

Bioeconomic Models and Sustainable Use of Marine Resources: Three Case Studies

by

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A Dissertation Presented in Partial Fulfillment  
of the Requirements for the Degree  
Doctor of Philosophy

Approved October 2014 by the  
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December 2014

## ABSTRACT

This dissertation consists of three substantive chapters. The first substantive chapter investigates the premature harvesting problem in fisheries. Traditionally, yield-per-recruit analysis has been used to both assess and address the premature harvesting of fish stocks. However, the fact that fish size often affects the unit price suggests that this approach may be inadequate. In this chapter, I first synthesize the conventional yield-per-recruit analysis, and then extend this conventional approach by incorporating a size-price function for a revenue-per-recruit analysis. An optimal control approach is then used to derive a general bioeconomic solution for the optimal harvesting of a short-lived single cohort. This approach prevents economically premature harvesting and provides an “optimal economic yield”. By comparing the yield- and revenue-per-recruit management strategies with the bioeconomic management strategy, I am able to test the economic efficiency of the conventional yield-per-recruit approach. This is illustrated with a numerical study. It shows that a bioeconomic strategy can significantly improve economic welfare compared with the yield-per-recruit strategy, particularly in the face of high natural mortality. Nevertheless, I find that harvesting on a revenue-per-recruit basis improves management policy and can generate a rent that is close to that from bioeconomic analysis, in particular when the natural mortality is relatively low.

The second substantive chapter explores the conservation potential of a whale permit market under bounded economic uncertainty. Pro- and anti-whaling stakeholders are concerned about a recently proposed, “cap and trade” system for managing the global harvest of whales. Supporters argue that such an approach represents a novel solution to

the current gridlock in international whale management. In addition to ethical objections, opponents worry that uncertainty about demand for whale-based products and the environmental benefits of conservation may make it difficult to predict the outcome of a whale share market. In this study, I use population and economic data for minke whales to examine the potential ecological consequences of the establishment of a whale permit market in Norway under bounded but significant economic uncertainty. A bioeconomic model is developed to evaluate the influence of economic uncertainties associated with pro- and anti- whaling demands on long-run steady state whale population size, harvest, and potential allocation. The results indicate that these economic uncertainties, in particular on the conservation demand side, play an important role in determining the steady state ecological outcome of a whale share market. A key finding is that while a whale share market has the potential to yield a wide range of allocations between conservation and whaling interests – outcomes in which conservationists effectively “buy out” the whaling industry seem most likely.

The third substantive chapter examines the sea lice externality between farmed fisheries and wild fisheries. A central issue in the debate over the effect of fish farming on the wild fisheries is the nature of sea lice population dynamics and the wild juvenile mortality rate induced by sea lice infection. This study develops a bioeconomic model that integrates sea lice population dynamics, fish population dynamics, aquaculture and wild capture salmon fisheries in an optimal control framework. It provides a tool to investigate sea lice control policy from the standpoint both of private aquaculture producers and wild fishery managers by considering the sea lice infection externality between farmed and wild fisheries. Numerical results suggest that the state trajectory

paths may be quite different under different management regimes, but approach the same steady state. Although the difference in economic benefits is not significant in the particular case considered due to the low value of the wild fishery, I investigate the possibility of levying a tax on aquaculture production for correcting the sea lice externality generated by fish farms.

Dedicated to My Family

## ACKNOWLEDGMENTS

It is quite overwhelming for me to recall and thank all those people who have contributed to my dissertation. First of all, I would like to thank Dr. Joshua. K. Abbott. The past five years have been quite a bumpy journey for me, but Dr. Abbott always exhibited an uncommon patience and has been providing guidance in the process of completing the dissertation, from discussing a dissertation topic to refining the analysis, formulating a storyline for a complete paper, and even pointing out a comma splice problem. His guidance is greatly appreciated. I would also like to thank Dr. Charles Perrings. Dr. Perrings also has been helping me greatly, from the formulation of my research idea to finding funding and providing insightful feedbacks. The weekly meeting with Dr. Perrings has really speeded up my research progress. Thanks also to Dr. Leah. R. Gerber for her unwavering confidence and support, also for introducing me to the field of whale conservation. My gratitude is also extended to my other Ph.D. committee member, Dr. Rachata Muneerpeerakul for his prompt reply to my matlab questions. I also appreciate Dr. Muneerpeerakul for his insightful comments on my dissertation chapters. Thanks to Dr. Michael Schoon for his constructive advice and open-door policy; it is always pleasant to talk with Dr. Schoon. Last but not the least, my gratitude is also extended to Dr. Eli Fenichel for his support during my study in ASU, the dissertation topic discussion, the insightful feedback on dissertation chapters, and for his straightforward attitude and Christmas presents as well. I would like acknowledge financial support for the last chapter from grant number 1R01GM100471-01 from the National Institute of General Medical Sciences (NIGMS) at the National Institutes of Health.

I could not have made it through the past five years without the support of my friends and family. I owe a debt to my mom, my dad, brother and sister for their love, support and encouragement. I would like to express my appreciation to my friend, Nicholas. C. Hayden for his support and encouragement over the past 10 plus years. I am also grateful for my friend, Daniel Lewis for many hours of insightful discussion, the Lewis family for their friendship and hospitality. I would also like to thank my friend Qiong Wang for helping each other through the hardship in ASU, David Shanafelt for insightful discussion, and the whole Ecoservices group, Gerber lab group. My gratitude is also extended to other friends in Phoenix here to make this journey a pleasant one.

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## Chapter 1

### INTRODUCTION

Policy-makers worldwide have been facing a common problem: how to efficiently manage the protection of environmental and natural resources. For policy makers, economics has been a powerful instrument to evaluate the design and implementation of policies that assure economic, ecology and sustainability. Research into environment management and natural resource protection from the standpoint of economics has generated many insights. Indeed, the close integration of various biophysical components and ecological processes with economic agents' decision behavior provides a useful tool to address environmental and natural resource management problems. It analyses the pattern of resource use and assesses the tradeoff between economic, environmental and natural resource sustainability objectives.

This dissertation consists of three substantive studies, all developing bioeconomic models for marine resources. In fishery science, premature harvesting-also known as growth overfishing, is modeled from the perspective of the yield per fish. The management recommendations from these models, however, are not based on decision theory. I show that a bioeconomic modeling approach that combines ecology and economics, and that considers the feedbacks between them, can be used to improve management of the growth overfishing problem. The first study, therefore, aims to test the economic efficiency of harvesting policies resulting from the conventional framework and from a bioeconomic modeling approach. This study is the first to synthesize the

conventional fishery science approach to growth overfishing. The fishery literature suggests that employing an impulse harvest at a particular time or a constant harvesting harvest for a period can generate maximal biomass yield. In some cases, however, fish size affects the unit price of fish, and the premium can be substantial. To accommodate this, I modify the conventional framework by incorporating this size-price information. There are a handful of fishery studies that include size-price information. These studies, however, treat human harvesting behavior as an external factor to the ecological system, and do not consider the feedback between ecology and economics.

As an alternative, a bioeconomic model is developed, which fully considers the dynamic nature of the system, and addresses the feedback arising from the economic and biological tradeoffs. The bioeconomic literature largely focuses on recruitment overfishing- where the adult fish population is depleted to a level that will not replenish itself- with less effort devoted to growth overfishing. Yet, growth overfishing remains an important issue in fishery management, and there are several studies investigating institutional arrangements to offset the incentive to discard low value fish.

This study adds to the current literature by extending the framework traditionally used in fishery science. The resulting harvest policies, including constant harvest and fishery closures, are not endogenous to the modelling process. On the other hand, a bioeconomic modeling does provide explanations for when and how much to harvest. By comparing the strategies from conventional and modified framework with that from bioeconomic modelling, I am able to test the economic efficiency of the conventional approach used to address growth overfishing.

The yield-per-recruit, revenue-per-recruit, and bioeconomic models are then applied to the Texas shrimp fishery within the state water area. Historically, the open access nature of the Texas shrimp fishery has given rise to excessive investment in capacity, resulting in a “race to fish” smaller shrimp. Large shrimp can fetch a price premium. Large (12-20 counts per pound) brown shrimp can be sold at about 2.4 times the price of small (60-70 counts per pound) shrimp. Intuitively, there may be value to gain by delaying harvest time. My numerical results suggest that there are significant gains to be had by relating the timing of harvest to harvest cost. When shrimp natural mortality is low, I find the gain from a revenue-per-recruit framework is close to that from the bioeconomic approach. However, when natural mortality is high, economic welfare is significantly higher using a harvest strategy resulting from bioeconomic modeling approach.

The second and third studies also use a bioeconomic modeling approach to deal with the problem of resource misuse. While the first study implicitly assumes a sole owner to investigate the potential of bioeconomic modeling, policy makers concerned about the problem of environmental management and natural resource protection have to consider many individual agents or groups. The privately rational behavior of individuals and groups when it comes to environmental management and natural resource protection will in many cases generate externality. The second and third studies investigate externalities created by the privately rational behavior of individuals (or groups). They therefore address an archetypal problem in environmental and resource economics, and follow strands developed in environmental and resource economics to correct the externality problem.



Indeed, environmental and resource economics deals for a large part with market failure due to externality, the public good nature of many environmental resources, and open access to common property resources. Two separate but related strands are provided for the correction of market failure. The first is due to A.C. Pigou (1920) and evolved ever since. In Pigou's example, a factory owner is responsible for the increased cost of washing cloth in a neighboring laundry where the soot falls. Without government intervention, the factory owner will not consider the cost that the factory incurs on its neighboring laundry and will produce until the marginal benefit equals to its marginal cost. The divergence between private and social cost provides a rationale for government intervention. By imposing a tax on factory production through government intervention, the factory owner will have to consider the cost that it inflicts on neighboring laundry and produce until its marginal benefit equals to its marginal cost plus the marginal social cost, and the externality that the factory generates will hence be corrected.

The second strand is due to R. Coase (1960): if the owner of a ranch raises cattle that trample neighboring farmers' crop, who should be responsible for the trampled crop? Coase approached this problem from the perspective of property rights and the role of transaction costs with respect to resource allocation. The insights that Coase's work (1959, 1960) offers into the efficiency of resource use are the following: with well-defined property rights, if transaction costs are sufficiently low, bargaining among parties involved will lead to an efficient outcome; [quote] *the delineation of rights is an essential prelude to market transactions*. Instead of governmental intervention, Coase suggested that the property right approach would allow a market to value the property right.

Following this line of thought, if resources, such as air (Crocker, 1966), water (Dale, 1986) and fish swimming in the ocean, can be delineated as a form of property right to access and emit and allocated to individual agents, then individuals can modify their initial endowments in a cost-effective way through market transactions. This is the intuition behind what have come to be known as “cap and trade” policies for many resources around the world.

The second study examines the long-run ecological implications of an idealized “cap and trade” market for whale conservation. Whale conservation is an impure public good, offering a range of services, including whale products, amenity, recreation, and scientific understanding. Despite the current moratorium policy imposed by the international whaling committee (IWC ) approximately 2,000 whales are harvested each year for “scientific purposes” (about 1,000 by Japan), for commercial use (600 by Norway and Iceland who openly deny the ban) and for subsistence (about 350 by Denmark, Russia and the United States) (Costello et al. 2012).

Gerber et al. (2014a) illustrated the potential of such a market for the management of minke, bowhead, and gray whales. They found that a market could ensure the persistence of whale populations while improving whaler and conservationist welfare. Opponents argue that whale permit market could potentially pose a greater threat to whales than the status quo due to issues of free-riding, incentives for “illegal” harvest and trade, monitoring and enforcement challenges, and increasing costs associated with creating and managing a whale permit market (Smith et al., 2014). These challenges relate to the details of market design, and are important for a well-functioned whale permit market (Gerber et al, 2014b). Indeed,

without credible international governance, whale management proves to be a difficult issue to solve among different users with competing interests, due to the highly migratory or widely dispersed nature of the species.

In my second study, instead of examining a global whale management problem and its economic welfare consequence for different groups involved, I focus on the possibility of a transferable minke whale permit market in Norway and evaluate the steady state ecological consequence of such a permit market. In particular, I consider uncertainty about maximal willingness to pay to harvest whales or to increase whale populations, and about free-riding associated with the public good nature of whale conservation. I present an annual series of static models, which are used to provide a bounded analysis of the economic uncertainties of a whale conservation market.

Although whaling is pursued in several countries around the world, Norwegian whaling has been chosen for this study. This has several advantages. First, the commercial Norwegian minke whaling takes place within its 200 nautical miles of exclusive economic zone in North Atlantic. Thus, this would ensure a credible regulator. Secondly, minke whale in North Atlantic is approaching carrying capacity. In the past decade, approximately 550 minke whales have been taken annually by Norway for commercial use, and the catch quota is set according to a procedure developed by the IWC. The combination of substantial current harvests and high abundance of minke whale stock makes it an attractive, relatively low-risk candidate for developing a whale permit market. Note here that a permit is delineated equivalent to one whale and is different from whale share. In addition, there is a quite well-developed institution in

Norway to manage whaling. This includes whale product DNA testing and registry, and the capacity to impose penalties for illegal whaling.

Specifically, this study aims to investigate the effect of a whale permit market on steady state minke whale population and harvest levels. A Monte Carlo simulation technique is used to construct the demand for whale conservation because the demand for different whale services is poorly known. The study contributes to the literature by providing insights into the economic conditions in which the introduction of a “cap and trade” program would result in harvesting all permits (full exploitation), conserving all permits (pure conservation) or a coexistence between whaling and conservation. The results indicate that economic uncertainties, in particular on the conservation demand side, play an important role in determining the steady state ecological outcome of a whale share market. A key finding is that while a whale shares market has the potential to yield a wide range of possible outcomes—from full exploitation to pure conservation—pure conservation is the most likely outcome.

The third study examines a negative externality problem that is similar with Pigou’s factory and laundry example. Salmon farms located near the migratory route of wild salmon are a potential reservoir for sea lice reproduction. Originally farms are free of lice. When the adult wild stocks migrate back to the parent river for spawning, they carry adult gravid lice. These adult gravid lice reproduce and spread into the farms close-by. They then infest wild juvenile salmon emerging from the parent river. The sea lice transmission from fishing farms to wild fisheries represents a negative externality during the fish farming production process, the value of which is the cost incurred by the wild

fishery. For fish farmers, this is an external cost that is not considered in their decision process without the relevant policy being in place.

The presence of salmon farming of high density and the high density of fish in farms around the coast may change the dynamics of sea lice and wild stocks. While the sea lice ecology is relatively well-known, to the author's knowledge, there are only two economic analyses of sea lice impact. Liu, Sumaila and Volpe (2011) explore the economic impact under fixed exploitation policy and target escapement policy by exogenously and randomly set sea lice induced mortality rates. Abolofia et al. (2014) used data from the Norwegian salmon industry to investigate the biological and economic impact of sea lice on salmon farms. Neither of these two articles incorporates sea lice dynamics within the economic model.

This study adds to the literature by investigating the externality between fish farms and the wild fishery. As far as I am aware, this is the first study to provide a dynamic bioeconomic assessment of the interaction between salmon farms and wild fishery by integrating a discrete macro-parasite and host interaction model with an economic model. It investigates the optimal control policy from the perspective of salmon aquaculture, and then from the perspective of a joint fisheries manager. By taking account the complex relationship between sea lice population, farmed fishery and the wild fishery, I am able to evaluate how much privately optimal treatment behavior deviates from the socially optimal solution, and to assess the economic impact of salmon aquaculture on the wild fishery due to sea lice transmission. In this study I also investigate possible institutional arrangements for sea lice control.

I conclude the dissertation with a summary of each study. This identifies the contributions of each chapter, and discusses their policy implications. It also provides directions for future research.

## Chapter 2

# CONVENTIONAL AND BIOECONOMIC APPROACHES TO ADDRESS PREMATURE HARVESTING

### 2.1 Introduction

There is currently widespread scientific concern that many marine capture fisheries have been overfished (Costello et al., 2012; Pikitch, 2012; Neubauer et al., 2013). Indeed, most marine fishery management is generally concerned with preventing overfishing, where overfishing is defined relative to an acceptable level, biologically or economically. In managed fisheries this might be quota set by the management authority. In unmanaged fisheries it might be the harvest level associated with an efficiency criterion. Overfishing takes two forms. It can occur through recruitment overfishing or harvesting too much of the stock so that potential recruitment is compromised, and premature harvesting, also known as growth overfishing, which reduces potential yield per fish (Haddon, 2001).

The ecological and economic fishery literatures largely focus on recruitment overfishing, with less effort devoted to growth overfishing. Yet growth overfishing is often an important ecological and economic issue in fishery management. Several recent articles argue that growth overfishing truncates the age structure of fish stocks and can lead to both stock fluctuations (Hsieh et al. 2006; Anderson et al. 2008) and evolutionary effects (Guttormsen et al. 2008). Economically, growth overfishing can reduce the profitability of fisheries, both within-season as fish are harvested at a size that fails to maximize their value, and across seasons through its recruitment and evolutionary

feedbacks. Traditionally, growth overfishing is addressed with yield-per-recruit analysis in fishery management. One policy response to this biological threat, derived from the recruit analysis, has been to impose gear restrictions (Quinn and Deriso, 1999). Gear restrictions, however, do not address the economic drivers of growth overfishing.

The problem of growth overfishing has multiple contributing dimensions, including gear restrictions, multiple cohorts (and/or species) of different age structure, and hold size restrictions that induce highgrading. This study abstracts from many of these complexities to focus on a single (homogeneous) annual cohort fishery. While most fisheries are thought of as a renewable resource, many shrimp (Ward and Sutinen 1994; Huang and Smith 2011) and cephalopod fisheries (Grant et al. 1981) do not show a significant relationship between the breeding stock and juvenile recruitment into the fishery (Grant et al. 1981; Wilen 1985; Swallow 1994), thereby justifying the applied value of the focus on a single cohort.

Fish size often affects the unit price of fish (Bjorndal 1988; Arnason 1992; Mistiaen and Strand 1999). The premia can be substantial, and tend to vary by species. In the U.S., for example, large (12-20 counts per pound) brown shrimp can be sold at about 2.4 times the unit price of small (60-70 counts per pound) shrimp. For Atlantic cod, the unit price of large (> 15 lb) fish is approximately 1.4 times that of small fish (2-4 lb) (derived from Smith et al. 2011). It is therefore important to consider the effect of size-based pricing on fishery management (Hilborn and Walters, 1992; Ye 1998), and its impact on yield-per-recruit and the paralleled revenue-per-recruit (Gallagher et al. 2004). These recruit models also treat human harvesting behavior as an external factor to the ecological system, and do not consider the stock-dependent cost of harvest.



As an alternative, bioeconomic analyses treat fishery harvest strategies as endogenous to the jointly determined human-ecological system, and have been widely used both to model the joint systems and inform management strategies (Conrad and Clark, 1976). Several bioeconomic studies incorporate size-price information into their analyses in the context of multi-cohorts (e.g., Zimmermann et al, 2011; Kristofersson and Rickertsen, 2008; Diekert, 2012), and have investigated options for removing the incentive for selective harvesting (Diekert, 2012; Smith and Gopalakrishnan, 2011). The annual cohort is regarded as an economic asset, capable of generating profits through harvesting. There is a sense in which the single annual cohort resembles a single even-aged forest stand. The fisher's problem, like the forester's problem, is to determine the optimal time to harvest the stock<sup>1</sup>.

By focusing on a single cohort fishery I compare the harvest strategies resulting from the yield-per-recruit and revenue-per-recruits frameworks traditionally used in fishery sciences with the harvest strategies obtained from a capital theoretic framework that solves the problem of when to harvest a physically growing fish stock when a size-price function is included. By comparing these strategies I am able to show how the socially optimal harvest (and hence the optimal intervention) depends on the modeling approach. I illustrate the full range of approaches through a numerical study of the Gulf of Mexico shrimp fishery.

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<sup>1</sup> The solution to the forester's problem was first found by Martin Faustmann in 1849, who calculated the optimal rotation for a forest stand. Later, Irving Fisher (1907) proposed a solution for a single rotation model in which the stand is harvested when the growth in its value is equal to interest rate. Each of these results can be seen as special cases of the foundational principles stated in their modern form by Harold Hotelling in 1931.

## 2.2 Modeling Approaches

### 2.2.1 Yield-per-Recruit Analysis and Extensions

Conventional yield per recruit (YPR) analysis focuses on the potential yield from a fish cohort, and specifies the effect of selection patterns and fish harvesting strategies on yield. It provides a biological reference point for fishery management policy (Pereiro 1992). I first examine the basic structure of yield-per-recruit, which I refer to as the “conventional” approach to growth overfishing. For a short-lived species, which does not show a significant relationship between spawning stock level and recruitment, maximum yield-per-recruit is analogous to MSY. When fishing effort is not bounded above, the maximum yield-per-recruit is achieved by applying an impulse or “knife edge” harvest at the age  $t_y^{\max}$  when the cohort biomass per recruit is maximized (Getz and Haight 1989; Quinn and Deriso 1999).

Consider an initial fish population stock level  $x_0$ , which is also referred to as recruitment for a short-lived species, like shrimp, having a known von Bertalanffy growth function,  $w(t) = w_\infty \left[ 1 - e^{-k(t-t_0)} \right]^\beta$ , where  $w_\infty > 0, k > 0, t_0 \geq 0$ , and  $\beta$  lies between 2.5 and 3. A value close to 3 implies that individual fish grow according to the exponential function  $\left( 1 - e^{-k(t-t_0)} \right)^\beta$  in three dimensions (width, length, height), ensuring that the von Bertalanffy function satisfies  $\frac{d(\dot{w}w^{-1})}{dt} < 0$  for  $t \geq 0$ , which implies the proportional rate of weight growth decrease with age (Getz and Haight 1989).

I assume that natural fish mortality is  $m$ . When there is no harvesting, the change of stock is therefore,  $\dot{x} = -mx$ . In the case of no upper bound for fishing mortality, the objective is to choose a harvesting time  $t$  to maximize yield-per-recruit. Mathematically,

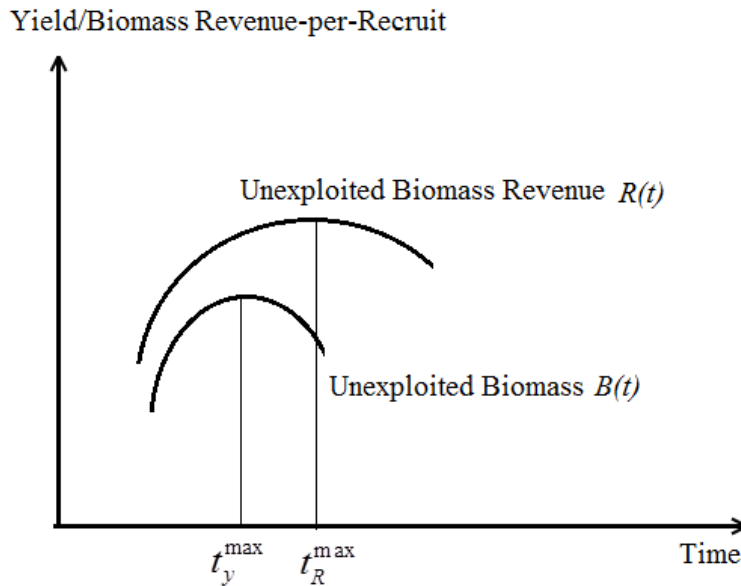
$$\max_t w(t) * x(t)$$

The first order condition with respect to  $t$  requires  $\dot{w} = wm$ . By substituting the von Bertalanffy growth function described above, time  $t_y^{\max}$  is found to be

$$t_y^{\max} = t_0 + \frac{1}{k} \ln \left( 1 + \frac{\beta k}{m} \right)$$

When fish are small, their per-capita growth is sufficient to offset the loss of biomass from natural mortality. Therefore, fisheries managers aiming to maximize YPR would prohibit harvest during this period. As biomass increases, the marginal biomass ( $\dot{w}$ ) gain associated with individual growth approaches the marginal biomass loss ( $wm$ ) due to natural mortality. With unbounded fishing effort, fisheries managers would wait until the single cohort biomass peaks at age  $t_y^{\max}$ , at which time  $\dot{w} = wm$ , and then harvest the entire cohort at this exact moment using an impulse harvest (Fig. 2.1). If the manager waits any longer, the marginal biomass loss from mortality associated with that delay exceeds the marginal biomass gain from individual growth. Alternatively if the harvest time is accelerated, then the foregone biomass gain exceeds the gains from reduced mortality.

Figure 2.1. Yield-per-Recruit versus Revenue-per-Recruit



Note: Impulse harvesting applies in both cases, but the harvest time is delayed in the revenue-per-recruit case in order to achieve maximum revenue.

Next consider the incorporation of an increasing size-price function into this traditional yield-per-recruit analysis. Define the revenue function for a single fish at age  $t$  as  $\psi(t) = p(w(t))w(t)$ , where  $p(w(t))$  is the unit price for fish with an individual weight function  $w(t)$ . The function  $p(w(t))$  is monotonically increasing. I assume that when their weight is very small, individual fish have no economic value so this function  $p(w(t))$  goes through the origin<sup>2</sup>, and takes the form  $p(w(t)) = bw(t)$ ,  $b > 0$ .

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<sup>2</sup> It might be more realistic that a minimum weight is required in order to achieve market value, which suggests that there is a negative intercept term for this  $p(w(t))$  function. This does not change the qualitative result, but the mathematical expression for  $t_R^{\max}$  will be messy. For the sake of tractability, the  $p(w(t))$  function is rescaled to pass through the origin.

Incorporating revenue considerations into the traditional yield-per-recruit model structure when there is no upper bound for fishing mortality, results in a model where the fishery manager's task is to choose a harvesting time  $t$  to maximize revenue-per-recruit<sup>3</sup>.

$$\max_t R(t) = \psi(t) * x(t) \quad (2.1)$$

Taking the derivative with respect to  $t$ , setting it equal to zero and substituting  $\dot{x} = -mx$ , yields

$$\dot{\psi}(t) = m\psi(t) \quad (2.2)$$

Time  $t_R^{\max}$  can be solved<sup>4</sup> by substituting the von Bertalanffy growth function described

$$\text{previously into Eq (2.2), } t_R^{\max} = t_0 + \frac{1}{k} \ln \left( 1 + \frac{2\beta k}{m} \right)$$

At this particular time  $t_R^{\max}$  the marginal revenue gains due to the biomass and unit price growth is equal to the marginal revenue loss due to natural mortality. At this particular point that an impulse harvesting strategy is applied in order to maximize revenues.

