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THREE ESSAYS IN ECONOMICS OF PREY-PREDATOR RELATION

by

Dong-Hun Go

A dissertation submitted in partial fulfillment
of the requirements for the degree

of

DOCTOR OF PHILOSOPHY

in

Economics

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2018

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ABSTRACT

Three Essays in Economics of Prey-Predator Relation

by

Dong-Hun Go, DOCTOR OF PHILOSOPHY

Utah State University, 2018

Major Professor: Reza Oladi
Department: Applied Economics

In establishing efficient and sustainable ecosystem management strategies, one of the basic steps requires an integrated approach to identifying the relationship between ecosystem and economy. In this dissertation, each essay presents such an integrated model to examine how the ecosystem consisting of individual representative species works and interacts with a regional economy at the micro-level in the cases of endangered species conservation and environmental shock.

This dissertation includes three essays, focusing on two case studies with respect to particular species in two regions. The second and third chapters (i.e., the first and second essays) cover the first case study in the Pacific Ocean between the United States and Canada, where endangered/threatened predators, as resources in whale watching industries, feed on their single prey, as a commercial species in fishery. The first essay describes how this predator-prey relation is linked with the two-sector regional economy, and then suggests and compares two management regimes for endangered species conservation. An integrated model in the essay is based on a two-factor, two-sector, and two-country general equilibrium framework concerning a predator-prey population dynamics.

Also, concerning the first case study, the second essay attempts to identify the levels of maximum sustainable yield and optimal harvest of the prey in the presence of endangered/threatened predators. Using dynamic optimization frameworks, the chapter also evaluates whether an existing

recovery plan is feasible for the endangered species, and suggests alternative feasible recovery plan scenarios and the corresponding nonmarket values of the species.

The third essay focuses on the second case study of the Great Salt Lake in Utah. A computable general equilibrium model constructed in the essay, integrating the lakes ecosystem and regional economy to identify how individual, aggregate ecological and economic components dynamically and interactively respond to a prolonged drought. In addition, the essay suggests multiple, species-specific regulatory regimes to mitigate the negative effects of drought. It also provides a methodological framework of measuring change in the values of the lakes entire ecosystem services.

(132 pages)

PUBLIC ABSTRACT

Three Essays in Economics of Prey-Predator Relation

Dong-Hun Go

This dissertation explores how natural ecosystem can be integrated with economic system through two case studies of multiple species interactions, or predator-prey relations. By the inclusion of biological, ecological and economic aspects, the integrated approaches aim at more clearly understanding of how regional ecosystem and economy interact with each other, given threats of resource extinction and environmental shock. I also explain strategies and policy regimes that can be considered to achieve efficient and sustainable ecosystem management in those circumstances.

The first case study focuses on a predator-prey relation in the Pacific Ocean between the United States and Canada, where endangered/threatened predators feed primarily on commercially valuable species as prey. Accounting for the importance of those predators as critical natural resources for whale watching industry, this case study synthesizes the species biological and the regional economic systems, and analyzes possible management strategies for both ecosystem conservation and sustainable economic growth.

A long-term drought and fragmented management has been one of the critical issues in the Great Salt Lake (GSL) ecosystem that is linked with its regional economy in Utah. For this issue, the second case study builds an integrated model for describing how the lakes main natural resources, such as water, brine shrimp, and migratory birds, are related to primary industries in the region including agriculture, mining, fishery, and recreation. With the model framework, the study presents how the prolonged drought affects both the GSL ecosystem and its regional economy, and suggests economic management strategies for the lakes ecosystem recovery in the presence of drought.

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Dong-Hun Go

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CHAPTER 1

INTRODUCTION

In the early studies of fisheries economics in the mid-1950s., a bioeconomic approach was first initiated by Gordon (1954) and Scott (1955), who attempted to build a relationship between fishing activities and biological growth. Since then, the bioeconomic approach through mathematical modeling has been extensively applied to ecology and environment and natural resource management. In particular, Sinclair et al. (2002) and van den Bergh et al. (2006) emphasize the importance of expanding the bioeconomic approach to analyze biological and economical interdependencies in the ecosystem.

In the transition from management of a single natural resource to management of interacting natural resources in the ecosystem, the foremost requirement for efficient ecosystem management is the understanding of biological/ecological interaction of resources. One representative example of this interaction can be a predator-prey relationship. Hartwick and Olewiler (1998) provide a well-defined predator-prey relationship as cross-effects meaning that one species benefits from the other while the other species suffers. From an economic perspective, the presence of another species can make harvesting more or less easy because the costs of harvesting a particular species may be affected by the presence of another species. Therefore, ecosystem-based management has become an important issue due to tension between endangered species protection and economic activities, where natural species are differently valued. Interest in the issue has been more rapidly growing, especially in the marine realm and in the saline lakes, in response to recognition of the declining state of fishery productivity and biodiversity.

This dissertation explores how an ecological, biological system can be integrated with an economic system through two case studies of multiple species interactions, or predator-prey relations. The integrated approaches include biological, ecological and economic aspects through multi-species population modeling, dynamic optimization, and general equilibrium modeling. The common objective of these approaches is to understand more clearly how regional ecosystem and

economy interact with each other, given threats of resource extinction and environmental shock. I also study strategies and policy regimes that can be used in achieving efficient and sustainable ecosystem management.

As stated by Anderson and Seijo (2011), managing fisheries with ecosystem considerations involves relevant ecological interdependencies among species along the trophic web. I argue that their statement can be also applied to some types of interacting natural species even in the saline lakes management or the forest management. The authors also suggest future research directions of bioeconomic modeling and analysis in incorporating multispecies and their biological interdependencies: (i) the relevant ecosystem management questions posed to address stock recovery and sustainability strategies within an ecosystem framework, (ii) the biological and economic data availability for parameter estimation of increasingly complex mathematical models required to address the identified relevant questions, and (iii) the assumptions and associated uncertainties of such complex models. I also argue that one of the best suitable modeling approaches is bioeconomic general equilibrium frameworks, the general equilibrium integrated with biological/ecological system.

In this dissertation, two chapters present such bioeconomic general equilibrium models. In traditional general equilibrium frameworks, production sectors conventionally consist of the factors of production, such as capital, labor, and land, where those factor amounts are implicitly assumed to be controlled by economic agents. One of the major distinctions in bioeconomic general equilibrium frameworks is that if at least one of the factors is a natural resource stock, then its amount is not controlled by economic agents but controlled by a natural system. One subsequent question is then how the resource stock size is determined, and possible answers may depend on types of natural resources and interacting relationships between those resources. This distinction includes any type of natural resource dynamic system in the modelling process.

The importance of this integrated modeling approach is also emphasized in a report by Millennium Ecosystem Assessment (MEA, 2005). According to the report, a full assessment of the interactions between people and biodiversity requires a multiscale approach to reflect the multiscale nature of decision-making. The approach allows for examination of driving forces from outside of particular regions, and provides a means of examining the differential impact of changes in biodiver-

sity, ecosystem services, and policy responses on different regions and groups within regions. Thus, my dissertation contributes to the literature by designing a holistic and adaptive natural resource management that bridges between biological/ecological modeling and economic modeling.

The dissertation includes two case studies with respect to particular species in two regions: the Pacific Ocean and the Great Salt Lake. The second and third chapters (or the first and second essays) cover the first case study in the Pacific Ocean between the United States and Canada, where predator (Southern Resident killer whales), as resources in whale watching industries, feed on their primary prey (Chinook salmon), as a commercial species in fishery. The first essay theoretically links this predator-prey population dynamics with a two-sector autarkic economy, and then suggests and compares management policy regimes for endangered species conservation. The policies include an optimal harvest limit (quota), free trade in the prey species, and the open-access scenario as a benchmark. The integrated model in the first essay is based on a two-factor, two-sector, and two-country general equilibrium framework synthesizing a predator-prey population dynamics. The essay studies theoretically and empirically whether free trade in the prey can contribute to the predator population conservation. Also, related to the first case study, the second essay attempts to identify the levels of maximum sustainable yield and optimal harvest of the prey in the presence of endangered/threatened predators. Using dynamic optimization framework, the essay also evaluates whether an existing recovery plan is feasible for endangered species. I conclude that the recovery plan is not feasible. Thus, I suggest alternative feasible recovery plan scenarios and the corresponding nonmarket values of the species.

The third essay deals with the second case study of the Great Salt Lake in Utah. A computable general equilibrium model is constructed by integrating the lakes ecosystem and regional economy in order to identify how individual, aggregate ecological and economic components dynamically and interactively respond to a prolonged drought. In addition, the essay also suggests multiple, species-specific regulatory regimes to mitigate the negative effects of drought, and provides a methodological framework of measuring change in the values of the lakes entire ecosystem services.

The rest of this dissertation is organized as follows: Chapters 2 and 3 cover the first case study

of Resident killer whales and Chinook salmon in the Pacific region. Particularly, Chapter 2 studies those species interaction in a bi-resource economy and the role of free trade by employing a two species (Southern resident killer whales and Chinook salmon) predator-prey population dynamics combined with a general equilibrium framework. In Chapter 3, the two species model is extended to a three species predator-prey system to explicitly account for Northern resident killer whales, and the chapter provides the more detailed study on optimal fisheries management of those species with respect to feasible recovery plan for and nonmarket value of those resident killer whales. Chapter 4 deals with the second case study of the Great Salt Lake ecosystem and its regional economy, and presents the study on the bioeconomic impacts of prolonged drought on the lake ecosystem by using a computable general equilibrium framework. Finally, Chapter 5 summarizes and concludes this dissertation.

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CHAPTER 2

Prey-Predator Interaction in a Bi-resource Economy and the Role of Free Trade

2.1 Abstract

We explore economic and ecological outcomes of an economy consisting of two ecological resources linked via a predator-prey relationship. This bi-resource economy is characterized by production functions that exhibit a tradeoff between prey production for harvest and for predator consumption. Where two bi-resource economies engage in trade, the prey-predator relationship leads to a source of comparative advantage in harvesting prey for a country with a lower proportion of predators to prey. This feature has not been noticed in the literature and leads to an important implication: given a high enough factor of production associated with the predator relative to the prey, free trade leads to conservation of prey and predator stocks in the importing country, making conservation via quota unnecessary. To illustrate our analytic findings, we present the stylized empirical example of the effect of Chinook salmon imports on killer whale populations.

2.2 Introduction

There has been considerable discussion in the economics literature on the relationship between trade and natural resource conservation. The primary focus of this work is on how a natural resource extraction sector interacts with a non-resource sector. For instance, the seminal Brander and Taylor papers (Brander and Taylor 1997a,b; 2008) on this subject present a model with two factors, labor and a natural resource, and two sectors, resource extraction and manufacturing. In this paper, we present an economy based on two interrelated resources such that production functions exhibit a tradeoff based on an environmental relationship. To our knowledge, this type of relationship has not been fully integrated into a general equilibrium framework. In the literature there has been some effort to model how land use affects natural resource production (see for instance Karp et al. (2001), Polasky et al. (2004), Smulders et al. (2004)), but these papers do not fully capture the complex

interactions between natural systems that have become increasingly important in understanding the dynamics of resource behavior.

Examples of resources used by two sectors that exhibit this type of tradeoff occur in groundwater and surface water, water and oil and gas production, and timber and wildlife production. In this paper we explore the specific case of two renewable resources linked via a predator-prey relationship. When predators and prey both have economic value, production functions exhibit a tradeoff between prey production for harvest and for predator consumption (Lester et al. 2013; Marshall et al. 2016). To account for this, we formulate a two-country, two species general equilibrium model where species are specific factors used by two sectors in the economy. The prey-predator relationship leads to a source of comparative advantage in harvesting prey for a country with a lower proportion of predators to prey. This feature has not been noticed in the literature and leads to an important implication: given a high enough factor of production associated with the predator relative to the prey, free trade leads to conservation of prey and predator stocks in the country with a higher proportion of predators to prey, making conservation via quota unnecessary.

The impact of trade on ecological and welfare outcomes has been shown to hinge on three factors: (i) the resource management regime; (ii) the dynamic nature of resource stock adjusting over time to the opposing forces of replenishment and harvesting; and, (iii) issues beyond those directly related to resource extraction, e.g. habitat conversion, non-use values, biodiversity, etc. (Barbier and Bulte 2004). Trade can increase resource overexploitation and may reduce welfare in an economy with open access management of renewable resources (Brander and Talyor 1997a; 1998; Chichilnisky 1994). The type of control over resource extraction is a key determinant of resource health and welfare under trade (Copeland and Talyor 2009), and the incentives to manage a resource are endogenous to the trade setting (Bulte and Damania 2005). Ferreira (2007) shows that a quantitative restriction (quota) on resource extraction makes trade unambiguously beneficial for a small open economy. There has been less focus on factor (iii), despite the importance of understanding the role of free trade in ecosystem outcomes. In this paper we provide a novel synthesis of existing models of multispecies management and international trade. Prior work has focused on the how ecological relationships between predator and prey affect the optimal choice of management

policy (Finnoff and Tschirhart 2003; Hoekstra and van den Bergh 2005; Link et al. 2002; Pikitch et al. 2004). In particular, this literature recognizes the inherent tradeoff between prey harvest and conservation of predator populations. We extend the prey-predator model developed by May et al. (1979) to include a non-consumptive value for the predator. We then compare outcomes of the predator and prey stocks under autarky and free trade, allowing for both open access and quota management regimes.

To illustrate a potential application of our analytic findings, we present a stylized empirical example from fisheries, which stand as some of the world's most valuable natural resources, with exports of wild and farm-raised fish estimated at \$136 billion in 2014 (FAO 2014). Many predators valued in tourism feed on economically valuable fisheries, for instance 590,000 shark watchers yearly spend in excess of \$314 million (Topelko and Dearden 2005). Yet lower shark populations can increase fishery production, for instance declines in apex predators like Blue Sharks in the deep-set Hawaiian longline fishery from 1996-2006 corresponded with increases in valuable commercial fisheries like mahimahi (Polovina et al. 2009). Whale watching is an even larger global industry, estimated to generate more than \$2.5 billion in yearly revenue and about 19,000 jobs (Cisneros-Montemayor et al. 2010). In this paper we examine the case of the Southern Resident Killer Whale (SRKW) and its primary prey, Chinook salmon. Federal spending under the Endangered Species Act (ESA) exemplifies the production tradeoff between prey and predator. In 2014, more than \$2.4 million was spent under the ESA to protect approximately 90 individual SRKW, around \$27,000 per individual (US Fish and Wildlife Service 2014). Chinook salmon, in turn, are a valuable commercial fishery with US harvest worth in excess of \$70 million in 2014 with nine subspecies in the US designated as endangered. ESA spending in 2014 on Chinook salmon exceeded \$181.5 million (US Fish and Wildlife Service 2014). We find the importation of Chinook salmon corresponding to increases in domestic whale populations. Although we do not establish a causal relationship due to data limitations, these results are consistent with the findings of the analytic model.

2.3 Prey-Predator Model

Consider two species where one, prey, serves as the food source for the other, predator. For

instance, a fish preyed upon by a whale. Denote the stocks of fish and whales at time t by $F(t)$ and $W(t)$.¹ Following May et al. (1979) and Beddington and Cooke (1982), the amount of fish preyed upon at any time, t , is given by γFW , where $\gamma > 0$ is the predation parameter. Thus, the dynamics of these species are given by:

$$\dot{F} \equiv \frac{dF}{dt} = g_f - \gamma FW \quad (2.1)$$

$$\dot{W} \equiv \frac{dW}{dt} = g_w \quad (2.2)$$

where g_f and g_w are the natural growth rates of fish and whales, respectively. We maintain that these natural growth rates follow logistic functions, defined as:

$$g_f = r_f F \left(1 - \frac{F}{K}\right) \quad (2.3)$$

$$g_w = r_w W \left(1 - \frac{W}{\tau F}\right) \quad (2.4)$$

where r_f and r_w are intrinsic growth rates of fish and whales, respectively. Moreover, K and τF are the carrying capacities for fish and whale stocks, respectively, where $\tau > 0$ is the proportionality coefficient. The carrying capacity for whales is a linear function of the fish stock. Equations (2.1)-(2.4) fully explain the prey-predator relationship in the natural state. By setting $\dot{F} = \dot{W} = 0$, using equations (2.1)-(2.4), and solving, we obtain the steady-state levels of fish and whales:

$$F = \frac{K}{1 + \nu} \quad (2.5)$$

$$W = \frac{\tau K}{1 + \nu} \quad (2.6)$$

where $\nu \equiv \frac{\gamma \tau K}{r_f}$. An interesting feature of this biological equilibrium is that the equilibrium stock size of the two species is independent of the intrinsic growth rate of the whale stock, r_w . Dividing equation (2.5) by equation (2.6) gives the proportionality coefficient, τ , the relative size of the whale stock to the fish stock, which can also be interpreted as the equilibrium stock ratio when there is no harvesting. This derivation and interpretation of τ is analogous to Flaaten (2012).

¹We drop t whenever it is convenient

2.4 Open Access General Equilibrium

Consider a resource-based economy with two goods: harvested fish and whale-watching recreation, a non-consumptive use. We first formulate the supply side of this economy.

Production functions for fish harvest and whale-watching services are given by:

$$H = \alpha FL_h \quad (2.7)$$

$$S = \beta WL_s \quad (2.8)$$

where H and S are the quantities of harvested fish and whale-watching tourism services, respectively. L_h and L_s denote labor usage in the fishing sector and whale-watching tourism sector. The labor requirement per-unit of output in each sector depends on its production parameter, $\alpha > 0$ and $\beta > 0$ for fish and whales respectively, as well as the resource stock. That is, $a_{Lh} = 1/(\alpha F)$ and $a_{Ls} = 1/(\beta W)$, where a_{Lh} and a_{Ls} denote unit labor requirements in the fishing and whale-watching sectors, respectively. The unit labor requirement in each sector is decreasing in its respective resource stock, with the usual interpretation in the resource economics literature.

Assuming that labor is intersectorally mobile and that the labor market as well as both good markets are competitive, equilibrium conditions require that:

$$P_h = \omega a_{Lh}(F) = \frac{\omega}{\alpha F} \quad (2.9)$$

$$P_s = \omega a_{Ls}(W) = \frac{\omega}{\beta W} \quad (2.10)$$

where P_h and P_s are the harvested fish price and the price of whale-watching tourism services, respectively, and ω is the economy-wide wage rate. Making tourism services the numeraire good, the relative price of fish can be expressed as:

$$p = \frac{\beta W}{\alpha F} \quad (2.11)$$

That is, the relative price of harvested prey is increasing (decreasing) in the stock of predator (prey).

Turning now to the demand side, we assume identical consumers with the following utility function:

$$u(c_h, c_s) = c_h^\varepsilon c_s^{1-\varepsilon} \quad (2.12)$$

where $\varepsilon \in (0, 1)$ and c_h and c_s are the consumption levels of harvested fish and tourism services, respectively.² Moreover, by assuming that each consumer is endowed with one unit of labor, the individual demand functions for fish and tourism services can be obtained as $c_h = \varepsilon\omega/P_h$ and $c_s = (1 - \varepsilon)\omega/P_s$, respectively. Thus, we have the following market demand functions:

$$C_h = \frac{\varepsilon\omega L}{P_h} \quad (2.13)$$

$$C_s = \frac{(1 - \varepsilon)\omega L}{P_s} \quad (2.14)$$

where L is the fixed endowment of labor. Using equations (2.9), (2.10), (2.13), (2.14) and the market clearing conditions, the equilibrium quantities of fish harvest and tourism services at any time are:

$$H = \alpha\varepsilon FL \quad (2.15)$$

$$S = \beta(1 - \varepsilon)WL \quad (2.16)$$

That is, ε is the share of labor used in the fishing sector at any temporal equilibrium.

While equation (2.2) remains valid in this harvesting economy, we have to modify equation (2.1) by incorporating the temporal equilibrium fish harvest (i.e., equation (2.15)) as follows:

$$\dot{F} = g_f - \gamma FW - \alpha\varepsilon FL \quad (2.17)$$

Characterizing the steady-state of this economy is important for understanding the sustainability of the economy and its renewable resources. Using equations (2.2), (2.3), (2.4), and (2.17), and setting

²The consumer's individual maximization problem is: $\max_{c_h, c_s} c_h^\varepsilon c_s^{1-\varepsilon}$, *s.t.* $P_h c_h + P_s c_s = \omega$.

$\dot{F} = \dot{W} = 0$, we obtain the steady state stock levels as:

$$\hat{F} = \frac{K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K} \quad (2.18)$$

$$\hat{W} = \frac{\tau K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K} \quad (2.19)$$

It follows that the corresponding steady-state levels of harvested fish and tourism services will be:

$$\hat{H} = \frac{\alpha \varepsilon L K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K} \quad (2.20)$$

$$\hat{S} = \frac{\beta \tau L K(1 - \varepsilon)(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K} \quad (2.21)$$

Notably, it directly follows from equations (2.11), (2.18) and (2.19) that $\hat{p} = \beta \tau / \alpha$. That is, the steady state relative price is determined by technology factors in the two production sectors (i.e., α and β) and the proportionality coefficient (i.e., τ). The former terms are the economic parameters that determine the long-run price while the latter is a biological term.

Moreover, as is evident from equations (2.18)-(2.21), positive steady-state values of our variables require that $r_f / \varepsilon L > \alpha$. The ratio of service flow from the prey resource stock to labor service flow (i.e., r_f / L_h) can be interpreted as the relative factor service flow (Brander and Taylor 1998). Recall also that $\varepsilon L = L_h$. Thus, we conclude that the necessary condition for positive steady state equilibrium with respect to those two stocks and their corresponding outputs is that the relative factor service flow (i.e., fish relative-factor proportion) should be sufficiently large. This is clearly the case for a fishing economy.³

Table 2.1 shows how any change in each parameter affects our steady-state values \hat{F} , \hat{W} , \hat{p} , \hat{H} , and \hat{S} . For example, “+” in the first row and the second column means $\partial \hat{F} / \partial r_f > 0$, stating that an increase in the intrinsic growth rate of prey increases the steady state level of the prey stock. Brackets shown in the table state conditions for which a sign holds.

³We provide stability analysis of equilibrium in the appendix.

Table 2.1: Comparative Statics of the Steady-State Equilibrium

Parameters	\hat{F}	\hat{W}	\hat{p}	\hat{H}	\hat{S}
r_F	+	+	0	+	+
K	+	+	0	+	+
ε	-	-	0	- (if $r_f K < 2\alpha\varepsilon L$)	-
α	-	-	-	- (if $r_f K < 2\alpha\varepsilon L$)	-
L	-	-	0	- (if $r_f K < 2\alpha\varepsilon L$)	- (if $\alpha\varepsilon L < 1$)
γ	+ (if $r_f K < \alpha\varepsilon L$)	+ (if $r_f K < \alpha\varepsilon L$)	0	+ (if $r_f K < \alpha\varepsilon L$)	+ (if $r_f K < \alpha\varepsilon L$)
β	0	0	+	0	+ (if $r_f K < \alpha\varepsilon L$)
τ	+ (if $r_f K < \alpha\varepsilon L$)	- (if $r_f K < \alpha\varepsilon L$)	+	+ (if $r_f K < \alpha\varepsilon L$)	+ (if $r_f K < \alpha\varepsilon L$)

2.5 Harvesting Control

In the preceding section we demonstrated that the prey stock size plays a key role in the multi-species ecosystem of an economy with interrelated renewable resources. Since some species may face endangerment, conservation is expected to have a critical impact on the ecosystem and the economy. We focus on non-transferable quota on fish harvest as a management tool.

Consider again the supply of harvest and tourism goods. At steady-state, it follows from equations (2.3), (2.15), and (2.17) that the value of harvested good must satisfy:

$$H^{ss} = r_f F^{ss} \left(1 - \frac{F^{ss}}{K}\right) - \gamma \tau F^{ss2} \quad (2.22)$$

where we also used the relationship between the stocks of those two species at steady-state, i.e., $W^{ss} = \tau F^{ss}$. Equation (2.22) states that the quantity of harvested good must be equal to the growth function of the prey minus its interaction with the predator stock (i.e., the quantity that become prey) at any steady-state equilibrium.

Next, using production functions (2.7), (2.8), and labor full employment condition, we can get the production possibility frontier, $H = \alpha FL - (\alpha F/\beta W)S$. Using this production possibility frontier and equation (2.22), we get:

$$S^{ss} = \beta \tau F^{ss} L - \frac{\beta \tau r_f F^{ss}}{\alpha} \left(1 - \frac{F^{ss}}{K}\right) + \frac{\beta \gamma}{\alpha} \tau^2 F^{ss2} \quad (2.23)$$

The quota manager's problem is to maximize the utility function, subject to the steady-state

resource constraints.⁴ Thus, we have to solve $\max_F u(C_h, C_s)$ subject to $C_h = H^{ss}$ and $C_s = S^{ss}$, where these constraints follow from market clearing conditions. As shown in Appendix, the optimal quota on fish stock can be obtained as:

$$F_q = \frac{\alpha L p^2 + r_f(1 - p^2)}{2 \left[\gamma \tau (1 - p) + \frac{r_f}{K} (1 - p^2) \right]} \quad (2.24)$$

where we maintain that $p = \beta \tau / \alpha < 1$ to ensure $F_q > 0$. Moreover, for optimal quota to be binding (i.e., $\hat{F} < F_q$), it follows from equations (2.18) and (2.24) that we must have $\sqrt{2}/2 < \hat{p} = \beta \tau / \alpha < 1$ and $L > r_f / \alpha$. Therefore, we assume for the rest of the paper that $\sqrt{2}/2 < \beta \tau / \alpha < 1$ and $\alpha \varepsilon L < r_f < \alpha L$.⁵ Recall also that at steady-state, we must have $W_q = \tau F_q$. Using equations (2.22)-(2.24), we obtain:

$$H_q = F_q \left[r_f - F_q \left(\frac{r_f}{K} + \frac{\alpha \gamma p}{\beta} \right) \right] \quad (2.25)$$

$$S_q = p F_q \left[\alpha L - r_f + F_q \left(\frac{r_f}{K} + \frac{\alpha \gamma p}{\beta} \right) \right] \quad (2.26)$$

From equations (2.18) and (2.24), and with our previous assumptions that $0 < \varepsilon < 1$ and $\beta \tau / \alpha < 1$, the equilibrium stock size of prey in open access can be compared with the prey stock under quota, F_q . We conclude that the level of open access equilibrium prey stock is less than that of prey stock under quota, i.e., $\hat{F} < F_q$.

As the prey-predator relation is at the heart of our paper and analysis, it is interesting to see how optimal stocks of prey and predators change when the proportionality coefficient τ changes. The following lemma formally addresses this issue. This lemma also helps us to derive our main result in the next section.

Lemma 1. *F_q is monotonically increasing (decreasing) in τ if β / α is sufficiently large (small), assuming that quota is binding.*

⁴see Brander and Taylor (1997b).

⁵Otherwise, we will have uninteresting cases where $\hat{F} = 0$, $F_q = 0$, and/or quota is non-binding for $p = \beta \tau / \alpha \geq 1$. Note also that this type of assumption is not uncommon in the literature (Brander and Taylor 1998)

Proof. Re-write equation (2.24), using the equilibrium relative price, as:

$$F_q = \frac{\Omega(\tau)}{2\Delta(\tau)} \quad (2.27)$$

where $\Omega(\tau) \equiv r_f(1 - \beta^2\tau^2/\alpha^2) + \beta^2\tau^2L/\alpha$ and $\Delta(\tau) \equiv \gamma\tau(1 - \beta\tau/\alpha) + (1 - \beta^2\tau^2/\alpha^2)r_f/K$. By differentiating equation (2.27), we obtain:

$$\frac{\partial F_q}{\partial \tau} = \frac{\Delta \frac{\partial \Omega}{\partial \tau} - \Omega \frac{\partial \Delta}{\partial \tau}}{2\Delta^2}$$

where, after a great deal of simplification, it can be shown that:

$$\Delta \frac{\partial \Omega}{\partial \tau} - \Omega \frac{\partial \Delta}{\partial \tau} = \frac{\beta}{\alpha} \tau \left[\beta L \left(\frac{2r_f}{K} + \gamma \tau \right) + \gamma r_f \left(2 - \frac{\beta}{\alpha} \tau \right) \right] - \gamma r_f,$$

which concludes this lemma. Recall that $1 - (\beta\tau/\alpha) > 0$ if quota is binding, implying that $2 - (\beta\tau/\alpha) > 0$. \square

Note that the relative (marginal) productivity factor in the tourism sector is β/α . The lemma states that if the tourism productivity factor (relative to that of fishing) is sufficiently high initially, then a higher τ will lead to a greater quantitative restriction on fish stock (i.e., a higher fish stock associated with harvest quota). In other words, If the predator stock is relatively higher than the prey stock at the steady state (higher τ), then a social planner would increase the harvest limit on prey stock to make unharvested prey stock more available to the predator. This result is particularly interesting when we compare different countries with different biological systems due to a differing levels of τ . We focus on this biological parameter in the next section when we compare two economies.

2.6 International Trade

Importing fish under free trade may be a perfect substitute for a policy of optimal quota on fish stock, resulting in conservation of the predator. In this section we examine the conditions under which a conservationist country can achieve its objective (i.e., optimal fish stock) by opening the

Table 2.2: Open Access for Home and Foreign

Open Access Home	Open Access Foreign
$\hat{p} = \frac{\beta\tau}{\alpha}$	$\hat{p}^* = \frac{\beta\tau^*}{\alpha}$
$\hat{F} = \frac{K(r_f - \alpha\varepsilon L)}{r_f + \tau\gamma K}$	$\hat{F}^* = \frac{K(r_f - \alpha\varepsilon L)}{r_f + \tau^*\gamma K}$
$\hat{W} = \frac{\tau K(r_f - \alpha\varepsilon L)}{r_f + \tau\gamma K}$	$\hat{W}^* = \frac{\tau^* K(r_f - \alpha\varepsilon L)}{r_f + \tau^*\gamma K}$
$\hat{H} = \hat{F}[r_f - \delta\hat{F}]$	$\hat{H}^* = \hat{F}^*[r_f - \delta^*\hat{F}^*]$
$\hat{S} = p\hat{F}[\alpha L + \delta\hat{F} - r_f]$	$\hat{S}^* = p^*\hat{F}^*[\alpha L + \delta^*\hat{F}^* - r_f]$

country to free trade. Assuming that both the home and foreign countries are large, we first compare the effect of trade on a country with an open-access policy. For the remainder of the paper we use an asterisk to distinguish foreign variables.

Since our focus in this paper is on the prey-predator relation and its effect on equilibrium under various scenarios, we assume that these two countries are similar in all aspects except for the proportionality coefficient, τ . In particular, we maintain that $\tau > \tau^*$. That is, the steady-state ratio of predator to prey stock is greater in the home than the foreign country. For convenience we present steady-state open access equilibrium values at autarky for both countries in Table 2.2, where $\delta \equiv \frac{r_f}{K} + \tau\gamma > \frac{r_f}{K} + \tau^*\gamma \equiv \delta^*$. The following proposition can be drawn from Table 2.2.

Proposition 1. *Assume that $\tau > \tau^*$. Then, (i) $\hat{p} > \hat{p}^*$; (ii) $\hat{F}^* > \hat{F}$; (iii) $\hat{W} > \hat{W}^*$; (iv) $\hat{H}^* > \hat{H}$; and (v) $\hat{S} > \hat{S}^*$.*

Proof. (i) and (ii) immediately follow from $\tau > \tau^*$. To verify validity of (iii), subtract \hat{W}^* from \hat{W} and simplify to obtain:

$$\hat{W} - \hat{W}^* = K(r_f - \alpha\varepsilon L) \left[\frac{(\tau - \tau^*)r_f}{(r_f + \tau\gamma K)(r_f + \tau^*\gamma K)} \right]$$

It follows that $\hat{W} - \hat{W}^* > 0$ since $\tau > \tau^*$. Thus, we conclude that $\hat{W} > \hat{W}^*$. To prove (iv), use the harvest production functions to obtain:

$$\begin{aligned} \frac{\hat{H}}{\hat{F}} &= r_f - \delta\hat{F} = r_f \left(1 - \frac{\hat{F}}{K} \right) - \tau\gamma\hat{F} = \alpha\varepsilon L \\ \frac{\hat{H}^*}{\hat{F}^*} &= r_f - \delta^*\hat{F}^* = r_f \left(1 - \frac{\hat{F}^*}{K} \right) - \tau^*\gamma\hat{F}^* = \alpha\varepsilon L \end{aligned}$$

Implying that $\hat{H}/\hat{F} = \hat{H}^*/\hat{F}^*$, which in turn implies that $\hat{H}/\hat{H}^* = \hat{F}/\hat{F}^*$. However, it follows from (ii) that $\hat{F}/\hat{F}^* < 1$. It then follows that $\hat{H}/\hat{H}^* < 1$, implying that $\hat{H} < \hat{H}^*$.

