rural Sociology, University of Idaho, 875 Perimeter Drive MS 2334, Moscow, ID 83843, USA, katherinelee@uidaho.edu
[f] National Oceanic and Atmospheric Administration Great Lakes Environmental Research Laboratory, 4840 S State Road, Ann Arbor, MI 48108, USA, Ed.Rutherford@noaa.gov, doran.mason@noaa.gov

* Corresponding Author

Abstract

Feedbacks between the ecosystem and the economy are important to consider when measuring impacts from a disturbance but are often omitted from general equilibrium bioeconomic models. These models usually focus on how humans respond to ecological change, but do not consider that in adapting to changed conditions, humans can further affect the ecosystem. We present a framework that couples a regional computable general equilibrium model with an Ecopath with Ecosim food web model with bidirectional feedbacks between the two systems. Our bioeconomic model uniquely represents a comprehensive mapping of the entire regional economy, including recreational and commercial fishing, harvest quotas, and fish biomass in the economic system. We simulate the bioeconomic impacts of a potential Asian carp invasion of Lake Erie's food web and regional economy with and without bidirectional feedbacks between the economy and the ecosystem. When feedbacks are omitted there are large ecological variations in the projected biomass levels of many target species, with differences in biomass of up to 80 percentage points. Results demonstrate the need to reflect bidirectional feedbacks between the economy and the ecosystem; omission of these feedbacks in this case may appear to be economically trivial yet have large ecological consequences.

Keywords: bidirectional feedbacks, Asian carp, computable general equilibrium, Ecopath with Ecosim, coupled systems.

1. Introduction

Unanticipated changes to ecological systems, such as the introduction of costly invasive species, are a major threat to resource-dependent economies and economic sectors. Thus, understanding the potential impact of natural or human-induced environmental changes are necessary to guide management strategies and recommendations for policymakers. The connection between the state of an ecosystem and human decision-making is critical in managing natural systems and quantifying the economic impacts of environmental change. A body of existing literature has identified that accounting for both economic and ecological systems is necessary to measure the economic impacts of environmental change (Bossenbroek et al., 2009; Ferraro et al., 2019; Finnoff and Tschirhart, 2008; Finnoff et al., 2009). There is also increasing realization that when measuring the impacts of an environmental shock, such as the introduction of an invasive species, the linkages between human decision-making and environmental response must be constantly updated (Finnoff et al., 2005; McDermott et al., 2013). For fishery related analysis, Armstrong (2007) further argues that economists need to not only incorporate multiple species into their fishery models, they must also collaborate with ecologists to develop models that are complex enough in both the economic and ecological components to better capture the reality of the fishery.

The ecological-economic interconnections are bidirectional feedback loops; actions by humans influence the future state of the environment, which then changes the circumstances humans face in the next decision period, and thus affects their behavior. For example, fishing pressure in a given year impacts the biomass of a species, which will affect fishing behavior in the next year and subsequent species response. Bidirectional linkages are present in much of the economic literature that uses partial equilibrium models (models that focus on one or few industries). For example, Fenichel & Abbot (2014), Hutniczak (2015), and Yun et al. (2017) heavily exploit the modeling of the feedbacks between aquatic and economic systems and recent work by Brockman et al. (2021) and Lee et al. (2021) make strides in modeling bidirectional feedbacks between an economic model and the ecosystem, but they omit the linkages between fishing sectors and the rest of the economy. Bidirectional feedback loops are less present in works that use general equilibrium economic models (models with many industries and market linkages).

These general equilibrium economic models developed with natural resource inputs often only include a unidirectional feedback (Gilliland et al., 2019; Manning et al., 2016). Often, the unidirectional feedback is captured through using the output of an ecological model as an input to the economic model (Avila-Foucat et al., 2009; Byron et al., 2015; Jin et al., 2012; Warziniack et al., 2017), but this ignores impacts of decision making in the economic model on the ecological system. There are some works that include the multi-directional feedback loop between the ecosystem and the economy in the general equilibrium framework, such as Finnoff & Tschirhart (2008, 2003) and Eichner & Tschirhart (2007), who developed a general equilibrium ecosystem model (GEEM) which models individual firms, households, and organisms together. Finnoff & Tschirhart (2008) use the GEEM to evaluate how the Alaskan economy and ecosystem respond to policy changes with bidirectional feedback loops, yet their model is based on state level aggregates that may not allow a clear focus on the areas dependent on the ecosystem services considered, they omit the key recreational fishing industry that represents a significant bidirectional feedback loop between ecological and economic systems, and the GEEM model has not been widely accepted by management agencies. We couple a well-known and widely accepted ecological model developed by members of our team (Zhang et al., 2016) with a regionally-specific computable general equilibrium (CGE) model that includes two bidirectional feedback loops. We push the research further by quantifying and addressing the remaining question: what are the consequences of misrepresenting bidirectional feedbacks as unidirectional in modeling coupled human and environmental systems? Here, we develop a coupled bioeconomic model that integrates an economic computable general equilibrium model and an ecological food web, Ecopath with Ecosim (EwE) (Christensen and Walters, 2004; Plagányi and Butterworth, 2004), into a single framework. In an iterative simulation, the EwE model updates the environmental "state" that the economic model is faced with, then decisions made by the economic model are used to update the ecological model. We quantify the ecological and welfare impacts of an environmental shock when accounting for bidirectional feedback loops. The economic model is developed for a regional economy, and includes recreational and commercial fishing activities, which allows us to measure the direct impacts of the shock on resource-dependent sectors and recreational activities, as well as the impacts to sectors indirectly dependent or connected to the resource. Lastly, we quantify the difference in biomass levels of target species following a projected introduction of two invasive species, bighead carp (Hypophthalmichthys nobilis) and silver carp (H. molitrix), commonly referred to as 'Asian carp,' on the Lake Erie ecosystem and regional economy when bidirectional feedbacks between the economic and ecological systems are considered or ignored.

2. Material and Methods

To jointly estimate the ecological and economic impacts from an environmental disturbance, an economic model is combined with the ecological food web model Ecopath with Ecosim (EwE). We begin by describing the economic model then turn to a description of EwE and how the models are integrated.

2.1 Economic Model

The economy is represented by a computable general equilibrium (CGE) model which includes production, consumption, government, and trade. The CGE model includes a comprehensive representation of the economic activities of both households and producers in the region in a way that allows their simulated behavior to adjust in response to changing market and ecological conditions. Each commercial sector $i \in I$ is modeled with a representative firm. Each of nine households $h \in H$, differentiated by income, is modeled with a representative household. 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. The economic model is defined for a single time period and is re-solved each year after fish biomass levels are input from EwE.

2.1.1 Producer Behavior

There are two types of firms, firms in commercial fishing sectors whose production depends on a biological component such as 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 and Tarr, 1992). In fishing sectors, fish stock is included as an input to production, so that effort is combined with a regional fish stock to produce the economic good, harvest. Some commercial fishing sectors face harvest restrictions. Regardless of sector, each firm behaves the same in the lower two nests (Figure 1). For firms without a biological component in production, production equals the output of Nest 2. We assume constant returns to scale in each production nest, which allows us to solve for a firm's behavior one nest at a time. We begin by presenting the decision-making processes in Nest 1 and Nest 2 that apply to all firms, then proceed to the behavior in Nest 3 that is exclusive to fishing firms.

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, 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, and δ_{Li} and δ_{Ki} are the firm's share parameters of each factor, which sum to one due to the constant returns to scale assumption. The parameter, ρ_i, is related to the firm's elasticity of substitution (σ_i) in the value-added function such that $\rho_i = (\sigma_i - 1)/\sigma_i$. 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:

$$L_i = (\frac{Z_i}{\varepsilon_i})(\frac{\varepsilon_i \delta_{Li} P_{Zi}}{P_L})^{\sigma_i} \qquad (1)$$

$$K_i = (\frac{Z_i}{\varepsilon_i})(\frac{\varepsilon_i \delta_{Ki} P_{Zi}}{P_K})^{\sigma_i} \qquad (2)$$

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:

$$P_{Zi} = (\frac{1}{\varepsilon_{Zi}})[\delta_{Li}{}^{\sigma_i} P_L{}^{(1-\sigma_i)} + \delta_{Ki}{}^{\sigma_i} P_K{}^{(1-\sigma_i)}]^{\frac{1}{1-\sigma_i}} \quad (3)$$

Equation 3 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 ($V_{i,j}$) from sector $j \in I$, 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:

$$Z_i = a_{Zi} E_i \quad (4)$$

$$V_{i,j} = a_{Vi,j} E_i \quad (5)$$

We 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:

$$P_{E_i} = a_{Zi} P_{Z_i} + \sum_j a_{Vi,j} P_j + T_i \quad (6)$$

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 *f* and firms from other sectors *o* diverge, such that, $o \subseteq I$, $f \subseteq I$, and $o \cup f = I$. 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,

$$P_{E_o} = P_o \tag{7}$$

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 1 - 7 . The fishing firm's harvest and effort levels depend on the fishing regulation they face and are described in the next section.

2.1.1.1 Fishing Sectors

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

$$H_f = q_f E_f^{\alpha_f} S_f^{\beta_f} \tag{8}$$

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.

Fishing sectors are categorized into two types, those that are limited by a total allowable catch (TAC) and those that are not. Fishing firms in sectors regulated by harvest restrictions are referred to as the regulated open access (ROA) fishing sectors. The representative firm from sector $r \in F$, is constrained by a TAC, so $H_r \leq TAC_r(S_r)$, where the total allowable catch is a function of stock. Firms do not pay for stock, so $P_{S_r} = 0$, and we assume the TAC constraint binds. Substituting $TAC_r(S_r)$ into Equation 8 and solving for effort gives the effort level for the regulated fishing sectors (Equation 9).

$$E_r = \left(\frac{TAC_r}{q_r S_r^{\beta_r}}\right)^{1/\alpha_r} \tag{9}$$

Fishing sectors that do not face a harvest restriction 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. Again, firms do not pay for stock. Solving Equation 8 for effort provides the effort level for the non-regulated fishing sectors (Equation 9').

$$E_n = \left(\frac{H_n}{q_n {S_n}^{\beta_n}}\right)^{1/\alpha_n} \qquad (9')$$

Free entry into fishing sectors implies zero-profits for both regulated and non-regulated fishing firms. 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 and Wainwright, 2004). For the fishing sectors, this implies:

$$(\alpha_f + \beta_f)\, H_f = E_f \frac{\partial H_f}{\partial E_f} + S_f \frac{\partial H_f}{\partial S_f} \qquad (10)$$

$\frac{\partial H_f}{\partial E_f}$ 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:

$$(\alpha_f + \beta_f)\, H_f P_f = E_f \frac{\partial H_f}{\partial E_f} P_f + S_f \frac{\partial H_f}{\partial S_f} P_f \qquad (11)$$

When written 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 and Wainwright, 2004).

The market determines P_f and because stock is a free input, effort is paid more than its marginal product. Manning et al. (Manning et al., 2016) show that the fishers incorrectly attribute the value of marginal product of the stock to factors of production and pay 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 Equation 11:

$$(\alpha_f + \beta_f)\, H_f P_f = E_f \left[\frac{\partial H_f}{\partial E_f} P_f + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f\right] \qquad (11')$$

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

$$P_{Ef} = \frac{\partial H_f}{\partial E_f} P_f + \frac{S_f}{E_f} \frac{\partial H_f}{\partial S_f} P_f \qquad (12)$$

Each species-specific representative fishing 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 12 shows that effort receives its value of marginal product plus the value of stock.