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<sup>3</sup>If the revenue-per-recruit analysis acknowledges that delaying harvest bears an opportunity cost, embodied by the discount rate  $\delta$ , the problem here is essentially identical to an optimal single rotation model (with costless harvest) in forestry economics,  $\max_t R(t) = e^{-\delta t} \psi(t) * x(t)$ . First order condition suggests

$\dot{\psi}(t) = (m + \delta)\psi(t)$ . This is very similar to Fisher's solution in that it is optimal to

harvest the entire cohort when the value of growth rate,  $\frac{\dot{\psi}(t)}{\psi(t)}$ , is equal to the mortality-

adjusted discount rate,  $m + \delta$ , where mortality is equivalent conceptually to the depreciation rate of the natural asset. This also suggests that discount rate has the same effect on harvest as mortality. The adjusted optimal harvest time is then,

$$t_R^{\max} = t_0 + \frac{1}{k} \ln \left( 1 + \frac{2\beta k}{m + \delta} \right). \text{ The greater the discount rate, the sooner the impulse harvest.}$$

As long as it is economically viable, an infinite discount rate will lead to a harvest decision at time  $t_0$ .

<sup>4</sup> Analytical solution for the "optimal" stopping time might not be available for some complex size-based price function.

Equation  $\psi(t) = m\psi(t)$  indicates the “optimum” harvesting strategy is to wait until all fish grow to age  $t_R^{\max}$ , and then to apply an impulse harvest (Fig. 2.1) at  $t_R^{\max}$ . The reason is that after  $t_R^{\max}$  the opportunity cost of delaying harvest due to mortality is greater than the appreciation rate in the value of the fish stock resulting from fish growth and the resulting increase in unit price. Note that this “optimum” harvesting strategy may not apply to an age-structured fishery. It also abstracts from some of the most important determinants of any harvesting strategy, such as fleet capacity and cost. Even though the coefficient  $b$  of size-price function does not enter the solution for “optimal” stopping time due to the linear assumption between weight and price, it does suggest that a size-price function will impact the “optimal” stopping time.

Without considering harvesting cost or fleet capacity, when the size-price relationship is incorporated into conventional yield per recruit analysis, the optimal harvesting time for an impulse harvest<sup>5</sup> is delayed from  $t_y^{\max}$  to  $t_R^{\max}$ . The delay is due to the fact that at  $t_y^{\max}$  the growth in the value of the stock still exceeds the opportunity cost of waiting because of the price premium on large-size fish. When the discount rate is

neglected, the time lag,  $\frac{1}{k} \ln(1 + \frac{\beta k}{m + \beta k})$  may be significant, depending on the

parameters of the Bertalanffy function and the natural mortality rate. In essence, while

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<sup>5</sup> In reality, maximum yield-per-recruit according to the YPR-model always involves exploring the combination of a probable range of two decision variables, a time-invariant harvesting mortality and the age of first capture (Getz and Haight 1989; Quinn and Deriso 1999, Jennings et al., 2001). The appendix (A) provides general equations for yield and revenue-per-recruit analysis for constant fishing mortality, which involves exploring fishing mortalities and fishing season opening and closing dates in order to maximize yield and revenue conditional on operating at maximum capacity when the season is open.

the relationship between biomass gain and biomass loss at a particular moment depends only on the biological growth function, the relationship between biomass-revenue gain and biomass-revenue loss depends both on the biological growth function and the size-price function.

Compared to conventional yield per recruit analysis, the paralleled revenue-per-recruit analysis combines a traditional biological based fishery management metric with a metric that jointly accounts for biological and some economic concerns. However, it fails to fully capture economic reality in at least three respects. First, the stock-dependent costs of harvest are not considered in revenue per recruit analysis. Second, revenue-per-recruit analysis fails to consider the opportunity cost of delaying harvest (i.e. discounting). Third, it does not allow for the fact that effort is bounded, requiring the solution of an optimal “window” for harvest rather than an impulse harvest. A capital theoretic (bioeconomic) approach addresses each of these shortcomings by considering the dynamic nature of the system, and addressing the feedbacks arising from the tradeoffs made between economic and biological variables. By comparing harvest strategies under yield per recruit, revenue-per-recruit and bioeconomic approaches, I am able to test the economic efficiency of the measures conventionally used to address growth overfishing.

### 2.2.2 A Capital-Theoretic Framework with an Increasing Size-Price Function

The social planner’s problem in the single, short-lived cohort case is to maximize the net present value of the fishery by choosing fishing effort  $E(t)$  and a terminal period  $T$  within a single planning season.. The state variable of concern is the biomass of shrimp  $B(t) = x(t)w(t)$ . With a known catchability coefficient  $q$ , fishing mortality is given by

$f(t) = qE(t)$  where  $0 \leq E(t) \leq E^{\max}$ . Assuming a Schaefer harvest function

$y = qE(t)B(t)$ , cost per unit fishing effort of  $c$ , biomass of  $x(t)$ , and a discount rate of

$\delta$ , the social planner's objective function can be expressed as

$$\max_{E(t)} \int_{t=0}^T [p(w(t))qE(t)B(t) - cE(t)] e^{-\delta t} dt \quad (2.3)$$

This maximization is conducted subject to the underlying biomass dynamics, in which

biomass declines due to fishing effort  $E(t)$  and natural mortality  $m$ ,

$$\dot{B}(t) = B(t) \left[ \frac{\dot{w}}{w} - m - qE(t) \right] \quad (2.4)$$

The fisher's problem can be solved using the maximum principle with inequality

constraints on the control effort (Leonard and Long, 1992). The current value

Hamiltonian is

$$\tilde{H} = p(w(t))qE(t)B(t) - cE(t) + \lambda(t) \left[ \frac{\dot{w}}{w} - m - qE(t) \right] B(t) \quad (2.5)$$

And the Lagrangean is

$$L(t) = \tilde{H}(t) + \mu(t)(E(t) - E^{\max}) + \eta(t)E(t) \quad (2.6)$$

where  $\lambda(t)$  is a costate variable denoting the shadow price of the stock (the opportunity

cost of harvest, the value of leaving a single fish in the water), and  $\mu(t)$  and  $\eta(t)$  are the

values of loosening in the constraint associated with upper and lower bound of fishing

effort, respectively. An optimal solution to the problem requires that four conditions be



met. The first, the maximum principle, requires the fishing effort level maximizes  $\tilde{H}$  subject to the constraint that  $0 \leq E(t) \leq E^{\max}$ . In term of Lagrangean, this means

$$L_E = \tilde{H}_E - \mu(t) - \eta(t) = 0$$

With Kuhn-Tucker condition  $\mu(t) \geq 0, (E^{\max} - E(t)) \geq 0, \mu(t)[E^{\max} - E(t)] = 0,$

$\eta(t) \geq 0, \eta(t)E(t) = 0$ ; and

$$\tilde{H}_E = p(w(t))qB(t) - c - \lambda(t)qB(t) \quad (2.7)$$

The marginal benefit of additional biomass harvest can be derived as

$p(w(t)) - c / qB(t)$ . By comparing this marginal benefit with the shadow price  $\lambda(t)$ , the fishing season can be determined.

The second, a condition on the evolution of the shadow price of the resource, requires that the growth of the shadow price is equal to the rate of return on investment in the fish stock.

$$\dot{\lambda}(t) = \delta\lambda(t) - H_B = \lambda(t) \left[ \delta + m - \frac{\dot{w}}{w} \right] + qE(t)(\lambda(t) - p(w(t))) \quad (2.8)$$

Here the discount rate or social rate time preference,  $\delta$ , is typically regarded as the opportunity cost of investment in the fish stock, and natural mortality is treated as a form of capital depreciation.

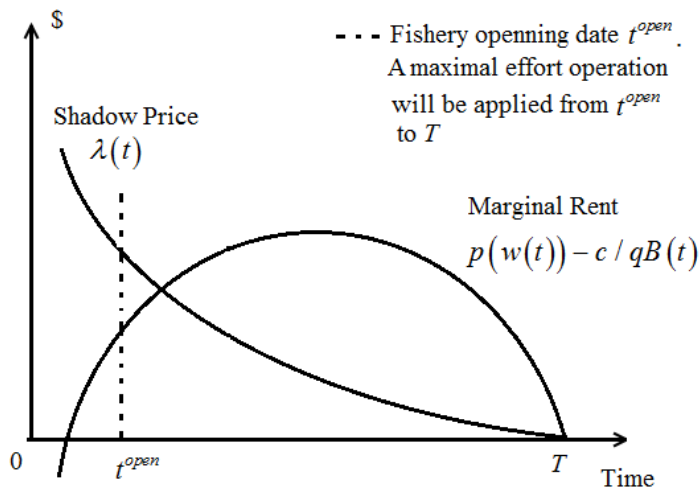
The third condition is that the state equation (Eq 2.4) must be satisfied. The fourth condition that needs to be met, the so-called transversality condition, defines the value of the fish stock at the terminal time (when fishing ceases). The terminal period of the problem and the terminal stock are both decision variables to the fishery manager.

Therefore, there are two transversality conditions (Leonard and Long, 1992) in this case. The first transversality condition is,  $H(T) = 0$ , due to the free terminal time and absence of a ‘scrap value’, stating that the economic benefit will be equal to zero at the terminal time  $T$ . The second transversality condition, required to find the optimal terminal biomass, is  $\lambda(T) = 0$ . Taken together, incorporating these two transversality conditions into Eq (2.5) imply a singular solution ( $\tilde{H}_E = 0$ ) at time  $T$ ,  $\lambda(T) = p(w(T)) - \frac{c}{qB(T)} = 0$ , or  $p(w(T))B(T) = \psi(T)x(T) = c/q$ .

To summarize, before the harvesting season opens, the opportunity cost of fishing is too high and it is worthwhile letting the fish stock grow. This is the period that no fishing effort is applied because  $p(w(t)) - c/qB(t) < \lambda(t)$ . With a growing stock, natural mortality will take effect and the shadow value of the stock,  $\lambda(t)$ , will decrease. When  $p(w(t)) - c/qB(t) = \lambda(t)$  and fishing capacity is unbounded, the optimal solution will be to apply a pulse harvest (also a singular solution) at that instant. Since a bounded fishing capacity prevents a pulse harvest in a single period from driving marginal profit to zero, the fishery manager must start harvesting earlier in order to satisfy the transversality condition. The manager will start harvesting at the point where the maximal effort from that point will exactly take all the profitable fish until the terminal time  $T$ , where  $p(w(T)) - c/qB(T) = \lambda(T) = 0$ . This suggests that the fishery opening date, denoted by  $t^{open}$ , is also determined by the terminal time  $T$  and the maximal effort. So, the point where effort turns on is not driven by crossing the singular solution.

Instead it comes from the upper bound on effort, which prevents an optimal stopping, "pulse" outcome. After  $T$  the shadow value of the fish stock will be equal to zero, and no fishing effort will be applied. Therefore, the whole season is divided into three intervals: no harvest, then harvest at maximal effort in the fishing season until terminal  $T$ , and then no harvest (Figure 2.2).

Figure 2.2. Three Stages in the Optimal Solution: No Harvest  $[0, t^{open}]$ , Maximal Effort Harvest  $(t^{open}, T]$ , and then No Harvest after  $T$ .



Note: The illustrative fishery opening date  $t^{open}$  should satisfy that maximal effort operation from  $t^{open}$  to  $T$  will ensure that shadow price and marginal rent at  $T$  are both equal to zero. The marginal rent is negative in the beginning due to no value related to the small shrimp but a positive value of biomass  $B(t)$ , parameter  $c$  and  $q$ .

### 2.3 Exploring the Difference between the Conventional and Capital Theoretic

#### Approaches: a Numerical Example

In this section, I will use a numerical example, based on the Texas shrimp fishery, to illustrate the implications of the different approaches described in section II for fishing

profits, harvest and fish stocks. Historically, the fishery manager has determined the opening/closing date, catch limit and location in order to improve the economic gain. It is hoped that the study can show how a capital theoretic approach can strengthen management options.

This shrimp fishery lies within Texas' territorial waters. From the shoreline seaward to nine nautical miles, the fishery is managed by the Texas Parks and Wildlife Department (TPWD). Beyond that it is regulated by the National Marine Fisheries Service and the Gulf of Mexico Fishery Management Council (GMFMC) (from nine to 200 nautical miles from the Texas coast) (TPWD 2002). This fishery was selected because it has been one of the most profitable 'single-species' fisheries (Johnson and Libecap 1982), and growth overfishing is a well-documented concern in the fishery (Onal et al. 1991; Nance et al. 1994; Ward and Sutinen 1994; Cailouet et al. 2008). Historically, growth overfishing was attributed to the open-access nature of the Texas shrimp fishery (Condrey and Fulley, 1992). Later on, it became a regulated access fishery, with harvesting rights delineated between inshore and offshore shrimpers. The fishery has many characteristics observed in other common pool resources, such as "the race to fish", and overinvestment in capacity (Johnson and Libecap 1982), such as increasing boat length and/or engine horsepower. Inshore shrimpers have exerted increasing effort to catch larger numbers of smaller shrimp, which has led to a reduction in the number of larger shrimp available to both inshore and offshore shrimpers (Funk et al 2003). With a premium unit price for larger shrimp, correcting this growth overfishing problem could potentially enhance the fishery's economic value.

The Texas fishery exploits three shrimp species: white, pink, and brown shrimp. These three shrimp species have similar life histories, and live approximately one year, but differ in terms of the timing of their migration between the Bay and the Gulf area. I focus on brown shrimp, which make up nearly three-quarters of the total catch (TPWD 2002).

Adult brown shrimp spawn in Gulf waters, and the post-larvae are carried to inshore waters from February to April. The juveniles then migrate from bay areas to the Gulf where the same process is repeated. The shrimp fishery was traditionally open access, and was initially an offshore industry. In the early days, the government ensured that all citizens could get access to this state-owned fishery, and rejected agreements among shrimpers for violation of anti-trust law (Johnson and Libecap 1982). When the inshore shrimp fishery first arose in the 1950s, conflict occurred between inshore and offshore shrimpers because inshore shrimpers harvested immature shrimp before they could move offshore. This conflict was exacerbated by the influx of Vietnamese fishermen, refugees as a result of the Vietnam war, to the Texas coast (Johnson and Libecap 1982). Historically, management policies favored offshore shrimpers with the goal of increasing the total harvested value by more harvesting larger shrimp (TPWD 2002). The Texas Shrimp Conservation Act adopted in 1959 initially defined the specific harvesting seasons for inshore and offshore shrimping. The act has since been amended and modified many times under influence of lobbyists for the inshore and offshore fisheries, which represent two different user groups, and by other interested parties. The open access nature of the Texas shrimp fishery had led to overinvestment in capacity (Johnson and Libecap 1982; Griffin et al. 1976). To offset this effect, the Texas

Legislature established an inshore shrimp license limitation plan in 1995 that included a moratorium on the sale of commercial shrimp harvest licenses, and the development of an inshore voluntary license buyback program.

Currently, the Texas inshore (bay) shrimp spring season runs from approximately May 15<sup>th</sup> to July 15<sup>th</sup> with a bag limit but no size limit, and fall open season runs from approximately August 15<sup>th</sup> to November 30<sup>th</sup> without a bag limit, and with no size limit for the winter open season. Commercial bait shrimp boats are allowed to operate in major bay areas all year round with a bag limit but with no size limit. For the offshore fishery (Gulf shrimp fishery), there is a traditional closed season from May 15<sup>th</sup> to July 16<sup>th</sup> to protect the small emigrating brown shrimp. The open season varies for different fishing locations in both the Northern and Southern shrimp zones, but generally imposes no bag or size limits. The season dates are reported to be based on the evaluation of the biological and economic information to maximize the benefit to the industry, and may vary depending on bay trawl and bag seine catch rates (TPWD 2012).

#### 2.4 Results: Numerical Study for the Brown Shrimp Fishery in the Gulf of Mexico

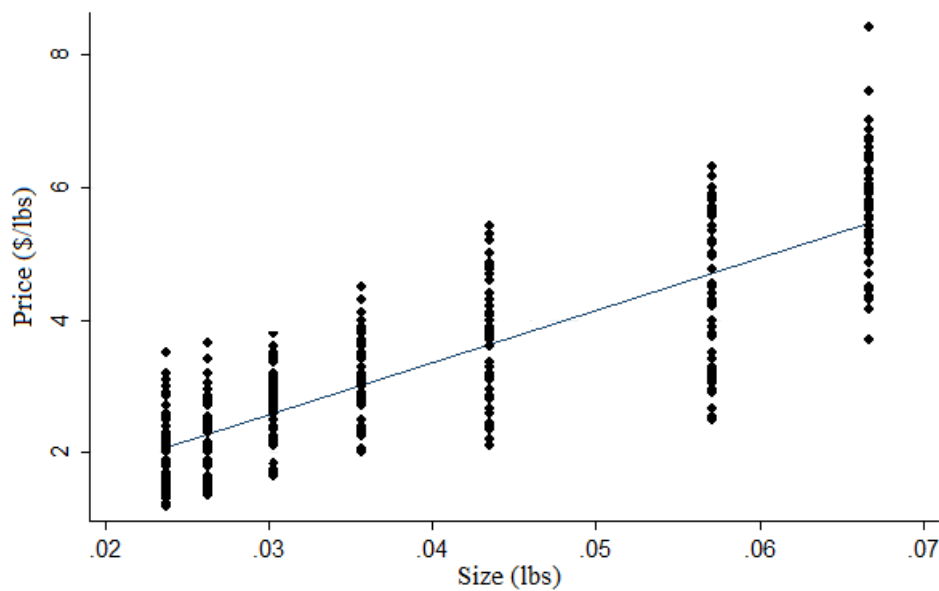
Shrimp size is measured as shrimp counts per pound. I use monthly ex-vessel price data in seven size categories collected by NOAA between 2006 and 2010 in Western Gulf (Texas port) because the greatest concentration of brown shrimp occurs in this area (Klima 1989). In practice, the estimated weight-price function is a step function; I smoothed the size-price function to make it continuous<sup>6</sup>. Using this monthly data set over this five-year period, I calculated the average size (lbs) for each shrimp size

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<sup>6</sup> For optimal control with a size-price step function, see Mistiaen and Strand (1999).

category, and used ordinary least squares to estimate the price per pound as a linear function of fish size (lbs). I suppressed the constant term in the regression<sup>7</sup>, and obtained a slope or marginal price premium of  $b = 83.4$  ( $t = 96$  and  $R^2 = 0.956$ ) see Fig.2.3 for the scatter plot.

Figure.2.3. Scatter Plot for Size Price Correlation



The social planner is assumed to maximize net present value by selecting the length of open and closed seasons corresponding to YPR, RPR and bioeconomic approach subject to an upper and lower bound on fishing effort. Table 2.1 summarizes the parameter values used. Many ecological and economic parameters were obtained directly from the literature, listed in Table 1. Huang and Smith (2011) set the initial number of shrimp as 100,000 for their numerical study in North Carolina. Here I set the initial number of shrimp as 10,000,000 for the numerical study. Since this is arbitrary, the

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<sup>7</sup> With a constant term in OLS regression, the estimated price-size function is  $p=0.21+78.8w$ .

resulting economic benefit does not reflect the true economic benefit for the Texas shrimp fishery.

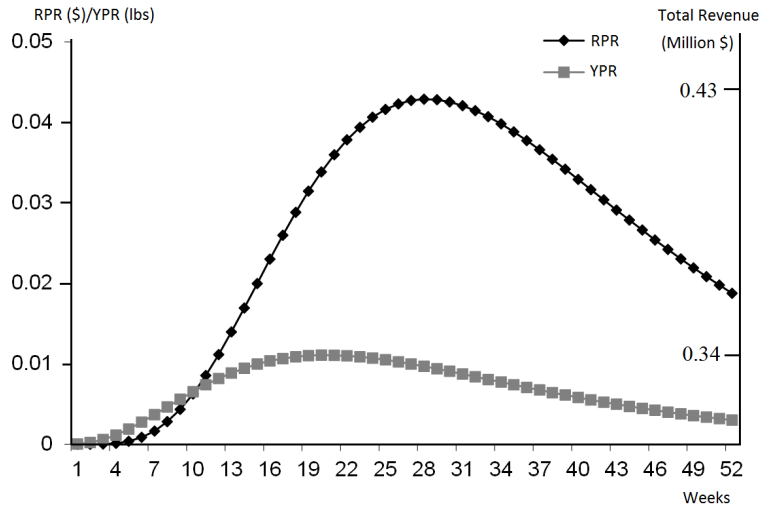
Table 2.1 Parameters Values Required for Simulation Model

Parameter	Value	Note	Parameter Sources
$\bar{w}$	0.08	Allometric function (pound)	Fontaine and Neal (1971)
$\beta$	2.94	Allometric function	Fontaine and Neal (1971)
$k$	0.0104	Von Bertalanffy growth function	McCoy (1968)
$t_0$	0	Von Bertalanffy growth function	Huang and Smith (2011)
$x_0$	10,000,000	Initial number of shrimp	
$\delta$	0.0035	Weekly discount rate	Huang and Smith (2011)
$c$	100	Unit fishing cost	
$q$	0.001	Catchability coefficient	Huang and Smith (2011)
$\bar{E}$	100	Fishing effort limit	Huang and Smith (2011)
$m$	0.063	Weekly instaneously natural mortality	Calibrated from Nance (1989)
$b$	83.4	Slope of the size price function	Estimated from NOAA data

Fig. 2.4 shows that YPR reaches a maximum at week 20, while RPR reaches a maximum at week 28. The initially convex phase of the YPR and RPR (without discounting) curves in this diagram results from the parameters  $\beta$  and  $k$  in the von Bertalanffy growth function. Fig. 2.4 suggests that if an impulse harvest were implemented at week 20, YPR management would generate approximately \$ 0.34 million in total revenue (assuming costless unbounded harvest) , while delaying the impulse to week 28, i.e., using RPR management, increases revenue by 26.5% (again assuming costless unbounded harvest).



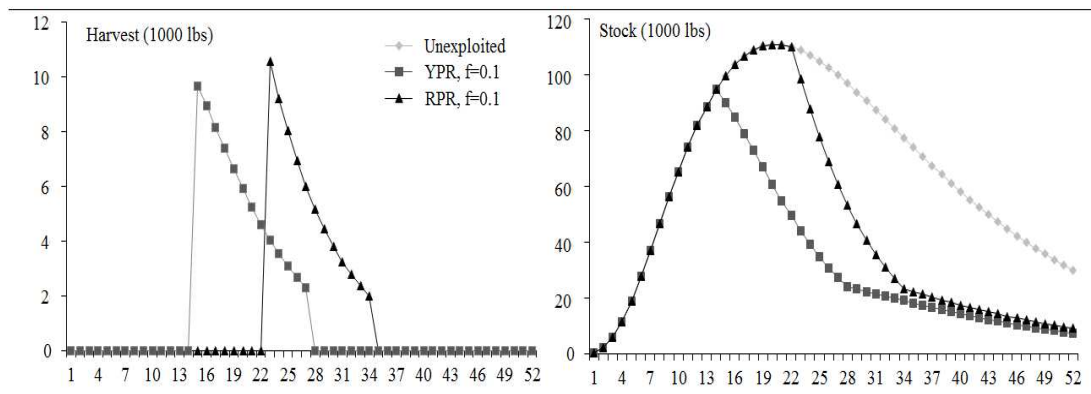
Figure.2.4. The YPR and RPR, and the Corresponding Maximum Revenue When a Costless Impulse Harvest is Applied at the Peak



Now consider the more realistic case where fishing effort is bounded above. Nance (1989) estimated monthly instantaneous rate of natural mortality to be 1. This has been converted into weekly instantaneous natural mortality for the purposes of this study. However the decreasing profit returns from the shrimping industry due to the competition from imported shrimp and shrimp aquaculture has caused some shrimpers to leave the industry. A license buyback program from 1996 to 2004 also reduced the number of licensees by 40% (Mamula 2009). If the catchability coefficient is equal to 0.001 and the weekly instantaneous fishing mortality  $f$  is equal to 0.1, then the upper bound on weekly fishing effort is approximately 100. Here I assume that the same upper bound exists on fishing effort ( $E_{\max} = 100$ ) for YPR, RPR and bioeconomic management due to the same catchability coefficient and fishing mortality  $f$ . Fig. 5 illustrates the harvest levels and stock levels implied by managing to maximize YRP and RPR using parameters specified in Table 1 and assuming a maximum possible fishing mortality rate of  $f = 0.1$  at 100

fishing trips. The figure shows YPR to be maximized when the fishing season opens at week 15, while RPR is maximized when the fishing season is delayed until week 22. This seven-week delay increases total revenue (still assuming costless harvest) from \$ 0.31 million for the YPR to \$ 0.36 million for the RPR. The corresponding stock levels for the YPR and the RPR both decline once the harvest season is open.

Figure.2.5. Harvest Level and Stock Level for the YPR and RPR When Maximum Fishing Mortality ( $f$ ) is Equal to 0.1



The optimal harvest path and stock level for different parameter values assuming a bioeconomic approach are shown in Fig. 2.6. The left column of the first row represents the optimal harvesting paths, while the right column describes the optimal stock level. Given the parameter values assumed, the optimal solution is divided into three phases. This is consistent with the analytical result that fishers will wait at the beginning, then apply maximal fishing effort ( $E_{\max} = 100$ ) until net profits fall to zero.