Finally, to verify that $\hat{S} > \hat{S}^*$. Note that from Table 2.2, the definition of δ and δ^* , and using equation (2.22), we can obtain:

$$\begin{aligned}\hat{S} &= p(\alpha L\hat{F} - \hat{H}) = p\hat{F} \left(\alpha L - \frac{\hat{H}}{\hat{F}} \right) = \frac{\alpha}{\beta} \left(\alpha L - \frac{\hat{H}}{\hat{F}} \right) \hat{W} \\ \hat{S}^* &= p(\alpha L\hat{F}^* - \hat{H}^*) = p\hat{F}^* \left(\alpha L - \frac{\hat{H}^*}{\hat{F}^*} \right) = \frac{\alpha}{\beta} \left(\alpha L - \frac{\hat{H}^*}{\hat{F}^*} \right) \hat{W}^*\end{aligned}$$

where we have also used $p\hat{F} = (\alpha/\beta)\tau\hat{F} = (\alpha/\beta)\hat{W}$, $p^*\hat{F}^* = (\alpha/\beta)\tau^*\hat{F}^* = (\alpha/\beta)\hat{W}^*$ to simplify these expressions. Recall also that $\hat{H}/\hat{F} = \hat{H}^*/\hat{F}^*$. Thus, it follows from (iii) that $\hat{S} > \hat{S}^*$. \square

It is interesting to note that the prey-predator relationship in our setup leads to a source of comparative advantage, a feature that has not been noticed in the literature. Recall that we maintain that countries are identical in all respects except for the prey-predator proportionality coefficient τ . The foreign country, with a lower predator to prey ratio than that of home country, has a comparative advantage in harvesting prey, while the home country has a comparative advantage in tourism services. The lower proportionality ratio is the source of comparative advantage in the harvested good and determines the pattern of trade. Next, we examine the consequences of free trade between these two countries.

Under free trade, the home (foreign) country exports (imports) tourism services and imports (exports) fish. The relative price of fish under free trade will be a convex combination of home and foreign country autarky prices. Then, the relative price under free trade, \hat{p}_T , must be between \hat{p} and \hat{p}^* , i.e., $\hat{p} > \hat{p}_T > \hat{p}^*$. Using $\hat{p} = (\beta\tau/\alpha)$ and $\hat{p}^* = \beta\tau^*/\alpha$ as shown in Table 2.2, \hat{p}_T can be also expressed as:

$$\hat{p}_T = \frac{\beta}{\alpha}\tau_T$$

Table 2.3: Open Access and Free Trade Outcomes

Home	Foreign
$\hat{p}_T = \pi \hat{p} + (1 - \pi) \hat{p}^*$	$\hat{p}_T^* = \hat{p}_T$
$\hat{F}_T = \frac{K(r_f - \alpha \varepsilon L)}{r_f + \tau_T \gamma K}$	$\hat{F}_T^* = \frac{K(r_f - \alpha \varepsilon L)}{r_f + \tau_T \gamma K}$
$\hat{W}_T = \tau \hat{F}_T$	$\hat{W}_T^* = \tau^* \hat{F}_T^*$
$\hat{H}_T = (r_f - \delta \hat{F}_T) \hat{F}_T$	$\hat{H}_T^* = (r_f - \delta^* \hat{F}_T^*) \hat{F}_T^*$
$\hat{S}_T = \hat{p}_T \hat{F}_T (\alpha L - r_f + \delta \hat{F}_T)$	$\hat{S}_T^* = \hat{p}_T^* \hat{F}_T^* (\alpha L - r_f + \delta^* \hat{F}_T^*)$

where $\tau_T \equiv \pi \tau + (1 - \pi) \tau^*$ for some $\pi \in (0, 1)$. Table 2.3 depicts all equilibrium quantities under free trade.⁶

We now turn to the main question of this paper. Can free trade be, at least to some extent, a substitute for prey conservation in fish importing country? The following propositions address the conditions under which trade leads to conservation of the predator:

Proposition 2. *Assume that $\tau > \tau^*$ and that quotas on prey stocks are binding in both countries. If β/α is sufficiently large, then $\hat{F} < \hat{F}_T = \hat{F}_T^* < \hat{F}^* < F_q^* < F_q$.*

Proof. It directly follows from Table 2.3 that $\hat{F}_T = \hat{F}_T^*$. Binding quota implies that (i) $\hat{F}^* < F_q^*$. In addition, p_T being a convex combination of \hat{p} and \hat{p}^* implies that $\hat{p} > \hat{p}_T = \hat{p}_T^* > \hat{p}^*$, which in turn implies from the second rows of Tables 2.2 and 2.3 that (ii) $\hat{F} < \hat{F}_T = \hat{F}_T^* < \hat{F}^*$. Moreover, due to Lemma 1 and since β/α is sufficiently large, $\tau > \tau^*$ implies (iii) $F_q^* < F_q$. Finally, inequalities (i)-(iii) imply that $\hat{F} < \hat{F}_T = \hat{F}_T^* < \hat{F}^* < F_q^* < F_q$ if the conditions of our proposition hold. \square

The ratio β/α is the relative marginal productivity factor in the tourism sector relative to the fishing sector. If the relative marginal productivity factor in the tourism sector is large enough, then substituting home harvested fish with imported fish does not provide enough prey stock conservation for predator consumption, due to the predator's high economic value (i.e., the specific factor of the tourism sector). Recall that in our setup, conservation of prey stock is necessary for protecting the prey from over-harvesting and also for protecting the predator stock. Thus, the above result highlights the conditions under which free trade cannot fully be a substitute for intervention to

⁶Note that τ and τ^* are ecological parameters affecting the carrying capacity of the predator. On the other hand, τ_T is an economic variable consequential to free trade equilibrium price.

conserve the prey and predator. Although free trade does lead to some conservation of both species compared with the open access scenario, it is suboptimal.

The more interesting result is the conditions under which free trade would make unnecessary government intervention in the form of quota:

Proposition 3. *Assume that $\tau > \tau^*$ and that quotas on prey stocks are binding in both countries. If $\varepsilon < 1/2$ and β/α is sufficiently small, then $\hat{F} < F_q < \hat{F}_T = \hat{F}_T^* < \hat{F}^* < F_q^*$.*

Proof. As in the proof of Proposition 2, binding quota implies that $\hat{F} < F_q$. Also, the chain of equalities (ii) is also valid here. Therefore, it is sufficient to show that $F_q < \hat{F}_T$ for $\varepsilon < 1/2$ and sufficiently small β/α . Using equation (2.24) and equation for \hat{F}_T in Table 2.3, we obtain:

$$\hat{F}_T = \frac{K(r_f - \alpha\varepsilon L)}{r_f + \gamma K \tau_T}$$

$$\lim_{\frac{\beta}{\alpha} \rightarrow 0} F_q = \frac{K r_f}{2(r_f + \gamma K \tau)}$$

It follows that $F_q < \hat{F}_T$ for sufficiently small β/α if

$$\lim_{\frac{\beta}{\alpha} \rightarrow 0} F_q - \hat{F}_T = \frac{r_f(r_f + \gamma K \tau_T) - 2(r_f - \alpha\varepsilon L)(r_f + \gamma K \tau)}{2(r_f + \gamma K \tau)(r_f + \gamma K \tau_T)} < 0$$

However, the above inequality holds if $r_f(r_f + \gamma K \tau_T) - 2(r_f - \alpha\varepsilon L)(r_f + \gamma K \tau) < 0$. By rearranging this expression, it becomes $r_f[(r_f + \gamma K \tau) + \gamma K(\tau - \tau_T)] > 2\alpha\varepsilon L(r_f + \gamma K \tau)$. The inequality, $[(r_f + \gamma K \tau) + \gamma K(\tau - \tau_T)] > (r_f + \gamma K \tau)$, holds since $\tau - \tau_T > 0$. For $\varepsilon < 1/2$, we have $r_f > 2\alpha\varepsilon L$ since $r_f > \alpha\varepsilon L$. It then follows that $F_q < \hat{F}_T$ for sufficiently small β/α and $0 < \varepsilon < 1/2$. \square

The above result has an important policy implication. Free trade alone can possibly make quota non-binding and therefore unnecessary. The conditions of these propositions revolve around the notion of the relative (marginal) productivity of factors, β/α , and the consumption share of the harvested good ε . If the harvested good consumption share is sufficiently low (less than one half of the total expenditure) and the relative productivity of tourism is not too large (i.e., the value of specific factor in tourism is not too high), then there will be no need to impose a quota on harvest since the optimal quota will not be binding in any case. Recall that the second condition for this

result can also be stated in terms of labor share used in fishing sector. Accordingly, fishing sector need to be sufficiently small (i.e., share of labor force employed in fishing should be less than half.)⁷

2.7 Case Study: Killer Whales and Chinook Salmon

2.7.1 Overview

We now turn to the stylized example of Chinook salmon and Southern Resident Killer Whales (SRKW) in the northwest United States. In recent years, ecosystem-based fishery management in this region has faced conflicting objectives for conserving both Southern Resident and close relative Northern Resident killer whales (*Orcinus Orca*) and their prey, Chinook salmon (*Oncorhynchus tshawytscha*) (Williams et al. 2011). A decline in the SRKW population led to the species being listed as endangered under the Species at Risk Act in Canada in 2003 and under the Endangered Species Act in the United States in 2005 (Krahn et al. 2004). A number of environmental and economic factors, including the overexploitation of Chinook salmon, construction of dams, vessel noise, and marine pollution, have been suggested as contributing causes. On the other hand, 9 of 26 evolutionarily significant units of Chinook salmon from Washington, Oregon, Idaho, and California are also listed as endangered or likely to become endangered under the Endangered Species Act (Good et al. 2005). There is a great deal of scientific uncertainty on the causal relationship between Chinook salmon abundance, harvest, and the SRKW population (Hilborn et al. 2012). Therefore, the discussion that follows is intended to be illustrative of the model's results, but not a rigorous empirical test.

Chinook salmon are the preferred prey of resident killer whales due to the Chinook salmon's long life span (Healey et al. 1991), year-round availability, body size, and lipid content (Ford and Ellis 2006). Residents are Chinook specialists, with northern residents consuming Chinook for 69% of their identified prey and southern residents 78% (NMFS 2008). While populations overlap, Northern Residents are primarily found in Canadian waters from northern Vancouver Island north, and Southern Residents in US waters from southern Vancouver island to inland Washington State

⁷It can be readily shown that these results will hold for the home economy if it is a small and open economy. Note that under this special case, $\hat{p}_T = p^*$. Then, it follows that $\hat{F} < \hat{F}_T = \hat{F}_T^* = \hat{F}^* < F_q^* < F_q$ in Proposition 2. Similarly, the chain of inequalities in Proposition 3 holds, except for we now have $\hat{F}_T^* = \hat{F}^*$.

(Ford and Ellis 2006). These ranges are illustrated in Figure 2.1. Salmon abundance faces the largest threat in the rivers of the United States, primarily the range of the SRKWs. We exploit the geographical differences in the whale populations to model the impact of trade in salmon on resident killer whales. To motivate our model we use the following stylized facts:

- United States salmon populations affect SRKW populations more than NRKW populations, and potentially only affect SRKW populations (Ford and Ellis 2006; Ford et al. 2000).
- Higher US imports of salmon from Canada lowers demand for domestic salmon and reduces the salmon catch in the US.
- Lower domestic catch leaves more fish for the SRKWs to eat (note: the magnitude of this effect is controversial, see discussion in Hilborn et al. (2012)).
- More abundant salmon increase whale fecundity and populations (Ward et al. 2009).

According to the Center for Whale Research in Washington, it is estimated that the average Southern Resident consumes about 18-25 adult salmon daily to maintain their energy requirements, which means that about 18,000 metric tons of salmon are required each year to sustain the SRKW population.⁸ The US salmon harvest is around 8,400 metric tons annually, and the US imports an additional 2,000-14,400 metric tons of salmon, of which a range of 1,000-7,000 metric tons are imported farmed Chinook from Canada.⁹ Farmed Chinook salmon produced in Canada have no predators, and thus the predator proportionality coefficient, $\tau^* = 0$. The analytic model suggests under this condition,

$$\hat{F} < \hat{F}_T \Rightarrow \hat{W} < \hat{W}_T.$$

that is, the US salmon population will be higher when imports of farmed salmon increase, leading to larger SRKW populations.

⁸ Author's calculation using data from: www.whaleresearch.com/about-salmon

⁹ US farmed Chinook imports from Canada accounts for approximately 97.37% of total US farmed Chinook imports from 1991 to 2014

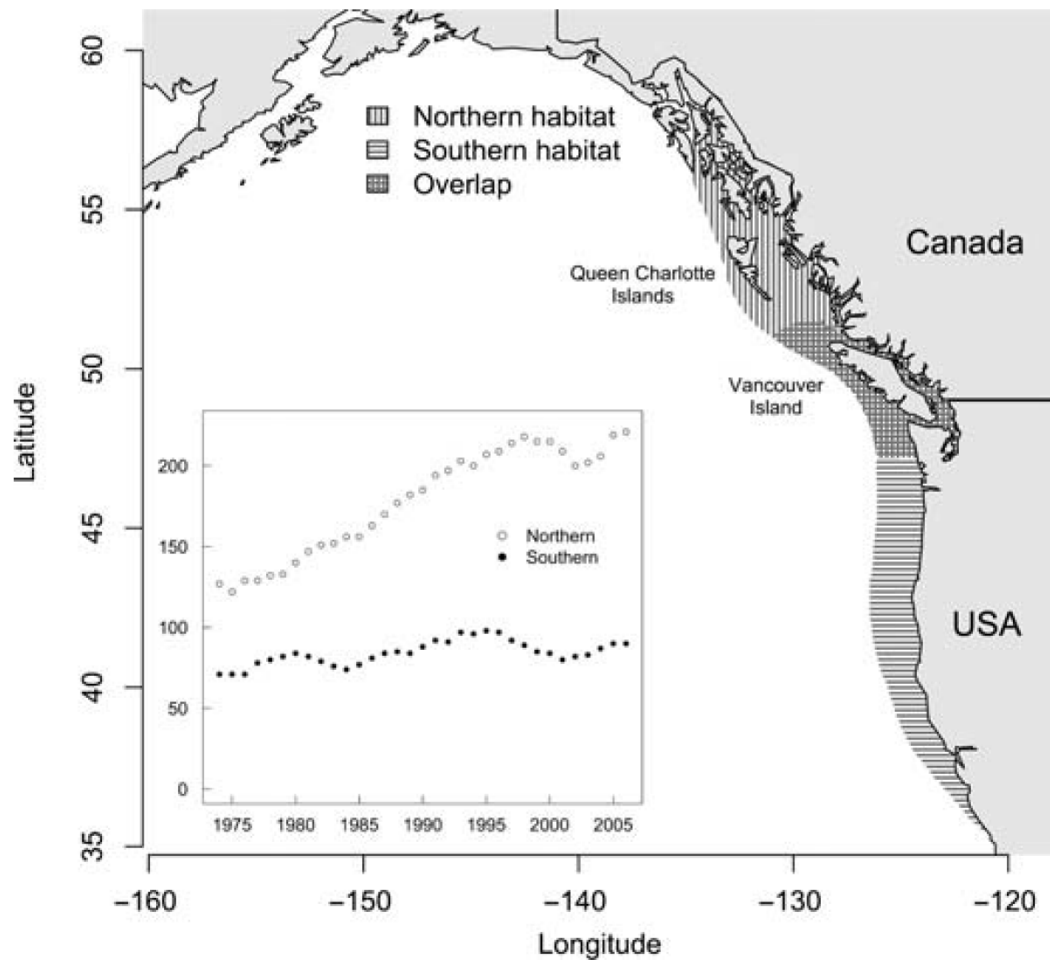


Fig. 2.1: Habitat regions of Southern and Northern Resident Killer Whales; inset whale populations over time. Taken from: Ward et al. (2009).

2.7.2 Setup

We compile data on whale populations, salmon catch, and imports from 1991-2014 to explore how US imports of Chinook salmon are related to the population of SRKW. Annual population estimates for SRKWs from 1991 to 2007 were obtained from the National Marine Fisheries Service (NMFS 2008) and unpublished data for the remaining years (2008-2014) were obtained from the Center for Whale Research.¹⁰ We also obtained unpublished annual average censuses of Northern Resident Killer Whales for 1991 to 2014 from the Pacific Biological Station, Fisheries and

¹⁰<http://www.whaleresearch.com>

Oceans Canada.¹¹ The annual imports to the United States of farmed Chinook salmon from Canada were obtained from the NMFS.¹² The annual domestic landings of Chinook salmon for the US and Canada were also available from the NMFS¹³ and Fisheries and Oceans Canada.¹⁴ Summary statistics for the data are displayed in Table 2.4. The NRKW population is larger throughout the sample period. While the volume of US domestic salmon harvest is larger than farmed Canadian imports throughout the sample, the ratio varies from 0.14 to 0.92. We use this variation to examine the relationship between imports and whale population.

Table 2.4: Summary statistics

Variable	Units	Mean	Std. Dev.	Min.	Max.
SRKW Population	Whales	88	5.40	78	98
NRKW Popualtion	Whales	229	28.02	196.5	288
Farmed Chinook Imports	Metric tons	3,332	1,845.97	976	7,399
Chinook Landings in the US	Metric tons	8,408	2,287.53	4,432	1,2974
Farmed Chinook Import Ratio	-	0.4	0.19	0.14	0.92

Our analysis is motivated by the relationships between resident killer whale populations and farmed Chinook imports from Canada shown in Figure 2.2. The SRKW population seems to be strongly correlated with the farmed Chinook imports from earlier periods. On the other hand, NRKW populations seem to have an upward trend, independent of farmed Chinook imports by the US.

This relationship is as anticipated by the analytic model. To statistically examine the correlation between whale populations and salmon imports, we set up an econometric model consistent with our understanding of the predator-prey system: changes in salmon harvest and imports the period prior affect the current change in whale population. First, we regress the change in population, *POP*, for each whale type separately, on the change in the ratio of farmed imports to domestic harvest, *RATIO*, in the current period and one period prior:

¹¹<http://www.pac.dfo-mpo.gc.ca/science/facilities-installations/pbs-sbp/index-eng.html>

¹²<http://www.st.nmfs.noaa.gov/commercial-fisheries/foreign-trade/applications/annual-product-by-countryassociation>

¹³<http://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index>

¹⁴<http://www.pac.dfo-mpo.gc.ca/fm-gp/species-especies/salmon-saumon/fisheries-peches/stats-donnees-eng.html>

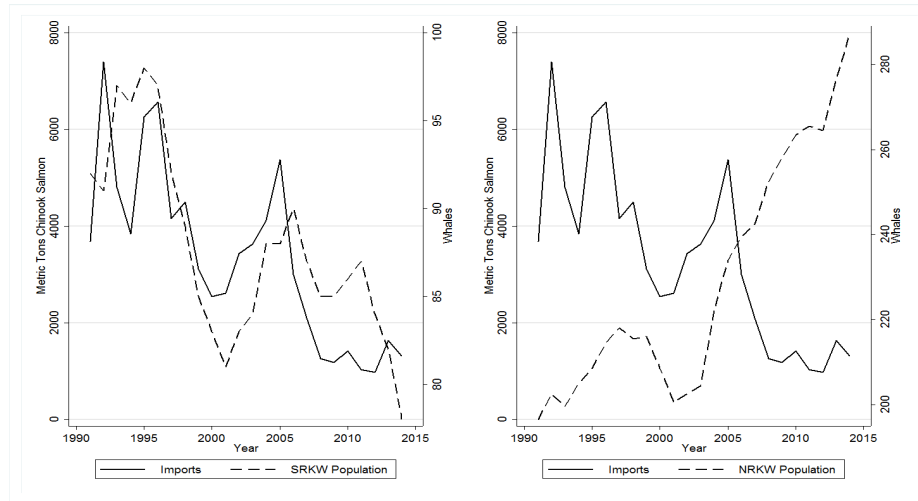


Fig. 2.2: Resident Killer Whale Populations and Farmed Chinook Imports from Canada

$$\Delta POP_t = \beta_0 + \beta_1 \cdot \Delta RATIO + \beta_2 \cdot \Delta RATIO_{t-1} + \tau_t + u_t \quad (2.28)$$

where u_t is the error term. The term τ_t is the unobserved time trend, for instance related to ocean temperature or other biological factors changing over time, which we control with a quadratic term. In the specification in equation (2.28) we expect the SRKW population to grow as the prior period ratio increases, as indicated by a positive coefficient on β_2 . However, we do not expect any significant relationship for the NRKW population. An alternative specification replaces $RATIO$ with just the change in imports, IMP . We also run a specification that uses both the change in imports and the level of US catch, DOM :

$$\Delta POP_t = \gamma_0 + \gamma_1 \cdot \Delta IMP + \gamma_2 \Delta IMP_{t-1} + \gamma_3 \Delta DOM + \gamma_4 \Delta DOM_{t-1} + \tau_t + u_t \quad (2.29)$$

We expect the interaction between the variable on the lagged change in imports to be positive and statistically significant, indicating that an increase in imports in the prior period caused positive growth in SRKW. Because the US salmon harvest is unlikely to affect NRKWs, we expect not to see a significant relationship between NRKW population growth and either US imports or US domestic catch.

2.7.3 Results

The results of our empirical specifications are shown in Table 2.5. The odd specifications use the population of SRKWs as the dependent variable, the even specifications use NRKW population. Specifications (1),(2),(5), and (6) use the ratio of imports to domestic catch, while the others use total quantity of imports. Specifications (5)-(8) also include domestic catch as a control, i.e. equation (2.29). The statistical power of the model is limited due to the small number of observations of whale population. Despite this, results are weakly consistent with there being a positive and statistically significant relationship between the change in salmon imports in the prior period and changes in SRKW population, but not for NRKW population. For instance, specification (2) suggests a 1000 metric ton increase in US salmon imports increases SRWK populations by about one individual, while specification (3) for NRKW shows a smaller, and not statistically significant, effect on NRKW populations.

Table 2.5: Results

VARIABLES	(1) $\Delta SRKW$	(2) $\Delta NRKW$	(3) $\Delta SRKW$	(4) $\Delta NRKW$	(5) $\Delta SRKW$	(6) $\Delta NRKW$	(7) $\Delta SRKW$	(8) $\Delta NRKW$
$\Delta Ratio$	4.350 (5.735)	15.73 (10.24)			3.060 (6.266)	19.01 (12.43)		
$\Delta Ratio_{t-1}$	6.741 (3.954)	8.866 (9.929)			7.435* (4.212)	9.242 (11.25)		
$\Delta Imports$			0.000608 (0.000534)	0.00118 (0.000815)			0.000553 (0.000650)	0.00196 (0.00117)
$\Delta Imports_{t-1}$			0.00101** (0.000444)	0.000794 (0.000906)			0.00101* (0.000494)	0.00104 (0.00115)
$\Delta Domestic$					0.000333 (0.000275)	0.000229 (0.000546)	0.000202 (0.000324)	-0.000519 (0.000569)
$\Delta Domestic_{t-1}$					0.000360 (0.000343)	-0.000278 (0.000464)	-5.36e-05 (0.000355)	-0.000533 (0.000602)
Observations	22	22	22	22	22	22	22	22
R-squared	0.174	0.243	0.312	0.220	0.274	0.249	0.329	0.248
Year Controls	Quadratic	Quadratic	Quadratic	Quadratic	Quadratic	Quadratic	Quadratic	Quadratic

Notes: This table presents results of regressions of the change in population from period $t - 1$ to period t on the change in imports, the ratio of imports to domestic harvest, and domestic harvest from period $t - 2$ to $t - 1$ and from $t - 1$ to t . there are 24 years of data, resulting in 22 observations due to using lagged differences. Robust standard errors in parentheses *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$

Because of the dynamic nature of the predator-prey relationship and limited data availability, establishing a causal relationship is beyond the scope of our work. However, our results are consis-

tent with the killer whale-salmon predator-prey relationship being illustrative of the analytic model. While the relationship between killer whales and salmon is complex, the model suggests that trade may increase both prey and predator stocks in the importing country, provided the whale-to-fish proportionality coefficient is smaller in the exporting country. This prediction is consistent with observed patterns of whale populations, as illustrated by the import of farmed Chinook salmon, for which this proportionality coefficient is zero.

A note of caution is warranted, however. The SRKW coefficient on imports, 0.00101, from specifications (2) and (7) suggests an increase of imports of about 1000 metric tons increases the SRKW population by one animal. Southern residents require about 341,917 to 410,350 Chinook salmon per year at their current populations (Wright et al. 2010), meaning that each resident eats up to 200 metric tons of salmon per year. Thus, the importation of enough fish to feed five whales directly is necessary to increase the population by a single whale through the implied indirect trade mechanism. Thus, while the potential for trade to increase predator populations exists, the mechanism may be problematic for cases where both countries have predator populations. Trade could increase predator populations in the importing country, but reduce these populations in the exporting country. In the case at hand, this is not an issue because the exports are farmed and therefore not available for predator consumption in the exporting country.

2.8 Conclusion

In this essay we demonstrate the effect of trade liberalization on renewable resources and biodiversity by exploring economic and ecological outcomes of a bi-resource economy consisting of two ecological resources linked via a predator-prey relationship. In an illustrative two-country, two species general equilibrium model, the predator species has a non-consumptive value and the prey a consumptive use value, where the predator and prey are specific factors used by the two sectors. The main theoretical result suggests that free trade from a country having a lower ratio of predators to prey leads to conservation of prey and predator stocks for its trading partner, which has a higher proportion of predators to prey. Under certain conditions trade may make conservation via quota unnecessary. This result has not been noticed in the existing literature. To show the practical

relevance of our finding, we present the stylized empirical example of killer whales and Chinook salmon. Statistical results are suggestive that US imports of Chinook salmon may serve to increase the population of Southern Resident killer whales. These findings are indicative of the important insight into the behavior of economies and the outcomes of trade gleaned from incorporating two related natural resource sectors into the general equilibrium framework.

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2.9 Appendix: Stability Analysis

Note that both the prey and predator stocks include the time variable, and thus to find the time path of each of the stocks, $F(t)$ and $W(t)$, the simultaneous steady-state equations mentioned earlier, i.e., $\dot{F} = g_f - \gamma FW - \alpha \varepsilon FL = 0$ and $\dot{W} = g_w = 0$, can be solved by setting up the nonlinear differential equations or the autonomous system. Instead of finding the exact time paths for prey and predator stocks, we try to focus on a qualitative approach, as an approximation, to figure out the dynamic relationship between the stocks of the two species in the open-access general equilibrium setting. The formal autonomous prey-predator system in this model is represented below as follows,

$$\begin{aligned}\dot{F} &= \frac{dF}{dt} = r_f F \left(1 - \frac{F}{K}\right) - \gamma FW - \alpha \varepsilon FL \\ \dot{W} &= \frac{dW}{dt} = r_w W \left(1 - \frac{W}{\tau F}\right)\end{aligned}$$

Demarcation curves for the two stocks, $DF(F, W)$ for the prey and $DW(F, W)$ for the predators respectively, i.e. the nonlinear differential equations for the two stocks at the steady state, are:

$$\begin{aligned}[\dot{F} = 0]; DF(F, W) &= r_f F \left(1 - \frac{F}{K}\right) - \gamma FW - \alpha \varepsilon FL = 0 \\ [\dot{W} = 0]; DW(F, W) &= r_w W \left(1 - \frac{W}{\tau F}\right) = 0\end{aligned}$$

Solving these two equations simultaneously for F and W yields the open-access steady state equilibrium points for the prey and predator stocks, i.e., $(F_A, W_A) = (0, 0)$, $(\frac{K(r_f - \alpha \varepsilon L)}{r_f}, 0)$, and $(\frac{K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K}, \frac{\tau K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K})$. Although the analyses of the first and second equilibrium points may have some important implications, they are trivial here because of our initial assumption that both of those two stock sizes are positive. Thus, we focus on the last equilibrium point. For analytic convenience, we denote the last point by $(a_1, \tau a_1)$. i.e., $(F_A, W_A) = (a_1, \tau a_1)$, where $a_1 = \frac{K(r_f - \alpha \varepsilon L)}{r_f + \gamma \tau K}$. The nonlinear autonomous system can become a linear system by using the Jacobian matrix (the matrix of first derivative of the demarcation curves) and thus the linear approximation at the last point is

summarized by the following matrix equation:

$$\begin{bmatrix} \dot{F} \\ \dot{W} \end{bmatrix} - \begin{bmatrix} r_f - \alpha \varepsilon L - (\frac{r_f}{K} + \gamma \tau) F & -\gamma F \\ \tau r_w \frac{W^2}{F^2} & r_w (1 - \frac{2W}{\tau F}) \end{bmatrix}_{(a_1, \tau a_1)} \begin{bmatrix} F \\ W \end{bmatrix} = \begin{bmatrix} \dot{F} \\ \dot{W} \end{bmatrix} - \begin{bmatrix} 0 & -\gamma a_1 \\ \tau^3 r_w & -r_w \end{bmatrix} \begin{bmatrix} F \\ W \end{bmatrix} = \begin{bmatrix} 0 \\ 0 \end{bmatrix}$$

From the matrix equation above, we denote the Jacobian matrix evaluated at the equilibrium point by J_E , i.e.,

$$J_E = \begin{bmatrix} 0 & -\gamma a_1 \\ \tau^3 r_w & -r_w \end{bmatrix}$$

Under the stability conditions, the determinant of the Jacobian matrix, $|J_E|$ evaluated at $(a_1, \tau a_1)$ is positive, $\gamma \tau^3 a_1 r_w > 0$, if a_1 is positive, which requires $r_f / \varepsilon L > \alpha$. The trace of the matrix, $Tr J_E$, which is the sum of diagonal elements of the Jacobian, is negative, i.e., $-r_w < 0$. Thus, we conclude that the open access steady state point, $(a_1, \tau a_1)$, is stable if $r_f / \varepsilon L > \alpha$, suggesting that any initial point located on the first quadrant in the predator-prey stock plane would eventually approach the steady state point, $(a_1, \tau a_1)$. The question of how an initial point would reach the equilibrium point over time depends on the magnitudes of parameters, which will determine the sign of the term, $(Tr J_E)^2 - 4|J_E|$.

2.10 Appendix: Deriving the optimal quota

The first order condition for the maximization problem of optimal quota is:

$$\frac{\partial u}{\partial C_h} \frac{dC_h}{dF^{ss}} + \frac{\partial u}{\partial C_s} \frac{dC_s}{dF^{ss}} = 0$$

Note that $\frac{\partial u}{\partial C_h} / \frac{\partial u}{\partial C_s} = \frac{1}{p}$ due to representative consumer's problem. Also, recall that at steady state $p = \beta\tau/\alpha$. It can also be readily verified from market clearing conditions and equations (2.22) and (2.23) that:

$$\begin{aligned} \frac{dC_h}{dF^{ss}} &= r_f - \frac{2r_f}{K}F^{ss} - \frac{2\alpha\gamma}{\beta}pF^{ss} \\ \frac{dC_s}{dF^{ss}} &= p\left(\alpha L - r_f + \frac{2r_f}{K}F^{ss} + \frac{2\alpha\gamma}{\beta}pF^{ss}\right) \end{aligned}$$

Thus, the equation of the first order condition, $\frac{\partial u}{\partial C_h} \frac{dC_h}{dF^{ss}} + \frac{\partial u}{\partial C_s} \frac{dC_s}{dF^{ss}}$, can be re-written as:

$$\frac{1}{p} \frac{dC_h}{dF^{ss}} + \frac{dC_s}{dF^{ss}} = \frac{1}{p} \left(r_f - \frac{2r_f}{K}F^{ss} - \frac{2\alpha\gamma}{\beta}pF^{ss} \right) + p \left(\alpha L - r_f + \frac{2r_f}{K}F^{ss} + \frac{2\alpha\gamma}{\beta}pF^{ss} \right) = 0$$

Solving the above equation for F^{ss} yields the following optimal level of fish stock F_q :

$$F_q = \frac{r_f + (\alpha L - r_f)p^2}{2 \left[\left(\frac{r_f}{K} + \gamma\tau \right) - \gamma\tau p - \frac{r_f}{K}p^2 \right]} = \frac{\alpha L p^2 + r_f(1 - p^2)}{2 \left[\gamma\tau(1 - p) + \frac{r_f}{K}(1 - p^2) \right]}.$$

CHAPTER 3

Feasible Recovery Plan under Optimal Fisheries Management: Chinook Salmon and Resident Killer Whale Complex

3.1 Abstract

This essay considers predator-prey relation between threatened Northern Resident and endangered Southern Resident killer whales competitively preying on Chinook salmon. Using the seemingly unrelated regression (SUR) and its iterative form, biological parameters in this particular predator-prey system are estimated to find the maximum sustainable yield (MSY) for Chinook salmon. With the goal of conserving those two killer whales, my result suggests that given my calculated MSY, harvest should be less than 26.59 % of the total catchable Chinook salmon. Furthermore, I trace optimal harvest levels of Chinook salmon in this two-predator-single-prey system and derive non-market values of those two killer whales by using an optimal control framework characterized as bang-bang and singular controls. An existing recovery plan for Southern Resident killer whales is also evaluated in this study by employing an optimal control technique known as the most rapid approach path (MRAP). Main findings suggest that (i) the existing recovery plan of achieving 95 Southern Resident Killer Whales by the year of 2020 may not be feasible due to the constraint of predator-prey system, (ii) there exist feasible recovery plan scenarios for Southern Resident Killer Whale starting from 84 whales as a maximum population size, with conservation of Northern Resident Killer Whale, and (iii) non-market values of Southern Resident Killer Whales depend on the feasible recovery plan scenarios. For example, the value of one additional Southern Resident Killer Whale from 75 to 76 in the recovery plan would be about 5,000\$, which may be the cost that an individual Chinook salmon fishery bears. On the other hand, the value of saving the species from 83 to 84 in the plan would be about 11,000\$.