2.1.2 Household Behavior

Recreational fishing can be a large part of the relationship between humans and the ecosystem; for this reason, the household and its behavior are explicitly defined in our CGE model. The household consumes both market and nonmarket goods. Goods acquired through markets are referred to as consumption goods, non-market goods are attained through recreation trips. Warziniack et al. (2017) describes the household's consumption of both market and non-market goods in a general equilibrium setting; here we apply that model specifically to recreational fishing.

A representative 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 generated by targeting each species f is produced by a combination of individual household efforts, $E_{h_{f,h}}$, and the levels of fish stock, S_f. Figure 2 shows the nesting structure for household consumption. A brief description of step 1 follows, see Warziniack et al. (2017) for the full model description.

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. As described in Warziniack et al. (2017), the households are endowed with varying amounts of capital labor and derive the income from factor payments. 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:

$$\frac{\partial U_h}{\partial R_h} = \lambda_{YD_h} P_R \tag{13}$$

$$\frac{\partial U_h}{\partial C_h} = \lambda_{YD_h} P_C \tag{14}$$

$$Y_{D_h} = P_R R_h - \ P_C C_h \tag{15}$$

These conditions require the household demand a mix of consumption goods and recreation so that marginal rate of substitution equals the price ratio of the two types of goods, while maintaining the budget constraint. Equations 13-15 identify R_h, C_h, and λ_{YD_h}. After spending $P_C C_h$ on consumption goods, the consumer has $Y_{D_h} - P_C C_h$ to spend on recreation. Similar steps are taken in the remaining nests of the household decision problem to allocate recreation spending across species-specific fishing trips.

2.1.3 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 foreign trade is the transfer of goods from outside the United States. 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 (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 ($Q_{M_{t,i}}$).

$$Q_i = \varepsilon_{Q_i}\left(\delta_{QD_i} {Q_{D_i}}^{\rho_{Q_i}} + \sum_t \delta_{QM_{t,i}} {Q_{M_{t,i}}}^{\rho_{Q_i}}\right)^{1/\rho_{Q_i}} \tag{16}$$

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.

$$Q_{m_{t,i}} = \left(\frac{Q_i}{\varepsilon_{Q_i}}\right)\left[\frac{\delta_{QM_{t,i}}\left({\delta_{QD_i}}^{\sigma_{Q_i}} P_i^{1-\sigma_{Q_i}} + \sum_t {\delta_{QM_{t,i}}}^{\sigma_{Q_i}} {P_{M_i}}^{1-\sigma_{Q_i}}\right)}{P_{M_i}}\right]^{\sigma_{Q_i}} \tag{17}$$

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.

$$X_i = \varepsilon_{X_i}\left(\delta_{XD_i} {X_{D_i}}^{\rho_{X_i}} + \sum_t \delta_{XE_{t,i}} {X_{E_{t,i}}}^{\rho_{X_i}}\right)^{1/\rho_{X_i}} \tag{18}$$

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.

$$X_{E_{t,i}} = \left(\frac{X_i}{\varepsilon_{X_i}}\right)\left[\frac{\delta_{XE_{t,i}}\left({\delta_{XD_i}}^{\sigma_{X_i}} P_i^{1-\sigma_{X_i}} + \sum_t {\delta_{XE_{t,i}}}^{\sigma_{X_i}} {P_{M_i}}^{1-\sigma_{X_i}}\right)}{P_{M_i}}\right]^{\sigma_{X_i}} \tag{19}$$

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 20).

$$Q_{D_i} = H_i + \sum_g Q_{G_{i,g}} + Q_{V_i} \tag{20}$$

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

$$Q_{G_{i,g}} = a_{QG_{i,g}} Q_i \tag{21}$$

$$Q_{V_i} = a_{QV_i} Q_i \tag{22}$$

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}.

$$Q_{D_i} = H_i + \sum_g Q_{G_{i,g}} + Q_{V_i} \tag{23}$$

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

$$X_{G_{i,g}} = a_{XG_{i,g}} X_i \tag{24}$$

$$X_{V_i} = a_{XV_i} X_i \tag{25}$$

Equations 16 - 25 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.

2.1.4 Model Closure

The model is closed with Equations 26, , and 28 and by defining the 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, $\overline{L}$ and $\overline{K}$, 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.

$$\textstyle\sum_i L_i = \overline{L} \tag{26}$$

$$\textstyle\sum_i K_i = \overline{K} \tag{27}$$

$$Q_i = X_i \tag{28}$$

2.2 Food-web Model

To represent the food web, we use an existing EwE model for Lake Erie (Zhang et al., 2016). EwE is an established model 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 and Walters, 2004; Cox and Kitchell, 2004; Kao et al., 2014; Langseth et al., 2012; Plagányi and Butterworth, 2004). EwE consists of two parts, Ecopath, a modeling system used to represent the flows between elements in an ecosystem (Christensen et al., 2005), and Ecosim, which allows for the simulation of potential disturbances to the systems through its dynamic modeling capabilities (Christensen and 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 either leaves the system or accumulates in the system (Christensen et al., 2005). In Ecopath, mass balances on the species level. The developed Lake Erie food web model consists of 51 model groups including birds, fish, benthos, zooplankton, phytoplankton, protozoa, bacteria, and detritus. The Ecopath model was mass balanced using data from 1999-2001, and the Ecosim is calibrated using observed time series of biomass for 14 trophic groups form 1999-2010 (see Zhang et al. (2016) for full details).

2.3 Model Dynamics

We consider two modeling frameworks: one with bidirectional feedbacks between the economic and ecological systems (hereafter named integrated economic and ecological model) and one without feedbacks. When feedbacks are included between the two systems, they are best described as recursive, bidirectional and dynamic. In the first stage, EwE simulates changes in species' biomass for a given disturbance scenario to the food web. In the second stage, the new biomass levels ($\boldsymbol{S_f}$) are fed into the economic model, and the agents make their optimal choices. In the third stage, the updated fishing mortality rates from commercial and recreational fishing from the economic model are reported to the

EwE model[1], which then simulates the trophic response to species' biomass. Starting over, the new biomass levels are reported back to economic model. We assume that the three stages occur within a year.

2.4 Study Area

While the model can be applied to a variety of regions and scenarios; we focus on Lake Erie and a potential Asian carp invasion. Lake Erie, the most productive lake among the five Laurentian Great Lakes, supports both recreational and commercial fishing sectors (Hubert and Quist, 2010; Zhang et al., 2016), and has been exposed to multiple anthropogenic stressors including contaminants, eutrophication, and climate change, and is also vulnerable to invasion by non-indigenous species through multiple pathways (Zhang et al., 2016). Recently, 'Asian carp,' have gained attention as a possible large-scale invader of the Great Lakes, and there is concern that they could cause ecological and economic damage on a scale similar to the invasive sea lamprey (Petromyzon marinus) and Dreissena mussels (zebra mussel Dreissena polymorpha and quagga mussel D. bugensis). The carp are filter feeders, can consume 20-40 percent of their body weight daily on species at the base of the food web, and are known to have high growth and fecundity rates (Great Lakes Commission and St. Lawrence Cities Initiative, 2012; Kolar et al., 2007; Zhang et al., 2016). These characteristics of Asian carp create concern that the species will out-compete forage fish and planktivorous larval fish, and consequently decrease commercial and recreational fish species (Wittmann et al., 2015; Zhang et al., 2016). Silver carp also impede recreational boating by leaping out of the water, creating a nuisance and sometimes causing injury to humans (Great Lakes Commission and St. Lawrence Cities Initiative, 2012). Asian carp have invaded the entire Mississippi River Basin, and two waterways have been identified as likely routes for the carp to pass into the Great Lakes – the Chicago Sanitary and Ship Canal (CSSC) that connects the Mississippi water basin and Lake

---

[1] The simulation time step in Ecosim is a month, but Ecosim reads in an annual fishing mortality rate at the beginning of a year and applies monthly fishing rates so that annual fishing rate holds.

Michigan and the Wabash River that occasionally connects to Lake Erie during high water periods (Kocovsky et al., 2012).

Other invasive species have been costly to the region; sea lamprey, a species that began to take hold in the Great Lakes in the 1940s and 1950s, is blamed for the collapse of native populations of lake trout (Salvelinus namaycush) and lake whitefish (Coregonus clupeaformis) and costs the region approximately $20 million annually to control (Great Lakes Commission and St. Lawrence Cities Initiative, 2012; Koonce et al., 1996). Dreissena mussels have changed nutrient dynamics, decreased lake pelagic productivity and are known for clogging intakes for power plants and drinking water treatment plants (Connelly et al., 2007); are estimated to annually cost the region $300-$500 million in damages and control (Great Lakes Commission and St. Lawrence Cities Initiative, 2012). The economic damages of Asian carp have not been studied with the same level of detail, but proposals have been made to re-structure the Chicago Area Waterway System (CAWS) at costs ranging from $7 billion to $18 billion in 2014 dollars; in 2010 a chain-link fence was installed at a cost of $200,000 to prevent passage of carp between the Wabash and Maumee Rivers during flood events (Hebert, 2010; U.S. Army Corps of Engineers, 2014).

Lake Erie is bordered by Michigan, Ohio, Pennsylvania, New York and the province of Ontario. For this analysis, we focus on a regional economy that includes the U.S. counties bordering Lake Erie, excluding counties with large cities such as Detroit, Toledo, and Cleveland. The Lake Erie Coastal Economy includes six counties in Ohio (Ashtabula, Erie, Lake, Lorain, Ottawa, Sandusky), one county in Michigan (Monroe), one county in Pennsylvania (Erie), and two counties in New York (Chautauqua, Erie), as shown in Figure 3. Counties with large cities are likely to have economic activity that overshadows lake related activities, so by omitting counties that include large cities, we focus on areas most directly affected by changing fishing conditions. Two of the Lake Erie fisheries, yellow perch and walleye, are regulated with a total allowable catch; a description of how these regulations are represented in the model can be found Appendix B.

2.5 Data and Parameterization

We use multiple data sources to define the initial values and parameters of the CGE model. For the social accounting matrix (SAM), an integral data component of CGE models, we use 2013 IMPLAN county-level data (IMPLAN Group, 2013). Though deficient in some areas, it is more comprehensive than any other data 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. A full description of the parameterization of the economic model and data used, including how the SAM was adjusted to include natural resources can be found in Appendix A. A description of the parameterization of the Lake Erie EwE can be found in Zhang et al. (2016).

2.6 Simulations

The primary objective of this analysis is to quantify the difference in estimated impacts of a shock when modeled with and without feedbacks between the ecological and economic systems. An interesting, yet secondary objective is to understand the ecological and economic impacts of an Asian carp (AC) invasion in Lake Erie. To address these objectives, we perform four simulations (shown in Table 1). The first simulation (S1) is an integrated economic and ecological model with an AC invasion. The second simulation (S2) is an integrated economic and ecological model without an AC invasion (S2). Simulations three (S3) and four (S4) only engage the ecological model; S3 includes an AC invasion while S4 does not.