Figure.2.6. Optimal Harvest Path and Stock Level for Various Parameter Values under the Bioeconomic Approach

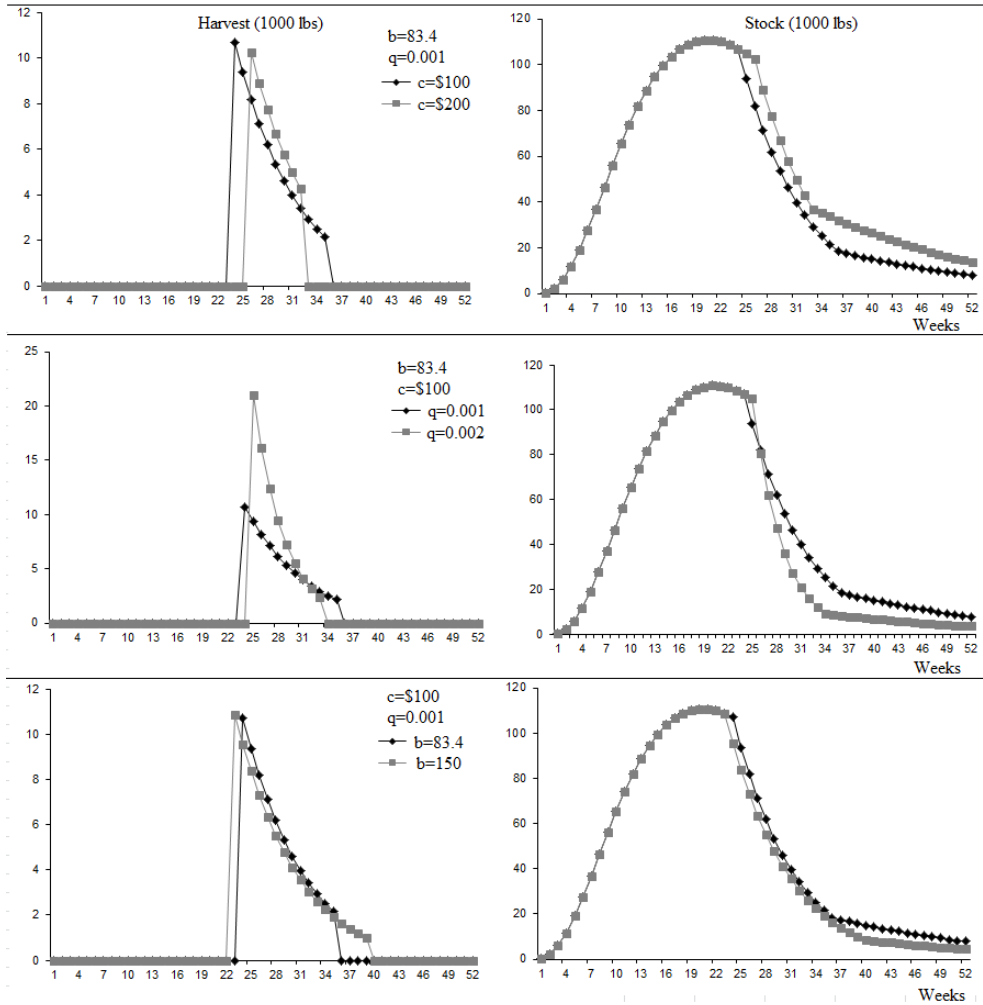


Fig. 2.6 shows the optimal paths with different costs per unit effort ( $c$ ), the catchability coefficient ( $q$ ), and the marginal price premium coefficient ( $b$ ). The black line represents the benchmark case with  $b = 83.4\$/lbs$ ,  $q = 0.001$ ,  $c = \$100$  /fishing trip. For the benchmark scenario, before week 24, the optimal management strategy is to close the stock to fishing because shadow price of additional unit of stock is greater than

the marginal gain from the harvest. The fishery is opened to harvest starting in week 24 with the maximal fishing effort until week 36. The fishery is then closed from week 36 forward, since in week 36 the marginal rent is equal to marginal harvesting cost, and the fishing effort switches from maximum to zero. After week 36, since marginal harvesting cost is greater than marginal revenue fishers have no incentive to continue fishing.

The first row of Fig. 2.6 depicts the scenario for two levels of cost,  $c$ , with the gray line representing the case for higher cost per unit effort. The left panel on this row shows that when unit effort cost is higher, the fishing season will be shorter and begin later. The intuition is that with a higher unit harvesting cost, the fishery is more profitable if the manager waits longer and lets the stock grow. If the fishing cost is great enough, the fishing season will only be open for a shorter period of time, and in extreme cases may not open at all. When cost per unit fishing effort decreases, the fishing season will be earlier and longer. Although the harvest level is higher for higher cost per unit effort, the stock level is also greater because of the later and shorter fishing season.

The second row of Fig. 2.6 represents the results for different values of the catchability coefficient, where the gray line again represents the greater catchability coefficient. When the price and cost per unit effort is the same, a greater catchability coefficient indicates a later harvest season in order to harvest larger size shrimp, which generate a price premium. With a greater catchability coefficient, stock is driven more quickly to a lower level. This is because greater catchability leads to a lower marginal cost of harvest  $c/qx(t)$ , and a lower terminal stock level where marginal rent is totally dissipated. The shifts associated with changes in catchability are not unidirectional, and cause the harvest and stock paths to cross, illustrating the interaction between biological

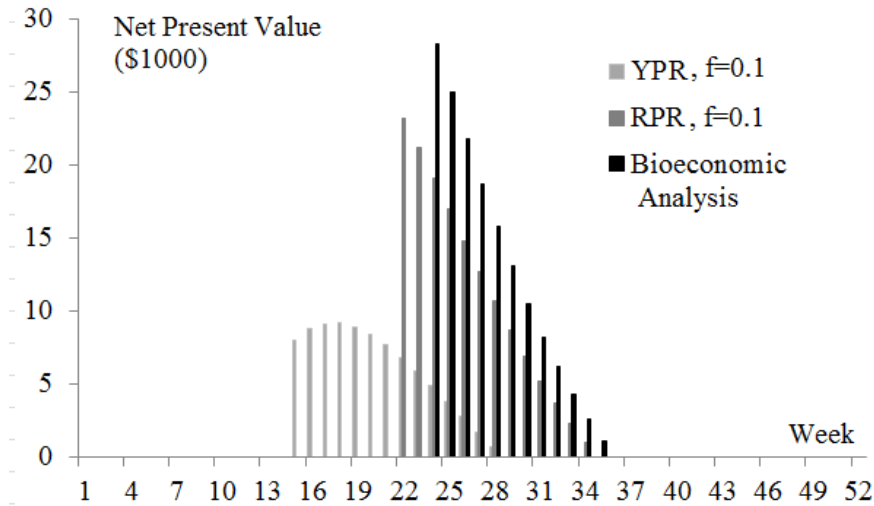
and economic parameters. Such a relationship would be further complicated if there was time varying catchability (Wilberg et al. 2010)

The last row of Fig. 2.6 shows results for different prices with the gray line representing higher price for the same fish size. An increase in  $b$  (note that  $b$  will not affect the “optimal” stopping time when effort is unbounded from above), the coefficient of the size-price function, will lead to an earlier and longer fishing season, an opposite effect to that of higher cost per unit effort. The intuition is that if smaller fish have a higher market price, it is profitable to harvest earlier. A combination of the same cost per unit effort and higher price ultimately leads to a higher harvest level, and decreasing stock levels, as shown in the right panel in the last row.

Fig. 2.7 demonstrates the impact of different fishery management strategies on net present value. It shows if the objective of the fishery management program is to maximize the YPR giving a capacity constraint of  $E_{\max} = 100$ , then the fishery will be open from week 15. A manager that sets the fishing season according to this strategy should find that no rational fishermen fish after week 29 when the harvesting cost is too high and offsets all the profit generated. Therefore, if fishing firms are profit-maximizing they will then continue to harvest from week 15 until week 29, after which harvest costs will exceed the value of harvest. Here the social welfare is calculated as weekly harvest yield multiplies its corresponding price, then less its weekly harvest cost before discounting to the present value. In the same manner, if the objective of the fishery management program is to maximize the RPR given the same capacity constraint, then the fishery will be open from week 22. And if fishing firms are profit-maximizing they

will then continue to harvest from week 22 until week 35, after which harvest costs will exceed the value of harvest.

Figure.2.7. Net Present Value and Fishing Season under BE, YPR, and RPR Approaches  
When Harvest Capacity is  $E=100$



### 2.5 Sensitivity Analysis of Natural Mortality

The sensitivity of YPR, RPR and BE approaches to fishery management to natural variation in fish dynamics is achieved by focusing on variation in natural mortality. Climate change has been recognized to modify marine ecosystems (Lubchenco et al. 1993; Harley et al.2006). For example, in the Gulf of Mexico it alters river flow, salinity, temperature level, and the size and severity of the hypoxic area (Ning et al. 2003; O'Connor and Whitall 2007). Huang and Smith (2011) investigate the impact of hypoxia on the optimal harvest path for shrimp in North Carolina, where hypoxia can create environmental stress and additional mortality. In this section, I conduct sensitivity analysis for baseline weekly instantaneous natural mortality (0.063), low natural

mortality (0.046), and high natural mortality (0.081). These values correspond to the estimates by Nance (1989).

Table 2.2 summarizes the effect of variation in natural mortality on the length of the fishing season, harvest, and social welfare under the YPR, RPR and BE approaches assuming the maximum fishing effort is constrained to  $E=100$ . With fishing capacity bounded above, the objective for fishery managers is to maximize the YPR or RPR by choosing date at which the fishing season opens. The date at which the fishery is closed need not be specified, since fishermen will stop fishing once it is no longer profitable to do so. I take the measure of welfare to be the discounted sum of net profits. The resulting fishing season and welfare level are also reported in Table 2.2.

For the same natural mortality, the YPR strategy has the earliest fishing season opening date, and the bioeconomic strategy has the latest. The YPR strategy also has the greatest biomass yield but generates the lowest social welfare, while the bioeconomic strategy has the least biomass yield, but generates the greatest social welfare. It does so by allowing the harvest of larger shrimp, which command a price premium. Two striking results deserve more attention. First, with a low natural mortality, the RPR strategy generates a welfare level that is approximately 97% of that generated by the bioeconomic strategy. Secondly, with high natural mortality, YPR strategy generates a welfare level that is only 11% of that generated by the bioeconomic strategy. These results indicate that in the face of ecological stressors that drive natural mortality rates to high levels, the gains from moving to a bioeconomic harvest strategy could be significant.

Table 2.2. Sensitivity Analysis of Varied Natural Mortalities for the YPR Strategy, the RPR Strategy and the Bioeconomic Strategy

Management scenario	Weekly instantaneous natural mortality	Harvest season (week)	Social welfare (1000 \$)
YPR strategy	low	18-36	256 (75)
RPR strategy	low	25-42	332 (97)
Bioeconomic strategy	low	27-43	341
YPR strategy	baseline	15-28	86 (55)
RPR strategy	baseline	23-34	146 (94)
Bioeconomic strategy	baseline	24-35	155
YPR strategy	high	13-24	11 (18)
RPR strategy	high	20-28	54 (87)
Bioeconomic strategy	high	22-30	62

Note: Social welfare for the YPR strategy and the RPR strategy is derived by maximizing the YPR and RPR, then deducting the corresponding harvest and discounting. Value in parentheses in the last column is the percentage of social welfare for YPR and RPR strategy compared to that from bioeconomic strategy at the same natural mortality level.

## 2.6 Discussion

The MSY concept was developed to address recruitment overfishing, and its analogue for growth overfishing is the maximum YPR. With unbounded fishing effort, the YPR for a short-lived single cohort occurs when the biomass yield loss from natural mortality is equal to biomass yield gain from per-capita weight gains. Since unlimited fishing effort is not feasible in practice, the maximized YPR typically generates a YPR-optimal fishing season which is sensitive to fishing effort constraint. Furthermore, the YPR approach neglects the opportunity cost of harvesting. Since fishers are more interested in the value of yield than in biomass, it is important to account for economic value of yield.



In this study I extend the traditional YPR approach by including size-dependent prices in an RPR analysis. Incorporating this price signal into YPR ensures fishers will harvest the fish stock at the time when the revenue loss from natural mortality equates biomass revenue gain. It is shown that the RPR analysis is similar to the management of a single rotation in forest economics when “harvest” effort is unbounded. The aim in both is to determine the optimal time to “harvest” the whole stock. The RPR is a step forward compared to the YPR. However neither the YPR nor the RPR approaches consider the stock-dependent costs of harvest, the potential cost of the harvesting due to discounting and the fact that fishing effort is always bounded above by fishing capacity. To address this I then consider a bioeconomic approach that recognizes the stock-dependent cost of harvest and considers the opportunity cost from harvesting due to discounting. My goal has been to compare and contrast the harvest strategy from the YPR, the RPR, and the bioeconomic approaches, in order to evaluate alternative options for addressing the growth overfishing of a short-lived species.

Including a size-price function changes the optimal season length in the YPR, the RPR and bioeconomic strategies. The fishing season is delayed, harvest is reduced, but profit is increased due to the inclusion of larger fish. For a shrimp fishery, the numerical results suggest that there are significant gains to be had by taking a capital-theoretic approach to the choice of when a fishery is opened. With the assumption that capacity is constrained, the fishery manager sets only the date at which the season opens under each strategy. This suggests that fishermen will fish maximally until they find no profit in doing so. I find that the value of the fishery under a RPR approach may be close to that from the bioeconomic approach if natural mortality is low. However, if natural mortality

is high, the value of the fishery can be significantly enhanced by adopting a bioeconomic, capital-theoretic strategy.

These results are subject to a number of caveats. I have assumed that the weight-price function is continuous. This is not really what we see in real world shrimp fisheries. A piecewise function would be more accurate, and would complicate the analysis and generate only a slightly different result. In addition, I have investigated a single cohort, while fisheries normally involve different species and different age classes. Therefore, I view this analysis as exploratory in nature. Nonetheless, the results suggest that incorporating the weight-price function and harvest cost in the decision process could improve fisheries management and benefit fishers.

## Chapter 3

# FORECASTING A WHALE PERMIT MARKET UNDER BOUNDED ECONOMIC UNCERTAINTY

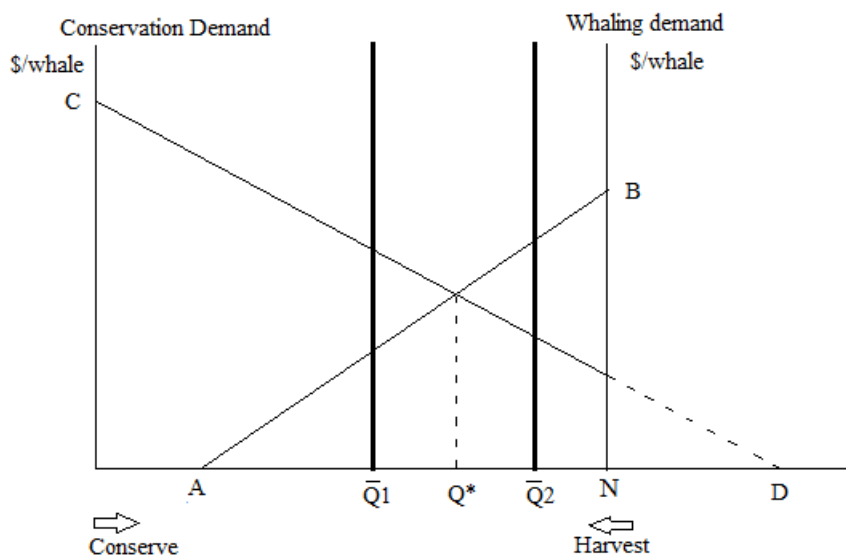
### 3.1 Introduction

The treatment of whales divides both individuals and societies, the arguments on all sides resting as much on cultural and ethical as on economic grounds. Indeed, different interest groups—notably whalers and conservationists—both appeal to moral or cultural imperatives to argue for the “essential” nature of either whale conservation or harvest. The result has been a global paralysis in any attempt to agree upon the allocation of whales between harvest and conservation. Recently, Costello et al. (2012) repeated previous calls (Kuronuma and Tisdell, 1994; Bulte et al., 1998) for a market-based approach to determine the number of whales harvested globally through a transferable whale permit program. This paper aims to evaluate the long run ecological consequence of such a market-based approach. It provides insight into the economic conditions in which the establishment of a “cap and trade” program will result in harvesting all permits (full exploitation), conserving all permits (pure conservation) or a coexistence between whaling and conservation.

Within a whale permit market, the regulator (e.g., the International Whaling Commission, IWC) first sets an allowable annual biological catch limit, at or below which a sustainable stock level will be assured. The regulator then allocates the annual catch limit in the form of whale permits (shares) to different whaling and conservation groups. The holder of the permit has the right, but not the obligation, to harvest a whale. In addition to allowing whaling interest groups to transfer the right to harvest among themselves, such a program enables conservationists to buy and retire harvesting rights.

The number of whales actually harvested relative to the allowable catch may then be determined by peoples' willingness to pay for whaling or conservation. By allowing each party to reveal the strength of its preferences in the form of willingness to pay (assuming that they are able to pay) and to reconcile the differences in these preferences through exchange (Figure 3.1), a market institution thus may provide a means to resolve the impasse created by diverse moral and cultural perspectives.

Figure 3.1. A Whale Permit Market System, Assuming Linear Demand for Whaling and Whale Conservation



Note:  $\bar{Q}_1$  represents an absolute constraint on the supply of whales to whalers. A relatively loose harvest limit ( $\bar{Q}_1$ ) suggests an “interior” solution, with whalers harvesting  $Q^*$ , and  $(\bar{Q}_1 - Q^*)$  is conserved, while a relatively conservative harvest limit ( $\bar{Q}_2$ ) would indicate that the maximal harvest level ( $\bar{Q}_2$ ) is exercised. Conservation demand is read from left to right and whaling demand is read from right to left. Here  $N$  denotes initial population level and  $D$  is carrying capacity level. See Gerber et al., 2014.

It is worth noting that similar programs have been applied to the US SO<sub>2</sub> and EU CO<sub>2</sub> pollution markets (Stavins, 1998; Ellerman and Bucher, 2007; Carlson et al., 2000) and to the management of fisheries within exclusive economic zones worldwide to create successful common governance structures and enhance fish stocks (Christy, 1973; Costello et al., 2008; Grafton et al., 2006). Relative to other right-based market approaches, conservation interests would have access to and likely play an important role in a whale permit market in a similar manner to, for example, how the Nature Conservancy and Environmental Defense Fund acquired trawl permits in the central coast of California in 2003 to protect essential fish habitat (Gleason et al., 2013).

Prior research on whale management has focused on biological uncertainty (Punt and Donovan, 2007; Taylor et al., 2000), while economic uncertainties, a crucial aspect of a proposed whale market, have been largely overlooked. Gerber et al. (2014) investigate the potential for such a market for a set of biological and economic conditions. They explored the functionality of the proposed whale permit market by examining minke, bowhead, and gray whales, finding that a market could ensure the persistence of whale populations while improving whaler and conservationist welfare. However, free-riding, incentives for “illegal” harvest and trade, monitoring and enforcement challenges, and increasing costs associated with creating and managing a whale permit market could reduce social welfare to suboptimal levels and, some say, potentially pose a greater threat to whales than the current moratorium (Smith et al., 2014).

Costello et al. (2012) implicitly assumed that all concerned parties would agree an authority (the IWC) for the global management of whaling and whale conservation.

However, concerns are raised that the IWC, as an international organization lacking the power to compel compliance from sovereign nations, would be credible in this role. Since the establishment of any “cap and trade” system requires an authority with the power to set quota, to allocate rights to quota shares, and, most importantly, to penalize non-compliance, it requires a body with more authority than the IWC has. Since a number of countries<sup>8</sup> currently ignore the existing moratorium on whaling there is no good reason to believe that conservation organizations would trust the IWC to police a cap and trade system. Indeed, for a global management of whale conservation managed by a regulator without a legal basis, creating appropriate institutions to efficiently deal with allocating, penalizing and policing issues may prove costly, politically and economically.

Considered this situation, it might be worthwhile to investigate the feasibility of such a “cap and trade” program in a nation where whaling is a traditional and relatively well-enforced practice, while at the same time maintaining the moratorium elsewhere.

In this study, I examine the ecological and economic outcomes of a whale conservation market for a range of assumptions demand for different whale uses. In particular, I explore uncertainties associated with the whaling demand (WD) and conservation demand (CD). I consider four types of uncertainty. The first is the uncertainty about the functional form of these demand curves – whether they are linear or highly non-linear. The second uncertainty is associated with the marginal willingness to pay for one more whale, either hunted (whalers) or saved (conservationists) (WTP, loosely, is the height of a particular demand curve measured at a particular quantity), and

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<sup>8</sup> Approximately 2000 whales are harvested by Japan for “scientific purpose”, 600 are harvested for commercial whaling by Norway and Iceland and 350 for subsistence by Denmark, Russia and the United States (Costello, et al, 2012).

with the response in marginal WTP to changes in whale stocks (i.e., slope, the responsiveness of marginal WTP to additional conservation or harvest). The third uncertainty, the so-called “choke price” for whaling demand, plays an important role in market dynamics. The choke price is the maximum WTP of whalers for an additional whale as the number of hunted whales approaches zero<sup>9</sup>. In addition, the degree of free-riding among conservation interests is an important feature for understanding the strength of conservation demand. Free-riding is the central problem in the provision of public goods: it refers to a situation where individuals benefit from a resource without paying their share of cost for its provision, thus “free-riding” on others’ contributions (Samuelson, 1954).<sup>10</sup> Uncertainty as to the extent of free-riding directly influences the degree to which demand for whale conservation manifests in a market, which in turn creates uncertainty as to the outcome of a whale market. Currently, each of the aforementioned uncertainties represents a serious impediment to advancing a real world whale market experiment.

Altogether, these uncertainties raise at least two important questions. First, under what circumstances can we expect market outcomes in which either conservation or whaling interests dominate? Second, which reductions in uncertainty would provide the greatest insight into the potential ecological outcome of a whale permit market? To shed light on these questions, I conducted extensive sensitivity analysis on a model applied to

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<sup>9</sup> There is also a choke price for the CD curve—the maximum WTP of the conservationists when the whale population approaches extinction—but this becomes irrelevant when the enforcement of conservative catch limits is in place.

<sup>10</sup> Free-riding is indistinguishable in its effects from a lack of participation by some conservationists due to moral repugnance. I therefore treat them symmetrically here.

a regional whale permit market that considers whale population dynamics, harvest policies, and whaling and conservation demands.

### 3.2 A Possible Whale Permit Market for Minke Whales

Although whaling has been enacted in several countries around the world, in this study I consider a whale transferable permit market for the management of Northeast Atlantic minke whales (*Balaenoptera bonaerensis*) in Norway. I focus on minke whales in Norway as a case study for several reasons. First, Norwegian whaling takes place in waters under Norwegian jurisdiction only. Minke stocks in this area are approaching carrying capacity (Gerber et al., 2014; North Atlantic Marine Mammal Commission, NAMMC, 1998). This minke whale stock has a wide range of migratory route in the Northeast Atlantic, but the harvests are taken almost exclusively by Norwegian whalers (Amundsen et al., 1995). The Norwegian regulator would set annual quota based on the procedures developed by the scientific committee of the IWC. In the past decade, approximately 550 minke whales have been taken annually by Norway for commercial use. Currently, all whaling vessels in Norway have monitoring systems and are assigned inspectors during the hunting season to ensure compliance with whaling regulations ([www.fisheries.no](http://www.fisheries.no)). There are a number of legal requirements on whalers and traders. Whalers are required to take an annual course to ensure the least pain inflicted on the hunted whale. The authority also uses a DNA registering system to monitor the whale meat market and to detect illegal, unreported whaling. The illegal hunting of whales is punished with a fine up to \$135,000 and the violator can be imprisoned for one year. Finally, the successful establishment of the Norwegian financial instrument market ( i.e,



future contract market) for salmon through an electronic trading system (Ewald, 2012) suggests that the institutional capacity for a whale share market exists within Norway and may provide insightful information for the establishment of such a market.

Similar with the property right-based fishery management where property rights are granted to fishers and can improve fisheries management (Arnason, 2012), this proposed whale conservation market assumes that the annual whale permits, which is based on stock abundance, are all allocated to whalers. Permits can be delineated equivalent to one whale. If the whale policy permits become a registered product on an electronic whale conservation market, then in theory any individual or conservation group with the necessary funds can purchase a permit. Once a conservation group has purchased a permit it is held in their electronic registry to retire to save a whale in that season. Importantly, as a global impure public good (Kuromuna and Tisdell, 1993), permit are tradable to conservationists anywhere regardless of geographic location through the electronic system. Therefore, the whale permits in Norway can be traded globally, though the resource management being addressed is more of a regional scale.

### 3.3 Material and Methods

I developed a model for the Norwegian whale permit market system, which includes regulatory, economic and biological components. It consists of the whale population, the whaling quota, and the whale permit market. In my model, an annual whaling quota ( $H_t^{\max}$ ) is set based on whale stock abundance, and the protection level ( $N_t - H_t^{\max}$ ) should reflect sustainable stock conservation levels. This annual whaling quota will be distributed to different whaling companies – thereby respecting the de facto

rights of whalers to whale under the weak governance of the IWC moratorium. Note that while the system for the initial distribution of quota may have important distributional implications, the equilibrium allocation of quota in a “cap and trade” system should be independent of its initial permit allocation in a well-functioning market, a finding that has been substantiated in real-world environmental markets (e.g., Fowlie and Perloff, 2013). Conservation groups therefore secure conservation by buying in to the whale permit market, and retiring seasonal whale permits. Whale permit market participants can modify their initial endowment according to their preferences by freely trading permits. Both regulated whalers and conservationists can choose between harvesting, selling all or some of their permits, or purchasing permits from the market.

My model assumes that all participants only engage in trades when it is in their own self-interest. This may be profit-maximization for whalers, or the optimization of a more complex objective for conservationists. The number of permits to hold is determined by the preferences and income endowments of whaling or conservation interests. The intensity of these monetarily-constrained preferences is revealed through demand curves. I assume that market clearance of whale permit happens each year, and banking of unused permits for future use is not allowed. The inclusion of permit banking can smooth price uncertainty across seasons (Fell et al., 2012); however, it may be undesirable in a species conservation context since it does not guarantee a firm upper bound on harvest in any season. Market trade would determine how many whales are actually being harvested each year, which impacts whale populations in subsequent years, and therefore the permits issued the following year.

This study focuses on the Central North Atlantic minke whales (*Balaenoptera bonaerensis*), whose populations are approaching carrying capacity (Gerber et al., 2014; North Atlantic Marine Mammal Commission, NAMMC, 1998). This population level suggests that the initial quotas for the first year would be at their maximum. The trajectory to the steady state population and harvest level – assuming predictable whale population dynamics – will be a smooth and declining curve.