3.2 Introduction

When threatened/endangered (predator) species feed on commercially valuable (prey) species, fisheries and ecosystem managers would inevitably face a challenging issue: A way of achieving both sustainable harvest on the prey for human and conservation of the predator for biodiversity. Such has been recently the case for two resident killer whales (*Orcinus Orcas*) competitively preying on Chinook salmon (*Oncorhynchus tshawytscha*) in the north-eastern Pacific Ocean. A sharp decline of the southern resident killer whale (SRKW) population, which had occurred during the period of 1996 to 2001, listed the species as endangered under both the Species at Risk Act (SARA) in Canada in 2003 and the Endangered Species Act (ESA) in the United States in 2005.¹ In addition, the northern resident killer whale (NRKW) population also showed its decreasing rate during the similar period of the SRKW's decline, which in effect resulted in the species being listed as threatened under the SARA.

Chinook salmon are the largest species (in body size) of the Pacific salmon including pink, sockeye, chum, and coho. They are the most valuable fish for commercial fishery and subsistence due to their large size and table qualities. Recent Chinook catches in Alaska, for example, have brought fishers nearly 19 million dollars per year.² On the other hand, economic importance of killer whales has also recently increased with an emergence of whale watching industry. The industry in the San Juan Islands alone, for example, has generated 10 million dollars in total for the last few years (Kriete 2007).

Although the biological and geographical interactions among those species have long been recognized and documented by marine scientists (Neil et al. 2014; Vélez-Espino et al. 2015; Ward et al. 2009, 2013), any type of multispecies or bioeconomic models incorporating those three species as well as Chinook salmon fishery has not been developed yet. The amount of Chinook salmon, which serve as the primary prey for those two killer whales' survival, cannot be ignored. In order to sustain the SRKW population, for example, one million salmon a year is required.³ Thus, any single species bioeconomic models excluding this predation effect would overestimate maximum

¹For SARA, see <http://www.cbc.ca/bc/news/bc-081009-killer-whale-recovery-strategy.pdf>.
For ESA, see <http://www.fisheries.noaa.gov/pr/pdfs/fr/fr70-69903.pdf>.

²<https://www.adfg.alaska.gov/static/education/wns/chinook-salmon.pdf>

³<https://www.whaleresearch.com/about-salmon>

sustainable yield (MSY) and optimal levels of Chinook harvest and hence would result in extinction of all of the species considered here.

A common objective of both the ESA and SARA is to identify, protect, and recover the population of imperiled species, although those two acts practically take some different legal approaches. For instance, while comprehensive evaluation on the species' status and socioeconomic factors is conducted by a single national scientific body under the SARA, the ESA does not have such a comparable national body, and it has more limited legal deadlines for listing actions and decisions, which are in turn restricted by the law to consider socioeconomic factors (Waples et al. 2013). According to McLendon's recent report, the ESA alone, for example, protects more than 1,600 species in the United States, along with nearly 700 from other countries.⁴ The report also finds that the performance of the ESA has been a controversial issue and as evidence, of more than 2,300 total listings (including species, subspecies and distinct population segments), only 47 have been delisted from the law. In the case of Southern Resident killer whales, a number of scientific researches have been periodically and mandatorilly performed (by those two acts), and summarized as recovery plans for the species (reported by management institutions from Canada and the US).⁵ However, most of the plans are limited to only two species (Southern Resident killer whales and Chinook salmon) interaction and fishery, not fully accounting for three species interaction including Northern Resident killer whales. This essay, therefore, focuses on evaluating one of those recovery plans. I show that the recovery plan is not feasible. Then, I attempt to find some alternative feasible plans, which is the main overlapping goal of those two acts for the endangered species conservation.

One of the major issues in multiple species management is how much a focal species should be harvested under the condition of sustaining the ecosystem health. Fisheries/ecosystem scientists have attempted to address such an issue by introducing the concept of ecosystem-based maximum sustainable yield. The concept has drawn attention of policy-decision makers in ecosystem and fishery sectors when they observed a growing interest in harvesting multiple species that biologically interact with one another. Several theoretical examples of those multiple harvesting models include May et al. (1979), Beddington and May (1980), and Beddington and Coolke (1982). I extend these

⁴<https://www.mnn.com/earth-matters/wilderness-resources/blogs/endangered-species-act-in-danger>

⁵Detail information of those recovery plans is provided in Section 3.5 in this essay.

frameworks by constraining our three species model. In determining an appropriate level of the ecosystem-based MSY, unharvested species can play an important role. In terms of multispecies bioeconomic model, those species indirectly affecting the focal species harvest has been extensively discussed through tradeoff between commercial fishery and biodiversity conservation. Those unharvested species can be nuisance species (Kasperski 2016), endangered species (Finnoff and Tschirhart 2003) or invasive species (Gutierrez and Regev 2005), and they are sometimes regarded as bycatch (Boyce 1996) and discards (Singh and Weninger 2009).

While multiple species bioeconomic models have been theoretically developed with simulation techniques, empirical methods consistent with those theoretical models have been developed relatively later, especially focusing on estimation of biological parameters which are critical components for analyzing the system of multiple species population dynamics. One of such empirical methods is the seemingly unrelated regression (SUR) with its simple extension, the iterate SUR. Application of the SUR and its iterate form is found in some empirical studies of pelagic fisheries from West Africa (Dedah et al. 1999), and pollock-cod-arrowtooth fisheries in the Bering Sea/Aleutian Islands (Kasperski 2015, 2016). Those two empirical methods are also applied in this study to estimate biological parameters for the purpose of deriving the level of MSY and optimal harvest in three species system, but the main difference between this study and the previous literature is that I focus on the case of two unharvested predators competing for a single harvested prey, more detail description of which will be discussed in the following sections.

The objective of this study is to understand how the Chinook salmon fishery can efficiently and sustainably interacts with the predator-prey environment. In particular, we explore sustainable and optimal harvest levels given the interdependent species population dynamics, and identify the feedback effects of those harvest levels and endangered species' conservation effort on the fishery's profit and species' potential valuation. First, using a set of biological parameters estimated by the seemingly unrelated regression and its iterative form, I derive the ecosystem-based MSY and optimal harvest of Chinook salmon by using an (infinite time horizon) optimal framework given the constraint of conserving those two killer whale populations. Next, I depict a dynamic relationship between the species populations and the Chinook fishery, and derive potential values of

those species. Finally, I evaluate one of the existing recovery plans for Southern Resident killer whales, identify what limitation makes the plan infeasible in the three species predator-prey context, and introduce alternative feasible recovery scenarios and the corresponding non-market values of Southern killer whales. Pursuing those goals, this essay contributes to the literature by providing useful information for sustainable co-management of Chinook salmon and Resident killer whales as well as future recovery plans for endangered species.

The next section introduces the three species predator-prey model, where two unharvested predators competitively and primarily prey on harvested prey, and our optimal control frameworks. In Section 3.4, we discuss biological parameters derived from the seemingly unrelated regression and its iterative form, and economic parameters obtained from various sources. Along with the feasible recovery plan scenarios suggested in this study, our simulated bioeconomic outcomes including the target species' valuation are presented in Section 3.5. The final section summarizes our findings and some limitations of our research.

3.3 Model

We focus on the structure of combined competition and predation in a three species model, where two unharvested predator species are competing for a single harvested prey species. A time invariant level of maximum sustainable yield (MSY) is derived from the model. As a simplest possible management objective, MSY is generally defined as the maximum amount of harvest (or yield) that can be taken from a renewable resource stock. This concept also serves as a reference point in determining whether or not a certain resource stock concerned is being over-harvested before a regulation controlling overexploitation is introduced. Here, we use the concept of MSY to analyze (i) in what time periods Chinook salmon have been over-harvested and (ii) how over-harvesting, if so, would be related to those two killer whale population dynamics. We first introduce theoretical derivation of MSY from our species population model and then, based on the model, we estimate ecological parameters using the seemingly unrelated regression (SUR) and its iterate form (ISUR). Lastly, we present our calculated MSY level, corresponding to the estimated parameter values, the observed harvest levels and species populations.

Three Species Predator-Prey

Consider a single prey (Chinook Salmon) population, N_1 , which serves as the resource for two competing predators (Southern and Northern Resident Killer Whales), N_2 and N_3 respectively. The prey is also harvested at a given level, h . Net growth of each species population is assumed to be logistic. Setting aside the time variable t for simplicity, a simple dynamic predator-prey system is then shown as follows,

$$\dot{N}_1 = \frac{dN_1}{dt} = r_1 N_1 \left(1 - \frac{N_1}{K} \right) - a_{12} N_1 N_2 - a_{13} N_1 N_3 - h, \quad (3.1)$$

$$\dot{N}_2 = \frac{dN_2}{dt} = r_2 N_2 \left(1 - \frac{N_2}{\alpha N_1} \right), \quad (3.2)$$

$$\dot{N}_3 = \frac{dN_3}{dt} = r_3 N_3 \left(1 - \frac{N_3}{\beta N_1} \right), \quad (3.3)$$

where r_i is the intrinsic growth rate of species i , and K is the carrying capacity for the prey population. αN_1 and βN_1 also represent the carrying capacities for the two predator populations, each of which is proportional to the prey population with the proportionality coefficient, α and β . As shown in equation (3.1), the two negative interaction terms, $a_{12} N_1 N_2$ and $a_{13} N_1 N_3$, are included to describe the predation effects by the predators feeding on the single prey at a rate of $a_{12} N_1$ and $a_{13} N_1$, respectively. The predator-prey system of equations (3.1)-(3.3) is my extension of the multispecies harvesting models presented by May et al. (1979), Beddington and May (1980), and Beddington and Cooke (1982). The main difference between my prey-predator system and these three previous studies is that while most of the species were harvested in their papers, only the prey species is harvested in our paper and the predator species should be conserved due to the risk of extinction. Some advantages of using this system is its generality and flexibility of application. For instance, the system can compare a single species harvest with multiple species harvest by simply adding or removing harvest across equations, and also compare a single species dynamics with multiple species dynamics by adding or removing both predation effect and the corresponding predator's population dynamics across equations.

In the steady state ($\dot{N}_i = 0$, $i = 1, 2, 3$), if all of the intrinsic growth rates and species populations are positive, then N_2 and N_3 equals αN_1 and βN_1 respectively from equations (3.2) and (3.3). Substituting αN_1 and βN_1 for N_2 and N_3 in equation (3.1) and setting the steady state condition ($\dot{N}_1 = 0$) yields the following quadratic function in terms of N_1 ,

$$DN_1^2 - r_1 N_1 + h = 0,$$

where $D = r_1/K + a_{12}\alpha + a_{13}\beta$. Thus, N_1 has the following steady-state values,

$$N_1^{SS} = \frac{r_1 \pm \sqrt{r_1^2 - 4Dh}}{2D}. \quad (3.4)$$

With the level of N_1^{SS} from equation (3.4), the steady state values of N_2 and N_3 can be readily derived as follows,

$$N_2^{SS} = \alpha \left(\frac{r_1 \pm \sqrt{r_1^2 - 4Dh}}{2D} \right), \quad (3.5)$$

$$N_3^{SS} = \beta \left(\frac{r_1 \pm \sqrt{r_1^2 - 4Dh}}{2D} \right), \quad (3.6)$$

for $r_1^2 \geq 4Dh$. The focus is on finding the maximum level of h , as a reference point, which would hold the steady state levels of N_1 , N_2 , and N_3 . Such a harvest level is called the ecosystem -based maximum sustainable yield (MSY) and we denote by h_{MSY} . Due to the quadratic functional form of N_1 , N_1 is symmetrically distributed between the two steady state values. Thus, N_1^{SS} has its maximum at the mid-point between the steady state values. In order for N_1^{SS} to be at the mid-point, the root term in the numerator of equation (3.4) should be zero, which eventually results in the level of h_{MSY} as follows,

$$h_{MSY} = \frac{r_1^2}{4D}. \quad (3.7)$$

The values of N_1^{SS} , N_2^{SS} , and N_3^{SS} at h_{MSY} can be readily found by considering the square root term

in each numerator of equations (3.4)-(3.6) as zero. Also, note that N_i^{SS} , $i = 1, 2, 3$, has two positive steady state values, which are identified by simply comparing r_1 with $\sqrt{r_1^2 - 4Dh}$, or r_1^2 with $r_1^2 - 4Dh$, resulting in $r_1 > \sqrt{r_1^2 - 4Dh}$.

Optimal Control

When a single prey is subject to harvest and two threatened and endangered predators, optimal control approach can suggest an ideal level of the prey harvest with conservation of the two predator populations. I first consider prey harvest technology as the following Schaefer production function,

$$h = qN_1E, \quad (3.8)$$

where q is the catchability coefficient and E the fishing effort. With cost function, cE , where c represents the unit cost, the following net profit function can be set up as a function of the prey stock and the fishing effort,

$$\pi(N_1, E) = ph - cE = [pqN_1 - c]E, \quad (3.9)$$

where p is the unit price. With initial conditions on the prey and predator populations (denoted as N_i^0 , $i = 1, 2, 3$), the following optimal control problem is set up by selecting E to maximize the present value of profit PV ,

$$\begin{aligned} \text{Max}_{\{E\}} \quad & PV = \int_0^\infty e^{-\theta t} [pqN_1 - c]E dt \\ \text{subject to} \quad & \dot{N}_1 = G_1(N_1) - a_{12}N_1N_2 - a_{13}N_1N_3 - qN_1E \\ & \dot{N}_2 = G_2(N_1, N_2) \\ & \dot{N}_3 = G_3(N_1, N_3), \end{aligned} \quad (3.10)$$

where $G_1(N_1) = r_1N_1[1 - N_1/K]$, $G_2(N_1, N_2) = r_2N_2[1 - N_1/(\alpha N_2)]$, and $G_3(N_1, N_3) = r_3N_3[1 - N_3/(\beta N_1)]$, and $\theta \in (0, 1)$ is the discount rate. Denoting the co-state variables, λ_1 , λ_2 , and λ_3 , for N_1 , N_2 , and N_3 respectively, the following Current Value Hamiltonian \mathcal{H}_C is presented as,

$$\mathcal{H}_C = [pqN_1 - c]E + \lambda_1[G_1(N_1) - a_{12}N_1N_2 - a_{13}N_1N_3 - qN_1E] + \lambda_2G_2(N_1, N_2) + \lambda_3G_3(N_1, N_3). \quad (3.11)$$

Since the Hamiltonian function is linear in the control variable E with the coefficient of $[pqN_1 - c] - \lambda_1qN_1$, either bang-bang control or singular control can be characterized as the maximum condition with a switching (or on-off) function to replace the condition, $\partial \mathcal{H}_C / \partial E = 0$. Following Ragozin and Brown (1985), the maximum conditions for the fishing effort under the bang-bang and singular controls over any interval are described as follows,

$$E = \begin{cases} E^{max}, & \text{if } \lambda_1 < p - c/qN_1 \\ E^{sv}, & \text{if } \lambda_1 = p - c/qN_1 \\ 0, & \text{if } \lambda_1 > p - c/qN_1, \end{cases} \quad (3.12)$$

where $E = E^{max}$ and $E = 0$ are optimal fishing effort levels derived from the bang-bang control and $E = E^{sv}$ from the singular control. An economic interpretation for this maximum conditions is that, in the case of the bang-bang control, when the shadow price for the prey is higher or lower than the prey fishing firm's net harvest value, then the firm will exert no effort or its maximum effort. In the case of the singular control, in which the shadow price equals the fishing firm's net harvest value, the firm will continue harvesting until it uses up available effort and neither profit nor loss will occur.⁶

Based on Pontryagin's Maximum Principle, the first order necessary conditions for equation (3.11) combined with the co-state variables are,

⁶Similar interpretation is also found in Ragozin and Brown (1985).

$$\dot{\lambda}_1 = \theta \lambda_1 - \frac{\partial \mathcal{H}_C}{\partial N_1} = \left[\theta - \frac{dG_1}{dN_1} + a_{12}N_2 + a_{13}N_3 + qE \right] \lambda_1 - pqE - \lambda_2 \frac{\partial G_2(\cdot)}{\partial N_1} - \lambda_3 \frac{\partial G_3(\cdot)}{\partial N_1} = 0 \quad (3.13)$$

$$\dot{\lambda}_2 = \theta \lambda_2 - \frac{\partial \mathcal{H}_C}{\partial N_2} = \left[\theta - \frac{\partial G_2(N_1, N_2)}{\partial N_2} \right] \lambda_2 + \lambda_1 a_{12} N_1 = 0 \quad (3.14)$$

$$\dot{\lambda}_3 = \theta \lambda_3 - \frac{\partial \mathcal{H}_C}{\partial N_3} = \left[\theta - \frac{\partial G_3(N_1, N_3)}{\partial N_3} \right] \lambda_3 + \lambda_1 a_{13} N_1 = 0 \quad (3.15)$$

$$\frac{\partial \mathcal{H}_C}{\partial \lambda_1} = G_1(N_1) - a_{12}N_1N_2 - a_{13}N_1N_3 - qN_1E = 0 \quad (3.16)$$

$$\frac{\partial \mathcal{H}_C}{\partial \lambda_2} = G_2(N_1, N_2) = 0 \quad (3.17)$$

$$\frac{\partial \mathcal{H}_C}{\partial \lambda_3} = G_3(N_1, N_3) = 0 \quad (3.18)$$

$$\frac{\partial \mathcal{H}_C}{\partial E} = [pqN_1 - c] - \lambda_1 qN_1 = 0. \quad (3.19)$$

Rearranging equation (3.16) in terms of λ yields $\lambda_1 = p - c/(qN_1)$. With this equation of λ_1 and its time derivative ($\dot{\lambda}_1$) together, singular control can arise. The time derivative of λ_1 is,

$$\dot{\lambda}_1 = \frac{c}{qN_1^2} \dot{N}_1 = \frac{c}{qN_1^2} [G_1(N_1) - a_{12}N_1N_2 - a_{13}N_1N_3 - qN_1E] \quad (3.20)$$

From equations (3.17) and (3.18), $N_2 = \alpha N_1$ and $N_3 = \beta N_1$ for positive $N_i, \forall i$. Inserting $N_2 = \alpha N_1$ and $N_3 = \beta N_1$ into equation (3.16) yields E as a function of N_1 as follows,

$$E = \frac{r_1 - (r_1/K + a_{12}\alpha + a_{13}\beta)N_1}{q}.$$

Using $N_2 = \alpha N_1$ and $N_3 = \beta N_1$, $\partial G_2(N_1, N_2)/\partial N_2$ and $\partial G_3(N_1, N_3)/\partial N_3$ in equations (3.14) and (3.15) are $-r_2$ and $-r_3$, respectively. Thus, each of λ_2 and λ_3 eventually becomes a function of λ_1 and N_1 as follows,

$$\lambda_2 = -\frac{\lambda_1 a_{12} N_1}{\theta + r_2} = -\frac{a_{12}(pqN_1 - c)}{q(\theta + r_2)} \quad \text{and} \quad \lambda_3 = -\frac{\lambda_1 a_{13} N_1}{\theta + r_3} = -\frac{a_{13}(pqN_1 - c)}{q(\theta + r_3)}.$$

In equation (3.13), since $r_1 - (2r_1/K)N_1$ comes from dG_1/dN_1 , and αr_2 and βr_3 come from $\partial G_2(N_1, N_2)/\partial N_1$ and $\partial G_3(N_1, N_3)/\partial N_1$ respectively, plugging these partial derivatives into equation (3.13) and then solving simultaneously equations (3.13) and (3.20) by using $N_2 = \alpha N_1$, $N_3 = \beta N_1$, and $\lambda_1 = p - c/(qN_1)$, yields the following quadratic functional form of N_1 ,

$$p\Phi_1 N_1^2 - \frac{c}{q}(\Phi_1 - \Phi_2)N_1 + p(\theta - r_1)N_1 - \frac{\theta c}{q} = 0, \quad \text{where}$$

$$\Phi_1 = \left(\frac{2r_1}{K} + a_{12}\alpha + a_{13}\beta \right) - \left(\frac{a_{12}\alpha r_2}{\theta - r_2} + \frac{a_{13}\beta r_3}{\theta - r_3} \right) \quad \text{and} \quad \Phi_2 = \frac{r_1}{K} + a_{12}\alpha + a_{13}\beta.$$

Under the singular control, solving the above equation for N_1 results in the optimal level of the prey stock and the corresponding levels of other stocks, shadow prices, and fishing effort as,

$$N_1^{sv} = \frac{[(c/q)(\Phi_1 - \Phi_2) + p(r_1 - \theta)] \pm \sqrt{[(c/q)(\Phi_1 - \Phi_2) + p(r_1 - \theta)]^2 + 4p(c/q)\theta\Phi_1}}{2p\Phi_1}$$

$$N_2^{sv} = \alpha N_1^{sv}$$

$$N_3^{sv} = \beta N_1^{sv}$$

$$\lambda_1^{sv} = p - \frac{c}{qN_1^{sv}}$$

$$\lambda_2^{sv} = -\frac{\lambda_1^{sv} a_{12} N_1^{sv}}{\theta + r_2} = -\frac{a_{12}(pqN_1^{sv} - c)}{q(\theta + r_2)}$$

$$\lambda_3^{sv} = -\frac{\lambda_1^{sv} a_{13} N_1^{sv}}{\theta + r_3} = -\frac{a_{13}(pqN_1^{sv} - c)}{q(\theta + r_3)}$$

$$E^{sv} = \frac{r_1 - \Phi_2 N_1^{sv}}{q}$$

The Most Rapid Approach Path (MRAP)

One possible optimal approach technique for infinite-horizon problems in case of endangered species conservation is the most rapid approach path (MRAP), which can be broadly applied if there is an explicit recovery plan for the species. In this study, the target species to be recovered is Southern Resident Killer Whale (as an endangered species), and there is an existing recovery plan with target population size and fixed endpoint in which the plan should be achieved. The MRAP is applied to this case in order to explore how the predator and prey populations, fishing effort, and the corresponding harvest and present value of profit should change over time, along with the

recovery plan to be implemented. Denoting the target population size for N_2 (Southern Resident Killer Whale) by N_2^R and the endpoint by R , we present our optimal framework for the MRAP as follows,

$$\begin{aligned} \text{Max}_{\{E\}} \quad & PV = \int_0^R e^{-\theta t} \pi(N_1, E) dt + \int_R^\infty e^{-\theta t} \pi(N_1, \xi(\dot{N}_1, N_i)) dt \\ \text{subject to} \quad & N_2 \geq N_2^R \quad \text{for } t \geq R \end{aligned} \quad (3.21)$$

where $\pi(N_1, E)$ is defined as in equation (3.9) and $\xi(\dot{N}_1, N_i), \forall i$ and $t \geq R$, is the rearranged equation in terms of E based on the first constraint in equation (3.10), i.e., $\xi(\dot{N}_1, N_i) = [\dot{N}_1 + a_{12}N_1N_2 + a_{13}N_1N_3 - G(N_1)]/qN_1$. Note that in order for the MRAP to be applied, the sufficient condition that $\xi(\dot{N}_1, N_i)$ is additively separable should be satisfied for $N_1(t+1)$ and $N_1(t)$ [?]. Thus, it is readily identified that our application of the MRAP satisfies this sufficient condition.

3.4 Data

The annual dataset of Chinook salmon, SRKW population, and Ocean-type Chinook catches during the period of 1979 to 2010 was available from National Oceanic and Atmospheric Administration (NOAA) Technical Memorandum (Ward et al. 2013). We also obtained (unpublished) annual data of average NRKW population from the Pacific Biological Station, Fisheries and Oceans Canada.⁷ Especially, Chinook salmon population is the indices of aggregated fall Chinook stocks, which are mostly related to those two Resident Killer Whales and are partitioned into Total Ocean Index and Terminal Run. Terminal Run represents the remaining Chinook salmon population after harvest, also called escapement, and Total Ocean Index is simply the sum of Terminal Run and Chinook harvest (For detail, see Ward et al. (2013)). The data is summarized in Table 3.1.

It is shown from Table 3.1 that during the period of 1979 to 2010, on average, Chinook salmon harvest accounts for around 44 % of the total Chinook salmon population (Total Ocean Index) and the remaining Chinook population after harvest (Terminal Run) accounts for approximately 55 % of the total population. The harvest and terminal run also show a high variation during the period. With respect to those two resident killer whale populations, the average number and variation of

⁷the NRKW population data is available by request at <http://www.pac.dfo-mpo.gc.ca/science/facilities-installations/pbs-sbp/index-eng.html>

Table 3.1: Summary Statistics, 1979-2010

Variable	Unit	Mean	Std. Dev.	Min	Max
Total Ocean Index	1,000's fish	2,139.48	604.129	1166.8	3,025.5
Terminal Run	1,000's fish	1,192.3	299.784	849.8	1,952.4
Catches	1,000's fish	947.18	530.083	299.1	1,956.1
Southern Residents	individuals	86.0	6.104	74	98
Northern Residents	individuals	198.8	33.505	140	263.5
Observations			32		

the SRKW population is relatively lower than those of the NRKW population. To understand the relationship between those species populations (and harvest) and the two legislations (the ESA and SARA), Figure 3.1 shows a time series plot of each species population with the specific years of having listed those resident killer whales by the two laws.

Biological Parameter Estimation

Since most of the previous biological studies have focused on the single species population dynamics or the two species population dynamics, i.e., the predator-prey relationship between Chinook and SRKW or between Chinook and NRKW, a set of biological parameters based on equations (3.1)-(3.3) is not available in the literature. In addition, since those three equations are based on the density-dependent models, any biological parameters from either of the two species systems may not be consistent with those from our three species system. A few studies have overcome the issue of simultaneously estimating biological parameters from their multispecies models by using the seemingly unrelated regression (SUR) and its iterate form. The recent examples are found in Kasperski (2015 and 2016). The predator-prey system, as shown in equations (3.1)-(3.3), can be also estimated by single equation models using the ordinary least squares (OLS) method, but it is generally known that although consistent estimates can be resulted from the OLS, they are not as efficient as the SUR, which is based on a feasible generalized least squares (GLS) estimator (Kasperski 2016).

Along with equations (3.1)-(3.3) and the dataset presented in Table 3.1, we present the predator-prey system (in discrete form) to be estimated as follows,

$$\begin{aligned}
N_{OI,t} - N_{OI,t-1} &= r_1 N_{OI,t-1} + \frac{r_1}{K} N_{OI,t-1}^2 + a_{12} N_{OI,t-1} N_{2,t-1} + a_{13} N_{OI,t-1} N_{3,t-1}, \\
N_{2,t} - N_{2,t-1} &= r_2 N_{2,t-1} + \frac{r_2}{\alpha} \frac{N_{2,t-1}^2}{N_{OI,t-1}}, \\
N_{3,t} - N_{3,t-1} &= r_3 N_{3,t-1} + \frac{r_3}{\beta} \frac{N_{3,t-1}^2}{N_{OI,t-1}},
\end{aligned} \tag{3.22}$$

where $t = 1979, \dots, 2010$. The same unobserved environmental and other factors, of course, can affect all of the species populations simultaneously, therefore by including an error term in each equation and allowing those error terms across equations to be correlated, the above system is estimated using the SUR and iterate SUR. It is also expected from the predator-prey estimation system (3.22) that some of the estimated parameters show negative values, i.e., r_1/K , a_{12} , a_{13} , r_2/α , and r_3/β , corresponding with equations (3.1)-(3.3). Such parameter values will be discussed after our result is presented.

While the second and third equations in the system (3.22) correspond to equations (3.2) and (3.3), N_{OI} in the first equation in system (3.22), needs some explanation. N_{OI} represents the Total Ocean Index of the Chinook population. The variable implies the aggregated Chinook population including the harvested amount of Chinook by fisheries. Thus, using the variable N_{OI} from the system (3.22), all of the biological parameters without the event of Chinook salmon harvesting, can be estimated. The biological parameter estimates from the SUR and iterative SUR for the system (3.22) are presented in Table 3.2, where the underlined Chinook, Southern Resident, and Northern Resident represent the first, second, and third equations of the system (3.22).

It is shown from the SUR column of Table 3.2 that the intrinsic growth rate of Chinook, r_1 , is much higher than those for Southern and Northern Resident killer whales, r_2 and r_3 . Specifically, r_1 is about 32 times higher than r_2 and about 39 times higher than r_3 . Also, all of these intrinsic growth rates seem to be statistically significant at 1 % level. On the other hand, the carrying capacity and its components, i.e., K , α , and β , can be readily calculated from the table. For example, since the estimated values of r_1 and r_1/K is given from the table, K is calculated by simply dividing the value of r_1 by that of r_1/K . From this simple calculation, the carrying capacity of (ocean) Chinook

salmon is estimated as about 9,414 thousands of fish, which implies that without human harvesting but with those two killer whales predations, the Chinook salmon population could reach about 9,414 thousands of fish as maximum population size. Based on our equations of the predator-prey system, the carrying capacity of southern resident killer whales depends on the values of α and N_{OI} . Given the value of α from Table 3.2 and the average number of Total Ocean Index from Table 3.1, the carrying capacity of southern resident killer whales without Chinook salmon being intercepted by fisheries could be approximately 109 whales. With the same logic, the carrying capacity of northern resident killer whales could be about 560 whales, which may not be significant since the coefficient of r_3/β seems to be statistically insignificant in Table 3.2. The iterate SUR column has the similar result. The carrying capacity of Chinook salmon from the iterate SUR is a little higher than that from the SUR.

Note that the MSY from this predator-prey system, as presented in equation (3.7), can be calculated by using those estimated parameters. My results suggest that the maximum sustainable yield for Chinook salmon is about 616, 842 fish for the SUR and about 616, 576 fish for the iterate SUR, which accounts for about 26.59 % and 28.81 % respectively of the average number of total ocean Chinook salmon population. Therefore, if the current ecosystem management objective is to conserve and recover these two threatened and endangered resident killer whale populations, then the result implies that the ocean-type Chinook salmon should be harvested less than at least 26.59 % of its given total population on average. Using the estimated parameter values from the SUR and iterate SUR and equations (3.1)-(3.3), we simulate those species populations and estimate the MSY level of Chinook, and compare our simulation and MSY level with the actual numbers of the populations and harvest. Figure 3.2 shows such comparison.

During the period of 1979 to 2010, the simulated curves (dotted lines) seem to follow the overall trends of the actual data. In addition, the last two graphs in this Figure also show the constant level of maximum sustainable yield (MSY, the red dotted line) and the observed values (Actual Harvest in the graph) of Chinook harvest. An interesting feature in the last two graphs is that Chinook harvest had exceeded the MSY level before the year of 1994, the harvest had been seemingly below the MSY between 1995 to 2001, it had been slightly over the MSY between 2002

and 2005, and it has been again below the MSY after 2006. During the overharvesting period, i.e., between 1979 and 1994, the observed Chinook harvest had been substantially declined. It may be due to decrease in the Chinook population or increase in those two killer whale populations that could intercept Chinook salmon from fisheries. During the period of 2000 to 2003 when the observed Chinook harvest levels are roughly below the MSY, those two whale populations are sustained at a constant or an increasing rate, and Chinook population at an increasing rate.

Economic Parameter

The economic parameter values regarded by the optimal control problem in equation (3.10) are obtained from various sources and presented in Table 3.3. I adjust the values of unit cost c and catchability coefficient q in Chinook salmon fishery from Morton et al. (2017) because my base year and measurement unit for Chinook are different from those of Morton et al. For instance, the catchability coefficient q per Chinook salmon was employed as 0.00003 in their study.⁸ However, since our measurement unit for Chinook salmon is thousand fish, this coefficient is adjusted to 0.03 by simply multiplying 0.00003 by thousand. Furthermore, Morton et al. (2017) used a unit cost of 766\$ with 2013 as its base year. For this study, the cost is adjusted to 725.39\$ with base year of 2010 using the GDP Deflator in United States (OECD 2017a). I also employ the discount rate of 5% as in their study.