In S1 and S2 the economic model is programmed in the General Algebraic Modeling System (GAMS). In S3 and S4 the economic model is not engaged, so the ecological impacts are purely driven by the responses modeled in EwE. In all four simulations harvest occurs, but in Simulations 1 and 2 harvest is driven by the economic model, while in Simulations 3 and 4, harvest is determined by the fishing mortality rate used by the EwE. In our application of the EwE model, we used annual estimates of nutrient loads and fishing mortality rates to drive biomass dynamics in the food web, and derived fishing mortality rates for each harvested species based on observed harvest. In all scenario simulations, the entire time horizon evaluated is 152 years. In the with-AC simulations, S1 and S3, the biomass of AC is kept at a low level for the first 32 years. Once the AC population is allowed to grow, we simulate an additional 120 years so that the system can move past the initial variability caused by the species introduction and so the food web can reach a new equilibrium among the species groups. Throughout the simulations we assume that no additional shocks occur to either the ecological or economic systems. Comparing S1 and S3 allows us to evaluate how estimated impacts differ when modeled with and without feedbacks between the economic and ecological systems. Comparing S1 and S2 allows us to quantify the impacts of an AC invasion. We use S4 as an internal validity check of the programming that connects the CGE and EwE; we confirm that running S4 in our linked model (but without feedbacks) matches the outcome of simulating the scenario in EwE alone.

3. Results & Discussion

Our results are addressed in two subsections; in the first subsection we discuss the impacts of an AC invasion. In the second subsection we discuss how the estimated impacts differ when modeled with and without feedbacks.

3.1 Effects of Asian carp in an Integrated Framework

3.1.1 Ecological Responses

We begin with a comparison of fish biomass levels under Simulations 1 and 2, which both use integrated economic and ecological models, but with and without an AC introduction. In the long-run, an AC introduction reduces the biomass levels of some species and increases the biomass of others. The percent difference in the biomass of commercially and recreationally targeted species between S1 and S2 can be seen in Figure 4. The biomass of white bass (Morone chrysops), walleye (Sander vitreus), rainbow trout (Oncorhynchus mykiss), and lake trout (referred to as Group A below) are smaller in S1 (with an AC invasion) compared to those in S2 (without an AC invasion), while the biomass of channel catfish (Ictalurus punctatus) and smallmouth bass (Micropterus dolomieu), (referred to as Group B below) are larger in S1. The remaining target species, white perch (Morone americana), common carp (Cyprinus carpio), yellow perch (Perca flavescens), suckers (Catastomidae family), lake whitefish, and freshwater drum (Aplodinotus grunniens) had similar changes in biomass in S1 and S2 (difference of less than 5%).

The AC invasion increased food competition on zooplankters, so zooplankters (cladocerans, copepods) and planktivorous fish (alewife (Alosa pseudoharengus), rainbow smelt (Osmerus mordax) and shiners (Cyprinidae family – primarily Emerald shiner Notropis atherinoides)) had lower biomass levels following the invasion (Figure D.1). Consequently, fishes (Group A) that prey on planktivorous fish had smaller biomass levels when Asian carp were present (Figure 4). Figure D.2 shows the target species consumption with and without AC (Tables 4 and 5 in Zhang et al. 2016 show the specific consumption proportions across species with and without AC).

Asian carp do not compete for food with benthic invertebrates and benthic fish, thus, fishes (Group B) that prey on benthivores fish (e.g., round goby (Neogobius malanostomus)) did not experience negative biomass effects from the invasion, but instead had positive biomass effects from the extra food source young AC provided. Furthermore, yellow perch and white perch feed both on benthic and pelagic prey and were less affected by AC invasion. Lake whitefish feed on benthic invertebrates and common carp feed on benthos and detritus, so were also less affected by AC invasion. AC may push out other species from their habitat (Phelps et al., 2017), however, EwE does not have spatial resolution. The food web responses in this study are mainly attributed to the changes in trophic interactions and external drivers (fishery catch and nutrient loads), and space competition was not considered in the model.

3.1.2 Economic Responses

Keeping with the comparison of the integrated simulations, S1 (with AC) and S2 (without AC), we next turn to the long-run impacts of an Asian carp introduction on the economic variables. The modelling framework includes many variables that could be evaluated (production across various industries, imports, exports, factor demands), here we focus only on the variables that measurably respond to an AC carp invasion. We first look at commercial harvest and recreational catch (Figure 4). For all species, the percent difference in commercial harvest moves in the same direction as the percent difference in each respective species stock, but to a smaller magnitude; lake whitefish, yellow perch, and channel catfish have larger commercial harvests in S1, and white bass has a smaller commercial harvest in S1. There is a negligible difference in the commercial harvests of freshwater drum and suckers between S1 and S2. For species that experience biomass increases, the resulting small change in commercial catch can be attributed to the effort required to harvest. Because additional resources are needed to catch additional fish, increases in biomass are not fully exploited. The percent difference in recreational landings also moves in the same direction as the percent difference in biomass; lake whitefish, yellow perch, and smallmouth bass have more recreational landings in S1, and lake trout, rainbow trout, walleye and white bass have less landings in S1. The percent differences in recreational catch are also smaller than the percent differences in biomass. Again, the different magnitudes of change are due to the effort required to catch fish.

Beyond looking at the changes in commercial and recreational landings, the impact of Asian carp can be measured through changes in net economic welfare, a measure of the wellbeing of households in the regional economy. Welfare[2] depends on household income and household consumption. In the general equilibrium model, household income comes from payments for factors of production, so changes in production across industries, including commercial fishing, can affect household income. Household consumption includes demand for goods, services, and recreational fishing; when market goods or recreational fishing become relatively more expensive, the household's welfare declines. In the long run, the total welfare impacts of an AC introduction are small, only 0.43 percent lower in S1 than in S2[3]. When the welfare measure is disaggregated by household income deciles, as shown in Figure 5, we see that all households have lower welfare in S1. Households 1 and 2, representative of the lowest and second

---

[2] Welfare is measured using an indirect money metric utility function as defined by (Varian, 1992) and calibrated by (Rutherford, 2008)

[3] Recall that we only examine U.S. counties surrounding Lake Erie. The Canadian fishery is generally larger, so the welfare impacts are likely on the conservative side.

lowest income deciles, experience the largest percent difference in welfare in the final period between the two scenarios with welfare 1.30 and 1.78 percent lower in S1 respectively[4].

There is a negligible percent difference in households' disposable income between S1 and S2. Household income constrains both market consumption and non-market activities, but the small differences in income between S1 and S2 suggest that the welfare impacts are being driven by price changes. There was little difference in the market prices of non-fishing sectors between the two simulations. White bass experiences the largest percent difference in market price, with a price 2% higher with-AC, a result that is consistent with the biomass impacts. The biomass level of white bass is substantially lower when AC are present. As a sought-after resource becomes more scarce, we expect the price to increase, which is what we see with white bass.

For recreational fishing, prices reflect the cost of a species-targeted fishing trip. As a species becomes more scarce, the price of a species-targeted fishing trip increases (i.e. becomes more expensive). The household targets species based on their relative non-market values, or the species that gives them the most 'bang for their buck' for their time, so the household moves away from more expensive trips leading to less exerted effort and in turn the landings of expensive species decrease. Initially, white bass provides the household with the highest non-market value of their time, but as its biomass declines following an Asian Carp invasion, targeting white bass becomes relatively more expensive (see Figure 6). The household reallocates their fishing trips among the remaining species. Species that are relatively cheaper to target in S1 see larger catch levels (lake whitefish, yellow perch, smallmouth bass), while the species that are relatively more expensive in S1 see smaller catch levels. Regardless of the biomass declines, the household still targets all species, and overall experiences a negative welfare impact driven by the large increases in the price of targeting some species (i.e. white bass). These changes to the household's cost drive the magnitude of total welfare effects.

[4] A sensitivity analysis was performed (discussed in Appendix C). Increasing the travel cost of fishing further reduced household welfare when AC were present.

While the magnitude of the welfare effects are driven by changes in the non-market value of the recreational fishing, the variation in welfare effects across households are driven by differences in consumption patterns across the households. Overall, HH2 has the lowest initial consumption of the households, but of their total consumption, they demand a higher share of the commercial fish species, water, and power sectors than the other households. These initial demands are used to represent the households' preferences and the proportions are held in the simulations. Household 2's higher proportion of consumption of white bass combined with its 2% increase in price, is likely driving the greater welfare loss experienced by HH2. This suggests that when welfare impacts are driven by price changes as opposed to direct income changes, a household's preferences may make them vulnerable to greater welfare consequences.

3.2 Effects of Modeling Feedbacks

In the previous section, comparing Simulations 1 and 2 allowed us to see how an introduction of AC can affect both the food web and the regional economy. Another objective of our work is to understand the significance of including feedbacks between the ecological and economic systems. To do so, we compare the ecological responses of an AC invasion to Lake Erie with and without the coupled economic model. In this section we evaluate the projected biomass levels of fish species following an AC invasion, with and without feedbacks between the ecological and economic models.

Figure 7 shows the percent change in biomass levels of harvested fish species between the first and final iterations of the simulations[5]. While results for some species are similar across the two simulations, we see very different results between S1 (coupled model) and S3 (ecological model only) for adult walleye, smallmouth bass, common carp. The ecological model (S3) <u>overestimates</u> the change in biomass of yellow perch, rainbow trout, lake trout and white perch and <u>underestimates</u> the change in

[5] A comparison of the biomass levels in S1 and S3 over the 152 iterations is shown in Figure D.3 in Appendix D.

biomass of common carp, white bass, adult walleye, smallmouth bass, lake whitefish, freshwater drum, and channel catfish, relative to the coupled economic-ecological results (S1).

In addition to differences in the equilibrium (ending) biomass change on species, the S1 and S3 vary in the biomass transition dynamics between the first and final iterations. In early iterations some of the species biomass levels differ greatly between S1 and S3 (Figure D.3) but have less deviation in the final iteration. For example, at one point there is a difference in the estimated biomass of adult walleye of 179 percentage points between S1 and S3, but in the final iteration there is only a difference of 80 percentage points. Inaccurately accounting for these transition dynamics could result in improper population estimates and management recommendations (total allowable catch), that could result in overfishing or unanticipated consequences through food web interactions.

We find that some of the deviation between S1 and S3 is driven by details of the economic model. One important observation is that for species fished recreationally, the deviation in biomass estimates between S1 and S3 is related to the change in demand for fishing trips. We observe that the biomass levels that are over or under estimated by the greatest magnitude in S3 are for fisheries that experience the largest percent change in the willingness to pay for recreational fishing trip. These results are shown in Table 2. As the biomass of species change, consumers' willingness to pay for species adjusts and their choices of species-targeted fishing trips change. S3 does not take into account that households respond to changes in value and overestimates the biomass of species households highly value. This result highlights how households adjust their recreational fishing choices, and that those adjustments are missing from S3. We also evaluated the relationships between the magnitude of the over- and under- estimation and effort parameters in both the commercial and recreational fishing, suspecting that there may be some relationship between effort intensity and the magnitude of over- or under- estimate, but were not able to confirm a relationship. Lastly, using the biomass values in the final period of S3, we ran one period of the economic model to evaluate the welfare consequences of misrepresenting the feedbacks. The resulting welfare using the S3 biomass values was smaller than the welfare in the final period of S1, but only by

0.14%, suggesting that for the Asian carp invasion, the primary consequence of misrepresenting the feedbacks is arriving at different population estimates.