### 3.3.1 Whale Population Dynamics

My assumptions about whale population dynamics mimic those of Gerber et al. (2014), who conducted prospective analyses for potential whale permit markets for Bering-Chuchki-Beaufort bowhead whales, Central North Atlantic minke whales and Eastern North Pacific gray whales. Population dynamics are assumed to take the form

$$N_{t+1} = N_t + r N_t (1 - N_t/K) - Q_t$$

Where  $N_t$  is the population stock level at time  $t$ ,  $r$  is the intrinsic growth rate,  $K$  is the carrying capacity,  $Q_t$  is the quantity of whales actually harvested. I use the biological characteristics for North Atlantic minke whales from Gerber et al.(2014), where initial population level and carrying capacity are  $N_0 = K = 72,130$  and the intrinsic growth rate is  $r = 0.04$ .

### 3.3.2 Whaling Quota Setting

The maximum allowable harvest is assumed to be based on the whale stock, and updates each year. In the United States the maximum allowable harvest is referred to as the potential biological removal (PBR) and is modeled as  $H_t^{\max} = 0.5rN_tF_r$ , where  $F_r$  is a recovery factor that adjusts upward – increasing the fraction of the stock of whales

$0.5rF_r$  that can be harvested – as the stock of whales increases toward carrying capacity:  
 $F_r = 0.1 + 0.4N_t / K$  (Gerber et al., 2014). The resulting quadratic harvest control rule,  
 $H_t^{\max} = 0.05rN_t + 0.2rN_t^2 / K$  reflects a conservative maximal harvest (ensuring a steady state of at least 79% of carrying capacity given model parameters) while maintaining recovery from a low abundance as quickly as possible (Taylor et al., 2013, Gerber et al., 2014).

### 3.3.3 Estimation of the Demand Curves

#### 3.3.3.1 Data Description

I used global population data for minke whales ([iwc.int/home](http://iwc.int/home)). The minke whales are hunted in Antarctica by Japan, in North Atlantic by Denmark, Iceland, Norway, St Vincent and the Grenadines, in North Pacific by Japan and Korea. I use global minke harvest data. The reason is that there is trade of whale meat products among Japan, Norway, Iceland and the Faroe Islands regardless of the ban on international trade in minke whale products. Norway reserved the right due to the “least concern” on the red list of The International Union for Conservation of Nature (IUCN). Norwegian fishery statistics ([www.fisheries.no](http://www.fisheries.no)) provide data on Norwegian minke whale prices.

Conservation information, such as the willingness to pay to increase whale population from one to another level, is extracted and calibrated from the literature (Loomis and Larson, 1994). Other parameters, such as the choke price and slope for whaling demand, slope for conservation demand and the degree of free-riding (the participation level of conservationists) are generated using Monte Carlo simulations drawn from plausible bounded intervals.

### 3.3.3.2 Whaling Demand Calibration

Due to differences in the type and availability of data, the WD and CD curves are calibrated differently. I rely upon an “input demand” function for whaling. An “input demand” function is built on the observation that the willingness to pay for the harvest of one more whale derives from that whales’ role as an essential input to the output of ex-vessel whale products. This willingness to pay is equal to the ex-vessel value of a whale to whalers net of the variable costs of harvesting, processing and delivering the whale products to market<sup>11</sup>. The willingness to pay for an additional whale is its "value marginal product," which is the marginal variable profit, therefore allowing harvest costs to be accommodated implicitly. Conventionally, a firm’s demand for any input is derived by choosing the input level that maximizes profit, given technology, and is assumed to be known. Due to the lack of input cost and production data, whaling demand was derived using approximation theory.

The WD curve is approximated by linear (Horan and Shortle, 1999) and quadratic functional forms, equivalent to first order and second order Taylor Series expansions around an average quantity-price pair  $(\bar{Q}, \bar{P})$  for the actual WD function.

The key parameters characterizing the linear WD curve are its slope and choke price. The linear whaling demand curve takes the form  $P_t = A - B * Q_t$  where  $P_t$  is the permit price and  $Q_t$  is the number of whales harvested at time  $t$ ,  $A$  is the whaling choke price. The choke price is reached when the quantity demanded of a good falls to zero. A finite choke price

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<sup>11</sup> If a whaling operation is vertically integrated with the operations that supply final consumer products, then the demand should reflect the final market value net of costs along the supply chain. If, on the other hand, whalers sell minimally processed whale products to downstream processors, then this value-added (and its associated costs) should not be reflected in the whale input demand curve.

is highly likely for whales as an economic input since there is little reason to think that the final market for whale products (i.e. the market for whale meat) should see skyrocketing prices as quantity supplied to market approaches zero, due to substitutability with other non-whale products.  $B$  is the slope of whaling demand, and it represents the change in price as the quantity harvested increases. The problem is to solve for parameter values of  $A$  and  $B$ . Here I sample with replacement from the whale price and quantity data to generate an average quantity-price pair  $(\bar{Q}, \bar{P})$ . For a linear whaling demand function passing through the average price and quantity pair  $(\bar{Q}, \bar{P})$ , once the slope  $B$  is simulated from a bounded distribution, the choke price  $A$  can be derived as  $A = \bar{P} + B * \bar{Q}$ .

The quadratic whaling demand function takes the form  $P_t = a * Q_t^2 + b * Q_t + c$ . This requires calibrating the three parameters  $(a, b, c)$ . This quadratic whaling demand also passes through the average price and quantity pair  $(\bar{Q}, \bar{P})$ , which is also generated by sampling with replacement from the whale price and quality data. The slope evaluated at the average price and quantity pair and the choke price  $c$  of this quadratic whaling function are then simulated from bounded distributions. These three data points are sufficient to uniquely determine the three parameters  $(a, b, c)$  of the quadratic whaling demand function.

To parameterize these two functional forms, I use data on the average price, quantity, and the whaling slope. Total value and quantity data over the period 1997-2007 were obtained from the IWC website and Norwegian fishery statistics. The marginal variable profit (“price” for whaling demand) was calculated by deducting the estimated participation cost adjusted from (2) from the value data, and then converting this into a

value per whale. Using 11 price- quantity data points I sampled with replacement to create the average price and quantity pair  $(\bar{Q}, \bar{P})$ .

There is little information provided in the literature regarding the slope and the choke price of whaling demand. Horan and Shortle (1994) used data from 1979-1983 along with the assumption that inverse demand elasticity<sup>12</sup> was equal to -1 to derive a linear whaling demand,  $p = 10991.04 - 0.73075Q$ . Here I assume that the slope of whaling demand takes a uniform distribution (-5, -0.5). I also assign a uniform distribution (\$5,000, \$50,000) for the choke price of whaling demand due to the lack of information on this value. The uniform distribution places the same weight on each value. These two uniform distributions are presented in Table 1. The slope and choke price of whaling demand in any given simulation are generated by sampling from these distributions.

### 3.3.3.3 Conservation Demand Calibration

Whale conservation is a public good, whose value is not measured by one but many individuals (Kuronuma and Tisdell, 1994). For instance, the value lost (e.g, its existence value or value related to non-direct use, such as whale-watching) associated with harvesting a whale would be the aggregated value of all “consumers”. The public good nature of whale conservation also implies that the benefits of conserving a given whale cannot be excluded to any one person or group of persons and one individual’s enjoyment of the benefits of conservation are not affected by anyone else’s enjoyment. This implies that aggregate conservation demand is derived by summing individuals’

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<sup>12</sup> Elasticity ( $Ed$ ) measures the proportional responsiveness of a quantity demanded of a good relative to a proportional price change. It is the slope when measured on a log scale,  $Ed=(\% \text{ change in quantity})/(\% \text{ change in price})$ .

willingness to pay for each successive conserved whale along the demand curve – “vertically summing” individual conservation demands. As a non-consumptive good, stated preference surveys, such as contingent valuation, are generally required to recover non-consumptive demands (Freeman, 1993). For the CD curve, I consider both linear and log-linear functional forms. The log-linear form is widely used in non-market evaluation studies and has the convenient property of constant elasticity, a parameter I use for derivation of linear and quadratic whaling demand and linear conservation demand. It is the slope of the demand curve when graphed on a logarithmic scale. To be different from the notation used for calibration of whaling demand, I use  $X$  to denote the quantity demanded for whale conservation. The nature of whale conservation management will not allow extinction to happen; therefore the properties of the demand curve in the vicinity of extinction are irrelevant to this analysis.

Unlike the case of whaling, conservation data associated with any particular population level and its corresponding marginal WTP are not available. Typically, a contingent valuation survey will elicit people’s WTP for a significant discrete change in species population, for instance, a 50% or 100% increase. This WTP is therefore the total WTP information, instead of marginal WTP for an incremental population change. Four statistics are used to characterize the log-linear functional form ( $\ln P_i = c + d \ln X_i$ ): the total WTP for a discrete population change, the slope of CD, the potential number of conservationists in the market, and the participation level of conservationists – where the last two data points speak to the extent of the “vertical summation” of conservation demands that actually materialize in the permit marketplace, accounting for the extent of free-riding.



I first consider a log-linear form,  $\ln P_t = c + 1/Ed * \ln X_t$  or  $P_t = \mu X_t^{1/Ed} \rightarrow$ , where  $X_t$  denotes the number of live whales (total whale population minus harvest),  $P_t$  is marginal willingness to pay, and in a market context it is marginal permit price. The elasticity is equal to,  $Ed$  defined as  $Ed = \frac{dX}{dP} * \frac{P}{X}$ . If conservation demand elasticity  $e$  is known, then  $\mu$  is all that is left to know. Loomis and Larson (1994) serve as a starting point for calibration of conservation demand. In their CV survey, the representative California household was found to be willing to pay \$25 per year, in 2012 US dollars, for a 50% increase in the gray whale population when the initial population level was 20,000. The integral of this conservation demand function corresponding to an increase from 20,000 to 30,000 is represented mathematically by,

$$\int_{20,000}^{30,000} \mu X^{1/Ed} dX = (WTPGWC) * (TNCH) \quad (3.1)$$

Where WTPGWC refers to mean willingness to pay for gray whale conservation, and TNCH is the total number of California households. The left-hand-side (LHS) of Eq (3.1) is the integral of the aggregated inverse conservation demand for gray whale population increases from 20,000 to 30,000. The right-hand-side (RHS) of Eq1 is the total WTP which equals to the product of average household's annual WTP for gray whale conservation and the number of California households. The strategy is to use values reported in the literature for the RHS of Eq 1 to calibrate  $\mu$  on the LHS of Eq1, while elasticity  $Ed$  is imputed from a reasonable bounded interval using Monte Carlo simulation technique.

Richardson and Loomis (2009) demonstrate that the WTP for species protection varies across species, it is therefore important to adapt these calibration results to minke whales. To accomplish this, I draw upon techniques from the benefit transfer literature (Boyle and Bergstrom, 1992). Benefit transfer is a calibration process by which a benefit value is transferred from data for a particular study site or species to another study site or species. The IWC whale population estimates indicate that minke whales are relatively more abundant than gray whales. Also an adult minke whale is much smaller than an adult gray whale. Altogether, this suggests that minke whales may be a less charismatic species, indicating a smaller total willingness to pay for the same magnitude of minke whale conservation as for humpback whales. Therefore, I consider a range of “transfer ratios” ( $1/n$ ,  $n=1,2,3,4,5$ ) between minke whales and gray whales, and will be applied on the right hand side of Eq (3.1) . The transfer ratio approach says that if a household is willing to pay  $\$n$  for gray whale conservation each year, then it is willing to pay  $\$1$  for minke whale protection each year. The number of California households is replaced with the number of conservationists participating in the whale share market. The number of conservation participants involves a free-riding problem related to the public good nature of whale conservation. Free-riding may be present in two ways: either market participation decreases, or market participation remains unchanged but the total WTP value decreases. Since WTP and the number of whale market participants are multiplicative in Eq (3.1), they are effectively indistinguishable, so I consider free-riding solely through market participation. The number of conservation market participants is equal to the product of the proportion of conservationists who actually participate in the

market, to which I refer as “the participation level of conservationists”, and the total number of whale conservationists (inclusive of free-riders). Eq (3.1) becomes,

$$\int_{20,000}^{30,000} \mu X^{1/Ed} dX = (WTPGWC) * (MGTR) * (TNWC) * (PLC) \quad (3.2)$$

Where MGTR refers to minke/gray whale transfer ratio, TNWC is the total number of whale conservationists, and PLC is the participation level of conservation. PLC is a ratio and equal to (1-the degree of free-riding). The LHS of Eq (3.2) becomes the integral of inverse conservation demand as the minke whale population increases from 20,000 to 30,000. The RHS of Eq (3.2) is the total WTP which equals to the product of WTP for gray whale conservation, minke/gray whale conversion ratio and the number of conservation market participants. With values on WTP for gray whale conservation, minke/gray whale conversion ratio and the number of conservation market participants I solve for  $\mu$  in Eq (3.2).

I also consider a linear functional form  $P_t = v - \tau * X_t$ , where  $v$  denotes the choke price related to whale conservation. It is the maximum WTP when  $X_t$  equals zero.  $\tau$  describes the decline in permit price as the quantity conserved increases. The choke price is not relevant for conservation demand because of the enforcement of conservative catch limits. Mathematically, the total WTP is

$$\int_{20,000}^{30,000} (v - \tau X) dX = (WTPGWC) * (MGTR) * (TNWC) * (PLC) \quad (3.3)$$

The strategy is to use values reported in the literature for the RHS of Eq (3.3) to calibrate  $v$  and  $\tau$  on the LHS of Eq (3.3).

Even if the information regarding WTP for gray whale conservation – the minke/gray whale conversion ratio and the number of conservation market participants – is known, the above expression only provides us with one calibration equation for two free parameters. Therefore, I need another equation in order to derive the linear conservation demand function.

The second equation comes from the definition of the elasticity of demand for conservation. This linear conservation demand  $P_t = \nu - \tau * X_t$  and the elasticity definition

suggest  $\frac{dX}{dP} = Ed * \frac{\bar{X}}{P} = -\frac{1}{\tau}$ , where  $\bar{X}$  is the average quantity between 20,000 and

30,000, which is 25,000, and  $P$  is the marginal willingness to pay (price), which is equal

to  $\nu - \tau \bar{X}$ . Substituting this information into the definition of elasticity gives the second

equation  $Ed * \frac{\bar{X}}{\nu - \tau \bar{X}} = -\frac{1}{\tau}$ , where  $\bar{X} = 25,000$  and  $Ed$  is sampled from plausible

bounds using Monte Carlo simulation. Little information can be gleaned from the literature regarding the elasticity of conservation demand. Here I choose a wide range of elasticities simulated from a uniform distribution (-5,-0.2). The endpoints, which span from highly elastic at the low end to relatively inelastic at the high end should be sufficient to bound the elasticity of conservation demand.

Since Loomis and Larson (1994) did not include WTP information from conservation groups, the average annual WTP per household after interest rate adjustment, which is \$25, does not include elicitation from conservation groups, whose annual expenditure on anti-whaling campaigns could be millions of dollars. For instance, a conservative estimate on annual anti-whaling expenditures by Greenpeace USA,

Greenpeace International, Sea Shepherd, WWF International and WWF UK is \$25 million<sup>13</sup> (Costello et al., 2012). The households in coastal California arguably have a high affinity for whales (hence more potential non-use value), while members from non-profit organizations with high expenditures for whale protection would likely pay more to conserve gray whales. I therefore scale up the annual WTP, and draw the value from the uniform distribution (\$25, \$75). The endpoints here do not reflect the annual expenditure from NGOs. However, once minke/gray whale transfer ratio is considered, these endpoints should be sufficient enough to bound the WTP information for minke whale conservation.

Since whale permits would be traded globally, instead of using California household information for conservationists participating in the market, I could use expenditure information from non-profit organizations (NGOs) dedicated to whale conservation. However, this information is impossible to retrieve from NGOs; nor is this information necessarily a reliable proxy for the expenditures that may be forthcoming in a conservation market context. Here, I assume the WTP information described previously, and use membership numbers from non-profit organizations, such as Environmental Defense Fund, Greenpeace, and Sierra Club as a proxy for the possible number of conservation market participants. While some non-profit organizations provide membership numbers on their websites, others do not. I use a conservative estimate of the membership of relevant non-profit organizations, 20 million, as the maximum number of conservation market participation. Due to the lack of information regarding the degree of free-riding, I also draw the value of this parameter from a uniform distribution (0,1). The

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<sup>13</sup> This anti-whaling expenditure is used for protection for all whale species. It is impossible to know the exact expenditure on minke whale protection.

Minke/gray whale transfer ratio is drawn from uniform integer distribution (1,5). All these distributions are provided in table 3.1.

Table 3.1. Parameter Distribution

Parameter	Distribution
Slope of whaling demand	Uniform (-5, -0.5)
Choke price of whaling demand	Uniform (\$5,000, \$50,000)
Elasticity of conservation demand	Uniform (-5, -0.2)
The participation level of conservationists	Uniform (0, 1)
Individual's annual WTP for gray whale conservation	Uniform (\$25, \$75)
Minke/gray whale transfer ratio	Integer Uniform (1, 5)

Note: Cases involving linear whaling demand only require simulation for the slope of whaling demand, and cases involving quadratic whaling demand require simulations for both slope and choke price of whaling demand.

### 3.4 Simulation

I considered four potential whaling and conservation demand combinations: (1) linear whaling demand and linear conservation demand, (2) linear whaling demand and log-linear conservation demand, (3) quadratic whaling demand and linear conservation demand, and (4) quadratic whaling demand and log-linear conservation demand. To examine how different parameters impact the steady state harvest under these four demand combinations, Matlab was used to randomly generate 1000 parameter sets from the distributions in Table 1, which are then used to produce 1000 pairs of whaling demand and conservation demand curves. The model is then simulated to the steady state harvest and population levels for each replicate.

For each harvesting season, the equilibrium harvest level is calculated based on the pair of whaling and conservation demand curves under consideration. Then, the equilibrium harvest is incorporated into the harvest policy. If the WTP to conserve the

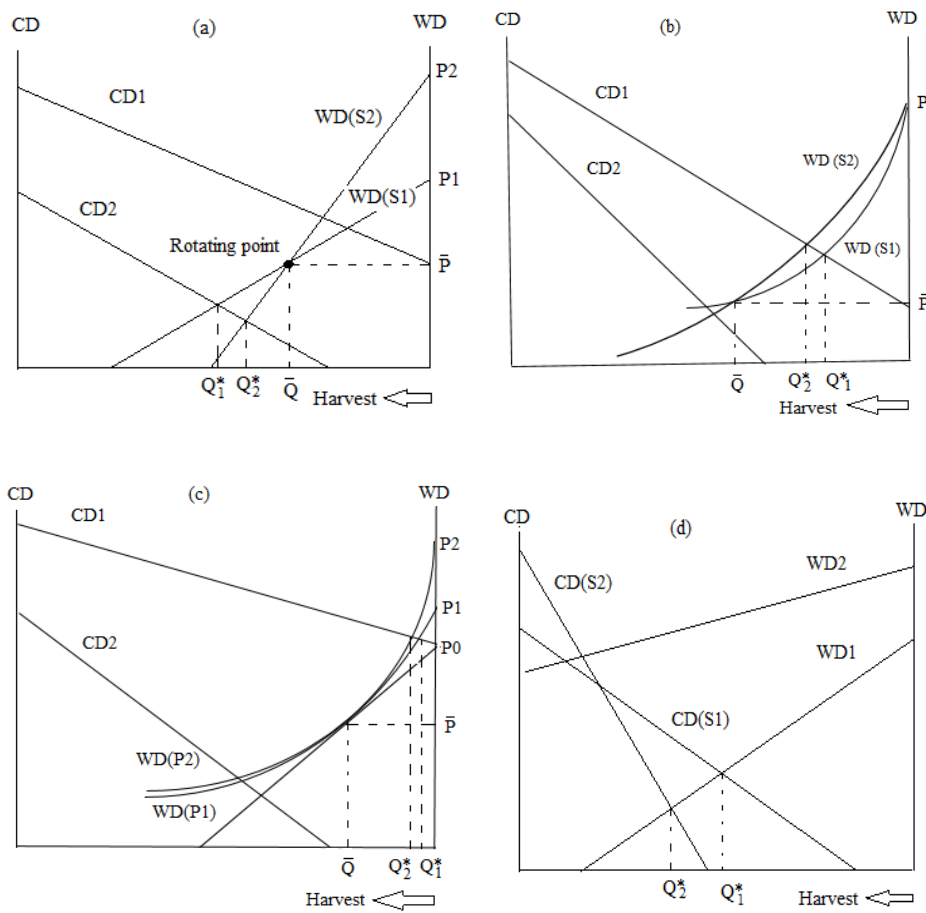
last whale is greater than the profit that the whalers will get from selling their first whale in the whale meat market, then the conservationists buy all whale permits leading to a pure conservation policy. If the equilibrium target harvest level  $Q_t^*$  exceeds the maximum allowable harvest  $(H_t^{\max})$ , then whalers will harvest the maximum allowable harvest since they value the last harvested whale more than the first whale “saved” by conservationists. If  $Q_t^*$  is less than  $(H_t^{\max})$ , then whalers would harvest  $Q_t^*$  (see Figure 1). This trading process is simulated until the model converges to the steady state harvest and population level. This was done for the 1000 pairs of whaling demand and conservation demand curves.

### 3.5 Results

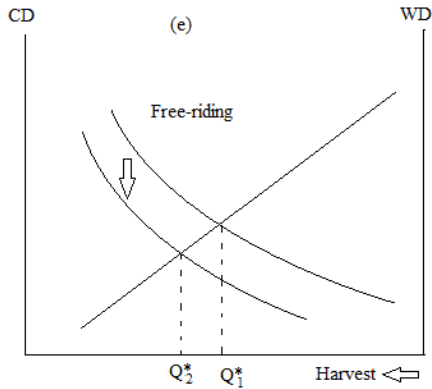
Fig 3.2 shows how the harvest level resulting from the whale permit market in a given season is affected by the uncertainties associated with the WD and CD curves – assuming “interior” equilibria where supply and demand are equal. Recall that in this study, I constrain these uncertainties by the (limited) available data: the average quantity-price pair  $(\bar{Q}, \bar{P})$  for WD, and the total WTP for a 50% increase in whale population for CD. Figure 2a depicts the effect of slope and choke price of linear whaling demands on harvest level. All else constant, increasing the whaling demand slope (its absolute value) rotates the WD curve counter-clockwise around the average quantity-and-price pair  $(\bar{Q}, \bar{P})$ . If  $(\bar{Q}, \bar{P})$  is on the right hand side of the equilibrium between WD and CD curves, increasing the slope of whaling demand will decrease the harvest level, and vice versa. The same holds for changing the slope of the quadratic WD curve at  $(\bar{Q}, \bar{P})$  with a fixed

choke price (Figure 3.2b). Increasing the choke price of a fixed slope of quadratic WD curve results in a greater harvest level (Figure 3.2c). Figure 3.2d shows the interplay between the two demand curves. If the WD curve is relatively low and steep, increasing the CD slope also increases the harvest level, and vice versa. A decrease in the participation level of conservationists (more free-riding) or in the total WTP for whale conservation will shift the CD down, resulting in less conservation effort and a higher harvest (Figure 3.2e).

Figure 3.2. A diagrammatic Exposition of How Each Factor Affects the Nature of the Whale Permit Market Equilibrium







Note: (a) illustrates the effect of slope for linear whaling demand on harvest level while keeping the average whaling quantity and price pair constant. Once the whaling quantity and price pair are fixed, the slope of linear whaling demand is correlated with choke price of whaling demand. (b) illustrates the effect of slope for quadratic whaling demand on harvest level while keeping choke price, the average whaling quantity and price pair constant. (c) illustrates the effect of choke price for quadratic whaling demand on harvest level while keeping slope, the average whaling quantity and price pair constant. (d) illustrates the effect of relative magnitude of slope for whaling demand and conservation demand on harvest level. (e) illustrates the effect of free-riding on the harvest while keeping whaling demand, willingness to pay, the number of conservationists and the slope of conservation demand curve constant. CD: conservation demand, WD: whaling demand.

For the remainder of the analysis, I focus on the steady-state harvest level,  $H^*$ , under different settings. To examine the sensitivity of  $H^*$  to different parameters, I estimate linear regressions of steady state harvest level on the choke price and slope of WD, the slope of CD, the participation level of conservationists, and the WTP for minke whale conservation under different whaling and conservation demand combinations. Since the independent variables have different units and domains of support, I standardize each variable so that each has a mean of 0 and a variance of 1 before running the regression analysis. The resulting standardized coefficient ( $a$ , for instance) of independent variable X is interpreted to mean that increasing X by one standard deviation will increase the dependent variable by  $a$  standard deviations. Table 3.2 suggests that, all

else being equal, increasing the participation level of conservationists, each participant's annual willingness to pay (WTP) for minke whale conservation, and the slope of CD, decreases the steady-state harvest level. On the other hand, increasing the choke price and the slope of WD increases steady state harvest. Of all the parameters examined, the steady harvest level is most sensitive to the participation level of conservationists, indicating its role in the viability of a market. The results also suggest that the steady-state harvest level is more sensitive to the parameters associated with the CD curve than those with the WD curve.