The value of unit price p is calculated based on various factors including landed values, catches of Chinook, the GDP Deflators, and exchange rates in United States and Canada. Since the common habitat of Chinook salmon and Resident Killer Whales is between Canada and the US, first I take into account annual Chinook catches and landed values for each country. I obtain regional data from British Columbia, Alaska, Washington, Oregon, and California.⁹ By dividing the landed values by Chinook catches for each region, I obtain (annual) unit prices for those five regions. Based on our data available, the Canadian unit price is represented by the Chinook salmon data (catches and landed values) in British Columbia and the US unit price is represented by the combined Chinook

⁸The catchability coefficient was originally estimated from Argue et al. (1983)

⁹Annual data of Chinook catches and values in British Columbia is available at <http://www.pac.dfo-mpo.gc.ca/fm-gp/species-especes/salmon-saumon/fisheries-peches/stats-donnees-eng.html>. Annual data of Chinook catches and values in those four states in the US is available at <https://www.st.nmfs.noaa.gov/commercial-fisheries/commercial-landings/annual-landings/index>.

data in Alaska, Washington, Oregon, and California. However, British Columbia in Canada and the other four states in the US have different measurement units for prices. For instance, a nominal unit price of Canadian Chinook is measured by 1,000 Canadian dollars per ton and that of US Chinook is measured by US dollars per metric ton. Thus, those two countries' nominal prices are adjusted to the real prices by using the GDP Deflators for both countries (OECD 2017a,b), the Canadian real price is again adjusted to the US real price by using exchange rates between those two countries,¹⁰ and then the unit prices for both countries (measured in per ton and per metric ton) are adjusted to a common unit price as US dollars per fish by using an average ocean Chinook caught weights of 4.98 kilogram per fish as shown in Morton et al. (2017). I finally use a weighted average price (US dollars per fish) for those two countries to represent a single unit price of Chinook salmon, following Morton et al. (2017).

3.5 Simulation Results

I first consider the optimal control problem as depicted in equation (3.10) for simulation. Given the predator-prey population dynamics as a constraint and the biological and economic parameters in the previous section, it is important to explore whether and how the variables approach the steady state.¹¹ I also calculate non-market values of these two killer whales using their shadow prices. Next, I describe feasible recovery plan scenarios from using the most rapid approach path, and show how all of the optimal values can change under those scenarios.

Optimal Harvest

I present all of the steady state values for control problem (3.10) in Table 3.4. As stated in the previous section, the MSY levels of Chinook salmon from our SUR and iterative SUR estimation are 616,842 and 616,576 fish respectively. It is suggested from Table 3.4 that the steady state levels of optimal harvest from our SUR and iterative SUR are approximately 414.78 and 415.11 thousand fish, which are roughly two thirds or 67% of each MSY level. In the Table, the optimal (steady state) shadow prices for those two Resident Killer Whales, λ_2^* and λ_3^* , suggests that the marginal net profit

¹⁰Canadian dollars to one U.S. dollar annual exchange rate (not seasonally adjusted) is available at <https://fred.stlouisfed.org/graph/?id=AEXCAUS>.

¹¹I use Solver, a Microsoft Excel add-in program, to find the corresponding optimal values.

of saving one SRKW (NRKW) is about 1,600\$ (122\$), which is the marginal cost that an individual fishing firm bears. As shown in the Table, the shadow price for SRKW is much higher than that for NRKW. For each species population size, the table suggests that, given the initial levels, the Chinook and SRKW populations measured in the year of 1979 would be reduced to 1,312,426 fish and 62 whales respectively in the long run, while the NRKW population would reach 317 whales, which is 2.25 times higher than the initial level.

Next, I simulate optimal trajectories for the variables to explore the transition path for each variable. The time paths of variables with the optimal fishing effort from SUR and iterative SUR are presented in Figures 3.3 and 3.4. Since the present value of profit goes almost to zero in the period 280 (see the graph in the first row and third column of Figure 3.3), all of the time paths are simulated until that period. In the first column of Figure 3.3, a rapid decrease in the Chinook salmon population for some initial periods is due to the high level of initial harvest, which in turn results in rapid decreases in harvest and present value of profit in subsequent early periods. It is interesting to see that while the SRKW population in the second column responds to the decrease in its prey population, the NRKW population does not. It is not known in the biological literature whether NRKW is more competitive predator than SRKW with respect to preying on their common prey, Chinook salmon. One possible explanation is that if it is the case, then the NRKW population could consistently increase since the species is likely able to find and catch its prey faster than its competitor can. The second and third rows of the second column in Figure 3.3 shows dynamic relations between the SRKW and Chinook populations and also between the NRKW and Chinook populations as well. Those two graphs also suggest that while the SRKW population rapidly responds to its prey population at the level of approximately 1,400 thousand Chinook salmon, the NRKW population seems to continuously increase even at the level of roughly 1,000 thousand Chinook.

Feasible Recovery Plan

Following Ward et al. (2013), recovery plan in this essay is defined as quantitative delisting criteria for SRKW as endangered species in terms of population size and endpoint. Several recovery plans are well summarized in Ward et al. (2013). With respect to population size, the simplest and

most explicit recovery plan that can be evaluated in this analysis is the Puget Sound Partnership (PSP)'s goal of reaching a SRKW population size of 95 whales by the year 2020 (PSP 2009). Thus, I apply this recovery plan to the optimal framework of the MRAP as shown in equation (3.21) by setting the target SRKW population size of 95 whales (i.e., $N_2^R = 95$) and the endpoint of 2020 (i.e., $R = 2020$).

However, result of my simulation suggests that the recovery plan proposed by the Puget Sound Partnership is infeasible, given our estimated biological and economic parameters. What makes the plan infeasible is due to a dynamic nature of the three species predator-prey relation as a constraint. A major finding indicated by my simulation analysis is that, given the constrained predator-prey relation, a feasible recovery plan with the endpoint of 2020 can start from a target SRKW population size of 84 whales at most. Any recovery plan with lower population size than this target population level are clearly feasible as well. Based on those simulation results, I establish feasible recovery plan scenarios for SRKW from 84 to 75 whales. Table 3.5 presents those scenarios and the corresponding present values of profit from two different sets of parameters estimated by the SUR and iterative SUR. It is shown from table that the individual present value of net profit from the SUR is a bit higher than that from the iterative SUR for each recovery plan scenario. Moreover, as one additional SRKW is added to the recovery plan, the present value of profit decreases nonlinearly. It is interpreted from the table that if the recovery plan sets its goal of saving the SRKW population from 75 to 84 whales, then an individual fishing firm's present value of profit would be reduced from 82,761\$ to 11,220\$ (based on SUR), or from 82,122\$ to 10,764\$ (based on iterative SUR). This result implies that as more SRKWs under the plan require more Chinook salmon as their prey, a decrease in fishing firm's discounted net profit responds to that predation effect by the firm's forgone fishery.

How does one additional SRKW in the recovery plan change the present value of profit? Based on the envelope theorem, answering this question is the same as finding a value of the shadow price for SRKW, which is in turn a non-market value of the species. To address the question, Table 3.6 presents a list of the shadow prices. It shows that the shadow price (non-market value) of one additional SRKW from 75 to 76 whales in the recovery plan is 5,174.78\$ (based on SUR), or 5,161.69\$ (based on Iterative SUR), measured in a representative salmon fishing firm's forgone

fishery. The table also shows that the shadow prices are different, depending on each recovery plan scenario. In effect, as a higher SRKW population level is set up in the recovery plan, the corresponding shadow price increases, implying that the non-market value of the species gets higher.

Finally, I present the time paths for species populations, harvest, and present value of profit following recovery plan scenarios. These scenarios include three cases of 84, 83, and 82 SRKW population levels that the feasible recovery plan should achieve by the year of 2020. Figures 3.5 and 3.6 from the SUR and iterative SUR shows those three cases. As expected, the first column of Figure 3.5 shows that the recovery plan of 84 SRKWs results in (relatively) the highest population size of Chinook salmon in each period. As a result, the time paths of harvest and net profit under the plan remain the lowest levels. The second column shows that, given MRAP, each of the SRKW populations eventually reach the desired level that each recovery plan should achieve. The dynamic relationship between the SRKW and Chinook salmon populations is also depicted in the second column and row of the Figure. As mentioned earlier, the SRKW population rapidly increases in the Chinook population for the Chinook population less than 1,800 fish, but the SRKW population gradually decreases after that. It is interesting to see the NRKW population. Regardless of the Chinook population and the recovery plan scenarios, the NRKW population shows a continuous increase over time.

3.6 Discussion

In this essay, I explore and identify the levels of maximum sustainable yield and optimal harvest in the threatened and endangered predator and harvested prey system by considering the case of two Resident Killer Whales and Chinook salmon. Within optimal control framework, I also evaluate whether the existing recovery plan for Southern Resident killer whale is feasible given the constrained three species predator-prey system. Next, I explore feasible recovery plans for Southern Resident killer whale and the corresponding non-market value of the species under each feasible recovery plan scenario. Biological parameters for our predator-prey system are estimated from the seemingly unrelated regression and its iterative form, and economic parameters for optimal control framework are obtained from various sources in the literature, such as Chinook salmon fishery and

macroeconomic data in Canada and United States. My main findings suggest that (i) the existing recovery plan of achieving 95 Southern Resident killer whales by the year of 2020 from PSP may not be feasible due to the constraint of predator-prey system, (ii) there exist feasible recovery plan scenarios for Southern Resident killer whale starting from 84 whales as a maximum population size, with conservation of Northern Resident killer whale, and (iii) the non-market values of Southern Resident killer whale depends on the feasible recovery plan scenario and those values are measured in Chinook salmon fishing firm's forgone fishery, or the present value of the firm's profit. For example, the value of one additional Southern Resident killer whale from 75 to 76 in the recovery plan would be about 5,000\$, which may be the cost that an individual Chinook salmon fishery bears. On the other hand, the value of saving the species from 83 to 84 in the plan would be about 11,000\$.

Several recovery plans for SRKW have been suggested in terms of its population size and growth rate. However, the evaluation on recovery plan in this essay is limited only to the population criterion. Evaluation on recovery plan with respect to the species' biological growth rate can be a future extension of this essay. Different predator-prey models may generate different estimates for biological parameters. Again, this could be a subject of future research. Although I account for a biological competition between those two predators for preying on Chinook salmon through the explicit functional form of three species predator-prey system, biological mechanism of why the Northern Resident killer whale population continuously increases regardless of its prey population size and feasible recovery plan scenarios is unknown. This question may be related to whether Northern Resident killer whale is the more competitive predator than Southern Resident killer whale in searching for Chinook salmon. Another limitation of our research is that we do not fully account for some environmental factors affecting those species' habitat, such as ocean temperature and quality, or climate change, which may explain variations in those species' populations. Further works for addressing those issues would also be an interesting extension of this essay.

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3.7 Appendix: Tables

Table 3.2: Biological Parameters from SUR and Iterate SUR

Parameter	SUR	Note	Iterate SUR	Note
<u>Chinook</u>				
r_1	1.478*** (0.427)		1.476** (3.22)	
r_1/K	-0.000157** (0.0000562)	$K = 9,414.01$	-0.000157* (-2.60)	$K = 9,416.17$
a_{12}	-0.0114* (0.00480)		-0.0114* (-2.22)	
a_{13}	-0.000792 (0.00110)		-0.000776 (-0.66)	
<u>Southern Resident</u>				
r_2	0.0458** (0.0147)		0.0458** (3.01)	
r_2/α	-0.972** (0.303)	$\alpha = 0.0471$	-0.971** (-3.10)	$\alpha = 0.0471$
<u>Northern Resident</u>				
r_3	0.0379** (0.0120)		0.0379** (3.05)	
r_3/β	-0.157 (0.0953)	$\beta = 0.2414$	-0.157 (-1.60)	$\beta = 0.2405$
Observations			31	

Standard errors in parentheses

*** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$

Table 3.3: Economic Parameters

Parameters	Values	Description
p	28.7 (\$)	unit price per Chinook salmon
c	725.39 (\$)	unit cost per boat-day
θ	0.05	discount rate
q	0.03	catchability coefficient

Table 3.4: Initial and Optimal Steady State Values

Description		SUR	Iterative SUR
Optimal Chinook Population	N_1^*	1,312.426	1,313.474
Optimal SRKW Population	N_2^*	61.84	61.91
Optimal NRKW Population	N_3^*	316.82	315.83
Optimal Harvest	h^*	414.78	415.11
Optimal Fishing Effort	E^*	10.5348	10.5346
Shadow Price for Chinook	λ_1^*	10.276	10.291
Shadow Price for SRKW	λ_2^*	-1,604.919	-1,615.027
Shadow Price for NRKW	λ_3^*	-121.521	-119.439
Initial Chinook Population	N_1^0		2,493.8
Initial SRKW Population	N_2^0		81
Initial NRKW Population	N_3^0		140

Table 3.5: Feasible Recovery Plan Scenarios and Present Values of Profit for Individual Fishing Firm

SRKWs	SUR Profit (\$)	Iterative SUR Profit (\$)
75	82,761.40	82,121.51
76	77,586.62	76,959.82
77	71,762.76	71,151.26
78	65,269.90	64,675.75
79	58,088.66	57,513.78
80	50,200.87	49,646.92
81	41,587.30	41,055.85
82	32,231.03	31,723.48
83	22,114.47	21,632.02
84	11,220.75	10,764.42

Table 3.6: Valuations on Southern Resident Killer Whale under Feasible Recovery Plan Scenarios

SRKWs	SUR	Iterative SUR
	Marginal Profit (\$)	Marginal Profit (\$)
75-76	5,174.78	5,161.69
76-77	5,823.86	5,808.56
77-78	6,492.86	6,475.51
78-79	7,181.24	7,161.97
79-80	7,887.79	7,866.86
80-81	8,613.57	8,591.07
81-82	9,356.28	9,332.37
82-83	10,116.55	10,091.45
83-84	10,893.73	10,867.60

3.8 Appendix: Figures

Fig. 3.1: Species Populations and Policies

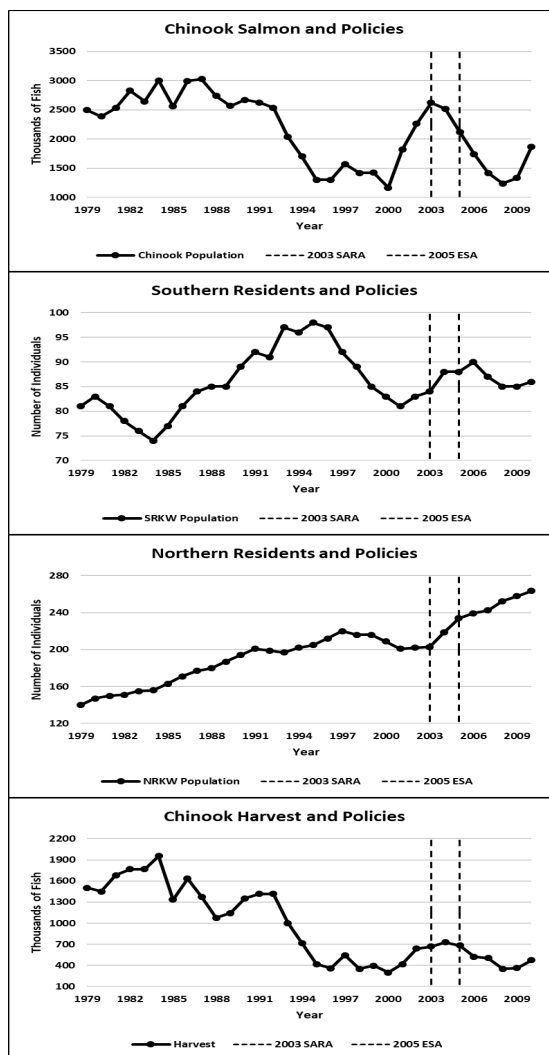


Fig. 3.2: SUR and Iterate SUR

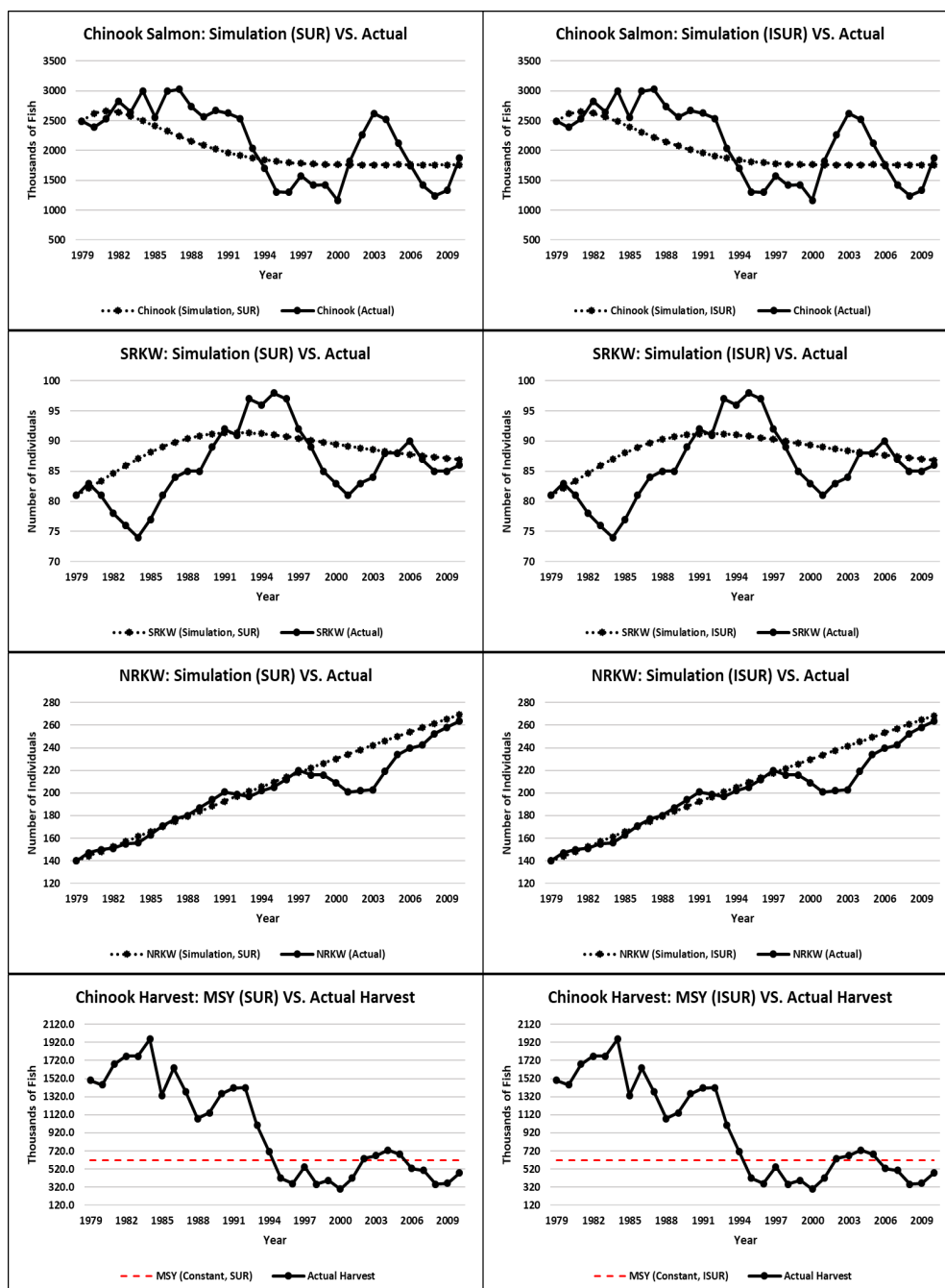


Fig. 3.3: Optimal Time Paths from SUR

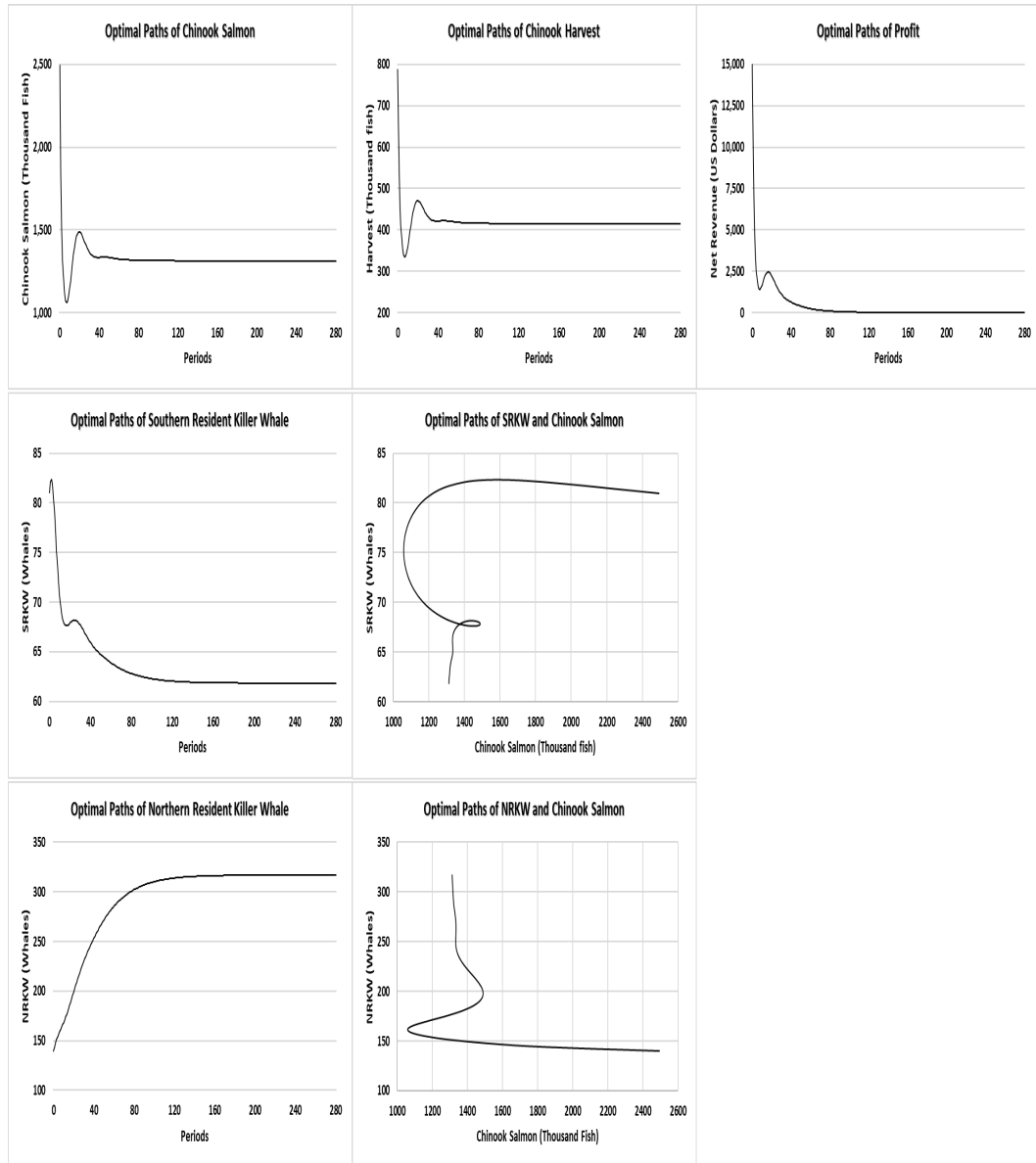


Fig. 3.4: Optimal Time Paths from Iterative SUR

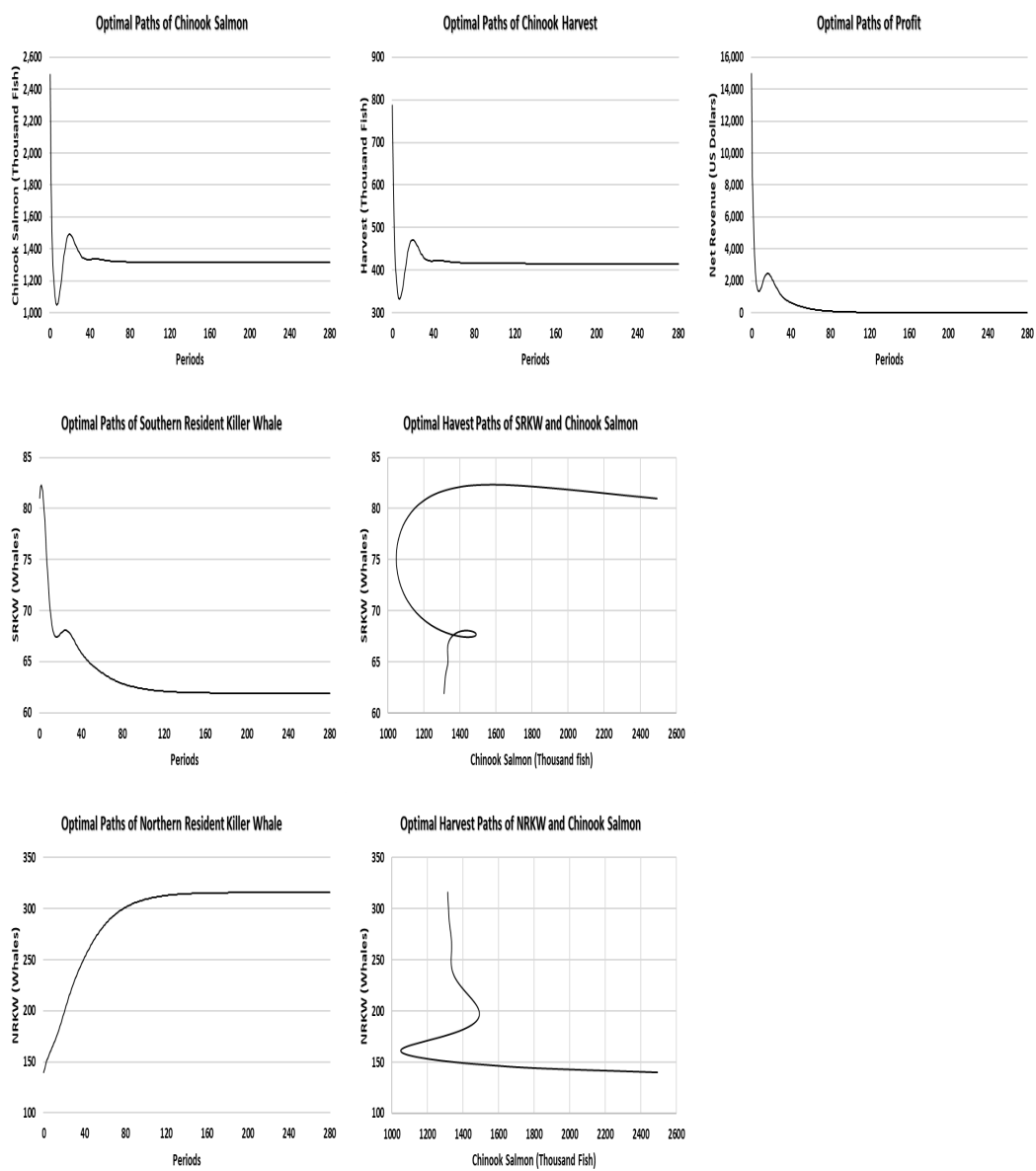


Fig. 3.5: Feasible Recovery Plan Scenarios from SUR

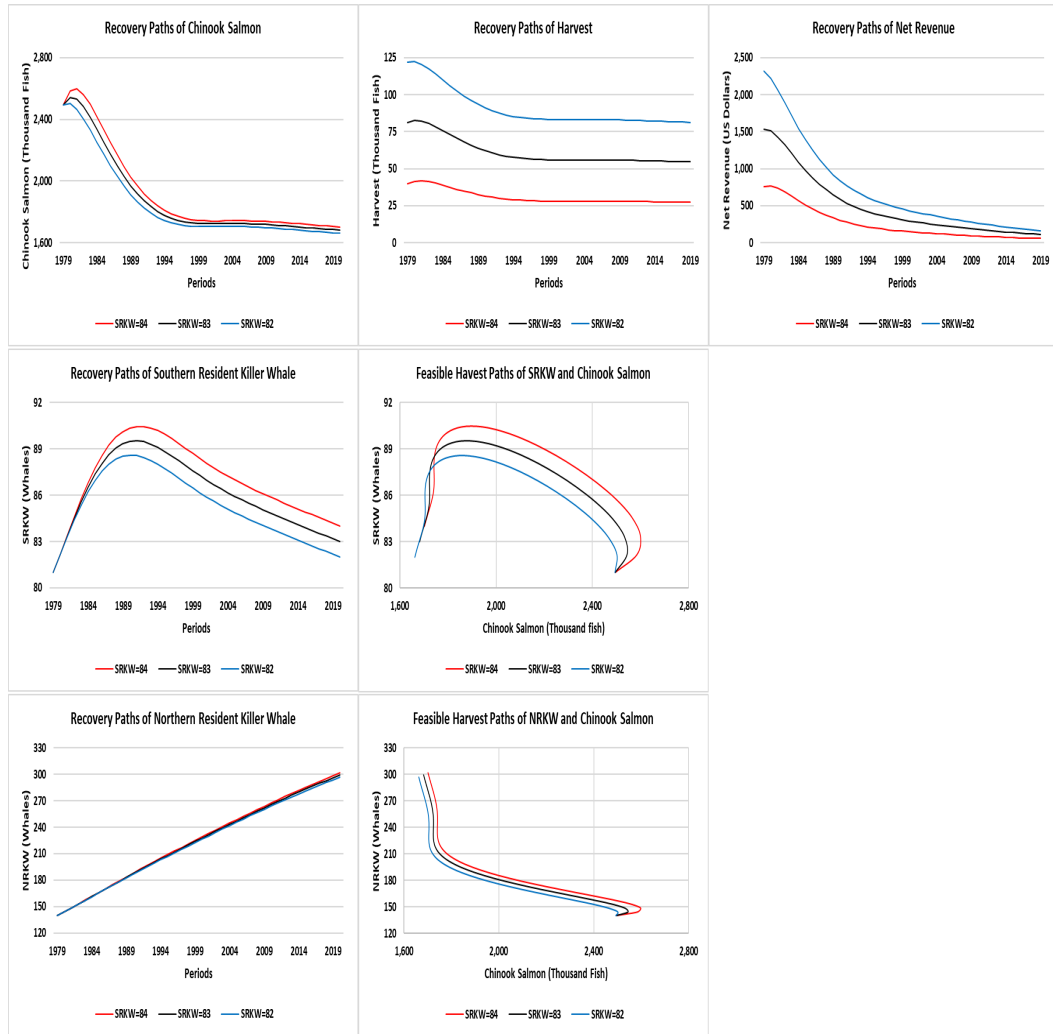
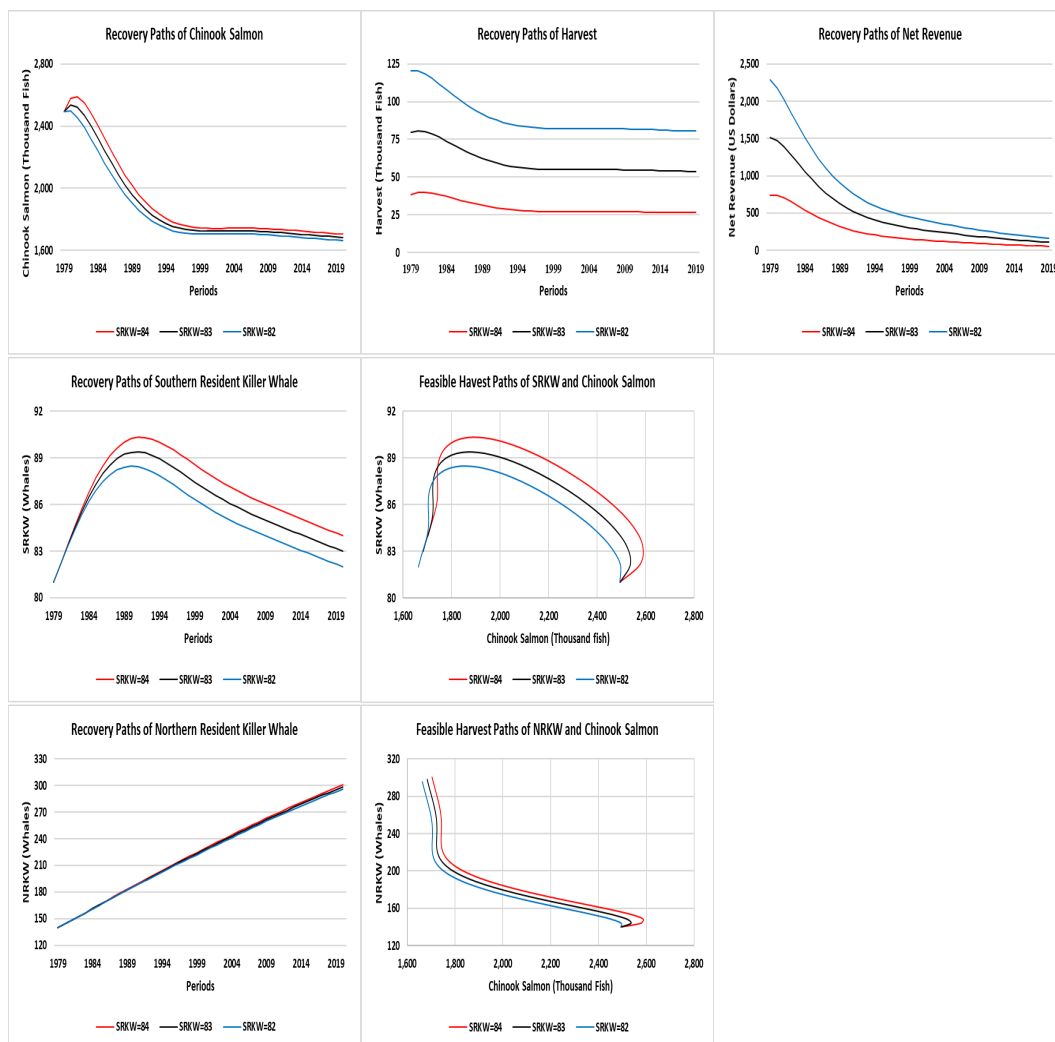


Fig. 3.6: Feasible Recovery Plan Scenarios from Iterative SUR



CHAPTER 4

Measuring the Bioeconomic Impacts of Prolonged Drought on a Lake Ecosystem: The Case of the Great Salt Lake, Utah

4.1 Abstract

We present a general-equilibrium (GE) bioeconomic model of Utah's Great Salt Lake (GSL) ecosystem that tightly links the lake's ecosystem with its regional economy and attendant international trading partners, thereby enabling full identification of the dynamic feedback effects occurring within the GSL bioeconomy in the presence of prolonged drought. The drought modeled here mimics a drying climate's impact on the lake's nutrient pool, which consists of the food web's base species, bacteria. The base species is preyed upon by brine shrimp (the ecosystem's focal species), which in turn are preyed upon by top-predator waterbirds. In this GE bioeconomic framework we demonstrate how (i) prolonged drought affects key bioeconomic variables over time, and (ii) these variables recover toward a new steady state. We also consider how two fishery-specific regulatory tools – a temporary harvest moratorium (quantity regulation) and ad valorem tax on the fishery's factors of production (price regulation) – alter the bioeconomy's recovery path. One key finding is that the two regulatory instruments induce relatively strong recovery effects on the brine-shrimp population, but noticeably weaker impacts on the ecosystem's other species populations. This suggests that multi-species recovery in the presence of prolonged drought requires imposition of multiple, species-specific regulatory instruments, similar to how the control of multiple pollutants require pollutant-specific regulation. Another key finding is that the fishery-specific regulations induce perverse social welfare effects as a consequence of shifting production from the fishery and recreation/wildlife-viewing sectors to mineral extraction, thereby increasing attendant pollution levels. Our model demonstrates how these perverse effects can be mitigated by imposing either an output or input tax on the mining industry in tandem with the fishery-specific regulations.