4. Conclusions

The interconnection between humans and the ecosystem creates feedback loops; human actions influence the future state of the environment, which creates a novel set of conditions and constraints for humans to base their decisions on in the next period. When the ecological-economic interconnection is modeled as unidirectional, as was done in S3, important human and ecosystem responses are missed. The coupled model we developed quantifies the differences in estimated impacts from invasive species when the human component of adaptability is omitted from the analysis and addresses the need for bidirectional feedback loops in bioeconomic models, particularly in general equilibrium models that emphasize the use of linkages between endogenous systems over multiple periods.

Using a model of this scale and complexity requires special care when calibrating and choosing parameter values and deserves the attention of a robust sensitivity analysis. Our sensitivity analysis (discussed in Appendix C) showed that commercial harvest, recreational catch, and biomass were not very responsive to changes in elasticity of supply, willingness to pay, and travel costs, but household welfare was affected, particularly by travel cost. If a model of this sort was to be used for management decisions, a more robust sensitivity analysis is warranted. When discussing the estimated impacts of Asian carp, we took care to compare variable levels in each period between a simulation with and without the ecological disturbance. We believe that this is the prudent way to evaluate such shocks, as opposed to only running a simulation with the shock and comparing the initial and final biomass levels. By evaluating variables with and without the shock, we can be more confident that the responses of the entire system are more accurately captured. It is also worth acknowledging that an integrated model of this scale is resource intensive to build, and may not be feasible for all policy questions or managers, but once built for a region, using the framework for multiple scenario and policy questions will allow managers to accrue additional benefits that may offset the costs.

The benefits of using a framework in which economic and ecological systems feed into each other include being better able to estimate the impacts of an exogenous shock to either system. In this analysis, we chose to focus on a potential Asian carp invasion to Lake Erie, but the modeling framework could be used for a variety of scenarios. For the AC scenario, only harvest, recreational catch, prices, biomass, and welfare were impacted, but in other scenarios, other relevant variables (for example imports, exports, factor demand) could be impacted and evaluated. If impacts were only estimated with the ecological framework, in this case an EwE model, the biomass levels of some species would be overestimated, and some species would be underestimated relative to models that include human responses. If the economic impacts were only estimated with the initial response from the EwE system, as a single input to an economic model, the long-run ecological impacts would not be accounted for and the initial noisy response of the population dynamics may be given too much credence. When including bidirectional feedbacks between the food web and economic model we find that some species have smaller biomass levels, and some have larger biomass levels when Asian carp are present. We also see that while total welfare impacts of an Asian carp invasion are small, the lowest-income households experience the greatest welfare losses following an invasion. The welfare impacts are driven by reduced purchasing power and consumer preferences.

We also note that food web–only analysis would miss the changes and substitutions that occur to harvest due to changing consumption allocations and recreational fishing value; omissions that could have important policy implications. Our comparison of biomass levels with and without the economic feedbacks show that even if the economic welfare impacts from an invasion are expected to be small (in this case a welfare difference of 0.43%), omitting the human component may lead to over/underestimates of ecological effects and misguided management decisions. The over/underestimates will be particularly apparent for species that are recreationally targeted and have large changes in perceived value by the household. Our findings highlight the need to include a bidirectional feedback between ecological and economic systems when modeling environmental disturbances.

Acknowledgements

This analysis would not be possible without the work of Jason Breck, Ph.D. for his work on programing the link between Ecosim and the economic model during his time as a Computer Specialist at the University of Michigan and as a graduate student at the University of Wisconsin.

Part of this research was funded by Environmental Protection Agency Great Lakes Restoration Initiative, NOAA Center for Sponsored Coastal Ocean Research awards NA09NOS4780192 and NA10NOS4780218. This is contribution number XXXXXX of the NOAA Great Lakes Environmental Research Laboratory.

References

Armington, P.S., 1969. A Theory of Demand for Products Distinguished by Place of Production. Staff Pap. 16, 159–178. https://doi.org/10.2307/3866403

Armstrong, C.W., 2007. A note on the ecological–economic modelling of marine reserves in fisheries. Ecol. Econ., Special Section: Ecological-economic modelling for designing and evaluating biodiversity conservation policiesEE Modelling Special Section 62, 242–250. https://doi.org/10.1016/j.ecolecon.2006.03.027

Avila-Foucat, V.S., Perrings, C., Raffaelli, D., 2009. An ecological–economic model for catchment management: The case of Tonameca, Oaxaca, México. Ecol. Econ. 68, 2224–2231. https://doi.org/10.1016/j.ecolecon.2009.01.010

Besedin, E., Mazzotta, M., Cacela, D., Tudor, L., 2004. Combining ecological and economic analysis: an application to valuation of power plant impacts on Great Lakes recreational fishing, in: American Fisheries Society Meeting Symposium: Socio-Economics and Extension: Empowering People in Fisheries Conservation, Madison, WI.

Bjørndal, T., Conrad, J.M., 1987. The Dynamics of an Open Access Fishery. Can. J. Econ. Rev. Can. Econ. 20, 74–85. https://doi.org/10.2307/135232

Bossenbroek, J.M., Finnoff, D.C., Shogren, J.F., Warziniack, T.W., 2009. Advances in Ecological and Economic Analysis of Invasive Species: Dreissenid Mussels as a Case Study, in: Keller, R.P., Lodge, D.M., Lewis, M.A., Shogren, J.F. (Eds.), Bioeconomics of Invasive Species: Integrating Ecology, Economics, and Management. Oxford Press, New York, pp. 244–265.

Brockmann, S., Zhang, H., Mason, D.M., Rutherford, E.S., 2021. Space and Species Interactions in Welfare Estimates for Invasive Species Policy. Front. Ecol. Evol. 9, 794. https://doi.org/10.3389/fevo.2021.703935

Byron, C.J., Jin, D., Dalton, T.M., 2015. An Integrated ecological–economic modeling framework for the sustainable management of oyster farming. Aquaculture 447, 15–22. https://doi.org/10.1016/j.aquaculture.2014.08.030

Campbell, H.F., 1991. Estimating the elasticity of substitution between restricted and unrestricted inputs in a regulated fishery: A probit approach. J. Environ. Econ. Manag. 20, 262–274. https://doi.org/10.1016/0095-0696(91)90012-8

Chiang, A., Wainwright, K., 2004. Fundamental Methods of Mathematical Economics, 4 edition. ed. McGraw-Hill Education, Boston, Mass.

Christensen, V., Walters, C.J., 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecol. Model. 172, 109–139. https://doi.org/10.1016/j.ecolmodel.2003.09.003

Christensen, V., Walters, C.J., Pauly, D., 2005. Ecopath with Ecosim: a User's Guide. Fisheries Centre, University of British Columbia, Vancouver.

Connelly, N.A., O'Neill, C.R., Knuth, B.A., Brown, T.L., 2007. Economic Impacts of Zebra Mussels on Drinking Water Treatment and Electric Power Generation Facilities. Environ. Manage. 40, 105–112. https://doi.org/10.1007/s00267-006-0296-5

Cox, S.P., Kitchell, J.F., 2004. Lake Superior Ecosystem, 1929–1998: Simulating Alternative Hypotheses for Recruitment Failure of Lake Herring (Coregonus Artedi). Bull. Mar. Sci. 74, 671–683.

De Melo, J., Tarr, D.G., 1992. A General Equilibrium Analysis of US Foreign Trade Policy. MIT Press.

Eichner, T., Tschirhart, J., 2007. Efficient ecosystem services and naturalness in an ecological/economic model. Environ. Resour. Econ. 37, 733–755. https://doi.org/10.1007/s10640-006-9065-4

Fenichel, E.P., Abbott, J.K., 2014. Natural Capital: From Metaphor to Measurement. J. Assoc. Environ. Resour. Econ. 1, 1–27. https://doi.org/10.1086/676034

Ferraro, P.J., Sanchirico, J.N., Smith, M.D., 2019. Causal inference in coupled human and natural systems. Proc. Natl. Acad. Sci. 116, 5311–5318. https://doi.org/10.1073/pnas.1805563115

Finnoff, D., Shogren, J.F., Leung, B., Lodge, D., 2005. The importance of bioeconomic feedback in invasive species management. Ecol. Econ. 52, 367–381. https://doi.org/10.1016/j.ecolecon.2004.06.020

Finnoff, D., Tschirhart, J., 2008. Linking dynamic economic and ecological general equilibrium models. Resour. Energy Econ. 30, 91–114. https://doi.org/10.1016/j.reseneeco.2007.08.005

Finnoff, D., Tschirhart, J., 2003. Protecting an Endangered Species While Harvesting Its Prey in a General Equilibrium Ecosystem Model. Land Econ. 79, 160–180.

Finnoff, D.C., Settle, C., Shogren, J.F., Tschirhart, J.T., 2009. Integrating Economics and Biology for Invasive Species Management, in: Keller, R.P., Lodge, D.M., Lewis, M.A., Shogren, J.F. (Eds.), Bioeconomics of Invasive Species: Integrating Ecology, Economics, Policy, and Managment. Oxford University Press, New York, pp. 25–43.

Gilliland, T.E., Sanchirico, J.N., Taylor, J.E., 2019. An integrated bioeconomic local economy-wide assessment of the environmental impacts of poverty programs. Proc. Natl. Acad. Sci. 116, 6737–6742. https://doi.org/10.1073/pnas.1816093116

Great Lakes Commission, St. Lawrence Cities Initiative, 2012. Restoring the Natural Divide: Separating the Great Lakes and Mississippi River Basins in the Chicago Area Waterway System.

Hebert, L., 2010. Carp fence completed to stop Maumee River invasion [WWW Document]. WNWO. URL http://nbc24.com/news/local/carp-fence-completed-to-stop-maumee-river-invasion (accessed 5.24.16).

Hubert, W.A., Quist, M.C., 2010. Inland Fisheries Management in North America, 3rd Edition 2010, 3rd edition. ed. American Fisheries Socirty, Bethesda, Md.

Hutniczak, B., 2015. Modeling heterogeneous fleet in an ecosystem based management context. Ecol. Econ. 120, 203–214. https://doi.org/10.1016/j.ecolecon.2015.10.023

IMPLAN Group, 2013. IMPLAN. Huntersville, NC.

Jin, D., Hoagland, P., Dalton, T.M., Thunberg, E.M., 2012. Development of an integrated economic and ecological framework for ecosystem-based fisheries management in New England. Prog. Oceanogr. 102, 93–101. https://doi.org/10.1016/j.pocean.2012.03.007

Kao, Y.-C., Adlerstein, S., Rutherford, E., 2014. The relative impacts of nutrient loads and invasive species on a Great Lakes food web: An Ecopath with Ecosim analysis. J. Gt. Lakes Res., The Continuing Effects of Multiple Stressors in Saginaw Bay 40, Supplement 1, 35–52. https://doi.org/10.1016/j.jglr.2014.01.010

Kocovsky, P.M., Chapman, D.C., McKenna, J.E., 2012. Thermal and hydrologic suitability of Lake Erie and its major tributaries for spawning of Asian carps. J. Gt. Lakes Res. 38, 159–166. https://doi.org/10.1016/j.jglr.2011.11.015

Kolar, C.S., Chapman, D.C., Jr, W.R.C., Housel, C.M., Williams, J.D., Jennings, D.P., 2007. Bigheaded carps : a biological synopsis and environmental risk assessment.