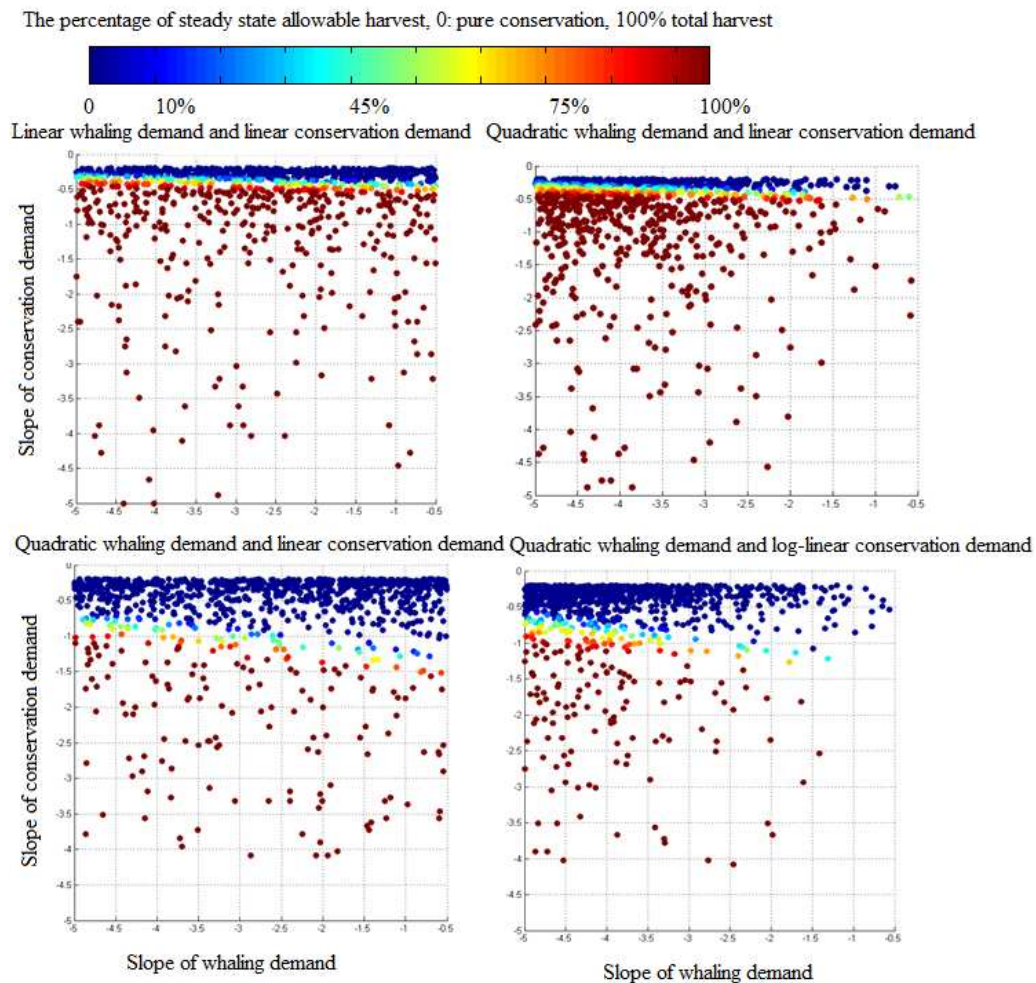
Table 3.2. Parameter Estimation from Standardized Regression of Different Whaling and Conservation Demand Combinations

Variable Name	L-L	L-N	Q-L	Q-N
Choke Price of Whaling Demand	0.09	0.08	0.1	0.18
Slope of Whaling Demand	NA	NA	0.01	0.09
Slope of Conservation Demand	-0.41	-0.37	-0.4	-0.36
The Participation Level of Conservationists	-0.55	-0.69	-0.57	-0.69
WTP for Minke Whale Conservation	-0.5	-0.53	-0.56	-0.58

Note: Since perfect multi-collinearity exists between the choke price and the slope of whaling demand in cases involving linear whaling demand, I drop the slope of linear whaling demand in the standardized regression, and NA denotes "non-applicable". In the variable names X-X, the first letter denotes the functional form of whaling demand, and the second denotes the functional form of conservation demand, L denotes linear whaling/conservation demand, N denotes nonlinear conservation demand, Q denotes quadratic whaling demand.

The resulting steady-state harvest levels can be broadly categorized into three solution/allocation types: pure conservation solutions ( $H^* = 0$ ), interior solutions where whaling and conservation coexist, and maximum harvest solutions. Figure 3.3 shows how the slopes of the two demand curves interplay to impact the type of solution  $H^*$ . The

Figure 3.3. Combinations of Slope for Whaling Demand and Conservation Demand Lead to Pure Conservation (Deep Blue Color), Maximum Harvest (Deep Red Color), and Coexistence between Whaling and Conservation

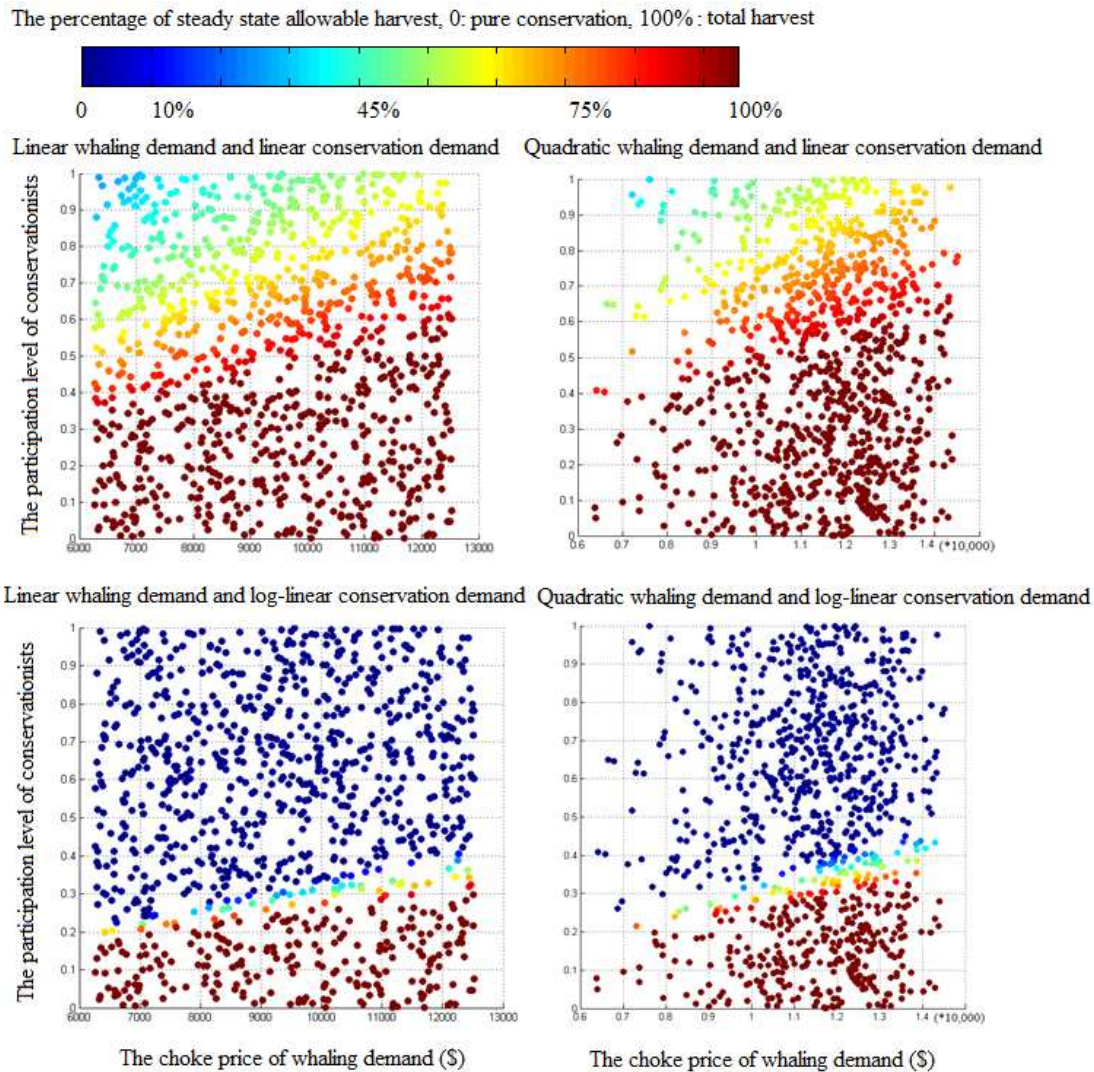


Note: The unevenly scattered points should not be interpreted as that the density is more located around low slope of conservation demand. This is because the elasticity of conservation demand is sampled from a uniform distribution instead of the equivalent slope. By sampling evenly from elasticities, a high slope of conservation will be under-sampled due to the nonlinear transformation between slope and elasticity. Basic parameter values used in constructing this figure include: free-riders are 50% of total conservationists, the total conservationist number is 20,000,000, and each conservationist's annual WTP for minke whale conservation is \$25.

results suggest that the CD slope is a more influential factor than the WD slope: as the CD slope becomes steeper, the market outcomes transition from pure conservation ( $H^* = 0$ ) to interior and ultimately to maximum harvest solutions. The resulting solution type depends on the distribution of these two demand slopes. Importantly the “likelihood” of any given scenario should not be judged by the size of regions or density of points in the figures as there is no a priori reason to believe that the uniform densities used in the Monte Carlo sampling scheme are reasonable subjective prior distributions.

Figure 3.4 shows that  $H^*$  undergoes the same transition—pure conservation to interior to maximum harvest—as the whaling choke price and free-riding levels increase (i.e., lower conservationist participation). Combinations involving linear conservation demand generate only interior and all harvest solutions. Combinations involving log-linear conservation demand generate all three solution types. Note that given a particular elasticity and price/quantity pair the conservation demand curve for a linear model lies below the nonlinear conservation demand curve – the result being that the willingness to pay in the marketplace does not fall off as rapidly at high levels of conservation as a linear demand with the same “local” behavior at average price/quantity values. The difference between combinations involving linear conservation demand and combinations involving nonlinear conservation demand also results from the fact that linear conservation demand suggests that the rate of change of the slope of the linear conservation demand curve (curvature) at lower levels of harvest and high levels of conservation is the same. In other words, marginal WTP falls fairly strongly regardless of stock status. A non-linear conservation model, on the other hand, suggests that marginal

Figure 3.4. Combination of the Participation Level of Conservationists and Choke Price of Whaling Demand Lead to Pure Conservation (Deep Blue Dot), Maximum Harvest (Deep Red Dot) and Coexistence between Whaling and Conservation.



**Note:** Basic parameter values used in constructing the figure include: the total conservationist number is 20,000,000, and each conservationist's annual WTP for minke whale conservation is \$25, slope for whaling demand is -2.5, and elasticity for conservation demand is -2.

WTP declines rapidly at the lower stock level, but relatively slowly as stock rise to high level. WD with a higher choke price requires a higher participation level of conservationists in order to achieve the same harvest level. Importantly, the transition from pure conservation to maximum harvest is much sharper for the log-linear CD curve (compare Figs 4c and 4d to Figs 4a and 4b). The transition process from one solution type to another one also depends on the distribution of the choke price of whaling demand and the participation level of conservationists.

### 3.6 Discussion

The regulation of whaling implies tradeoffs between the preferences of two distinct groups, whalers and conservationists. The appeal of a market system is that it has the potential to replace an adversarial system of arguing in support of one extreme allocation or the other with a mechanism to enable whalers and conservationists to peaceably act upon their convictions given the resources that each group can marshal. A whale permit market provides a platform for participants to “put their money where their mouth is” – allowing each party to reveal their values in an environment that reflects the scarcity of resources (i.e. that neither whale conservation nor the maintenance of whaling are infinitely valued by society). A market also reallocates resources in a manner that acknowledges the de facto property rights of whalers under the current system while allowing them to be compensated for forsaking these rights through voluntary transactions that mutually benefit both parties. Hence a market approach avoids deciding if one group is intrinsically right and the other wrong, instead operating in shades of gray where compromise is implicitly reached. If the market outcomes resulted in complete

conservation – effectively a “buyout” of the whaling fleet – this would suggest that whale harvest is not as essential as claimed by whalers. On the other hand, if the market outcome did not yield significant conservation, then this either suggests that the real willingness to pay of conservationists is far lower than is suggested by the claim that the conservation of whales is essential, or that this conviction is at least not sufficiently strong to overcome strong free-riding incentives.

The utility of the Norway example is that whale conservation market can serve as an experimental whale conservation market to manage the disputes between pro-and anti-whaling stakeholders in nations that still have whaling. A potential whaling buyout may be the result of such a regional whale conservation market, while the moratorium could be maintained elsewhere. This would truthfully reflect the *de facto* property rights as they currently are – conservationists implicitly have the rights to the whales that would have been hunted in compliant countries while whalers have the rights to those whales that they have historically been able to hunt for scientific whaling, noncompliance and subsistence purposes. This also suggests an interesting extension of the market system to other countries whose whaling takes place beyond its exclusive economic zone (EEZ), although considerable monitoring and enforcement challenges arise in that context (Smith et al., 2014).

While it is impossible to disentangle the exact expenditure that NGOs (nongovernmental organizations) spend on minke whale protection, a conservative estimate of the total annual expenditure by NGOs on all anti-whaling activities is approximately \$25 million (Costello et al., 2012). If the Norwegian government set up a transferable whale permit market it would be expected that demand for whale

conservation would be high. The average equilibrium price in all whaling and conservation combination from the simulations carried out is in the range of approximately \$7,300 to \$8,900. With whaling quotas in the steady state of 475 whales, this suggests that the implied annual expenditure from conservationists to whalers, if conservationists buy all permits, is approximately \$3.47 million to \$4.23 million. This is around 14% and 17% of the annual expenditure by NGOs, respectively.

Despite such potential, a market-based approach to whale conservation management is complicated by uncertainties that must be taken into account to anticipate outcomes and design institutions to achieve its goals. In this study, I use (limited) data on minke whales to devise an experimental whale permit market in order to evaluate the potential of the market approach under these uncertainties. The study has demonstrated how different sources of economic uncertainty impact the whale permit market forecast.

An important political and economic question related to a whale permit market system is what the long-run equilibrium allocation and policy might look like. I find that the market approach might result in a variety of outcomes ranging from pure conservation to full harvest, or a balance between the two. Which of these different outcomes actually results could be reflected by replacing the "ignorance priors" of the uniform distribution with priors on parameters that reflect a sense of knowledge about them – perhaps drawing upon expert elicitation methods (Martin et al., 2011). Then the frequency of different outcomes in the simulations could provide information on their relative likelihood.

The greater influence of the CD curve, compared to the WD curve, stems from the available data and the corresponding calibration procedures. Linear and quadratic WD



curves are both calibrated with the average quantity-price pair,  $(\bar{Q}, \bar{P})$ . This  $(\bar{Q}, \bar{P})$ , combined with a reasonable range of WD slope, results in a relatively low choke price of whaling, therefore generating a relatively lower and steep WD. This seems quite consistent with the common perception of whale meat products demand around the world. On the other hand, the CD curve is calibrated on information regarding total WTP for minke whale conservation and the number of conservationists. This leads to a much higher and flatter CD. These two calibrated WD and CD will produce an equilibrium harvest that is relatively close to the WD axis (i.e. at low levels of harvest). This is the reason that parameters associated with CD side can result in a significant change in harvest level while parameter changes in the WD side have less impact on the harvest level. The above argument applies only to this particular (very) limited set of data, and the sensitivity of the steady-state harvest level,  $H^*$ , highlights the importance of data collection related to both whaling and conservation.

As discussed above, free-riding is a serious concern for the voluntary provision of a public good (Bergstrom et al., 1986; Cornes and Sandler, 1986). Intuitively, how many people participate vs. free-ride in the market directly impacts the potential outcome of the whale permit market, which is also confirmed by the statistical results. This outcome resulting from free-riding will be expected to deviate from 100% participation equilibrium. When conservationists have to compensate whalers to forgo some harvesting, there is an incentive for some conservationists to free-ride on other conservationists' contributions, resulting in an under-provision of whale conservation in the market. In the case where whalers must compensate conservationists to harvest (not the scenario modeled here), there is of course no incentive for conservationists to free-

ride.<sup>14</sup> Therefore, there are reasons to doubt the usual “neutrality” results in economics, which proposes that the initial allocation of rights in a market should not affect the equilibrium outcome (Fowlie and Perloff, 2013). The potential harm from free-riding is bounded under the management scenario envisioned in this paper due to the fact that harvest is constrained by the allowable catch limit. If the resulting market equilibrium harvest level—with no free-riding—is already greater than the maximum allowable harvest, then the imposition of the maximum allowable harvest will make the level of free-riding irrelevant under such circumstances. In other cases, if a “whaling buyout” occurs even with a significant degree of free-riding, then the level of free-riding is also not important to the result. Furthermore, the possibility that large conservation or animal-rights NGOs may participate in a whale permit market may mitigate the severity of free-riding due to their ability to consolidate the contributions of their donors. While incentives to free-ride between NGOs will persist, these incentives will likely be far less severe than in the individual donor case. Also, the transaction costs of overcoming these free-riding incentives may be relatively low given the small number of large NGOs and their existing cooperation in other domains.

The whaling choke price, unlike the parameters on CD, might actually be estimable, at least to a “ballpark” level, from market data to some reasonable extent, assuming profit maximization on the part of whalers. Alternatively, the choke price and elasticity for whaling demand can be estimated from market data using econometric

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<sup>14</sup> However, moral abhorrence (and potential shaming of individuals and NGOs) at the prospect of selling a whale permit to a whaler (and, thereby, “sentencing a whale to death”) could significantly elevate conservationists willingness to accept payment for this transaction relative to their willingness to pay to avoid the same outcome.

methods. While whale meat consumption is quite common in Norway, the elasticity of whale meat demand is probably in the ballpark of the elasticity of demand for beef arguably because beef is in the similar market and can serve as a close substitute for whale meat. This may provide a probable range for the whaling choke price, perhaps ranging between \$9000 to \$11,000 (Horan and Shortle, 1999)<sup>15</sup>. This indicates that for a particular level of free-riding, it may be possible to get a reasonable sense of just how likely the persistence of whaling is in a market. The degree of free-riding suggested in experimental research for the provision of public goods ranges from 40% to 60% – the percentage measuring the share of free-riding-of the optimal endowment level in a one-shot game, with the upper bound reflecting outcomes in a repeated public good game (Chaudhuri et al., 2006).

In the context where the conservation side of the whale market is dominated by NGOs, it is possible that free-riding could be contained by the use of a third-party “charity rating” that reports to potential donors the numbers of whales “saved” by competing NGOs (Chhaochharia and Ghosh, 2008). A low “rating” on saving whales will likely turn potential donors to other NGOs who receive higher ratings. Such a rating system utilizes the stiff competition between NGOs for donors to reduce free-riding behavior. A brief browse on Sea Shepherd website demonstrates the stiff competition fund raising among NGOs: Sea Shepherd accused Greenpeace of fraud, and that Greenpeace has been raising millions of dollars from anti-whaling campaigns and yet betray whales. While we have primarily viewed the conservation market as operating

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<sup>15</sup> Horan and Shortle (1999) calibrated the linear inverse whaling demand  $p=10991.04-0.73075Q$  with elasticity equal to -1, and 1979-1983 whaling quantity and price data from Amundsen et al. (1995). Here the probable lower bound \$9000 is adjusted by lower average price and whaling quantity.

through NGOS, private participation is possible as well, driven in part by philanthropists motivated in part by warm glow or through social approval through conspicuous display of documentation of their activities as a whale “life saver” (Andreoni, 1988, 1995). On balance therefore, a 40-60% participation rate of conservationists seems entirely plausible. Combined with the probable range of the choke price for whaling demand, this participation level would most likely generate a pure conservation policy and least likely result in a pure harvesting policy under a nonlinear conservation demand.

This study only focuses on long run steady state population and harvest level. The trajectory to the steady state, however, also deserves attention and may impact the stability and success of such a market. The whale permit market makes it possible to reallocate whale permits among stakeholders. Within the market system the whalers are entitled the initial whale permit due to the grandfathering allocation; conservation groups on the other hand have to buy into the market. The effectiveness of a whale permit market for conservation depends on the response of the fishery manager to changes in stocks. If an increase in whale numbers prompts an increase in permits, as dictated by the PBR rule assumed in this paper, then conservationists may face increasing costs over time – driven in part by the success of their own conservation efforts in previous market periods! For a whale species that is approaching carrying capacity, such as minke whale in Central North Atlantic, a quota setting will ensure that the initial permit quota for the first year will be maximal. As the market progresses year by year, the permit quota allocated to whalers will be less and less because minke whale population will decrease from the carrying capacity level to a relatively lower steady state level and conservationists will buy less in a high stock level. On the other hand, if whale species is

of low abundance, the permit quotas allocated to whalers will be increasing. The reason is that whale population will be increasing from low level to a relatively higher steady state level because in a low stock level conservation groups will buy and retire whaling permit. However, this suggests an increasing cost for conservationists. Under such a grandfathering rule if conservationists foresee this trajectory of increased costs, they have a “rent-seeking” incentive to oppose the market.<sup>16</sup> Instead, they resort to alternative strategies by lobbying and disrupting whalers to get the same or even greater conservation with cheaper cost. By doing so, they are also able to claim a moral high ground for anti-whaling activities and potentially secure benefits from fundraising. The issues of the implied income distribution and resource allocation need to be addressed before the establishment of such a market.

The correct shape of the CD curve is essential in evaluating the robustness of conservation success against free-riding, as indicated in Figure 3.4. Despite being calibrated to the same dataset, the log-linear CD curve admits a pure-conservation, or “buyout,” outcome even with high levels of free-riding (i.e., only 20-30% participation level) over the range of whaling choke prices explored, whereas the linear CD curve does not do so even with the full participation level. A linear conservation demand curve indicates that marginal WTP decrease quite rapidly regardless population level. If conservation demand mostly is driven by the conservation concerns or animal welfare, then there is a “saturation effect” and the marginal WTP should decline fairly strongly at a low stock level but then decrease slowly as stocks rise to high levels. In that case, a

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<sup>16</sup> One possibility to stave off this behavior would be to replace the strictly increasing PBR rule with one in which whaling quota is only allowed to increase to a maximum, “ceiling” level (perhaps tied to recent harvest levels).

nonlinear conservation demand is appropriate. One might also think that the conservation demand is driven by concerns for individual whales or the avoidance of suffering, in which case conservation demand may be very unresponsive to population levels across a wide array of population levels – causing a fairly flat conservation demand. The importance of the correct shape of the demand curves again highlights the importance of better data collection.

Many concerns about whale permit markets are dominated by uncertainties—both biological and economic. Using the limited data on minke whales, I focus on uncertainties associated with the whaling demand and conservation demand. The study shows that the whale permit market mechanism is particularly sensitive to features of the conservation demand curve; unfortunately, the data needed to fully examine these uncertainties are not currently available. This study illustrates a comparatively low-risk case where an experimental market for the adaptive management of a small-scale minke whale stock can help understand the ecological effect and risk of the management policy, and could yield substantial transferable insights. These, along with better data availability and comparison across different whale populations, can help shape subsequent policy and to assess market viability for systems on a larger scale.

## Chapter 4

# A BIOECONOMIC PERSPECTIVE ON THE MANAGEMENT OF SEA LICE EXTERNALITIES BETWEEN SALMON AQUACULTURE AND WILD FISHERIES

### 4.1 Introduction

Aquaculture is a rapidly growing industry that has become a major supplier of fish and shellfish to the global market (Naylor and Burke, 2005). According to the Food and Agriculture organization (FAO, 2011), aquaculture now accounts for 40.1% of global fish production. There are a number of issues that have constrained the development of aquaculture along the coast worldwide. For instance, shrimp and salmon, two of the most lucrative and widely traded aquaculture products, are subject to criticism for the environmental impacts of waste discharges, escapees, use of chemicals and drugs, consumption of fishmeal and fish oil, and disease and parasite spreading (Naylor and Burke, 2005; Morton et al. 2004; Morton and Routledge 2005; Krkošek et al. 2005 & 2006; Asche et al., 1999). The economics of some of these issues have already been investigated. For instance, Sylvia and Anderson (1993) proposed a tax for aquaculture waste pollution. This study aims to evaluate the impact of disease transmission from aquaculture to wild fisheries, already recognized as one of the most serious threats to the sustainability of marine aquaculture and the viability of wild fisheries (Bulter, 2002; Naylor et al, 2000; Morton et al. 2003 and 2004; Krkošek et al. 2007).

Marine fish farms may have a reservoir-host effect on disease transmission and pose a threat to adjacent wild stocks. Disease transmission from fish farms to wild fisheries represents a negative externality during the fish farming production process, the

value of which is the costs incurred by the affected adjacent wild capture fishers<sup>17</sup>. For fish farmers, this is an external cost. It is not considered in their decision process without the relevant policy being in place or through spatial consolidation implemented by credible regulators. Regulatory authorities currently use both trigger thresholds and farm-fallowing after harvest to protect the wild fishery. These policy recommendations come from traditional disease ecology models (such as Anderson and May's (1978) susceptible-infected model, Grenfell and Dobson, 1995; Krkošek et al. 2005 & 2006). While such epidemiological models are useful for understanding the system, they have limited utility in terms of providing management recommendations because they treat human behavior as external factor, and are not rooted in decision theory (Albers, et al., 2010; Fenichel et al., 2010). Moreover, the causes of many disease problems, including pest, parasite and invasive species, are economic, and hence require economic solutions (Perrings et al., 2002). As an alternative for providing management guidance, bioeconomic models incorporate information from epidemiological systems into an economic decision-making framework in which the level of disease control is endogenous (Horan et al., 2010).

This study focuses on the management of sea lice externalities between salmon aquaculture and wild salmon fisheries. When wild stocks migrate to a fresh water environment in the fall for spawning, lice from wild stocks disperse into fish farms located on the migration route of wild stock and infest the farmed fish. If not treated in the farms, the lice will grow rapidly and re-infest wild juveniles when they emigrate into

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<sup>17</sup> This study only focuses on the disease impact on wild fishery. However, disease may also influence the marine ecosystem, for instance by changing the biotic structure and distribution (Burge et al, 2014).



marine environment from May to July. The salmon louse (*L. Salmonis*) has a free living phase and a parasitic phase in its 6-8 week life cycle. Once the lice attach to a host salmon, they feed on mucous, blood, and skin which causes morbidity and mortality (Pike and Wadsworth, 2000; Costello 2006). Although the disease problem associated with fish farms is widely recognized<sup>18</sup>, there is very limited work to quantify the ecological and economic impact of the disease on both farmed and wild fisheries. It is reported that sea lice may cost the salmon industry US\$480 million a year and 6 percent of product value (Costello, 2009). Liu, Sumaila and Volpe (2011) explore the ecological and economic impact under fixed exploitation policy and target escapement policy by exogenously and randomly set sea lice induced mortality rates. There is, however, no sea lice population dynamics in their framework.

Prior bioeconomic studies have used an optimal control framework to study disease and pest control policy from both private and/or social perspectives (Gramig et al., 2009; Gramig and Horan, 2011; Epanchin-Niell and Wilen 2012; Sims and Finnoff 2012; Horan et al., 2008; Fenichel et al., 2013; Richard et al., 2010). This study adds to the literature by investigating the externality between farmed fisheries and wild fisheries due to sea lice transmission. To the author's knowledge, this is the first to integrate sea lice population dynamics with economic modeling to determine the optimal control policy from the perspective of salmon aquaculture, and then from the perspective of the joint fishery. By taking account of the complex relationship between sea lice populations, farmed fisheries and the wild fisheries, it is able to evaluate how much privately optimal

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<sup>18</sup> Natural conditions produce a far lower infestation on wild juveniles than salmon farms (Morton, 2011). Krkošek et al. (2005) found that the maximum infection pressure (sea lice larval production per unit space) near farms was 73 times greater than ambient levels.