4.2 Introduction

Effective management of any given bioeconomy requires knowledge rooted in two perspectives.¹ One perspective is micro-orientated, whereby on-going findings from laboratory and field studies of the foraging and reproductive behaviors of an ecosystem's individual plant and animal species continually inform species-specific management processes. The other is more macro-orientated in scope, whereby species interactions (i.e., predator-prey relationships) within the ecosystem's food web (or, ecology) are quantified. General-equilibrium (GE) bioeconomic analysis formally links these two perspectives and extends them by explicitly accounting for the anthropocentric feedback effects within the ecosystem that occur as a consequence of regional economic activities and natural occurrences (such as drought and climate change) that are precipitated and exasperated by these activities.

In this essay we develop a GE bioeconomic model of Utah's Great Salt Lake (GSL) watershed with the overriding goal of demonstrating how the onset of prolonged drought conditions can affect the GSL bioeconomy over time, and how various regulatory policies might serve as management tools to mollify these effects (Figure 4.1 in Appendix C displays the GSL, its freshwater river sources, and geographic boundaries of the counties surrounding the lake). The GSL is representative of a multitude of ecosystems worldwide under persistent threat from prolonged drought.² Here, we adopt Belovsky's ecological framework for the GSL, where the focal species, brine shrimp (*Artemia franciscana*), preys upon cyanobacteria (bacteria) (Belovsky et al. 2011). We also model the broad impacts of changes in the lake's salinity level on its nutrient pool (in particular, on the variable respiration efficiencies of the bacteria existing within the pool) as well as on the lake's brine shrimp population. The salinity level is determined primarily by the extent of endogenously determined agricultural- and mining-sector diversions of the ecosystem's water supply for production purposes, as well as exogenously determined weather conditions such as prolonged drought.³ We follow

¹Strictly speaking, "bioeconomy" refers to an interconnected ecosystem and regional economy. An ecosystem may be located within a widely or narrowly demarcated watershed or ecoregion (whichever the case may be Omernik and Robert (1997)), and the regional economy may likewise span a wide or narrow zone encompassing the ecosystem.

²Solomon (2010) provides several examples, from the Aral Sea watershed in Central Asia to sub-Saharan Africa's Lake Chad to the wide expanses of the Middle East and the world's industrial democracies. Closer to our particular study area in the US Great Basin, ecosystems such as Lake Albert located in southern Oregon (Rosner 2016) and Mono Lake in northern California (Hart 1997) have historically been prone to cyclically severe drought conditions.

³A full schematic of the GSL bioeconomy is presented in Section 4.3.

Finnoff and Caplan (2004) in modeling the GSL regional economy as consisting of four primary production sectors – agriculture, mining, brine shrimp fishery, and wildlife viewing/recreation. Our GE economy model extends the small-country, international trade equilibrium framework developed in Gilbert and Tower (2012) to more than two sectors and two input markets.

A major innovation in this essay is the investigation of two alternative regulatory regimes (one quantity, the other price) that might be used to mitigate the effects of prolonged drought on the GSL bioeconomy. Both regimes target maintenance of the brine shrimp population over time. The quantity regime – a temporary moratorium imposed on brine-shrimp cyst harvesting once the adult shrimp population has fallen beneath a predetermined threshold concentration (henceforth “harvest moratorium”) – mimics the current regulatory policy practiced by the GSL’s regulatory authority. A price regime – ad valorem taxation of labor and capital employed in the brine shrimp fishery (henceforth “input taxation”) – is investigated as a benchmark to which the efficacy of the harvest moratorium is compared. The use of a harvest moratorium as an overt form of regulation is similar to the total allowable catch (TAC) regulatory approach adopted by Finnoff and Tschirhart (2003a,b).⁴ The key difference between the two regulatory approaches is that a TAC is an aggregate harvest limit set prior to the onset of the harvesting season. This limit is then divided into individual transferable harvest quotas (ITQs), which are allocated across individual firms. In contrast, a harvest moratorium is a lower-bound species population threshold that once crossed on the downside during the harvest season triggers a complete shutdown in harvest until the population later crosses the threshold on the upside.

A broader goal of this research is to demonstrate how a relatively simple GE bioeconomic model can be extended to incorporate international trade and alternative regulatory regimes aimed at protecting an ecosystem’s food web in the face of external shocks to the ecosystem and/or regional economy. We consider the model to be a first step in this direction. With respect to the GSL ecosystem in specific, our contribution is the development of an analytical framework to guide management of the GSL bioeconomy in the face of both environmental and economic shocks. The particular shock modeled in this essay is prolonged drought, which has been an on-going concern

⁴As Finnoff and Tschirhart (2003a,b) note, the TAC approach was initially modeled in Homans and Wilen (1997) in the context of a regulated open-access fishery model.

of the GSL regulatory authority for the past several years (UNDR 2013; Wurtsbaugh et al. 2016). However, the GE bioeconomic model is flexible enough to handle any number of different possible shocks, such as sustained climate change and increasing residential and commercial development that encroaches upon the ecosystem's resource base.

We find that the onset of prolonged drought can lead to relatively severe responses in the GSL bioeconomy. In particular, both brine shrimp and waterbird populations decrease in response to the drought, while the corresponding equilibrium biomass-market energy price paid by waterbirds for brine shrimp initially increases and then undershoots its steady-state level on its return to equilibrium. These ecological adjustments accompany decreased shrimp harvests and exports, corresponding reductions in recreation/wildlife viewing, reduced imports of the composite good, and economy-wide shifts in factors of production that lead to increases mineral extraction and its attendant pollution levels. Regional GDP and household welfare both fall in response to the prolonged drought. As expected, both the harvest-moratorium and input-taxation regulations imposed on the brine shrimp fishery cause the ecosystem's full set of ecological variables to recover more quickly from the drought shock to their respective steady-state levels than would otherwise be the case. One key finding is that the two fishery-specific regulations have fishery-specific effects in the face of multiple-species population externalities. This result suggests that multi-species recovery in an ecosystem suffering from prolonged drought requires the imposition of multiple, sector-specific regulatory instruments, similar to how the control of multiple pollutants require pollutant-specific regulation (e.g., separate permit markets for different nonpoint-source pollutants (Lence et al. 1988; Montgomery 1972) and for correlated externalities (Ambec and Coria 2013; Caplan 2006; Caplan and Silva 2005), and separate emissions taxes for different greenhouse gas emissions (Michaelis 2017)).

A second key finding is that the fishery-specific regulations induce perverse social welfare effects akin to the classic general-equilibrium welfare effects associated with environmental regulation found in Hazilla and Kopp (1990), Jorgenson et al. (1992), Pizer and Kopp (2005), and USEPA (2015). What drives this result is the exacerbated shift of production factors out of the brine shrimp fishery and into the mining sector in response to the regulations. This shift ultimately leads to a

decline in regional gross domestic product (GDP) and an increase in the emissions of pollutants, both of which negatively affect social welfare. We show that mining-sector taxation can mitigate this perverse social welfare effect, thus demonstrating that speeding the recovery of an ecosystem in the face of prolonged drought (e.g., by initially regulating the sector of the economy that directly impacts the ecosystem's focal species) may require offsetting regulations imposed on the sectors toward which resources flow in response to initial regulation of the focal species.

Our bioeconomic analysis of the GSL extends two strands of the literature – one concerned with the modeling of multi-species ecosystems in the context of a GE model, the other in what is effectively a partial equilibrium context.⁵ With respect to GE modeling of multi-species ecosystems, our approach is firmly rooted in the theoretical framework pioneered by Tschirhart (2000) and later applied to the Alaskan marine and North Carolina estuary ecosystems (Finnoff and Tschirhart 2003a,b and 2011). Similar to how our model integrates the GSL ecology with the region's economy to inform the ecosystem's management in the face of prolonged drought, Finnoff and Tschirhart (2003a,b) (henceforth FTA) develop an Alaskan marine bioeconomy to inform endangered species policies, where the trade-offs between harvested fish (pollock) and endangered Steller sea lions are examined. Similar to our use of a harvest moratorium to mitigate declines in the GSL's brine shrimp population levels, FTA investigate the role of a recovery plan for endangered sea lions located in Alaska's Aleutian Islands and Eastern Bering Sea that limits the harvest of pollock (the sea lion's prey species) via a TAC (defined as a linear function of each period's pollock population level). FTA's numerical simulations show that reducing sea lion mortality by 50 percent requires a

⁵Our analysis also extends the literature concerned with the social-welfare impacts of environmental regulation in the context of a GE setting. This literature originates with Hazilla and Kopp's estimates of the social costs associated with environmental regulations mandated by the US Clean Air and Water Acts promulgated in the 1970s and 1980s (Hazilla and Kopp 1990). The authors demonstrate that the general-equilibrium impacts of these acts were significant, i.e., that neglecting the secondary, or indirect, costs associated with the regulations resulted in a pronounced understatement of their full cost, to the tune of over \$325 billion (roughly six percent of real GNP) during the 1980s. In their survey of the numerical literature, Pizer and Kopp (2005) report that in aggregate environmental regulations enacted during the 1970s contributed to anywhere from 10 to 40 percent of the productivity slowdowns experienced during that time period. Jorgenson et al. (1992) estimate similar social welfare losses associated with imposition of a (budget-balanced) carbon tax aimed at stabilizing US CO_2 emissions at 1990 levels; losses ranging from roughly \$190 - \$250 billion (1990 dollars), or 0.15 to 0.2 percent of aggregate household wealth for the 30-year period from 1990 to 2020. To the contrary, Nordhaus and Boyer (2000) estimate potential discounted, global net benefits associated with an optimally set global carbon tax of \$3.4 trillion accrued by the end of the 22nd century. As the results of these particular empirical studies suggest, environmental regulation may foster negative welfare effects in a GE economy, similar to the simulated negative effect we find for the GSL bioeconomy for the case of fishery-specific regulation in the presence of prolonged drought.

slightly larger percentage reduction in pollock harvest (by 51.2 percent). The authors also estimate a relatively inelastic response in pollock population size to incremental tightening of the TAC limit.⁶

The key difference between our framework and that of FTA is the extent to which the regional economy is represented in the two models. FTA isolate the pollock fishery as the sole sector affecting (and therefore affected by) the Alaskan marine bioeconomy, while our model explicitly accounts for the feedback effects of multiple sectors simultaneously, including international trade in the harvested focal species. Finnoff and Tschirhart's (2011) (henceforth FTE's) bioeconomic model of agricultural runoff in North Carolina's Neuse Estuary is more closely aligned with our GSL model in this respect, i.e., in terms of accounting for multi-sectoral impacts in the regional economy. The authors develop a model that accounts for the effects of hypoxia and associated oxygen depletion on multiple species populations across both space (shallow vs. deep channels) and time, and assesses the impact of reduced agricultural runoff on the region's ecological and economic welfare. As expected, FTE find that as runoff is reduced total blue crab populations increase, ultimately leading to increased crab harvests and reduced crab prices in the product market. Discounted social welfare is impacted negatively in scenarios where agricultural costs associated with runoff reductions have increased (by 1 percent and 10 percent).

In addition to the inclusion of multiple sectors in the regional economy model, our GSL bioeconomic model shares several key features with the FTE model. In both models economic agents are assumed to be perfectly myopic and all savings and investment behavior is omitted, thus factor stocks remain constant over time. The dynamics of both models therefore emanate from species population adjustments. Government involvement in the economy occurs solely through its regulatory practices, and cumulative aggregate welfare (equivalent variation) measures corresponding to these practices are calculated for each policy intervention. There are also several differences between the two models. FTE model a closed economy, while we assume a small open-

⁶Kasulo and Perrings (2006) assess the productivity of a multi-species, freshwater lake fishery in a similar, albeit more reduced-form fashion. They empirically model the bioeconomics of the fishery using a modified Gordon-Schaefer production function, where the modifications account for water pollution effects (proxied by rainfall in the catchment area) and biodiversity (measured as an index of species harvest diversity that is negatively related to catch per unit of effort). Surprisingly, the authors find that an open access regime is associated with more biodiversity than a regulated profit-maximizing regime. In a similar vein, Huang and Perrings (2017) explore management options for wild fish stocks in the presence of infectious diseases transmitted to and from neighboring fish farms. They find that the stability of the wild stock (the harvest of which is subject to a total allowable catch contingent upon fluctuations in disease infestation) is negatively related to disease-induced mortality and value of the wild fishery itself.

economy framework. FTE ignore potential non-market linkages between households and the bio-economy, while we explicitly model the impacts of pollution from mining and the periodicity of recreation/wildlife viewing opportunities on household welfare. Nutrient flows are exogeneously determined in FTE, while in our model salinity flows are determined endogenously as a consequence of water usage by the agriculture and mining sectors.

Several of these differences also extend to a comparison between our model and that of Finnoff and Caplan (2004) (henceforth FC). While both models are dynamically myopic, the dynamics in FC unfold through two channels – (1) the evolution of factor stocks and household saving decisions and (2) species population updating – while the dynamics in our model, like that of FTE's, occurs solely through the latter channel. Further, FC depict regulation of the brine shrimp fishery as occurring through an industry-wide quota (or upper bound) on total harvest at the outset of each season. In contrast, a harvest moratorium is imposed seasonally in our model once the shrimp population falls beneath a certain threshold level, i.e., the moratorium is effectively a lower-bound regulatory instrument that better reflects both the extent to which and immediacy with which the regulatory authority is capable of managing the fishery. Lastly, FC's simulation exercises are purely demonstrative. The shock occurring in our model instead mimics a particular natural occurrence – prolonged drought. The overriding goal of the simulation exercises is to measure welfare effects associated with different regulatory regimes.

The second strand of literature to which our work contributes – more as a basis of comparison than via extension per se – uses what we are calling a partial-equilibrium (PE) approach to assess the unidirectional impacts of changes in regional economic conditions (which in turn may be initiated by global drivers such as climate change) on an ecosystem or watershed. Examples of these types of studies abound. For instance, Xu et al. (2018) assess the impacts of potential land-use changes on nonpoint-source phosphorous loadings in a Lake Erie Watershed; Perrings and Walker (1997) investigate the efficacy of event-driven, or 'opportunistic' management of rangeland in the presence of periodic wildfire occurrences; and Baggio and Perrings (2015) similarly investigate the trade-offs associated with adaptive vs. non-adaptive policies on a fishery's productivity and resource rents. Erickson et al. (2007) provide a host of case studies that develop practical, quantitative-based

frameworks of analyses within which to assess sustainable watershed management scenarios. While this strand of the literature does not explicitly incorporate dynamic feedbacks into the evolutions of the bioeconomic outcomes (as is done in the GE models), their practicality (in terms of quantitative applications) and the sheer breadth of these applications are nevertheless suggestive of the efficacy of the PE approach.

The next section describes the salient features of the GSL ecosystem. Section 4.4 then derives the various welfare measures used to value the bioeconomy-wide effects of prolonged drought, as well as the fishery-specific and mining regulations imposed to mitigate the drought's effects (specific details of the regional economy are coalesced in Appendix A). Section 4.5 presents results from a series of numerical simulations of the GE bioeconomic model developed in Sections 4.3 and 4.4. Section 4.6 concludes with a summary of the simulation results and a discussion of possible avenues for future research.

4.3 The GSL Ecosystem

The GSL bioeconomy is depicted in Figure 4.2. The figure describes how the ecosystem (described in Belovsky et al. (2011)) interacts with human and environmental factors comprising the surrounding watershed. As in Finnoff and Caplan (2004), the ecosystem is restricted to the GSL's south arm (see Figure 4.1), where the lake's salinity level has historically supported a viable brine shrimp fishery.⁷ As mentioned in Section 4.2, the GSL ecology is comprised of two food webs. One is the benthic food web, where algae (*Phytoplankton*) located in the nutrient pool are preyed upon exclusively by brine flies (*Ephydra Larvae*), which in turn are preyed upon by Adult Ephydras-feeding waterbirds. The other grouping is the pelagic food web, where bacteria (*Cyanobacterial*) from the nutrient pool are exclusively preyed upon by brine shrimp (*Artemia franciscana*), which in turn produce hard-cased eggs, or cysts. The shrimp cysts are preyed upon by insects (*Trichocorixa*) and waterbirds, and are commercially harvested and exported to Asian and South American aquaculture markets as feed for prawns and table shrimp (Lavens and Sorgeloos 2000; UNDR 2013).

⁷The north and south arms of the lake were physically separated in 1959 when the original Southern-Pacific Railroad causeway was in-filled, thus preventing mixing of water flows between the two arms (GSLEP 2017). Because freshwater inflow into the north arm is much less than that flowing into the south arm, the north's salinity level exceeds the maximum concentration permissible for shrimp survival.

Lastly, the lake's salinity level – a critical factor determining the survival of algae, bacteria, shrimp, and fly species – is governed by net water inflow, which in turn is determined by both endogenous (e.g., mineral extraction, agricultural production, residential and commercial development, etc.) and exogenous (e.g., drought, climate change, etc.) factors. As mentioned in Section 4.2, mineral extraction also leads to loadings of pollutants (mercury and selenium) which negatively impact the ecosystem's health.

To simplify our ensuing numerical analysis while retaining the essential features of the ecosystem, we focus on the predator-prey relationships existing between waterbirds (predator species) and the brine shrimp and (adult and larvae) flies (respective prey species), and thus ignore the relatively minor predation role played by *Trichocorixa* on shrimp (Belovsky et al. 2011). We also assume that pollution emitted from mineral extraction impacts households directly, rather than indirectly via negative effects introduced into the ecosystem. Thus, our focus is retained on the role that the lake's salinity level plays in determining the ecosystem's sustainability. For future reference identification numbers are assigned to each respective species according to Table 4.1.

Species' Net-Energy Maximization Problem

Following Finnoff and Caplan (2004), we represent the net-energy (R_i) maximization problem for each predator species' $i = \{s1, f1, 2, \dots, 4\}$ as predator i 's optimal choice of prey species j 's biomass ($x_{ij}, i \neq j$) in any given time period.⁸ For example, consider the interrelated net-energy maximization problems of (representative members of) the bacteria, brine shrimp and waterbird species (i.e., the Pelagic food web in Figure 4.2),

$$\begin{aligned} \text{Max}_{x_{s10}} \quad R_{s1} &= [e_0 - e_{s10}]x_{s10} - e_{s1} [1 + t_{12}e_{21}]y_{12}(x_{s10}) - f_{s1}(x_{s10}; SA_{s1}) - \beta_{s1} \\ \text{Max}_{x_{21}} \quad R_2 &= [e_{s1} - e_{21}]x_{21} - e_2 [1 + t_{24}e_{42}]y_{24}(x_{21}) - f_2(x_{21}; SA_2) - \beta_2 \\ \text{Max}_{x_{42}} \quad R_4 &= [e_2 - e_{42}]x_{42} + [e_3 - e_{43}]x_{43} - f_4(x_{42}, x_{43}) - \beta_4, \end{aligned} \quad (4.1)$$

where R_i is measured in power units (e.g., watts or kilocalories/time). The first terms on the right-hand sides (RHSs) of each problem in (4.1) represent inflows of energy from the nutrient pool and

⁸For expository clarity we henceforth drop the time-period subscript $t = 1, \dots, T$ from the model's variables except where needed.

respective prey species. For instance, the term $[e_0 - e_{s10}]x_{s10}$ for R_{s1} represents the energy inflow from the nutrient pool to bacteria. Constant e_0 is measured in power units (kilocalories/kilogram) and represents the amount of energy embodied in a unit of biomass from the nutrient pool. Embodied energy e_{s10} represents the energy expended by bacteria per unit of predation on the nutrient pool (e.g., to locate, capture, and handle a unit of biomass from the pool), i.e., it is the equilibrium “energy price”, established in the bacteria - nutrient pool “biomass market”, which a member of the bacteria species pays to capture a unit of biomass from the nutrient pool.⁹ The choice variable in bacteria’s maximization problem, x_{s10} , is biomass (kilograms/time) transferred to, or demanded by, bacteria from the nutrient pool. Similar interpretations are made for the initial RHS terms in the brine shrimp and waterbird problems. Note that two energy inflow terms exist for waterbirds, one representing inflow from brine shrimp, $[e_2 - e_{42}]x_{42}$, the other inflow from brine flies, $[e_3 - e_{43}]x_{43}$.

The second terms in (4.1) represent energy outflow via predation. The term $e_{s1} [1 + t_{12}e_{21}]y_{12}(x_{s10})$ in R_{s1} , for instance, is the energy lost by bacteria to brine shrimp through predation, and the term $e_2 [1 + t_{24}e_{42}]y_{24}(x_{21})$ in R_2 is similarly the energy lost by brine shrimp to waterbirds. There is no energy lost through predation for waterbirds since waterbirds are the top predator species in the GSL ecosystem. The second bracketed terms in R_{s1} and R_2 , $[1 + t_{ik}e_{ki}]$ for $k > i$, represent “taxes” on the energy expended by species i to avoid predation by species k , with constant t_{ik} representing the associated tax rate. The total tax paid by a member of species i is assumed to increase with the energy price paid by members of predator species k , e_{ki} , which introduces a “convex damages” predation effect on species i ’s net energy level.¹⁰ Variable y_{ik} represents the biomass transferred from, or supplied by, a member of prey species i to predator species k . This supply is assumed to be a strictly increasing, concave function of x_{ij} . For example, y_{24} is the biomass supplied by brine shrimp to waterbirds, which is an increasing function of shrimp’s biomass demand for bacteria, x_{21} . Similarly, y_{12} is the biomass supplied by bacteria to brine shrimp, which in turn is a function of bacteria’s biomass demand for nutrients, x_{s10} . The specific functional forms for the set of $y_{ik}(x_{ij})$ in both food webs of the GSL ecosystem are shown on the left side of Table 4.2, with constants $\delta_{ij} > 0$

⁹Assuming energy prices $e_{ij} > 0, \forall i > j$ represent equilibrium prices implies that both the predator and prey species populations are constrained in size in any given time period.

¹⁰In other words, the rate at which the marginal damages associated with predation increase is proportional to the rate at which the energy price paid by the predator species increases with predation.

and $0 < \alpha_{ij} < 1 \forall i, j$.

Following Finnoff and Caplan (2004), energy lost via respiration occurs through variable ($f_i(x_{ij}; SA_i)$) and fixed, or basal metabolic (β_i) capacities $\forall i, j$. Variable respiration, which accounts for energy lost in reproduction, defecation, defense of territory, etc., is assumed to be strictly increasing and concave in species i 's biomass demand for its prey, x_{ij} , and increasing in salinity level, SA_i , which in turn is determined via natural fluctuations in water availability and the amount of water diverted for use in the agriculture and mining sectors. As described in the lower half of Table 4.2, SA_i equals the summation of species-specific salinity effects occurring through agricultural (s_{ia}) and mining (s_{im}) production. These effects are calculated as constant multiples of the proportions of water used in agricultural (W_a) and mining (W_m) production, respectively, where σ_{ia} and σ_{im} are each assumed greater than zero $\forall i$. We define $f_i(\cdot)$ according to the expressions listed on the right-hand side (RHS) of Table 4.2, with constants r_i and γ_i both greater than zero $\forall i$.¹¹ Lastly, basal metabolism for each species, β_i , is assumed to be independent of energy intake since it represents the minimum amount of energy necessary for a species to maintain respiration, circulation, and other vital body functions while at rest. First-order optimality conditions (FOCs) corresponding to the net-energy maximization problems expressed in (2.1) for the Pelagic food web result in,¹²

$$\begin{aligned}
 e_0 &= e_{s10} + e_{s1} [1 + t_{12}e_{21}] y_{12}'(x_{s10}) + f_{s1}'(x_{s10}; SA_{s1}), \\
 e_{s1} &= e_{21} + e_2 [1 + t_{24}e_{42}] y_{24}'(x_{21}) + f_2'(x_{21}; SA_2), \\
 e_2 &= e_{42} + f_{42}'(x_{42}, x_{43}), \\
 e_3 &= e_{43} + f_{43}'(x_{42}, x_{43}),
 \end{aligned} \tag{4.2}$$

where $y_{ik}'(x_{ij}) > 0$ and $f_{ij}'(\cdot) > 0$ are the partial derivatives of $y_{ik}(x_{ij})$ and $f_i(\cdot)$ with respect to x_{ij} , respectively, $i, j, k \in \{0, s1, 2, 3, 4\}$. The expressions in (4.2) are standard optimality conditions, which in our case indicates that the biomass demand of each species occurs up to the point where the marginal benefit from energy inflow is equated to the marginal cost of energy outflow. Marginal benefit equals the energy per-unit of prey biomass, e_i , on each equation's left-hand side (LHS).

¹¹As in Finnoff and Caplan (2004), we express the top predator's (waterbird's) variable respiration as a flexible translog function, which accounts in a relatively unrestrictive way for the dual effects that a waterbird's brine-shrimp and brine-fly biomass demands have on its variable respiration.

¹²Similar FOC expressions are calculated for the benthic food web in Figure 4.2.

The marginal cost of energy outflow on the RHS is the summation of (1) energy price paid per unit of biomass demanded, e_{ij} , (2) energy lost to predator k per unit of biomass demanded, $e_i [1 + t_{ik}e_{ki}]y_{ik}'(x_{ij})$, and (3) energy lost via respiration per unit of biomass demanded, $f_{ij}'(\cdot)$. We note that the last two expressions in equations (4.2) represent the waterbird's optimality conditions, the first with respect to its demand for brine shrimp, the second with respect to its demand for brine flies. Neither of these two expressions include energy lost due to predation in the calculation of marginal cost because waterbirds are the ecosystem's top predator species. The specific functional forms used in our ensuing numerical analysis for the full set of species' net-energy equations and corresponding FOCs are presented in Table 4.3.

The resulting ecosystem FOCs can be solved each period for a set of six optimal (equilibrium) biomass demands, $x_{ij}(e_i, e_j, e_{ij}, e_{kii,k}, t_{iki,k}, SA_j) \forall i, j, k$, and, via the corresponding functions $y_{ik}(x_{ij})$, four optimal biomass supplies (recall that waterbirds do not exhibit biomass supplies). For example, a brine shrimp's optimal biomass demand for bacteria in any given period is expressed as $x_{21}(e_{s1}, e_2, e_{21}, e_{42}, t_{24}, SA_2)$, resulting in optimal biomass supply $y_{24}(e_{s1}, e_2, e_{21}, e_{42}, t_{24}, SA_2)$.

Short-Run Ecosystem Equilibrium

Similar to how equilibrium commodity and factor prices are determined in a competitive economy, energy prices play a central role in the determination of an ecosystem equilibrium. In particular, a species' choice of prey depends upon the prey's relative energy price determined in any given biomass market. Each predator species is assumed to be an energy price-taker in the market. However, the equilibrium energy price is determined endogenously within the market via the forces of aggregate predator demand and prey supply. In a short-run equilibrium each species population level is fixed, analogous to the fixed-number-of-firms condition that distinguishes a market economy's short-run competitive equilibrium. A short-run, ecosystem-wide equilibrium is attained when all biomass markets reach equilibrium simultaneously.

For example, short-run equilibrium in the brine shrimp-bacteria biomass market occurs when

brine-shrimp demand for bacteria biomass equals the bacteria population's biomass supply. Notationally speaking, equilibrium in this biomass market satisfies the clearing condition,

$$N_2 x_{21}(e_{s1}, e_2, e_{21}, e_{42}, t_{24}, SA_2) = N_{s1} y_{12}(e_0, e_{s1}, e_{s10}, e_{21}, t_{12}, SA_{s1}),$$

where the LHS of the condition corresponds to brine-shrimp market demand and the RHS corresponds to bacteria market supply. Variables N_2 and N_{s1} represent the given brine-shrimp and bacteria population units, respectively. As in an economy's market equilibrium, where the price of what is demanded must equal the price of what is supplied, the unique equilibrium energy price determined in the brine-shrimp-bacteria biomass market is e_{21} , i.e., the price paid by shrimp for a unit of bacteria biomass. Short-run equilibrium conditions for the full set of biomass markets in the GSL ecosystem are provided in Table 4.3.

Population Adjustment

Given its fixed population size in the short run, a species may exhibit negative, positive, or zero net energy in a short-run equilibrium. The adjustment from short-run to long-run (steady-state) equilibrium in the ecosystem is therefore driven by the simultaneous movement of species' net-energy levels toward zero, which in turn governs associated changes in species population sizes. In effect, natural variations in species population sizes drive, and are driven by, the transition from short-run to long-run equilibrium. The relationship between a species' net-energy and population levels is characterized by what is known as a population-updating equation.¹³

Following Finnoff and Caplan (2004), we begin by constructing the population-updating equation for the top predator species in the GSL ecosystem – waterbirds – and note that in a long-run steady-state equilibrium there is by definition no change in the waterbird population, i.e., births equal deaths. Denote the lifespan of a representative waterbird as s_4 and the size of the waterbird population at time t as N_4^t . The total number of waterbird births and deaths is then calculated as

¹³A similar updating process occurs in a competitive market economy with respect to the relationship between the different "population sizes" of firms and their respective profitability levels – positive profitability in the short run leads to more firms in a long-run equilibrium (via free entry), while negative profitability leads to fewer firms in long-run equilibrium (via free exit).