Koonce, J.F., Busch, W.-D.N., Czapla, T., 1996. Restoration of Lake Erie: contribution of water quality and natural resource management. Can. J. Fish. Aquat. Sci. 53, 105–112. https://doi.org/10.1139/f96-003

Lake Erie Committee, Great Lakes Fishery Commission, 2015. Lake Erie Walleye Management Plan 2015-2019.

Lake Erie Committee Yellow Perch Task Group, 2016. Report of Lake Erie Yellow Perch Task Group.

Lake Erie Committee Yellow Perch Task Group, 2013. Yellow Perch Task Group Executive Summary Report.

Langseth, B.J., Rogers, M., Zhang, H., 2012. Modeling species invasions in Ecopath with Ecosim: An evaluation using Laurentian Great Lakes models. Ecol. Model. 247, 251–261. https://doi.org/10.1016/j.ecolmodel.2012.08.015

Lee, K., Apriesnig, J., Zhang, H., 2021. Socio-Ecological Outcomes of Single-Species Fisheries Management: The Case of Yellow Perch in Lake Erie. Front. Ecol. Evol. 9, 703813. https://doi.org/10.3389/fevo.2021.703813

Lofgren, H., Harris, R.L., Robinson, S., 2002. A Standard Computable General Equilibrium (CGE) Model in GAMS. Intl Food Policy Res Inst.

Manning, D.T., Taylor, J.E., Wilen, J.E., 2016. General Equilibrium Tragedy of the Commons. Environ. Resour. Econ. 1–27. https://doi.org/10.1007/s10640-016-0066-7

McDermott, S.M., Finnoff, D.C., Shogren, J.F., 2013. The welfare impacts of an invasive species: Endogenous vs. exogenous price models. Ecol. Econ. 85, 43–49. https://doi.org/10.1016/j.ecolecon.2012.08.020

National Oceanic and Atmospheric Administration, U.S. Geological Survey, 2013. Great Lakes Commercial Fishery Landings [WWW Document]. Natl. Mar. Fish. Serv. URL http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/other-specialized-programs/great-lakes-landings/index

Ohio Division of Wildlife, 2012. Ohio's Lake Erie Fisheries 2012. Ohio Deparment of Natural Resources, Division of Wildlife, Lake Erie Fisheries Units.

Plagányi, é. E., Butterworth, D.S., 2004. A critical look at the potential of Ecopath with ecosim to assist in practical fisheries management. Afr. J. Mar. Sci. 26, 261–287. https://doi.org/10.2989/18142320409504061

Rutherford, T.F., 2008. Calibrated CES Utility Functions: A Worked Example.

U.S. Army Corps of Engineers, 2014. The GLMRIS Report, Great Lakes & Mississippi River Interbasin Study.

U.S. Department of the Interior, U.S. Fish & Wildlife Service, U.S. Department of Commerce, U.S. Census Bureau, 2014. 2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.

Varian, H.R., 1992. Microeconomic Analysis. Norton & Company.

Warziniack, T., Finnoff, D., Bossenbroek, J., Shogren, J.F., Lodge, D., 2011. Stepping Stones for Biological Invasion: A Bioeconomic Model of Transferable Risk. Environ. Resour. Econ. 50, 605–627. https://doi.org/10.1007/s10640-011-9485-7

Warziniack, T., Finnoff, D.C., Apriesnig, J.L., 2017. General Equilibrium Model of Ecosystem Services (GEMES), General Technical Report. USDA Forest Service, Rocky Mountain Research Station.

Wing, I.S., 2004. Computable general equilibrium models and their use in economy-wide policy analysis. Tech. Note Jt. Program Sci. Policy Glob. Change MIT.

Wittmann, M.E., Cooke, R.M., Rothlisberger, J.D., Rutherford, E.S., Zhang, H., Mason, D.M., Lodge, D.M., 2015. Use of structured expert judgment to forecast invasions by bighead and silver carp in Lake Erie. Conserv. Biol. 29, 187–197. https://doi.org/10.1111/cobi.12369

Yun, S.D., Hutniczak, B., Abbott, J.K., Fenichel, E.P., 2017. Ecosystem-based management and the wealth of ecosystems. Proc. Natl. Acad. Sci. 114, 6539–6544. https://doi.org/10.1073/pnas.1617666114

Zhang, H., Rutherford, E.S., Mason, D.M., Breck, J.T., Wittmann, M.E., Cooke, R.M., Lodge, D.M., Rothlisberger, J.D., Zhu, X., Johnson, T.B., 2016. Forecasting the Impacts of Silver and Bighead Carp on the Lake Erie Food Web. Trans. Am. Fish. Soc. 145, 136–162. https://doi.org/10.1080/00028487.2015.1069211

Zhang, J., Smith, M.D., 2011. Estimation of a generalized fishery model: A two-stage approach. Rev. Econ. Stat. 93, 690–699.

Appendices

A.1 CGE Data & Parameterization

This section provides background and parameterization of the economic components required for simulation. We begin by describing the social accounting matrix, and then describe the harvest and recreational fishing data and parameters.

A.1.1 Social Accounting Matrix

The foundation for the economic model used in this analysis is a social accounting matrix (SAM). A SAM is a succinct and comprehensive way of representing economic activity in equilibrium for 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 et al., 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 in 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 we are 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 (IMPLAN Group, 2013). 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.

A.1.1.1 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 A1. However, a few sectors, like recreational and commercial fishing had to be constructed using other data; those sectors are shown in Table A.2.

The species-specific commercial fishing sectors were disaggregated from IMPLAN's commercial fishing sector using the USGS Lake Erie Landing report from 2013 (National Oceanic and Atmospheric Administration and 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 A.3. 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, we 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, we assume all remaining activities can be attributed to the fishing of other species both in Lake Erie and the surrounding lakes.

Recreational fishing was disaggregated from other IMPLAN sectors using data from the 2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation (U.S. Department of the Interior et al., 2014). The survey reports daily expenditures of fishing the Great Lakes and the number of days fishing Lake Erie, from which we 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 composite sector of all other activities in a manner similar to the commercial fishing disaggregation.

A.1.1.2 SAM and Biomass Reconciliation

Traditionally, social accounting matrices only include market inputs. Little guidance for extending a SAM to include nonmarket inputs is provided by the literature, so additional steps were 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, we 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. We use the yellow perch sector and its effort demand as an example to further describe how these values are reconciled.

Table A.4 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 reconcile the economic IMPLAN data with the harvest data for the fish species, it is assumed that the initial output price of fish, P_f, is such that Euler's equation holds, and there are zero profits, $(\alpha_f + \beta_f)\, P_f H_f = P_{Ef}^{ROA}\, E_f$. Thus, the initial output price of fish equals $P_f = \frac{E_f P_{Ef}^{ROA}}{(\alpha_f+\beta_f)\, H_f}$, where P_f is the output price of fish, and H_f is the total harvest from either the GLFC or the ODNR depending on the species, and α_f and β_f are the effort and stock elasticities of harvest respectively (their derivation is discussed in Section A.1.3). For yellow perch, $\alpha_f + \beta_f = 1$, so the benchmark output price is: $P_f = \frac{\$14.57 million}{H_f}$. While units of goods in a traditional SAM are generally thought to be arbitrary, and thus defined so benchmark prices are equal to one, 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.

Another parameter that must be considered is the catchability coefficient in the harvest function. With the estimated elasticities, the catchability coefficient is calculated from the benchmark data in a way to ensure that the initial harvest level is consistent with the stock and the effort levels present, $q_f = \frac{H_f}{E_f^{\alpha_f} S_f^{\beta_f}}$. By calculating the catchability coefficient in this way, we ensure that data from the SAM and data from outside sources is reconciled.

A.1.2 Initial Household Recreation Values

There are multiple variables in the households' recreational choices that require 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 we 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, we use a willingness to pay from a species group with similar characteristics. The willingness to pay values from the study are shown in Table A.5, we converted these values to 2013 dollars for the simulations. 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.

A.1.3 Elasticities

The exogenous parameters in the economic model can be classified into two groups: 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. There are two sets of elasticities that are needed for the CGE. The elasticities for each of the constant elasticity of substitution functions are taken from the literature. In their 2008 work, Finnoff & Tschirhart (2008) 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, we use their values in this analysis (Table A.6).

A.1.3.1 Harvest Elasticities

Because the standard SAM does not include payments to fish stock, it cannot be used to estimate the production function that combines effort and stock to produce harvest. Instead, we use regression analysis to estimate the fisher production (harvest) function. The specifics of estimating the fishery production functions are described here.

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

$$H_f = q_f E_f^{\alpha_f} S_f^{\beta_f} \tag{A.1}$$

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. Because stock is not included in the SAM, these parameters cannot be estimated from the benchmark data and must be obtained via alternative means. We 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, we use the natural log version of the Cobb-Douglas harvest function (Equation A.2) in the estimate of the Cobb-Douglas parameters.

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

In Equation A.2, y represents the year, and f indicates the fishing firm and species, and $\epsilon_{f,y}$ is the error term. 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.

A.1.3.1.1 Yellow Perch Data

Harvest, stock, and effort data for yellow perch 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 Committee Yellow Perch Task Group, 2013). We believe that harvest is a better indicator of the fisher's behavior than the TAC, so for the purposes of this estimation we use harvest as the dependent variable; this approach follows the estimation techniques of Bjørndal and Conrad (1987), Campbell (1991), Zhang and Smith (2011).

A.1.3.1.2 Walleye Data

The walleye data were also obtained from annual reports published by the GLFC 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 a large commercial fishery.

A.1.3.1.3 Other Species Data

Three other species we 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 the Ohio Department of Natural Resource (ODNR) (Ohio Division of Wildlife, 2012). 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 we only had access to the method of that harvest (i.e. net type) for 1999, 2003, 2012, 2015. We decided to use 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 us 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. We 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 us to calculate the average weight of each species by using the length-weight relationship represented by Equation A.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.

$$log(Weight_f) = c_f + d_f log(length_f) \tag{A.3}$$

Through these steps, we were 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, we used millions of pounds of harvest, number of trap net lifts, and relative abundance in millions of pounds from 1990-2015.

A.1.3.1.4 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 whitefish, the null was rejected. The estimated coefficients and the lags of autoregressive integrated moving average models for each species is shown in Table A.7.

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 we 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 we also test to see if they exhibit constant returns to scale. We test the null hypothesis of $\gamma_1 + \gamma_2 = 1$, with a χ^2 – distribution Wald test, and reject the null of constant returns to scale for white bass but cannot reject the null for yellow perch. We assume that constant returns to scale exists for yellow perch and use the coefficient estimated at the highest significance to calculate the other, so for yellow perch $\alpha_f = \gamma_1 = 0.763$ and $\beta_f = 1 - 0.763 = 0.237$. Because we reject the null that constant returns to scale exists for white bass, and increasing returns to scale seems unlikely, we chose to let the two elasticities be equal, and set $\alpha_f = \beta_f = \gamma_2 = 0.185$. Given the lack of relationship between white bass harvest and stock we could set the elasticity to zero, but this would imply that effort has no impact on harvest. Instead, we, chalk the lack of significance up to poor data. In addition, we test the null hypothesis that $\alpha_f + \beta_f = 0.37$ and cannot reject the null. Stock and effort data is not available for the other species of interest in the model, so we 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 $\beta_f = 0.234$.