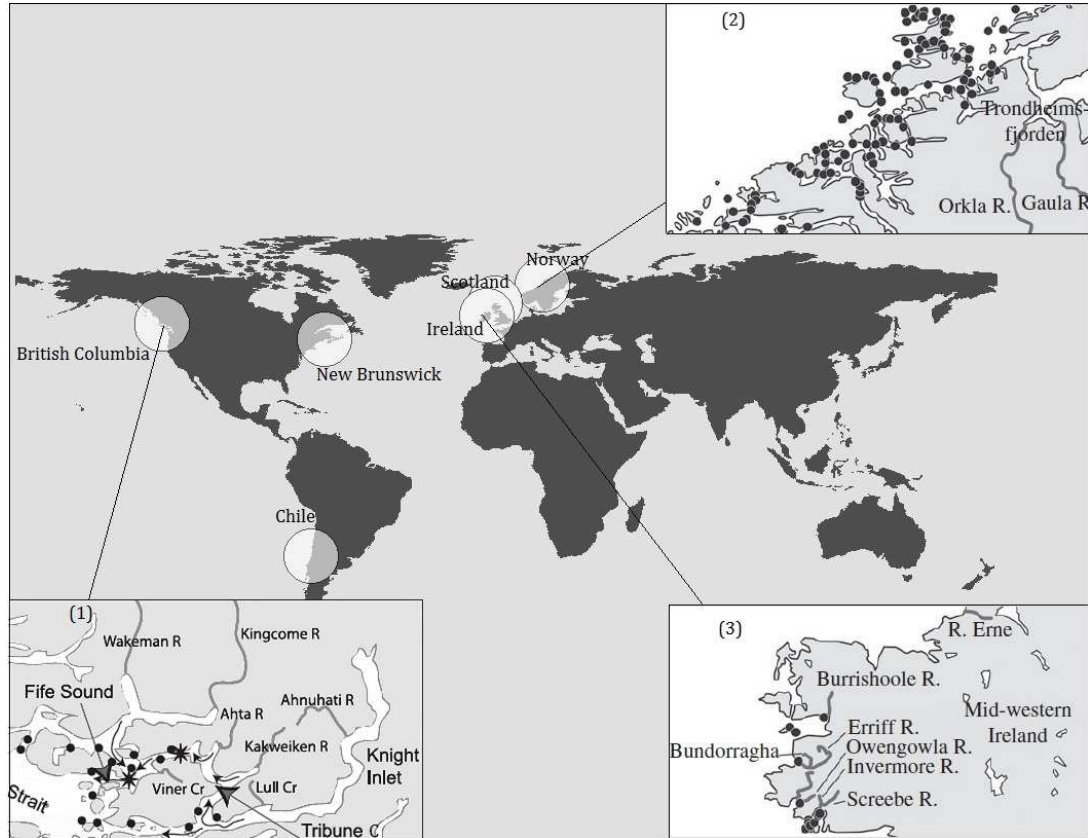
treatment behavior deviates from the socially optimal solution, and to assess the economic impact of salmon aquaculture on the wild fishery due to sea lice transmission.

The outline of the chapter is as follows. Section II describes the general structure of salmon aquaculture and the wild fishery. The various components of the bioeconomic model are provided in section III. The main results and the outcome of numerical simulations are presented in section IV. Finally, section V discusses the results and draws conclusions.

#### 4.2 The General Structure of Salmon Aquaculture and Wild Fishery

The past several decades have witnessed the transformation of salmon aquaculture into a global industry. As shown in Figure 1, salmon farms are located along coasts that anadromous wild stocks swim through during their inward and outward migration. Salmon farms are typically stocked with several hundred thousand fish (Orr, 2007), and farms are highly concentrated in a small area. For instance, in the Broughton Archipelago of about  $117 \text{ km}^2$  in British Columbia, Canada, there are twenty-nine licensed salmon farms owned by three companies (Liu, et al, 2011). These farms are clustered near the river outlets. The farm sites are chosen initially for easy access to the market and suitable environmental conditions for raising fish. Elsewhere, salmon farming may be less concentrated than in the Broughton Archipelago area, but is more widespread. There are, for example, approximately 850 licenses for fish farming in Norway (Färe et al., 2009).

Figure 4.1. Major Salmon Aquaculture Production Regions around the World



Note: Pic (1) is from Krkošek et al. (2007); pic (2) and (3) are from Krkošek et al. (2012).

Anadromous species, such as salmon and trout, migrate between freshwater and seawater environment during their life cycles. The spawning stage takes place in freshwater environment, and the main growth stage takes place in the ocean. Depending on the species and location, most wild stock runs from a river are likely to mingle with the runs from other rivers within the 200 nautical mile zone of a country's exclusive economic zone. In some cases, salmon runs from the rivers in one country will mingle with those from other countries in the open ocean (Ebbin, 2005). Most wild salmon fishing activity takes place during the spawning migration, and the fish are harvested offshore, inshore and then in the rivers.

Sea lice are native ectoparasite copepods, common on both farmed and wild adult salmon<sup>19</sup>, but unable to survive in a freshwater environment. Lice infection on Pink salmon and wild salmon is most reported and studied (Krkošek, et al., 2005, 2007; Morton et al., 2004; Marty et al., 2010) due to their vulnerability. When the wild stocks migrate back to the parent river for spawning, they carry adult gravid lice. These adult gravid lice reproduce and spread into any farms close-by, and re-infest the wild juvenile stocks when wild juvenile migrate into ocean environment. The presence of highly concentrated salmon farms around the coast may change the dynamics of sea lice and wild stocks. The impact of sea lice on wild stocks has been debated extensively. Some believe that sea lice are only one of many factors that affect wild stock levels (Brooks, et al, 2005; Brooks and Jones, 2008; Riddell et al., 2008; Marty et al., 2010); while others claim that where salmon net-pens provide ideal conditions for sea lice, they are a primary source of loss of vulnerable migrating wild juveniles (Krkošek, et al., 2005, 2007; Morton et al., 2004; Morton and Routledge, 2005; Frazer, 2008).

Wild salmon fisheries differ from salmon aquaculture in terms of management objectives, institutional structure and regulations. The typical management objective for a wild salmon fishery manager is to conserve salmon populations and restore their habitat to avoid overexploitation. The management objective for salmon aquaculture, on the other hand, is to maintain and promote the profitability and competitiveness of the aquaculture industry. Wild fishery managers would typically impose restrictions on the

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<sup>19</sup> Early stage sea lice are also found on sticklebacks, but no mature sea lice have been found on sticklebacks (Jones et al., 2006). Hence, sticklebacks may serve as a temporary reservoir for early stage sea lice. However, the epizootics on juvenile salmonids consist of the same early life-stages of sea lice, which are also found on sticklebacks. This suggests that both juvenile salmonids and sticklebacks are infested by the same sources rather than contributing to each other's infestations (Costello, 2009).

limit (cap) on overall licensed numbers harvested in the wild capture fishery, along with input/gear restrictions, closed seasons and areas.

One commonly used management objective for wild fisheries is to ensure escapement spawning targets to ensure potential future harvest. Another is to set a fixed harvest rate in order to maintain the harvest level for fishermen. Both aquaculture and wild fisheries can be managed by the same or by different authorities. In Norway, for instance, wild salmon stocks are managed by the Ministry of Environment, while salmon aquaculture is managed by the Directorate of Fisheries (Liu, et al, 2010). In Canada, salmon aquaculture has been traditionally regulated by the provincial Ministry of Agriculture and Lands, while the conservation of wild Pacific salmon is in the hands of the federal Department of Fisheries and Oceans (DFO). In 2009, the British Columbia Supreme Court transferred regulatory authority of salmon aquaculture from provincial authority to DFO, which indicates that DFO is responsible for managing both wild salmon and salmon aquaculture in British Columbia (Krkošek, 2010).

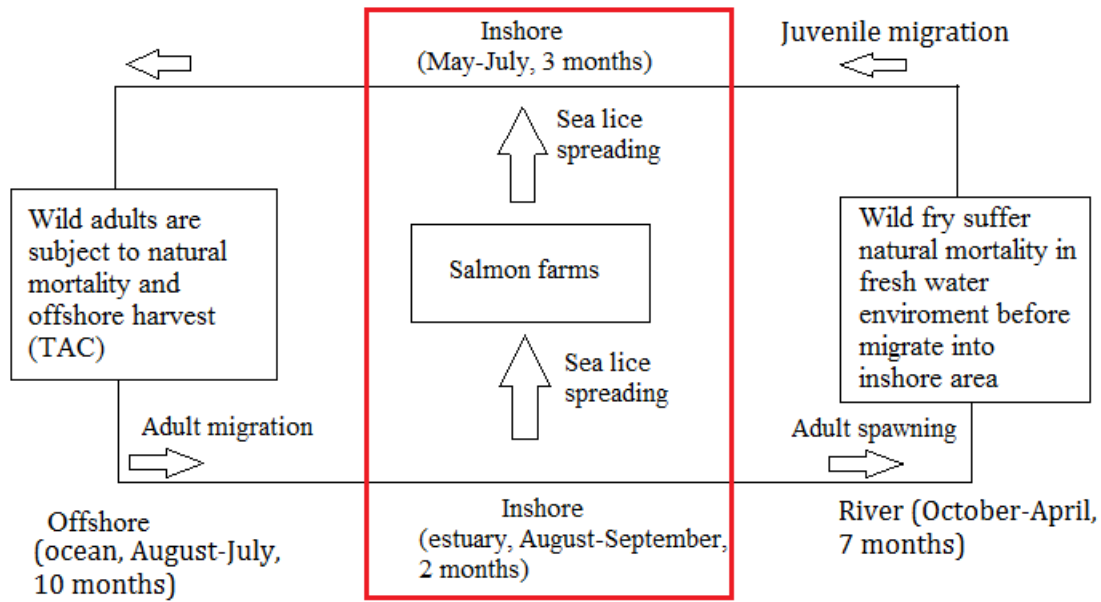
## 4.3 Model Description

### 4.3.1 Sea Lice Dynamics

This study does not refer to any specific system; instead it is intended to abstract the core characteristics of the general system described in previous section. The simplified structure of the wild fishery and salmon aquaculture is illustrated in Figure 4.2. Consider a coastal area where an Atlantic salmon (*Salmo salar*) farm (or a coordinated aquaculture industry consisting of many farms) is connected by the free-living stage of sea lice transmission with wild fish stocks when they migrate in and out. The farm

manager releases salmon smolts ( $F_{f,0}$ ) into fish farms right before wild adults head home for spawning and harvests all fish 24 months after they are released<sup>20</sup>. The subscript  $f$  denotes farmed fish, and subscript  $w$  denotes wild fish. Note that wild juvenile pink salmon would typically remain inshore for several months before going out to sea<sup>21</sup>.

Figure 4.2. The structure of the Wild Pink Salmon Fishery and the Lice Dispersal between Wild Salmon Stock and Atlantic Salmon Farms



Note: TAC: total allowable catch. The red rectangle depicts the area where sea lice dispersal takes place between wild fisheries and salmon farms.

<sup>20</sup> Fish farmers either use batch harvest of fixed interval to target specific markets, or grading harvesting operation during the whole out-growth season. Here this study focuses on aquaculture production of fixed interval under batch harvesting operation. For grading harvesting operations, some farmers may release smolts at different times in order to supply the market all year around. This is also implied by the literature investigating the optimal harvesting time instead of setting  $T$  exogenously.

<sup>21</sup> See, <http://www.adfg.alaska.gov/FedAidpdfs/FRED.FishCultureManual.pdf>. However some papers model wild pink juvenile salmon swimming routes as a linear pattern from inshore area to the sea (Krkošek, 2006).

I lump sea lice production into a free-living copepodite phase and an adult lice phase. Sea lice cannot survive in a fresh water environment. Due to the relatively brief spawning migration (August and September) of wild pink salmon from the marine environment to fresh water environment, gravid lice from homecoming wild adult stocks would pass through coastal areas and infest farms by spreading the copepodite produced by gravid lice ( $L_{w,t}$ ). A higher sea lice infection rate in the abundant wild population will lead to a greater risk to salmon aquaculture. I assume that there is always a fixed proportion of total gravid lice ( $\varsigma$ ) remaining at the coast when the wild stock migrate upstream for spawning. These lice produce copepodites. Copepodites have some probability  $\rho$  of attaching to farmed ( $F_{f,t}$ ) or wild ( $F_{w,t}$ ) hosts if present, and survive to adult lice stage with probability of  $\psi_t$ . The settlement success  $\psi_t$  is subject to environmental factors, such as salinity and water temperature (Tucker, et al., 2000; Bricknell, et al., 2006; Groner et al., 2014). In this study, settlement success  $\psi_t$  is assumed to be periodically forced, and takes the form,

$$\psi_t = \varepsilon_1 + \varepsilon_2 \sin\left(\frac{2\pi}{12}t\right) \quad (4.1)$$

This simple sinusoidal function assumes a 12 month periodicity and  $t=1, 2, \dots, 12$ , and has a seasonal force impact coefficient  $\varepsilon_2$  and a base settlement success  $\varepsilon_1$ .

The emigrating wild juveniles, when they pass through fish farms located close to wild migratory routes in May to July, are subjected to lice infestation from fish farms. These wild juvenile are at their most vulnerable because of their small size, and also subject to the environmental stress because they are transitioning from fresh water environment to marine environment. I further assume that when juvenile wild stocks

migrate into the ocean environment, a fixed proportion of non-attached copepodites from farms re-infest the juvenile wild stocks. I also assume that the chemical treatment  $u_t$ , if applied, kills both adult sea lice and copepodites on farmed fish. The mortality associated with chemical treatment for copepodites and adult sea lice is denoted  $k$  and  $z$ , respectively.

Consider a system where copepodites are well mixed instantaneously, and the copepodite density is the same across the coastal area. Since I focus on dispersal among farms and wild stocks, instead of the dynamics between infectious fish and susceptible fish, I also assume that sea lice within farms are well-mixed. Natural mortality for adult lice is denoted by  $v$ . Adult lice can produce copepodites at a rate of  $\lambda$ , which are also subject to natural mortality  $\xi$ , and attach to fish at rate  $\rho$ . Copepodite transmission between farmed and wild fish only happens during the spawning migration of wild adults and the emigration of juvenile wild stocks.

Let  $X_t$  denotes total copepodite abundance in the coastal area at time  $t$ , then a discrete model<sup>22</sup> for sea lice dynamics in farms is,

$$L_{f,t+1} = \rho \psi_t X_t F_{f,t} + L_{f,t} * (1 - v) * f_l(u_t) * F_{f,t} / F_{f,t-1} \quad (4.2)$$

$$X_{t+1} = \lambda (L_{F,t} + L_{w,t}) + X_t (1 - \rho F_{F,t} - \rho F_{w,t}) * (1 - \xi) * f_c(u_t) \quad (4.3)$$

I assume that in this free-mixing system, the copepodites will not attach to wild spawning adults. This is because that the wild spawning adult will migrate upstream, where the sea lice cannot survive free water environment. However, when wild juvenile

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<sup>22</sup> In these equations,  $\rho$  is calibrated from Frazer et al. (2012) to make sure the term  $(1 - \rho F_{F,t} - \rho F_{w,t})$  is bounded between 0 and 1. I conduct the sensitivity analysis for this parameter in later section.



migrate into the ocean, they will be infested. Therefore,  $F_{w,t}$  in Eq (4.3) denotes the abundance of wild juveniles at the end of month  $t$ .  $L_{w,t}$  is total gravid lice from spawning wild stocks, and  $L_{w,t} = sF_{w,t}$ . Here  $s$  represents the gravid lice number on each individual spawning wild fish, and is assumed to be maintained by processes occurring in the offshore ocean environment. This variable is exogenous and assumed to be constant (Frazer et al., 2012).

Eq (4.2) gives the total lice number in farms at the beginning of each time unit (month) as the sum of the newly mature adult and the lice remaining from last period. The first term on the right hand side (RHS) of Eq (4.2) is the number of copepodites attached to hosts that become adult. The second term on the RHS is the number of lice remaining from last period. There are four components of this term. The first is the total lice number at the beginning of last time unit, the second,  $(1-\nu)$ , and the third,  $f_l(u_t)$ , are the surviving proportions after natural mortality  $\nu$  and chemical treatment  $u_t$ ,

respectively. The surviving number of lice after the natural and chemical mortality

$L_{f,t} * (1 - \nu) * (1 - ku_{f,t})$  will lead to a loss of fish population from the initial  $F_{f,t-1}$  to

$F_{f,t}$ . Note here  $F_{f,t}$  is the farmed fish population at the end of month  $t$ . Therefore, the

last term,  $F_{f,t} / F_{f,t-1}$ , denotes the effect of fish killed between time  $(t-1)$  and  $t$ . I assume

that if 10% of fish are killed, then 10% of total lice will also be killed, and hence if all fish are harvested, then all lice would be killed as well.

Eq (4.3) models the dynamics of copepodites in the coastal area. The first term on the RHS is the number of copepodites produced by lice on farmed  $L_{f,t}$  and wild stocks

$L_{w,t}$ . Note that  $L_{w,t}$  is equal to zero when there are no adult wild stocks in the inshore area.

The second term has two components, the first component is total copepodite abundance after dispersal, and the second component is the surviving proportion after natural mortality and chemical treatment  $f_c(u_t)$ .

The efficacy interval of chemical treatment varies depending on sea location, lice genetics, and water temperature (Saksida, et al., 2010; Stone et al., 1999). Here I assume that chemical treatment has a decreasing efficacy over a three-month period but that treatment from three months back still kills lice and copepodites, which is commonly documented (Gustafson et al., 2006; Stone et al., 1999). The term that captures the effect of chemical treatment application for adult sea lice and copepodites, the kill function, is then modified as  $f_l(u_{t-2}, u_{t-1}, u_t) = (1 - ku_t) * (1 - 0.8ku_{t-1}) * (1 - 0.6ku_{t-2})$  and  $f_c(u_{t-2}, u_{t-1}, u_t) = (1 - zu_t) * (1 - 0.8zu_{t-1}) * (1 - 0.6zu_{t-2})$ , respectively.

In this study I make the assumption that the fish killed by lice do not have any economic value, but that infested but still live fish fetch the same price as un-infested fish. That is, the effect of sea lice is only evident through mortality, not other sub-lethal effects. Sea lice can, however, also reduce fish growth and feed conversion rates (Costello, 2006; Mustafa et al, 2001), although the literature provides little information on these affects. Therefore, in this study I fix the fish growth and feed conversion rates, and conduct sensitivity analysis for these two parameters. If the feedback of adult sea lice on weight accumulation is known, it could be easily incorporated in the model presented here. The impact of sea lice on the farmed salmon is modeled as

$$F_{f,t+1} = F_{f,t} \exp(-d_f L_{f,t} / F_{f,t}) \quad (4.4)$$

where  $F_{f,t}$  is Atlantic salmon abundance in farms at the end of time  $t$ (month), and  $F_{f,0}$  is the total Atlantic smolts released at farms. The farm manager harvests all fish in two years (batch operation) once the juveniles are released in the farms. When they harvest all fish, they also fallow the farms for a short time and all lice will be killed, and they manage to release the same amount of farm juvenile  $F_{f,0}$  in the same month.  $d_f$  is sea lice-induced mortality rate for farmed Atlantic salmon, and  $L_{f,t}$  is adult sea lice abundance in farms at time  $t$ .

Wild salmon juveniles ( $F_{w,t}$ ) are free of lice infestation in the river environment but are subject to re-infestation from fish farms when migrating into marine environment from May to July. The number of lice in month  $t$  is the sum of new adults developed from copepodites and attaching to hosts, and the remaining lice from last period. Note the lice remaining at month  $t$  after natural mortality ( $1-v$ ) will changes in the wild fish population level from  $F_{w,t-1}$  to  $F_{w,t}$ . The lice ( $L_{w,t+1}$ ) dynamics in a freely-mixed system is modeled as,

$$L_{w,t+1} = \rho\psi F_{w,t} X_t + L_{w,t} * (1-v) * (F_{w,t} / F_{w,t-1}) \quad (4.5)$$

I only model the lice dynamics inshore. Lice re-production dynamics also take place in the open ocean, and are subject to the processes occurring in offshore ocean environments, but are not modeled here. The effect of lice on wild fish stocks is incorporated into the constant natural mortality rate assumed to occur in the open ocean.

### 4.3.2 Wild Pink Salmon Population Dynamics

Homecoming wild pink salmon are harvested offshore before migrating to their home river for spawning. Pink salmon are the smallest salmon found in North America. They have less commercial value than other salmon due to their low oil content. Commercially caught pink salmon are often canned and prepared for complete packaged meals. Pink salmon (*O.gorbuscha*) have a two-year life cycle, and have even and odd-numbered year runs. These two runs are reproductively isolated. Spawning wild adults of even-numbered year runs do not encounter and infest wild juveniles of odd-numbered year runs during their first early marine life stage (Krkošek, et al, 2006).

Assume an initial escapement level,  $S_0$ , is known and is the same for odd and even-numbered year runs. Let  $S_n$  denote the escapement level of wild stocks at year  $n$ . This escapement level becomes the spawning stock, which has a concave fry production function following a Ricker relationship  $F_{1,n+1} = S_n \exp(\gamma - S_n / b)$ , where  $F_{1,n+1}$  is the number of fry in the next generation ( $n+1$ ) of 1 month old,  $\gamma$  is the population growth rate, and  $b$  determines density dependent mortality and is related to the carrying capacity of the system. The breeding stock dies after spawning. A natural mortality rate  $m$  applies to juvenile wild salmon in fresh water, therefore the population of surviving wild juveniles before migrating into the inshore area is  $(1 - m) F_{1,n+1}$ .

When wild pink juveniles migrate into and remain in the inshore area (May to July), they are subject to re-infestation from fish farms. For every farmed salmon grow-out cycle, 24 months, there are 2 copepodite infestations from wild stocks (even and odd-numbered) to farmed stocks, and 2 re-infestations from farm stocks to wild stocks. The

wild juvenile mortality rate induced by sea lice infestation<sup>23</sup> is modeled using the Ricker equation (Krkošek *et al.*, 2007; Marty *et al.*, 2010), and a monthly time scale in accordance with time scale of the farmed fishery and lice dynamics. The damage function is estimated by Krkošek (2010), and takes the form  $F_{t+1,n} = F_{t,n} * \exp(-d_w L_{wt} / F_{t,n})$ , where  $F_{t,n}$  is juvenile numbers at age  $n$  at month  $t$ , and  $d_w$  is the mortality rate induced by sea lice population  $L_{wt}$  at month  $t$  attached to wild juveniles ( $F_{t,n}$ ). Therefore, the wild juvenile population for each year right before migrating into the open ocean from inshore area would be

$$F_{n+1} = (1 - m) F_{1,n+1} \prod_t^{May, June, July} \exp(-d_w L_{w,t} / F_{t,n}) \quad (4.6)$$

It is expected that sea lice-induced mortality among farmed Atlantic salmon,  $d_f$ , is much smaller than  $d_w$ . This is because when the farmed Atlantic salmon are released into the marine environment, they are on average 50 grams<sup>24</sup>, while wild pink salmon juveniles are only about 0.2 grams without scales (Brauner *et al.*, 2012; Morton and Williams, 2003; Morton *et al.*, 2004, 2005). The greater size of farmed fish relatively to wild juvenile fish enhances their resistance to infection. The stress born by wild juveniles migrating into the marine environment also makes them more vulnerable to disease.

Before turning into two year old adults, wild juveniles in the marine environment experience a fixed natural mortality rate  $\varphi$ . This rate includes the effect of many

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<sup>23</sup> It is well-established in the literature that *L.Salmonis* in Pacific Ocean and Atlantic Ocean are genetically different, hence have differently pathogenic impact on fish host.

<sup>24</sup> Juvenile Atlantic salmon can be stocked up to 75 gram before releasing to the net pen. <http://www.dfo-mpo.gc.ca/aquaculture/lib-bib/nasapi-inpasa/BC-aquaculture-CB-eng.htm>. This might be a way to prevent farmed fish from getting out of net.

environmental factors along with sea lice, which are considered exogenous. Therefore, the wild adult abundance of each run at year  $n+1$ ,  $F_{w,n+1}$ , is,

$$F_{w,n+1} = (1-m) * (1-\varphi) * F_{1,n+1} * \prod_t^{May, June, July} \exp(-d_w L_{w,t} / F_{t,n}) \quad (4.7)$$

Note that  $L_{w,t}$  is the total lice infestation of wild juveniles migrating into the ocean environment from May to July each year. At other wild salmon stages, sea lice have no impact on wild salmon. Depending on the management objective, the harvesting policy varies. For instance, there are two harvesting policies in British Columbia, a fixed harvest rate and a target escapement (Liu et al, 2011). I assume that the regulatory authority can choose fishing effort level by license limitation, and each year a constant harvest proportion  $\alpha$  is applied. Therefore, the total allowable catch ( $TAC_{n+1}$ ) for each run is,

$$TAC_{n+1} = a(1-m) * (1-\varphi) * F_{1,n+1} * \prod_t^{May, June, July} \exp(-d_w L_{w,t} / F_{t,n}) \quad (4.8)$$

Eq (4.8) suggests that  $TAC_{n+1}$ , equal to  $\alpha F_{w,n+1}$ , is subject to fluctuation due to sea lice infestation during the juvenile stage. The survival  $(1-\alpha) F_{w,n+1}$  will then be the escapement level of wild spawning stock for the next generation.

### 4.3.3 Economic Model

Consider an aquaculture producer. For every 24-month grow-out season, he feeds farmed Atlantic salmon up to month 24 to weight  $w_f$  then harvests all fish at the last period instantaneously with harvest cost  $c_f$  per unit of weight. The adult Atlantic salmon weighs about 7 kg, fetching a price of \$6.5/kg, compared to 1.4k for an adult pink salmon fetching a price of \$0.36/kg. The farmer targets a specific market and sells the fish at a

fixed price  $p_f$  per kilogram. Once he harvests all fish, he will release the same amount of farmed juvenile stock in the farms<sup>25</sup>. Due to its close location to wild migratory routes and other adjacent fish farms, sea lice infestation could occur within the farm and from surrounding areas that cannot be controlled. Each period the producer has to decide whether or not to apply chemical treatment ( $u_t$ ) to his farms, the copepodite and adult sea lice kill function being  $f_c(u_{t-2}, u_{t-1}, u_t)$  and  $f_l(u_{t-2}, u_{t-1}, u_t)$ , as defined previously. I assume that fish farmers manage the fish farms optimally with respect to other inputs than chemical treatment effort, and this decision is separable from other productive inputs<sup>26</sup>.