N_4^t/s_4 , where $1/s_4$ equals both the per-capita waterbird steady-state birth and death rates. Given a representative waterbird's maximized net energy in period t , denoted as $R_4^{t*} = R_4^t(x_{42}^t, x_{43}^t)$, where x_{42}^t and x_{43}^t represent the waterbird's optimal biomass demands for brine shrimp and brine flies, respectively, a long-run ecosystem equilibrium is consistent with the waterbird exhibiting zero net energy, i.e., $R_4^{t*} = 0$.¹⁴

Letting v_4^{ss} represent the waterbird's steady-state variable respiration level, and recalling that variable respiration is used partially for reproduction purposes, we denote energy required for reproduction in the steady state as ρv_4^{ss} , with constant proportionality factor $\rho > 0$. Note that the production level of waterbird offspring in the steady state (i.e., the waterbird population's steady-state birth rate) may therefore be denoted as $N_4^{ss}[\rho v_4^{ss}]$, which equals the total number of waterbird deaths, N_4^{ss}/s_4 , in an ecosystem steady state. This in turn yields the result $\rho = 1/[v_4^{ss}s_4]$. When the waterbird species is instead in a non-steady state, i.e., $R_4^{t*} \neq 0$, its variable respiration level is (henceforth) denoted simply as v_4^t . It therefore follows that in general (i.e., for any period t , regardless of whether a steady state has been attained), the waterbird species' per-capita birth rate can be defined as $[R_4^{t*} + v_4^t]/[v_4^{ss}s_4]$.¹⁵

Assuming the waterbird species' death rate is independent of energy available for reproduction, the waterbird population adjustment equation can therefore be written as,

$$N_4^{t+1} = N_4^t + \frac{N_4^t}{s_4} \left[\frac{R_4^{t*} + v_4^t}{v_4^{ss}} - 1 \right]. \quad (4.3)$$

Note that in the steady state, where $R_4^{t*} = 0$ and $v_4^t = v_4^{ss}$, equation (4.3) collapses to $N_4^{t+1} = N_4^t$, implying no change in the waterbird population from the current period t to the next period $t + 1$. Outside of the steady state we see that changes in the waterbird population depend upon the values of a waterbird's maximized net energy, R_4^{t*} , and variable respiration level, v_4^t .

The population-updating equation for brine shrimp is of special interest. Brine shrimp is the ecosystem's focal species, which is preyed upon by waterbirds and commercially harvested by the

¹⁴ R_4^{t*} may also be referred to as the waterbird's optimal-value, or "indirect net energy" function for period t . As discussed previously, optimal biomass demand exhibited by each species is itself a function of ecosystem parameters pertinent to that species. Hence $R_i^{t*}, \forall i, t$ are also functions of those same respective parameters.

¹⁵Note that in a steady-state equilibrium $R_4^{t*} = 0$ and $v_4^t = v_4^{ss}$ by definition. Thus, the waterbird birth rate expression collapses to $1/s_4$, which is the waterbird species death rate.

regional economy's brine shrimp fishery. As a result, the number of offspring equals the sum of natural deaths, predation mortality, and commercial harvest (which we henceforth denote H_2^t) in the steady state. Recall from equation (4.1) that a member of the brine shrimp species loses its biomass to waterbird predation according to supply function $y_{24}(x_{21})$. Thus, brine shrimp population loss through predation is represented as $N_2 y_{24}(x_{21})$. In the steady state, brine shrimp births equal the sum of natural deaths and predation mortality (net of natural deaths), i.e.,

$$N_2^{ss} \rho_2 v_2^{ss} = N_2^{ss} \left[y_{24}^{ss} \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right],$$

resulting in,

$$\rho_2 = \left[y_{24}^{ss} \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right] \frac{1}{v_2^{ss}},$$

where s_2 and v_2^{ss} are a brine shrimp's lifespan and the steady-state variable respiration level, respectively, and y_{24}^{ss} is the steady-state supply of brine shrimp to waterbirds.¹⁶

Analogous to the waterbird population-updating equation, the population-updating equation for brine shrimp can be written as,

$$N_2^{t+1} = N_2^t + \frac{N_2^t}{v_2^{ss}} \left[y_{24}^{ss} \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right] \left[v_2 + R_2^* \right] - N_2^t \left[y_{24}^t \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right] - H_c^t. \quad (4.4)$$

Population-updating equations for each species in the GSL ecosystem are presented in Table 4.4.

4.4 The GSL Regional Economy

The GLS regional economy is an extended version of the small-country trading equilibrium model (Gilbert and Tower 2012). As in Finnoff and Caplan (2004), a “myopically dynamic” approach is adopted, whereby a sequence of short-run bioeconomic optimizations determines a dynamic equilibrium path via the evolution of species population updates in the GSL ecosystem. The model described here is of a small open-economy, where the (given) prices of the tradable commodities – brine shrimp, minerals, and a composite import good – are exogenously determined in

¹⁶Recall that y_{24}^{ss} is a function of the steady-state bacteria biomass demanded by brine shrimp. In order to change units of measure from biomass to population (which we must do for simulation purposes), the total biomass of brine shrimp is divided by a representative brine shrimp's weight, w_2 . Thus, $y_{24}^{ss} = y_{24}(x_{21}^{ss})/w_2$

world markets. International trade in these commodities enables the regional economy to export and import at given world prices, subject to the constraint that in any market equilibrium (short- or long-run) the aggregate values of exports and imports are equated.

Production

The GSL regional economy consists of four main production sectors: agriculture (a), mineral extraction (m), brine shrimp fishery (f), and recreation/wildlife-viewing (r). Following Gilbert and Tower (2012), each sector is endowed with constant elasticity of substitution (CES) production technology. Specifically,

$$Q_a(K_a, L_a, W_a) = \gamma_a[\delta_{K_a}K_a^{\rho_a} + \delta_{L_a}L_a^{\rho_a} + \delta_{W_a}W_a^{\rho_a}]^{1/\rho_a} \quad (4.5a)$$

$$Q_m(K_m, L_m, W_m) = \gamma_m[\delta_{K_m}K_m^{\rho_m} + \delta_{L_m}L_m^{\rho_m} + \delta_{W_m}W_m^{\rho_m}]^{1/\rho_m} \quad (4.5b)$$

$$Q_f(K_f, L_f, N_2) = \gamma_f[\delta_{K_f}K_f^{\rho_f} + \delta_{L_f}L_f^{\rho_f} + \delta_{2f}(cyst_2N_2)^{\rho_f}]^{1/\rho_f} \quad (4.5c)$$

$$Q_r(K_r, L_r, N_4) = \gamma_r[\delta_{K_r}K_r^{\rho_r} + \delta_{L_r}L_r^{\rho_r} + \delta_{4r}N_4^{\rho_r}]^{1/\rho_r}, \quad (4.5d)$$

where Q_i represents sector i 's output level, $i \in a, m, f, r$, γ_i represents sector i 's productivity scale factor, δ_{ji} input j 's share factor in the production of sector i 's output, $j \in K, L, W, 2, 4$, and ρ_i the degree of input substitutability in sector i (time subscript t is again dropped for expository convenience). Variables K and L are mobile factors of production – labor and capital, respectively – while W represents water input, which is mobile across the agriculture and mining sectors. The term $cyst_2N_2$ in the expression for Q_f accounts for the input role that the brine-shrimp population plays in the quantity of cysts ultimately harvested in any given period¹⁷ Here, proportionality factor $cyst_2$ converts the adult brine-shrimp population to its corresponding cyst population, which, as explained below, is ultimately exported. Similarly, waterbird population, N_4 , serves as an input in the production of recreation and wildlife-viewing, Q_r , but without the need for any attendant conversion factor

¹⁷As mentioned earlier, precedent for the inclusion of a commercial species' population level as an input in its associated production function can be found in Gordon (1954) and Smith (1969).

per se.

Lastly, it is assumed that homogeneous and perfectly mobile inputs capital, labor, and water satisfy their respective finite full-employment conditions each period, expressed as,

$$\bar{K} = K_a + K_m + K_f + K_r, \quad (4.6a)$$

$$\bar{L} = L_a + L_m + L_f + L_r, \quad (4.6b)$$

$$\bar{W} = W_a + W_m, \quad (4.6c)$$

where \bar{K} , \bar{L} and \bar{W} represent period-specific, total available levels of capital, labor, and water, respectively.

The Household Sector

Following Gilbert and Tower (2012), we assume the regional economy's preferences (i.e., the household sector's aggregate preferences), are represented by a standard Cobb-Douglas utility function (U). Specifically,

$$U(C_a, C_m, C_r, C_c, G) = \alpha C_a^{\beta_a} C_m^{\beta_m} C_r^{\beta_r} C_c^{\beta_c} G^{\beta_G}, \quad (4.7)$$

where the set $\{C_a, C_m, C_r, C_c\}$ represents domestic consumption levels of the agricultural, mineral, recreation/wildlife viewing, and composite import goods, respectively. Parameter $\alpha > 0$ is the utility function's shift parameter and the β_i 's represent the function's set of taste parameters, $i \in \{a, m, r, c, G\}$. Variable G represents the composite air and water pollution level resulting from mineral extraction, which is assumed proportional to the mining sector's production level Q_m , specifically $G = \phi Q_m$, $\phi > 0$.¹⁸ While β parameters, β_a , β_m , β_r , and β_c , are each greater than zero, β_G is less than zero, reflecting the fact that pollution damages the environment and thus reduces social welfare. For simplicity we assume β_G is the (negative) average of the other (positive) taste parameters, i.e., $\beta_G = -(\beta_a + \beta_m + \beta_r + \beta_c)/4$.

¹⁸See Farrell (2005) and Henetz (2005) for evidence on the link between GSL mercury pollution and regional mining activities.

Market-Clearing and Trade Balance Conditions

Following Issacson (2002), USFWS (2001), and UNDR (2013), we model the agriculture and recreation sectors as producing solely for the domestic regional market. Brine shrimp cysts are produced solely for export and the mineral extraction industry produces partially for the domestic regional market and partially for export to the world market. Households import the composite good. As a result, the regional economy's per-period market-clearing and trade-balance equations may be expressed as,

$$Q_a = C_a, Q_r = C_r, \quad (4.8a)$$

$$Q_m = C_m + X_m, Q_f = X_f, C_c + X_c = 0, \quad (4.8b)$$

$$\hat{P}_m X_m + \hat{P}_f X_f + \hat{P}_c X_c = 0, \quad (4.8c)$$

where equations (4.8a) and (4.8b) represent the respective market-clearing conditions and equation (4.8c) the trade balance. Here, X_m , X_f , and X_c are net exports of the mining products, brine-shrimp cysts, and composite good, respectively. Their respective values are negative (imports) or positive (exports), whichever the case may be. Given our small open-economy assumption, output prices for the traded goods (\hat{P}_m , \hat{P}_f , and \hat{P}_c) are exogeneously determined in their respective world markets. As the corresponding trade balance equation (4.8c) shows, the total value of net exports sums to zero.

Formal Problem

We can now represent the regional economy's per-period optimization problem in the form of a Lagrangian function, where the objective is to maximize the household sector's welfare function (4.7) subject to full-employment conditions (4.6), production functions (4.5), and market-clearing and trade-balance conditions (4.8) period-by-period, i.e., myopically. Because the agricultural and recreation/wildlife-viewing goods are non-traded, market-clearing conditions associated with these two goods endogenously determine their associated output prices. World prices for minerals, brine shrimp, and the composite import good are determined exogeneously and taken as given in the prob-

lem. The detailed optimization problem with corresponding FOCs is presented in Table 4.6. Here we present a compact form of the problem's per-period Lagrangian function,

$$\mathcal{L} = U(\mathbf{C}, G) + \Lambda[\mathbf{Q}, \mathbf{C}, \mathbf{X}] + \mathbf{M}[\bar{\mathbf{R}}, \mathbf{R}_K, \mathbf{R}_L, \mathbf{R}_W] + \phi[\mathbf{P} \cdot \mathbf{X}], \quad (4.9)$$

where consumption vector $\mathbf{C} = (C_a, C_m, C_r, C_c)$, $\Lambda = (\lambda_a, \lambda_m, \lambda_f, \lambda_r, \lambda_c)$ is the multiplier vector corresponding to the set of market-clearing and trade-balance conditions included in (4.8) (which also represent the equilibrium output prices under competitive market conditions), $\mathbf{Q} = (Q_a, Q_m, Q_f, Q_r)$, $\mathbf{X} = (X_m, X_f, X_c)$, and $\mathbf{M} = (\mu_K, \mu_L, \mu_W)$ is the multiplier vector corresponding to the full-employment conditions, with $\bar{\mathbf{R}} = (\bar{\mathbf{K}}, \bar{\mathbf{L}}, \bar{\mathbf{W}})$, $\mathbf{R}_K = (K_a, K_m, K_f, K_r)$, $\mathbf{R}_L = (L_a, L_m, L_f, L_r)$, and $\mathbf{R}_W = (W_a, W_m)$. Further, $\mathbf{P} = (\hat{P}_m, \hat{P}_f, \hat{P}_c)$, where ϕ is the shadow price of foreign exchange normalized to unity.

Measuring Welfare Change

We adopt equivalent variation (EV) as our welfare measure of the effects of prolonged drought, as well as of the various regulations adopted to mitigate these effects. Because drought imposes a negative social welfare shock on both the ecosystem and the regional economy, our EV measure represents the maximum amount of wealth the aggregate household sector would willingly sacrifice (i.e., its willingness-to-pay (WTP)) to avoid the bioeconomic impacts of the drought. We calculate EV using two methods. One method, which mimics the approach used in Finnoff and Tschirhart (2011), calculates annualized EV as the difference between the current period's GDP and next period's minimum expenditure (evaluated at the current period's prices). We henceforth name this measure "Rolling EV". The second method calculates annualized EV as the difference between the initial period's GDP (i.e., GDP in period $t = 1$) and each subsequent period's minimum expenditure (evaluated at period $t = 1$'s prices). We henceforth label this measure "Base EV". Clearly, Rolling EV is a fully dynamic EV measure, in the sense that it explicitly accounts for period-by-period adjustments in GDP as the minuend in the EV equation. To the contrary, Base EV fixes GDP at its initial level in period $t = 1$ as the equation's minuend. In this sense, Base EV is not confounded by the period-to-period changes in GDP as is the Rolling EV measure.

To calculate the two welfare measures, we begin by expressing the household-sector expenditure function evaluated at current prices \mathbf{P}^t and next period's indirect utility level V^{t+1} as,

$$E(\mathbf{P}^t, V^{t+1}) = \left[\left(\frac{P_a^t}{\beta_a} \right)^{\beta_a} \left(\frac{P_m^t}{\beta_m} \right)^{\beta_m} \left(\frac{P_r^t}{\beta_r} \right)^{\beta_r} \left(\frac{P_c^t}{\beta_c} \right)^{\beta_c} \left(\frac{V^{t+1}}{\alpha(G^t)^{\beta_G}} \right) \right]^{\frac{1}{(\beta_a + \beta_m + \beta_r + \beta_c)}}, \quad (4.10)$$

where the value of V^{t+1} is calculated using (4.7) and the demand expressions for \mathbf{C} that result from the optimality conditions derived in Table 4.6. In specific,

$$V^{t+1} = \alpha \left(\frac{\beta_a GDP^{t+1}}{P_a^{t+1}} \right)^{\beta_a} \left(\frac{\beta_m GDP^{t+1}}{P_m^{t+1}} \right)^{\beta_m} \left(\frac{\beta_r GDP^{t+1}}{P_r^{t+1}} \right)^{\beta_r} \left(\frac{\beta_c GDP^{t+1}}{P_c^{t+1}} \right)^{\beta_c} \left(G^{t+1} \right)^{\beta_G}. \quad (4.11)$$

Rolling EV in period $t + 1$ is then expressed as,

$$EV^{t+1}(\mathbf{P}^t, GDP^t, V^{t+1}) = GDP^t - E(\mathbf{P}^t, V^{t+1}), \quad (4.12)$$

while Base EV in period $t + 1$ is expressed as,

$$EV^{t+1}(\mathbf{P}^1, GDP^1, V^{t+1}) = GDP^1 - E(\mathbf{P}^1, V^{t+1}). \quad (4.13)$$

4.5 Numerical Results

In this section we present two sets of numerical results. The first set provides answers to this study's two main questions: how does the GSL bioeconomy respond to prolonged drought and what are the broader impacts of regulations exclusively targeting recovery of the ecosystem's focal species, brine shrimp? Because one of the regulations investigated here – a harvest moratorium – mimics the regulatory framework currently in force in the GSL bioeconomy (Wotipka 2014), and drought conditions also persist in the GSL region (Rosner 2016), we can interpret this first set of results as loosely representing the bioeconomy's status quo. The second set of results answer's the following question: given that the fishery-specific regulations imposed in the status quo generate perverse social welfare effects – which are driven by the shifting of production from the fishery

and recreation/wildlife viewing sectors to the mining sector, in turn leading to an increase in the emissions of air and water pollution – to what extent can these effects be mitigated by an input or output tax levied on the mining industry?

Prolonged Drought and Fishery-Specific Regulation

As mentioned previously, here we perform numerical simulations of the GSL bioeconomy in order to demonstrate (i) the effects of prolonged drought on the ecosystem and regional economy, and (ii) the extent to which two regulatory policies – brine-shrimp harvest moratorium and fishery input taxation – might speed the bioeconomy’s recovery from this environmental shock.¹⁹ We model the drought as an exogenous 25-percent decrease in the lake’s nutrient pool, in particular a 25-percent decrease in bacteria and algae biomass, that recurs for the first five consecutive periods of the simulation.²⁰ To account for a potential lag in the regulatory response we assume regulations take effect three periods after the onset of the drought and continue for four consecutive periods.²¹ All initial and calibrated parameter and variable values for the ecosystem and regional economy are provided in Tables 4.5 and 4.7, respectively. These values are taken from Finnoff and Caplan (2004), which in turn are derived from an extensive review of the pertinent GSL ecological and microbiological literatures. The economic component of the bioeconomic model is calibrated based on the relative sizes (values of production) of each sector in the regional economy.²²

We begin by exploring the drought’s impacts on species populations in Figure 4.3, along with the impacts that the fishery harvest moratorium and input taxes have on mitigating the drought and speeding species population recoveries to their original steady-state levels over time. The figure’s first column depicts the steady-state species population paths when the GSL ecosystem remains

¹⁹GAMS version 24.7.4 r58773WEX-WEI x86 64bit/MS Windows was used for the simulation exercises.

²⁰While it is possible to model any size shock for any length of time (consecutive or otherwise), we chose 25 percent strictly for demonstrative purposes. As mentioned previously, drought is at least partially an endogenous occurrence, whereby dry weather conditions coincide with what are already relatively high water-use conditions to push water availability beneath an implicit threshold. For simplicity, we impose the drought shock on the bioeconomy during the first five periods, thus avoiding the need to account for any endogenous factors that would potentially precede and ultimately compound the drought’s impacts.

²¹For simulation purposes we compare the harvest moratorium with 50-percent ad valorem taxes levied on both wages (for labor) and the market interest rate (for capital) employed in the brine shrimp fishery. It is assumed that the revenues obtained by the government from imposition of the taxes are redistributed lump-sum to the household sector each period.

²²A goal of future research is to more firmly calibrate the regional-economy model with disaggregated sector-specific data.

in its natural state (sans harvest) versus with harvest, respectively. As expected, harvesting brine shrimp reduces the shrimp population along with its predator species, waterbirds. The brine fly population also decreases as the declining waterbird population substitutes predation on brine flies for predation on brine shrimp. Further, bacteria and algae populations increase due to the falling populations of their respective predator species, brine shrimp and brine flies. One of the concerns with increased base-species populations in the ecosystem is that as the brine shrimp and fly populations decrease due to harvest, a significant increase in the bacteria and algae populations may result in the lake's eutrophication, potentially leading to hypoxic conditions (Bedford 2005). Although the detailed relationship between bacteria and algae populations and the lake's water quality is beyond the scope of this paper, the relationship between brine shrimp harvest and those species' populations is readily estimated via our numerical simulations.

The second and third columns of Figure 4.3 show the effects of the drought on species populations, along with the countervailing effects of the harvest moratorium and input tax regulations, respectively. We see in column two that drought reduces the population size of each species, and that the harvest moratorium has a relatively pronounced countervailing effect on the brine shrimp population recovery path, but relatively small effects on the population recovery paths of the remaining species. Results are similar for the input tax in column three. Both sets of results demonstrate (in a relative sense) the shrimp-specific effects of the harvest moratorium and input tax regulations mentioned in Section 4.2.

To more carefully assess the effects of each regulation on the brine shrimp and waterbird populations we must compare the actual numbers behind the graphs. Table 4.8 presents the population sizes for periods 3 - 15 under drought (with neither regulation imposed) and with each regulation imposed during periods 3 - 6.²³ As the table indicates, the moratorium has slightly stronger recovery effects on both brine shrimp and waterbird populations than does the input tax.

Figure 4.4 presents our corresponding results for species energy prices. As indicated by the figure's first column of graphs, the equilibrium energy prices paid by brine shrimp and brine flies for bacteria and algae, respectively, decrease to lower steady state levels in the presence of harvesting,

²³The table's assessment begins with the imposition of the regulations in period three. Reversion back to the species' respective steady-state levels occurs at roughly the 15th period.

reflecting the relatively larger population sizes of these two prey species as a result of shrimp harvest and waterbirds substituting brine flies for brine shrimp in their predation decisions. Energy prices paid by waterbirds for shrimp and flies each rise initially (tracking the reduced shrimp and fly populations) and then decrease to lower steady-state levels (tracking the reduced waterbird population itself in the new steady state).

The second and third columns of graphs indicate that the drought has relatively large, short-term impacts on the equilibrium energy prices paid by each species; impacts that attenuate as prices return to their respective steady-state levels. It is interesting to note that the prices paid by each predator species (i.e., shrimp for bacteria, fly for algae, and waterbird for shrimp and fly) experience periods of relatively marked over- or under-shooting (i.e., high volatility) along their return paths.²⁴ Similar to their effects on species population adjustments, both the harvest moratorium and input tax reduce this volatility, most notably with respect to the price brine shrimp pay for bacteria. In the case of equilibrium prices paid by waterbirds for shrimp and flies, the initial price increases track the strength of the drops in the shrimp and fly populations relative to the reduction in waterbird population. The overshooting effects suggest that these relative population adjustments are later reversed, causing energy prices to temporarily fall beneath their corresponding steady-state levels. The price paths for shrimp and flies (i.e., e_{21} and e_{41}) indicate that the initial, drought-induced reductions in bacteria and algae populations cause a sharp yet brief spike in prices, the paths of which then track the consequent falls in shrimp and fly populations downward. Overshooting later occurs as the effect of lower bacteria and algae populations outweigh the effect of lower shrimp and fly populations.

Figures 4.5 and 4.6 contain our corresponding results for the regional economy. To begin, we note from Figure 4.5 the drought-induced shifts in production from the lower-valued fishery and recreation/wildlife viewing sectors to the higher-valued agriculture and mineral extraction sectors.²⁵ Corresponding changes in output prices lead to a decrease in GDP during the drought, which in concert with the increase in pollution associated with increased mining activity leads to a relatively

²⁴Price volatility of this nature is a common artifact of general-equilibrium models (c.f., Petri and Hahn (2003)). Although anticipated, the provenance of this volatility in the GSL ecosystem is nevertheless difficult to pin down.

²⁵Recalling that the waterbird population, i.e., waterbird stock, appears as an input in the recreation/wildlife production function, the fall in this stock in response to drought is what leads to the sector's noticeable production decline.

pronounced decline in the household sector's welfare level. Figure 4.6 tracks the changes in input levels that lie behind these adjustments in production. We see that increases in labor and capital employed in agriculture effectively outweigh the reduction in water usage, leading to an increase in production. The mining sector increases its use of each factor. Capital and labor shift out of the fishery, and despite increases in capital and labor employed in the recreation/wildlife viewing sector, the drought-induced reduction in the waterbird population ultimately leads to a fall in that sector's production level.

Unlike their countervailing effects on the drought's impacts on the ecosystem itself, the two fishery-specific regulations for the most part exasperate the drought's effects on the regional economy, with the harvest moratorium inducing the more pronounced effects. As indicated in Figure 4.5, the regulations offset the initial decline in the recreation/wildlife viewing sector and mitigate the amplitude of the sector's return path to its steady state. These effects on the recreation/wildlife viewing sector are a direct consequence of the regulations' negative impacts on the brine shrimp harvest. The corresponding positive impact on the shrimp population transfers directly to its predator species, waterbirds, and via the sector's production function to recreation/wildlife viewing services available to the household sector. With respect to the remaining sectors of the regional economy, the exasperated decline in the shrimp harvest due to regulation in turn exasperates the drought's original impacts. As Figure 4.6 indicates, these 'exasperated' effects are uniform across the sectors' respective factor demands and the markets' equilibrium prices.

The corresponding paths of Rolling and Base EV values associated with the drought and fishery-specific regulations are provided in Figure 4.7. As expected, by fixing GDP at $t = 1$ Base EV essentially tracks the path of the household sector's welfare level, rising as welfare declines over the five consecutive periods of drought. Except for brief time periods occurring near the conclusion of the drought, the fishery-specific regulations exacerbate this increase in Base EV, reflecting the perverse welfare effects associated with regulation (similar in nature to Hazilla and Kopp (1990), Jorgenson et al. (1992), and Pizer and Kopp (2005), which report perverse social welfare effects associated with environmental regulation in a general-equilibrium setting).

Because Rolling EV simultaneously accounts for reduced household welfare levels and the

economy's reduced GDP levels over time as a consequence of the drought, its time paths naturally deviate from those representing Base EV. Specifically, while it initially reflects the negative welfare effects associated with the drought, Rolling EV's path suggests that these welfare effects are eventually outweighed by the drought's negative impact on regional GDP, causing it to drop beneath its benchmark path beginning in period seven. Thus, the efficacy of Rolling EV as a theoretically consistent measure of welfare loss associated with the drought is drawn into question. It suggests an eventual increase in monetized welfare in the face of what are clearly negative impacts of the drought on social welfare itself and regional GDP. As expected, the fishery-specific regulations exasperate the Rolling EV measure of the drought on both its upside (in reference to the spike occurring from roughly periods 2 – 4) and downside (periods 5 – 8).

Table 4.9 presents the corresponding discounted sums of both EV measures, where we assume a constant discount rate of 2.9% over a 15-period time frame.²⁶ We note that both measures increase in response to the drought, and then increase again in response to the imposition of the fishery-specific regulations. Larger EV values are associated with the harvest moratorium, indicating that on average the household sector is willing to pay the most to avoid the combination of the prolonged drought and the moratorium.

Mining regulations

As discussed in Section 4.2, the perverse social welfare effects associated with fishery-specific regulations beg the question of how we might subsequently boost welfare through additional regulation, e.g., via the imposition of multiple sector-specific regulations. The natural sector to target in our case is the mining industry, since the increased mineral extraction induced by the fishery-specific regulations ultimately causes a welfare-reducing increase in pollution. In this section we investigate two forms of mining regulations – ad valorem input and output taxation – each levied at 30 percent for the same four consecutive periods (periods 3 – 6) as the fishery regulations.²⁷ For

²⁶The 2.9% discount rate used in this study reflects Utah's historical rate of inflation (The Salt Lake Tribune 2013).

²⁷Although taxation per se has heretofore not been identified by the lake's regulatory authority as a possible pollution control mechanism for the mining industry, the need for some form of control has found at least some support in the literature. See for instance Bedford (2005). As with the fishery input taxes discussed in the previous subsection, revenue obtained by the government from imposition of the mining taxes are redistributed lump-sum to the household sector each period.

ease of reference, the full set of regulations considered in the paper are now summarized in Table 4.10. We focus the ensuing discussion on results for the Moratorium and Mining Output Tax scenario.²⁸

Figures 4.8 and 4.9 depict the effects of the mining output tax (in concert with the harvest moratorium) on species populations and energy prices, respectively. Both figures demonstrate that relative to the moratorium's impacts on the drought the imposition of the mining output tax has no discernible, additional effects. Potential feedback effects within the ecosystem may therefore be considered second-order small at best.

To the contrary, the mining tax has relatively pronounced effects on the regional economy. As shown in Figures 4.10 and 4.11, these effects are in some cases countervailing and in others intensifying. For example, in Figure 4.10 we see that, as expected, the tax induces increases in agricultural and recreation/wildlife production as well as in consumption of the composite good. Mining production and net exports decline along with associated pollution emissions. Regional GDP also declines, but household welfare increases, mainly in response to the reduced pollution from mining. As Figure 4.11 demonstrates, these output effects are driven by a reallocation of factors of production from the mining sector to the agriculture and recreation/wildlife viewing sectors.

As shown in Figure 4.12, the increase in household welfare associated with imposition of the mining tax is mirrored in the Base and Rolling EV time paths. Base EV now tracks beneath the reference path corresponding to valuation of the drought itself, both during and shortly after the drought's occurrence. Rolling EV demonstrates similar countervailing tendencies, tracking beneath its reference drought valuation path during the bulk of the drought period, and then spiking above the reference path for 2 – 3 periods in the drought's wake. Similar to valuation results obtained for the fishery-specific regulations, the efficacy of Rolling EV as a theoretically consistent measure of welfare is drawn into question. In this case the eventual increase in monetized welfare in the face of what is clearly the mining tax's positive impact on social welfare itself suggests that the household sector's WTP to avoid the triumvirate of drought, harvest moratorium, and mining tax should fall relative to its WTP to avoid only the drought and moratorium, as is reflected in the time path of Base EV.

²⁸Results for the other three scenarios are qualitatively similar and available from the authors upon request.

As Table 4.11 indicates, both discounted EV measures are nevertheless consistent with the effects of the mining tax on household welfare – the measures associated with the combination of harvest moratorium and mining tax are lower than those for the moratorium alone.

4.6 Summary and Conclusion

We have developed a general-equilibrium (bioeconomic) model of Utah's Great Salt Lake (GSL) ecosystem that tightly links the lake's ecosystem with its regional economy and attendant international trading partners. The bioeconomic model initially developed in Finnoff and Caplan (2004) has been updated to reflect new information regarding mutually exclusive species interaction in the ecosystem's food web. The ecosystem has been linked with an extended version of the small-country trading equilibrium model developed by Gilbert and Tower (2012) in order to identify the dynamic feedback effects of prolonged drought on the GSL bioeconomy. The drought modeled here mimics a temporally drying climate's impact on the lake's nutrient pool, which consists of the food web's base species, algae and bacteria. The base species are preyed upon by brine shrimp (the ecosystem's focal species) and brine flies, which in turn are preyed upon by waterbirds. In this framework we have demonstrated numerically (i) how the effects of prolonged drought change key bioeconomic variables over time, and (ii) how these variables eventually recover from the drought toward a new steady state. We have also considered how two fishery-specific regulatory tools – a temporary brine-shrimp harvest moratorium (i.e., quantity regulation) and an ad valorem tax on the fishery's factors of production (price regulation) – might be used to alter the bioeconomy's recovery path.

Numerical simulation exercises have generated four key findings. First, the fishery-specific regulatory instruments have relatively strong recovery effects on the brine shrimp population, but noticeably weaker effects on the ecosystem's other species populations, suggesting that multi-species recovery in an ecosystem suffering from prolonged drought requires the imposition of multiple species-specific regulatory instruments, similar to how the control of multiple pollutants require pollutant-specific regulation. In other words, if society's goal is to simultaneously speed recovery from an environmental shock of both a focal prey species and a predator species, two separate reg-

ulatory policies will be necessary – one directed toward the recovery of the prey species, the other directed toward recovery of the predator species. Second, the reallocation of production in the regional economy wrought by these regulations affects social welfare perversely, as a consequence of an increase in pollution damage accompanying the reallocation of production resources from the shrimp fishery to the mining sector. This result extends the classic general-equilibrium literature on welfare effects associated with environmental regulation to a bioeconomic setting.

Third, we show that these perverse welfare effects associated with fishery-specific regulations can be offset via imposition of either an input or output tax on mineral extraction. This suggests that, in addition to multiple species-specific regulations imposed within the ecosystem itself, multi-sector regulation within the regional economy is also likely necessary to maintain, or perhaps improve, social welfare. Fourth, we have evaluated the efficacy of two dynamic measures of equivalent variation in measuring the welfare effects associated with prolonged drought, the shrimp harvest moratorium, and a mining output tax. What we have named Base EV calculates annualized EV as the difference between the initial period's GDP (GDP in period $t = 1$) and each subsequent period's minimum expenditure (evaluated at period $t = 1$'s prices). In contrast Rolling EV calculates annualized EV as the difference between the current period's GDP and next period's minimum expenditure (evaluated at the current period's prices). With respect to each EV measure's time path, we find that the efficacy of Rolling EV as a theoretically consistent measure of welfare loss associated with the drought is drawn into question. However, summary (i.e., discounted) measures of EV are theoretically consistent for both Base and Rolling EV.

The path(s) for future research are clear, each relating to a series of updates to the numerical model presented here as more scientific information about the effects of prolonged drought on the bioeconomy come to light. For instance, the specific effects of drought on agricultural productivity and water allocation in general in the GSL regional economy have not been fully explored in this particular version of the model (Coppock 2011; Elias et al. 2016). Neither have the effects of pollution from mining and nutrient loading from agriculture on the GSL ecosystem. Indeed, the array of possible adjustments to the model, as new information about the bioeconomy becomes available, seems almost limitless. Ultimately, the usefulness of this type of bioeconomic model is

measured by the extent to which it can be utilized to inform management policy. As it evolves with new data and better understanding of the bioeconomy, the model can provide a holistic framework within which to assess the effects of drought and climate change on both the ecology of the GSL as well as the region's economy and social welfare.