B.1 Lake Erie Harvest Restrictions

A final aspect of the model that needs to be identified are the harvest restrictions on yellow perch and walleye. This section describes how those restrictions are included in the model.

B.1.1 Yellow Perch Total Allowable Catch

Catches of yellow perch from Lake Erie are 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 Committee Yellow Perch Task Group, 2016). The Lake Erie Yellow Perch Task Group (YPTG) has set target fishing (harvest) rates, $\mathrm{F_m}$, for each management unit that they believe “support viable sport and commercial fisheries without inviting excessive biological risk,” (Lake Erie Committee 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 B.1), we derive a function that approximates the TAC for the U.S. proportion of Lake Erie for a given biomass of the yellow perch (Equation B.1), where the subscript yp represents yellow perch.

$$TAC_{yp} = S_{yp} \sum_{m} R_M F_m = 0.31291 S_{yp} \tag{B.1}$$

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 our 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, we assume that each type of fishing gets a proportion of the TAC. We 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 B.2 and the commercial TAC *($TACC_{yp}$)* is shown in Equation B.3.

$$TACR_{yp} = .628\, TAC_{yp} = 0.19651 S_{yp} \qquad \text{(B.2)}$$

$$TACC_{yp} = .372\, TAC_{yp} = 0.11640 S_{yp} \qquad \text{(B.3)}$$

B.1.2 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 TAC formulation during this time period sets the TAC such that the fishing mortality rate is equal to 60% of the maximum sustainable yield (MSY) fishing rate (Lake Erie Committee and 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 our simulations, we use the 'rule of thumb' equation for MSY (Hubert and Quist, 2010, p. 461) as shown in Equation B.4 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.

$$MSY_w = p_w M_w S_w \tag{B.4}$$

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. We do not have access to that information, so instead use the projected fish stock. We 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, $M_w = 0.32$, so we do the same. With MSY calculated, we are 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 B.5 and B.6 respectively.

$$F_{MSY\ w} = \frac{MSY_w}{S_w} \tag{B.5}$$

$$F_{60MSY\ w} = 0.60\frac{MSY_w}{S_w} \tag{B.6}$$

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 our simulations we 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 us 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 B.7, 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).

$$TAC_W = \frac{F_{60MSY}\,(0.086SJ_W + \ 0.412SA_W)}{M_f + F_{60MSY}} \tag{B.7}$$

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 B.8.

$$TACR_W = \frac{F_{60MSY}\,(0.086SJ_W + 0.412SA_W)}{M_f + F_{60MSY}} 0.5694 \quad \text{(B.8)}$$

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, we set the commercial TAC of walleye equal to the 2013 commercial landings reported by the USGS. USGS reports the number of fish harvested. We 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 B.9 shows the commercial TAC of walleye.

$$TACC_W = 986.97\ lbs \quad \text{(B.9)}$$

C.1 Sensitivity Analysis

A robust sensitivity analysis of a model with this complexity is a cumbersome, yet important, undertaking. Across all the industries and households there are over a hundred parameters in the economic model alone; many of which are calibrated using the base data. There are various approaches that could be used for a sensitivity analysis, for example, when evaluating their CGE model, Warziniack et al. (2011) performed a Monte Carlo style sensitivity analysis by taking random draws of parameter values and performing thousands of simulations. That style is of sensitivity analysis can be useful, particularly when the modeling framework is used to drive specific policy insights as part of decision support, but given the integrated nature of the model and current resources (computing power, time, etc.), it is not something we were able to undertake. Instead, we chose to perform a sensitivity analysis in line with Finnoff and Tschirhart (2008) and focus on a few key parameter values that lie at the intersection of the ecological and economic models and that were taken from the literature instead of being calibrated from the base data. We simulated 50% increases and decreases of the elasticity of substitution (σ_{Q_i}) in the supply function (Equation 16), the households' initial travel cost for recreational fishing, and the households' species-specific initial willingness to pay values. For the elasticity and WTP sensitivity analyses, we chose to change only the yellow perch and white bass parameters, instead of changing the values across all the species. By focusing on two species, we were able to keep the sensitivity analysis manageable while allowing for relative values across the species to vary. Table C.1 shows the base values and the values used in the sensitivity analysis.

For each sensitivity analysis scenario we ran an S1 and S2 simulation and compared the differences in biomass, recreational catch, commercial harvest, household welfare, and non-market values. The WTP, elasticity of substitution, and 0.5x travel cost, sensitivity analyses resulted in negligible differences (differences of up to 0.02 percentage points) from our original comparison of S1 and S2 across all variables. The minimal impacts of the changes in elasticity of substitution are in line with what Finnoff and Tschirhart (2008) found. Recreational fishing and biomass levels were less sensitive to changes in WTP values than we expected, this is one area where a more extensive sensitivity analysis would be ideal. More noticeable difference in the household welfare existed in the 1.5x high-travel cost scenario. In that simulation, a 50% increase in the travel cost for recreational fishing led to larger decrease in welfare across all the households (of 0.13-0.56 percentage points) when Asian carp were present. While the increased travel cost affected household welfare, it did not induce changes in recreational fishing or biomass levels, a result consistent with changing a fixed cost. After seeing the welfare implications of the 1.5x travel cost sensitivity analysis we added an additional sensitivity analysis and simulated an increase in travel cost 3x much. The results were consistent; welfare was further reduced when Asian carp were present, but the increased travel cost did not induce changes in fishing patterns. Total welfare was 0.64 percent smaller when AC were present in the 1.5x travel cost scenario and 1.22 percent lower in the 3x travel cost scenario. The percent difference in household welfare for the 1.5x and 3x travel cost simulations are shown in Table C.2.

D.1 Additional Figures and Tables

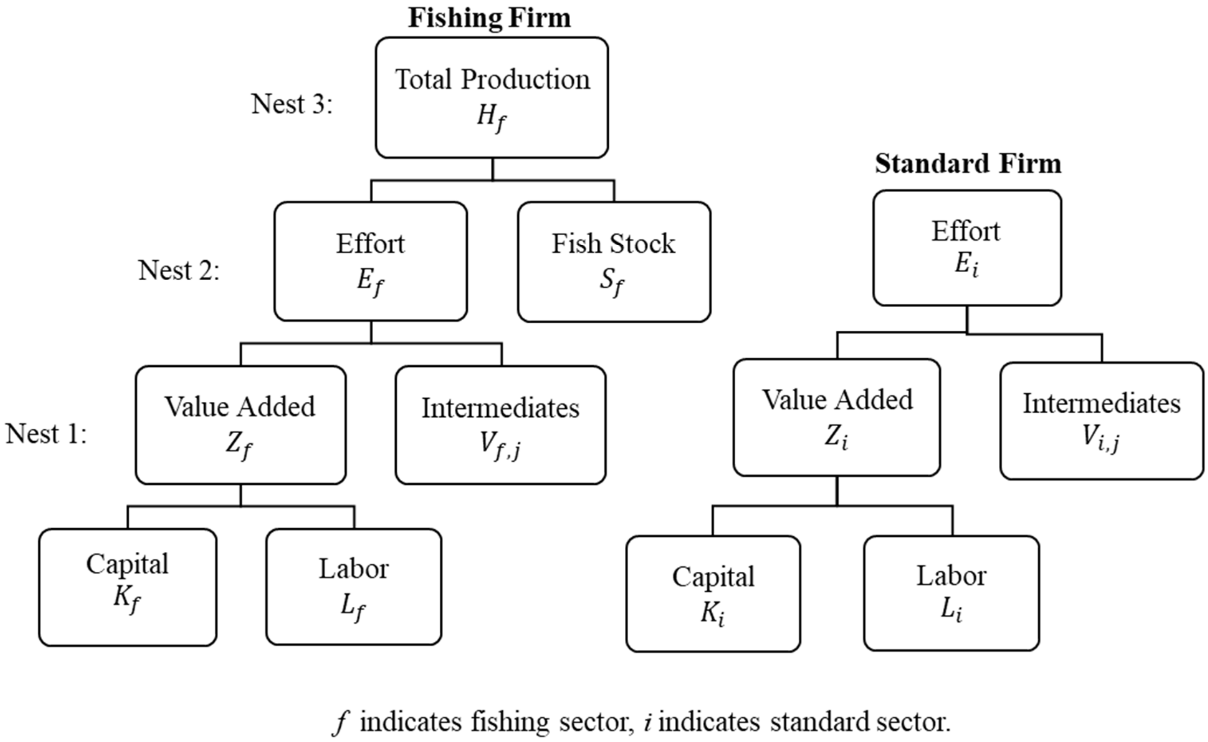

Figure 1: Firm Production Trees
The production nests for fishing firms and the standard firm from non-fishing sectors are shown. Fishing firms include stock as an additional input to production.

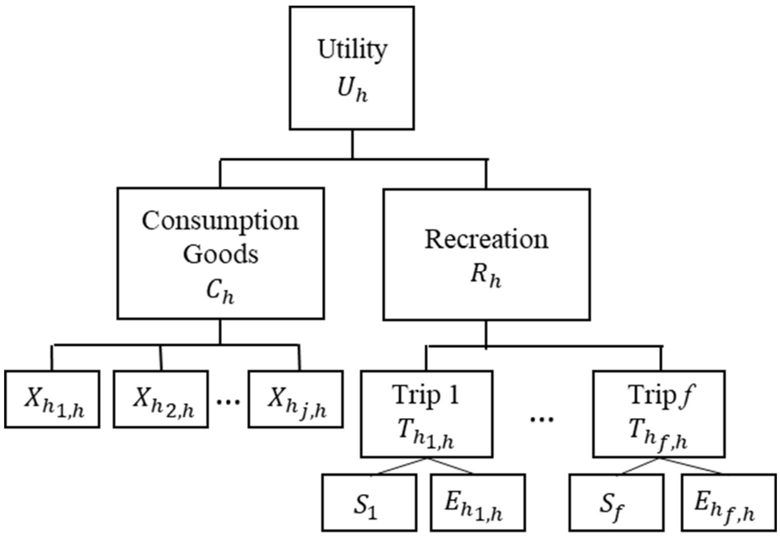

Figure 2: Nesting Structure for Household Behavior
Substitution possibilities at each level in the nest described by constant elasticity of substitution (CES) functions.

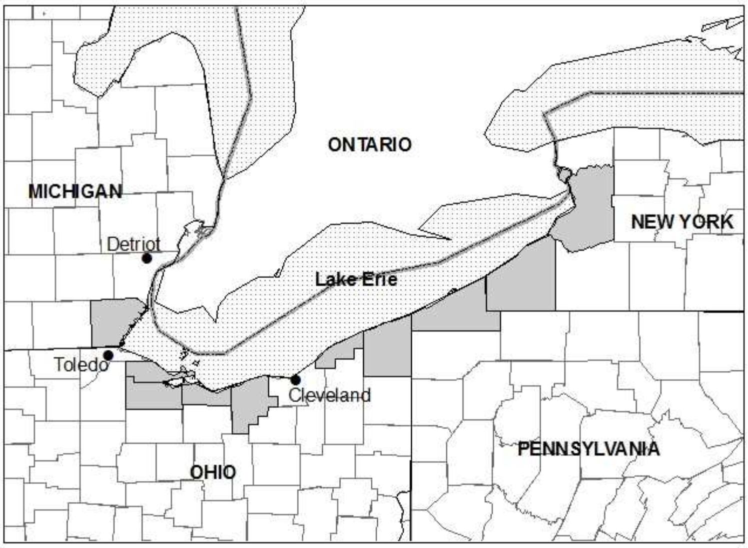

Figure 3: Lake Erie Coastal Economy
Shaded counties are included in the Lake Erie Coastal Economy. Dashed line indicates Canadian and U.S. border.