Farmed Atlantic salmon weight growth is modeled as a polynomial function of time  $w_{f,t} = a_1 t^2 - a_2 t^3$ , where  $t_1 = 1, \dots, 24$  (Asche et al, 2012). In reality lice can affect this weight growth function, but here I assume that lice only affect salmon mortality. A farmed fish requires a certain feeding quantity for weight growth. The conversion ratio (

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<sup>25</sup> Here I assume that a batch operation will kill all lice. Also, I assume that a short-time fallow period, mandatory in many places such as British Columbia, will kill all lice and still allows manager to release the juvenile farmed salmon in the same month that they harvest the adult farmed fish. Therefore the grow-out rotation is not affected by the lice and assumed fixed here. Admittedly, this is simplified from the real world where fish farmers would release fish in spring and fall; also they rotate different farms for fallowing in order to supply the market all year around. The farmed smolts releasing time is not a decision variable arguably due to the low mortality rate from lice from the returning wild stocks.

<sup>26</sup> Another important input decision is stocking density. However the literature suggests that salmon farm are typically stocked at a rate of about 10,000 for a medium- size cage (12m\*12m\*6m). Stocking rates also depend on water temperature and current flows and the size of the sea cage. A paper by Sylvia and Anderson (1993), for instance, assumes a constant 50 metric ton expected harvest per cage when they examine the effluent stock effect on fish production, and no rotation decision is involved.

$f_{con}$ ) is defined as the relationship between the quantity of feed and fish weight growth.

The feeding quantity for each time step for a fish is thus  $g(t) = f_{con} * (w_t - w_{t-1})$ .

#### 4.3.3.1 Solving the Private Producer's Problem

I first solve the problem from the private producer's perspective. Recall that I do not try to solve the problem for multiple farms; instead I assume that there is a single farm, or that the regulatory authority imposes a policy that coordinates management among all farms in the inshore area, meaning that the coordinated farms are operated as if there was only one owner. Given the discount rate  $\delta$ , feeding cost  $c_{feed}$  per kilo, and antibiotic cost of  $c_{treatment}$ , the problem for the aquaculture producer is to maximize the value of aquaculture production, net of control and feeding investment,  $V^P$ , by choosing a treatment policy:

$$V^P = \max_{u_t} \sum_{j=1}^{40} \left( \frac{1}{1+\delta} \right)^{24j-1} (p_f - c_f) w_f F_{f,24n} - \sum_{t=1}^{960} \left\{ \left( \frac{1}{1+\delta} \right)^{t-1} \left( (w_{f,t+1} - w_{f,t}) f_{con} c_{feed} F_{f,t} + u_t c_{treatment} \right) \right\} + S^P(F_{f,960}) \quad (4.9)$$

subject to the state dynamics Eq (4.2-4.4) and the initial conditions  $F_{f,0}$  and  $S_0$ .

The control  $u_t$  is a binary choice variable,  $u_t \in [0,1]$ . Here I model it long enough (40 grow-out seasons, or 80 years) for the steady-state equilibrium to be reached. The time step is monthly, so I have a total of 960 months. For the weight difference term,  $w_{f,2} - w_{f,1} = w_{f,j*24+2} - w_{f,j*24+1}$ , where  $j=1, \dots, 40$ , denotes the fish farm out-growth season, the weight difference is determined exogenously, and lice have no impact on the weight accumulation. For a finite time horizon, a departure from steady state toward the



end of terminal time will be present (Epanchin-Niell and Wilen, 2011). To deal with this problem, I follow the logic of Epanchin-Neill and Wilen (2011), and I set a terminal value to lock the steady state equilibrium once it is reached in 40 rotations (year 80). This terminal value will then account for the benefit of the harvest and control cost accrued value after 80 years (40 rotations) — the last term in Eq (4.9),  $S(F_{f,40})$ . This terminal value is the present value of steady state farmed fish harvest net of control costs from time 81 year to infinitely. Since the system reach steady state at rotation year 40, therefore I am able to use the value at the 40th rotation year to calculate terminal value. Let  $TVP^p = (P_f - C_f)W_f F_{f,40}$  represent the profit from harvesting at the end of the 30<sup>th</sup> rotation, and  $TVC^j = \sum_{936}^{960} \left( (w_{f,t+1} - w_{f,t}) f_{con} c_{feed} F_{f,t} + u_t c_{treatment} \right)$  represents the cost of feed and treatment during the 30<sup>th</sup> rotation (from month 936 to month 960) under privately optimal control. The terminal value is then,

$$S^p(F_{f,40}) = \sum_{j=41}^{\infty} \left( \frac{1}{1 + \delta} \right)^{24j-1} (TVP^p - TVC^p)$$

#### 4.3.3.2 Solving the Joint Fishery Manager's Problem

The joint fishery manager, on the other hand, has to consider the effect that the aquaculture production has on the commercial wild pink salmon fishery. Note in this case there is a fixed proportion of wild pink salmon to be harvested each season, determined exogenously. Therefore, the joint fishery manager's problem considered here is a not a true social planner's problem. To understand the joint fishery management problem, it is necessary to understand the objectives of the individual fisheries.

Wild fishery managers would typically impose restrictions on the license numbers for the wild capture fishery (along with other input restrictions). However, the efficiency of such regulations depends on substitution between restricted and unrestricted inputs as argued by Dupont (1991) and Deacon, Finnoff and Tschirhart (2010). Hence, these policies are often accompanied by other regulations such as allowable harvesting time and areas. Along with the minimum escapement policy imposed, wild fishery managers would ideally seek to prevent rent from being fully dissipated. I assume that with a homogeneous catchability and cost parameter, the wild capture fisher would choose an optimal fishing effort level  $E$  by license limitation. This will lead to the maximization of the net benefit of harvest at the level of the total allowable catch ( $TAC$ ). The harvesting function for wild capture fishery for each wild fishery season is assumed to take the form  $h_n = F_{w,n} (1 - e^{-qE_n})$ , which represents the discrete-time version of the Schaefer-Gordon harvest function for a wild fishery at year  $n$ , where  $F_{w,n}$  is total number of wild adult fish (number not biomass weight) at year  $n$ ,  $q$  is the catchability coefficient and  $E_n$  is the corresponding fishing effort at year  $n$ . Substituting  $h_n = TAC_n$ , and assuming that the unit cost per effort is  $c_w$ , total annual cost ( $TC$ ) thus has the form

$$TC = \frac{c_w}{q} \ln \frac{F}{F - TAC} = \frac{c_w}{q} \ln \frac{1}{1 - \alpha}.$$

Assuming an adult pink salmon weight of  $w_w$  at harvest and price  $p_w$ , the joint fishery problem is to maximize the net present value  $V^J$  from fish farms and the wild fishery, net of control and harvesting cost over time,

$$\begin{aligned}
V^J = \max_{u_t} & \sum_{j=1}^{40} \left( \frac{1}{1+\delta} \right)^{24j-1} (p_f - c_f) w_f F_{f,24j} \\
& - \sum_{t=1}^{960} \left\{ \left( \frac{1}{1+\delta} \right)^{t-1} \left( (w_{f,t+1} - w_{f,t}) f_{con} c_{feed} F_{f,t} + u_t c_{treatment} \right) \right\} \\
& + \sum_{n=1}^{80} \left( \frac{1}{1+\delta} \right)^{12n-1} \left( TAC_n w_w p_w - \frac{c_w}{q} \ln \frac{1}{1-\alpha} \right) + S^J (F_{f,40}, F_{w,40})
\end{aligned} \tag{4.10}$$

subject to the state dynamics, Eq (4.2, 4.3, 4.5, 4.8), and the initial conditions  $F_{f,0}$  and  $S_0$ .

The last term of Eq (10),  $S^J (F_{f,40}, F_{w,40})$ , is the scrap value of the state variables after 40 rotations (80 years),  $S^J (F_{f,80}, F_{w,80}) = \sum_{j=41}^{\infty} \left( \frac{1}{1+\delta} \right)^{24j-1} (TVP^J - TVC^J + TVW^J)$ .

Here I also let  $TVP^J = (P_f - C_f) W_f F_{f,40}$  represent harvesting profit for the 40<sup>th</sup> rotation,

and  $TVC^J = \sum_{t=1}^{960} \left( (w_{f,t+1} - w_{f,t}) f_{con} c_{feed} F_{f,t} + u_t c_{treatment} \right)$  represent feed and control costs

during the 40<sup>th</sup> rotation under joint control (note that the time in  $TVC$  is month).

$TVW^J = (TAC_{79} + TAC_{80}) w_w p_w - \frac{2c_w}{1-q} \ln \frac{1}{1-\alpha}$  represents profit from harvest in the wild

fishery in year 79 (odd-numbered year run) and 80 (even-numbered year run). These two years correspond to 40<sup>th</sup> rotation for fish farming operation.

Here  $n$  denotes a yearly time scale as wild salmon are harvested every year. By taking into consideration the externality that farms create by the spatial migration of lice from farms to wild juveniles and the externality imposed by the migration of gravid lice from wild spawning stock to farms, the solution for the social planner's problem will differ from the private producer's problem. The aquaculture producer only controls lice

until the marginal value of the damage inflicted on the farms is equal to the marginal cost of applying chemical treatment to control lice in the farms, including future growth on the producer's farms.

#### 4.4 Numerical Results

Given the complex nature of this integer non-linear, non-convex optimal control problem, closed form solutions for the value function and optimal control policy do not exist. Furthermore, multiple locally optimal solutions may occur. Therefore, I rely upon numerical implementation to derive a solution for this problem. Premium Solver Platform (multi-starting point) was used for the simulation and able to handle 1000 variables and 1000 constraints (including integer constraints) with Premium Platform's Branch and Bound method. I draw the economic and biological parameters of the models from the literature. I also make assumptions for missing information, then conduct sensitivity analysis over these parameters. All notation, parameters and abbreviations are described in Table 4.1 in the end of this chapter. The values for all parameters are provided in Table 4.2. Some of the parameters are taken directly from the literature, and some are calibrated for this study. Nonetheless, none are empirically derived and they do not reflect any particular system.

Table 4.2 Parameter Definition and Values Used for the numerical Simulation

Parameter	Value	Note	Reference
$F_{f,0}$	1000000	The fish numbered released at the beginning of each out-growth season	Mustafa et al. (2001)*
$S_0$	1000000	The initial escapement level for wild stock	Liu et al. (2011)**
$\rho$	0.00000015	The propability that a copepodite attached to host	Frazer et al. (2012)***
$\varepsilon_1$	0.25	Base settlemet success for copepodite survival to adult stage on host	Frazer et al. (2012)***
$\varepsilon_2$	0.15	Seasonl force impact coefficient	Bricknell et al. (2006)****
$\nu$	0.15	Natural mortality for adult lice (one per unti of time)	Frazer et al. (2012)***
$k$	0.85	Chemical treatment efficiency for sea lice	Gustasfon et al. (2006)□
$z$	0.85	Chemical treatment efficiency for copepodite	Gustasfon et al. (2006)□
$\lambda$	50	Natality (copepodite produced per adult female louse per unti of time)	Frazer et al. (2012)***
$\zeta$	0.5	Copepodite natural mortality	Frazer et al. (2012)***
$s$	5	Fixed number of gravid lice on each individual wild spawning salmon	Frazer et al. (2012)***
$p_f$	6.5	Market price for farmed fish (\$/kg)	Asche et al. (2011)□□
$p_w$	0.32	Market price for wild pink fish (\$/kg)	Liu et al. (2011)
$\delta$	0.004074	Mothly discout rate	Laukkanen (2001)□□□
$m$	0.94	Natural mortality rate for wild juvenile in fresh environment	Liu et al. (2011)
$\varphi$	0.5	Natural mortality rate for wild stock in marine enironment	Liu et al. (2011)
□	5.2	Wild population growth rate	Liu et al. (2011)
$b$	4456618	Density dependent mortality	Liu et al. (2011)
$d_w$	0.4	Mortality rate for wild juveile induce by sea lice	Krkosek et al. (2007)
$d_f$	0.05	Mortality rate for farmed salmon induce by sea lice	Krkosek et al. (2007)
$c_f$	1.5	Harvesting cost per kilo farmed fish	Asche et al. (2011)□□
$c_{feed}$	1	Feeding cost per kilo feeding	Asche et al. (2011)□□
$C_{treatment}$	50000	Treatment cost per chemical application	Mustafa et al. (2001)*
$f_{con}$	1.1	Food conversion ratio	Asche et al. (2011)
$c_w$	30	Unit cost per fishing effort for wild fishery	Laukkanen (2001)□□□
$q$	0.016	Catchability coefficient for wild fishery	Laukkanen (2001)□□□
$a$	0.76	The harvesting proportion for wild fishery	Liu et al. (2011)
$W_w$	1.43	Adult pink salmon weight (kg)	Liu et al. (2011)

Note: "\*" Mustafa et al. (2001) estimated sea lice control cost in East coast of Canada. The cost of chemical treatment per section was Can \$150-2000/per cage raising 10000 fish. Since I focus on a large farm (100 cages), I scale up the cost proportionally and conduct sensitivity analysis. "\*\*\*" Liu et al. (2011) estimated that the equilibrium population level for pink salmon is about 760,000 in Broughton Archipelago BC. I scale up the wild population level to farmed salmon population level. "\*\*\*\*": While there is uncertainty about these parameters, the multiplication of louse natality and settlement success should be much greater than louse mortality (Frazer et al., 2012). In their paper the settlement success is 0.22, and louse natality is 6.4 per day. "\*\*\*\*\*" Bricknell et al. (2006) estimated the effect of salinity on sea lice settlement success. The literature provides few empirical data on how seasonality impacts the lice settlement success. "□" Frazer et al. (2012) used 0.90 as the chemical-induced sea lice mortality rate. Gustasfon et al. (2001) found that chemical has efficacy over 90 days. I make the assumption that the chemical-induced mortality rate is the same for lice and copepodite."□□" Asche et al (2012) used a value that is slightly higher in Norway."□□□": Laukkanen (2011) used a higher unit cost per fishing effort for Baltic salmon. I assume a high abundance level and hence a lower cost per fishing effort. The ones without any symbol in the reference column denote values retrieved directly from the literature.

The frequency and timing of chemical treatment determines the effect on the bio-control of invasive species and disease alike. The binary control policies for private producer and the joint fishery manager in the first four fish farm grow-out seasons (the first 8 years) are clearly different, while the rest of period sees fewer differences. Both the privately and jointly optimal policy involves chemical treatment from August to November, and from June to December. Therefore, I am able to use the first four fish farm grow-out seasons to summarize and characterize the control policy differences between the private producer and the joint fisheries manager (Fig.4.3).

In the first year of the grow-out period, the private producer only applies chemical treatment from August to October, then from the next May to December. Specifically, the chemical treatment for the odd-numbered year run lasts from August to December, then from the next May to July. I refer to the treatment strategy starting August to December as the first period treatment strategy, which corresponds to the odd-numbered year wild spawning stock migration back to natal river. I refer to the treatment strategy starting next May to July as the second period treatment strategy, which corresponds to the lice reproduction after the first period treatment for the odd-numbered year run. The treatment policy for even-numbered year runs is the same for all periods- only applying chemicals from August to December. The distinction between the odd and even-numbered year runs is artificial. I assume that the odd-numbered years are first to be subjected to the lice infection. The time lag between odd and even numbered year runs makes a difference due to discounting. This is confirmed through simulations that (a) set initial infection levels in odd-numbered year runs to zero, and (b) apply different discount factors. The second, third and fourth grow-out seasons see chemical treatment from

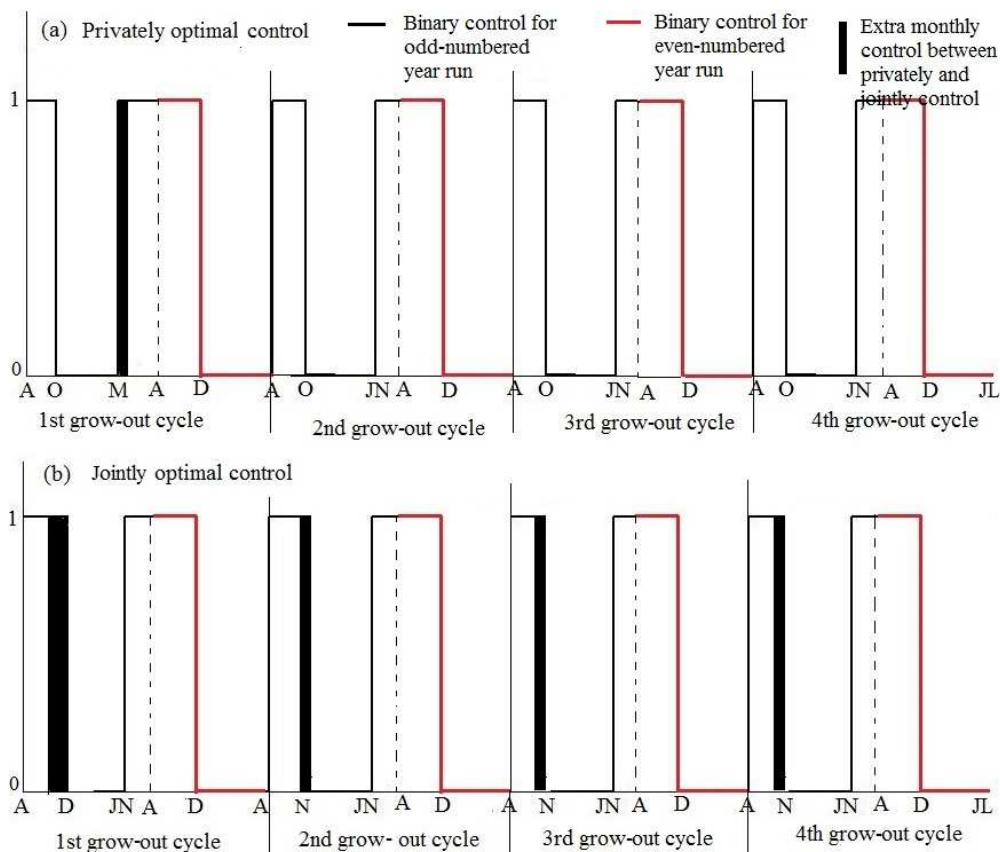
August to October then from the next June to July for odd-numbered year runs. Two factors may contribute to the delay in chemical applications in the second period treatment for these grow-out seasons. They are the effect of discounting and feedback from the control policy in the first year, which results in a lower spawning wild stock, and hence lower infection risk to farm and less chemical control.

The joint fisheries problem, on the other hand, requires that chemical controls be applied from August to December, and then from June to July for odd-numbered year runs. The treatment policy under jointly optimal control for even-numbered year runs lasts from August to December—the same as that from privately optimal control. This is also due to the effect of discounting and the cheap price of wild stock. The second, third and fourth grow-out seasons have a chemical control policy applying from August to November, due to feedback from the control policy in the first year.

In sum, the private producer applies less effort to control lice than the joint fisheries manager because they focus only on farm profits, while joint fisheries manager has to consider the wild fishery externality as well. In the first year of the first grow-out season, the private producer applies chemical controls earlier than a joint fishery manager would, from May to July versus June to July for jointly optimal control. Even though the private producer does not consider the wild fishery—wild juveniles emigrating inshore from May to July—the reduced control effort in the first period leads to earlier controls in later periods due to the lice reproduction in the farm. Symmetrically, the greater the control effort on the part of joint fisheries manager, and the three-month period of chemical efficacy, leads to later chemical treatment in second and subsequent grow out periods. Finally, feedback from the chemical treatment under jointly optimal control

leads to a more abundant spawning wild stock than that under privately optimal control, hence more lice will disperse into farms, leading to more control by the joint fisheries manager.

Figure 4.3. Privately Optimal Control and Jointly Optimal Control Policy During the First Four Atlantic Salmon Grow-Out Periods



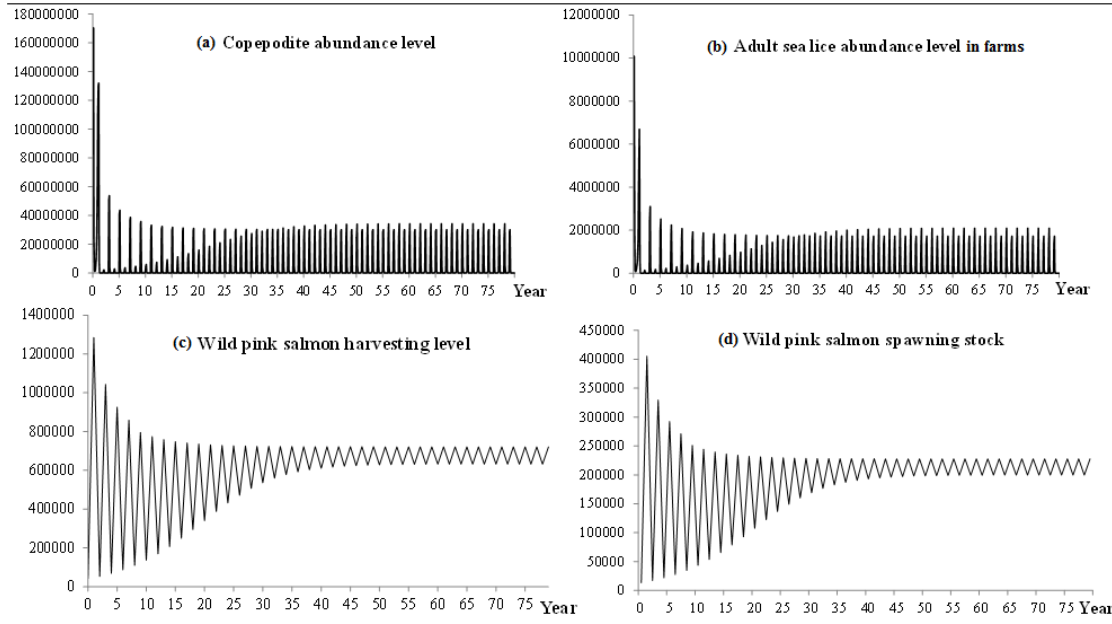
Note: The letter in the axis denotes the first letter of the month. M: May, JN: June, and JL: July. Private producer applies less effort for lice treatment than joint fisheries manager for the odd-numbered year run, but same control effort for even-numbered year run. Both private producer and joint fisheries manager apply more control effort in the early stage than in the later stage for each grow-out season. The effect of discounting and the feedback from control policy in the early stage/season also determine the control pattern in the later stage/season for both private producer and joint fisheries manager.



The treatment policy by the fish farmer deviates from that of the joint fishery manager only in the early years. This is understandable, since sea lice control in the early stages will have a greater impact on welfare than control in at later stages, partly due to the effect of discounting. This difference in chemical treatment between the fish farmer and the joint fishery manager will result in different impacts on the wild population. Since the control of the fish farmer happens during the spawning stock migration period, it internalizes at least part of the external impact of lice control decisions. Furthermore, the binary nature of control also helps the alignment of fish farmer's and joint fishery manager's objectives.

The consequences of the private producer's control policy are illustrated in Fig 4.4. The peak levels for copepodite and adult sea lice abundance are related to the time when wild spawning stocks migrate back to the natal river. The peak levels for copepodite and adult sea lice abundance corresponding to odd-year runs falls to a very low level then climbs back up, while the peak level for copepodite and adult sea lice abundance corresponding to even-year runs decreases until it levels off when the wild population approaches a steady state. These treatment policy differences for odd and even years generates an oscillating pattern of sea lice infestation. Note that one month less lice control in the first four grow-out periods for odd-numbered years in privately optimal control relative to joint control really drives down the odd numbered wild juvenile stock when they emigrate into inshore from freshwater environment. Intuitively, all else equal, a reduction in wild spawning stocks implies less infection risk to farmed salmon stocks, and hence to wild juvenile stocks next year. Since wild juvenile salmon mortality falls, this increases the abundance of wild adults in the ocean, resulting in a high spawning

Figure 4.4. Trajectories for Copepodites and Adult Sea Lice Abundance (*Ex-Ante* Treatment), Wild Salmon Harvesting and Spawning Stock under Private Optimum Control



Note: The privately optimal control policy results in an oscillation pattern in the peak levels of steady state copepodite, adult sea lice abundances, wild pink salmon harvest and spawning stock levels. There are two trends in a, b, c and d. The increasing trend corresponds to odd-year numbered wild salmon run, and the declining one corresponds to even-numbered year wild salmon run.

stock, high infestation risk to farms, and high infection rates of wild juvenile stocks, and therefore fewer adults and a smaller wild spawning stock in the following year. This oscillation pattern between the peak level of copepodite and adult sea lice abundance associated with odd and even-numbered year run also contributes to the presence of a more obvious oscillation pattern for wild pink salmon harvesting and spawning levels, illustrated in Fig. 4.4 (c) and (d).

The oscillating pattern is also present in the joint fisheries management problem (Figure 4.5). The trajectory path, on the other hand, is quite different. The copepodites and adult sea lice abundance levels corresponding to odd and even-numbered year run both decrease smoothly. This leads to a relatively smoother path for wild fishery harvest and spawning stock level, which is in stark contrast to harvest and spawning stocks associated with privately optimal control of the fishery.