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4.7 Appendix A: GSL Ecosystem

Table 4.1: Species Identification Numbers

Species Numbers	Species Names
0	Nutrient Pool
s1	Bacteria
f1	Algae
2	Brine Shrimp
3	Brine Flies (larvae and adult)
4	Waterbirds

Table 4.2: Biomass Supply and Variable-Respiration Functions

Biomass Supply	Variable-Respiration
$y_{12}(x_{s10}) = \delta_{12}x_{s10}^{\alpha_{12}}$	$f_{s1}(SA_{s1}; x_{s10}) = r_{s1}[SA_{s1} + 1]x_{s10}^{\gamma_{s1}}$, where $SA_{s1} = s_{1a} + s_{1m}$
$y_{13}(x_{f10}) = \delta_{13}x_{f10}^{\alpha_{13}}$	$f_{f1}(SA_{f1}; x_{f10}) = r_{f1}[SA_{f1} + 1]x_{f10}^{\gamma_{f1}}$, where $SA_{f1} = SA_{s1}$
$y_{24}(x_{21}) = \delta_{24}x_{21}^{\alpha_{24}}$	$f_2(SA_2; x_{21}) = r_2[SA_2 + 1]x_{21}^{\gamma_2}$, where $SA_2 = s_{2a} + s_{2m}$
$y_{34}(x_{31}) = \delta_{34}x_{31}^{\alpha_{34}}$	$f_3(SA_3; x_{31}) = r_3[SA_3 + 1]x_{31}^{\gamma_3}$, where $SA_3 = s_{3a} + s_{3m}$
	$f_4(x_{42}, x_{43}) = r_4(x_{42} + x_{43}) + 0.5r_4(r_{43}x_{42}x_{43} + x_{42}^{\gamma_4} + r_{43}x_{43}^{\gamma_4})$

$$s_{1a} = \sigma_{1a} \left[\frac{W_a}{W_a + W_m} \right] \quad \text{and} \quad s_{1m} = \sigma_{1m} \left[\frac{W_m}{W_a + W_m} \right]$$

$$s_{2a} = \sigma_{2a} \left[\frac{W_a}{W_a + W_m} \right] \quad \text{and} \quad s_{2m} = \sigma_{2m} \left[\frac{W_m}{W_a + W_m} \right]$$

$$s_{3a} = \sigma_{3a} \left[\frac{W_a}{W_a + W_m} \right] \quad \text{and} \quad s_{3m} = \sigma_{3m} \left[\frac{W_m}{W_a + W_m} \right]$$

Table 4.3: Species' Net-Energies, the First Order and the Short Run Equilibrium Conditions

Species	Net-Energy (Watts or Kilocalories)
Bacteria	$R_{s1} = [e_0 - e_{s10}]x_{s10} - e_{s1} [1 + t_{12}e_{21}]y_{12}(x_{s10}) - f_{s1}(SA_{s1}; x_{s10}) - \beta_{s1}$
Algae	$R_{f1} = [e_0 - e_{f10}]x_{f10} - e_{f1} [1 + t_{13}e_{31}]y_{13}(x_{f10}) - f_{f1}(SA_{f1}; x_{f10}) - \beta_{f1}$
Brine Shrimp	$R_2 = [e_{s1} - e_{21}]x_{21} - e_2 [1 + t_{24}e_{42}]y_{24}(x_{21}) - f_2(SA_2; x_{21}) - \beta_2$
Brine Flies	$R_3 = [e_{f1} - e_{31}]x_{31} - e_3 [1 + t_{34}e_{43}]y_{34}(x_{31}) - f_3(SA_3; x_{31}) - \beta_3$
Waterbirds	$R_4 = [e_2 - e_{42}]x_{42} + [e_3 - e_{43}]x_{43} - f_4(x_{42}, x_{43}) - \beta_4$
Control Variables	First Order Conditions
x_{s10}	$e_0 = e_{s10} + e_{s1} [1 + t_{12}e_{21}]y_{12}'(x_{s10}) + f_{s1}'(x_{s10}; SA_{s1})$
x_{f10}	$e_0 = e_{f10} + e_{f1} [1 + t_{13}e_{31}]y_{13}'(x_{f10}) + f_{f1}'(x_{f10}; SA_{f1})$
x_{21}	$e_{s1} = e_{21} + e_2 [1 + t_{24}e_{42}]y_{24}'(x_{21}) + f_2'(x_{21}; SA_2)$
x_{31}	$e_{f1} = e_{31} + e_3 [1 + t_{34}e_{43}]y_{34}'(x_{31}) + f_3'(x_{31}; SA_3)$
x_{42}	$e_2 = e_{42} + f_{42}(x_{42}, x_{43})$
x_{43}	$e_3 = e_{43} + f_{43}(x_{42}, x_{43})$
Biomass Markets	Short Run Equilibrium Conditions
Nutrient Pool – Bacteria	$N_{s1}x_{s10}(e_{s10}, e_{21}, SA_{s1}) = Aress$
Nutrient Pool – Algae	$N_{f1}x_{f10}(e_{f10}, e_{31}, SA_{f1}) = Aresf$
Brine Shrimp – Bacteria	$N_2x_{21}(e_{21}, e_{42}, SA_2) = N_{s1}y_{12}(e_{s10}, e_{21}, SA_{s1})$
Brine Flies – Algae	$N_3x_{31}(e_{31}, e_{43}, SA_3) = N_{f1}y_{13}(e_{f10}, e_{31}, SA_{f1})$
Waterbirds – Brine Shrimp	$N_4x_{42}(e_{42}, e_{43}) = N_2y_{24}(e_{21}, e_{42}, SA_2)$
Waterbirds – Brine Flies	$N_4x_{43}(e_{42}, e_{43}) = N_3y_{34}(e_{31}, e_{43}, SA_3)$

Note: $Aress$ and $Aresf$ represent the fixed sizes of the nutrient pool for bacteria and algae, respectively. As benchmarks, $Aress$ is calibrated by $Aress = N_{s1}x_{s10}$, and $Aresf$ by $Aresf = N_{f1}x_{f10}$. The explicit functional forms of partial derivatives of $f_i(\cdot)$ and $y_{ik}(x_{ij})$ are shown as follows,

$$\begin{aligned}
f_{s1}'(x_{s10}; SA_{s1}) &= \gamma_{s1}r_{s1}[SA_{s1} + 1]x_{s10}^{\gamma_{s1}-1}, \quad y_{12}'(x_{s10}) = \alpha_{12}\delta_{12}x_{s10}^{\alpha_{12}-1} \\
f_{f1}'(x_{f10}; SA_{f1}) &= \gamma_{f1}r_{f1}[SA_{f1} + 1]x_{f10}^{\gamma_{f1}-1}, \quad y_{13}'(x_{f10}) = \alpha_{13}\delta_{13}x_{f10}^{\alpha_{13}-1} \\
f_2'(x_{21}; SA_2) &= \gamma_2r_2[SA_2 + 1]x_{21}^{\gamma_2-1}, \quad y_{24}'(x_{21}) = \alpha_{24}\delta_{24}x_{21}^{\alpha_{24}-1} \\
f_3'(x_{31}; SA_3) &= \gamma_3r_3[SA_3 + 1]x_{31}^{\gamma_3-1}, \quad y_{34}'(x_{31}) = \alpha_{34}\delta_{34}x_{31}^{\alpha_{34}-1} \\
f_{42}'(x_{42}; x_{43}) &= r_4 + 0.5r_4[r_{43}x_{43} + \gamma_4x_{42}^{\gamma_4-1}] \\
f_{43}'(x_{42}; x_{43}) &= r_4 + 0.5r_4[r_{43}x_{42} + \gamma_4r_{43}x_{43}^{\gamma_4-1}].
\end{aligned}$$

Table 4.4: Species' Population-Updating Equations

Species	Population-Updating Equations
$s1$	$N_{s1}^{t+1} = N_{s1}^t + \frac{N_{s1}^t}{v_{s1}^{ss}} \left[y_{12}^{ss} \left(1 - \frac{1}{s_{s1}} \right) + \frac{1}{s_{s1}} \right] \left[v_{s1} + R_{s1}^* \right] - N_{s1}^t \left[y_{12}^t \left(1 - \frac{1}{s_{s1}} \right) + \frac{1}{s_{s1}} \right]$
$f1$	$N_{f1}^{t+1} = N_{f1}^t + \frac{N_{f1}^t}{v_{f1}^{ss}} \left[y_{13}^{ss} \left(1 - \frac{1}{s_{f1}} \right) + \frac{1}{s_{f1}} \right] \left[v_{f1} + R_{f1}^* \right] - N_{f1}^t \left[y_{13}^t \left(1 - \frac{1}{s_{f1}} \right) + \frac{1}{s_{f1}} \right]$
2	$N_2^{t+1} = N_2^t + \frac{N_2^t}{v_2^{ss}} \left[y_{24}^{ss} \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right] \left[v_2 + R_2^* \right] - N_2^t \left[y_{24}^t \left(1 - \frac{1}{s_2} \right) + \frac{1}{s_2} \right] - H_c^t$
3	$N_3^{t+1} = N_3^t + \frac{N_3^t}{v_3^{ss}} \left[y_{34}^{ss} \left(1 - \frac{1}{s_3} \right) + \frac{1}{s_3} \right] \left[v_3 + R_3^* \right] - N_3^t \left[y_{34}^t \left(1 - \frac{1}{s_3} \right) + \frac{1}{s_3} \right]$
4	$N_4^{t+1} = N_4^t + \frac{N_4^t}{s_4} \left[\frac{v_4 + R_4^*}{v_4^{ss}} - 1 \right]$

Note: $y_{12}^{ss} = y_{12}(x_{s10}^{ss})/w_{s1}$, $y_{13}^{ss} = y_{13}(x_{f10}^{ss})/w_{f1}$, $y_{24}^{ss} = y_{24}(x_{21}^{ss})/w_2$, $y_{34}^{ss} = y_{34}(x_{31}^{ss})/w_3$

Table 4.5: Values of Ecological Parameters and Initial Variables

<u>Embodied Energy</u> $e_0 = 1500$ $e_{s1} = 1300$ $e_{f1} = 1300$ $e_2 = 1000$ $e_3 = 500$	<u>Tax in Supply</u> $t_{12}=0.0000688280$ $t_{13}=0.0000130773$ $t_{24}=0.0090030127$ $t_{34}=0.0115926425$	<u>Supply</u> $\alpha_{12}^{GSL}=0.5$ $\alpha_{13}^{GSL}=0.5$ $\alpha_{24}^{GSL}=0.5$ $\alpha_{34}^{GSL}=0.5$	<u>Basal Metabolism</u> $\beta_{s1}=1.4198593^*$ $\beta_{f1}=1.88082521^*$ $\beta_2=56.03115663^*$ $\beta_3=29.21489220^*$ $\beta_4=61,870.1$
<u>δ^{GSL} Parameters</u> $\delta_{12}^{GSL}=0.01053687^*$ $\delta_{13}^{GSL}=0.00233150^*$ $\delta_{24}^{GSL}=0.13640192^*$ $\delta_{34}^{GSL}=0.24634991^*$	<u>Respiration Parameters</u> $\gamma_{s1}^{GSL}=1.3$ $\gamma_{f1}^{GSL}=1.3$ $\gamma_2^{GSL}=1.3$ $\gamma_3^{GSL}=1.3$ $\gamma_4^{GSL}=1.3$ $\gamma_{43}^{GSL}=1.3$	<u>r^{GSL} Parameters</u> $r_{s1}^{GSL}=2,381.249^*$ $r_{f1}^{GSL}=2,391.057^*$ $r_2^{GSL}=514.0552^*$ $r_3^{GSL}=553.338^*$ $r_4^{GSL}=157.5466^*$ $r_{43}^{GSL}=0.02332145^*$	<u>Life Spans</u> $s_{s1}=5$ $s_{f1}=5$ $s_2=5$ $s_3=5$ $s_4=15$
<u>Salinity Adjustment</u> $\sigma_{1a}=0.75$ $\sigma_{2a}=0.75$ $\sigma_{3a}=0.75$ $\sigma_{1m}=0.75$ $\sigma_{2m}=0.75$ $\sigma_{3m}=0.75$	<u>Solar Input</u> $ps=0.09$ $qs=0.4$ $pf=0.09$ $qf=0.04$	<u>Weights</u> $w_{s1}=0.007$ $w_{f1}=0.007$ $w_2=0.776154462$ $w_2=0.776154462$ $w_3=0.887$	<u>Species and Energy Constraint</u> $A_{ress}=1,478,332^*$ $A_{resf}=2,892,059^*$ $etaco=0.005$

Variables	Initial Values
Biomass Demand	$x_{s10}=0.007068638$ $x_{f10}=0.007068638$ $x_{21}=0.656089993$ $x_{31}=0.401$ $x_{42}=156$ $x_{43}=156$
Species Populations	$N_{s1}=209,139,538$ $N_{f1}=409,139,538$ $N_2=386,392$ $N_3=200,000$ $N_4=200$
Biomass Energy Price	$e_{s10}=191.6095^*$ $e_{f10}=250.6058^*$ $e_{21}=117^*$ $e_{31}=195^*$ $e_{42}=90^*$ $e_{43}=45^*$

4.8 Appendix B: GSL Regional Economy

Table 4.6: Formal Optimization Problem for the GSL Regional Economy

Objective: $U = \alpha C_a^{\beta_a} C_m^{\beta_m} C_r^{\beta_r} C_c^{\beta_c} G^{\beta_G}$

Subject to,

i) $Q_a = C_a, Q_m = C_m + X_m, Q_f = X_f, Q_r = C_r, C_c + X_c = 0$ (Market-Clearing Conditions)

ii) $\bar{K} = K_a + K_m + K_f + K_r, \bar{L} = L_a + L_m + L_f + L_r, \bar{W} = W_a + W_m$ (Full-Employment Conditions)

iii) $\hat{P}_m X_m + \hat{P}_f X_f + \hat{P}_c X_c = 0$ (Trade-Balance Conditions)

First Order Conditions (FOCs):

i) $\frac{\partial U}{\partial C_a} = \lambda_a, \frac{\partial U}{\partial C_m} = \lambda_m, \frac{\partial U}{\partial C_r} = \lambda_r, \frac{\partial U}{\partial C_c} = \lambda_c$

ii) $\lambda_m = \phi \hat{P}_m, \lambda_f = \phi \hat{P}_f, \lambda_c = \phi \hat{P}_c$, where ϕ is normalized to unity

iii) $\lambda_a \frac{\partial Q_a}{\partial K_a} = \lambda_m \frac{\partial Q_m}{\partial K_m} = \lambda_f \frac{\partial Q_f}{\partial K_f} = \lambda_r \frac{\partial Q_r}{\partial K_r} = \mu_K$

iv) $\lambda_a \frac{\partial Q_a}{\partial L_a} = \lambda_m \frac{\partial Q_m}{\partial L_m} = \lambda_f \frac{\partial Q_f}{\partial L_f} = \lambda_r \frac{\partial Q_r}{\partial L_r} = \mu_L$

v) $\lambda_a \frac{\partial Q_a}{\partial W_a} = \lambda_m \frac{\partial Q_m}{\partial W_m} = \mu_W$

vi) market-clearing, full-employment, and trade-balance conditions hold

Note that the first order conditions are derived from the following Lagrangian function,

$$\begin{aligned} \mathcal{L} = & U + \lambda_a(Q_a - C_a) + \lambda_m(Q_m - C_m - X_m) + \lambda_f(Q_f - X_f) + \lambda_r(Q_r - C_r) + \lambda_c(C_c - X_c) \\ & + \mu_K(\bar{K} - K_a - K_m - K_f - K_r) + \mu_L(\bar{L} - L_a - L_m - L_f - L_r) + \mu_W(\bar{W} - W_a - W_m) \\ & + \phi(\hat{P}_m X_m + \hat{P}_f X_f + \hat{P}_c X_c). \end{aligned}$$

Table 4.7: Values of Economic Parameters and Initial Variables

Sectors	Factor Productivity	Factor Substitutability
Agriculture (<i>a</i>)	$\gamma_a=2.89937061^*$	$\rho_a=0.1$
Mining (<i>m</i>)	$\gamma_m=2.76880520^*$	$\rho_m=0.1$
Brine-Shrimp Fishery (<i>f</i>)	$\gamma_f=0.19452063^*$	$\rho_f=0.1$
Recreation (<i>r</i>)	$\gamma_r=0.04637760^*$	$\rho_r=0.1$
Composite (<i>c</i>)		
Share Parameter (Capital)	Share Parameter (Labor)	Share Parameter
$\delta_{Ka}=0.27361600^*$	$\delta_{La}=0.27361600^*$	$\delta_{Wa}=0.45276800^*$
$\delta_{Km}=0.51294360^*$	$\delta_{Lm}=0.21217673^*$	$\delta_{Wm}=0.27487967^*$
$\delta_{Kf}=0.87841463^*$	$\delta_{Lf}=0.12158537^*$	$\delta_{2f}=0.25^*$
$\delta_{Kr}=0.91491276^*$	$\delta_{Lr}=0.08508724^*$	$\delta_{4r}=0.375^*$
Output Price(exogenous)	Taste Parameter	Shift Parameter in Utility
$\hat{P}_f=1$	$\beta_a=0.23076923^*$	$\alpha=9.81493045^*$
$\hat{P}_m=3$	$\beta_m=0.23076923^*$	
$\hat{P}_c=2$	$\beta_r=0.23076923^*$	
	$\beta_c=0.30769231^*$	
	$\beta_{poll}=-0.25^*$	

Variables	Initial Values
Output, Consumption, and Net Exports	$Q_a=C_a=150$ $Q_m=C_m=150$ ($X_m=0$) $Q_f=X_f=200$ $Q_r=C_r=150^*$ $C_c=-X_c=200^*$
Capital	$K_a=40$ $K_m=80$ $K_f=180^*$ $K_r=140^*$
Labor	$L_a=40$ $L_m=30$ $L_f=20$ $L_r=10$
Specific Factor	$W_a=70^*$ $W_m=40^*$
Factor Price	$r_k=1$ $r_l=1$ $r_w=1$
Output Price (endogenous)	$P_a=2$ $P_r=2$
Utility	$U=650$
Gross Domestic Product	$GDP=650$
Pollution	$G=37.5^*$
Harvest	$H_c=104,000^*$

Table 4.8: Brine Shrimp/Waterbird Populations under Drought, Moratorium, and Input Taxation

Key Periods	Brine Shrimp			Waterbirds		
	Drought	Moratorium	Input Tax	Drought	Moratorium	Input Tax
3	257,590	257,590	257,590	194.32	194.32	194.32
4	236,116	255,794	250,816	189.61	189.61	189.61
5	222,623	252,195	244,485	184.51	185.01	184.89
6	214,660	248,850	239,709	179.42	180.63	180.33
7	230,108	265,267	255,664	175.95	178.03	177.49
8	249,291	265,840	261,187	174.34	177.25	176.49
9	264,395	272,023	269,844	174.22	177.29	176.48
10	274,927	278,213	277,257	175.06	177.97	177.20
11	281,804	282,944	282,597	176.44	179.04	178.35
12	286,048	286,136	286,089	178.05	180.30	179.70
13	288,499	288,100	288,193	179.68	181.59	181.08
14	289,782	289,191	289,341	181.24	182.84	182.41
15	290,337	289,706	289,870	182.65	183.97	183.62

Table 4.9: Present Values of Rolling and Base EV

Scenario	Rolling EV	Base EV
With Harvest	4.78	43.98
Drought	9.07	119.23
Moratorium	10.03	161.02
Fishery Input Tax	9.47	141.73

Table 4.10: Multi-Sector Regulatory Scenarios

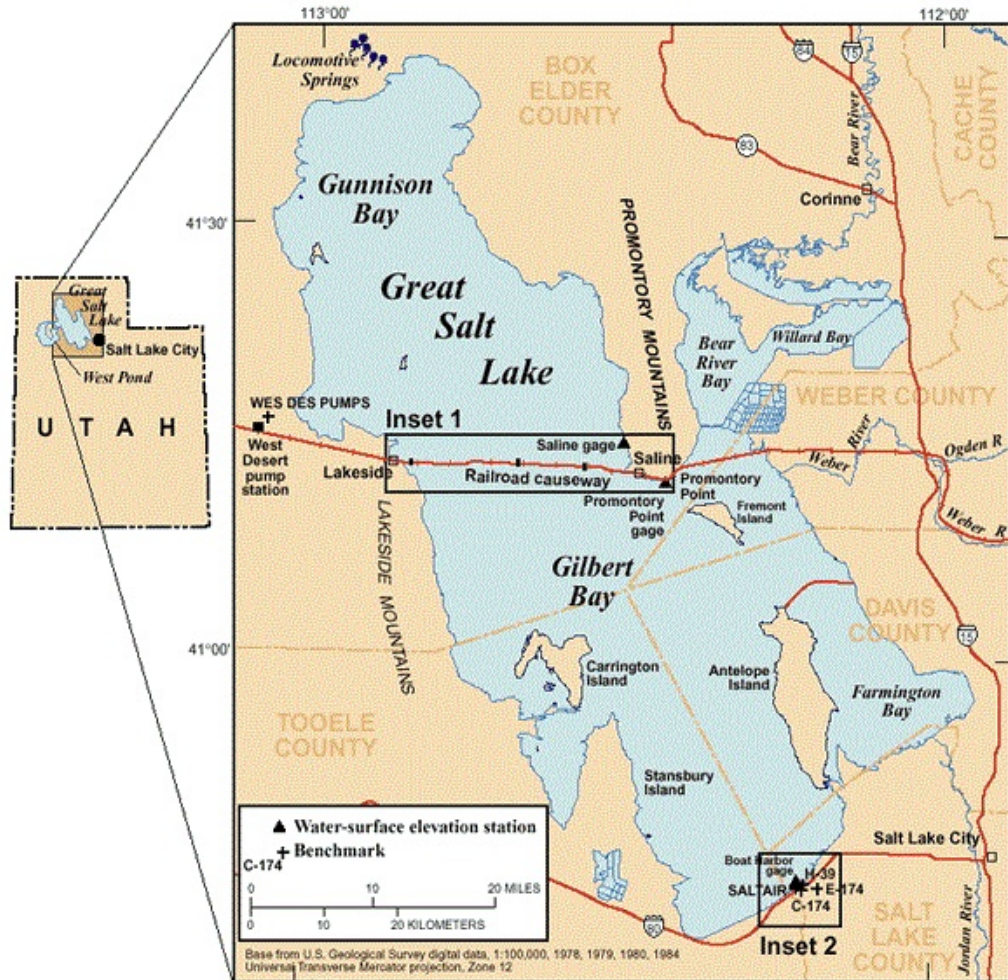
Scenarios
Moratorium and Mining Input Tax (30%)
Moratorium and Mining Output Tax (30%)
Fishery Input Tax (50%) and Mining Input Tax (30%)
Fishery Input Tax (50%) and Mining Output Tax (30%)

Table 4.11: Present Values of Rolling and Base EV

Single/Multiple Sector Regulations	Rolling EV	Base EV
Drought	9.07	119.23
Moratorium	10.03	161.02
Moratorium and Output Tax on Mining	6.78	74.33

4.9 Appendix C: Figures

Fig. 4.1: The Great Salt Lake (GSL) Map



Source: U.S. Geological Survey (USGS)