Table 1: Simulation Definitions

	With Feedbacks	Without Feedbacks
With Asian Carp	Simulation 1	Simulation 3
Without Asian Carp	Simulation 2	Simulation 4

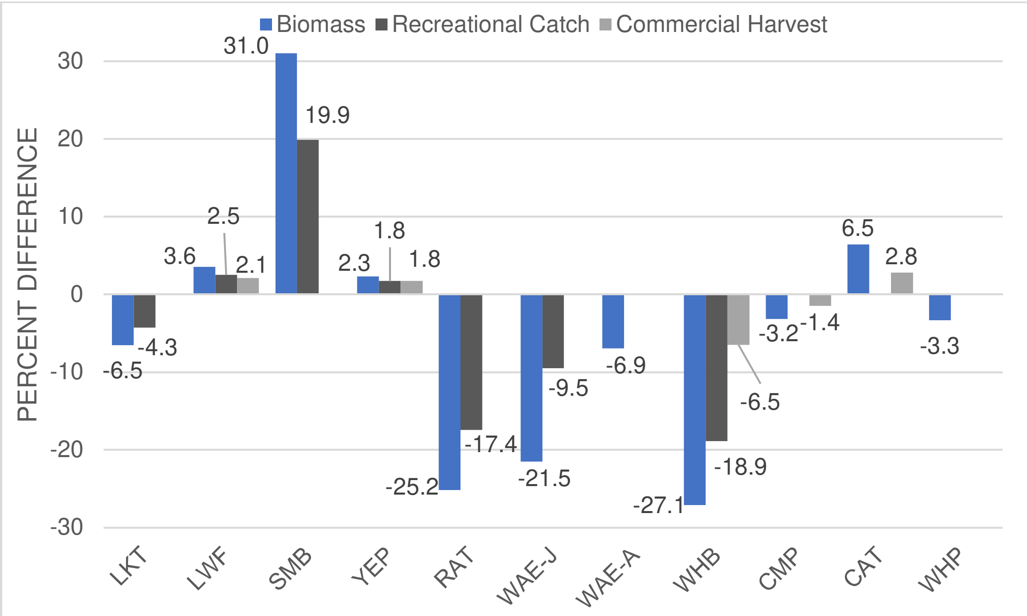

Figure 4: Percent difference in biomass, commercial harvest, and recreational catch of target species in final simulation period between S1 and S2

Percent difference calculated as [([S1 value] – [S2 value])/S2 value] in final simulation period. Freshwater Drum and Sucker fish are omitted from figure because they have a percent different in biomass of less than 1 percent, and even smaller percent differences in commercial harvest. Note: Species are as follows: LKT (Lake Trout, adult), LWF (Lake Whitefish), SMB (Smallmouth Bass), YEP (Yellow Perch, adult), RAT (Rainbow Trout, adult), WAE-J (Walleye, juvenile), WAE-A (Walleye, adult), WHB (White Bass), CMP (Common Carp), CAT (Channel Catfish and Brown Bullhead), WHP (White Perch).

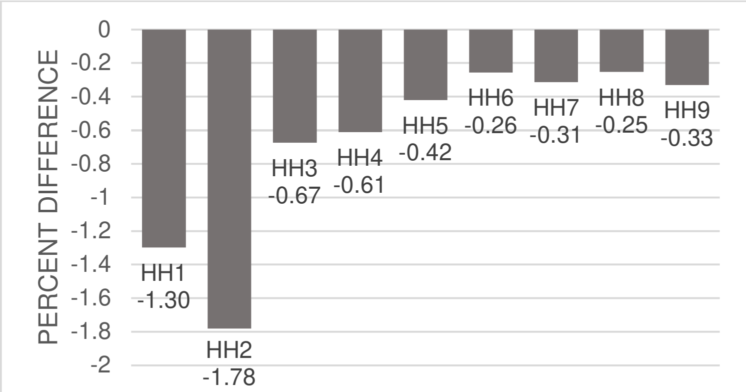

Figure 5: Percent Difference in Total Welfare by Household in Final Simulation Period
Percent difference calculated as [([S1 value] – [S2 value])/S2 value] in final simulation period.
Note: household income brackets are: HH1: less than \$10K, HH2: \$10K-\$15K, HH3: \$15K-\$25K, HH4: \$25K-\$35K, HH5: \$35K-\$75K, HH6: \$50K-\$75K, HH7: \$75K-\$100K, HH8: \$100K - \$150K, HH9 greater than \$150K

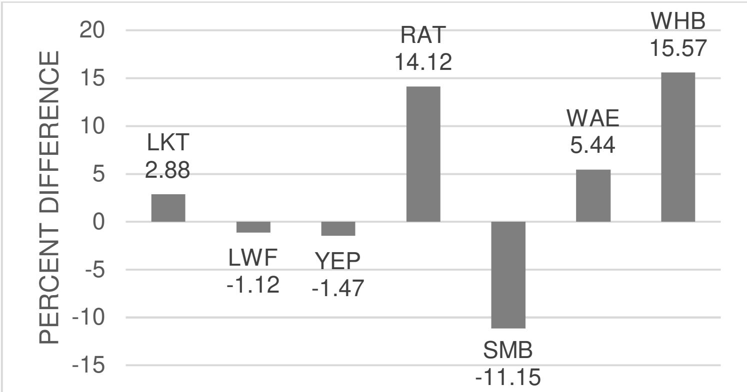

Figure 6: Percent Difference in the non-market value for species targeted trip
Percent difference calculated as [([S1 value] – [S2 value])/S2 value] in final simulation period. Note: Species are as follows: LKT (Lake Trout, adult), LWF (Lake Whitefish), RAT (Rainbow Trout, adult), SMB (Smallmouth Bass), WAE (Walleye), WHB (White Bass),YEP (Yellow Perch, adult)

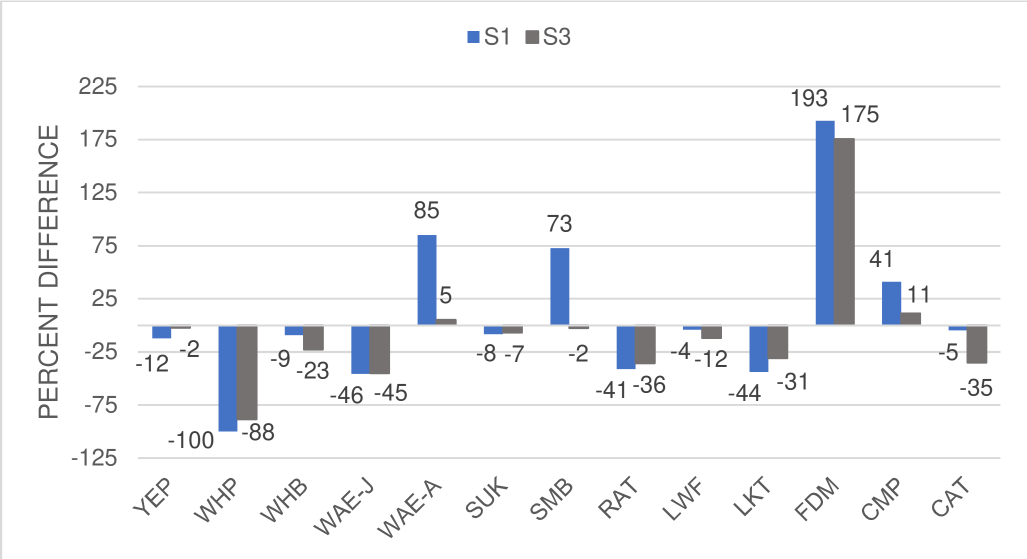

Figure 7: Percent Change in Biomass Levels from Benchmark of Target Species in Final Simulation Period by Simulation Type

Percent Change calculated as: $\%\Delta = 100[S_{f,t} - S_{f,0}]/S_{f,0}$, where $S_{f,t}$ is the biomass of species f in period t, and $S_{f,0}$ is the benchmark biomass of species f. Note: Species are as follows: CAT (Channel Catfish and Brown Bullhead), CMP (Common Carp), FDM (Freshwater drum), LKT (Lake Trout, adult), LWF (Lake Whitefish), RAT (Rainbow Trout, adult), SMB (Smallmouth Bass), SUK (Suckers), WAE-A (Walleye, adult), WAE-J (Walleye, juvenile), WHB (White Bass), WHP (White Perch), YEP (Yellow Perch, adult).

Table 2: Comparison of Percent Change in Willingness to Pay and Magnitude of Over – and Underestimation of Biomass

Species	% Change in WTP[a]	Difference in % Change of Biomass[b]
Smallmouth Bass (SMB)	-46.16	-75.08
Adult Walleye (WAE-A)	-20.77	-79.97
Lake Whitefish (LWF)	5.05	-.754
White Bass (WHB)	12.15	-13.58
Yellow Perch (YEP)	25.46	10.08
Lake Trout (LKT)	89.73	12.89
Rainbow Trout (RAT)	99.33	5.4

Notes: [a] $\%\Delta WTP = 100[WTP_{f,t} - WTP_{f,0}]/WTP_{f,0}$, where $WTP_{f,t}$ is the willingness to pay for recreational fishing trip targeting species f in period t and $WTP_{f,0}$ is the benchmark willingness to pay of recreational fishing trip targeting species f. [b] calculated by subtracting the percent change in biomass with feedbacks from the percent change in biomass without feedbacks in the final period, where percent change in biomass is calculated as: $\%\Delta = 100[S_{f,t} - S_{f,0}]/S_{f,0}$, where $S_{f,t}$ is the biomass of species f in period t, and $S_{f,0}$ is the benchmark biomass of species f.

Table A.1: IMPLAN Social Accounting Matrix Sectors
Sectors provided by IMPLAN. The miscellaneous sector is a composite of activities from all other industries.

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

Table A.2: Commercial Fishing Sectors

Sectors disaggregated from IMPLAN Commercial fishing sector.