Figure 4.5. Trajectories for Copepodites and Adult Sea Lice Abundance (*Ex-Ante* Treatment), Wild Salmon Harvesting and Spawning Stock under Joint Optimum Control

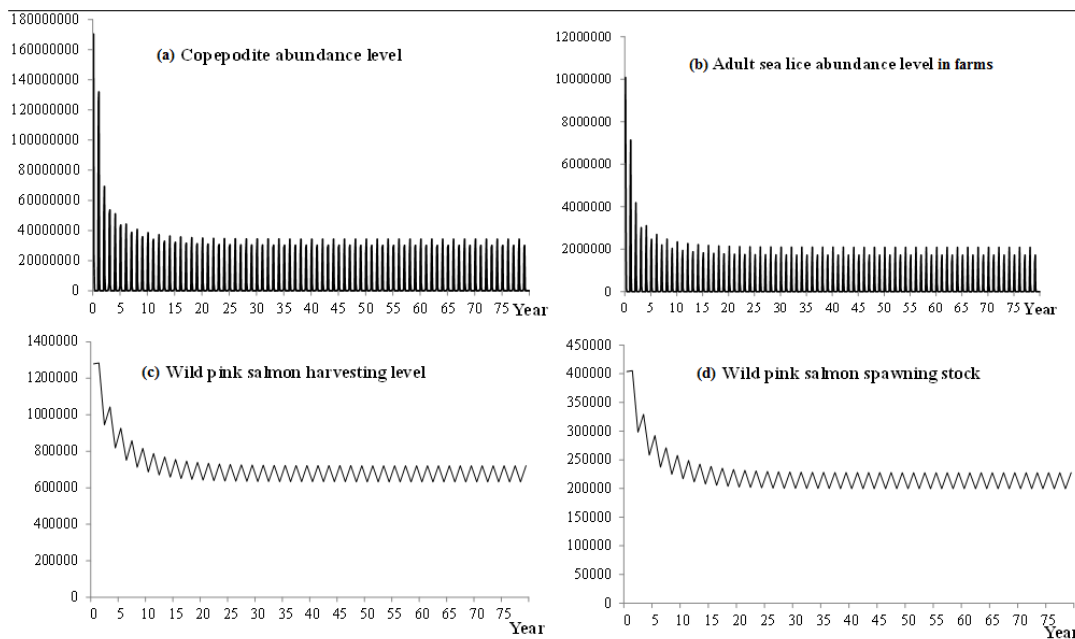


Figure 4.5. Oscillation pattern is also seen in steady state under jointly optimal control. Two different levels of declining trend stand in starkly contrast to that from privately optimal control with the lower level corresponding to odd-numbered year wild salmon run and higher level corresponds to even-numbered year wild salmon run.

Table 4.3 shows the net present value (NPV) and the steady state wild fishery harvest and wild spawning stock level under socially and privately optimal control. The

difference between total NPV under the two control strategies is not significant (approximately 0.6% of total NPV under joint fisheries management). In other words, the externality generated by farms is quite small. This is because of the low price and small size of wild pink salmon (\$0.36/kg and 1.43kg) relative to farmed Atlantic salmon (\$6.5/kg and 7.3kg). The steady state harvest and spawning stock level are approximately the same under both approaches. However the outcome is quite different for odd and even-numbered year runs, with less harvest and spawning stock levels in odd-numbered years due to differences in treatment policy.

Table 4.3 A Comparison of Key Results between Private Producer and Joint Fisheries Manager

	Private Producer	Joint Fisheries Manager
Total Net Present Value	266,865,337	268,501,973
Net Present Value for Aquaculture	261,431,513	260,469,439
Net Present Value for Wild Fishery	5,446,824	8,032,534
Wild Fishery Harvesting at Steady State (Odd Year)	632,100	632,100
Wild Spawning Stock at Steady State (Odd Year)	199,610	199,610
Wild Fishery Harvesting at Steady State (Even Year)	721,550	721,550
Wild Spawning Stock at Steady State (Even Year)	227,858	227,858

#### 4.5 Potential Institutions for Sea Lice Control

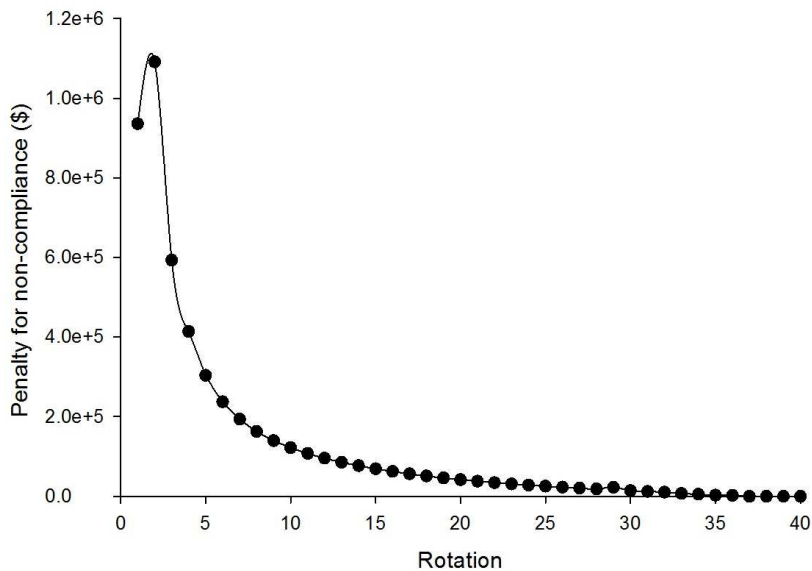
Since analytical solutions for any market-based approach to control sea lice is not available due to the integer non-linear, non-convex optimal control problem, I also rely on numerical simulation to investigate a potential institution for sea lice control to align privately and jointly optimal control policy. I use Bellman's principle of optimality, which allows me to induce farmers to adopt the jointly optimal control policy via a penalty for non-compliance with the jointly optimal policy, using a backward induction method. The amount of penalty for not compliance is determined as followed,

- Solve for the private optimum in the last rotation ( $T=40$ ) when the wild stock level inherited at the beginning of this rotation is exactly the same as the beginning of  $T$  rotation along the jointly optimal trajectory.
- If treatment in period  $T$  differs from the jointly optimum, increase the penalty in the month that deviates from the jointly optimum until the privately optimal control “flips” to match the jointly optimal control. In months when the privately optimal control does not differ from jointly optimal control, a zero penalty is assumed.
- Move back to  $T-1$  rotation. The wild stock level inherited at the beginning of  $T-1$  rotation is set equal to that along the jointly optimal trajectory. Solve for the privately optimal control in rotation  $T-1$  and  $T$ , with the penalty found for rotation  $T$  still in place.
- If the lice control pattern at either rotation  $T$  or  $T-1$  differ from those along jointly optimal control, adjust the penalty in rotation  $T-1$  only until the lice control in both rotation align jointly optimal control.

The simulation is run twice for each rotation. The first run corrects a control deviation in November. After a penalty imposed in November, a second run corrects for a control deviation in October, and requires a penalty in October. Where simulation in the first rotation finds control deviations in two months (November and December), the penalties in these two months and increased simultaneously until the control “flips”. This leads to an increasing penalty for non-compliance from the first to the second rotation. Figure 4.6 showed the penalty regime for non-compliance for private producer over all 40

rotations. I found that the penalty for non-compliance begins from 37<sup>th</sup> rotation and increases when moving backward. Penalties occur mostly in October and November, except for the first rotation where they occur in November and December, or only in November (36<sup>th</sup> and 37<sup>th</sup> rotations).

Figure 4.6. The Penalty Regime for Non-Compliance for Private Producer over 40 Rotations



The decreasing value of the penalty is due to a combination of discounting and the decreasing value of the externality resulting from declining wild stock levels and hence declining wild harvests. The aggregate value of penalties needed to induce compliance is approximately \$5.2 million, more than three times the value of the externality. While this discrepancy is large, the reason arguably lies in the binary nature of the lice control. Little information exists on the magnitude of tax, subsidy or penalty to internalize the externality under the context of binary control. Intuition suggests that the total amount of these instruments required can be greater or smaller than the economic benefit loss

resulting from the negative externality. The reason lies in the gap between the penalty threshold of “flipping” and the true benefit loss due to the externality.

#### 4.6 Sensitivity Analysis

Sensitivity analysis of key parameters was carried out relative to the maximized net present value (NPV). In principle, all parameters could be evaluated by how the result changes in response to change in each parameter in the model. I focus on a subset of parameters and approach the sensitivity analysis in two steps. First I consider an economic parameter, the price of wild salmon,  $p_w$ , and then I consider other key biological parameters and another economic parameter, treatment cost.

The price of wild pink salmon is particular important as the price may vary due to market integration. Asche et al. (2005) found that farmed and wild salmon are close substitutes on the Japanese salmon market, and that the increasing supply of the farmed salmon would lead to a price decrease in all salmon species. However, in North America there might be niche markets for wild salmon due to preferences for wild caught fish or to market labelling laws that requires the source of fish to be identified. Such product differentiation has also been promoted by many wild salmon conservation coalitions and environmental associations. Eco-labelling can change consumer’s behavior (Teisl, et al, 2002), and consumers may be willing to pay a premium for this environmental friendly product. Therefore, there might be a price drift and fishermen might be able to sell wild salmon in North America at a higher price.

Since the producer’s behavior will not be affected by wild salmon price changes under privately optimal control, I examined the control policy and its biological

consequences under socially optimal control over a range of salmon prices, low (base) (\$0.36/kg), medium (\$0.6/kg) and high (\$0.72/kg). Lice infection is most common in pink salmon (\$0.36/kg) and chub salmon (\$0.94/kg) in North America. As figure 4.7 indicates, when the wild salmon price increases, a social planner would apply chemicals in farms more frequently in order to reap more benefit from the wild fishery. So there is a ‘stair pattern’ between the total number of monthly treatments and the wild pink salmon price.

Figure 4.7. Total Monthly Sea Lice Control Level and Harvesting Level When the Pink Salmon Price Is Low (\$0.36/kg), Medium (\$0.6/kg) and High (\$0.72/kg)

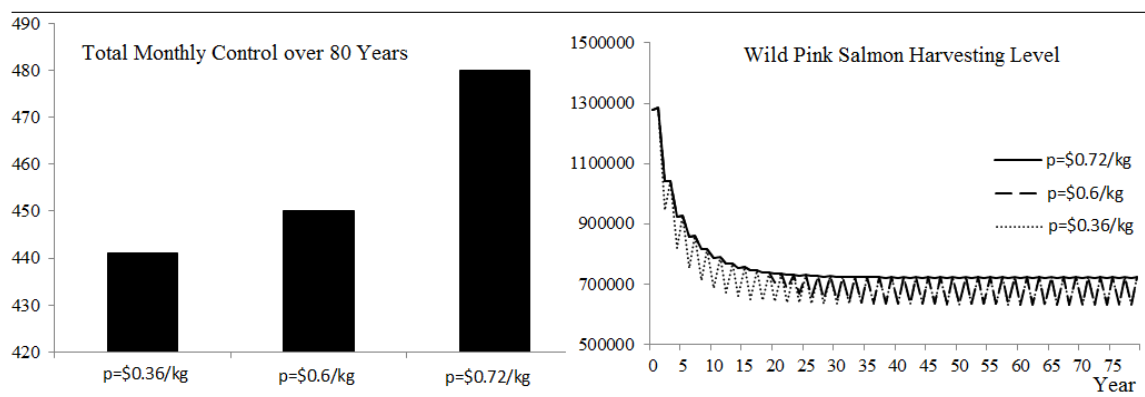


Figure 4.7 also illustrates the consequences for wild pink salmon harvesting. There is an initially smooth decrease, then an oscillating pattern. By the end of the planning horizon the oscillation is quite close to the wild salmon price. When the price is doubled, the oscillation pattern is still present, but the amplitude of the oscillation is significantly smaller. However, a further increase in wild salmon prices will not result in a higher level of harvest and spawning stock levels, and the control policy will increase to 361 months out of the possible 720 months.



I also conducted sensitivity analysis for a subset of biological parameters and for treatment costs. These biological parameters are chosen because of their high degree of uncertainty. These uncertainties highlight the center of current debate on the effect of lice on wild salmon. To conduct sensitivity analysis for attachment and settlement success, louse and copepodite mortality, chemical efficiency, natality, mortality rate for wild juvenile induced by sea lice, mortality rate for farmed salmon induced by lice, food to body weight conversion ratio and treatment cost, I increased these parameter values individually by 10%, and examined the percentage change of the resulting total maximized NPV, while keeping others at the base value.

Table 4.4 reports the results of sensitivity analysis for these parameters. It shows that the impact of each key parameter is consistent between the private and social problems except for sea lice-induced mortality rate for wild juveniles. Under privately optimal control, all else equal, a 10% increase of sea lice-induced mortality will lower the wild juvenile population, hence less wild spawning adult and less infestation for farm fish. Under socially optimal control, the gain for aquaculture producers is less than the cost to the wild fishery. The total NPV under these two control scenarios are most sensitive to chemical efficiency  $k$  on lice and conversion ratio  $f$ . A 10% increase in chemical efficacy will result in approximately 2.2% and 2.1% increase in maximized NPV for privately optimal control and socially optimal control, respectively. On the other hand, a 10% increase in conversion ratio will result in approximately the same, 3%, for both privately optimal control and socially optimal control. This result is consistent with the literature. For instance, Costello (2009) found that one of the most significant costs of sea lice management is the reduced feed conversion efficiency. A 10% increase in control

cost has negative but relatively weak impact on the net present value of the fishery. While increasing the value of this parameter also changes the value of steady state population for odd-number year runs but not for even-numbered year runs. I include this information in the last column in Table 4.4. It shows that a 10% increase in chemical efficiency results in a 11.5% of population increase, while a 10% increase in lice natality and copepodite settlement success to adult stage both lead to a 10% decrease in steady state odd-numbered year run stock levels. This also suggests that the greatest divergence in the trajectory between privately optimal control and jointly optimal control comes from these three parameters. A 10% increase in chemical efficiency under privately optimal control results in a significant reduction of chemical controls, decreasing applications from 437 to 360 over 80 years.

Table 4.4 Sensitivity Analysis

Parameter	Privately Optimal Control	Jointly Optimal Control	Steady State Odd-Number Year Run under Privately Optimal Control
$\rho$	-0.4423	-0.3293	-0.9551
$\nu$	0.1195	0.0642	1.7520
$k$	2.2252	2.0923	11.497 (270 controls )
$z$	0.1457	0.0542	2.9000
$\lambda$	-0.445	-0.3323	-9.9210
$\xi$	0.0598	0.0138	1.2078
$d_w$	0.0245	-0.0003	-1.1432
$d_f$	-0.1886	-0.2541	0.0757
$\psi$	-0.4539	-0.3323	-9.9209
$f_{con}$	-2.9947	-2.9796	0.0010
$C_{treatment}$	-0.2705	-0.2102	0.0010

Note: Table entries represent the percentage of change in maximized net present value, in response to a 10% increase in parameter value relative to its value in Table 2, while holding all other parameter values constant.

#### 4.7 Discussion

Salmon aquaculture is one of the fastest growing industries worldwide. Salmon farming production surpassed wild caught salmon in 1996, and has now become an economic driver for many regional economies. Despite its potential, local governments are reluctant to introduce salmon farming due to its environmental effects. At the top of the list of environmental effects is disease transmission between salmon farms and wild fisheries, which concerns fishermen and environmental groups tremendously. It is important to manage this issue based on both biological and economic factors. To the author's knowledge, this study is the first attempt to integrate sea lice population dynamics, fish population dynamics and economic model, and to provide a tool to investigate the sea lice management policy. Also, it investigates an institutional arrangement for addressing the externality issues generated by the salmon aquaculture that prevents the whole system from being efficient.

Several assumptions were made when constructing the bioeconomic model. The first is that copepodites and adult lice both have a one-month life cycle. This is based on the lumping of free-living stages and stages during which copepodites attach to fish hosts. In reality, the timing of different stage varies with temperature, and the copepodite's life span could be longer in lower temperatures. Changes in environmental factors are found not only in the impact on louse survival and growth, but also on the adult lice's natality rate. Secondly, lice are assumed not to evolve and develop any resistant strains in response to chemical treatment. The assumption here is that lice are either killed or not infected by the chemical treatment, and lice surviving the chemical treatment still have the same growth and natality rates. Increasingly, evidence suggests the emergence of

resistance to chemical uses (Murray, 2011). The development of resistant lice would come at a cost—it takes time for the surviving and exposed lice to develop the same growth and natality rate as lice in the absence of chemical treatment. Finally, it is assumed that wild juveniles with a non-lethal infestation level may still maintain the same growth rate. Evidence suggests that parasite infestation would influence wild stock dynamics, that the physical impact is additive, and that the exposed wild stock during infestations would suffer significantly lower growth rates, leading to a smaller spawning sizes (Krkošek, et al., 2007; Skilbrei and Wennevik, 2006).

The analysis shows how sea lice management strategy changes in response to two different property right regimes. While the state trajectory paths are different, both regimes reach the same steady state harvesting and spawning stock levels with the same oscillation pattern. While this study only deals with a single private aquaculturist, or a coordinated aquaculture industry, the numerical results suggest that a coordinated aquaculture industry would result in decline of wild pink salmon population regardless of the management regime. This result is consistent with the conclusion reported in many sea lice studies. However, the economic benefit to joint fishery management is not significantly different from that to separate fishery management. There are two possible reasons for this. The first is that the nature of binary control helps the alignment of private and joint fishery objectives. Private producers, to some extent, already internalize part of the externality when they make their own treatment decisions. Secondly, the low price and weight of adult wild pink salmon, relative to that of Atlantic salmon, means that the damage they sustain is relatively low value. This study also shows that a penalty for non-compliance, derived using backward induction logic from Bellman's principle of

optimality, is able to replicate the control behavior of jointly optimal control strategy. The relatively large magnitude of the total penalty compared to the externality cost is arguably due to the binary nature of sea lice control.

This study is based on modeling assumptions for the sake of mathematical tractability, evaluation and calibration of parameters because of the vast uncertainty associated with these parameters. Some of these inputs and models are used directly from the literature, others have been modified to fit the numerical setting in which the study is performed. While it is desirable to simplify the related models in order to focus on the core questions framed in the objective outlined above, it should be noticed that the problem inherently is very complicated. This simplified model only measures the net effect of sea lice loaded on fish, and cannot identify various underlying causal mechanisms. To the extent that this simplified model and parameters used in this study differ significantly from the real world, the estimated results may be biased and the extrapolation of the results should be taken cautiously. I believe that the model built here is sufficiently rich and captures the essential epidemiology and economic necessary to understand the problem at hand, and provides insight for the regulation of sea lice transmission between farmed fish and wild fish population.

#### Acknowledgements

This work on the disease control aspects of this problem benefits from support from grant number 1R01GM100471-01 from the National Institute of General Medical Sciences (NIGMS) at the National Institutes of Health.

Table 4.1 Variables, Parameters and Abbreviations in the Theoretical Model

Symbol	Definition
$t$	Month
$L_{w,t}$	Total lice abundance in wild population in time t
$\varsigma$	Proportion of total gravid lice dispersed into farms
$\rho$	The propability that a copepodite attached to host
$F_{f,0}$	The fish numbered released at the beginning of each out-growth season
$F_{f,t}$	Total farmed fish abundance
$F_{w,t}$	Total wild fish abundance
$\psi_t$	Settlement success for copepodite survival to adult stage on host
$\varepsilon_1$	Base settlement success for copepodite survival to adult stage on host
$\varepsilon_2$	Seasonal force impact coefficient
$u_t$	Chemical treatment binary variable
$k$	Chemical treatment efficiency for sea lice
$z$	Chemical treatment efficiency for copepodite
$\nu$	Natural mortality for adult lice (one per unit of time)
$\lambda$	Natality ( Copepodite produced per adult female louse per unit of time)
$\xi$	Natural mortality for copepodite ( one per unit of time)
$X_t$	Total copepodite abundance in the coastal area at time t
$s$	Fixed number of gravid lice on each individual wild spawning salmon
$d_f$	Sea lice-induced mortality rate for farmed Atlantic salmon
$S_n$	Escapement level of wild stock in year n
$S_0$	The ininitial escapement level for wild stock
$R_{t,n+1}$	The wild fry abundance of age (n+1) at month t
$\gamma$	The population growth rate for wild stock
$b$	Density dependent mortality
$m$	Natural mortality rate for wild juvenile in fresh water environment
$d_w$	The wild juvenile mortality rate induced by sea lice
$\phi$	Natural mortality rate for wild stock in marine environment
$F_{n+1,w}$	The wild adult abundance of at year n+1
$\alpha$	The harvesting proportion for wild fishery
$TAC_{n+1}$	Total allowable catch for wild stock at year n+1
$w_f$	The weight of adult farmed fish (kg)
$c_f$	Harvesting cost per kilo farmed fish
$p_f$	Market price for farmed fish
$w_{f,t}$	The weight of farmed fish at month t
$f_{con}$	Food conversion ratio
$g(t)$	Feeding quantity at time t for farmed fish
$\delta$	Monthly discount rate
$c_{feed}$	Feeding cost per kilo feeding
$c_{treatment}$	Chemical treatment cost per section
$j$	Fish farm out-growth season
$h_n$	Harvest level for wild fishery at year n, same as TAC
$q$	Catchability coefficient for wild fishery
$E_n$	Fishing effort at year n for wild fishery
$c_w$	Unit cost per fishing effort for wild fishery
$w_w$	Adult pink salmon weight (kg)
$p_w$	Market price for wild fish

## Chapter 5

### CONCLUSION

This dissertation has presented three substantive papers to investigate the effect of size-based prices on fishery management and harvest, the effect of market uncertainty in the ecological consequence of an establishment of whale permit markets, and the problem of disease externalities between farmed salmon stocks and wild salmon stocks, respectively. Each paper has introduced a new model, contributes to the bioeconomic literature, and has management implications with regard to the traditional fishery science, whale conservation policy and sea lice control policy.

Traditionally, fishery science tackles the growth overfishing problem with yield-per-recruit (YPR) and revenue-per-recruit (RPR) models. These models are conventionally used to determine fish harvesting rates where there is a trade-off between the size of the fish and the number of fish available for harvesting. One goal of the first paper has been to compare and contrast three strategic approaches for managing the growth over-fishing problem when there is a price premium for larger size of shrimp: conventional YPR without consideration of economic trade-offs, RPR when there is size-based price information, and harvest strategy based on a bioeconomic model with the size-based price effect. The chapter adds to the current literature by extending the framework traditionally used to deal with growth overfishing in fishery science. The numerical results suggest that there are potential economic advantages to the inclusion of size-based price information. An interesting feature is that strategy based on revenue-per-

recruit that considers the sized-based price effect can generate a rent that is close to that from bioeconomic strategy when there is low natural mortality, but not when there is high natural mortality.

The second paper aims to quantify the effect of economic uncertainty on the conservation potential of a whale permit market that allows environmental organizations to buy and retire permits, and to forecast the possible steady state ecological outcome. This paper contributes to the literature by investigating the conditions in which transactions in a whale permit market will lead only to whaling, to complete conservation, and to the coexistence between whaling and conservation. One of the biggest concerns about a global whale permit market versus the current moratorium policy is the free-riding issue due to the public good nature of whale conservation and the implementation challenges. The possibility of opening up a regional whale permit market, such as in Norway, is considered. The objective of this paper is not to estimate the resulting social welfare for whaling and conservation groups, instead it places emphasis on the long run harvesting and conservation status by considering key economic uncertainties. The simulation results suggest the possibility of three types of outcome: harvesting all permits, whaling buyout and the coexistence of whaling and whale conservation, depending on key parameter distribution. However, under probable parameter ranges, the pure conservation policy is the most likely outcome.

The third paper develops a unique bioeconomic model to analyze the disease transmission problem between farmed fisheries and wild fisheries. It investigates the



incentive for the management of sea lice from the perspective of private aquaculture producer and then a joint fisheries manager. This allows me to evaluate how much privately optimal treatment behavior deviates from the jointly optimal control solution, and to assess the economic impact of salmon aquaculture on the wild fishery due to sea lice transmission. An institutional arrangement is considered to correct the externality that fish farms create. A feature of this problem is the discreteness of sea lice transmission between wild fishery and aquaculture farms and the binary nature of sea lice control. My results suggests that private aquaculture does have an incentive, to some extent, to treat lice during the wild salmon migration period, and that the binary nature of sea lice control can partially help private producers to internalize the sea lice externality to wild fisheries. The numerical results also indicate different state trajectories under two different management regimes, privately optimal control and jointly optimal control. Another finding is that the externality that fish farms generate is relatively insignificant compared to the total net present value of the farmed fisheries.

Concerns about the sustainable exploitation and development of marine resources are likely to persist and intensify with the presence of poorly defined property right regimes over many common property marine resources. Indeed, with the demand for marine resources increased, the problem that is associated with sustainable resource allocation is more imperative. Bioeconomic models are developed to investigate the inter-temporal decision with the presence of negative externality, and to evaluate the economic and ecological tradeoff.

The bioeconomic model developed in the first study can be generalized and applied to other annual species or species that do not show a significant relationship between the breeding stock and juvenile recruitment into the fishery. The second study examines the steady state equilibrium harvest and population level, the trajectory to the steady state, however, is also important. The conservation effort would also benefit the whaling groups by increasing whale population level and hence more whaling permits grandfathered to whaling group. How to address the implied increasing cost for whale conservation for market design, and possibly the increasing free-riding as well because of this increasing cost pattern? The second and third study both make assumptions due to the limited data information and highly uncertainty associated with parameters. The implied results can be refined by further studies, for instance, through a contingent valuation for whale conservation, and through field study on how temperature changes the dynamic of sea lice. The sea lice study also assumes a coordinated farm industry. In reality, this also raises question: how to design institutional systems that effectively control sea lice dispersal among different farms and wild fishery? Or are these institutional systems worthy? These questions provide directions for future research.

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APPENDIX A  
YIELD AND REVENUE-PER-RECRUIT ANALYSIS  
WITH BOUNDED FISHING MORTALITY



Impulse harvesting is generally infeasible. If capacity and therefore fishing mortality ( $f$ ) is bounded above, the maximized yield-per-recruit involves solving for the opening ( $t_c$ ). The biomass yield for a particular recruit is then (Getz and Haight, 1989),

$$Y(f, t_c) = \int_{t_c}^{t_{\max}} f w(t) x(t) dt$$

Here  $f$  is fishing mortality, it is a constant throughout the season,  $x_t$  is the number of individual alive,  $w_t$  is the weight of each individual,  $t_c$  is the fishing open season, and  $t_{\max}$  is the maximized age of cohort. Note that in the numerical example, the shrimp is recorded in discrete categories (week), therefore a discrete version of yield is used.

In the case of constrained fishing mortality  $f$ , the maximized revenue-per-recruit similarly involves solving fishing opening season  $t_R$ . The revenue from catch is

$$R(f, t_R) = \int_{t_R}^{t_{\max}} f \psi(t) x(t) dt$$

where  $t_R$  is the fishing open season, and  $t_{\max}$  is the maximized age of the cohort,

$$\psi(t) = p(w(t)) w(t) .$$