<http://ut.water.usgs.gov/greatsaltlake/elevations/gslcorrection.html>

Fig. 4.2: The Great Salt Lake (GSL) Ecosystem

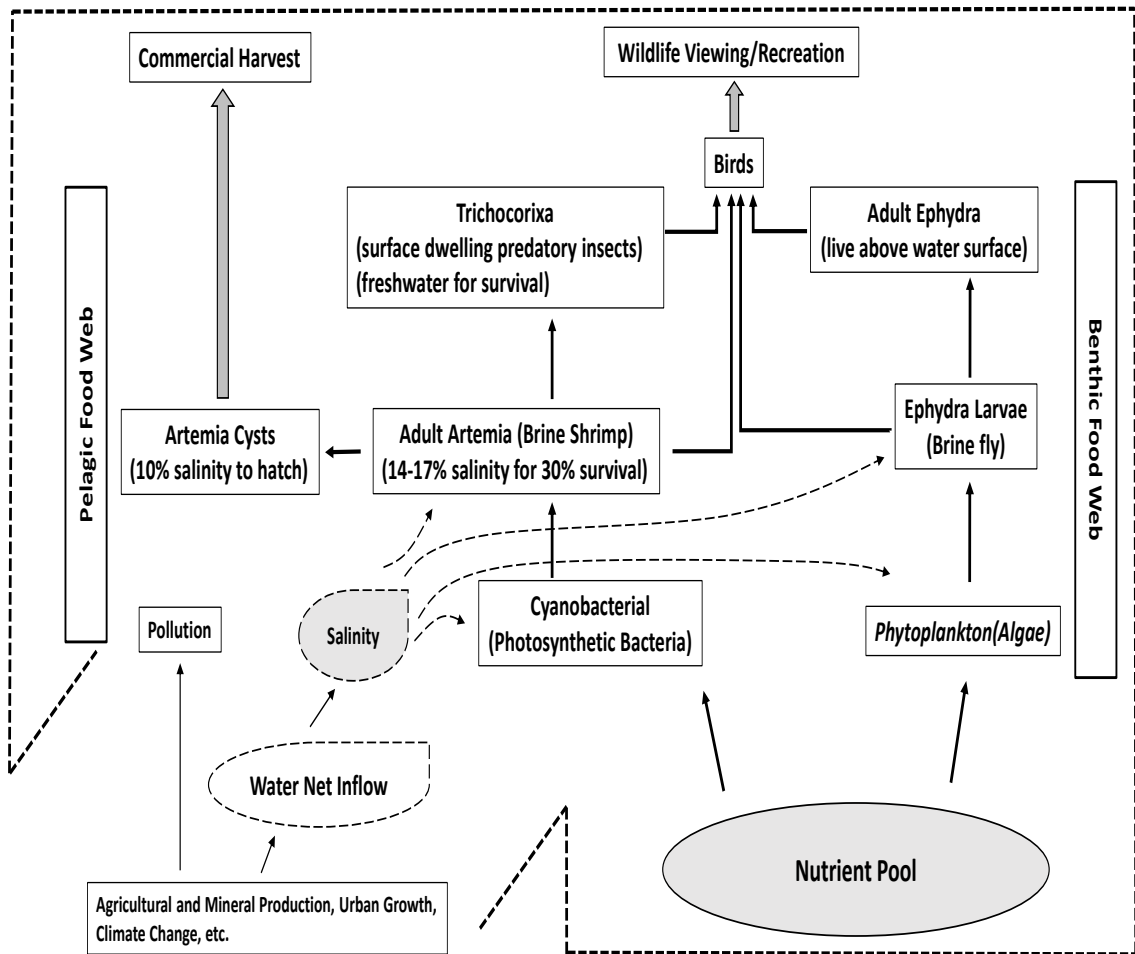


Fig. 4.3: Species Populations

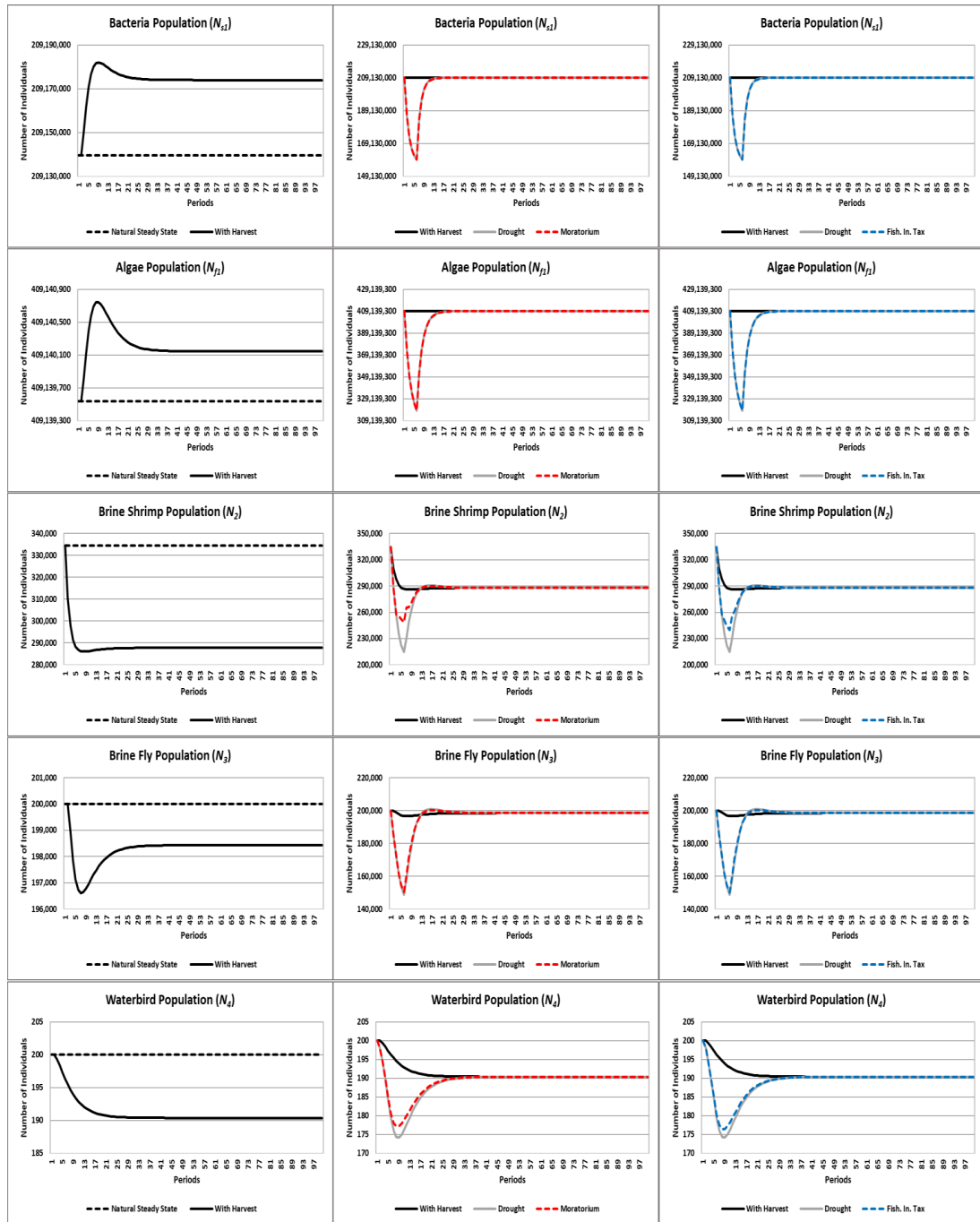


Fig. 4.4: Species Energy Prices

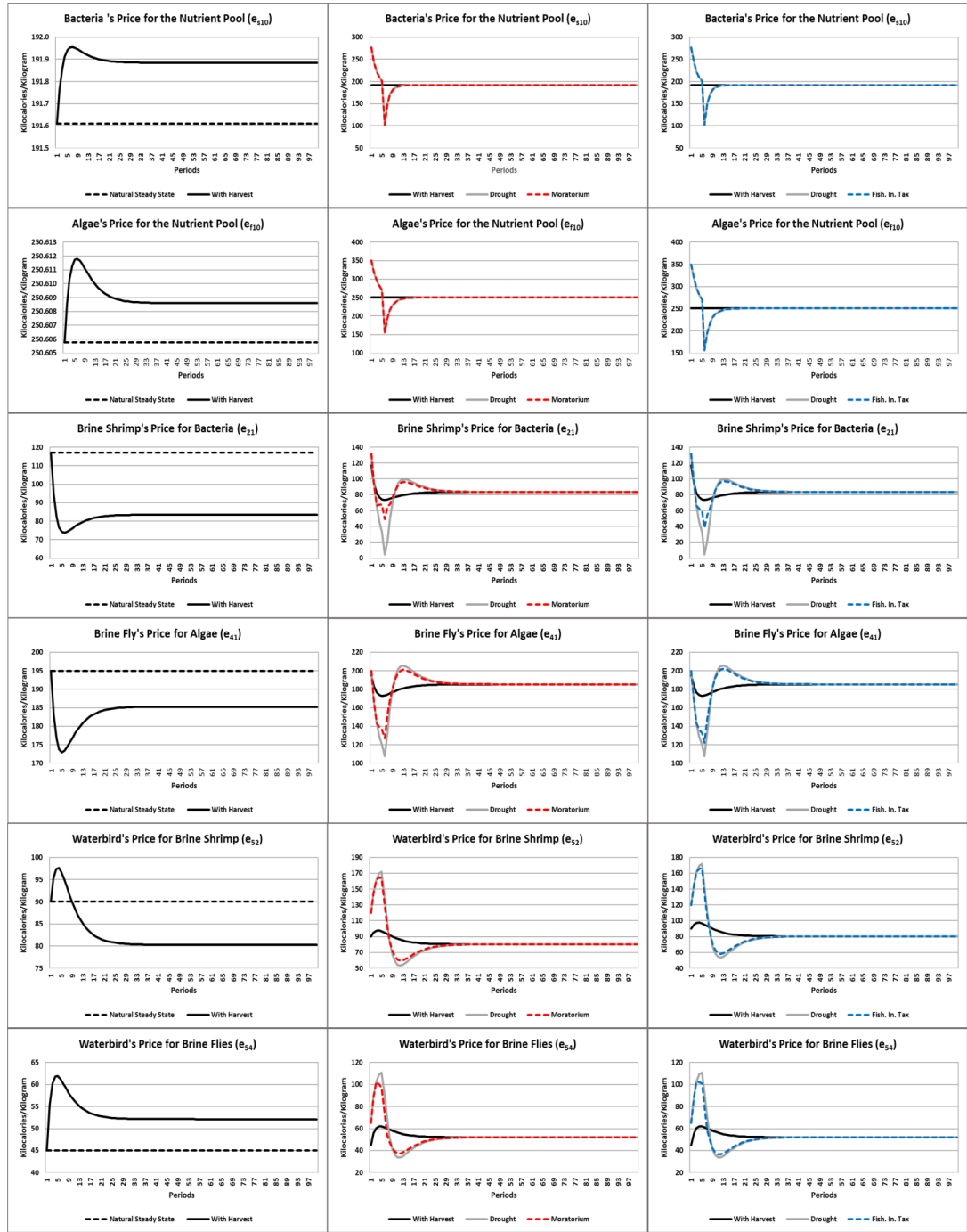


Fig. 4.5: Output Markets

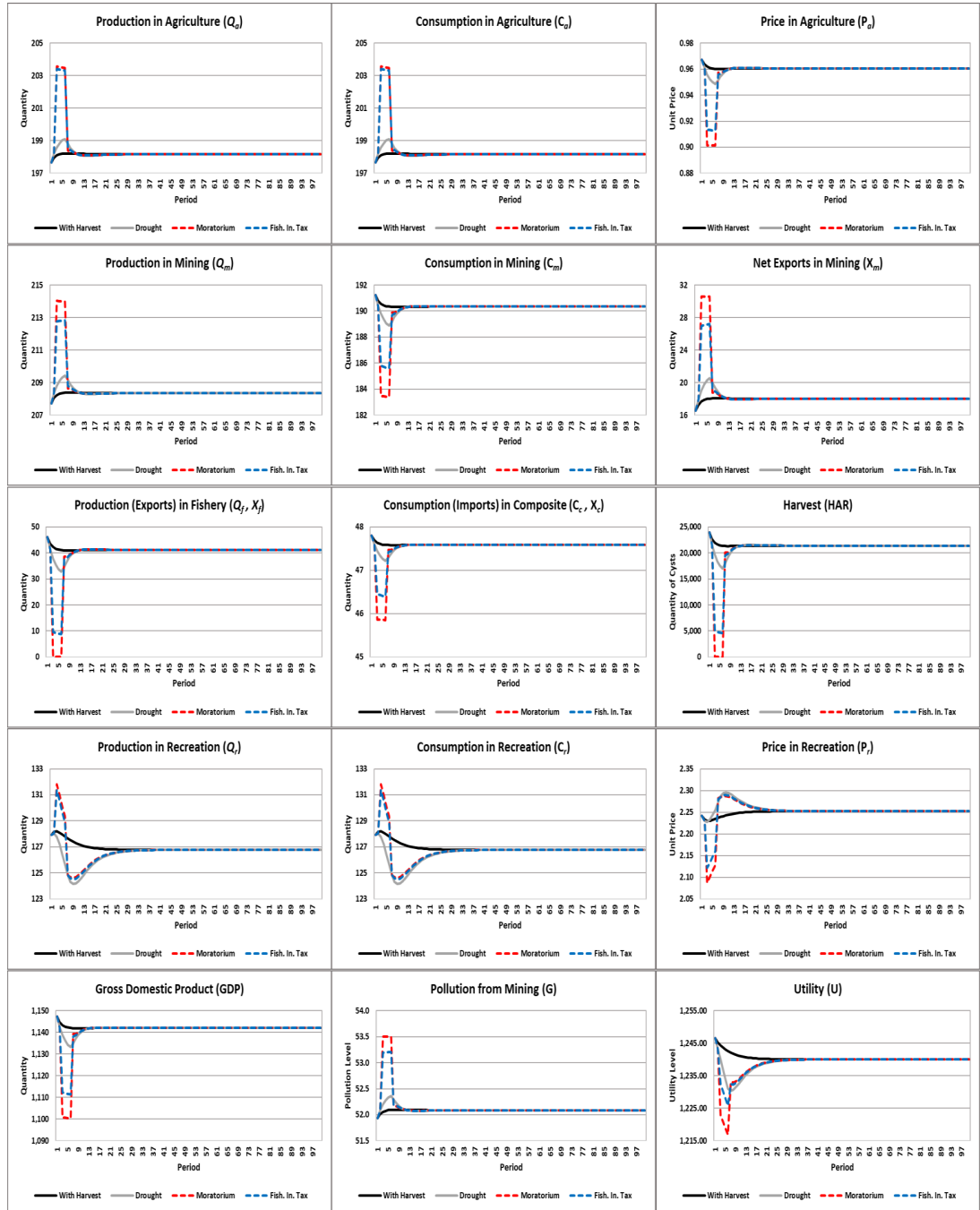


Fig. 4.6: Input Markets

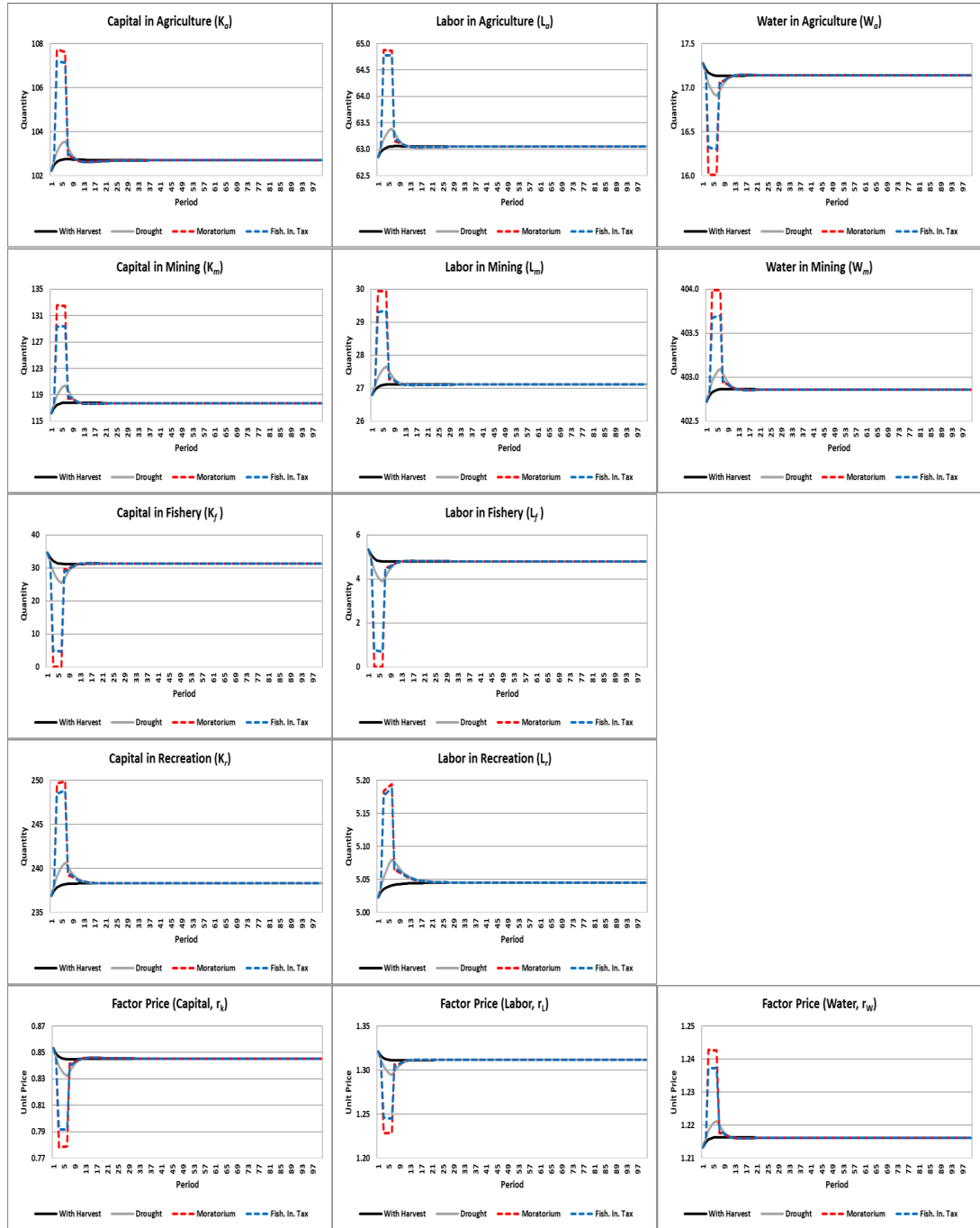


Fig. 4.7: Rolling and Base Equivalent Variation (EV) Paths

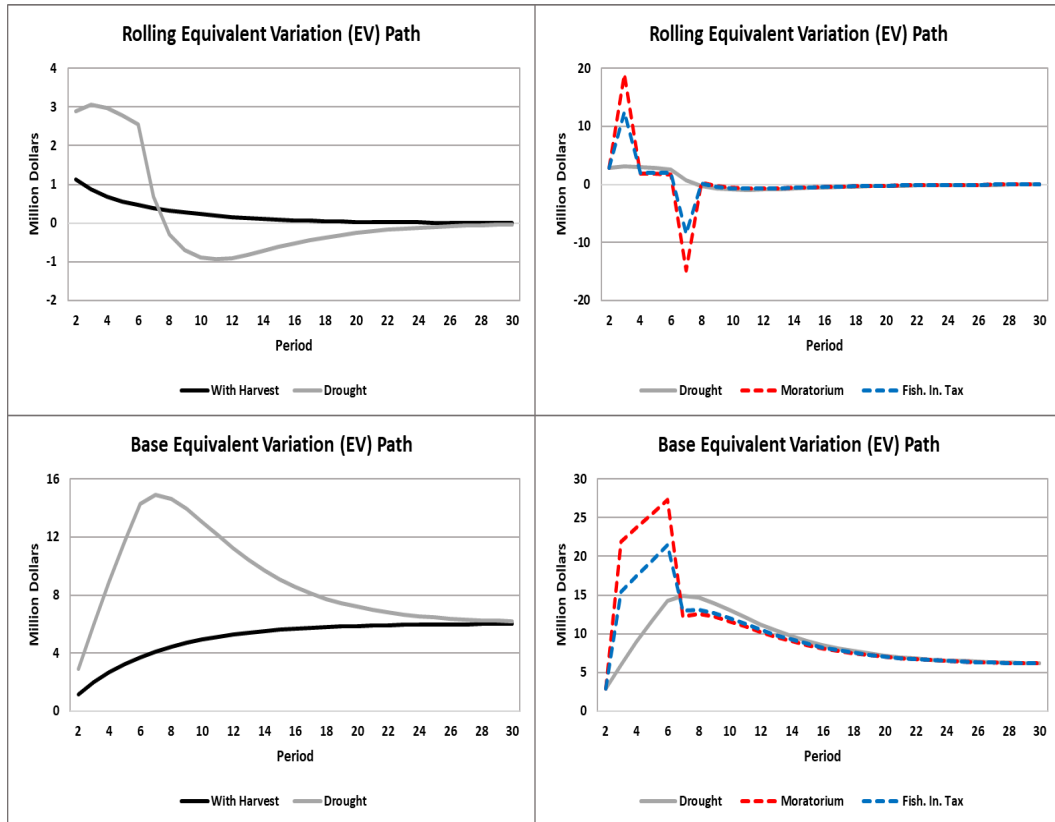


Fig. 4.8: Species Populations: Moratorium and Output Tax on Mining

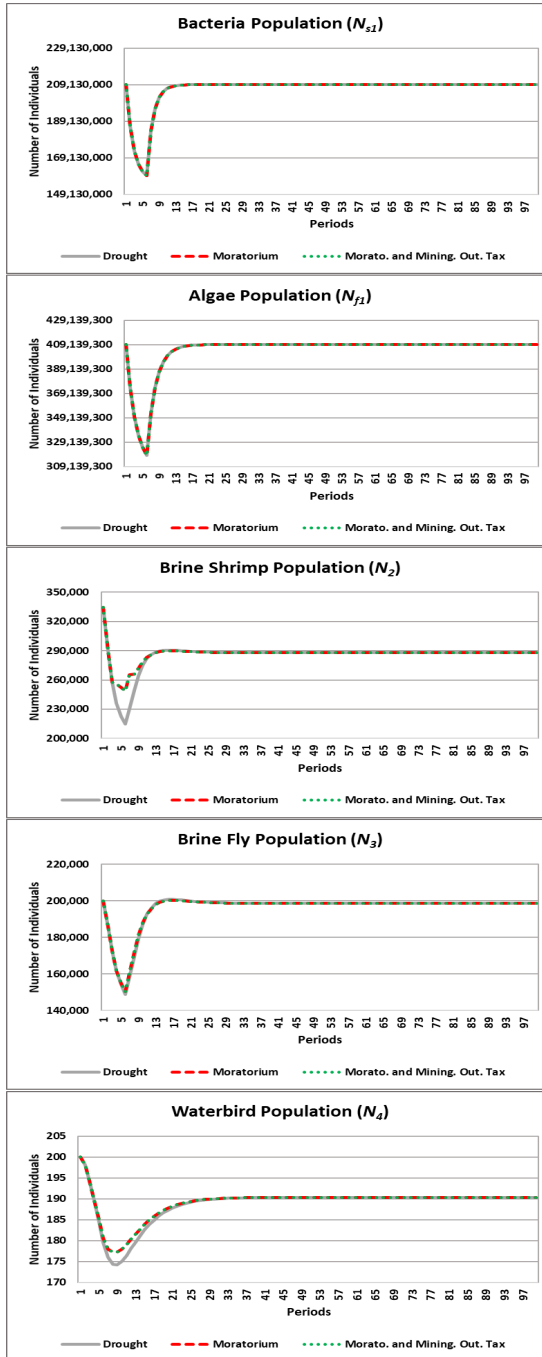


Fig. 4.9: Species Energy Prices: Moratorium and Output Tax on Mining

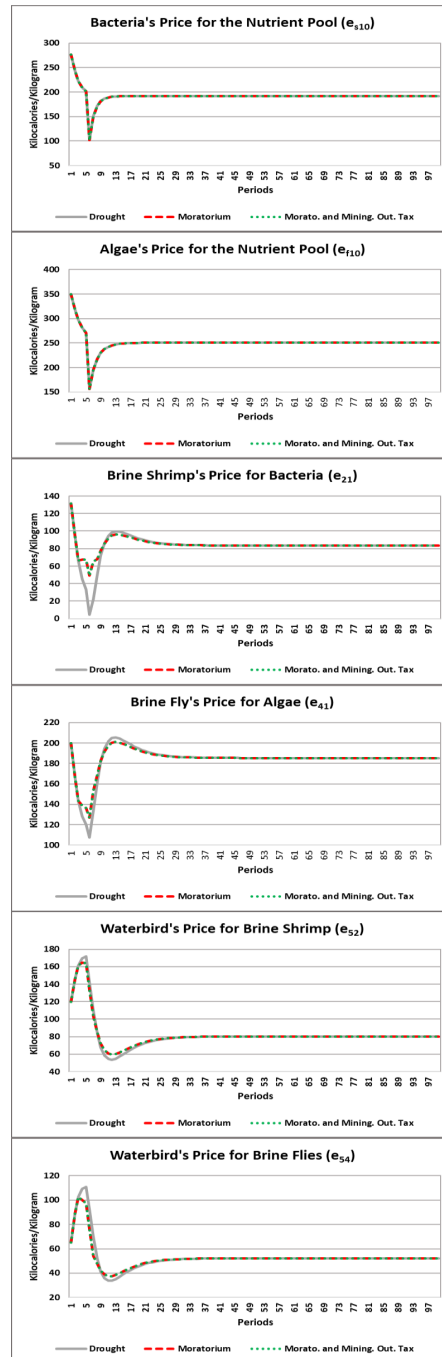


Fig. 4.10: Output Markets: Moratorium and Output Tax on Mining

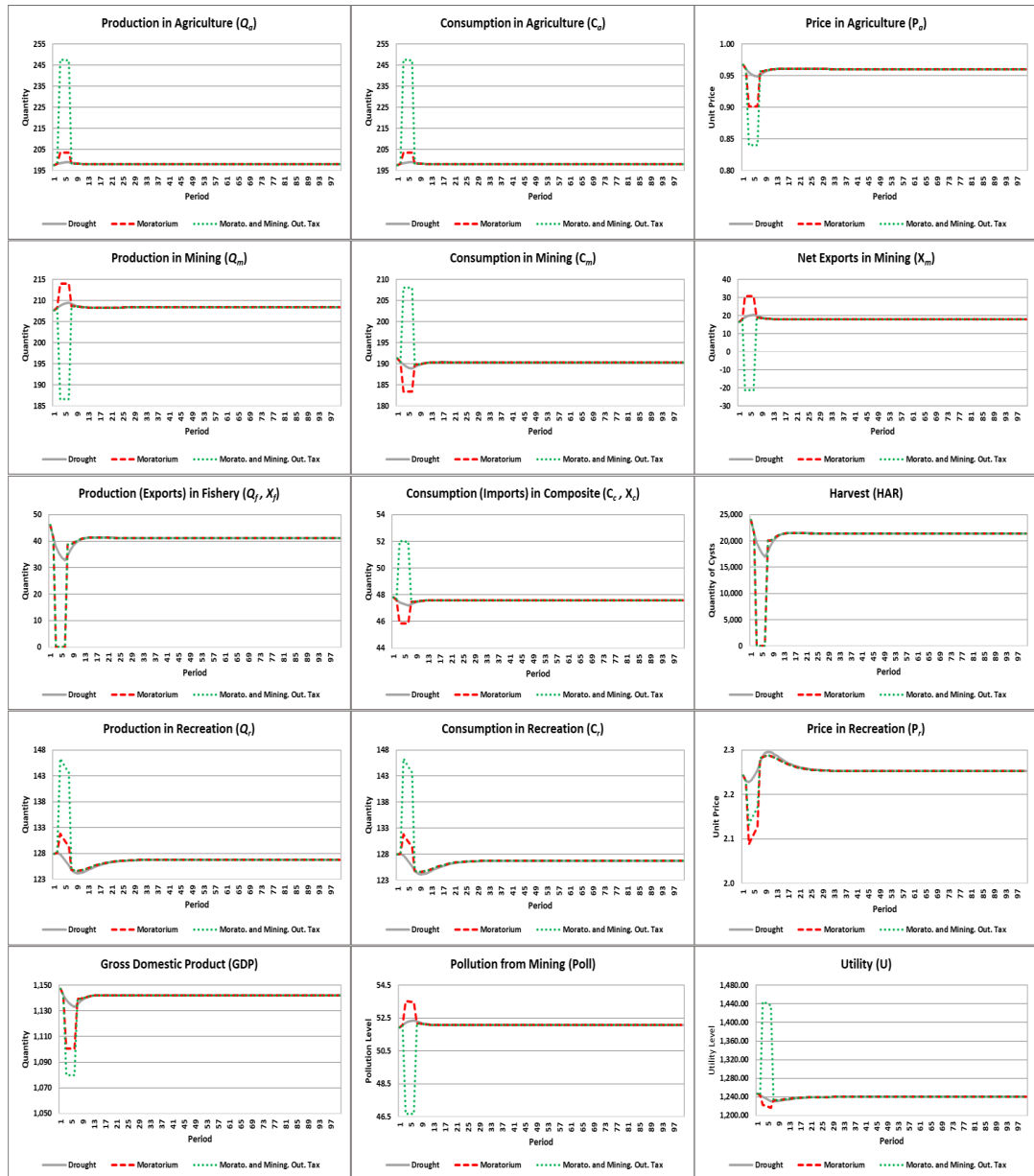


Fig. 4.11: Input Markets: Moratorium and Output Tax on Mining

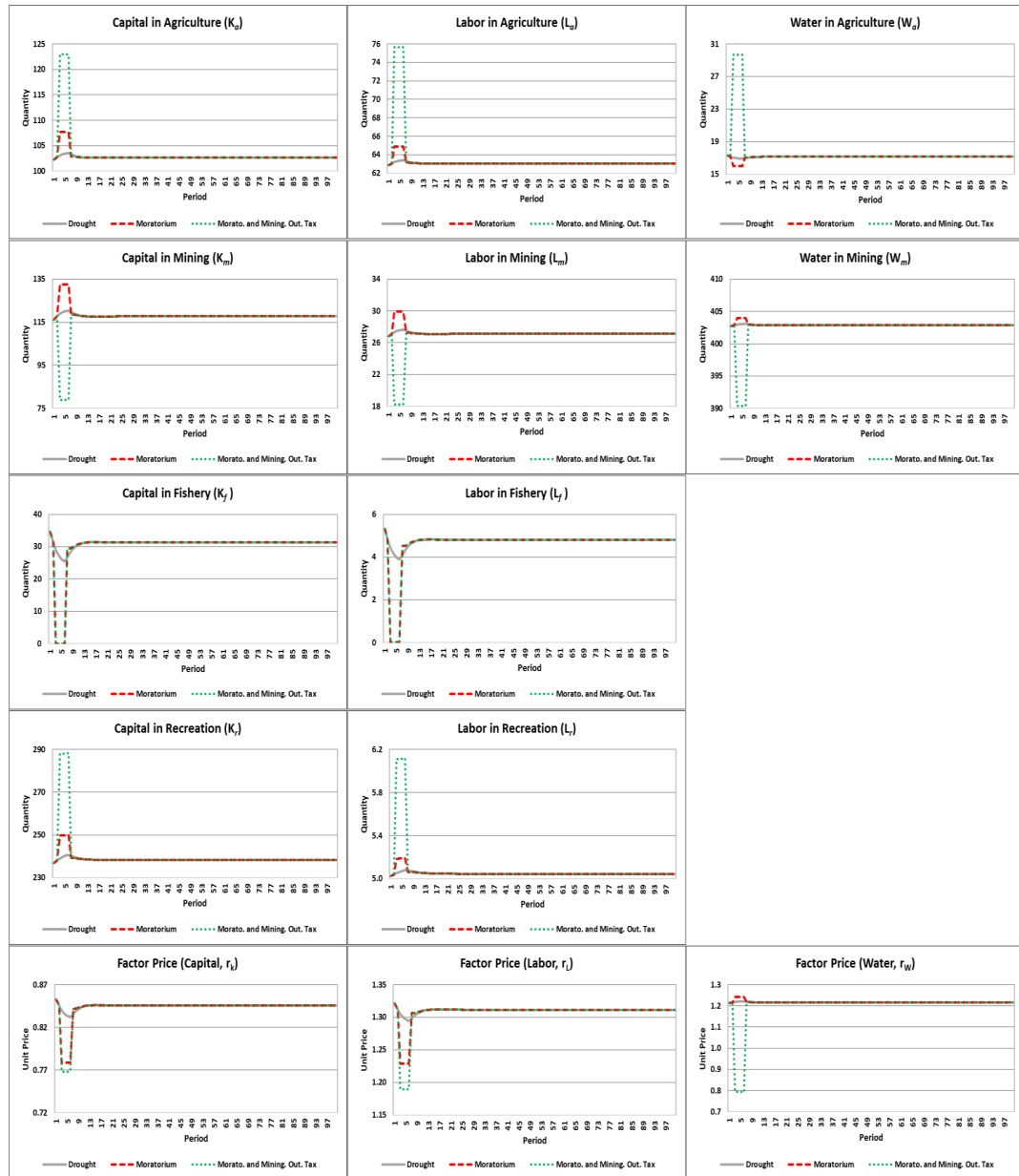
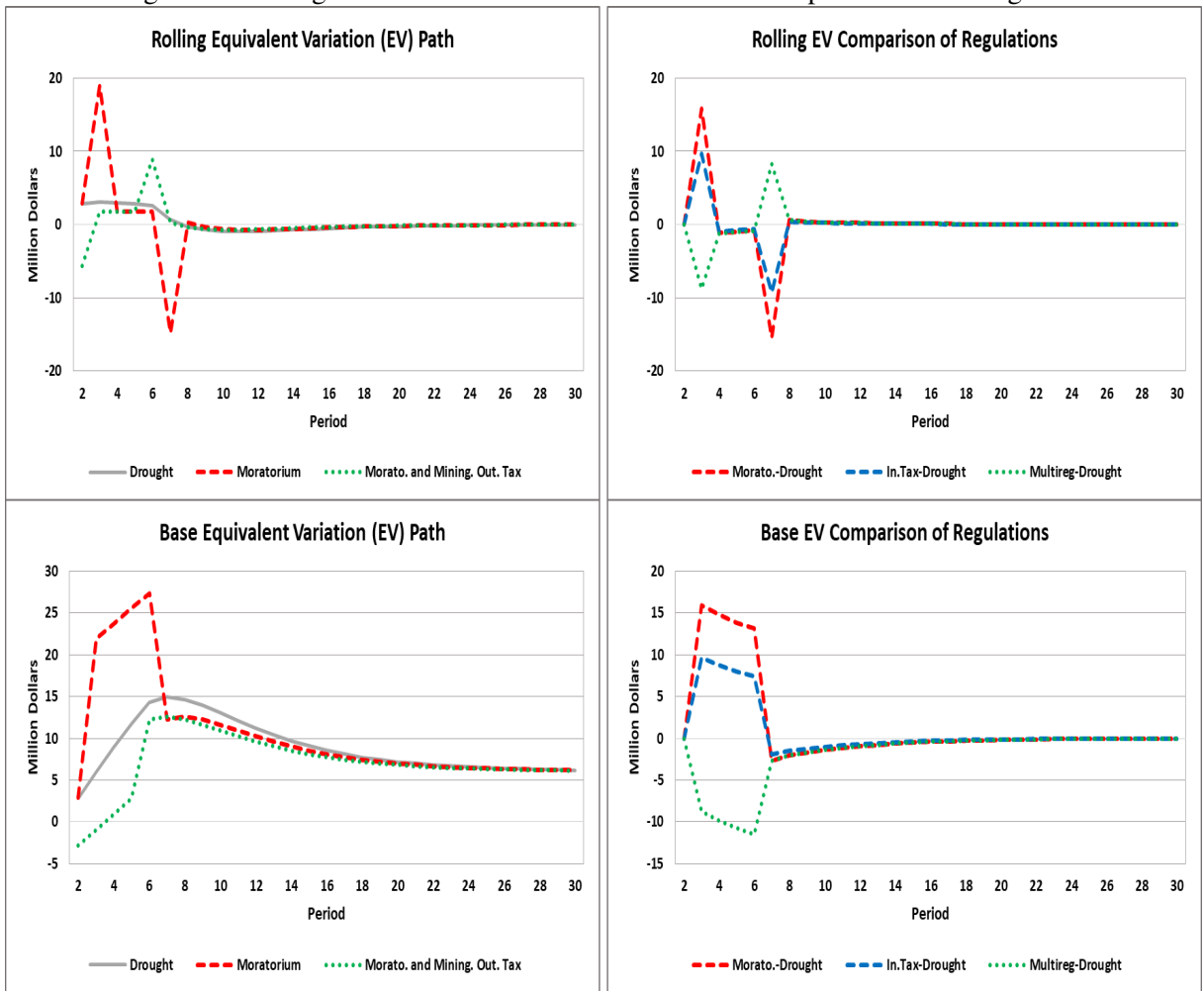


Fig. 4.12: Rolling and Base EV Paths: Moratorium and Output Tax on Mining



CHAPTER 5

SUMMARY AND CONCLUSIONS

This dissertation consists of three essays that explore how natural ecosystem can be integrated with economic system through two case studies of multiple species interactions, or predator-prey relations. Two case studies are carried out with theoretic and empirical aspects. The first case study focuses on the Pacific Ocean between the United States and Canada, where endangered/threatened predators and their commercially valuable prey coexist. The second case study covers the Great Salt Lake in Utah, where the regional economy relies on the saline lakes ecosystem.

The first essay demonstrates the effect of trade liberalization on renewable resources and biodiversity by exploring economic and ecological outcomes of a bi-resource economy consisting of two ecological resources linked via a predator-prey relationship between Resident killer whales and Chinook salmon. In an illustrative general equilibrium model, the predator has a non-consumptive value and the prey has a consumptive value, where those species are specific factors used by the whale watching and fishery sectors. The theoretical result suggests that free trade with a country having a lower ratio of predators to prey leads to conservation of prey and predator stocks for its trading partner, which has a higher proportion of predators to prey. Under certain conditions trade may make conservation via quota unnecessary. Statistical results are suggestive that US imports of Chinook salmon from Canada may serve to increase the population of Southern Resident killer whales.

The second essay investigates the levels of maximum sustainable yield and optimal harvest with the first case study of two Resident killer whales (Northern and Southern) and Chinook salmon. With optimal framework, using the most rapid approach technique, the essay also evaluates whether the existing recovery plan for Southern Resident Killer Whale is feasible given the constraint imposed by the three species predator-prey system. Biological parameters for our predator-prey system are estimated from the seemingly unrelated regression and its iterative form, and economic parameters for our optimal control framework are obtained from various sources in the literature, such as

Chinook salmon fishery and macroeconomic data in Canada and United States. The main findings suggest that the existing recovery plan of achieving 95 Southern Resident Killer Whales by the year of 2020 may not be feasible, that there exist feasible recovery plan scenarios for Southern Resident killer whale starting from 84 whales, and that the non-market values of the killer whale depends on the feasible recovery plan scenarios and those values are measured in Chinook salmon fishing firms forgone fishery.

The third essay develops a general-equilibrium bioeconomic model of the Great Salt Lake ecosystem that tightly links the lakes ecosystem with its regional economy. A computable general equilibrium model is constructed by integrating the lakes ecosystem and regional economy to identify how individual, aggregate ecological and economic components dynamically and interactively respond to a prolonged drought. It is also considered how two fishery-specific regulatory tools might be used to alter the bioeconomys recovery path in the presence of the drought. Four main findings are suggested in this essay. First, the fishery-specific regulatory instruments have relatively strong recovery effects on the brine shrimp population, but weaker effects on other species populations. Second, the reallocation of production in the regional economy caused by these regulations has perverse effects on welfare. Third, these perverse effects associated especially with fishery-specific regulations can be offset via imposition of either an additional input or output tax on mineral extraction. Fourth, two dynamic measures of equivalent variation are graphically shown with prolonged drought, the shrimp harvest moratorium, and a mining output tax.

As mentioned by Anderson and Seijo (2011), an ecosystem management has also an inherent spatial dimension. Characterizing and understanding its physical and bioecological attributes over space are essential. However, such spatial dimensions are beyond the scope of this dissertation. It would be interesting in the future research to understand and analyze the geographic boundaries of population distribution over space, estimate a required habitat size for the ecosystem conservation, and identify the interdependent relationship between species with dynamic geographic patterns.

References

Anderson LG, Seijo JC (2011) *Bioeconomics of fisheries management*. John Wiley & Sons

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 Salt Lake City, Utah
- B.S. in Tourism Development, Gyeongju University February, 2008
 Gyeongju, South Korea

EXPERIENCE

- Research Assistant 2014-Present
 Applied Economics Dept., Utah State University
- Research Assistant 2011-2013
 Applied Economics Dept., Pukyong National University, South Korea

RESEARCH

- Predator-Prey Interaction in a Bi-Resource Economy and the Role of Free Trade, (with Edwards, E.C. and Oladi, R), *Under Review*
- Measuring the Bioeconomic Impacts of Prolonged Drought on a Lake Ecosystem: The Case of the Great Salt Lake, Utah, (with Caplan, A.J.), *Under Review*
- Feasible Recovery Plan under Optimal Fisheries Management: Chinook Salmon and Resident Killer Whale Complex, (with Edwards, E.C. and Oladi, R), *In Progress*
- The Impact of the Great East Japan Earthquake on Inbound Tourism Demand across Countries: Similarities and Differences, (with Pang A., Tejada, H.A., and Kim, M-K.), *In Progress*

PROFESSIONAL SERVICE AND PRESENTATION _____

- Reviewer, the Agricultural and Applied Economics Association (AAEA) Selected Presentations Committee, 2017-2018
- Go, D-H., A., Pang, H.A., Tejada, and M-K., Kim. “The Impact of the Great East Japan Earthquake on Inbound Tourism Demand across Countries: Similarities and Differences.”Invited presentation at the Agricultural and Applied Economics Association (AAEA) Annual Meeting in Chicago, July 30 - August 1, 2017.

UNIVERSITY SERVICE AND HONOR _____

- President of Korean Student Association, Utah State University, 2015-2016
- Year of Student, Utah State University, 2015