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

Table A.3: USGS Lake Erie Commercial Landings

Commercial species included in the 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
16	Walleye	271	925	0.02

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

Table A.4: 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

Table A.5: Initial Willingness to Pay Values for Recreational Targeted Species

	Willingness to Pay ($/fish)	
Species	**2002 Dollars**	**2013 Dollars**
Smallmouth Bass	12.86	16.72
White Bass	12.86	16.72
White Perch	2.47	3.21
Yellow Perch	2.47	3.21
Walleye	18.43	23.96
Lake Trout	20.13	26.17
Rainbow Trout	20.13	26.17
Lake Whitefish	2.47	3.21

2002 Values from Besedin et al., (2004)

Table A.6: 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

Table A.7: 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	AR1 MA1	AR3 MA3	AR4 MA2	MA1	AR2 MA4

Standard errors in parentheses

*** p<0.01, ** p<0.05, * p<0.1

Table B.1: 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 Committee Yellow Perch Task Group, 2016)

Table C.1: Sensitivity Analysis Parameter Values

Sensitivity Analysis	Updated Values	Original Values
1: 1.5x Elasticity of Substitution	Yellow Perch & White Bass: 5.85	Yellow Perch & White Bass: 3.90
2: 0.5x Elasticity of Substitution	Yellow Perch & White Bass: 1.95	Yellow Perch & White Bass: 3.90
3: 1.5x Yellow Perch WTP	$4.82	$3.21
4: 0.5x Yellow Perch WTP	$1.61	$3.21
5: 1.5x White Bass WTP	$25.08	$16.72
6: 0.5 White Bass WTP	$8.36	$16.72
7: 1.5x Travel Cost	$47.19	$31.46
8: 0.5x Travel Cost	$15.73	$31.46
9: 3.0x Travel Cost	$94.38	$31.46

Table C.2: Sensitivity Analysis Results. Percent Difference in Household Welfare

Percent difference calculated as [([S1 value] – [S2 value])/S2 value] in final simulation period

	Base	1.5 x Travel Cost	3x Travel Cost
Household 1 (Income < $10K)	-1.30	-1.86	-3.28
Household 2 (Income: $10k - $15K)	-1.78	-2.51	-4.24
Household 3 (Income: $15K-$25K)	-0.67	-0.99	-1.84
Household 4 (Income: $25K - $35K)	-0.61	-0.90	-1.68
Household 5 (Income: $35K - $75K)	-0.42	-0.62	-1.19
Household 6 (Income: $50K - $75K)	-0.26	-0.38	-0.74
Household 7 (Income: $75K - $100K)	-0.31	-0.46	-0.90
Household 8 (Income: $100K - $150K)	-0.25	-0.38	-0.73
Household 9 (Income > $150K)	-0.33	-0.49	-0.95
Total	-0.43	-0.64	-1.22

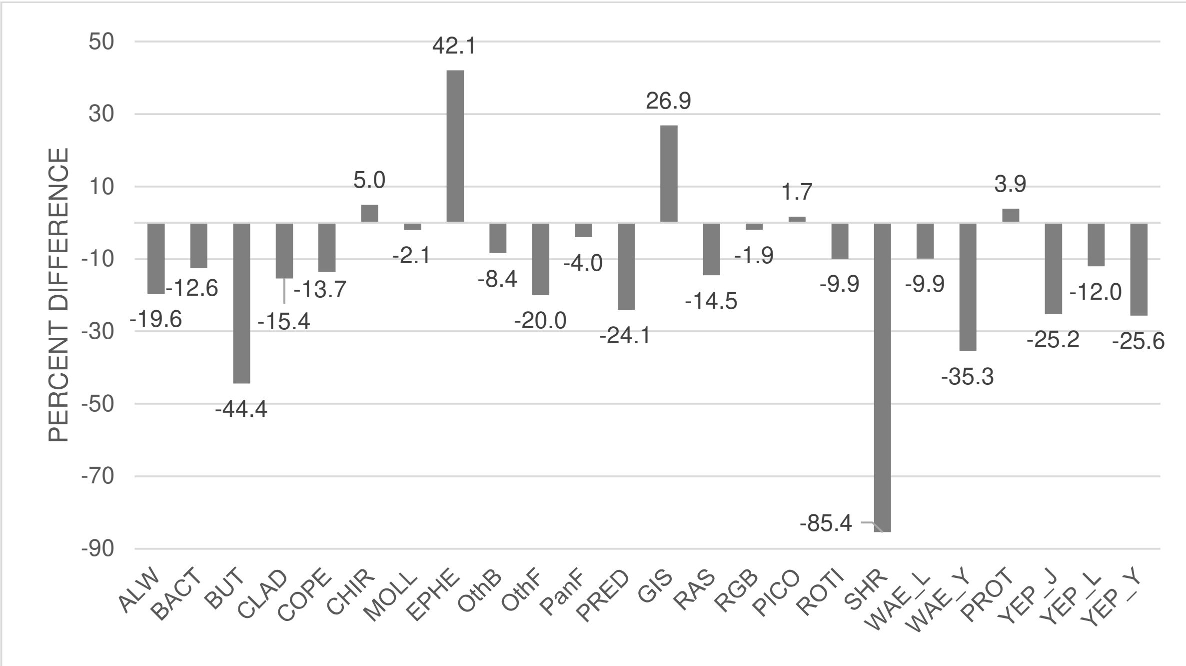

Figure D.1: Percent Difference in Biomass of Non-Target Species in Final Simulation Period
Percent difference calculated as [([S1 value] – [S2 value])/S2 value] in final simulation period. Species with magnitude of percent difference greater than 1% shown. Species name shown in Table D.1

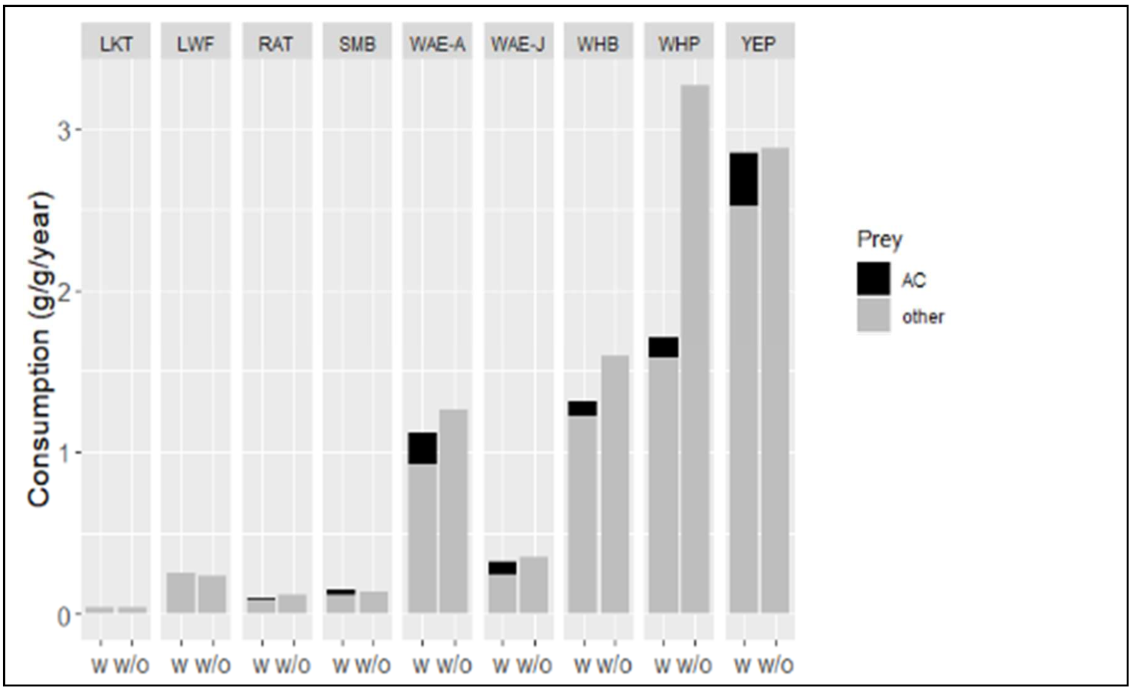

Figure D.2 Diet composition with (w) and without (w/o) Asian carp

Figure was generated based on study of Zhang et al. 2016, which used harvest as fishery forcing while this study used fishing mortality.

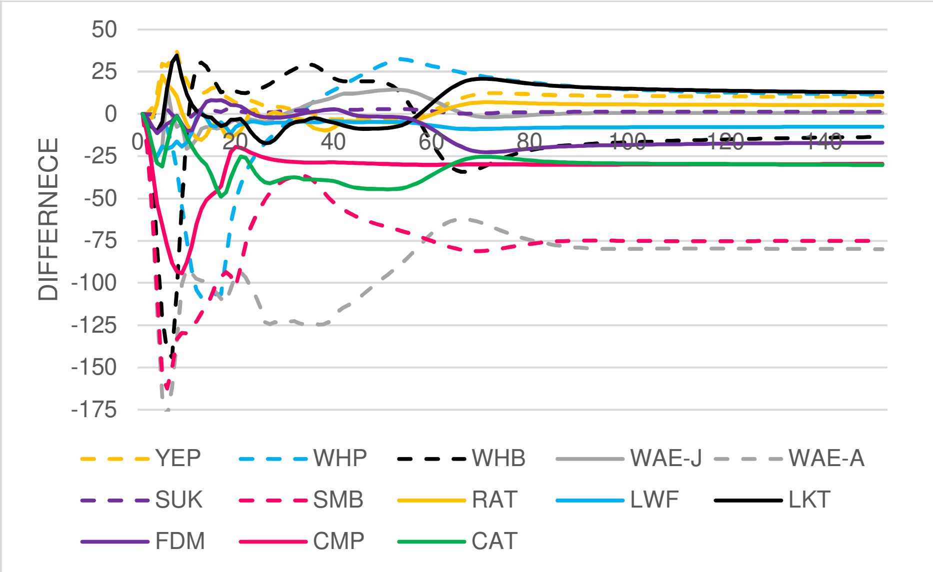

Figure D.3: Difference in Percent Change in Stock Levels in Asian Carp Scenario Projected with and without Feedbacks Over all Simulation Periods

The difference between projected percent changes of biomass levels with-AC when modeled with and without feedbacks between the ecological and economic systems (percent change in biomass without feedbacks – percent change in biomass with feedbacks). Percent Change calculated as: $\%\Delta = 100[S_{f,t} - S_{f,0}]/S_{f,0}$, where $S_{f,t}$ is the biomass of species *f* in period *t*, and $S_{f,0}$ is the benchmark biomass of species *f*. Species names shown in Table D.1.

Table D.1: Species Abbreviations

Abbreviation	Definition
ALW	Alewife *Alosa pseudoharengus*
BACT	Bacteria
BUT	Burbot *Lota lota*
CAT	Channel Catfish *Ictalurus punctatus* and Brown Bullhead *Ameiurus nebulosus*
CHIR	Chironomidae
CLAD	Herbivorous cladocerans
CMP	Common Carp *Cyprinus carpio*
COPE	Calanoida and Cyclopoida
EPHE	Gastropoda, Sphaeriidae, and Bivalvia
FWD	Freshwater drum
GIS	Gizzard Shad
LKT	Lake Trout, adult
LWF	Lake Whitefish
MOLL	Ephemeroptera
OthB	Other benthos (mainly insect larvae)
PanF	Panfish
PICO	Picoplankton
PRED	*Leptodora kindtii*, and spiny water flea *Bythotrephes longimanus*
PROT	Ciliates and heterotrophic flagellates
RAS	Rainbow Smelt *Osmerus mordax*
RAT	Rainbow Trout, adult
RGB	Round Goby *Neogobius melanostomus*
ROTI	Rotifera
SHR	Emerald Shiner *Notropis atherinoides* and Spottail Shiner *Notropis hudsonius*
SMB	Smallmouth Bass
SUK	Suckers
WAE_L	Walleye, larva
WAE_Y	Walleye, age 0
WAE-A	Walleye, adult (age 3C)
WAE-J	Walleye, juvenile (age 1–2)
WHB	White Bass *Morone chrysops*
WHP	White Perch *Morone americana*
YEP	Yellow Perch, adult (age 2C)
YEP_J	Yellow Perch, juvenile (age 1)
YEP_L	Yellow Perch, larval
YEP_Y	Yellow Perch, age 0